Allied-Signal Inc.

Morristown, NJ

Risk Assessment Report UOP Site, East Rutherford, New Jersey

Volume 2

Part I - Ecological Assessment

Part II - Human Health Food Chain Assessment

ENSR Consulting and Engineering

(Formerly ERT)

November 1989

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ENSR

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November 17, 1989

Mr. James Schnitzer, Case Manager
NJDEP Bureau of Federal Case Management
401 East State Street
5th Floor, West Wing
Trenton, New Jersey 08625

Re: Comments and Revisions to Ecological and Human Health Food Chain Risk Assessments, UOP Site, East Rutherford, New Jersey, May 1989

Dear Mr. Schnitzer:

In response to your letter and comments dated 28 September 1989 concerning the referenced site, we are pleased to submit a new document titled "Risk Assessment Report, UOP Site, East Rutherford, NJ, Volume 2" as prepared by ENSR Consulting and Engineering and dated November 1989.

In the revised report and this letter, we respond to all the technical comments and questions in your letter. The revised report contains the vast majority of our responses identified by revision bars in the righthand margin of the text. This letter addresses information and ideas on certain comments that are more appropriately handled in a letter of transmittal.

Here are our thoughts using the same comment numbers as in your letter:

<u>Response to General Comment 1</u>

Overall, we believe that our original Risk Assessment Report analyzed the most important exposure pathways for the NJDEP-approved list of Indicator Species. However, in response to your letter, we have extended our analyses in the Risk Assessment Report to include qualitative and quantitative discussions of direct exposures to pore water and sediments. Chapter 5 now includes quantitative toxicity quotient analyses of exposures to pore water for the Indicator Species. This chapter also includes qualitative discussions on pore water and sediment exposure pathways.

Toxicological effects of direct exposure to sediments and ingestion of sediments are addressed primarily by reviewing laboratory studies of sediment toxicity. Where laboratory benthic organisms or natural populations are exposed to contaminated sediment, effects of direct exposure and ingestion are not distinguishable. In both cases, however, it is the total exposure that induces effects, and so the laboratory studies reviewed here should be relevant and meaningful for the UOP site.

Response to Comment 9:

The following description for the chromium VI method used was provided by the laboratory responsible for the analyses, Cambridge Analytical Associates (CAA):

> "Hexavalent Chromium Analyses:
>
> Samples for hexavalent chromium analysis were prepared according to New Jersey DEP methods summarized in NJDEP Lab Report #0002, prepared by Tom Costantino and Denise Costantino of NJDEP Bureau of Radiation and Inorganic Analytical Services. In summary, an aqueous leacheate is prepared from the soil/sludge sample. This is designed to extract all soluble hexavalent chromium from the sample. Following extraction of the hexavalent chromium, the leacheate is treated using EPA SW-846 Method 7195, 7196 or 7197. Method 7195 (coprecipitation) was used on these samples as it is the same method used in the past on samples from this site, and it is the best option when leacheates are colored. Final analysis was performed by Inductively Coupled Plasma Emission Spectroscopy (ICP-EPA Method 6010)."

The question of which chromium VI extraction procedure to use was and is the object of vigorous debate within NJDEP and U.S. EPA. The extraction method selected, NJDEP Lab Report #0002, was based on numerous communications between ENSR staff and researchers at NJDEP, U.S. EPA, universities, and in industry. We are involved with chromium issues at several sites and therefore keep abreast of this evolving issue. A copy of Lab Report #0002 is included herein as Attachment A.

Response to Comment 10:

The benchmark values in the tables in Chapters 5 and 6 were double-checked, and in a few instances, some modifications were made. There are differences in the benchmark values chosen between the chapters, as the methodologies applied in each chapter (Chapter 5: Toxicity Quotient Method, Chapter 6: AEE Method) necessitate

different benchmark selection criteria. These criteria are enumerated and discussed in each chapter.

Response to Comment 12:

As discussed with Dr. David Ludwig of NJDEP on November 3, 1989, ENSR called the U.S. Fish and Wildlife Service Patuxent Wildlife Research Center in an effort to obtain better documentation on the daily feeding ration for the mallard duck. Dr. Gary Heinz at the Center conducts toxicological research on mallards. He estimates that mallards eat approximately 300 grams (wet weight) of food per day in the natural environment. He bases this estimate on his laboratory's daily feeding ration of 100 grams (dry weight) for mallards in captivity. According to Dr. Heinz, mallards principally consume aquatic vegetation, seeds and insects. He states that mallards eat very few fish because they are not physically equipped to catch them effectively.

Based on this conversation, ENSR has modified the mallard risk assessment in Chapter 7 to use a value of 300 rather than 600 grams of vegetation as the daily food ration. The assumptions regarding the dietary components of the mallard's intake have not been changed, but it should be recognized that the assumption of fish consumption is a conservative-case scenario.

Response to Comments 14 and 15:

The text in Chapter 7 has been modified to clarify the reasoning for the application of safety factors and the logic supporting the magnitude of those safety factors.

Response to Comment 16:

We cannot resolve the direct contradictions between recognized experts - we simply do not know whether mink exist in the Meadowlands or not. However, in a cooperative spirit of "what if" analyses, this Ecological Risk Assessment Report considers the possible presence of mink. However, we agree with the NJDEP's assertion that if the mink issue becomes critical to decisions about the site, then additional study will be necessary.

Response to Comment 21:

NJDEP points out that either an estimate of a higher percentage of fish consumption by mink or inclusion of other foods that are potentially PCB-contaminated would contribute a higher PCB exposure to the mink. Because either of these two scenarios would present higher exposure to mink than that calculated by ENSR, ENSR has

redefined the mink assessment in this report as a conservative-case estimate, rather than a worst-case estimate.

ENSR believes the mink diet assessed in this report is a valid conservative-case scenario. References obtained by ENSR indicate that mink diet in a marshland environment consists of approximately one-third fish, hence this amount was used as a realistic estimate in the risk assessment. Other items in the mink diet consist of a variety of small animals and aquatic invertebrates. The potential concentration of PCBs in these food sources is difficult to quantify due to a sparsity of bioconcentration data for these species.

Even though a worst-case scenario cannot be quantified for mink exposure to PCBs, ENSR believes that the risk assessment represents an unlikely and conservative scenario, and thus additional safety factors are not needed in the development of remediation goals.

Response to Comment 25:

Extraction and holding times were met for all sediment sample analyses except one. Sample Da was received by the laboratory 26 hours after sampling and thus exceeded the 24-hour holding time. The sample was analyzed as soon as it was received; thus the effect on the Chromium VI results is believed to be small.

Fish analyses do not have required holding and extraction times, but it would be difficult to justify longer times for this complex media than are appropriate for soils. The fish were extracted for PCBs from 29 to 31 days after receipt and were prepared for mercury analyses at 39 days after receipt. For comparison, the required times for soils are: 14 days for PCBs and 28 days for Hg. The effect on PCB results is believed to be negligible; PCBs are stable and would not leave the matrix. Mercury results could be reduced because the mercury can volatilize. The magnitude of the effect on these mercury results is unknown.

Response to Comment 28:

The graphs in Appendix C represent data in tables in Appendix D. Language has been added to the text of Appendix C to indicate the exact location of the data for each graph.

Response to Comment 31:

An assessment of ingestion of crabs caught from Ackerman's Creek was performed under the assumption that this scenario could possibly occur. However, for the reasons given in the assessment,

we do not believe that people crab directly in Ackerman's Creek. Our judgment in this matter should be included in the report because there is no extensive survey of fishing behavior in Ackerman's Creek.

Response to Comment 32:

The NYDEC (1981) document does not address the frequency of consumption of crab hepatopancreas. The analyses provided do not indicate whether fat associated with muscle tissue is assumed to be part of the "muscle tissue" or "soft body parts". In the risk assessment, it is assumed that PCBs partition equally in all soft body parts. Thus the concentration of PCBs per unit weight of soft-body parts is identical in muscle tissue and fat, including the fat in shell points. If this fat is eaten, it is assumed to have the same concentration of PCBs as muscle tissue. A copy of NYDEC (1981) has been ordered and will be submitted to NJDEP.

Response to Comment 33:

ENSR does not assume that the compounds lacking BCFs for crab tissue are not bioconcentrated by crabs. For several compounds, BCFs are not available for crabs, nor are they available for a closely-related species such that a BCF can be estimated through a comparative ratio approach. Although BCFs cannot be quantified for these chemicals, a qualitative assessment has been added to the revised Part II: Human Health Food Chain Assessment report.

Response to Comment 34:

If the NJDEP has information on the partitioning of PCB into shell versus soft body parts, ENSR will change the assumption about 1:1 partitioning. ENSR is unaware of any information on such partitioning and used best professional judgment in deriving the assumption of 1:1 partitioning.

Response to Comment 35:

As indicated in NJDEP's comments, reference doses are typically based on administered dose and the NJDEP states that absorption adjustment factors are not appropriate. ENSR disagrees with this statement. Absorption adjustment factors are justified regardless of whether the reference dose is based on an administered or an absorbed dose. In applying a reference dose, several factors must be accounted for. The critical ones are: differences in metabolism between the test species and humans; differences in the route of experimental exposure and the exposure in the real world; and differences in availability of the compound from the experimental

vehicle versus the environmental media in the real world. When such differences exist, the reference dose must be adjusted, regardless of whether it is based on administrated or absorbed dose.

The need for modifying reference doses using absorption adjustment factors is recognized by some regulatory agencies including the Massachusetts Department of Environmental Protection (relevant portions of their risk assessment guidelines are in Attachment B) and U.S. EPA Region I (relevant portions of their risk assessment guidance are in Attachment C).

Finally, as a practical point, please note that even if all the absorption adjustment factors were changed to a value of 1, the hazard index would still remain below 1 and the conclusion of the human health food chain assessment would not change.

Change to AEE Method:

An additional revision was made to Chapter 6 and Appendix H concerning the results of the AEE method. After submission of the Ecological Risk Assessment, an error was found by ENSR in Suter et al.'s (1986) published paper on the AEE method. This error was confirmed through direct communication with Suter. To correct this error, the total variance, and thus the probability, had to be recalculated for all the species-compound analyses involving the double extrapolation step. The recalculations have been performed resulting in revisions to Table 6-2 in Section 6 and several tables in Appendix H of the Ecological Risk Assessment. However, the results do not change significantly.

Please call Mark Kamilow, of Allied-Signal at (201) 455-2119 if you have any questions regarding this submittal.

Sincerely,

Michael C. Worthy

Michael C. Worthy, P.E.
Project Manager

Wm A. Duvel

Dr. William A. Duvel, Jr., P.E.
Vice President

cc: Mark Kamilow, Allied-Signal
Janet Feldstein, U.S. EPA

054584.NAB, 6020-006-270

ATTACHMENT A

METHOD FOR THE EXTRACTION OF CR(VI) IN NEUTRAL MEDIA

METHOD FOR THE EXTRACTION OF CR (VI) IN NEUTRAL MEDIA

Lab Report #0002

The following method is under evaluation by the New Jersey Department of Environmental Protection, Bureau of Radiation and Inorganic Analytical Services, for use with Cr(+6) contaminated non-aqueous materials. The authors of this method withhold comment regarding its universal application until research has validated its performance. The method is provided for informational purposes only.

METHOD FOR THE EXTRACTION OF CR (VI) IN NEUTRAL MEDIA

Lab Report #0002

1.0 Scope and Application

1.1 This method is designed to extract soluble hexavalent chromium from soils, sludges and solid wastes.

2.0 Summary of Method

2.1 This method utilizes a neutral extraction with deionized water to solublize all available hexavalent chromium prior to its potential reduction to a trivalent state.

2.2 The extraction technique is not designed as a rigorous digestion, and is not recommended for use in the extraction of insoluble hexavalent chromium material.

3.0 Interferences

3.1 Slightly soluble trivalent chromium may be partially oxidized while hexavalent chromium may be slowly reduced under conditions of rigorous soil extraction or temperature during the extraction. Strongly buffered wastes may resist extraction and decrease extraction efficiency. Organic sludges and polymeric compounds may not allow hexavalent Cr to be solublized by this method.

4.0 Apparatus

4.1 125ml Erlenmeyer flasks, borosilicate.
4.2 Centrifuge
4.3 Centrifuge tubes-100 ml. capacity
4.4 100 ml volumetric flasks, borosilicate
4.5 Magnetic stir plate
4.6 One-inch stir bars
4.7 1.0 liter volumetric flask
4.8 Glass funnel filtration assembly with glass filtration flask
4.9 Millipore HAWP 047 00 filters and AP15 047 00 prefilter or equivalent (0.45 micron).

5.0 Reagents

5.1 Potassium dichromate solution (1.0 ml.= 50ug.Cr): Dissolve 141.4mg. of dried potassium dichromate in deionized water in a 1.0 liter volumetric flask and dilute to the mark.

5.2 Deionized water

5.3 Nitric acid, reagent grade

6.0 Sample Handling and Preservation

6.1 Collect the soil, sludge or waste samples in a 250ml brown-glass bottle which has been rinsed with deionized water and dried. Store at 4 degrees Centigrade from point of collection.

6.2 Analyze samples immediately (within 24 hours) to retard adulteration of hexavalent chromium by chemical activity.

7.0 Procedure

7.1 Weigh out 10 grams of well-mixed sample into a 125 ml. Erlenmeyer flask rinsed with 1:1 nitric acid, then rinsed several times with deionized water

7.2 Add 90 ml. of deionized water.

7.3 Mix with magnetic stirring at a moderate rate for 15 minutes.

7.4 Remove flask from stirrer. Remove stirring bar and lightly rinse the bar off into flask contents with deionized water. Transfer contents to a 100ml centrifuge tube rinsed with 1:1 nitric acid, then rinsed several times with deionized water.

7.5 Centrifuge at 400 rpm for 10 minutes. Decant liquid into a 100 ml volumetric flask. If sample extract is turbid, filter using a 0.45 micron filter. Transfer filtrate from flask to 100ml volumetic flask and fill to correct volume. Stopper and mix well.

7.6 Select a method for determining the concentration of hexavalent chromium in the extraction and analyze immediately (7195, 7196 or 7197). Method 7196 is the recommended method due to sensitivity and reproducibility.

7.7 Calculate the amount of hexavalent chromium in the sample in mg/kg. Sample calculation would be as follows:

An extracted 10 gram soil sample was found to contain 7.3 ug/l in the final extraction, or 7.3 mg/10grams of hexavalent chromium in the soil. 7.3 mg/L in 0.01 kg sample extracted in 0.1L=73mg/kg Cr(VI).

8.0 Quality Control

Quality control consists of duplicate and spiked analyses of samples. A deionized water blank should be analyzed for hexavalent chromium to confirm the integrity of deionized water used in this extraction process.

8.1 Duplicate Analysis

A duplicate extraction and analysis will be performed at the minimum rate of one every 20 samples. Percent difference between duplicates should be <20%.

8.2 Spiked Analysis

An aliquot of soil spiked with Cr(VI) in a concentration sufficient to verify absence of interference will be performed at the minumum rate of one every 10 samples. The spike solution is added to the sample at Step 7.2. Acceptable recovery is between 80-120%. If recovery is not acceptable, data is considered invalid. Further analysis may be required to confirm interferences or obtain acceptable spike recovery.

Method developed by Tom Costantino, Denise Costantino.
New Jersey Department of Environmental Protection
Bureau of Radiation and Inorganic Analytical Services
380 Scotch Rd.,CN 411, W.Trenton, New Jersey 08628
(609) 530-4100

ATTACHMENT B

GUIDANCE FOR DISPOSAL SITE RISK CHARACTERIZATION AND RELATED PHASE II ACTIVITIES IN SUPPORT OF THE MASSACHUSETTS CONTINGENCY PLAN

ANDERSON

GUIDANCE FOR DISPOSAL SITE RISK CHARACTERIZATION

AND RELATED PHASE II ACTIVITIES

- IN SUPPORT OF THE MASSACHUSETTS CONTINGENCY PLAN

MASSACHUSETTS DEPARTMENT OF ENVIRONMENTAL QUALITY

ENGINEERING

OFFICE OF RESEARCH AND STANDARDS

$8.10

May 17, 1989

#70

Published by

Office of the Massachusetts Secretary of State, Michael J. Connolly, Secretary

E. ESTIMATION OF AVERAGE DAILY DOSE (ADD)

This section applies to risk assessments conducted per Method 3.b. and to risk assessments in which guidelines are developed per Method 3.a.

These operations should be used in calculating doses for the estimation of total site risks (Method 3.b.). These expressions for daily dose should also be used to calculate guidelines for Method 3a. as described in Appendix A. [However, if the risk estimate or proposed guideline is based upon a chemical's Unit Risk (See Section III.B.3.- Unit Risk) and a lifetime average concentration then a Lifetime Average Daily Dose would not be calculated.]

The US EPA Exposure Assessment Group defines exposure as the amount of material contacted and available for absorption. The dose is defined as the amount of material actually absorbed into the body. The application of a Bioavailability Adjustment Factor (BAF) or a "Relative Absorption Factor" (an EPA Region I term) is generally assumed to convert an exposure to a dose, although the actual role of such a factor is discussed in detail in Appendix B, Section K of this document. Each equation given in the following pages includes a Bioavailability Adjustment Factor, and, under certain situations, the result will in fact be an average daily exposure rather than a dose. For simplicity, we have retained the term "average daily dose" to apply to the product of an "average daily exposure" and a Bioavailability Adjustment Factor.

The equations presented below outline the procedure for the calculation of an Average Daily Dose of an OHM. Depending upon the period of time over which the total intake of contaminant is averaged (Averaging Period, or AP), the calculations may yield:

- Lifetime Average Daily Dose (LADD): The averaging period for a LADD is 70 years. A LADD should be calculated in order to estimate carcinogenic risk. (The exception being the use of Unit Risk, as noted above.) The actual exposure period could range from a day to an entire lifetime.

- Average Daily Dose, Chronic (ADD_C): The averaging period for an ADD_C may range approximately from several months duration to somewhat less-than-lifetime. The chronic daily dose is used to evaluate non-carcinogenic health effects associated with long-term exposure.

- Average Daily Dose, Subchronic (ADD_s): The averaging period for an ADD_s may range approximately from a few days to several months.. The subchronic daily dose is used to evaluate non-carcinogenic health effects associated with short-term or seasonal exposures.

- Average Daily Dose, Acute (ADD_a): The averaging period for an ADD_a is generally one day or less. The acute dose is used to evaluate non-carcinogenic health effects associated with onetime or episodic exposures.

The Averaging Period (AP) used in the following equations may not always be equal to the Duration of the Exposure Period (D_2).

Example: The risk assessor is asked to evaluate the carcinogenic risk associated with a five year exposure to chemical A. Estimation of carcinogenic risk requires the calculation of a Lifetime Average Daily Dose. Thus, the Averaging Period used for calculating the LADD would be 70 years while the Duration of the Exposure Period would be equal to 5 years.

The risk assessor is also asked to evaluate the likelihood of non-carcinogenic health effects associated with a five year exposure to chemical A. The assessor may calculate an Average Daily Dose Chronic (ADD_c) where AP = 5 yrs and D_2 = 5 yrs.

Frequently, it will be necessary to calculate several different daily doses of a chemical to a receptor in order to evaluate all the exposures scenarios which have been identified as being of concern.

The general form of the ADD equation is presented as:

$$ADD = \frac{\text{(Total Amount of OHM Taken In)}}{\text{(Body Weight}_{avg}\text{)(Averaging Period)}}$$

The doses of an OHM received via different routes of exposure are assumed to be additive unless there is evidence otherwise.

General equations for the calculation of Average Daily Dose are presented in this section for some frequently encountered exposure pathways. These equations are not intended to represent the universe of potential models and they must be tailored to site-specific conditions. It is expected that additional exposure pathways may be identified, and an ADD may be calculated, using appropriate models, for each receptor of concern.

The Daily Dose(s) of each OHM calculated for each potential receptor may be summarized in the Phase II Report. In addition, summary tables presenting the equations and the exposure assumptions used to calculate the daily dose should be presented and well referenced. (See Appendix G)

1. Air

The equations which follow represent estimations of the actual dose of OHM received at an exposure point. The equation for calculating the dose associated with the Allowable Threshold Concentration (ATC) may be found in Appendix J.

a. Inhalation of OHM Contaminated Particulates

Airborne particulates (fugitive dust) may carry OHM to receptors both on- and off-site. An Average Daily Dose due to the inhalation of OHM contaminated particulates (ADD_{inhp}) may be calculated:

$$ADD_{inhp} = \frac{[RP]_{air} * [OHM]_{part} * VR * BAF * D_1 * D_2 * F * C}{BW_{avg} * AP}$$

Where:

$[RP]_{air}$ = Representative concentration of respirable particulates (PM_{10}) in the air at the Exposure Point during the exposure event. (dimensions: mass/volume)

$[OHM]_{part}$ = Representative concentration of OHM in the respirable particulates at the Exposure Point during the period of exposure. (dimensions: mass/mass)

VR = Daily respiratory volume for the receptor of concern during the period of exposure. (dimensions: volume/time)

BAF = Bioavailability Adjustment Factor

F = Number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: events/time)

D_1 = Average duration of each exposure event (dimensions: time/event)

D_2 = Duration of the exposure period (dimension: time)

BW_{avg} = Average body weight of the receptor of concern during the averaging period (dimension: mass)

AP = Averaging Period (dimension: time)

C = Appropriate units conversion factor(s)

b. Inhalation of Gaseous OHM

Gaseous Oil or Hazardous Material (for example, OHM volatilized from contaminated soil) may be inhaled both on- and off-site. An Average Daily Dose due to the inhalation of gaseous OHM (ADD_{inhg}) may be calculated:

$$ADD_{inhg} = \frac{[OHM]_{air} * VR * BAF * D_1 * D_2 * F * C}{BW_{avg} * AP}$$

Where:

$[OHM]_{air}$ = Representative concentration of gaseous OHM in the air at the Exposure Point during the period of exposure (dimensions: mass/volume)

VR = Daily respiratory volume for the receptor of concern during the period of exposure. (dimensions: volume/time)

BAF = Bioavailability Adjustment Factor

F = Number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: events/time)

D_1 = Average duration of each exposure event (dimensions: time/event)

D_2 = Duration of the exposure period (dimension: time)

BW_{avg} = Average body weight of the receptor of concern during the averaging period (dimension: mass)

AP = Averaging Period (dimension: time)

C = Appropriate units conversion factor(s)

ADD_{sw}	-	Total Average Daily Dose received via exposures to contaminated surface water: $ADD_{sw} = ADD_{swd} + ADD_{swi} + ADD_{swih}$
ADD_{swd}	-	Average Daily Dose received via dermal contact with contaminated surface water
ADD_{swi}	-	Average Daily Dose received via incidental ingestion of contaminated surface water
ADD_{swih}	-	Average Daily Dose received via inhalation of contaminants volatilized from surface water
AF	-	Fraction of OHM in soil absorbed through the skin (unitless)
AIC	-	Allowable Intake, Chronic
AIS	-	Allowable Intake, Subchronic
AP	-	Averaging Period (units: days)
Background	-	The level of oil or hazardous material in the environment which would exist in the absence of the disposal site
BAF	-	Bioavailability Adjustment Factor (unitless)
BW_{avg}	-	Average Body Weight of the receptor of concern during the period of exposure (units: mass)
C	-	Appropriate units conversion factor
D_1	-	Average duration of each exposure event (units: hours/event)
D_2	-	Duration of the exposure period (units: days)
DEQE	-	The Massachusetts Department of Environmental Quality Engineering
Disposal Site	-	Any structure, well, pit, pond, lagoon, impoundment, ditch, landfill or other place or area, excluding ambient air or surface water, where uncontrolled oil or hazardous material has come to be located as a result of any spilling, leaking, pouring, abandoning, emitting, emptying, discharging, injecting, escaping, leaching, dumping, discarding, or otherwise disposing of such oil or hazardous material. The

ATTACHMENT C

SUPPLEMENTAL RISK ASSESSMENT GUIDANCE FOR THE SUPERFUND PROGRAM

United States
Environmental Protection
Agency, Region I

EPA 901/5-89-001
June 1989

Supplemental Risk Assessment Guidance for the Superfund Program

PART 1 - Guidance For Public Health Risk Assessments

PART 2 - Guidance For Ecological Risk Assessments

DRAFT FINAL

June 1989

**Prepared by the U.S. EPA Region I
Risk Assessment Work Group**

4.0 EXPOSURE ASSESSMENT

The exposure assessment identifies pathways and routes by which site contaminants may reach receptors and contaminant concentrations at the points of exposure. This information is then integrated with known or predicted activity patterns such that an estimated exposure dose for each pathway and each contaminant of concern is generated.

In this section, general methods and approaches used for conducting an exposure assessment are described. Four basic elements of any exposure assessment are identified below along with the section in which each is addressed.

1. Identification of exposure pathways (Section 4.1)

2. Estimation of exposure point concentrations for each selected pathway (Section 4.2)

3. Estimation of exposure dose for each selected pathway (Section 4.3)

4. Development of exposure scenarios (Section 4.4)

Exposure pathways may involve several media, and the exposure assessment must address the exposure scenarios for each media. Methods and approaches for exposure evaluations specific to each media are presented in Sections 4.5 (Groundwater), 4.6 (Surface Water), 4.7 (Soil) and 4.8 (Air).

4.1 IDENTIFICATION OF EXPOSURE PATHWAYS

As described in the Superfund Public Health Evaluation Manual (USEPA 1986), an exposure pathway consists of four elements:

1. A source and mechanism of chemical release into the environment.

2. An environmental transport medium for the released contaminant (e.g., air) and/or a mechanism of contaminant transfer from one medium to another (e.g., deposition of particles onto soil).

3. A point of potential contact of humans or biota (the receptors) with the contaminated medium (the exposure point).

4. An exposure route (e.g., inhalation) at the exposure point.

When all four of these elements are present, an exposure pathway is considered "complete". For each public health risk assessment, exposure pathways considered complete should be identified by the risk assessor. Some of the more common exposure pathways quantitatively evaluated in risk assessments are briefly identified below by the environmental medium of exposure. Note that there often are many potential pathways of exposure, only some of which are presented below.

Air: Both indoor and outdoor exposures could result from inhalation of particulate matter and/or gaseous compounds from a nearby site.

Groundwater: Exposures to chemicals migrating through groundwater from a site could result in the ingestion of contaminated water or in the inhalation of volatile organic compounds resulting from volatilization from household water usage.

Surface Water: Exposures to chemicals in surface water could occur as a result of incidental ingestion, dermal absorption, or inhalation of volatile organic compounds released from surface water.

Soil: Exposures to contaminants in soil could occur as a result of direct contact with soil and subsequent incidental ingestion or dermal absorption of these contaminants. Chemicals released from the soil into the air could also be a pathway of concern if the chemicals were subsequently inhaled.

Food Chain: Exposures to chemicals could also occur as a result of ingestion of produce, fish, or game in which site-related chemicals may have bioaccumulated (e.g., PCBs) or upon which such chemicals could have been deposited.

Based on the list of exposure pathways, a screening process should be performed to determine which pathways should be evaluated in detail in the risk assessment. This screening process has not been precisely described in EPA guidance manuals but should be based on consideration of several important factors. These include whether there is sufficient data to evaluate exposure, the frequency and duration of likely exposures, and the relative contribution of the exposure to the total site risk.

The number of pathways ultimately selected for detailed evaluation in a risk assessment varies depending upon site-specific conditions such as accessibility and availability of sampling data. In

general, anywhere from two or three to ten or more pathways may be selected for detailed evaluation. In some cases, it will be extremely difficult to quantitatively evaluate a potentially important pathway due to inadequate sampling data and the availability of adequate fate and transport models. In these cases, it may be appropriate to qualitatively evaluate the pathway in the risk assessment. It may also be appropriate to obtain additional site information or data to further evaluate potential pathways of concern. This may be done as part of a phased Remedial Investigation effort.

4.2 ESTIMATION OF EXPOSURE POINT CONCENTRATIONS

The exposure point concentration refers to the amount of chemical in an environmental medium to which one may be exposed (i.e. the measured concentration in the medium of concern). The exposure point concentration is usually expressed in units of mass per unit volume (mg/l or mg/m^3) or mass per unit weight (mg/kg). Exposure point concentration should be differentiated from the term exposure dose - a term used to describe the resulting exposure from a particular concentration, expressed in units of mg/kg/day (see Section 4.3).

Average and maximum chemical concentrations (exposure point concentrations) should be developed for each exposure pathway based on site sampling data or on modeling results. The sampling data used to characterize exposure point concentrations should be representative of the specific exposure pathway of concern. In some instances (if there is a discrete "hot spot" of contamination or if there are separate areas where different exposures are expected), it may be appropriate to identify exposure point concentrations for discrete areas of a site. In this case, a single exposure pathway may be evaluated more than once using exposure point concentrations derived from different portions of the site sampling data.

4.3 ESTIMATION OF EXPOSURE DOSE

To quantitatively estimate the risk posed by the presence of one or more contaminants in an environmental medium, one must estimate the exposure dose of each contaminant. The exposure dose is similar to the administered dose of a laboratory experiment from which cancer potency factors and reference doses are usually derived.

Other terms important to understand include absorbed dose, metabolized dose, and body burden. Each of these terms relate the exposure dose to a particular biological function. The absorbed dose corresponds to the amount of the exposure dose that is

absorbed either through the skin, the gastrointestinal tract, or the lungs. Similarly, the metabolized dose describes the amount which is metabolized. The metabolized dose is an important parameter when the toxic compound is thought to exert its toxic effects via a metabolite.

Finally, the body burden is used to describe the level of a particular compound in a biological tissue or fluid. It is influenced by the exposure dose, absorption, metabolism, and excretion. The body burden is ideally the best measure of exposure to contaminants. Such measurements are however, beyond the scope of CERCLA risk assessments in which data describing exposure point concentrations is usually all that is available. Thus estimation of exposure dose forms the basis for the quantitative risk assessment process.

The general equation that may be used to calculate an exposure dose is the total amount of chemical exposure divided by body weight and the period over which exposure is to be averaged. The exposure dose is also normalized by the use of a relative absorption factor (RAF) which takes into account differences in absorption between the medium and the exposure route of concern, and the medium and exposure route used in the laboratory study from which the cancer potency factor or reference dose was derived. The total amount of chemical exposure is estimated from information on the chemical concentration in the environmental medium of concern, the frequency and duration of exposure, and the magnitude of exposure to the medium of concern. Thus the general equation for computing the exposure dose is as follows:

$$EXP_i = \frac{(C_m)(M_m)(F)(D)(RAF_{mi})}{(BW)(AVG)(365\ \text{days/year})}$$

where

EXP_i = average daily exposure dose for pathway i (normalized) (mg/kg/day)

C_m = chemical concentration in medium m (e.g., mg/kg)

M_m = magnitude of exposure to medium m per day of exposure (e.g., kg/day of exposure)

F = frequency of exposure (days of exposure/yr)

D = duration of exposure (yrs)

RAF_{mi} = relative absorption factor for medium m and pathway i (unitless)

BW = body weight (kg)

AVG = number of years over which exposure is averaged (70 years for carcinogenic effects, D for noncarcinogenic effects)

In estimating exposure doses, the time over which the total dose is averaged differs when assessing carcinogenic and noncarcinogenic effects as discussed in Section 5.0. For carcinogenic effects, the total dose is averaged over a 70 year lifetime and over 365 days per year to yield an average daily lifetime dose. For noncarcinogenic effects, the total dose is averaged over the total period of exposure (i.e. 10 years) and over 365 days per year. This yields an average daily dose for the period of exposure. Representation of exposure dose in these terms facilitates future estimates of lifetime cancer risk and hazard indices.

For noncarcinogenic effects, the standard averaging approach is most appropriately applied to exposure scenarios in which the frequency of exposure is high, such as several times a week throughout the entire year. For infrequent exposures, this approach may not accurately represent the potential for acute health effects. The implicit assumption to dose averaging is that a high dose over a short period of time is toxicologically equivalent to a low dose over a longer period of time. For many contaminants, acute or subchronic effects may occur from a single high dose or a limited number of moderately high doses. Averaging the total dose over an entire year may thus be inappropriate if it is anticipated that such acute noncarcinogenic health effects may occur. In this case, the risk assessor should evaluate the exposure dose incurred over a short period of time (such as a single exposure event) by making the necessary adjustments to F, D, AVG, and 365 days in the general exposure dose equation. The risk assessor should then compare this exposure dose to a health based criterion that is derived for a shorter exposure period such as a one or ten day Health Advisory.

For some exposure pathways, such as incidental ingestion of soil, the bioavailability of the chemical from the environmental medium to which an individual is exposed should be taken into account as chemicals present in a soil matrix may be less bioavailable than chemicals present in a solvent or in water. The relative absorption factor (RAF) is defined as the ratio of the estimated absorption factor for the site-specific medium and route of exposure to the known or estimated absorption factor from the laboratory study from which the cancer potency factor or the reference dose was derived. The use of this factor allows the risk assessor to make appropriate adjustments to the assessment if the efficiency of absorption is known or expected to differ because of physiological, matrix, or vehicle effects. Relative absorption factors for each contaminant in each medium for each exposure pathway should be derived from the literature. However, due to a lack of data on the absorption of contaminants from soils, Region I has listed some default values for this pathway in Section 4.7.1 and 4.7.2.

To the extent possible, EPA Region 1 has standardized the exposure parameters that should be used in risk assessments prepared for the Regional Office. These parameters (primarily for soils) are

presented in the appropriate exposure sections. Each of the exposure parameters contained in this guidance are subject to some inherent uncertainty as a result of limited experimental data and due to variability in individual activity patterns. While these values have some uncertainty in them, it was the feeling of the Region I Office that in order to afford consistency across risk assessments, it was advantageous to specify parameters to the extent possible. The Regional Office intends to reevaluate these parameters as new information becomes available. However until such time, they are to be used in risk assessments prepared for the Region I Office. The risk assessor should identify the limitations of this method in the risk assessment as these values will ultimately influence the risk estimate and the selection of target clean-up levels.

4.4 DEVELOPING EXPOSURE SCENARIOS

Exposure scenarios are developed to simulate possible exposures in the absence of site specific exposure information and to account for the variability in contaminant distribution and land use. Usually there are two levels to consider in developing exposure scenarios. The first involves identifying present and future potential exposure scenarios. Much of this task involves the identification of present and future land and water use at a site and at adjacent properties and identification of populations of concern. Present exposure scenarios are thus fairly easy to identify though as a word of caution, present exposure scenarios include those that have a present potential of occurring as well as those known to occur. Future exposure scenarios are often more difficult to identify though Region I has maintained the position that future land use at most Superfund sites could be residential. Any deviations from a residential future exposure scenario should be approved by the EPA risk assessment specialist prior to proceeding with the development of the exposure assessment. The outcome of the first level of exposure scenario development should thus identify potential receptor populations and their exposure pathways to contaminated media considering both present and potential future land use. It is therefore not uncommon to identify some exposure scenarios as only being a future potential exposure scenario (i.e., ingestion of ground water) or to have more than one exposure scenario for a particular medium (i.e., present exposure to soils for adults and present exposure to soils for children).

The second level in developing an exposure scenario involves the estimation of a range of conditions characterized by an "average case" and a "reasonable worst case" exposure scenario for each selected exposure pathway. The "average case" exposure scenario combines average values for exposure point concentrations with reasonably conservative exposure assumptions from the sections that

follow. Exposures represented by this scenario may reasonably be assumed to occur. However, the actual level of risk described by the "average case" exposure is likely to be lower than the value reported due to the use of conservatively derived health criteria. The "reasonable worst case" exposure scenario combines maximum values for exposure point concentrations with the reasonably conservative parameters applied to the "average case" estimate. The reasonable worst case scenario is intended to provide an upper bound of the possible risk.

In selecting exposure parameters, (i.e., ingestion rate, relative absorption factor, body weight) the same parameter value should be used for both the average and the reasonable worst case exposure scenarios except for contaminant concentration Many of these parameter values are specified in the exposure sections that follow recognizing that certain parameters (ingestion rate, body weight, and exposure duration) may vary from scenario to scenario according to the population (adult vs. child) and land use (residential vs. industrial) for a given site. Values for the frequency and duration have not been specified in the guidance but should be determined on a case-by-case basis. Once determined, the same frequency and duration of exposure should be applied to both the average and reasonable worst case scenarios. Only when considering present vs. future exposures is there room for flexibility in the values assigned to exposure frequency and duration. Because frequency of exposure is very site specific, the risk assessor should carefully consider the size and location(s) of contaminated area(s) described by the average and maximum contaminant concentrations. Additionally, it is in the risk assessor's best interest to conduct a site visit to gain a better understanding of the site and potential exposure scenarios.

4.5 EXPOSURES VIA GROUNDWATER

There are several routes of exposure to contaminants that may be present in groundwater. These include: (1) direct ingestion of groundwater; (2) inhalation of chemicals released into indoor air as a result of household water use; (3) dermal absorption of chemicals from household water use; and (4) groundwater seepage into basements and subsequent volatilization into indoor air. Indirect pathways of exposure could also occur if contaminated groundwater is used to irrigate gardens and contaminants are subsequently deposited on or accumulated in food crops.

4.5.1 INGESTION OF GROUNDWATER

Exposures via ingestion of groundwater are almost always evaluated in a risk assessment. Exposures can be evaluated under both present site use conditions (e.g., nearby residents with contaminated well water) and under future use conditions (e.g.,

Non-detect (ND): The term used to refer to a chemical that is not present in sufficient quantity to be accurately quantified.

Pica: An abnormal desire to eat substances not normally eaten.

Receptor: An organism that receives, may receive, or has received environmental exposure to a chemical.

Reference Dose (RfD): The RfD is based on EPA's identification of the threshold effects level with an added margin of safety. The RfD represents an estimate of the daily dose level (mg/kg/day) for a particular compound that is likely to be without appreciable risk of deleterious effects when exposure occurs over a given period (usually a lifetime).

Relative Absorption Factor: The ratio of the estimated absorption factor for the site specific medium and route of exposure to the known or estimated adsorption factor for the laboratory study from which the cancer potency factor or reference dose was derived.

Risk: The potential for realization of unwanted negative consequences or events.

Suggested No Adverse Response Level (SNARL): A term formerly used by EPA and the National Academy of Sciences corresponding to a contaminant level in drinking water at which adverse health effects would not be anticipated.

Superfund Public Health Evaluation Database (PHRED): Computer database available through EPA with chemical, physical, and toxicity information on various compounds.

Teratogenicity: The capacity of a phys_cal or chemical agent to cause non-hereditary congenital malformations (birth defects) in offspring.

Toxic/Target-Endpoint: The most sensitive yet significant noncarcinogenic effect caused by the administration of a compound. Examples include enzyme, weight, gross morphological, and functional changes. Also the target endpoint serves as the basis from which some of the noncarcinogenic criteria (RfDs) are derived.

Toxicity: The quality or degree of being poisonous or harmful to plant, animal or human life.

Uncertainty Factor (U.F.): A factor used to account for the interspecies and intraspecies differences, severity of adverse effects, and the adequacy of data when determining the reference dose.

Xenobiotic: A chemical compound that is foreign to a living organism.

PREFACE

In May 1986, UOP Inc. entered into an Administrative Consent Order (ACO) with the New Jersey Department of Environmental Protection (NJDEP) for completing remedial investigations and initiating feasibility studies and remedial actions at the UOP National Priority Listed site in East Rutherford, NJ. In order to organize the investigations and studies, the site was delineated into six areas as described in the general work plan (Geraghty & Miller Inc., September 1986). As a part of the Remedial Investigation process, a human health risk assessment was completed for four of the six site areas: 1, 1A, 2 and 5. These areas represent the terrestrial portion of the site. The first draft of the risk assessment report was submitted to the NJDEP in May 1987, subsequently revised (Revision 1), and re-submitted to the NJDEP in October 1987. In review comments dated March 10, 1988, the NJDEP required that Area 4, representing the on-site tidal stream channels, be incorporated into the risk assessment and also that ecological risks be considered. Further, the NJDEP required that the human health assessment include both direct and indirect exposure pathways from the stream channels to humans.

An assessment of direct exposure pathways from the stream channels was incorporated into the human health risk assessment report which was submitted to the NJDEP as Revision 2 in October 1988. NJDEP comments on Revision 2 were received on April 28, 1989. The work plan was revised, and retitled as: "Risk Assessment Report, UOP Site, East Rutherford, NJ, Volume 1, Human Health Risks." This version was submitted to the NJDEP in June 1989.

A work plan (ENSR, 1988) was submitted to the NJDEP in October 1988 to address indirect exposure pathways from the stream channels and ecological risks. The activities described in the work plan were initiated in October 1988 under NJDEP approval of parts of the plan. NJDEP comments on the plan were

received in February 1989, and these comments were addressed in Revision 1 of the work plan submitted to the NJDEP later the same month. The work described in the plan was completed and the report: "Ecological Risk Assessment Report for the UOP Site, East Rutherford, NJ", was submitted to the NJDEP in May 1989. NJDEP comments on this report were received by letter dated September 28, 1989. The report was revised and retitled for consistency with the human health risk assessment: "Risk Assessment Report, UOP Site, East Rutherford, NJ, Volume 2." This report is provided herein. A summary of the risk assessment volumes and their contents is provided below.

- Volume 1

  Areas 1, 1A, 2, 5: Human health risks for all exposure pathways

  Area 4: Human health risks for direct contact with stream channel sediments

- Volume 2

  Part I: All site areas, Ecological Risks

  Part II: Area 4, Human health risks of exposure to sediments through the food chain.

PART I

ECOLOGICAL ASSESSMENT

EXECUTIVE SUMMARY

This document presents an ecological risk assessment for the UOP site in East Rutherford, New Jersey.

Focussed Environmental Assessment

The assessment encompasses the entire 75-acre site with the exception of the two former wastewater lagoons (known as Area 3) that amount to about one acre. The methodology generally follows the format used in public health risk assessments. The major components in this report include:

- a description of both the Indicator Species and Indicator Compounds selected for this analysis;
- a discussion of exposure including a description of the model used to estimate contaminant concentrations in water;
- a characterization of ecological risks by three methods; and
- a discussion of uncertainties associated with such analyses.

The three ecological risk methods used are the Toxicity Quotient Method and the Analysis of Extrapolated Error Method described by Oak Ridge National Laboratory for aquatic species and a dose-response method for mallard ducks and mink.

Preliminary Survey

A preliminary survey of terrestrial plants and wildlife on the site was conducted in October 1988. The survey of terrestrial animals and both woody and herbaceous vegetation indicated no differences between study and reference areas that might be associated with environmental impact. A study of aquatic invertebrates, however, indicated a difference in

composition of macroinvertebrates between the areas and no difference in zooplankton composition.

During the preliminary survey, mummichog (Fundulus heteroditus) tagging and recapture studies were initiated. These studies were completed in November 1988 during an effort called the Focused Environmental Assessment. Results indicated a mummichog population of over 57,000 fish on the site. Data showed that the fish range at least 1,000 m.

During October and November, 1988 a Focused Environmental Assessment was completed. This Assessment involved sampling stream sediment and fish (Fundulus heteroclitus) on the site and in a reference area across Berry's Creek. Mummichogs (Fundulus heteroclitus) and sediments collected in the Focused Environmental Assessment were analyzed for PCBs, chromium, and mercury. Ranges of fish body burdens averaged for four different channels on-site and in the reference area are as follows: PCBs, 31-89; total chromium, 1.4-28; and total mercury, 1.2-2.3 all in µg/g-dry weight (ppm-dry weight).

Ranges of sediment concentrations for four different channels on-site and in the reference area are: PCBs, 0.69-123; total chromium, 6.6-26,900; chromium (VI), undetectable - 64; and total mercury, 5.9-150 all in µg/g-dry weight (ppm-dry weight). Sediment concentrations of PCBs and chromium are highest at the most instream portion of the study site and lowest at the reference area. The concentrations of mercury in the reference area are higher than on site, which is indicative of an off-site source. Comparison of sediment to fish tissue concentrations for these chemicals suggests little or no relationship for mercury and chromium, and a positive relationship for PCBs.

Indicator Compounds and Indicator Species

Selection of Indicator Compounds is based on: (1) concentrations of chemicals reported on the site in both sediments and water, (2) the frequency of detection for the

contaminants, (3) federal Ambient Water Quality Criteria, and (4) knowledge of potential toxicity of the chemicals. The final selection includes both metals and organic compounds. The 11 Indicator Compounds for the ecological risk assessment are cadmium; chromium; mercury; lead; zinc; 1,2-dichlorobenzene; 1,2-diphenylhydrazine; di(2-ethylhexyl)phthalate; PCBs; phenol; and total PAHS.

Based on the preliminary survey, the ecological risk assessment is limited to the tidal channels and surrounding area on the site. Indicator Species for the channel habitats are selected to represent a variety of taxonomic groups and life histories. The seven indicator species are Chironomidae or non-biting midges, *Palaemonetes pugio* or grass shrimp, *Congeria leucophaeta* or platform mussel, *Callinectes sapidus* or blue crab, *Fundulus heteroditus* or mummichog, *Anas platyrhychos* or mallard duck, and *Mustela vision* or mink.

Exposure Assessment

For the aquatic species, the route of exposure to the Indicator Compounds is assumed to be from the column water and the sediment pore water. These species are the midges, grass shrimp, platform mussel, blue crab, and mummichog. For at least some of these species and chemicals, published studies indicate that uptake comes primarily from column water. For many of the Indicator Compounds, however, concentrations in the tidal channels are below test detection limits. To provide estimates of chemical concentrations and a measure of uncertainty associated with those estimates, column water and pore water contaminant concentrations are modeled.

The modeling methodology begins with the measured contaminant concentrations in sediment. Equilibrium relationships are used to compute the concentrations in pore water surrounding the contaminated sediments. Further modeling simulates the mixing of pore water into the stream channel water column. This algorithm provides the contribution of

dissolved contaminants to the water column. A second analysis considers the contribution of organic contaminants adsorbed to suspended sediment. The water column concentration for this contribution was estimated by using suspended sediment values from Berry's Creek and assuming the suspended sediments are equal in contaminant concentration to the bed sediments. The two concentrations, dissolved and adsorbed, are added to arrive at a total concentration in the water column for each contaminant.

Toxicity Quotient Method

The modeled water concentrations (column and pore water) are then compared to toxicological benchmarks in the Toxicity Quotient Method (QM) described by Oak Ridge National Laboratory. The QM is simply an arithmetic comparison made by forming a ratio of an expected environmental concentration (EEC) to a benchmark concentration (BC) from a toxicological study.

First, modeled column water concentrations of Indicator Compounds are used as EECs. Three types of BCs are used: Ambient Water Quality Criteria prepared by the USEPA and both acute and chronic test results for the Indicator Species or their close surrogates. Results indicate that concentrations of mercury, zinc, and PCBs in column water exceed one or more toxicological benchmark concentrations or Ambient Water Quality Criteria.

Second, the toxicity quotient analysis is also conducted using modeled sediment pore water concentrations of Indicator Compounds as the EECs. Results indicate that pore water concentrations of cadmium, lead, mercury, zinc, phenol, and PCBs exceed the Ambient Water Quality Criteria and/or one or more of the toxicological benchmark concentrations. However, this comparison may not be accurate since these standards and benchmarks are based on an ambient open water environment, not on a sediment/interstitial water environment.

Analysis of Extrapolation Error Method

For several of the the Indicator Compounds for which the Toxicity Quotient Method indicates the modeled water column concentrations are above the toxicological benchmark concentrations, and for chromium, which is below benchmark concentrations but occurs in high concentrations in sediments on the site, a second analytical technique is applied. This technique is the Analysis of Extrapolation Error (AEE), also described by Oak Ridge National Laboratory. AEE is a relatively new method for ecological risk assessment that produces an estimate of risk based on estimates of the toxicological endpoint (or benchmark) and of the uncertainty around that estimated endpoint. The AEE method allows a statement of uncertainty about the environmental concentrations exceeding toxicological benchmarks. The method has two components. First, a mean and a variance for a chronic toxicological benchmark for each Indicator Species are estimated from known toxicity test results and known relationships among these results. The mean is used as the chronic toxicity endpoint and, where necessary, may have been determined by extrapolation between species. The probability that the environmental concentration will exceed the chronic toxicity endpoint is then calculated based on the mean and variance of the endpoint and the mean and variance of the modeled channel water data. The AEE method was developed to be applied to water column exposure only and that is how it is applied in this risk assessment.

The AEE method was applied for the following Indicator Compounds: chromium, mercury, zinc and PCBs and for the following Indicator Species: mummichog, blue crab and grass shrimp. The AEE results are interpreted as the probability or certainty that a species will not experience chronic effects from exposure to a particular compound.

The AEE results indicate that it is probable that the mean modeled water column concentrations of zinc exceed extrapolated chronic toxicity endpoints for mummichog and blue crab.

There is also a relatively high probability that channel water concentrations of PCBs and inorganic mercury will exceed chronic toxicity endpoints, even though the mean modeled water column concentrations are less than the estimated toxicity endpoint values for the assessed species. On the other hand, there is a relatively high probability that modeled water column concentrations of chromium will not exceed the chronic toxicity endpoints for mummichog, blue crab or grass shrimp. For methyl mercury, the AEE results are less clear, although literature studies indicate that the bacteria in Berry's Creek sediments do not methylate mercury. Hence, species on the site may not even be exposed to methyl mercury.

Mallard and Mink Risk Assessments

Risk assessments were conducted for higher trophic level organisms, using mallards and mink as the Indicator Species. The risk assessments are based on comparisons of doses of contaminants ingested to toxicological benchmarks obtained from the literature. Foodchain pathways were evaluated for mallards consuming mercury-contaminated and PCB-contaminated food items and for mink consuming PCB-contaminated food items. PCB body burdens were directly measured in *Fundulus heteroclitus* collected at the UOP site and were found to be in a range of 6.49 mg/kg to 34.08 mg/kg with a mean of 17.6 mg/kg (wet weight). This data is used in the mink and mallard risk assessments as a means of evaluating biomagnification of PCBs via the foodchain.

Mallards eat a variety of foods, so for both mercury and PCBs, two feeding regimes are assumed based on reported diets. The first diet is assumed to be 300 grams of plants, and the second diet is assumed to be 30 grams of mummichogs and 200 grams of plants. Measured mummichog body burdens were used in the analyses. Hazard indices for mercury (defined as dose received/benchmark dose) are 0.03 for the vegetarian diet and 42.3 for the mixed diet. For PCBs, the hazard indices are 0.76

and 0.975 for the vegetarian and mixed diets, respectively. The results indicate that a vegetarian diet does not appear to pose any potential adverse effects, while a mixed diet shows a high potential for adverse effects.

The risk assessment for mink is performed in a similar manner. Only risk from PCBs is considered. Mink eat a variety of foods, and one New York study indicates a diet of about on third fish. Assuming a conservative case, mink eat a diet of one third fish with mean body burden equal to the mean measured body burden of mummichogs on the UOP site. After estimating average mink weight and average total food intake, a maximum possible dosage can be calculated. By knowing the areal ranges that mink travel and comparing that to the area of the channels where the contaminated mummichogs are available, a fraction of diet coming from the site is calculated under the conservative assumption that both a male and female mink have the center of their range on the site next to the channels. Because of the difference in ranges between sexes, the predicted risks are different. The conservative-case estimated exposure dose for a female mink with a den on the site is 109 µg PCB/kg body weight-day, or 70 times higher than the estimated tolerable dose. For the male, however, the conservative-case estimated exposure dose is 0.09 µg/kg-day, or about 17 times less than the estimated tolerable dose. An opposing scenario is that mink do not inhabit the site, and so are unexposed. Because of the small area of the site compared to the Meadowlands and the relative proximity of the site to a major road and human activities, the probability of female mink making dens on the site and experiencing conservative-case conditions remains questionable and may be quite small.

Risk Assessment Uncertainties

There are many uncertainties associated with this risk assessment and the various techniques employed, and using the AEE method, we have explicitly incorporated estimates of the

uncertainty in the analysis. Under the best conditions, ecologists have limited ability to predict future dynamics of single populations and less ability to predict the behavior of complex systems. Each step of this risk assessment makes simplifying assumptions without which the task would be overwhelming, but with which uncertainty increases.

Uncertainties are reviewed in each section, but several sources pervade the entire assessment. First, many concentrations of chemicals in channel water are below test detection limits some or all of the time, and so field concentrations are unknown and model results cannot be completely verified. Second, there is natural variation in almost all aspects of animal physiology and great variability in most physical aspects of estuarine habitats, but some sort of mean or typical condition must be assumed for each method. Then, toxicological results for the Indicator Species exposed to the Indicator Compounds long enough to make an ecologically meaningful observation are not always available, and so these assessments rely on extrapolations, surrogate species, and other assumptions that cannot be completely evaluated.

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1. INTRODUCTION

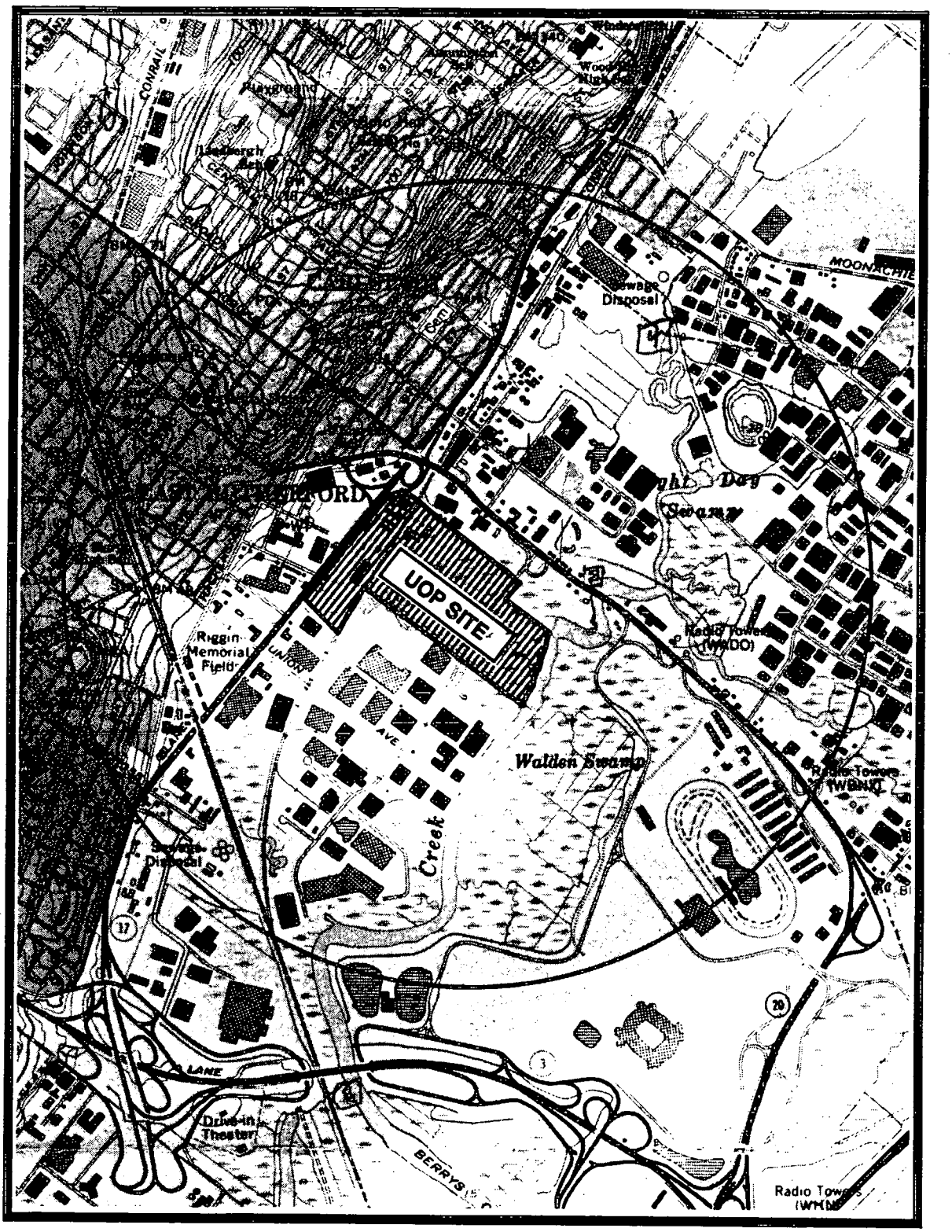
1.1 Background

This report presents the ecological risk assessment for the UOP site, East Rutherford, NJ (see Figure 1-1). The initial requirement to perform an ecological risk assessment was made in a NJDEP letter dated 10 March 1988 and was elaborated on in a letter dated August 22, 1988. During a meeting on 20 September 1988 the specific requirements for the assessment were agreed to by NJDEP and UOP. The assessment requirements were developed into a formal work plan: "Ecological Risk Assessment Work Plan, UOP Site," which was submitted to the NJDEP by ENSR in October 1988. The plan called for field investigations with laboratory analyses and literature based assessments. The field investigations began in October 1988 while the plan was under NJDEP review. NJDEP comments on the Work Plan were received by letter dated 1 February 1989. The Work Plan was revised and submitted on 24 February 1989.

This report has nine chapters and nine appendices. Following Section 1, Introduction, the remaining chapters are:

2. Receptor Characterization
3. Hazard Assessment
4. Exposure Assessment
5. Toxicity Quotient Method
6. Analysis of Extrapolation Error
7. Ecological Risk Assessments for Ducks and Mink
8. Uncertainty Analyses
9. Summary and Conclusions

The first eight appendices contain detailed analyses and data too voluminous to present in the main text. The last appendix contains the references for the report and the appendices.



SOURCE:
USGS Topo Map- Weehawken Quadrangle
New Jersy- New York

0 .5 1 Mile

Figure 1-1
One Mile Radius Surrounding the UOP Site

1.2 Ecological Risk Assessment Methods

The methodology used in this ecological risk assessment follows concepts that are similar to those used in public health risk assessments, but are modified for different receptors as outlined in ORNL (1986) and USEPA (1988). The major components of the methodology include:

- selection of indicator compounds and organisms (Sections 1.3 and 1.4)
- receptor characterization (Section 2)
- hazard assessment (Section 3)
- exposure assessment (Section 4)
- risk characterization (Sections 5, 6 and 7)
- uncertainty analysis (Section 8).

The toxicity quotient method was used to screen for potential adverse ecological effects to the various receptor organisms. This method is a simple, quantitative form of risk characterization which is effective as a screening tool for identifying areas of concern. The method is one in which an estimated exposure concentration, EEC (surface water concentration, pore water concentration, food item concentration), is compared to a toxicological endpoint or benchmark concentration (BC). In this report, conservative assumptions were used in selecting benchmark concentrations as follows:

1) The toxicology literature was reviewed to select endpoint concentrations (acute and chronic) for the same species as the indicator organisms of concern.
2) In the absence of data for the particular Indicator Species, endpoints were chosen for the next most closely related species. If data were available for more than one closely related species at the next taxonomic level, the toxicity value for the most sensitive species was selected.

3) If data were available on the chronic effects of exposure to the contaminant of concern, that quotient was given preference in the analysis and discussion over a quotient developed on the basis of acute effects.

4) Lowest effect concentrations were selected as endpoints whenever possible.

The two concentrations were then compared as a quotient:

$$\frac{EEC}{BC} = \text{Toxicity Quotient}$$

While this is not a risk estimate, because it does not provide a probability of an adverse effect, it does help answer the question: Is the estimated exposure concentration of ecological concern? Ratios for chronic effects which are less than 0.1 are expressed as "no concern", ratio of 0.1 to 10 are expressed as "possible concern", and ratios greater than 10 are expressed as "high concern" (USEPA, 1988).

For the ecological risk assessment, a second method was also applied: the analysis of extropolation error (AEE) method described by Suter et al. (1986) in Barnthouse et al. (1986). This method provides an objective and quantitative estimate of risk that is defined in terms of toxicological benchmarks. One of the advantages of the method is that it allows a statement of uncertainty about the result. AEE was developed using data for fresh water fish and some invertebrates, and so cannot be applied to all species. In this report, AEE is applied to a subset of the data used in the quotient method.

Analysis of extrapolation error is a method that allows the risk assessor to predict the probability that chronic effects would not occur. Because the method allows the calculation of probability that an expected environmental concentration (EEC) will exceed some end point, AEE is considered to be a true risk assessment method because it

yields an estimate of the probability of an event or a condition. AEE allows extrapolation between biological groups and extrapolation between types of tests (e.g., acute and early life stage). The uncertainty associated with the extrapolations is calculated and contributes to the estimation of uncertainty to the risk.

A third method was also applied in this ecological risk assessment to assess species living in the Meadowlands that are not themselves immersed in the aquatic environment but which feed on aquatic plants and animals. A dose-response approach was combined with the toxicity quotient methodology to assess the likelihood of adverse effects on mallard ducks and mink.

1.3 Selection of Indicator Compounds

Indicator Compounds for in-depth analysis of potential ecological risks were selected from two sources. First, the list of compounds detected in the water and sediments as reported in the public health risk assessment (ENSR, 1988) were considered. Secondly, the entire list of compounds found at the site were examined for potential ecotoxicity by comparing water and sediment concentrations to:

- ambient water quality criteria
- frequency in which each contaminant was detected
- toxicological benchmarks (LOEL and LC_{50}).

Using this process, a subset of chemicals was selected for evaluation in the ecological risk assessment (Table 1-1). The indicator compound list includes organics and metals which are representative of the range of contaminants detected in the water and sediments at the site. Details of the Indicator Compound selection process are presented in Appendix G.

TABLE 1-1

INDICATOR COMPOUNDS FOR THE RISK ASSESSMENTS

Ecological

mercury
chromium
1,2-dichlorobenzene
zinc
lead
cadmium
phenol
total PAHs
PCBs
1,2-diphenylhydrazine
di-2-ethylhexyl phthalate

Public Health

mercury
chromium
1,2-dichlorobenzene
zinc
lead
1,2-diphenylhydrazine
benzene
arsenic
carcinogenic PAHs
PCBs
1,1,2,2-tetrachloroethane

1.4 Selection of Indicator Species

Although numerous species exist at the site, it was not necessary to examine each one specifically to determine the potential for ecological risk posed by contaminants at the site. Based on the results of the preliminary survey (Appendix A) and available data review, Indicator Species, as shown in Table 1-2, were selected for the site to represent various trophic levels and exposure pathways. The Indicator Species and the rationale for their selection are:

- Chironomidae - aquatic larvae of non-biting midges. Chironomid midges represent a taxonomically and ecologically diverse group of insects whose larvae inhabit most aquatic habitats. Their natural life histories are diverse. These benthic invertebrates spend the majority of their life-cycle feeding on detritus and microbes in sediments. Chironomid larvae are eaten by both birds and fish, thus they represent a typical trophic link. Some toxicological information is also available for midge larvae.
- *Palaemonetes pugio* - The grass shrimp is an important pelagic component of the food web at the UOP site and is found in both the on-site stream channels and in Walden Swamp. Toxicity studies concerning this species and important indicator compounds (e.g., PCBs) are available.
- *Congeria* sp. - One species of mussel is abundant in on-site and reference areas. As a filter-feeding, stationary benthic invertebrate, some individuals of this species may be exposed to high indicator compound concentrations in sediments.
- *Callinectes sapidus* - The blue crab is found in and around the Berry's Creek area. It is a scavenger, and is readily exposed to sediments. This species may be consumed by humans.

TABLE 1-2

INDICATOR SPECIES AT THE UOP SITE

Common Name	Scientific Name
Mink	*Mustela vison*
Mallard duck	*Anas platyrhynchos*
Blue crab	*Callinectes sapidus*
Mummichog	*Fundulus heteroclitus*
Midge	Chironomidae
Grass shrimp	*Palaemonetes pugio*
Platform Mussel or Blue mussel	*Congeria leucopheata* or *Mytilus edulis*

- Fundulus heteroclitus - The mummichog is probably the most common forage fish in the stream channels. It was also designated by the NJDEP as the indicator organism for the focused environmental assessment (Appendix C). This fish's behavioral patterns suggest that it may be significantly exposed to sediments and, as a forage species, it represents a potential link to higher trophic levels.
- Anas platyrhynchos - Mallards nest in the meadowlands area and their presence has been reported on site (Appendix A). These ducks feed by dabbling in stream channels, and they may be hunted and consumed by humans. Toxicity studies concerning mallards and important Indicator Compounds (e.g., PCBs) are available.
- Mustela vison - According to historical records, mink have been observed in the meadowlands area. Based on the possibility that mink do inhabit the Meadowlands, this species was assessed at the request of NJDEP. Mink feed on a variety of prey including fish, reptiles, amphibians, crustaceans, ducks and mammals. Mink are sensitive to PCBs and will show signs of toxicity at low doses.

The above seven taxa represent several trophic levels of the aquatic ecosystem of the Berry's Creek area. No primary producers were chosen because this is predominantly a detrital-based food web. No reptiles or amphibians were chosen as aquatic Indicator Species because:

- little or no evidence was found of aquatic/semi-aquatic species in the on-site or reference areas, and
- birds (such as mallards) are more representative of higher order vertebrates which utilize this habitat.

2. RECEPTOR CHARACTERIZATION

The receptors that are analyzed in the risk assessment are the Indicator Species selected in Section 1.3. For the analyses there are insufficient data to project beyond populations of individual species to quantitative assessment of community-level or ecosystem-level effects. However, there is some qualitative discussion in the following sections which address potential effects at those higher levels based upon reports from the published literature compared to available data.

2.1 *Fundulus heteroclitus*

The mummichog is the most common forage fish in the stream channels. It has also been designated by the NJDEP as the indicator organism for the focused environmental assessment (see Appendix C). The behavioral patterns of this fish suggest that it may be exposed to sediments, and, as a forage species, it represents a potential link to higher trophic levels.

The mummichog has an extensive Atlantic coast range from the Gulf of St. Lawrence to northeast Florida. This species, whose habitat requirements differ greatly from those of other *Fundulus* sp., is normally found in salt water marshes and tidal creeks, but does enter fresh water to a limited extent (Robin et al., 1986). Few quantitative data are available on seasonal ranges of this fish species. Lotrich (1975) described a summer range of 36 m. The exact homerange of Fundulus on the UOP site is unknown. However, during the marked recapture study described in Appendix C, herein, mummichogs were recaptured after one month at a distance of more than 1,000 meters from their original capture site. This comparatively large range may be due in part to the tidal hydraulic characteristics in the area. During careful observations, mummichogs were observed to move with the tidal currents and not burrow into the sediment during ebb tide.

The mummichog is an omnivore. Digestive tract analyses have shown the following dietary components: diatoms, amphipods and other crustaceans, molluscs, fish eggs, small fishes and vegetation such as eel grass (Scott and Crossman, 1973). Available information indicates that the mummichog may consume five times as much detritus as other foods (Jeffries, 1972). F. heteroclitus will neither grow nor maintain weight when fed detritus, but may ingest detritus accidentally along with prey (Prinslow et al., 1974).

Fundulus heteroclitus may live about four years, while significant mortality has been recorded at three years of age (Valiela et al., 1977; Fritz and Garside, 1975).

2.2 Callinectes sapidus

The blue crab is an estuarine species that ranges from Nova Scotia to Florida and Texas (Gosner, 1981). It lives in close contact with sediments, may bury itself at low tide (Teal and Teal, 1969), and ingest sediment while feeding (Williams, 1984). Blue crabs may be consumed by humans. The following description of blue crab ecology is excerpted from Williams (1984):

> "The blue crab is a coastal creature occurring on a variety of bottoms in freshwater, estuaries, and shallow ocean from water's edge to approximately 90 m...but mainly in the shallows to depths of 35 m. Hatching in the mouths of estuaries and shallow ocean, development of larvae progresses in the ocean..., followed by migration of megalopae and young crabs back into estuaries to mature into adults...
>
> "During its life the blue crab leads a migratory existence. The migratory patterns have been studied in greatest detail in Chesapeake Bay but the pattern seen there appears to be true of other areas as well. Mating usually takes place in water of reduced salinity well up

in the estuaries. After this, the females migrate downstream to areas of higher salinity near the mouths of estuaries where the eggs are laid and hatched, whereas the males tend to remain in the low-salinity areas for the remainder of their lives. For this reason, samples of adult crabs (or commercial catches) near the sea contain greater numbers of females, whereas those from the middle or upper reaches of bays contain larger percentages of males except at the breeding season. Once in the spawning areas, the females tend to remain for the duration of their lives or move a short way out to sea. Once hatched, the zoeae lead a planktonic existence until they transform to the megalopa stage. This stage also is planktonic, at least in part, for large numbers of megalopae enter estuaries in surface plankton..., their abundance influenced to some degree by light, current or pressure... King (1971) found three waves of abundance in Texas, January-March, May-June, October, but there as elsewhere they occur all through the year. Migration either as megalopae or first crab stage leads up the estuary to the maturing grounds. Early recruits may reach these areas in their first summer, but the remainder may not reach them until early in the second year of life.

"In Chesapeake Bay, it has been demonstrated that crabs spawned in June of one year are mature about 14 months later and at that time mate. Most mating pairs are found in July, August , or September, although the mating season extends from May to October. At this time, females ready to molt into the mature stage...are carried about...under the males' bodies. Usually, a female mated in late summer casts the first batch of eggs the following spring at the age of two years, but egg laying may be any time from two to nine months after mating. Three years is judged to be about the normal maximum age for this species and the animals are estimated to undergo some 18 to 20 or more molts before reaching maturity..."

Normal diet includes a variety of materials including fishes, benthic invertebrates, and plant material. Blue crabs may be a major predator on infauna, and scoop up mud to feed on the infaunal organisms living in it.

2.3 *Palaemonetes spp.*

Grass shrimp have been identified from the Hackensack meadowlands (Hackensack Meadowlands Development Commission, 1987) and probably occur on this site. *Palaemonetes* has been used in bioassay testing, and therefore some toxicological information exists for this genus. The genus *Palaemonetes* (family Palaemonidae) ranges from Cape Cod to the Gulf of Mexico. Grass shrimp inhabit a variety of estuarine habitats, but are more likely found on the bottom than swimming free. Grass shrimp are said to tolerate substantial pollution levels in general (Gosner, 1978).

2.4 *Congeria leucopheaeta* (Similar Species: *Mytilus edulis*)

Congeria (= *Mytilopsis*) is a common bivalve in brackish to fresh water near rivers from New York to Florida to Texas and Mexico. It attaches itself to rocks and twigs by a short byssus. Colonies occur in clumps similar to those formed by the blue mussel, *Mytilus edulis*, which it superficially resembles (Abbott, 1954). Little research has been done on *Congeria* compared to *Mytilus*, therefore *Mytilus* may be used as a surrogate species. Both species are filter feeders that may be exposed to toxicants that are dissolved or borne on organic particles.

The blue mussel has been reported in the Meadowlands area and is representative of other bivalve inhabitants. As these organisms are filter-feeding benthic invertebrates, exposure to contaminated sediments is likely to be significant.

The blue mussel ranges from the Arctic to South Carolina and Alaska to Baja, California (Gosner, 1981). This species is euryhaline, living both off shore and in brackish water intertidal areas (Gosner, 1978). In an intertidal environment the mussel attaches itself to rocks, pilings or any other solid object (Gosner, 1981). In prime habitats this species reaches full growth within one year. Elsewhere, two to five years are necessary for full maturation (Gosner, 1978).

2.5 Chironomidae spp.

Larvae of the midge family are an important component of fresh water and low salinity food webs. These organisms may come in direct contact with the potentially contaminated sediments and, as a food source for many organisms, indirectly expose higher order consumers to the sediments and their contaminants.

Members of the midge family are very common in nearly all water infiltrated environments: ponds, lakes, streams, and estuaries (Borror and White, 1970). The North American range of *Chironomus attenuatus* is from British Columbia to Hudson Bay and Quebec south to California and Florida (Swan and Papp, 1972).

The larvae of most midge species are aquatic, inhabiting the sediments for much of their life cycle (Swan and Papp, 1972). Most midge species are herbivorous, feeding on algae, plankton and decaying vegetation (Peterson, 1960). Other midges such as *Procladius* are considered carnivores and opportunists, feeding on dead insects and newly hatched larvae of other insects (Wetzel, 1983; Swan and Papp, 1972). One stomach content analysis of a *Chironomus* sp. reported the presence of diatoms, infusoria, eggs of aquatic animals, and grains of sand. Many midge species do not feed in the winged state (Wetzel, 1983).

Eggs, approximately 500 for Chaoborus (phantom midge), are deposited in a gelatinous clump in the water or at its edges (Wetzel, 1983; Swan and Papp, 1972). The eggs hatch within two to four days, larvae of many species submerging into the sediments. The lifespan of the adult Chaoborus is less than 6 days (Wetzel, 1983).

2.6 Anas platyrhynchos

The mallard duck was chosen as an indicator organism as a representative of the higher order vertebrates. This species is known to inhabit the Meadowlands area and may be hunted and consumed by humans.

The mallard is widely distributed throughout the United States. In the eastern United States, breeding occurs from Nova Scotia to northern Virginia. Overwintering covers a broader range, from Nova Scotia to southern Mexico (Forbush and May, 1939).

As with most ducks, the mallard prefers wet environments, but actually requires very little water and feeds readily on land. Dietary components of the mallard include seeds, reeds, fresh water snails and other molluscs, aquatic insects, fish, fish eggs, and all grains (Forbush and May, 1939).

Mallards have one brood yearly and have lifespans of up to twenty-nine years. This species has shown a preference for nesting among Phragmites (Terres, 1980).

2.7 Mustela vison

The NJDEP required that mink be included as an indicator organism. This species was chosen because of its known sensitivity to polychlorinated biphenyls. Reproductive failure and/or fatality of mink have been recorded at low PCB concentrations (Ringer et al., 1981).

Mink, found in all contiguous 48 states, Alaska and all Canadian provinces, prefer fresh water habitats, but are often found in esturine environments. Fish and muskrat are the mink's major dietary components, but these carnivores will feed upon a wide range of mammalian, reptilian, amphibian, and avian organisms (Rue, 1981).

Distinctly different homeranges have been determined for male and female mink. The female's range is normally restricted to less than 0.08 square kilometers, but if food is scarce, this range will be increased by 3 to 5 times. While with young, the female has one den. During the winter months, the females are as nomadic as the males, which have a range of up to 64.75 square kilometers (Rue, 1981).

Both male and female are capable of breeding at one year of age, the yearly litter consisting of three to eight young. Mink has a reported lifespan of 10 years (Rue, 1981).

3. HAZARD ASSESSMENT - ECOTOXICITY REVIEW

The ecological toxicity of each of the Indicator Compounds is summarized below. More detailed ecotoxicological profiles of the Indicator Compounds can be found in Appendix E. Most of the information on each compound cited in this chapter comes from the Ambient Water Quality Criteria documents published by the U.S. EPA under the Clean Water Act of 1972, as amended. References for this material are cited in Appendix E, and provided in Appendix I.

Cadmium

Cadmium is a metal which forms no stable organic compounds. Cadmium releases to the environment are generally associated with mining, smelting, alkaline battery disposal, and various manufacturing operations.

Compared to other heavy metals, cadmium is relatively mobile in the aquatic environment and may be transported in solution as organic or inorganic complexes. Dissolved cadmium can be removed from solution through adsorption onto mineral surfaces. Cadmium is strongly bioaccumulated by both salt water and fresh water organisms at all trophic levels.

In general, fresh water animal species are more sensitive to cadmium toxicity than are salt water species. *Daphnia magna*, one of the most sensitive species reported in the available literature, shows chronic toxicity at a concentration of 0.15 µg/L.

To protect salt water life, the four-day average concentration of cadmium should not exceed 9.3 µg/L more than once every three years on the average and the one-hour average concentration should not exceed 43 µg/L more than once every three years on the average. The values for fresh water life are related to water hardness:

Hardness ($CaCO_3$)	Four-Day Average Concentration	One-Hour Average Concentration
50 mg/L	0.66 µg/L	1.8 µg/L
100 mg/L	1.1 µg/L	3.9 µg/L
200 mg/L	2.0 µg/L	8.6 µg/L

Chromium

Chromium is a heavy metal that generally exists in either a trivalent or hexavalent oxidation state. Chromium salts are used extensively in metal finishing and the tannery industry.

Hexavalent chromium (CrVI) is rapidly converted to trivalent chromium (CrIII) in the presence of reducing agents. Hexavalent chromium is soluble and mobile in water, and trivalent chromium tends to form stable complexes with negatively charged organic and inorganic species. Therefore, if the aquatic conditions favor Cr(III), accumulation will occur in sediments, and if Cr(VI) is favored, accumulation will occur in water.

The acute toxicity of chromium is dependent on the hardness of the water. As hardness and pH increase, hexavalent chromium becomes less toxic, whereas trivalent chromium tends to be more toxic in soft water.

The federal Ambient Water Quality Criteria for trivalent chromium are as follows:

Fresh Water (µg/L)	
Acute	Chronic
1,700	210

The lowest observed effect level for trivalent chromium in salt water is 10,300 µg/L for acute exposure.

The federal Ambient Water Quality Criteria for hexavalent chromium are as follows:

Fresh water (µg/L)		Salt water (µg/L)	
Acute	Chronic	Acute	Chronic
16	11	1,100	50

1,2-Dichlorobenzene

1,2-Dichlorobenzene is used primarily as a process chemical in the production of toluene diisocyanate, and in the manufacturing of dyes, herbicides, and degreasers.

1,2-Dichlorobenzene has a low water solubility and volatilizes quickly, although it may also be adsorbed by sedimented organic material. 1,2-Dichlorobenzene has a high affinity for organic materials, and probably bioaccumulates. There is evidence that this compound is persistent and is probably degraded slowly by microorganisms already growing on another hydrocarbon source.

Among fresh water species tested, as reported by U.S. EPA, rainbow trout (*Salmo gairdneri*) was the most sensitive species. The mysid shrimp (*M. bahia*) was the most sensitive salt water species tested, as reported by U.S. EPA. Only the fathead minnow was tested under chronic conditions. This embryo/larval test resulted in an effect concentration of 2,000 µg/L.

Federal Ambient Water Quality Criteria for protection of aquatic life have not been established. The lowest observed effect level for fresh water is 1,120 µg/L for acute exposure and 763 µg/L for chronic exposure. The lowest observed effect level for salt water is 1,970 µg/L for acute exposure.

Di(2-ethylhexyl)phthalate

Di(2-ethylhexyl)phthalate (DEHP) is primarily used as a plasticizer for resins such as polyvinyl chloride for the manufacturing of materials such as tubing and containers for blood products. DEHP is not believed to undergo any significant direct photolysis, hydrolysis, or volatilization. DEHP is likely to be adsorbed onto sediments and organic matter.

Federal Ambient Water Quality Criteria have not been established for DEHP. The lowest observed effect level is 940 µg/L for acute exposure and 3 µg/L for chronic exposure. The lowest observed effect level for salt water is 2,944 µg/L for acute exposure and 3.4 µg/L for chronic exposure.

1,2-Diphenylhydrazine

1,2-Diphenylhydrazine is used in the synthesis of phenylbutazone and as an intermediate in the production of dyes. 1,2-Diphenylhydrazine is rapidly oxidized to azobenzene in aerated solutions, where azobenzene can then be reduced. 1,2-Diphenylhydrazine is resistent to hydrolysis and has little tendency to volatilize from water. There is a strong tendency for 1,2-diphenylhydrazine and azobenzene to be adsorbed to organic matter.

Only two species have been used in toxicity tests as reported by U.S. EPA. The acute value for *Daphnia magna* was 4,100 µg/L, and the acute value for bluegill (*Lepomis macrochirus*) was 270 µg/L. There are no data available for chronic toxicity.

Federal Ambient Water Quality Criteria have not been established for 1,2-diphenylhydrazine.

Lead

Lead is a metallic element which is used extensively in the manufacture of storage batteries and formerly in gasoline and paint.

Lead tends to form complexes with naturally occurring organic materials and therefore has an affinity for clays and mineral surfaces. Most lead compounds, with the exception of tetramethyl lead, are not volatile. The adsorption of lead to sediments is highly pH-dependent. At a pH of 7 or above essentially all lead is in the solid phase, and at low pH lead

is negatively sorbed (repelled from the adsorbent surface). Because lead is easily complexed, precipitation can be important in controlling its mobility.

Lead, in general, is not bioaccumulated or biomagnified as the pH of most natural waters render lead unavailable for uptake. However, when it is available, mussels and oysters can accumulate high levels of lead in edible tissues.

The toxicity of lead to aquatic organisms is often dependent on the hardness of the water, with species being more sensitive in soft water.

The federal Ambient Water Quality Criteria for the protection of aquatic life are as follows:

Fresh water (µg/L)		Salt water (µg/L)	
Acute	Chronic	Acute	Chronic
82	3.2	140	5.6

Mercury

Mercury is a silver white metal found as a liquid at room temperature. It is used in electrical apparatus, thermometers, and barometers.

Elemental mercury can be oxidized to divalent mercury under natural conditions. Divalent mercury can be methylated by bacteria and in the slime coat and intestines of fish. Methyl mercury is the most water soluble form of mercury, and comprises most of the mercury residual in tissues of aquatic organisms.

The acute and chronic toxicity of mercury is not only species-specific, but it also depends on the chemical form of mercury: divalent inorganic mercury, methyl mercury, and other organic compounds.

The federal Ambient Water Quality Criteria for mercury is as follows:

Fresh water (µg/L)		Salt water (µg/L)	
Acute	Chronic	Acute	Chronic
2.4	0.012	2.1	0.025

Phenol

Phenol is a widely used industrial chemical which is slowly reduced in the aqueous environment through photo-oxidation. Phenol is resistant to hydrolysis, there is little tendency for it to volatilize from water, and it is not likely to become sorbed to organic sediments. There is a wide range of species sensitivity to phenol for both fresh water and salt water species.

There are no federal Ambient Water Quality Criteria for the protection of aquatic life. The lowest observed effect level for fresh water is 10,200 µg/L for acute exposure and 2,560 µg/L for chronic exposure. The lowest observed effect level for salt water is 5,800 µg/L for acute exposure.

Polynuclear Aromatic Hydrocarbons

Polynuclear aromatic hydrocarbons (PAHs) are a class of compounds consisting of substituted and unsubstituted polycyclic and heterocyclic aromatic rings. As a group, they are widely distributed in the environment, having been found in animal and plant tissue, sediments, air, and surface water.

Most PAHs undergo direct photolysis in the aqueous environment, and PAHs with four or more rings are unlikely to undergo any significant volatilization. Most PAHs adsorb onto particulate matter and can be transported in water. Sorption onto sediments is strongly correlated with organic carbon levels in sediments (adsorption increases with carbon content). Bacteria have been shown to use some PAHs as a sole source of carbon for growth, and biodegradation is thought to be an important fate process for PAHs with four or fewer aromatic rings.

There are no federal Ambient Water Quality Criteria available for PAH. The lowest observed effect level for salt water is 300 µg/L for acute exposure.

Polychlorinated Biphenyls

Polychlorinated biphenyls (PCBs) are no longer manufactured in the U.S. However, due to the long life of products containing PCBs, many of the PCBs previously manufactured are still in use.

In general, PCBs are extremely resistant to oxidation and strongly resistant to both acid and base hydrolysis. In the aquatic environment PCBs are very likely to be sorbed to sediments and are strongly bioaccumulated.

In acute toxicity tests the most sensitive salt water species was the eastern oyster (*C. virginica*). Among the fresh water species, the rainbow trout (*S. gairdneri*) and fathead minnow (*P. promelas*) were some of the most sensitive. In chronic tests, the midge (*T. dissimilis*) was one of the most sensitive fresh water species. Only one salt water species was tested, the sheepshead minnow (*C. variegatus*), which was found to be more sensitive to Aroclor® 1254 (value = 0.098 µg/L) than to Aroclor® 1016 (value = 7.14 µg/L).

The federal Ambient Water Quality Criteria for the protection of aquatic life are as follows:

Fresh water (µg/L)		Salt water (µg/L)	
Acute	Chronic	Acute	Chronic
2.0	0.014	10	0.03

Zinc

Zinc is a metallic element which is ubiquitous in the environment and is present in most foodstuffs including meats, seafoods, whole grains, and dairy products. Zinc is widely used in galvanizing processes and in various medicinal preparations.

The predominant fate of zinc is adsorption to sediments and clay minerals. Zinc is bioaccumulated readily from detrital organic material, although little uptake occurs when zinc is co-precipitated with hydrous iron or manganese oxides.

In acute toxicity tests, Daphnia magna was one of the most sensitive fresh water species (EC_{50} or LC_{50} = 525 µg/L), while the hardshell clam, Mercenaria mercenaria, was one of the most sensitive salt water species (EC_{50} or LC_{50} = 195 µg/L). In chronic toxicity tests, D. magna was one of the most sensitive fresh water species (46.7 µg/L). Only one salt water species, the mysid shrimp (M. bahia), was tested. The chronic value was 166.5 µg/L. Reported bioconcentration factors range from 51 to 23,820.

The federal Ambient Water Quality Criteria for the protection of aquatic life are as follows:

Fresh water (µg/L)		Salt water (µg/L)	
Acute	Chronic	Acute	Chronic
120	110	95	86

4. EXPOSURE ASSESSMENT

4.1 Sources

The main pathways of exposure for the Indicator Species are water, food, and sediment. The selection of pathways for analysis depends in part on the biology of the species and in part on the requirements of the method of analysis. Three types of analysis are used to estimate the effects of exposure on the Indicator Species. First, a screening level analysis is applied to all 11 Indicator Compounds and all seven Indicator Species. For three aquatic species, a relatively new type of analysis, called analysis of estimated error, is also applied. For mink and mallard duck, an analysis based on dose and dose-response is used.

4.2 Fate and Transport

Sediment samples taken from the tidal creek and channels on the site were analyzed to determine the geographic distribution of indicator chemicals. Samples taken at a reference area designated by the NJDEP were analyzed to provide a measure of the levels of these chemicals off the site. Geographic patterns, if any, in the distribution of indicator chemicals can be used to make inferences about the fate and transport of these chemicals.

Surface water data can be used to make inferences about water-borne transport of the chemicals. The surface water analyses were performed several years ago and many of the detection levels were higher than is needed for the risk analysis. For this reason, sediment data were used in contaminant concentration modelling to predict water-borne concentrations. The results were checked against measured values in water where possible.

4.3 Exposure Concentrations

Many exposure pathways are possible in the estuarine environment at the UOP site, including:

- direct contact with and ingestion of contaminated column water,
- direct contact with and ingestion of contaminated sediments and pore water, and
- ingestion of contaminated food.

Measured concentrations of indicator chemicals in sediment and water are shown in Appendix G. Samples were collected by Geraghty & Miller (1987) in 1983 and 1985 and by ENSR in 1988. Columns in the table represent locations, which have different station designations depending on the year in which samples were collected. Sampling was conducted in 3 channels, which are designated A, B, and C (Figure G-2). The northern and most inland channel is A. B is a meandering channel that may be the original creek, and C is the southernmost channel which opens into Berry's Creek. Channel B connects channel A with B. The reference area is designated D.

In 1988, three chemicals (i.e., PCBs, chromium, and mercury) were measured in sediments, while none were measured in water. In the 1983 and 1985 data, many chemical concentrations in water were below detection limits.

In 1988, levels of Aroclor® 1248 ranged from 1.1 mg/kg dry weight in sediments at location D to 123 mg/kg at a location in channel A. On the site, the mean concentration of Aroclor® 1248 at all stations within a channel was highest in channel A (mean = 104 mg/kg, n = 3), intermediate in channel B (mean = 49.6 mg/kg, n = 3), and lowest in channel C (mean = 20 mg/kg, n = 3).

In that year, total chromium showed a similar pattern. Concentrations ranged from 1,022.7 mg/kg at location D to

26,900 mg/kg at a location in channel A. On the site, the mean concentration of total chromium at all locations within a channel was highest in channel A (mean = 18,913 mg/kg, n = 3), intermediate in channel B (mean = 6,921 mg/kg, n = 3), and lowest in channel C (mean = 1,205 mg/kg, n = 3).

The distribution of mercury in that year was slightly different. Location D had a higher concentration of mercury than any of the average channel concentrations. (There were no replicate samples at D, however). Concentrations ranged from 5.9 mg/kg at a location in channel C to 108 mg/kg at a location in channel A. On the site, the mean concentration of total mercury at all locations within a channel was highest in channel A (mean = 63.3 mg/kg, n = 3), intermediate in channel B (mean = 48.8 mg/kg, n = 3), and lowest in channel C (mean = 10.8 mg/kg, n = 3).

In general, concentrations of PCBs and chromium measured in 1988 decreased in sediments with increasing distance away from the lagoon. Concentrations at reference location D were lower than on-site channel concentrations for Aroclor® 1248 and total chromium, but higher for mercury.

In order to have usable surface water data (i.e., concentrations below the high detection limits obtained in 1983 and 1985), the concentration of contaminants in pore and channel water at the UOP site are predicted using available data on sediment concentration. This is done by using a Monte Carlo model to predict the average pore water concentration and a standard deviation about that average. This is followed by the use of an analytic algorithm to describe mixing of the sediment pore water into the channel water.

The first step in this process is the calculation of an areally averaged sediment concentration. To do this, each sediment sample is assigned a fraction of the total channel area. The sediment concentration is weighted by this fraction in the calculation of the site wide average and standard deviation. For metal contaminants, distribution coefficients

(means and standard deviations) are available from the literature. As explained in Appendix F, a Monte Carlo analysis is used to generate pore water concentration estimates (mean values and standard deviations) from the sediment concentration and distribution coefficients. Similarly, pore water concentrations are estimated by a Monte Carlo analysis using the sediment concentration distribution, literature values of organic partition coefficients, and sediment fraction organic carbon distributions derived from site data.

The calculation of channel water concentration is a linear transformation of the pore water concentration. As such, the mean and standard deviations of the pore water concentration distribution can be manipulated in closed form expressions to solve for mean and standard deviations of the channel water concentration distributions.

The algorithm describing mixing of the pore water with the channel water is an USEPA screening procedure and assumes that there is a boundary layer of pore water at the channel bottom and that pore water mixes with the channel water depending upon the hydraulic parameters of the channel. This algorithm has been modifed to depict the tidal flushing of the channels at the UOP site. Most importantly, the channels are assumed to retain some contaminated water after each tidal cycle. The mixing algorithm is run iteratively until the channel water concentration converges to a stable value.

4.4 Exposure Characterization

In order to make some estimates about the effect of the exposure concentrations on biological species, some assumptions about exposure must be made in accordance with and the biology of the Indicator Species and the method of estimation used. Three types of analysis are used to estimate the effects of exposure on the indicator organisms. The first two types of analysis, the quotient method and analysis of extrapolation

error, are described in the User's Manual for Ecological Risk Assessment (Barnthouse et al., 1986). The third method, for ducks and mink, involves the estimation of risk from dose and toxicity.

The first is a screening level analysis. This "quotient method" (QM) is applied to all 11 Indicator Compounds and all seven Indicator Species. For this method, expected exposure concentrations in column water and pore water are compared to benchmark concentrations from both acute and chronic toxicity tests. In this analysis, the measured concentrations are used where the concentrations are above detection limits. Also, expected environmental concentrations predicted by contaminant concentration modelling are also used. For this analysis, it is assumed that the organisms are exposed to the mean concentration continuously, so that the mean concentration can be used to select the benchmark concentration.

The second method is the analysis of estimated error (AEE), which is applied to three Indicator Species and four Indicator Compounds. The three species are blue crab, grass shrimp, and mummichog, and the four compounds are chromium, zinc, mercury and PCBs. AEE also uses results from laboratory toxicity tests to establish the most closely related toxicological endpoints. Expected exposure concentrations in column water are used. As in the QM analyses, both measured concentrations above detection limits and predicted concentrations from contaminant concentration modeling will be used. In addition to the mean concentration, AEE also uses the variance of the concentrations. Like the analysis by QM, the AEE analysis assumes that the organisms are exposed to the mean concentration continuously, so that the mean concentration can be used to select an appropriate toxicological end point.

For ducks and mink, the third method is to estimate a risk from PCBs and mercury based on assumptions about dosage. Here the route of exposure is through food. The analyses of PCBs in fish tissue are used. Measured concentrations of PCBs and

mercury in fish tissue from field collections in 1988 are given in Appendix D. Additionally, concentrations of PCBs and mercury in aquatic plants were estimated.

5. TOXICITY QUOTIENT METHOD

5.1 Introduction

ENSR has used a "Toxicity Quotient Method" as a screening method to estimate the ecological risk from exposure to the Indicator Compounds in the channel water. The indicator species used in this assessment are: mummichog (*Fundulus heteroclitus*), blue crab (*Callinectes sapidus*), midge (Chironomus spp.,) grass shrimp (*Palaemonetes pugio*), and mussel (*Congeria* spp.). Based on the natural history of these five species, ENSR has computed the Toxicity Quotients using the estimated water column concentrations because the organisms would be predominantly exposed to this media.

ENSR has also computed the Toxicity Quotients using the estimated pore water concentrations, even though this is not the dominant exposure route. The effects of sediment toxicity has also been addressed by calculating a sediment quotient for PCBs using the U.S. EPA (1988) interim sediment criteria method. Additionally, ENSR has discussed the effects of additivity, synergism, and antagonism in sediments based on recent work by Swartz et al. (1988), and reviewed a recent empirical investigation on the toxicity of New York Harbor sediments.

5.2 Methodology

G.W. Suter II has written extensively on the use of Toxicity Quotients as a way to assess ecological conditions at a site (see for example, Chapter 3 in ORNL, 1986). As he explains:

> The quotient method is simply the direct arithmetic comparison of a benchmark concentration (BC) from a toxicity test with an expected environmental concentration (EEC). It is typically calculated as the quotient of the

ratio EEC/BC. It is the basis for nearly all assessments of the environmental hazards of chemicals. In this basic form, the method amounts to an assumption that the test benchmark is a good model of the assessment end point (i.e., the level of toxic effect that is not to be exceeded in the ambient ecosystem). This assumption is most likely to hold when the toxicity tests have been performed for the particular assessment, using the anticipated temporal pattern of exposure and dilution water and organisms from the site. When it is recognized that this assumption may not hold, multiplicative factors are often applied to the quotients.

For exposures in the aquatic environment (the estuarine channels), ENSR has estimated two sets of Toxicity Quotients: first with the estimated water column concentrations as the EECs, and second with the estimated pore water concentrations as the EECs.

Interim sediment criteria values are also applied in a method analogous to the quotient method for water concentrations. Of the Indicator Compounds considered here, only one (PCBs) has a Sediment Quality Criterion in the Interim Sediment Criteria document (U.S. EPA, 1988). After normalizing sediment concentrations for organic carbon, a toxicity quotient can be calculated as a ratio of the expected environmental concentration to the Sediment Quality Criterion.

5.3 Expected Environmental Concentrations in Water

Table 5-1 lists the estimated average concentrations of the Indicator Compounds in the channel water and in the pore water (see Appendix F for a full discussion).

Table 5-1 distinguishes between the two most prevalent valences of chromium (III and VI) and two of the possible forms of mercury (inorganic ion and methyl mercury). For chromium, the trivalent species is the thermodynamically preferred

Table 5-1

Estimated Average Concentrations of Indicator Compounds in the Water

Indicator Compound	Average Water Column Estimated from Sediment Data and Model (µg/l)	Average Pore Water Estimated from Sediment Data and Model (µg/l)	Note
cadmium	1.16E+00	2.10E+02	
chromium (III)	6.14E+00	1.20E+03	a, e
chromium (VI)	2.39E-01	4.20E+01	
lead	2.62E+00	4.40E+02	
mercury	3.51E-01	1.10E+02	e
methyl mercury	3.51E-01	1.10E+02	b, e
zinc	1.29E+02	1.49E+04	
1,2-dichlorobenzene	3.91E-02	9.00E+00	
phenol	6.03E+00	1.70E+03	
di-2-ethylhexyl phthalate	1.24E-01	9.53E-05	
1,2-diphenylhydrazine	3.08E-02	1.20E+01	
PAHs	1.50E-02	6.02E-02	c
PCBs	5.26E-01	3.00E+00	d, e

a = total chromium minus chromium (VI)
b = assumed for worst case
c = sum of three compounds
d = aroclor 1248
e = sampling in fall 1988

species in natural waters. No information on the relative speciation of mercury is available, except to note that bacteria can methylate the metal in anaerobic sediments.

5.4 Exposure Routes

Four of the Indicator Species for this study are pelagic or epifaunal, i.e., they live in the water column or on the surface of the sediments: mummichog, blue crab, grass shrimp, and mussel. Mummichog, blue crab and grass shrimp are mobile and likely to migrate; whereas mussels tend to attach to the substratum and do not migrate.

Chironomid midges, the fifth Indicator Species, are not a species but a diverse group of insects; there are over 4,000 species worldwide. Classification to genus and species is confused and difficult, although dipteran taxonomists are agreed on higher taxonomic categories. Most, but not all, chironomids have aquatic larvae. Different chironomid species feed on different types of food sources utilizing a variety of feeding behaviors. Some are carnivorous, others herbivorous; some are microphagous, meaning they filter microscopic organic materials from the surrounding water. Some species have a generation every two years, others have several generations in one year. In North America, a single water body may have only a few species, but highly productive streams may have more than 30 species, and large lakes and reservoirs may have twice that number (Pennak, 1978).

Most chironomid larvae are herbivorous and microphagous; most build tubes of some sort through which a current of water passes. Therefore, the most probable route of exposure to contaminants for midge larvae on the UOP site is through channel water. Sediment feeding or non-tube building species which could accumulate contaminants directly from sediment might be present, however, and so pore water exposure is possible. Larsson (1984) found that uptake of PCBs by particle-feeding larvae of Chironomus plumulosus-type midges

was directly related to PCB concentrations in sediments. In any case, taxonomic and life history diversity and problems with systematics suggest that more caution be exercised in interpreting conclusions for midges than for single species.

5.5 Benchmark Concentrations

Tables 5-2, 5-3, and 5-4 show the "benchmark" concentrations used in the denominators of the toxicity quotients:

- Table 5-2 lists the Ambient Water Quality Criteria published by the U.S. EPA for these Indicator Compounds in salt waters. The U.S. EPA publishes these acute and chronic values as those concentrations which will protect the most sensitive life stages of the most sensitive species in a given (salt water) environment.

- Table 5-3, drawn from the corresponding federal Ambient Water Quality Criteria Documents published by U.S. EPA, summarizes the concentrations known to cause acute effects (LC50 or EC50) in the Indicator Species in salt water. The table lists the lowest reported concentrations known to cause acute effects, along with the species, the chemical tested, and the citation, all reported in the U.S. EPA criteria documents (U.S. EPA Tables 1 or 6).

- Table 5-4, in the same format and drawn from the same sources, summarizes the lowest reported concentrations known to cause chronic effects in the Indicator Species in salt water. The table lists the lowest reported concentrations known to cause chronic effects, along with the species, the chemical tested, and the citation, all reported in the U.S. EPA criteria documents (U.S. EPA Tables 2 or 6).

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Table 5-2

Ambient Water Quality Criteria
US Environmental Protection Agency

Indicator Compound	Marine Acute Criteria (µg/l)	Note	Marine Chronic Criteria (µg/l)	Note	date reference
cadmium	4.30E+01		9.30E+00		50FR30784
chromium (III)	1.03E+04	*			50FR30784
chromium (VI)	1.10E+03		5.00E+01		50FR30784
lead	1.40E+02		5.60E+00		50FR30784
mercury	2.10E+00		2.50E-02		50FR30784
methyl mercury					50FR30784
zinc	9.50E+01		8.60E+01		52FR6213
dichlorobenzenes	1.97E+03	*			45FR79318
phenol	5.80E+03	*			45FR79318
di-2-ethylhexyl phthalate	2.94E+03		3.40E+00		45FR79318
1,2-diphenylhydrazine					45FR79318
PAHs	3.00E+02	*			45FR79318
PCBs	1.00E+01		3.00E-02		45FR79318

Notes: * = Insufficient Data to Develop Criteria
Value is the LOEL - Lowest Observed Effect Level

FR = Federal Register

Table 5-3
Concentrations for Acute Effects

Indicator Compound	Conc (µg/l)	Mummichog (Fundulus heteroclitus)	Conc (µg/l)	Blue Crab (Callinectes sapidus)	Conc (µg/l)	Midge (Chironomus sp.)	Conc (µg/l)	Grass Shrimp (Palaemonetes)	Conc (µg/l)	Mussel (Congeria sp.)
cadmium	7.80E+03	Fundulus heteroclitus (larvae) cadmium chloride Middaugh & Dean, 1977	3.20E+02	Callinectes sapidus (juveniles) cadmium chloride Frank & Robertson, 1979	1.20E+03	Chironomus sp. Rehwoldt, et al. 1973	4.20E+02	Palaemonetes vulgaris cadmium chloride Eisler, 1971	1.20E+03	Mytilus edulis (embryo) cadmium chloride Martin et al, 1981
chromium III	3.15E+04	Fundulus heteroclitus chromium chloride Dorfman, 1977			1.10E+04	Chironomus sp. Rehwoldt, et al. 1973				
chromium VI	4.40E+04	Fundulus heteroclitus potassium chromate Eisler & Hennekey, 1977	3.40E+04	Callinectes sapidus potassium dichromate Frank & Robertson, 1979	5.73E+04	Tanytarsus dissimilis potassium dichromate Call et al, 983	2.10E+04	Palaemonetes pugio potassium chromate Fales, 1978	4.47E+03	Mytilus edulis (embryo) potassium dichromate Martin et al, 1981
lead	3.15E+02	Fundulus heteroclitus lead nitrate Dorfman, 1977	5.75E+02	Cancer magister lead nitrate Martin, et al. 1981	2.58E+02	Tanytarsus dissimilis (embryo) lead nitrate Anderson, et al.1980	3.13E+03	Mysidopsis bahia lead nitrate Lussier et al, ms	4.76E+02	Mytilus edulis (larvae) lead nitrate Martin, et al. 1981
mercury	6.74E+01	Fundulus heteroclitus (embryo) mercuric chloride Sharp & Neff, 1982	1.00E+01	Carcinus maenas mercuric chloride Connor, 1972	2.00E+01	Chironomus sp. mercuric nitrate Rehwoldt, et al. 1973	1.00E+01	Palaemonetes vulgaris mercuric chloride Shealy & Sandifer, 1975	5.80E+00	Mytilus edulis mercuric chloride Martin et al, 1981
methyl mercury	5.11E+01	Fundulus heteroclitus (embryo) methylmercuric chloride Sharp & Neff, 1982								
zinc	1.75E+04	Fundulus heteroclitus zinc chloride Dorfman, 1977	1.00E+03	Carcinus maenas zinc sulfate Connor, 1972	1.82E+04	Chironomus sp. LC50 Rehwoldt, et al. 1973	4.99E+02	Mysidopsis bahia zinc chloride Lussier et al, 1985	2.50E+03	Mytilus edulis planulatus zinc chloride Ahsanullah, 1976
1,2-dichlorobenzene	9.66E+03	Cyprinodon variegatus 1,2-dichlorobenzene US EPA, 1978			1.18E+04	Tanytarsus dissimilis 1,2-dichlorobenzene US EPA, 1980	1.97E+03	Mysidopsis bahia 1,2-dichlorobenzene U.S. EPA, 1978	1.00E+05	M mercenaria (embryo) 1,2-dichlorobenzene Davis & Hindu, 1969
phenol							5.80E+03	Palaemonetes pugio phenol Tatem et al. 1978	5.26E+04	Mercenaria mercenaria phenol Davis & Hidu, 1969
di-2-ethylhexyl phthalate	5.80E+04	Cyprinodon variegatus dimethyl phthalate US EPA, 1978			1.80E+04	Chironomus plumosus di-2-ethylhexyl phthalate Streufert, 1977	7.37E+04	Mysidopsis bahia dimethyl phthalate US EPA, 1978		
1,2-diphenylhydrazine										
PAH										
PCB (Aroclor 1248)	9.30E-01	Cyprinodon variegatus Aroclor 1254 Schimmel, et al. 1974								

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Table 5-4
Concentrations for Chronic Effects

Indicator Compound	Conc (µg/l)	Mummichog (Fundulus heteroclitus)	Conc (µg/l)	Blue Crab (Callinectes sapidus)	Conc (µg/l)	Midge (Chironomus sp.)	Conc (µg/l)	Grass Shrimp (Palaemonetes)	Conc (µg/l)	Mussel (Congeria sp.)
cadmium							7.14E+00	Mysidopsis bahia cadmium chloride Gentile et al,1982		
chromium III										
chromium VI			1.50E+03	Callinectes sapidus (larvae) sodium chromate reduced survival Bookhout et al, 1984	1.00E+02	Chironomus tentans potassium dichromate suppressed respiration Batac-Catalan & White, 1983	1.32E+02	Mysidopsis bahia potassium dichromate Lussier et al, ms		
lead	1.00E+02	Fundulus heteroclitus (embryo) lead nitrate depressed axis formation Weis & Weis, 1977					2.50E+01	Mysidopsis bahia lead nitrate Lussier et al, ms	5.00E+02	Mytilus edulis lead nitrate LT50 Schulz-Baldes, 1972
mercury	1.00E+01	Fundulus heteroclitus mercuric chloride development abnormalities Weis & Weis, 1977					1.13E+00	Mysidopsis bahia mercuric chloride Gentile et al, 1982, 1983	3.20E+01	Mytilus edulis mercuric chloride abnormal development Okubo & Okubo, 1962
methyl mercury	5.00E+01	Fundulus heteroclitus (embryo) methylmercuric chloride teratological effects Weis et al, 1981							4.00E+02	Mytilus edulus methylmercuric chloride 90% reduced feeding Dorn, 1976
zinc	7.60E+02	Fundulus heteroclitus (juvenile) zinc chloride inhibited scale calcification Sauer & Watabe, 1984					1.66E+02	Mysidopsis bahia zinc chloride Lussier et al, 1985	6.00E+01	Mytilus edulis zinc chloride shell growth Stromgren, 1982
1,2-dichlorobenzene										
phenol									1.00E+02	Mercenaria mercenaria phenol cellular damage Fries & Tripp, 1977
di-2-ethylhexyl phthalate							1.00E+05	Palaemonetes pugio dimethyl phthalate decreased survival Laughlin et al., 1978		
1,2-diphenylhydrazine										
PAH										
PCB (Aroclor 1248)	9.80E-02	Cyprinodon variegatus Aroclor 1254 Schimmel, et al. 1974			8.00E-01	Tanytarsus dissimilis Aroclor 1254 Nebeker & Puglisi, 1974	2.50E+01	Palaemonetes pugio Aroclor 1254 Altered metabolism Rocijadi et al, 1976		

Tables 5-3 and 5-4 include a value only if the U.S. EPA documents listed the identical or a close surrogate species. Tables 5-3 and 5-4 list values for salt water species only (with the exception of values for chironomids). Empty cells in the two tables reflect the absence of field or laboratory studies.

5.6 Estimated Toxicity Quotients

The toxicity quotient is commonly defined:

$$\text{Toxicity Quotient} = \frac{\text{EEC}}{\text{BC}}$$

The EEC and BC values must have common units; here micrograms per liter are used. As detailed above, one possible EEC for the numerator and several possible BCs for the denominator are available. The range in the BC values arises from and expresses the differences between: (i) acute and chronic exposures, and (ii) sensitivity of different organisms.

For each Indicator Compound, Tables 5-5 through 5-7 report the estimated Toxicity Quotients.

Table 5-5 reports the results for the estimated water column concentrations and pore water concentrations compared to the acute and chronic federal ambient water quality criteria for salt water. Tables 5-6a and 5-7a report the results for the acute and chronic criteria, respectively, using estimated water column concentrations of contaminants. Tables 5-6b and 5-7b report the results for acute and chronic criteria, respectively, using estimated pore water concentrations of contaminants. Although the Ambient Water Quality Criteria and toxicity endpoints are not enforceable standards (i.e., under the Federal Clean Water Act), it is interesting to compare them to the water concentration in the channel or sediments and then compare that ratio or "toxicity quotient" to a value of "one."

Table 5-5

Quotient Method
Ratio of Estimated Average Water Concentrations to Ambient Water Quality Criteria

Indicator Compound	Ratio of Water Column to Marine Acute (frac)	Ratio of Water Column to Marine Chronic (frac)	Ratio* of Pore Water to Marine Acute (frac)	Ratio* of Pore Water to Marine Chronic (frac)
cadmium	2.70E-02	1.25E-01	4.88E+00	2.26E+01
chromium (III)	5.96E-04		1.17E-01	
chromium (VI)	2.17E-04	4.78E-03	3.82E-02	8.40E-01
lead	1.87E-02	4.68E-01	3.14E+00	7.86E+01
mercury	1.67E-01	1.40E+01	5.24E+01	4.40E+03
methyl mercury				
zinc	1.36E+00	1.50E+00	1.57E+02	1.73E+02
dichlorobenzenes	1.98E-05		4.57E-03	
phenol	1.04E-03		2.93E-01	
di-2-ethylhexyl phthalate	4.21E-05	3.65E-02	3.24E-08	2.80E-05
1,2-diphenylhydrazine				
PAHs	5.00E-05		2.01E-04	
PCBs	5.26E-02	1.75E+01	3.00E-01	1.00E+02

* See text for discussion on inappropriateness of comparing pore water concentrations to water quality criteria

Table 5-6a

Quotient Method
Ratio of Estimated Average Water Column Concentrations to Concentrations for Acute Effects

Indicator Compound ***	Mummichog (frac) ***	Blue Crab (frac) ***	Midge (frac) ***	Grass Shrimp (frac) ***	Mussel (frac) ***
cadmium	1.49E-04	3.63E-03	9.67E-04	2.76E-03	9.67E-04
chromium III	1.95E-04		5.58E-04		
chromium VI	5.43E-06	7.03E-06	4.17E-06	1.14E-05	5.35E-05
lead	8.32E-03	4.56E-03	1.02E-02	8.37E-04	5.50E-03
mercury	5.21E-03	3.51E-02	1.76E-02	3.51E-02	6.05E-02
methyl mercury	6.87E-03				
zinc	7.37E-03	1.29E-01	7.09E-03	2.59E-01	5.16E-02
1,2-dichlorobenzene	4.05E-06		3.31E-06	1.98E-05	3.91E-07
phenol				1.04E-03	1.15E-04
di-2-ethylhexyl phthalate	2.14E-06		6.89E-06	1.68E-06	
1,2-diphenylhydrazine					
PAHs					
PCB (Aroclor 1248)	5.66E-01				

Table 5-6b

Quotient Method
Ratio of Estimated Average Pore Water Concentrations to Concentrations for Acute Effects

Indicator Compound ***	Mummichog* (frac) ***	Blue Crab* (frac) ***	Midge* (frac) ***	Grass Shrimp* (frac) ***	Mussel* (frac) ***
cadmium	2.69E-02	6.56E-01	1.75E-01	5.00E-01	1.75E-01
chromium III	3.81E-02		1.09E-01		
chromium VI	9.55E-04	1.24E-03	7.33E-04	2.00E-03	9.40E-03
lead	1.40E+00	7.65E-01	1.71E+00	1.41E-01	9.24E-01
mercury	1.63E+00	1.10E+01	5.50E+00	1.10E+01	1.90E+01
methyl mercury	2.15E+00				
zinc	8.51E-01	1.49E+01	8.19E-01	2.99E+01	5.96E+00
1,2-dichlorobenzene	9.32E-04		7.63E-04	4.57E-03	9.00E-05
phenol				2.93E-01	3.23E-02
di-2-ethylhexyl phthalate	1.64E-09		5.29E-09	1.29E-09	
1,2-diphenylhydrazine					
PAHs					
PCB (Aroclor 1248)	3.23E+00				

* See text for discussion on inappropriateness of comparing pore water concentrations to water quality criteria

Table 5-7a

Quotient Method
Ratio of Estimated Average Water Column Concentrations to Concentrations for Chronic Effects

Indicator Compound ***	Mummichog (frac) ***	Blue Crab (frac) ***	Midge (frac) ***	Grass Shrimp (frac) ***	Mussel (frac) ***
cadmium				1.62E-01	
chromium III					
chromium VI		1.59E-04	2.39E-03	1.81E-03	
lead	2.62E-02			1.05E-01	5.24E-03
mercury	3.51E-02			3.11E-01	1.10E-02
methyl mercury	7.02E-03				8.78E-04
zinc	1.70E-01			7.77E-01	2.15E+00
1,2-dichlorobenzene					
phenol					6.03E-02
di-2-ethylhexyl phthalate				1.24E-06	
1,2-diphenylhydrazine					
PAHs					
PCB (Aroclor 1248)	5.37E+00		6.58E-01	2.10E-02	

Table 5-7b

Quotient Method
Ratio of Estimated Average Pore Water Concentrations to Concentrations for Chronic Effects

Indicator Compound ***	Mummichog* (frac) ***	Blue Crab* (frac) ***	Midge* (frac) ***	Grass Shrimp* (frac) ***	Mussel* (frac) ***
cadmium				2.94E+01	
chromium III					
chromium VI		2.80E-02	4.20E-01	3.18E-01	
lead	4.40E+00			1.76E+01	8.80E-01
mercury	1.10E+01			9.73E+01	3.44E+00
methyl mercury	2.20E+00				2.75E-01
zinc	1.96E+01			8.98E+01	2.48E+02
1,2-dichlorobenzene					
phenol					1.70E+01
di-2-ethylhexyl phthalate				9.53E-10	
1,2-diphenylhydrazine					
PAHs					
PCB (Aroclor 1248)	3.06E+01		3.75E+00	1.20E-01	

* See text for discussion on inappropriateness of comparing pore water concentrations to water quality criteria

5.7 Sediment Criteria Quotient Method

The method for using interim sediment criteria values is given in U.S. EPA (1988). In applying the criteria, the fraction of organic carbon, foc, and the salinity of the water must be taken into account. On this site, salinities are roughly one-eighth seawater. Since estuarine organisms such as those inhabiting this site are included by U.S. EPA in the salt water Ambient Water Quality Criteria, and those criteria are used in development of the sediment criteria, the site will be considered salt water. Fraction of organic carbon is calculated from total organic carbon (Table D-6). U.S. EPA interim Sediment Quality Criteria (SQC) for salt water are listed in U.S. EPA (1988) Table 4. U.S. EPA lists both mean and 95 percent confidence limits for Sediment Quality Criteria, and recommends that the lower confidence limit be used to be 97.5 percent certain of protecting benthic fauna. The mean and lower confidence limit for PCBs (the only Indicator Compound for which SQCs are available) are used in Table 5-8.

A mean sediment value, normalized for organic carbon, is calculated as a geometric mean of the normalized concentrations weighted by the area each represents. This method is consistent with methods used in the channel water contaminant concentration model (Appendix F). The ratio of the averaged site concentration of PCBs in sediment to the mean SQC is 22.92 and the ratio to the lower confidence limit of the SQC is 115.56.

5.8 Comparison to Laboratory Studies

Rubinstein et al. (1983) investigated the acute (100 day) effects and bioaccumulation of PCBs, mercury, and cadmium by three invertebrate species from contaminated sediments from New York harbor. Their results can be applied to the UOP site because: (i) the invertebrate species (clamworms, hard clams, and grass shrimp) are local, and one (grass shrimp) occurs on

TABLE 5-8

QUOTIENT METHOD APPLIED TO PCBs IN SEDIMENTS

Ratio of Normalized Sediment Concentrations to Two Measures of Sediment Quality Criteria

Location	Conc (mg/kg)	Foc	Norm Conc (mg/kg-C)	Areal Wt	Ratio 1	Ratio 2
Aa	66.0	0.055	1200.000	0.04456		
Ab	82.0	0.086	953.488	0.03308		
Ac	120.0	0.023	5217.391	0.09207		
Ba	100.0	0.049	2040.816	0.13822		
Bb	41.0	0.015	2733.333	0.05740		
Bc	7.7	0.096	80.208	0.03535		
Ca	5.3	0.009	602.273	0.47734		
Cb	14.0	0.025	560.000	0.04154		
Cc	23.0	0.050	460.000	0.08006		
Site			900.183*		22.92	115.56

Conc is from Table G-1.
Foc is from Table D-3.
Norm Conc is Conc/Foc, normalized in terms of organic C.
Areal Wt is from Table F-1.
Ratio 1 is Norm Conc/Mean Saltwater SQC (= 41.8).
Ratio 2 is Norm Conc/Lower CI Saltwater SQC (= 8.29).
Site is the value for the site using areal weighting.

*The geometric mean after weighting by area.
That is, antilog (sum(log(Norm Conc*Areal Wt))/N).

the UOP site and is an Indicator Species; (ii) the contaminants (PCBs, mercury, and cadmium) are Indicator Compounds at the UOP site; and (iii) the sediments are probably of similar origin due to geographic proximity and are of comparable percent organic composition (6 to 22 percent) to those at the UOP site. Rubinstein et al. characterized the sediment contaminants at four sites in terms of estimated concentrations (µg/g, dry weight, from their Table 1) as:

Site	PCBs	Hg	Cd
Site A	0.46	4.13	5.32
Site B	0.71	2.85	11.49
Site C	7.28	34.89	38.60
Site D	0.72	2.71	5.16

Contamination of the sediments at the UOP site can be tabulated in this format using areally averaged concentrations and 1988 data for PCBs and Hg (µg/g dry weight, from Table F-2) as:

Site	PCBs	Hg	Cd
UOP	17.3	14.4	1.4

For comparison and contrast, contamination at the UOP site falls within the general range investigated by Rubinstein et al. and might be considered to fall between their Sites B and C in terms of Hg, although PCB concentrations are higher and Cd concentrations are lower.

Rubinstein et al. (1983) report that none of the sediments (Sites A through D) was acutely toxic to the three test species (clamworms or sandworms, *Nereis virens*; hard clams, *Mercenaria mercenaria*; and grass shrimp, *Palaemonetes pugio*) in 100 days. Mortality did not exceed 10 percent in any of the species or conditions tested. Of the three contaminants measured, only

PCBs accumulated above the control concentrations. For grass shrimp and hard clams, accumulation of PCBs was statistically different from control accumulation in sediments B, C, and D, and no accumulation appears to exceed 0.08 µg/g in tissue. The authors conclude that "grass shrimp and (hard) clams are primarily susceptible to water-mediated uptake," (p. 254), despite close proximity to the sediment, "whereas the sandworm is subject to uptake via interstitial water, ingestion, and absorption from sediments."

For clamworms and PCBs, bioaccumulation factors (BAFs) were calculated as:

$$BAF = \frac{\text{Tissue Concentration as wet weight}}{\text{Sediment Concentration as wet weight}}$$

BAFs ranged from 0.15 to 1.59 and were dependent on the organic content of the sediment. For example, sediment C had the highest concentrations of PCBs, but the lowest BAF. It also had the highest percentage of organic matter. These results suggest that the availability of PCBs to clamworms decreases (linearly) with an increasing percentage of organic material in the sediments.

Contaminants occurring as mixtures in sediments may interact additively, synergistically, antagonistically, or may not interact. Swartz et al. (1988) reported effects of mixtures of sediment contaminants on a (west coast) marine amphipod. Three of the four contaminants tested are the same as the compounds on the UOP site: zinc, mercury, and PCB (Aroclor 1254). The toxic effect (10-day LC50) of the mixture of these three contaminants was less than additive using the "toxic unit" model. The results of Swartz et al. may explain the lack of acute toxicity in Rubenstein et al.'s most contaminated sediments. This suggests that the total toxicity of the sediments on the UOP site is greater than that attributed to any one chemical, but less than the sum of the toxicities attributed to all chemicals.

Chapman et al. (1982) found that the presence of sediments increased tolerances of 12 species of oligochaetes (nine fresh water, three salt water) exposed to five contaminants (cadmium, mercury, pentachloraphenol, pulp mill effluent, and sewage sludge) and extremes of four environmental factors (pH, temperature, salinity, and anoxia). To explain how the presence of sediments would increase tolerance to both contaminants and temperature extremes, the authors suggested that in addition to chemically binding and physically adsorbing contaminants, sediments were a more natural environment for the worms, which reduced physiological stress and increased tolerance to other stresses.

The results of these studies present a consistent picture. Rubenstein et al.'s (1983) experiments indicate that acute toxicity may not be present where it might be expected on the basis of sediment contamination levels. Furthermore, species in close proximity to the sediments such as hard clams may take up the contaminants primarily from water, which lowers exposure and accumulation. Swartz et al.'s (1988) results with amphipods indicate that the toxicological interaction of some of the contaminants present in Rubenstein et al.'s sediments and in sediments at the UOP site may be less than additive. Chapman et al. (1982) found that presence of sediments increases tolerance of oligochaetes to both contaminants and environmental extremes. These findings suggest that methods presented here using pore water concentrations for comparison with toxicological benchmarks yield conservative or worst-case estimations of potential effects.

5.9 Discussion of Results

In the lefthand side of Table 5-5, the portion showing the ratios based on estimated water column concentrations, one or both of the quotients for chronic and acute exposures to mercury, zinc, and PCBs exceed "one." In the righthand side of Table 5-5, the portion showing the ratios based on estimated

pore water concentrations, several quotients exceed "one". Although the AWQC are not enforceable regulations for the water column, the toxicity quotients greater than one in Table 5-5 may identify potential problems.

In Tables 5-6a and 5-7a, only the ratios for chronic effects on mussels exposed to zinc and mummichogs exposed to PCBs (Table 5-7a) exceed "one." Yet, apparently healthy mussels and mummichogs exist on the site. In Tables 5-6a and 5-7a, the ratios for cadmium, chromium (both III and VI), lead, mercury, dichlorobenzene, phenol, and di(2-ethylhexyl) phthalate never exceed "one."

In Tables 5-6b and 5-7b, several toxicity quotients exceed "one". As noted before, the primary route of exposure to mummichogs, blue crabs, mussels and grass shrimp appears to be through the water column rather than the pore water. Table 5-6b indicates acute (mortality) effects to mummichogs, mussels, crabs, and grass shrimp in spite of apparently healthy populations on-site. Table 5-7b indicates chronic effects to mummichogs, grass shrimp and mussels for several compounds. Because of the sedentary life history of the platform mussel, they are more likely to experience chronic exposure than are mummichogs and grass shrimp which can migrate from the area. Platform mussels, however, were found to inhabit the site. The contradiction between these predicted results of acute and chronic effects and the observations of apparently healthy populations can be explained as follows: the primary route of exposure for at least some of these species may not be pore water, and/or estimates of effects based on pore water are conservative or worst-case and are not being realized.

The overall uncertainties and variabilities in the toxicity quotients for channel water and pore water may span several orders of magnitude for the same compound, reflecting the variability and uncertainty in the EEC values (the numerators) and in the BC values (the denominators). One

uncertainty is the availability of individual contaminants to organisms. A second uncertainty is the effects of mixtures, i.e., do mixtures of toxicants act in an additive or antagonistic manner when present in a complex matrix?

Although the toxicity quotients exceeding "one" suggest that some of the Indicator Species are stressed by the presence of the Indicator Compounds, the Focused Environmental Assessment results (Appendix C) suggest that the water concentrations are not high enough to cause any apparent population gradients or local extinctions. However, the Toxicity Quotients for PCBs in sediments, as based on the sediment quality criteria, are large enough to warrant concern even though many aquatic organisms can sustain high body burdens with no apparent effects in the individuals or in the populations.

The presence of apparently healthy populations of grass shrimp and mummichogs on the site combined with experimental studies on toxicity of sediments from New York harbor by other investigators suggest that sediment contaminants are not available to these species.

Thus, the use of the Toxicity Quotient Method as a screening method has identified the most important species and compounds for more detailed consideration in the next chapter.

6. ANALYSIS OF EXTRAPOLATION ERROR

Analysis of extrapolation error (AEE) is a relatively new technique for ecological risk assessment and is described by Suter et al. in Barnthouse et al. (1986). This method provides an objective and quantitative estimate of risk that is defined in terms of toxicological benchmarks. One of the advantages of the method is that it quantifies the level of uncertainty in the risk estimate; uncertainty which results from interspecies differences in sensitivity to chemicals and from the variable relationship between acute and chronic effects of chemicals. AEE was developed using data for fresh water fish and some invertebrates, and so cannot be applied to all species. Here, AEE is applied to a subset of the data used in the quotient method described in Chapter 5.

6.1 Overview of the AEE Method

Analysis of extrapolation error (AEE) is a method that allows the risk assessor to predict the probability that chronic effects would not occur. Because the method allows the calculation of the probability that an expected environmental concentration (EEC) will exceed some end point, Suter et al. (1986) consider AEE to be a true risk assessment method. AEE allows extrapolation between biological groups and extrapolation between different types of tests (e.g., acute and early life stage effects). The uncertainty associated with the extrapolations is calculated and contributes to the estimation of uncertainty in the risk estimate.

To implement the test, the first step is to define the end point of the risk assessment. In this case, the end point will be the probability of exceeding the threshold for chronic effects for three Indicator Species (mummichogs, blue crabs, and grass shrimp). These species were chosen because all are inhabitants of the site with conspicuous populations. Blue crabs also may be consumed as human food.

The next steps are to identify the test result for the chemical of interest that is most closely related to the chosen end point and to choose a logical progression to extrapolate from the existing datum to the chosen end point. For example, to estimate the risk to blue crabs for chronic effects from Cr (III), the closest test result is from an acute test of a gammarid amphipod. The logical steps are to extrapolate from amphipod acute tests to decapod acute tests (crabs are decapods) and then from acute tests to chronic tests. Because all possible taxonomic extrapolations are not included in Suter et al. (1986)--nor are they possible--the choice of logical steps depends on which taxonomic relationships have been done already. Lastly, the distribution parameters of the end point and the estimated risk that the EEC will exceed this end point concentration are calculated. The AEE method is described in greater detail in Appendix H.

Analysis of extrapolation error is applied to those Indicator Compounds identified by the quotient method as having estimated environmental concentrations which are near or exceeding the toxicological benchmarks for the Indicator Species. The Indicator Compounds selected for AEE are chromium (III), chromium (VI), total mercury (considered first as inorganic mercury and then as methyl mercury), zinc, and PCBs (as Aroclor® 1248). Although Section 5 indicates that water concentrations of chromium are well below toxicological benchmarks, chromium is selected for further investigation because sediment concentrations are well above natural crustal abundances (Appendix G). In the case of Indicator Species, AEE was developed primarily for fresh water fish, although some invertebrate extrapolations are possible. The method is not appropriate for ducks or mink, which are examined separately, or for mussels and midges, for which there are no extrapolation methods developed by Suter et al. (1986). AEE is applied to the remaining mummichogs, blue crabs, and grass shrimp.

In two places, a minor variation from Suter et al.'s (1986) method is employed. Carcinus maenas is considered a

surrogate for blue crabs exposed to inorganic mercury and P. setiferus (white shrimp) is considered a surrogate for grass shrimp. These surrogates are relatively close taxonomically and Suter et al. (1986) provide no extrapolations for these invertebrates.

For application of AEE to the UOP site, some assumptions were made about the Indicator Compounds, the populations of Indicator Species, and the routes of exposure. The test results or toxicological benchmarks were taken from U.S. EPA Ambient Water Quality Criteria documents, and so are all based on exposure of aquatic organisms by immersion in a test solution. Therefore, the route of exposure for this analysis is limited to water. AEE provides an extrapolation to chronic effects for each Indicator Species, which provides a benchmark for protection of that species.

Some assumptions about chromium concentrations are made. Most of the chromium at this site is believed to be Cr (III) (Appendix F), and so the environmental concentration of total chromium is assumed to be in the form of Cr (III). Cr (VI) in sediment was measured separately in 1988, and those concentrations were used to estimate Cr (VI). Two assumptions about the form of mercury are used. First, all mercury is assumed to be methyl mercury. This assumption is based on the observation that tidal creek sediments are typically organic and anaerobic and so would favor methylation. The second assumption is that all mercury is inorganic. This is based on studies of mercury in Berry's Creek that indicate that methylation may not occur in Berry's Creek and that fish are taking up inorganic mercury (Weis et al., 1986). For PCBs, the only Aroclor® detected was 1248, therefore total PCBs are assumed to be only Aroclor® 1248.

Because the AEE method was developed using only water-borne exposure as a basis for extrapolation, only modeling results of water contamination concentrations (Appendix F) are considered. Where more than one year of water concentrations were modeled, only model results from 1988 were

used because they are the most current and because 1988 sediment data represent surface sediments. Water quality modeling produces estimates of both mean and variation, and both are needed for AEE.

Several assumptions were also made with regard to the biological populations. First, because the shrimp, crabs and fish are mobile and because the creeks are well flushed by tides, the average site concentration is assumed to be a representative measure of exposure (average of log transformed concentrations, see Appendix F). The second assumption is that the populations inhabit the site long enough to incur chronic effects. With such mobile organisms and tidal fluctuations, this assumption may not be true. Since this assumption increases the possibility of predicting effects, it is considered conservative.

6.2 Results

The estimated concentrations of Indicator Compounds in channel water are shown on Table F-4. The estimated means and variances of log (10) transformed concentrations in µg/l are the expected environmental concentrations (EECs) for this analysis. For each of the three Indicator Species and the six Indicator Compounds, the most closely related toxicological endpoint is shown on Table 6-1. These endpoints are from U.S. EPA Ambient Water Quality Criteria documents. No tests of fish with Aroclor® 1248 were listed in the Ambient Water Quality Criteria document, so tests on Aroclor® 1242 and 1254 were used.

The endpoints on Table 6-1 are chosen to be the closest taxonomic endpoints, with salt water species chosen over fresh water species when possible and with chronic endpoints chosen over acute (Suter et al., 1986). This selection, however, is under the constraint that the endpoints must be taxonomic categories given by Suter et al. (1986) for making extrapolations. Also, fathead minnow data are used as the most

TABLE 6-1

EXISTING TOXICOLOGICAL RESULTS MOST CLOSELY RELATED TO INDICATOR SPECIES ENDPOINTS*

Indicator Compound	Mummichog (*Fundulus heteroclitus*)	Blue Crab (*Callinectes sapidus*)	Grass Shrimp (*Palaemonetes* sp.)
Chromium (III)	*Fundulus heteroclitus*, SW LC_{50} = 31,500	*Gammarus* sp., FW LC_{50} = 3200	*Gammarus* sp., FW LC_{50} = 3200
Chromium (VI)	*Fundulus heteroclitus*, SW LC_{50} = 74,010	*Callinectes sapidus*, SW LC_{50} = 93,390	*Gammarus pseudolimnaeus*, FW LC_{50} = 67.1
Inorganic Mercury	*Fundulus heteroclitus*, SW LC_{50} = 67.4	*Carcinus maenas*, SW LC_{50} = 14	*Penaeus setiferus*, SW LC_{50} = 17
Methyl Mercury	*Fundulus heteroclitus*, SW LC_{50} = 51.1	*Gammarus duebeni*, SW LC_{50} = 150	*Gammarus duebeni*, SW LC_{50} = 150
Zinc	*Pimephales promelas*, FW Lifecycle = 106.3	*Carcinus maenas*, SW (Surrogate) LC_{50} = 1000	*Corophium volutator*, SW LC_{50} = 4683
PCB	Aroclor® 1242 *Pimephales promelas*, FW Lifecycle = 9.0 Aroclor® 1254 *Pimephales promelas*, FW Lifecycle = 2.9	Aroclor® 1248 *Gammarus pseudolimnaeus*, FW LC_{50} = 29	Aroclor® 1248 *Gammarus pseudolimnaeus*, FW LC_{50} = 29

* All data are from USEPA Ambient Water Quality Criteria documents.
All concentrations are in µg/L.

** No usable test identified.

FW = Fresh Water Species

SW = Salt Water Species

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closely related data points for AEE extrapolations in Table 6-1 when specific data are unavailable as suggested by Suter et al. (1986). Species mean acute values from AWQC documents are used when several values are reported. Because of these selection criteria, the toxicity values chosen for Table 6-1 may differ from those chosen for Tables 5-3 and 5-4.

Table 6-2 shows estimates of certainty of not exceeding the threshold for chronic effects for each of the species and for each of the contaminants. The table can be read as a percent certainty or probability that a chronic effect would not occur. For example, we are 96.6 percent certain that a chronic effect would not occur in the mummichog population due to Cr (III). Calculations are shown in Appendix H.

6.3 Discussion

The results of this analysis agree well with the results of the quotient method, and add to that method by associating estimates of certainty with the results. Of the Indicator Compounds AEE indicates with about 90 percent certainty or better that only chromium (III and VI) is not exceeding the threshold for chronic effects on Indicator Species.

Section 5 indicates that chromium (III and VI) concentrations in water are well below toxicological benchmarks, and the AEE result extends that to extrapolated chronic thresholds for the Indicator Species.

For two species, the estimated mean environmental concentration of a chemical in channel water exceeds the estimated threshold for chronic effects: blue crabs and mummichogs exposed to zinc. For these two species, the estimated certainty of not exceeding the threshold for chronic effects is very low, less than 30 percent (Table 6-2). The reason that there is uncertainty about exceeding the threshold is that there is uncertainty about both the threshold for chronic effects for each species and about the modeled environmental concentration.

TABLE 6-2

ESTIMATIONS OF CERTAINTY OF NOT EXCEEDING

THE THRESHOLD FOR CHRONIC EFFECTS

Indicator Compound	Indicator Species: Mummichog (Percent)	Blue Crab (Percent)	Grass Shrimp (Percent)
Chromium (III)	96.6	95.8	95.8
Chromium (VI)	99.7	99.7	93.1
Mercury, Inorganic	90.1	75.2	77.3
Mercury, Methyl	88.1	96.9	96.9
Zinc	29.8	27.8	86.2
PCB: Aroclor® 1248	**	84.6	84.6
Aroclor® 1242	88.5		
Aroclor® 1254	76.7		

**No closely related endpoint for extrapolation.

In terms of estimates of exceeding the threshold for chronic effects of Indicator Species, methyl mercury, inorganic mercury and PCBs are in a middle area, neither clearly exceeding threshold for chronic effects nor so low that the estimate of not exceeding the threshold is fairly certain. The certainties of not exceeding the thresholds for chronic effects extrapolated to the most sensitive Indicator Species are as follows: inorganic mercury for blue crabs, 75.2 percent; methyl mercury for mummichogs, 88.1 percent, and PCBs for mummichogs 76.7 percent. These values are not high, but Suter et al. (1986) and Suter (Suter, personal communication) give no guidance or acceptable estimates.

Regarding blue crabs and chronic exposure to inorganic mercury in the channels, blue crabs will not spend their lives in the channels. Adults overwinter in the main basins of estuaries rather than in tidal creeks (Schaffner and Diaz, 1988), and we are aware of no studies suggesting that blue crabs return to the same marsh streams they left following overwintering. Indeed, female blue crabs migrate to the mouths of the estuaries to spawn and remain there (see Section 2.2). After hatching, the young crabs may not complete the migration up the estuary to the maturing grounds until the end of the first year or beginning of the second year of life. Considering that the larval stages, which are usually most sensitive, are spent at sea and that only a portion of the adult life of these crabs is spent in the channels, estimates of risk based on chronic effects should be conservative (protective) for this species.

Regarding mummichogs and chronic exposure to methyl mercury and to PCBs in the channels, the presence of an apparently healthy and sustained population on the site suggests that the chronic threshold is not being exceeded. Unlike blue crabs, mummichogs do not undertake extensive migrations and tend to stay in the same reach of a marsh creek (see Section C.4). This behavior should favor detection of population level effects. While the mummichogs are undoubtedly

incurring body burdens of contaminants (Section C.4), population level effects resulting from chronic responses are not evident.

Mercury concentrations in sediments are roughly equal on the site and in the reference area. Bacteria in sediments can methylate mercury, but methylation in Berry's Creek sediments appears to be inhibited (Weis et al., 1986). Mercury may be a generalized contaminant in this area, and the extrapolation in Table 6-2 to inorganic mercury may be more accurate than the extrapolation to methyl mercury.

Even with these certainty estimates from the AEE method, there are some uncertainties. The estimated environmental concentrations and the associated variances used in Table 6-2 are based on contaminant concentration modeling. Actual concentrations of the Indicator Compounds were below detection limits. Estimates of variance from the model account for some of the uncertainty, but field validation cannot be done.

There are uncertainties with application in this estuarine environment. The method depends on extrapolations that have already been done. Suter et al. (1986) used data from fresh water fish and invertebrates, and so there is some uncertainty about how well the results actually predict estuarine conditions. Their data come primarily from fish, but here the technique is also applied to invertebrates. The results presented here are therefore less certain for the invertebrates than for the fish. The AEE method also considers contaminants individually, such that any effects from additivity, synergism, or antagonism among contaminants is not addressed by this method. A qualitative discussion on additive, synergistic and antagonistic effects is presented in Section 5.8.

Also, this method does not address biomagnification of pollutants through the food chain. That is because the experiments that form the basis of the extrapolations are bioassay tests on the effects of exposure to water-borne contaminants. To some extent, the extrapolation to chronic

effects may account for the accumulation of contaminants. The degree to which bioaccumulation is accounted adds another level of uncertainty.

For Nereis diversicolor and N. virens, estuarine infaunal polychaetes, uptake of PCBs and mercury from contaminated sediments may be the principal route of accumulation (Fowler et al., 1978; Rubenstein et al., 1983). For marine species living in the water column, however, PCB accumulation may be due to direct partitioning from the water rather than from accumulation through the food chain (Scura and Theilacker, 1977). This may be the case for mummichogs. Grass shrimp (P. pugio) also appear to accumulate PCBs from the water rather than from the sediment (Rubenstein et al., 1983), possibly because they are epifaunal and swim readily. This may also be the case for blue crabs, which are also epifaunal crustaceans. In light of this, estimates of chronic thresholds for these Indicator Species by AEE should represent the field situation.

The accumulation of PCBs in infaunal and epifaunal organisms from sediments and water does, however, provide a continuous path of these contaminants from the sediments. Elder et al. (1977) indicated that the polychaete N. diversicolor remobilized PCBs from sediments, and Larsson (1984) suggested that chironomid midges could even transport PCBs from aquatic sediments to terrestrial environments since they live in aquatic sediments as larvae and terrestrial environments as adults. In the present analysis, mobilization of PBCs from the sediments into water is assumed to take place entirely by a physical process, with the conservative assumption that all water column total suspended solids come from the contaminated channel beds. Considering tidal flushing twice per day, this assumption may be conservative compared to biological remobilization, although there are no field data to verify this assumption.

There are also uncertainties about application of the analysis of extrapolated risk itself. This is a relatively new technique, and so lacks extensive field verification.

Subsequent work by the authors of these methods indicate that maximum acceptable toxicant concentration (MATC) based on early life stage tests, as is the case with many toxicological benchmarks used here, may not always be a good predictor of chronic effects in a population (Barnthouse et al., 1987; Suter et al., 1987).

In the final analysis, the objective of an ecological risk assessment is to predict the response of populations and communities. An underlying assumption of using AEE to predict population response is that knowledge about chronic (life-cycle) effects on the most sensitive life stage can, in fact, be translated to population response. On one hand, the translation seems straightforward: populations are groups of individuals, which implies that an effect on an individual will affect the population. On the other hand, populations have attributes individuals lack, such as birth rate, death rate, intrinsic rate of increase, ability to evolve, etc., which implies that populations can react in ways individuals cannot. Moreover, although methods such as those used here yield some indication of the levels of contamination that may cause individual response, the nature of the response is not predicted. So, the uncertainty of the predictions is increased.

To some extent, fisheries-derived population models can be applied to quantify the effects of contaminants and reduce the uncertainty of the translation. We know, for example, that a certain fractional change in mortality rate applied to fish at one life stage will cause a different response than the same fractional change applied at a different life stage. Also, some populations exhibit density-dependent response to increased mortality, which protects the population from extinction over some ranges of densities. Field populations of white suckers exposed to metal contamination and laboratory populations of cladocerans exposed to cadmium may exhibit such responses (reviewed in Barnthouse et al., 1987). Then too, field populations are known to adapt to environmental conditions.

Translation to ecological community response is more tenuous. The ways in which populations interact are often unknown, as are the processes that make ecological communities identifiable units.

In this risk assessment, the AEE method is used to quantify the uncertainty in estimating risks to individuals of a given species. The effects on populations and communities are qualitatively addressed from this information. A quantitative determination of the effects to populations and communities, however, would require additional analysis.

7. ECOLOGICAL RISK ASSESSMENT FOR MALLARD DUCKS AND MINK

7.1 Mallard Duck Risk Assessment

The UOP meadowlands site is an area which meets the primary requirements for mallard duck nesting. Requirements include dense vegetation at least 24 inches high and water within 100 to 500 yards (Bellrose, 1976). Because both the male and female ducks stay close to the nest during egg laying and incubation, and females rear the young, ENSR assumed that some ducks will nest at and remain at the UOP site for at least four months (30 days pre-nesting activities, 28 days incubation, 56 days rearing of young). It is reported that many mallards live in the meadowlands for the entire year (Smith, personal communication). Taking these factors into consideration, it is assumed, for purposes of worst case analysis, that some mallards nest in and remain near the UOP site for 365 days a year.

7.1.1 Toxicity Assessment for Ducks

Mallard ducks eat a wide variety of foods, including the leaves and stems of submerged aquatic plants, seeds of grasses and weeds, snails, insects, crustaceans, worms, and various marine invertebrates (Bellrose, 1976). When wildlife authorities in New Jersey were contacted, different reports were received about the types of foods mallards in the meadowlands area consume. One source (Gunther, personal communication) reported that the ducks eat a strictly vegetarian diet comprised mostly of grasses and roots, but another (Smith, personal communication) said that the ducks also eat small fish and infauna from waterways.

The diet of the mallard ducks varies greatly with the season of the year (Fitch, personal communication). Typically, the ducks will feed intensively in the spring, summer, and fall and usually try to consume high energy foods such as

mummichogs, crustacea, and insects. In the winter there is less food available from animal sources (fish) and the ducks will most likely eat a vegetarian diet.

Mallards in captivity in a laboratory setting consume approximately 100 grams (dry weight) of dry food per day (Heinz, personal communication). In a natural setting, Heinz estimates that mallards will eat 20 to 30 percent of their body weight or approximately 300 grams (wet weight) of food which is predominantly composed of aquatic vegetation, seeds and insects.

Mercury - Risk Assessment for Ducks

It is assumed that a mallard duck will consume about six killifish a day and seven ounces (0.198 kg) of vegetation (based on: Fitch, personal communication). Note that this diet totals approximately 230 grams of food per day; slightly less than the 300 grams estimated by Heinz due to the higher caloric content of fish relative to vegetation.

As a base case, the first risk assessment assumes that the ducks eat a diet of 100 percent aquatic plant roots. The second risk assessment assumes that the diet is comprised of both vegetation and mummichogs. The only data readily available concerning uptake by plants or bioaccumulation by invertebrates were for mercury. As toxicity information was also available for mercury, this compound was chosen for this risk assessment. Based on a study by Skripnichenko and Zolotoryeva (1980); the root uptake factor for plants is estimated as follows:

In a solution of 500 µg Hg as $HgNO_3$/L water, 60 µg Hg/kg plant tissue is taken up. Therefore, the ratio of mercury in water to mercury in plant tissue is:

$$\frac{60\ \mu g/kg}{500\ \mu g/L} = 0.12\ L\ water/kg\ plant\ tissue$$

When the root uptake factor is multiplied by the UOP mercury (inorganic) concentration in water, a value is estimated for the concentration of mercury in the UOP site plant roots.

$$0.12 \text{ L/kg} \times 2.50\text{E-01} \frac{\mu g}{L} = \frac{0.03 \ \mu g \text{ Hg}}{\text{kg plant}}$$

The only available toxicity value concerning ducks and mercury is one which is reported to cause reduced fertility and inhibited food conversion ability. This was a two-generation feeding study and the mercury concentration was estimated to be 0.1 mg/kg in the natural succulent food of the wild duck (Heinz, 1976). The toxicity index is then estimated as follows:

$$\frac{EEC}{BC} = \frac{0.00003 \text{ mg/kg}}{0.1 \text{ mg/kg}} = 3 \times 10^{-4}$$

Even with the inclusion of safety factors (10 or 100) in the denominator, the estimated toxicity quotients for ingestion of mercury-contaminated vegetation are low (3×10^{-3} or 3×10^{-2}).

The second scenario includes mummichogs in the diet. To determine the average concentration of mercury in mummichogs, body burden data from sites A, B, C, and D from both small and large fish were averaged. The mean value is 0.41 µg/g (see Appendix D and Table 7-3). The mean weight of a mummichog was calculated by averaging the raw data of all fish captured. This value is 5.13 g.

The total consumption of fish is multiplied by the average concentration of mercury in fish to estimate the average daily intake of mercury from fish. The total consumption of vegetation is multiplied by the concentration of mercury in plant roots to estimate the average daily intake of mercury from vegetation. The total intake of mercury per day then equals the daily intake of mercury from fish plus the daily intake of mercury from vegetation.

mummichog avg. weight	x	# eaten daily	=	total grams consumed daily
5.13 g	x	6	=	30.78 g fish/day

total grams consumed daily	x	average conc. Hg in fish	=	average daily intake of Hg from fish
30.78 g fish/day	x	0.41 µ/g	=	12.62 µg/day

kg of vegetation consumed daily	x	concentration of Hg in plant roots	=	average daily intake of Hg from vegetation

0.198 kg x 0.03 µgHg/kg plant = 0.006 µg/day

(average daily intake Hg from fish	+	average daily intake Hg from vegetation)	x	(1 mg/1000 µg)	=	total mg Hg consumed/day
$12.62 \frac{\mu g}{day}$	+	$0.006 \frac{\mu g}{day}$	x	$\frac{1\ \mu g}{1000\ mg}$	=	0.013 mg/day

To calculate the dose received per kg body weight per day:

$$0.013 \frac{mg\ Hg}{day} \div \frac{\text{avg. wt of duck}}{1.15\ kg} = 0.011\ mg/kg/day$$

The benchmark concentration which was previously used is that of 0.1 mg/kg in the natural succulent food of the duck (Heinz, 1976). To convert this benchmark to a dose it is assumed that ducks eat 300 grams of vegetation per day and the following calculation is performed:

Conc. Hg in plants	x	kg plant consumed/day	=	Hg ingested/day
$\frac{0.1\ mg\ Hg}{kg\ plant}$	x	$\frac{0.300 kg}{day}$	=	0.03 mg Hg/day

The amount of mercury consumed per day is then divided by the body weight of a duck to calculate a dose:

Hg ingested/day ÷ duck body weight	=	daily dose causing reduced fertility
0.03 mg Hg/day ÷ 1.15 kg	=	0.026 mg/kg/day

It is assumed that the availability of mercury from mummichog flesh to the GI tract of the duck is equal to the availability of mercury from plants to the duck's GI tract. In order to calculate a tolerable dose, which is assumed not to cause any adverse effects, a safety factor of 100 is applied to the dose which causes reproductive effects (0.026 mg/kg/day ÷ 100 = 0.00026 mg/kg/day). A hazard index is then calculated as follows:

$$\frac{\text{Dose received}}{\text{Tolerable dose}} = \frac{0.011 \text{ mg/kg/day}}{0.00026 \text{ mg/kg/day}} = 42.3$$

An index of 42.3 indicates that chances are good that adverse effects may occur to the mallard duck from exposure to mercury at the UOP site.

PCBs Risk Assessment for Ducks

A diet of 100 percent vegetation will be used to calculate risk to mallards from consuming PCB contaminated food. The assessment will assume that 10.6 ounces of vegetation (0.300 kg) a day is consumed. In the second assessment, the same diet of 7 ounces (0.198 kg) vegetation and 6 mummichogs per day will be used to calculate risk to mallards.

Data regarding the uptake of PCBs by plants could not be located, but literature on the uptake of PCBs by algae was obtained. The authors (Wang et al., 1982) report that the uptake of PCBs also occurs in plants that are dead, indicating that the process may represent absorption in lipid cell components. As the uptake mechanism is probably not a result of some physiological process unique to algae, the uptake values were assumed to also apply to aquatic plants.

The concentration of PCBs in plants is thus calculated:

Channel water concentration = 6.6×10^{-3} µg/L (from Table F-3).
PCBs in total suspended solids (tss) = 5.26×10^{-1} µg/L.

Adding the two sources of PCBs together = 6.6×10^{-3} + 0.526 = 5.326×10^{-1} µg/L

The concentration in plants is equal to the water and TSS concentrations multiplied by the BCF:

$$5.326 \times 10^{-1} \text{ ppb} \times 5500 = 2929 \text{ ppb or } 2929 \ \frac{\mu g \text{ PCB}}{\text{kg plant}}$$

The dose to the ducks eating a vegetarian diet is then calculated as:

Concentration of PCB in plant x kg plant consumed/day = µg PCB consumed daily

$$2929 \ \frac{\mu g \text{ PCB}}{\text{kg plant}} \times 0.300 \text{ kg} = 878.7 \ \mu g \text{ PCB/day}$$

Dividing by the average weight of a duck, a "dose received" is derived as follows:

$$\frac{878.7 \ \mu g \text{ PCB/day}}{1.15 \text{ kg}} = 764.1 \ \mu g/kg/day \text{ or } 0.76 \text{ mg/kg/day}$$

The chronic value which causes a highly significant decrease in hatchability of chicken eggs after 8 weeks of PCB treatment in food is 10 mg/kg/day (Scott et al., 1975). The dose which caused <u>no</u> significant decrease in hatchability of eggs was 1 mg/kg/day. This value (1 mg/kg/day) will be used to calculate the toxicity index. A safety factor is not needed in applying this value as a tolerable dose because 1 mg/kg/day of

PCB was observed (by Scott et al.) to have no adverse effects on chickens. Additionally, a safety factor is not necessary for extrapolating between species since chickens are more sensitive to PCBs than ducks (Lillie et al, 1974). The toxicity index is then calculated as follows:

$$\frac{\text{dose received}}{\text{tolerable dose}} = \frac{0.76 \text{ mg/kg/day}}{1.0 \text{ mg/kg/day}} = 0.76$$

The second scenario, in which ducks consume 0.198 kg plants and 6 mummichogs a day involves the following calculation:

$$2929 \frac{\mu g \text{ PCB}}{\text{kg plant}} \times \underset{\text{consumed daily}}{0.198 \text{ kg plant}} = \underset{\text{consumed daily}}{580 \ \mu g \text{ PCB}}$$

Dividing the amount of PCB's consumed daily by the average weight of a duck, a dose is derived:

$$\frac{580 \ \mu g \text{ PCB}}{1.15 \text{ kg (avg. weight of duck)}} = 504 \frac{\mu g}{kg}; \text{ converting to mg} =$$

$$\frac{504 \ \mu g}{kg} \times \frac{mg}{1000 \ \mu g} = 0.504 \frac{mg}{kg\text{-day}}$$

The next dose to be calculated is that received from fish ingestion. The average concentration of PCBs in fish is 17.6 mg PCB/kg whole body wet wt. fish (see Appendix D and Table 7-3).

The fish-ingestion dose is then calculated:

$$17.6 \frac{\text{mg PCB}}{\text{kg fish}} \times 0.03078 \frac{\text{kg fish consumed}}{\text{day}} = 0.542 \frac{\text{mg PCB}}{\text{day}}$$

0.542 mg/day ÷ 1.15 kg (avg. wt. duck) = 0.471 mg/kg/day.

Adding the plant and fish doses together, a final daily dose of PCBs is arrived at:

0.504 mg/kg/day + 0.471 mg/kg/day = 0.975 mg/kg/day
(from vegetation) (from fish)

The toxicity index is calculated by dividing the dose received by the tolerable dose:

$$\frac{0.975 \text{ mg/kg/day}}{1 \text{ mg/kg/day}} = 0.975$$

The toxicity indices for PCBs are calculated for ducks ingesting a vegetation and fish diet and a strictly vegetarian diet are both less than "1." Thus, exposure to PCBs at the UOP site is not considered to be a health threat to ducks.

The toxicity indices calculated for mallard duck exposure to PCBs and mercury are summarized in Table 7-1.

7.1.2 Conclusion and Uncertainties

Mercury

The low toxicity index calculated for the vegetarian diet implies that mercury poses no imminent threat to mallard ducks which consume vegetarian diets. While some ducks may eat a strictly vegetarian diet, others may also consume killifish (Smith, 1989), and exposure to mercury through the ingestion of fish and vegetation may pose a somewhat more serious threat. The toxicity index of 42.3 implies that the mallards may be in danger of experiencing adverse reproductive effects. Marine invertebrates and plants bioaccumulate mercury (Kopfler, 1974; Havlik et al., 1979) and as the duck is a higher food chain predator it may ingest mercury contaminated plants and animals which have very high levels of mercury.

In the vegetarian diet scenario, the toxicity index was estimated using a benchmark concentration which was derived using organic mercury, and the root uptake factor was derived using inorganic mercury. For this reason, a safety factor of 10 is appropriate. While it is not known if the benchmark

TABLE 7-1

MALLARD DUCK RISK ASSESSMENT

Compound	Diet	Ratio of Estimated Dose to Tolerable Dose
Mercury	Vegetation	0.03
	Vegetation and Fish	42.3
PCB	Vegetation	0.76
	Vegetation and Fish	0.975

value of 0.1 mg/kg was a lowest observable effect level or a mean of some doses, and what percentage of animals were affected, a second safety factor of 10 is appropriate as well. Even with these two safety factors of 10 included, the Toxicity Quotient is still low (0.03).

Another source of uncertainty involves calculating plant root mercury concentration strictly from water uptake. As both inorganic and organic mercury will adsorb to sediments (Ramamoorthy and Rust, 1976) root uptake through sediments should be considered. Once adsorbed, a dynamic process of desorption and re-adsorption begins, making it difficult to predict the chemical concentration available for uptake (Reimers and Krenkel, 1974). In addition, bacteria in the sediments may methylate some unknown fraction of the mercury in the sediments.

PCBs

In the PCB risk assessment the toxicity index using the vegetarian diet scenario is less than "1," indicating that the dose received by the ducks is less than that which causes adverse reproductive effects. Similar to the mercury assessment, the fish and vegetation diet in the PCB assessment resulted in a higher toxicity index than the vegetarian diet. The toxicity index for the fish and vegetarian diet is close to "1" (0.975), indicating that the dose received nearly equals the tolerable dose, and some sensitive individual ducks may experience adverse health effects. While it is tempting to draw the conclusion that the ducks will definitely be impacted by ingesting the PCB-contaminated vegetation and fish, some uncertainties need to be addressed.

The use of the chronic value, which was derived from a study using chickens, may have caused the toxicity index to be higher than it should be, as chickens are known to be more sensitive to PCBs than ducks (Lillie et al., 1974). Thus, the safe daily intake of PCBs for ducks may actually be more than

the value of 1 mg/kg/day which was used to calculate the toxicity index.

While the two sample diets used in the risk assessment calculations are representative of what some of the ducks are eating it cannot be confirmed that all of the ducks eat the same diet. Much as the diet of a human population can vary tremendously, so can the diet of the mallard duck. The best case is probably represented by the vegetarian diet scenario, as plants may be less likely to accumulate PCB's and mercury than fish and invertebrates which live in the sediments. This is demonstrated in the actual plant and fish concentration values used in the risk assessment. The worst case diet would probably be that of the mallards consuming only fish and invertebrates such as snails and mussels. As this scenario was not used in risk calculations, and it is likely to present a higher risk than that which would occur using a vegetarian diet, it should be concluded that the risk to mallards from both PCBs and mercury is highly variable, and that the type of diet eaten by individual birds will play the major role in intensity of exposure.

7.2 Mink Risk Assessment

Many opinions exist as to whether mink now inhabit the Hackensack Meadowlands. According to Edward Konsevick, Laboratory Manager and Supervisor of Research of the Environmental Operations Research Laboratory, Hackensack Meadowlands Development Commission, mink do not live in the Meadowlands (personal communication, February 20, 1989). The last reported mink sighting was in 1948 and no sightings have been reported during the Commission's Wildlife Survey for the past 17 years. Staff Naturalist, Don Smith has been trapping in the Meadowlands for more than 30 years and has not found any indications that mink are in the area. Smith believes that if a population existed to sustain the 10 minks caught in 1987-1988 it would be fairly large and he would have seen mink

or some signs of them. He also believes the Meadowlands as they now exist are not an ideal habit for mink.

Dr. Sydney Anderson of the New York Museum of Natural History (personal communication, April 12, 1989) has not researched the area in any great detail, but his impression is that mink did inhabit the Meadowlands at one time, but do not live there now.

Ms. Pat McConnell of the New Jersey Fish, Game and Wildlife Service has never visited the Meadowlands in search of mink, but based on trapping records for Bergen County and the habitat of the Meadowlands believes mink do live in the area (personal communication, April 26, 1989). This assumption is based strictly on habitat since trapping surveys are submitted anonymously and there is no way of knowing where in Bergen County the mink were caught. In a later communication, Ms. McConnell said that she may have located a trapper who has caught mink in the Meadowlands and will provide any additional information as soon as possible.

Dr. Richard Van Gelder, retired from the New York Museum of Natural History, had no specific information as to whether mink inhabit the Meadowlands or not. He believes, based on habitat, that if mink were trapped in Bergen County they probably came from the Meadowlands, but he is not certain of this. He has sighted mink in a residential area 15 miles away from the Meadowlands.

Dr. John Fitch, vertebrate ecologist and Executive Director of Mainewatch Institute agrees that the Meadowlands habitat is ideally suited to mink but was not convinced they now live there (personal communication, April 26, 1989). Since mink can travel a large area, Dr. Fitch believes a conduit or connection of some kind must be available for mink to use to travel from some other area into the Meadowlands. If this is possible, i.e., the Meadowlands are not isolated by roadways and buildings, the mink may migrate in and out of the Meadowlands. Don Smith reported that there is no good

connecting habitat in the area (personal communication, May 3, 1989). In fact, many miles separate the nearest appropriate habitat from the meadowlands. It was Mr. Smith's opinion that mink are more likely to inhabit the northeast and northwest sections of Bergen County where habitat is highly suitable. These areas are wooded watershed areas abundant in wildlife.

Based on the possibility that mink may inhabit the Meadowlands, the following ecological risk assessment is provided at the request of the New Jersey Department of Environmental Protection (NJDEP).

7.2.1 Mink - PCB Toxicity Assessment

Mink are sensitive to PCBs and will develop signs of toxicity at low doses. Due to this high degree of sensitivity, mink will be affected more by the presence of PCBs in the environment than any other compound. Therefore, only exposure to PCBs will be considered in this assessment.

This special study assesses the risk of PCB exposure to mink in a manner similar to human risk assessments. The toxicity of PCBs to mink will be discussed, followed by an exposure assessment, and risk characterization.

Of all species tested, mink are the most sensitive to PCBs. Levels as low as 0.64 ppm PCBs (type not specified) in the diet for 160 days caused reproductive failure in female ranch mink. One ppm may cause death (Eisler, 1986).

Aulerich and Ringer (1977) showed that the lethal dose varied inversely with the chlorine content of PCBs. Reproductive effects did not show the same relationship. Aroclor® 1254 (54 percent Cl) exerted a detrimental effect on reproduction when fed to mink at levels as low as 2 ppm for 8 months. However, Aroclors® 1016 (42 percent Cl), 1221 (21 percent Cl) and 1242 (42 percent Cl) fed at levels of 2-5 ppm for 9 months produced no adverse reproductive effects. No information on the reproductive effects of Aroclor® 1248 (48

percent Cl) in the mink were located. The reproductive failure induced by Aroclor® 1254 was not permanent. The reason for the varied reproductive response is not known, but may be due to the ability of the mink to metabolize selected PCB congeners (Eisler, 1986).

Signs of PCB poisoning in mink include anorexia, lethargy, bloody stools, fatty liver, kidney degeneration and hemorrhagic gastric ulcers (Aulerich and Ringer, 1977).

Aroclor® 1254 residues in subcutaneous fat of adult mink were up to 38 times the dietary levels with some individual congeners accumulating up to 200 times. Elimination of 50 percent of PCBs from the adipose tissues was approximately 98 days with 100% elimination by day 199 (Eisler, 1986).

The feeding level at which no measurable effect occurs is not known. Since death and reproductive toxicity have been reported at levels of 100 to 640 µg PCB/kg diet, the calculated maximum tolerance for mink is less than 1.54 µg PCB/kg body weight/day (Eisler, 1986). This value was derived by known growth rates of female mink between the ages of 7 to 13 weeks (weight increased from 560 g to 1,130 g) and by the percentage of body weight consumed as food on a daily basis (16.4 to 27.2). This growth rate along with a safety factor of 100 was applied to a dietary level of 0.64 mg PCB/kg diet to obtain a tolerable level of 1.54 µg/kg/day (Eisler, 1986). Due to the lack of a No Observable Adverse Effect Level (NOAEL), Eisler further speculated the tolerable level of PCBs for mink may be less than 1.0 µg/kg/day.

7.2.2 Mink Exposure Assessment

The wildlife makeup in the Hackensack Meadowlands is an ideal food source for mink. The mink is a carnivorous animal (weighing approximately 1.5 to 3 pounds) with a diet dependent upon the fauna of the environment. Usually, the mink's diet is composed of muskrats, fish, frogs, crayfish, mice, rabbits,

grasshoppers, small birds, eggs, salamanders, snakes, turtles, squirrels, chipmunks, fresh water mussels, clams, crabs, diving beetles, ducks and earthworms (Rue, 1981). A survey of organisms in the Berry's Creek area of the Hackensack Meadowlands was performed by the Hackensack Meadowlands Development Commission (HMDC, 1987). Compatable mink prey taken from this survey are listed in Table 7-2.

Since such a wide variety of game is available, the mink diet is difficult to predict. A study of summer and winter dietary habits of mink living in a New York marsh reported a diet of 34 percent fish in the winter and 32.4 percent fish in the summer (Rue, 1981). A similar study performed in North Carolina showed fish consumption to be as high as 61 percent. PCBs are known to bioaccumulate in fish and fish are likely to contribute the major portion of the mink's dietary exposure to PCBs. This assessment will focus on the PCB exposure to the mink from the fish portion of their diet. Since the New York study was conducted in a similar environment to the Hackensack Meadowlands, an average daily fish intake of 33 percent will be used. Mink eat approximately 3.5 ounces (100 g) of food per day (Rue, 1981), so 33 grams of fish are expected to be consumed daily.

The amount of time the mink spend on the UOP site can be estimated by the travel range of the mink and the area of the water bodies (stream channels) located on the site. The on-site stream channels occupy 150,000 ft^2 and are on the edge of the meadowland bordering East Rutherford (Appendix F). Male mink cover a maximum range of 25 $miles^2$ (6.96×10^8 ft^2) while the female mink usually cover less than 20 acres (871,200 ft^2) daily (Rue, 1981; J. Fitch, personal communication).

For this assessment, it is assumed that mink spend all their time foraging for food, while traveling the range. Since female mink appear to be more susceptible to the toxic effects of PCBs than males, and the females stay in smaller areas than

TABLE 7-2

POTENTIAL COMPONENTS OF MINK DIET IN THE BERRY CREEK AREA OF THE HACKENSACK MEADOWLANDS EAST RUTHERFORD, NEW JERSEY

Invertebrates:

- *Dytiscidae*: diving beetles

Fish:

- *Fundulus heteroclitis*: mummichog
- *Fundulus majalis*: striped killifish
- *Menidia beryllina*: inland silverside
- *Menidia menidia*: Atlantic silverside
- *Morone americana*: white perch
- *Morone saxatilis*: striped bass
- *Anguilla rostrata*: American freshwater eel

Reptiles:

- *Chelydra serpentina*: snapping turtle
- *Kinosternon s. subrubrum*: E. mud turtle
- *Clemmys guttata*: spotted turtle
- *Malaclemys t. terrapin*: N. diamondback terrapin
- *Chrysemys p. picta*: E. painted turtle
- *Natrix s. sipedon*: N. water snake
- *Storeria d. dekayi*: N. brown snake
- *Thamnophis s. sirtalis*: E. garter snake
- *Thamnophis s. sauritus*: E. ribbon snake
- *Coluber c. constrictor*: N. black racer
- *Lampropeltis t. triangulum*: E. milk snake

TABLE 7-2 (Continued)

Amphibians:

- *Rana clamitans melanota*: green frog
- *Rana utricularia*: S. leopard frog

Crustaceans:

- *Callinectes sapidus*: blue crab

Mammals:

- *Sciurus carolinensis*: E. grey squirrel
- *Tamias striatus*: E. chipmunk
- *Peromyscus leucopus*: white footed mouse
- *Microtus pennsyvanicus*: meadow mole
- *Ondatra zibethica*: muskrat
- *Rattus norvegicus*: Norway rat
- *Mus musculus*: house rat
- *Zapus hudsonius*: meadow jumping mouse
- *Sylvilagus floridanus*: E. cottontail rabbit

Numerous Ducks and Small Birds

Source: Hackensack Meadowland Development Committee, 1987

the males, this assessment will focus on PCB exposure to female mink. For a worst case assessment, the female mink is assumed to live on the UOP site with her den directly on one of the channels connecting Berry's Creek to the larger stream channels on the UOP site. To obtain the amount of time the mink can be expected to visit the UOP streams to collect 33 grams of fish daily, the total area occupied by UOP water is divided by the total area the mink travels in search of food.

$$\frac{150{,}000\ ft^2}{871{,}200\ ft^2} = 0.17$$

Daily, the mink may spend an estimated 17 percent of her time in the UOP stream channels. Furthermore, the mink is conservatively assumed to eat fish each time she visits the water site.

Analysis of fish found in water on the UOP site revealed an average concentration of 17.6 mg/kg PCBs (whole body analysis, wet weight). Concentrations expressed in wet weight were estimated by multiplying concentrations expressed in dry weight by the fraction of solids present in the tissue sample. Both dry weight values and percent solids are located in Appendix D. See Table 7-3 for all wet weight values. The following equation is used to calculate the amount of PCB-contaminated fish that mink living on the UOP site may ingest from the UOP water.

Fish Ingestion for UOP site water = Total Ingestion x Fish intake x UOP stream intake

= 100 g/day x 0.33 x 0.17

= 5.61 g/day

Therefore, mink can feasibly ingest 5.61 g of PCB containing fish from the UOP site daily. To calculate the maximum-daily dose of PCBs mink may ingest:

TABLE 7-3

CONCENTRATIONS OF PCBs AND MERCURY IN MUMMICHOG AT UOP SITE
EAST RUTHERFORD, NEW JERSEY

Sample Location	PCB (Aroclor® 1248) (mg/kg, wet weight)	Hg (µg, wet weight)
Aa	22.85	0.39
Aa	9.92	0.45
Ab	19.72	0.31
Ab	29.71	0.53
Ac	19.36	0.38
Ac	14.62	0.31
Ba	19.66	0.33
Ba	18.43	0.42
Bb	22.84	0.31
Bb	22.28	0.34
BC	18.14	0.40
Ea[2]	7.63	0.42
Ea[2]	16.5	0.40
Ca	19.65	0.41
Ca	18.76	0.45
Cb	21.84	0.28
Cb	19.23	0.47
Cc	26.2	0.31
Cc	14.82	0.48
Da	9.69	0.44
Da	13.73	0.47
Db	6.92	0.75
Db	34.08	0.27
Dc	7.23	0.39
Dc	6.49	0.50
	$\bar{X}$ = 17.61 ± 7.09	$\bar{X}$ = 0.41 ± 0.10

Source: Appendix D

$$PCB_{dose} = \text{Concentration of PCB in fish} \times \text{fish intake} \times \frac{1}{\text{mink body weight}}$$

$$= 17.61 \text{ mg/kg} \times 0.00561 \text{ kg/day} \times \frac{1}{0.91 \text{ kg}}$$

$$= 0.109 \text{ mg/kg/day}$$

$$\cong 109 \ \mu\text{g/kg/day}$$

7.2.3 Mink Risk Characterization

The estimated exposure dose of 109 µg/kg/day is approximately 70 times higher than the estimated maximum tolerable dose of 1.54 µg/kg/day set by Eisler (1986). However, it must be kept in mind that all the factors used to calculate the exposure dose assumed a conservative, worst case scenario and this exposure dose is probably an overestimate of the actual PCB intake of mink, even if they are in the area.

Due to the location of the UOP site (i.e., at the edge of the Meadowlands with the highly developed city of East Rutherford and several major roadways nearby) mink are probably very unlikely to actually live or range on the UOP site. Large parts of the Meadowlands are relatively uninhabited by people and support a large variety of suitable game for the mink to live on, and would therefore be far more suitable habitat for the mink.

A male mink can travel great distances with speeds ranging from 2-4 miles per hour (Rue, 1981). At this rate, the male could easily cover the entire Meadowlands area each day. Male mink cover an estimated maximum range of 25 square miles (Rue, 1981; J. Fitch, personal communication). This large range would make it extremely unlikely that the male mink would visit the UOP site and obtain fish from the UOP streams and channels with any greater frequency than the female mink living on the site. While the speed that mink travel provides an indication of how mink may use the UOP site, for the purpose of estimating

chronic risk, travel speed is less important than fraction of time spent on the site. Here we assume that the probability of the occurrence of a male mink in the UOP area is equal to the ratio of the area of the UOP site to the entire area of the mink's range. This assumes that there is an equal probability of the mink being in any one place within his range at any one time.

Thus, the exposure dose to male mink eating PCB containing fish from this site would be extremely small, as they would visit the site only 0.02 percent of the time (calculations based on previous formulas). Assuming the male mink weighs 1.25 kg, his estimated exposure dose would be 0.09 µg/kg/day, which is well below the maximum tolerable dose of 1.54 µg/kg/day and results in a toxicity index of 0.06.

The toxicity index value is directly proportional to the ratio of the UOP site area to the total range area. The limited information available indicates that male mink range about 25 square miles. Even if that range were overestimated by a factor of two, the resulting toxicity index would still be only 0.12. The summary of mink risks to PCB exposure are summarized in Table 7-4.

The farther the female mink establishes her den away from the UOP site, the less likely she is to travel to the UOP stream channels to obtain fish. Even if the female established her den at the point where the channel enters into Berry's Creek, her range would not be great enough to take her up the channel to the UOP streams. Living at any point beyond the channel on the UOP site would reduce the risk of female mink eating fish containing PCBs from the UOP streams to that of the male mink or lower.

7.2.4 Conclusion and Uncertainties

The ecological risk assessment for the mink was done by estimating the exposure dose to the animal. Due to the uncertainty associated with the actual location of mink in the

TABLE 7-4

MINK RISK ASSESSMENT

Mink Gender	PCB Ratio of Estimated Dose to Tolerable Dose
Male (25 mi^2 range)	0.06
Male (12.5 mi^2 range)	0.12
Female	70

Meadowlands and intraspecies variability, several assumptions were made:

- female and male mink were assumed to weigh 0.91 and 1.25 kg, respectively;
- female mink were assumed to travel 20 acres daily in search of food. Male mink were assumed to travel: a) 25 square miles daily; or b) 12.5 square miles daily;
- all mink were assumed to eat 33 grams of fish daily; and
- each time mink traveled to the UOP streams, they were assumed to eat PCB-containing fish.

Due to a lack of data on no-observable effect levels of PCBs in mink, the estimated maximum tolerable does used in this risk assessment is also uncertain.

This analysis presents two extremes of risk to mink. The worst possible case includes high exposure to female mink living directly in the channel. The best case is that mink do not frequent this site.

8. UNCERTAINTY ANALYSIS

8.1 Introduction

An ecological risk assessment estimates the risks to the ecology of an area, based on a limited knowledge of the complex dynamics of the ecosystem and limited toxicological data on the compounds of interest and their effects on the species in the environment. Thus, numerous assumptions must be made when assessing the potential ecological impacts of chemical effects on an ecosystem. Some of the assumptions have a strong scientific basis, while others are more uncertain due to inadequate scientific data. Because we are not certain about each of the assumptions, some uncertainty is introduced into the risk assessment process every time an assumption is made. The sources of the uncertainty are described in qualitative terms, because in most instances, with the exception of the Analysis of Extrapolation Error method, there is not enough information to quantify the magnitude of those uncertainties.

There are two categories of uncertainty associated with risk assessment: 1) that which is inherent in the risk assessment methodology; and 2) that which is associated with our knowledge of the site, referred to as site-specific uncertainty. Site-specific uncertainties include uncertainties in the scientific understanding of the biological effects of environmental stressors and the complexity and inherent variability of natural populations and ecosystems, as well as uncertainties about the system dynamics of the chemicals of interest on the site. The following discussion will qualitatively identify many of these sources of uncertainty.

8.2 Natural Variability in the Environment

The physical and biological relationships in an ecosystem are intertwined into a complex web of interdependent species

co-existing in a dynamic environment. There is a great deal of natural variability inherent in the environment due to successional changes over many years, seasonal changes during a year, weather changes from day to day, and, in an estuarine environment such as the UOP site, tidal changes within a day. With the flood and ebb of the tide, the physical environment and the species living there are exposed to wide variations in water availability and flow, salinity, nutrients, oxygen, and temperature. The environmental conditions in an estuary are always changing. This natural variation results in an ever-changing physical and biological system dynamic that is difficult to quantify.

Sediment concentrations and water concentrations measured at the UOP site may be highly variable due to the natural variability in this estuarine environment. An assumption is made in conducting this risk assessment that the samples collected at the site are truly representative of the average physical conditions of this dynamic environment. Because the sediments are more stable than the flowing water, this assumption will tend to be more appropriate for the sediment measurments than the water measurements.

The fauna living in this dynamic estuarine environment also demonstrates considerable variability in their movements and habitats. The pelagic organisms, such as the Fundulus spp., may move in and out of stream segments along with the tide, and thus will be exposed to varying water concentrations of chemicals. Benthic organisms, on the other hand, may vary their movements, according to tidal flow, in a vertical direction, possibly burying into the sediments at low tide and returning to the sediment surface as the tide returns. These movements will result in a varied exposure to concentrations of chemicals within the sediments. The methods of the ecological risk assessment assume that, despite the movement dynamics of the organisms within the environment, their exposure to chemical concentrations in the soil and the water is relatively constant. It is also assumed that the natural variation in the

physical environment would neither lessen nor increase the potential effects of the chemicals of concern on these organisms. These simplifying assumptions are necessary for the practical application of the risk assessment methodology.

8.3 Uncertainty in Sources

The fate and transport of the chemicals of concern in this dynamic, fluctuating environment are uncertain. Samples were not collected at Berry's Creek, so it is not known how much, if any, chemical loading is being contributed by other sources. The chemical transport by means of tidal fluctuations from one creek to another is uncertain. It should be noted that elevated concentrations of several chemicals were measured at the reference (control) area. This implies that the meadowlands area is not pristine and experiences many other sources of chemical loading (additional to the UOP site).

Several of the water chemistry analyses measured chemical concentrations that were below the analytical detection limit. This limitation in analytical methodology lends uncertainty to the true chemical concentrations in the water. For the purposes of the risk assessment based on water concentrations, it is conservatively assumed that these chemical concentrations were at levels of half the detection limit. Further data on water concentrations were developed by means of a model. Uncertainties inherent in the water model are discussed in Appendix F.

8.4 Uncertainties in Quotient Method

The quotient method measures the potential risk to a species by comparing the estimated environmental concentration to a toxicological benchmark concentration. Uncertainties in the estimation of the environmental concentration are discussed in Section 8.3. Many uncertainties also accompany the use of toxicological benchmarks. First, the toxicological data

available are limited in its scope of species covered, chemicals, and measured effects. Secondly, the data are inherently variable due to differences in scientific methodology applied by different researchers.

In some cases, the toxicological benchmark used for the evaluation of potential risk to an indicator species was not that of the same species. In some instances, the toxicological benchmark selected, in the absence of more specific data, may not measure the same effect as the chosen endpoint for the indicator species.

These limitations and uncertainties in the quotient method result in an uncertain ratio as the measurement of potential ecological risks.

8.5 Uncertainty in AEE Method

The Analysis of Extrapolation Error (AEE) method carries with it the same uncertainties as that of the quotient method in the use of toxicological benchmarks. However, the AEE approach attempts to quantify that uncertainty by estimating the variance about the assumptions made in extrapolating toxicological data between different species and between different endpoints. The AEE method allows for this measure of uncertainty by expressing results as a function of probability.

8.6 Uncertainty in Mallard Duck and Mink Risk Assessments

The uncertainties in the risk assessment for the mallard duck and the mink are discussed in Sections 7.1.2 and 7.2.4, respectively.

9. SUMMARY

This document presents an ecological risk assessment for the UOP site along Berry's Creek in East Rutherford, New Jersey. A preliminary survey of the vegetation and wildlife on the terrestrial portions of the site indicated no need for an ecological risk assessment for those areas. The habitats on the site designated for risk assessment are the tidal channels. The methodology generally follows the format used in public health risk assessments. The major components in this report include a description of both the Indicator Species and Indicator Compounds selected for this analysis, a discussion of exposure including a description of the model used to estimate contaminant concentrations in water, a characterization of ecological risks by three methods, and a discussion of uncertainties associated with such analyses. The three methods are the Toxicity Quotient Method, the Analysis of Extrapolated Error Method (described by Oak Ridge National Laboratory for aquatic species), and a dose-response method for mallard ducks and mink.

Selection of Indicator Compounds is based on concentrations of chemicals reported on the site in both sediments and water, the frequency of detection for the contaminants, federal Ambient Water Quality Criteria, and knowledge of potential toxicity of the chemicals. The final selection includes both metals and organic compounds. The eleven indicator chemicals for ecological risk assessment are cadmium; chromium, mercury; lead, zinc; 1,2-dichlorobenzene; 1,2-diphenylhydrazine; di-2-ethylhexyl phthalate; PCBs; phenol; and total PAHs. Indicator Species are selected to represent a variety of taxonomic groups and life histories. The seven indicator species are Chironomidae or non-biting midges, *Palaeomonetes pugio* or grass shrimp, *Congeria leucophaeta* or platform mussel, *Callinectes sapidus* or blue crab, *Fundulus heteroclitus* or mummichog, *Anas platyrhychos* or mallard duck, and *Mustela vison* or mink.

For the aquatic species, two routes of exposure to the Indicator Compounds are assessed: from the column water and from the sediment pore water. These species are the midges, grass shrimp, platform mussel, blue crab, and mummichog. For at least some of these species and chemicals, published studies indicate that uptake comes primarily from column water. For many of the Indicator Compounds, concentrations in the tidal channels are below test detection limits. To provide estimates of chemical concentrations and a measure of uncertainty associated with those estimates, column water and pore water contaminant concentrations are modeled.

Water concentration modeling is based on field data for surface sediment concentrations. Sediment samples are assumed to represent the reach of a channel where they are collected, and by defining and estimating the area of the reaches, an areally weighted average sediment concentration is obtained. From this, an average pore water concentration is calculated using the conservative (worst-case) assumption of equilibrium of contaminants between pore water and sediment solids. A water quality model that treats the contaminants as a slick on the bottom of a river is modified to predict concentration of dissolved contaminants in the channels, with the slick being the contaminants at pore water concentration. Monte Carlo simulations are employed to transform frequency distributions from areally averaged sediment calculations and the sediment-water partition coefficient into frequency distributions for sediment pore water and channel water concentrations. For organic contaminants likely to partition to suspended sediments, suspended sediment are assumed to have the same concentration of contaminant as bed sediments. For these compounds, the channel estimated water concentrations are the sum of the predicted dissolved concentration and the suspended sediment concentration.

The modeled water concentrations (column and pore water) are then compared to toxicological benchmarks in the Toxicity Quotient Method (QM) described by Oak Ridge National

Laboratory. The QM is simply an arithmetic comparison made by forming a ratio of an expected environmental concentration (EEC) to a benchmark concentration (BC) from a toxicological study.

First, modeled column water concentrations of Indicator Compounds are used as EECs. Three types of BCs are used: Ambient Water Quality Criteria prepared by the U.S. EPA and both acute and chronic test results for the indicator species or their surrogates. Results indicate that mercury, zinc, and PCB concentrations in column water exceed the Ambient Water Quality Criteria and/or the chronic toxicological benchmark concentrations.

Second, the toxicity quotient analysis is also conducted using modeled sediment pore water concentrations of Indicator Compounds as the EECs. Results indicate that pore water concentrations of cadmium, lead, mercury, zinc, phenol and PCBs exceed the Ambient Water Quality Criteria and/or one or more of the acute and/or chronic toxicological benchmark concentrations. However, this comparison may not be accurate since the Ambient Water Quality Criteria and benchmarks are based on an ambient open water environment, not on a sediment/interstitial water environment. Also, this pore water analysis assumes that the indicator species are constantly exposed to the pore water, whereas, in actuality, much of their exposure is through the column water.

For several of the Indicator Compounds for which the Toxicity Quotients exceed one, and for chromium, which occurs in high concentrations in sediments on the site, a second analytic technique is applied. This technique is the Analysis of Extrapolation Error (AEE), also described by Oak Ridge National Laboratory. AEE is a relatively new method for ecological risk assessment that allows a statement of uncertainty about the environmental concentration exceeding toxicological benchmarks. The method has two components. First, a mean and variance for a chronic toxicological benchmark for the indicator species are estimated from other

known test results and known relationships among results. Then, the probability that the environmental concentration will exceed the extrapolated endpoint is calculated based on the results of the first component and the mean and variance of the modeled water quality data.

The AEE method indicates that it is probable that the mean modeled channel water concentrations of zinc exceed extrapolated mean thresholds for chronic effects on mummichog and blue crab. There is some uncertainty in this conclusion due to the uncertainty in both the concentration in the water and the extrapolated benchmark concentrations for the species. There is also a relatively low probability that channel water concentrations of PCBs and inorganic mercury will not exceed chronic benchmarks, although mean modeled water concentrations are below mean toxicological benchmark extrapolations. On the other hand, it is highly probable that modeled channel water concentrations of chromium do not exceed the endpoint concentrations for chronic effects on the Indicator Species. For methyl mercury, the assessment results are less clear. However, literature studies indicate that bacteria in Berry's Creek sediments do not methylate mercury. Therefore, species on the site may not even be exposed to methyl mercury.

Risk assessments for mallard and mink are based on comparisons of doses of ingested contaminants to toxicological benchmarks obtained from the literature on dosage experiments. Foodchain pathways were evaluated for mallards consuming mercury-contaminated and PCB-contaminated food items and for mink consuming PCB-contaminated food items. PCB body burdens were directly measured in Fundulus heteroclitus collected at the UOP site and were found to be in a range of 6.40 to 34.08 mg/kg with a mean of 17.6 mg/kg (wet weight). This data is used in the mink and mallard risk assessments as a means of evaluating biomagnification of PCBs via the foodchain.

Mallards eat several different food items, so for each of these two Indicator Compounds, two feeding regimes were assumed based on reported diets. The first diet is assumed to be all plants, and the second is assumed to be about 30 grams of

mummichogs and 200 grams of plants. Measured mummichog body burdens were used in the analyses. Hazard indices for mercury (defined as dose received/benchmark dose) are 0.03 for the vegetation diet and 42.3 for the mixed diet. For PCBs, the hazard indices are 0.76 and 0.975 for the vegetarian and mixed diets, respectively. The results indicate that a vegetarian diet does not appear to pose any potential adverse effects, while the mixed diet shows a high potential for adverse effects.

The risk assessment for mink is performed in a similar manner. Only risk from PCBs is considered because of the mink's sensitivity to the effects associated with that compound. Mink also eat a variety of foods, with one New York study indicating a diet of about one third fish. Assuming a conservative case, mink eat a diet of one third fish with an estimated body burden equal to the mean measured body burden of mummichogs on the UOP site. After estimating average mink weight and average total food intake, a maximum possible dosage can be calculated. By knowing the areal ranges that mink travel and comparing that to the area of the channels where the contaminated mummichogs are available, a fraction of diet coming from the site is calculated under the conservative assumption that both a male and female mink have the center of their range on the site next to the channels. Because of the difference in ranges between sexes, the predicted risks are different. The conservative-case estimated exposure dose for a female mink with a den on the site is 109 ug PCB/kg body weight-day, or 70 times higher than the estimated tolerable dose. For the male, however, the conservative-case estimated exposure dose is 0.09 ug/kg-day, or about 17 times less than the estimated tolerable dose. An opposing scenario is that mink do not inhabit the site and, therefore, would not exposed. Because of the small area of the site compared to the Meadowlands and the relative proximity of the site to a major road and human activities, the probability of female mink making dens on the site and experiencing such conservative-case conditions remains unclear and may be quite small.

There are many uncertainties associated with this risk assessment and the various techniques employed. Under the best conditions, ecologists have limited ability to predict future dynamics of single populations and less ability to predict the behavior of complex systems. Each step of this risk assessment makes simplifying assumptions without which the task would be overwhelming, but with which uncertainty increases. Uncertainties are reviewed in each section, but several sources pervade the entire assessment. First, many concentrations of chemicals in channel water are below test detection limits some or all of the time, and so actual concentrations are unknown and model results cannot be completely verified. Second, there is natural variation in almost all aspects of animal physiology, as well as great variability in most physical aspects of estuarine habitats. However, some sort of mean or typical condition must be assumed for each method. In addition, toxicological results for the Indicator Species exposed to the indicator chemical long enough to make an ecologically meaningful observation are not always available, and so these assessments rely on extrapolations, surrogate species, and other assumptions that cannot be completely evaluated. Comparisons between the reference area and the site area for mercury are not made because mercury concentrations are somewhat higher in the reference area than the site area. Berry's Creek estuary is well known for its mercury contamination with its likely source from upstream of Ackerman's Creek.

APPENDIX A

PRELIMINARY SURVEY

APPENDIX A
PRELIMINARY SURVEY

A.1 Introduction

ENSR conducted a preliminary site assessment on 6 and 7 October, 1988 to obtain baseline data on the planktonic and benthic organisms present in the stream channels. These samples were used to select and estimate abundance of indicator species. In addition, terrestrial vegetation and vertebrates were observed, and data were collected on species presence, relative abundance (vegetation), or activity (vertebrates) in order to assess the need for including terrestrial portions of the site in the ecological risk assessment. The following sections contain data summaries; the raw data from the preliminary survey are presented in Appendix B.

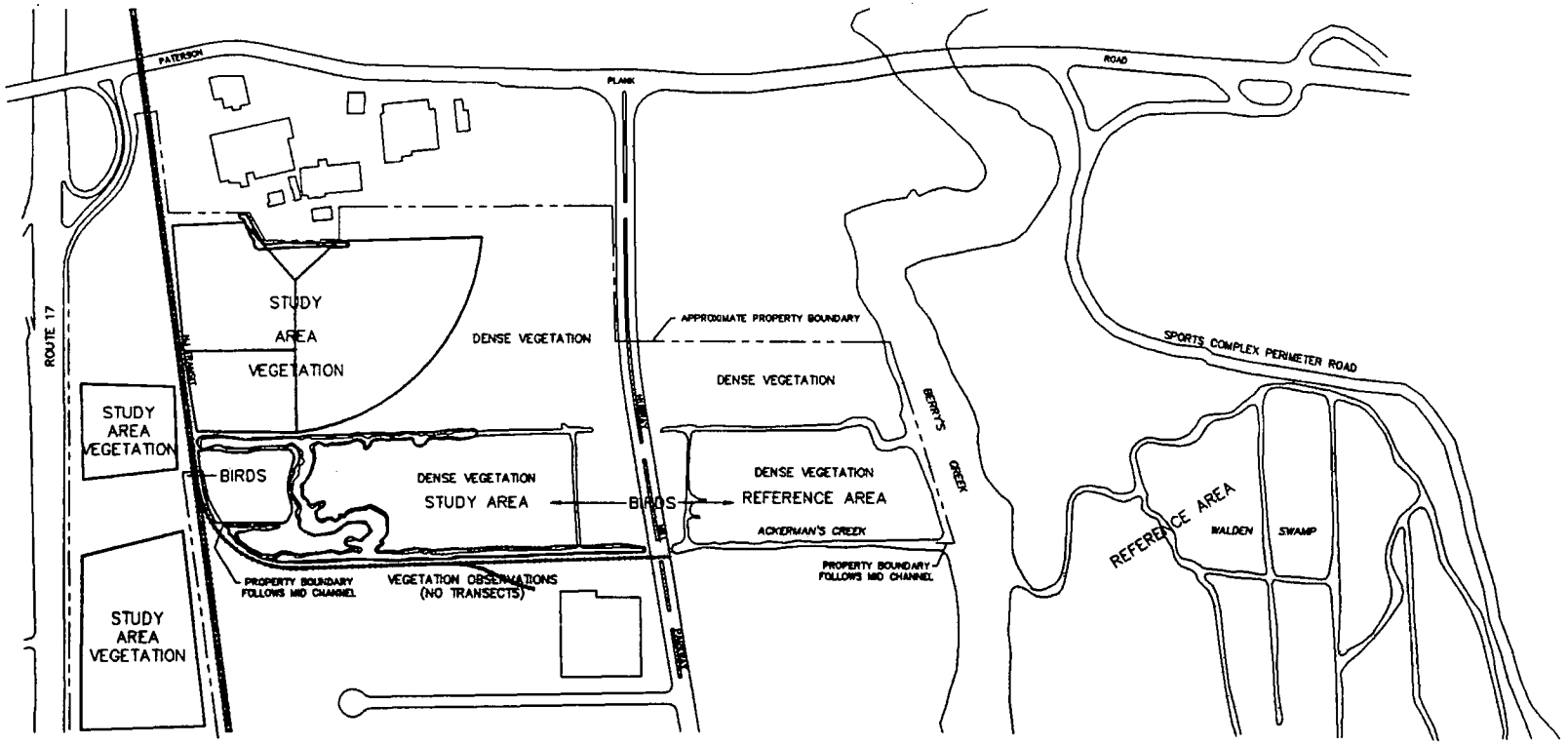
A.2 Description of Study Site and Reference Area

Reference areas for the survey were designated by NJDEP (Figure A-1). The "study site" was considered to be the upland portion of the UOP-owned property (for the vegetation survey) and stream channels A, B, and C (for the aquatic species survey). Stream channel D and the area depicted on Figure A-1 are respectively, the aquatic and terrestrial reference areas.

A.3 Terrestrial Survey

A.3.1 UOP Site Vegetation Study

Plant species identified on 6 and 7 October, 1988 at the UOP study site are listed in Table A-1. This list was checked against Hough's (1983) list on the biogeography of plants in New Jersey. All of the species have been reported previously from the area with the following exceptions: Ambrosia



NO.	DATE	REVISIONS	BY	CHKD	APP'VD

ENSR.
ENSR CONSULTING AND ENGINEERING

SITE AREAS
UOP SITE
EAST RUTHERFORD, NEW JERSEY

DESIGN: MCW	DATE: 4/21/89	FIGURE NUMBER: Figure A-1
DRAWN: KLU	SCALE: AS NOTED	
CHECKED: MCW	APPROVED:	PROJECT NUMBER: 6020-00(REV NO.

TABLE A-1

PLANT SPECIES OBSERVED AT THE UOP SITE

DISTRIBUTION AREA

Plant Species	On-Site[a]
Acer rubra	X
Achillea sp.	X
Agrostis sp.	X
Ailanthus altissima	X
Ambrosia artemesifolia	X
Ambrosia trifida	X
Andropogon sp.	X
Apocgnum androsapmifolium	X
Artemisia sp. (b)	X
Asclepias sp.	X
Asclepias syriaca	X
Aster dumosum (c)	X
Aster subulatus	X
Baccharis halmifolia	X
Bidens frondosa	X
Chenopodium sp.	X
Cirsium discolor Cuscuta sp.	X
Cyperus strigosus	X
Daucus carota	X
Eupatorium rugosum	X
Eupatorium serotinum (d)	X
Galinsoga sp.	X
Lepidium virginicum	X
Linaria vulgaris	X
Liquidamkar styraciflua	X
Lythrum Salicaria	X
Lycopus virginicus	X

TABLE A-1 (Continued)

Plant Species	On-Site
Melilotus sp.	X
Morus sp.	X
Oenothera biennis	X
Oxalis sp.	X
Parthenocissus quanquefolia	X
Phargmites sp.	X
Phytolacca americana	X
Plantago lanceolata	X
Pluchea purpurescens	X
Polygonum persicaria	X
Populus deltoides	X
Pyrus sp.	X
Rhamnus sp.	X
Rhus radicans	X
Rhus sp.	X
Rubus sp.	X
Setaria faberii	X
Solanum dulcamara	X
Solidago graminifolia	X
Solidago rugosa	X
Rolidago sempervirens	X
Sonchus eleraceus	X
Vitis sp.	X
Xanthium strumarium	X

(a) *Ambrosia psilostachya* was identified (probably incorrectly), but was not found to be indigenous to NJ according to Hough (1983).
(b) *absinthium* only sp. recorded here
(c) West and south of Meadowlands
(d) South and west Jersey
(e) Should be *pluchea odorata* due to an error in the type specimen i.d.

psilostachya - not reported from New Jersey; Aster dumosum - reported south and west of the Meadowlands area; Eupatorium serotinum - reported from south and west New Jersey; and Pluchea purpurescens - probably should be Pluchea odorata due to an error in the identification of the type specimen. A survey of the fall-blooming and fruiting plant species was conducted to assess any need for extending the focus of the ecological risk assessment to the upland areas of the site.

A more in-depth survey of the woody and herbaceous vegetation was undertaken in order to investigate differences in the terrestrial vegetation between the study area and the designated reference area. Results are summarized below.

A.3.2 Woody Vegetation

Woody vegetation (trees at least 1.5 m tall) in the upland areas were sampled at 10-m intervals along a 100-m transect by the point-centered quarter method (ENSR SOP No. 670-088) on 6 and 7 October, 1988. One transect was located in the study area and one in the designated reference area. Plants were identified to species according to Petrides (1973). The data were analyzed to determine density, dominance and frequency according to Mueller-Dombois and Ellenberg (1974). Because there were so few individuals in the community, some plants were sampled twice. These "repeat plants" were not included in the calculations of frequency, density, and dominance.

There were few species of trees at the two sites, and individual trees were separated by relatively large distances (Table A-2). All of the species of trees at both sites are colonizing species and are sufficiently small (mean dbh ± one standard deviation: study area = 4.79 cm ± 2.97; reference area = 13.7 cm ± 14.12) that they could easily have become established since the industrial activity ceased and the buildings were levelled.

TABLE A-2

WOODY VEGETATION SUMMARY STATISTICS

	Study Area	Reference Area
Average Distance Between Plants	15.0 m	11.1 m
Plants per Acre	18	33
Number of Species	2	4
	P. deltoides	*P. deltoides*
	A. altissima	*A. altissima*
		Morus sp.
		R. pseudoacacia

Cottonwood (Populus deltoides) was the most common species at both sites (Tables A-3 and A-4) (i.e. had the highest density and frequency). Some large individuals of Morus sp. caused that species to have the highest dominance value in the reference area (Table A-4), despite the fact that there were so few individuals.

The woody vegetation is sparse at both the study and reference sites. There are only a few plants, and they are apparently young and are widely spaced. The differences in the two communities can probably be explained by differences in dispersal to and colonization of microsites.

A.3.3 Herbaceous Vegetation

The herbaceous vegetation was sampled on 6 and 7 October, 1988 at 1-m intervals along a series of 25-m transects in the study and reference areas using the line-intercept method (ENSR SOP 670-087, contained in Ecological Risk Assessment Work Plan, February, 1989). Height and cover were measured for individuals directly beneath the transect line. "Bare ground" was considered to be equivalent to a species for the purposes of these analyses. Frequency, dominance, and density were calculated according to Mueller-Dombois and Ellenberg (1974). Graminoids were pooled for analyses with the exception of Phragmites communis. Reproductive status was measured according to a presence/absence (evidence of flowering or fruiting) scale.

Plant species distributed in both sites with sample sizes greater than one had a greater mean height in the reference area (with the exception of Aster dumosum which had a greater mean height in the study area) (Table A-5). There were more bare ground and paved surfaces in the study area than reference area transects (Table A-5). Plants were reproducing in both the study and reference areas, although reproduction was variable between species (Table A-5).

TABLE A-3

SUMMARY STATISTICS ON WOODY VEGETATION IN THE

UOP SITE STUDY AREA

	Relative Density (%)	Relative Dominance (%)	Relative Frequency (%)
Populus deltoides	92	80	80
Ailanthus altissima	8	20	20

TABLE A-4

SUMMARY STATISTICS ON WOODY VEGETATION IN THE REFERENCE AREA

	Relative Density (%)	Relative Dominance (%)	Relative Frequency (%)
Populus deltoides	78	33	62
Ailanthus altissima	7	3	13
Morus sp.	7	64	13
Robinia pseudoacacia	7	0	13

TABLE A-5

HERBACEOUS VEGETATION SUMMARY DATA

Species*	Mean Height	Mean Cover	% Repro.	Number
STUDY AREA				
Ambrosia artemisifolia	46.1±34.3	8.4±3.4	32	28
Artemisia sp.	8	8	0	1
Aster dumosum	51.5±25.5	22.4±11.1	100	7
Bare ground				37
Eupatorium serotinum	80.8±18.3	15.0±1.4	75	4
Graminoids*	22.0±22.8	16.7±15.4	61	28
Moss**				2
Phragmites communis	126.9±30.8	42.8±28.0	20	10
Pluchea purpurea	60	13	100	1
Solidago rugosa	70	14	100	1
Solidago sempervirens	55.7±38.4	14.8±4.1	50	6
REFERENCE AREA				
Achillea millefolium	63	20	100	1
Ambrosia artemisifolia	55.6±25.3	13.7±6.2	43	35
Aster dumosum	47.2±16.7	24.9±8.8	100	12
Bare ground				21
Centaurea sp.	39.5±4.9	11.5±2.1	100	2
Eupatorium serotinum	53	28	100	1
Graminoids*	27.4±4.6	16.8±5.0	23	13
Melilotus alba	46	20	0	1
Phragmites communis	136.6±42.4	21.1±8.3	64	11
Populus deltoides				1
Setaria sp.	53	33	100	1
Solidago sempervirens	87.8±18.5	20.4±8.3	89	9

*Graminoids = all "grasses" except *Setaria* and *Phragmites*.
**moss = unidentified moss.

A.3.4 Terrestrial Animal Survey Observations

A site walkover was conducted on October 6 and 7, 1988 during which general observations were made of terrestrial animals in the reference and study areas. Bird activities for these observation periods are listed in Table A-6. On October 6, observations of bird activity were made from 9:05 to 9:40 a.m. at separate sites overlooking Ackerman's Creek (study area) and a Phragmites wetland in the reference area. On October 7, 1988 bird activity was observed along the railroad track (while walking north) from 9:07 to 10:30 a.m. From 1:18 to 1:40 p.m. on October 7, 1988 bird activity was observed from the railroad trestle looking south over the study area. The species which were sighted during these observation periods are listed in Table A-7.

The following evidence of mammals was observed while conducting the vegetation survey of the study area:

- rabbit feces
- dead mouse/vole

The following evidence of animal activity was observed while conducting the vegetation survey at the reference area:

- dragonfly
- bees
- grasshopper
- butterflies (3 species)
- rabbit feces
- field mouse
- small burrow under a fence

TABLE A-6

BIRD ACTIVITY OBSERVED AT THE STUDY SITE AND REFERENCE AREA

Species	Common Name	Reference Area	Study Area
Spizella passerina	Chipping Sparrow	Perching	Flying
Corvus brachyrynchos	Common Crow	Flying	Flying
Branta canadensis	Canada Geese	Flying	
Sturnus vulgaris	Starling	Perching	Flying
	Gull	Flying	
	Duck	Flying	
Anas oscuta	Northern Pintail		Flying
Larus argentatus	Herring Gull		Flying
Anas platyrhynchos	Mallard		Flying

TABLE A-7

DAILY BIRD ACTIVITY OBSERVED AT THE STUDY SITE

Species	Common Name	Morning Observations	Afternoon Observations
Passer domesticus	House sparrow	Flying	
Melospiza melodia	Song sparrow	Feeding	Flying
	Duck	Flying	
Cyanocitta cristata	Blue Jay		Perching
Corvus brachyrynchos	Common Crow		Perching
Carpodacus mexicanus	House finch		Flying
	Gull		Flying

A.3.5 Conclusions of Terrestrial Survey

The survey of plants and animals in terrestrial areas showed no significant differences between on-site and reference areas. Any minor differences observed are easily attributed to remnants of development (foundation pads and paved areas) rather than to contamination. This conclusion is consistent with earlier perceptions that the stream channels were critical in terms of contamination, exposure routes and sensitive species.

A.4 Aquatic Survey

A.4.1 Zooplankton Survey

Zooplankton samples were collected by horizontal tows with a Wisconsin-style plankton tow net (mesh size = 64 µm) in the stream channels at each of the study area sites and at the reference site (Walden Swamp). The samples were preserved in order to retard degradation of the organisms. A qualitative analysis was performed in order to determine general trends in species diversity and community similarity between the study sites and reference site. As this analysis was qualitative and not quantitative in nature, the presence of an organism was recorded as being either rare (R), infrequent (I) or abundant (A) in each sample (Table A-8).

Analysis of the samples indicated the presence of several different species of adult copepods (cyclopoid and calanoid forms) and an even larger number of juvenile copepods (i.e. copepodites). Rotifers were slightly more abundant than copepods. Large numbers of invertebrate "resting" eggs were also abundant in most samples. Other organisms present included both nematodes and coleopterans. All of the organisms retrieved by the zooplankton tows are considered to be indigenous to the Hackensack Meadowlands area (Hackensack Meadowlands Development Commission, 1987). Identifications were according to Pennak (1983), Gosner (1978) and Smith (1964).

TABLE A-8

RELATIVE ABUNDANCE OF ZOOPLANKTON FROM UOP SITE STREAM CHANNELS

Site	Cyclops copepodid	Calanoid copepod	Miscellaneous copepod	Nauplius	Rotifers	Eggs	Nematode	Coleoptera (beetle)	Unknown
A1			R	A	A	A			
A2	R	R		R		A			
A3							R		
B2	R								
B3	R					A			R
B4				I				R	
C2						I			
C3									
DE1		R		I	A	I			
DE2	I			I		A			
DW1					A	A			
DW2					A	A			

Explanation:

R: rare

I: infrequent

A: abundant

Data from this qualitative analysis indicate no difference in the diversity nor community composition of micro-invertebrates between the study area and the reference area. Significant variability exists for both of these measures even within the reference area. If the replicates for each site are composited, both the numbers and types of organisms in the study areas are very similar to those of the reference area. Site C is an exception to this, but this difference may be explained by natural variability or the presence of anthropogenic (concrete and gravel) rather than naturally occuring sediment.

Although no significant qualitative differences are apparent between the microinvertebrates of the study and reference areas, this does not necessarily indicate that the organisms at the UOP site have not been impacted. Sediment contamination concentrations are also high off-site, and therefore there is some question as to whether the reference area is appropriate as a presumed non-impacted area.

A.4.2 Macro-Invertebrates

Representative sediment samples were collected with a ponar sampler from the study sites (A, B and C) and the reference site in Walden Swamp (D) (Figure A-1). The samples were hand picked for macro-invertebrates in order to determine species diversity of the study areas relative to that of the reference area. As only representative species were collected from the samples (actual numbers of each species were not counted) the ensuing analysis is qualitative in nature.

In order to determine the effect of xenotoxins at the UOP site on invertebrate organisms, species richness and relative presence (percent of subsites at which an organism appeared per site - see Tables A-9 and A-10) were analyzed. Area D, the reference area possessed the greatest species diversity and relative presence. The effects of toxins were manifested by a decrease in both of these indices in the study areas. Of the

TABLE A-9

BENTHIC MACROINVERTEBRATES OBSERVED FROM THE UOP SITE STREAM CHANNELS

Organism	A1	A3	B2	B3	B4	C1	C2	C3	DE1	DE2	DE3	DW1	DW2	DW3
Balanus eburneus	*					*	*	*	*	*		*	*	*
Hydrobia minuta		*	*	*	*				*	*				*
Gastropod-unknown 1									*			*		*
Gastropod-unknown 2												*		
Mytilopsis leucophaeta				*		*	*	*				*	*	*
Bivalve-unknown 1												*		
Bivalve-unknown 2													*	
Shrimp species								*			*			
Nematode-unknown	*		*								*			
Coleopteran-unknown					*									

*indicates presence of specified organism

DE 1-3 and DW 1-3 are controls

TABLE A-10

PRESENCE OF BENTHIC INVERTEBRATES: SITE-SPECIFIC REPLICATE PERCENTAGES[a]

	Site			
	A (%)	B (%)	C (%)	D (%)
Balanus eburneus	50	0	100	83
Hydrobia minuta	50	100	0	50
Gastropod-unknown 1	0	0	0	50
Gastropod unknown 2	0	0	0	17
Mytilopsis leucopheata	0	33	100	50
Bivalve-unknown 1	0	0	0	17
Bivalve-unknown 2	0	0	0	17
Shrimp species	0	0	33	17
Nematode-unknown	50	33	0	0
Coleopteran-unknown	0	33	0	0

[a] Replicates found of each organism are presented for four sites.

three study areas, C was the least negatively affected. Areas A and B were affected to a greater extent, with area A appearing to be slightly more stressed than B.

The degree for which each area was stressed (decreased diversity and relative presence) was most likely a function of chromium and/or PCB toxicity. Sediment concentrations of these two toxins were highest in area A, intermediate in area B and lowest in area C. These general trends appear to indicate an inverse relationship between both species diversity and numbers and one or both of the toxins (PCBs, chromium).

APPENDIX B
PRELIMINARY SURVEY DATA

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APPENDIX B

PRELIMINARY SURVEY DATA

Appendix B contains data from a preliminary survey conducted in October of 1988 at the UOP Site in East Rutherford, New Jersey. The data contained in this appendix include a list of indicator organisms chosen for the site and measurements of trees and herbs found at the site during the preliminary survey.

TABLE B-1

TREE MEASUREMENTS AT THE UOP SITE CONTROL AREA

POINT	TOP LEFT					TOP RIGHT					BOTTOM LEFT					BOTTOM RIGHT				
	Specie	Dist. (m)	Height (m)	DBH (m)	Basal Area (cm)	Specie	Dist. (m)	Height (m)	DBH (m)	Basal Area (cm)	Specie	Dist. (m)	Height (m)	DBH (m)	Basal Area (cm)	Specie	Dist. (m)	Height (m)	DBH (m)	Basal Area (cm)
1	Cot	6.0	9.0	20.6	333.1	Cot	8.1	4.8	5.9	27.3	Cot	47.4	3.9	4.7	17.3					
2	Cot	11.4	6.9	9.6	72.3	Cot	4.5	9.0	6.9	37.4	TL1	8.7	9.0	20.6	333.1	TR1	4.8	4.8	5.9	27.3
3	Cot	3.9	3.3	3.5	9.6	Cot	8.7	4.2	5.5	23.7	Cot	9.6	7.8	9.5	70.8	Cot	6.6	9.0	14.1	156.1
4	Cot	6.6	8.7	10.9	93.3	Mul	24.6	4.5	34.7	945.2	Cot	4.8	9.3	14.0	153.9	Cot	12.0	3.3	3.5	9.6
5	Cot	3.3	9.3	15.2	181.4	Ail	27.6	4.8	17.7	245.9	Cot	3.9	3.0	3.3	8.5	TR4	25.2	4.5	34.7	945.2
6	Cot	0.0	15.0	25.0	490.6	Mul	23.4	6.9	73.0	4183.3	Cot	9.9	9.3	14.6	167.3	TR5	29.4	4.8	17.7	245.9
7	Cot	7.8	9.6	31.7	788.8						TL1	13.5	9.0	20.5	329.9					
8	Cot	3.6	8.4	15.7	193.5	Ail	11.1	2.1	1.6	2.0	TL7	3.9	9.6	25.7	518.5					
9	Cot	15.9	3.0	4.3	14.5	TR8	3.3	2.1	1.6	2.0	Loc(w/t)	3.3	2.1	1.6	2.0					
10	Cot	11.1	2.4	2.0	3.1	Cot	5.4	1.5	0.6	0.3	Loc	9.6	2.4	1.5	1.8	TR9	8.1	2.1	1.6	2.0

TABLE B-2

TREE MEASUREMENTS AT THE UOP SITE STUDY AREA

POINT	TOP LEFT					TOP RIGHT					BOTTOM LEFT					BOTTOM RIGHT				
	Specie	Dist. (m)	Height (m)	DBH (cm)	Basal Area (cm)	Specie	Dist. (m)	Height (m)	DBH (cm)	Basal Area (cm)	Specie	Dist. (m)	Height (m)	DBH (cm)	Basal Area (cm)	Specie	Dist. (m)	Height (m)	DBH (cm)	Basal Area (cm)
1	Cot	9.3	3.6	5.7	25.8	Cot	9.3	2.1	1.6	2.0						Cot	13.8	1.2	-	-
2	TL1	3.6	3.6	5.7	25.8	Cot	8.1	3.6	5.6	24.4						TR1	6.9	2.1	1.6	2.0
3	Cot	38.7	5.4	7.6	45.9	Cot	11.1	1.8	1.0	0.7	Cot	3.6	3.6	3.7	10.5	TR2	6.3	3.6	5.6	24.4
4	Cot	26.1	3.6	2.5	5.1	Cot	12.0	7.5	8.0	49.8	Cot	12.0	3.6	3.8	11.5	TR3	12.0	1.8	1.0	0.7
5	TL4	16.5	3.6	2.5	5.1	Ail	16.5	2.1	7.3	42.1	BL4	21.3	3.6	3.8	11.5	TR4	117.3	7.5	8.0	49.8
6	TL4	8.1	3.6	2.5	5.1	TR5	9.9	2.1	7.3	42.1	BL4	29.7	3.6	3.8	11.5	TR4	18.0	7.5	8.0	49.8
7	Cot	3.0	9.0	14.6	168.5	Cot	5.1	2.1	1.9	2.9	TL6	2.1	3.6	2.5	5.1	TR6	11.1	2.1	7.3	42.1
8	Cot	1.5	1.5	0.6	0.3	Cot	6.0	2.4	2.9	6.4	Ail	8.1	2.1	10.2	81.5	Cot	3.9	2.1	1.9	2.9
9	Cot	15.3	5.4	5.1	20.4	Cot	5.1	5.4	5.7	25.8	Cot	6.3	1.5	0.3	0.1	Cot	4.5	2.4	4.8	17.9
10	Cot	27.3	5.1	4.8	17.9	Cot	33.6	7.5	7.3	42.1	TL9	16.5	5.4	5.1	20.4	Cot	9.9	5.4	5.7	25.8

TABLE B-3

HERB MEASUREMENTS AT THE UOP SITE

0.394

UOP - transects #1 - 7

TRANS.#	SITE	INTER.#	SPECIE	HEIGHT (in)	HEIGHT (cm)	COVER (in)	COVER (cm)	REPRO.
1	1	1	gra one		10			0
1	1	2	gra six		116			1
1	1	3	bar gro					
1	1	4	amb art		14			0
1	1	5	sol sem		55			1
1	1	6	amb art		33			0
1	1	7	amb art		93			1
1	1	8	bar gro					
1	1	9	bar gro					
1	1	10	bar gro					
1	1	11	amb art		115			1
1	1	12	bar gro					
1	1	13	bar gro					
1	1	14	bar gro					
1	1	15	bar gro					
1	1	16	amb art		93			1
1	1	17	bar gro					
1	1	18	gra one		47			0
1	1	19	gra one		15			1
1	1	20	gra one		20			1
1	1	21	bar gro					
1	1	22	gra one		28			1
1	1	23	amb art		16			0
1	1	24	amb art		38			0
1	1	25	bar gro					
2	1	1	phr com		110		30	0
2	1	2	phr com		120		55	0
2	1	3	bar gro					

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0.394

UOP - transects #1 - 7

TRANS.#	SITE	INTER.#	SPECIE	HEIGHT (in)	HEIGHT (cm)	COVER (in)	COVER (cm)	REPRO.
2	1	4	amb art		5		5	0
2	1	5	amb art		70		12	1
2	1	6	ast dum		100		22	1
2	1	7	amb art		70		10	1
2	1	8	amb art		25		11	0
2	1	9	amb art		90		16	1
2	1	10	amb art		58		5	0
2	1	11	bar gro					
2	1	12	bar gro					
2	1	13	amb art		115		12	1
2	1	14	bar gro					
2	1	15	amb art		82		12	1
2	1	16	amb art		10		5	0
2	1	17	amb art		65		7	0
2	1	18	bar gro					
2	1	19	art sp		8		8	0
2	1	20	ast dum		40		30	1
2	1	21	bar gro					
2	1	22	gra two		25		7	1
2	1	23	gra two		1		17	0
2	1	24	gra thr		2		40	1
2	1	25	gra thr		2		27	0
3	1	1	amb art		15		6	0
3	1	2	gra two		24		5	1
3	1	3	gra two		19		6	1
3	1	4	gra two		22		11	1
3	1	5	gra two		25		50	1
3	1	6	gra two		35		35	1

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0.394

UOP - transects #1 - 7

TRANS.#	SITE	INTER.#	SPECIE	HEIGHT (in)	HEIGHT (cm)	COVER (in)	COVER (cm)	REPRO.
3	1	7	gra two		25		30	1
3	1	8	ast dum		24		11	1
3	1	9	bar gro					
3	1	10	bar gro					
3	1	11	bar gro					
3	1	12	bar gro					
3	1	13	amb art		52		12	0
3	1	14	bar gro					
3	1	15	gra two		15		32	1
3	1	16	gra two		10		14	1
3	1	17	bar gro					
3	1	18	moss					
3	1	19	bar gro					
3	1	20	gra fou		20		10	1
3	1	21	ast dum		70		32	1
3	1	22	ast dum		42		11	1
3	1	23	ast dum		40		13	1
3	1	24	eup ser		58		16	0
3	1	25	ast dum		42		38	1
4	1	1	sol rug		70		14	1
4	1	2	amb art		8		4	0
4	1	3	amb art		23		4	0
4	1	4	sol sem		77		20	1
4	1	5	amb art		33		7	0
4	1	6	amb art		22		6	0
4	1	7	amb art		15		9	0
4	1	8	bar gro					
4	1	9	bar gro					

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0.394

UOP - transects #1 - 7

TRANS.#	SITE	INTER.#	SPECIE	HEIGHT (in)	HEIGHT (cm)	COVER (in)	COVER (cm)	REPRO.
4	1	10	gra fiv		15		3	0
4	1	11	gra fiv		1		0	0
4	1	12	bar gro					
4	1	13	bar gro					
4	1	14	gra fiv		25		7	1
4	1	15	gra fiv		7		4	0
4	1	16	sol sem		24		18	0
4	1	17	gra six		59		45	0
4	1	18	sol sem		16		13	0
4	1	19	gra fiv		7		2	0
4	1	20	gra fiv		20		5	0
4	1	21	amb art		29		10	0
4	1	22	moss					
4	1	23	amb art		12		5	1
4	1	24	amb art		21		7	0
4	1	25	gra fiv		2		5	0
5	1	1	gra sev		20		10	1
5	1	2	bar gro					
5	1	3	plu pur		60		13	1
5	1	4	sol sem		120		13	1
5	1	5	phr com		150		35	1
5	1	6	phr com		110		50	0
5	1	7	phr com		140		30	0
5	1	8	phr com		94		20	0
5	1	9	bar gro					
5	1	10	bar gro					
5	1	11	eup ser		100		16	1
5	1	12	bar gro					

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0.394

UOP - transects #1 - 7

TRANS.#	SITE	INTER.#	SPECIE	HEIGHT (in)	HEIGHT (cm)	COVER (in)	COVER (cm)	REPRO.
5	1	13	phr com		80		116	0
5	1	14	bar gro					
5	1	15	amb art		70		12	0
5	1	16	phr com		125		40	0
5	1	17	sol sem		42		10	0
5	1	18	bar gro					
5	1	19	bar gro					
5	1	20	eup ser		90		13	1
5	1	21	phr com		180		27	1
5	1	22	phr com		160		25	0
5	1	23	eup ser		75		15	1
5	1	24	bar gro					
5	1	25	bar gro					
6	2	1	sol sem	26	66	7	18	1
6	2	2	amb art	17	43	5	13	0
6	2	3	amb art	9	23	4	10	0
6	2	4	amb art	21	53	6	15	0
6	2	5	amb art	13	33	5	13	0
6	2	6	amb art	16	41	6	15	0
6	2	7	amb art	23	58	5	13	1
6	2	8	amb art	10	25	4	10	0
6	2	9	amb art	12	30	4	10	0
6	2	10	amb art	10	25	3	8	0
6	2	11	bar gro		0		0	
6	2	12	bar gro		0		0	
6	2	13	bar gro		0		0	
6	2	14	bar gro		0		0	
6	2	15	bar gro		0		0	

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0.394

UOP - transects #1 - 7

TRANS.#	SITE	INTER.#	SPECIE	HEIGHT (in)	HEIGHT (cm)	COVER (in)	COVER (cm)	REPRO.
6	2	16	amb art	18	46	5	13	0
6	2	17	amb art	39	99	4	10	1
6	2	18	bar gro		0		0	
6	2	19	amb art	13	33	4	10	0
6	2	20	amb art	15	38	3	8	0
6	2	21	gra A	7	18	9	23	0
6	2	22	amb art	32	81	4	10	1
6	2	23	amb art	8	20	4	10	0
6	2	24	amb art	19	48	4	10	0
6	2	25	amb art	28	71	6	15	1
6	2	26	amb art	17	43	5	13	0
6	2	27	bar gro		0		0	
6	2	28	bar gro		0		0	
7	2	1	phr com	78	198	10	25	1
7	2	2	gra B	10	25	5	13	0
7	2	3	ach mil	25	63	8	20	1
7	2	4	gra B	11	28	6	15	0
7	2	5	gra B	12	30	6	15	0
7	2	6	gra B	12	30	6	15	0
7	2	7	gra B	11	28	6	15	0
7	2	8	gra B	11	28	5	13	0
7	2	9	sol sem	33	84	6	15	0
7	2	10	gra B	11	28	5	13	0
7	2	11	amb art	37	94	16	41	1
7	2	12	gra A	9	23	5	13	0
7	2	13	sol sem	35	89	6	15	1
7	2	14	sol sem	50	127	6	15	1
7	2	15	gra A	10	25	5	13	0

TABLE B-3 (Page 7 of 9)

0.394

UOP - transects #1 - 7

TRANS.#	SITE	INTER.#	SPECIE	HEIGHT (in)	HEIGHT (cm)	COVER (in)	COVER (cm)	REPRO.
7	2	16	bar gro		0		0	
7	2	17	ast dum	9	23	6	15	1
7	2	18	phr com	72	183	16	41	1
7	2	19	amb art	20	51	5	13	0
7	2	20	amb art	26	66	8	20	1
7	2	21	amb art	27	69	12	30	1
7	2	22	phr com	42	107	6	15	0
7	2	23	bar gro		0		0	
7	2	24	pop del		0		0	
7	2	25	phr com	51	129	8	20	1
7	2	26	phr com	44	112	6	15	0
7	2	27	phr com	68	173	8	20	1
7	2	28	phr com	59	150	6	15	0
8	2	1	sol sem	34	86	5	13	1
8	2	2	amb art	16	41	4	10	0
8	2	3	set sp	21	53	13	33	1
8	2	4	amb art	28	71	6	15	1
8	2	5	gra C	10	25	8	20	1
8	2	6	gra C	15	38	11	28	1
8	2	7	bar gro		0		0	
8	2	8	ast dum	26	66	14	36	1
8	2	9	gra C	12	30	9	23	1
8	2	10	ast dum	23	58	15	38	1
8	2	11	mel alb	18	46	8	20	0
8	2	12	bar gro		0		0	
8	2	13	ast dum	16	41	6	15	1
8	2	14	phr com	51	129	5	13	1
8	2	15	bar gro		0		0	

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0.394

UOP - transects #1 - 7

TRANS.#	SITE	INTER.#	SPECIE	HEIGHT (in)	HEIGHT (cm)	COVER (in)	COVER (cm)	REPRO.
8	2	16	bar gro		0		0	
8	2	17	bar gro		0		0	
8	2	18	ast dum	17	43	13	33	1
8	2	19	phr com	38	96	6	15	0
8	2	20	phr com	69	175	11	28	1
8	2	21	phr com	56	142	10	25	1
8	2	22	ast dum	18	46	6	15	1
8	2	23	ast dum	22	56	9	23	1
8	2	24	ast dum	20	51	10	25	1
8	2	25	ast dum	23	58	12	30	1
8	2	26	ast dum	29	74	13	33	1
9	2	1	sol sem	26	66	8	20	1
9	2	2	amb art	16	41	5	13	0
9	2	3	amb art	12	30	4	10	0
9	2	4	bar gro		0		0	
9	2	5	ast dum	12	30	7	18	1
9	2	6	sol sem	39	99	8	20	1
9	2	7	amb art	31	79	5	13	1
9	2	8	bar gro		0		0	
9	2	9	bar gro		0		0	
9	2	10	amb art	26	66	5	13	1
9	2	11	amb art	15	38	4	10	0
9	2	12	bar gro		0		0	
9	2	13	ast dum	8	20	7	18	1
9	2	14	sol sem	31	79	15	38	1
9	2	15	sol sem	37	94	12	30	1
9	2	16	amb art	27	69	6	15	1
9	2	17	amb art	33	84	6	15	1

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0.394

UOP - transects #1 - 7

TRANS.#	SITE	INTER.#	SPECIE	HEIGHT (in)	HEIGHT (cm)	COVER (in)	COVER (cm)	REPRO.
9	2	18	eup ser	21	53	11	28	1
9	2	19	amb art	16	41	4	10	0
9	2	20	bar gro		0		0	
9	2	21	bar gro		0		0	
9	2	22	amb art	27	69	6	15	1
9	2	23	cen sp	17	43	4	10	1
9	2	24	cen sp	14	36	5	13	1
9	2	25	amb art	42	107	5	13	1
9	2	26	amb art	47	119	7	18	1

APPENDIX C
FOCUSED ENVIRONMENTAL ASSESSMENT

APPENDIX C

FOCUSED ENVIRONMENTAL ASSESSMENT

C.1 Introduction

Based upon the recommendation of the NJDEP, Fundulus heteroclitus was chosen as an indicator species for a site specific study of uptake of contaminants. Body burdens of Hg, Cr, and PCBs were measured in F. Heteroclitus collected at sites in which contaminant levels in the underlying sediments were also measured. In addition, fish were trapped during the Preliminary Site Visit and marked. Fish caught during the Focused Environmental Site visit were inspected for markings for use in a mark-recapture study. Finally the fish in each sample were measured (for length) and weighed to determine their condition factors. The raw data for the various physical and chemical analyses are contained in Appendix D.

C.2 Methods

C.2.1 Fundulus Population Survey (10/6/88 - 10/7/88)

Fundulus spp. were collected as part of a mark-recapture study during the preliminary site assessment in order to estimate the population size and range. Fundulus from each of the three study areas (A, B, and C, Figure A-1) described in the work plan (Section 3.2) were collected on 6 October, 1988 using minnow traps baited with cat food and bread. Fish from each area were tagged by fin clipping and returned to the sample area from which they were captured. Fish from each of the study areas were tagged as follows: Site A - left pectoral fin; Site B - right pectoral fin; Site C - caudal fin. The site-specific tagging of the fish was used to estimate the homerange of Fundulus in this tidal environment. The total number of fish tagged in the pre-census sampling effort was 1,448 (Site A = 520, Site B = 336, Site C = 592).

The recapture sampling effort of November 1988 yielded data on the total number of fish collected, number of tagged fish collected and the tag location (left pectoral, right pectoral or caudal fin) length and weight. Formulae for estimating a population size for Fundulus spp. are:

$$N = \frac{nT}{t}$$

$$S.E. = N^* \frac{(N-T)\ (N-n)}{Tn(N-1)}$$

where N = population estimate, T = number of fish marked in precensus, t = number of marked fish in recapture, n = total number of fish trapped in recapture, and S.E. = standard error (Smith, 1980). This method assumes that there is a homogeneous distribution and a closed population.

C.2.2 Focused Environmental Assessment - Sample Collection (11/3/88 - 11/4/88)

In order to complete the mark-recapture study for population estimates and to measure PCB concentrations in the sediments and fish tissues, a second sampling effort was required. Two minnow traps per site (A, B, C and D) were baited with bread and cat food at each of the twelve locations designated by the workplan (sample locations - Section 3.2). The traps were set at low tide and left for most of one tidal cycle (approximately 8 hours). The captured fish were kept in separate buckets by duplicate and site. Each of these buckets was filled with tap water which was held for greater than 12 hours prior to holding fish to vaporize the chlorine and prefitted with suspended 1/4" wire mesh for the purpose of facilitating gut clearance. The mesh barrier isolated the fish from sediments, detritus and feces. Gut clearance was allowed to take place for the majority of the samples for 18 to 22 hours. The samples from one test area (D) were held for

27 hours. (For the remainder of the discussion these duplicates and sites will be referred to as: Aa, Ab, Ac, Ba, Bb, Bc, Ca, Cb, Cc, Da, Db, Dc.) One fish sample (Bc) was split in order to create a duplicate sample (Ea) for quality assurance purposes in the laboratory analysis. After gut clearance, fish were sorted into two size classes (large/small) determined by visual inspection. A best attempt was made to isolate F. heteroclitus from the fish samples. However, rapid identification of mummichogs is difficult under field conditions due to their small size. Approximately 250 grams of fish from each size class were stored on ice for transport to the lab for determination of tissue concentrations of mercury, chromium, and PCBs. All fish were weighed to the nearest tenth of a gram with a pesola scale and their length was measured to the nearest millimeter.

The measurements were used to calculate Fulton's condition factor (K):

$$K = \frac{W * 10^5}{L^3}$$

where W is the weight of the fish in grams and L is the length of the fish in millimeters (Carlander, 1977). This condition coefficient is a measure of the well-being of a fish (Carlander, 1977) and can also be used to standardize fish data.

To determine the weight of the mummichog, we used a pesola scale. Given the inherent precision of the scale, and in view of the difficulty measuring the weights of thousands of fish rapidly in the field, we believe the weights reported in Table D-5 and within ±10 percent of the reported value. Similarly, we believe the lengths, measured rapidly in the field, are within ±10 percent. Thus, using the standard error propagation formulae, we believe that the overall precision of the condition factor (K) is ±40 percent. Because the condition

factors are not used to buttress any arguments or conclusions in the risk assessment, the overall statistics of K will have no impact on the conclusions of this assessment.

All fish were examined for fin clips. Those fish not collected for lab analyses from each of the sites were checked for fin clips. Some of these fish were subsequently returned to the stream channels. Measurements of fish in all samples could not be completed on site because of the short daylength. Those fish were preserved with alcohol and transported to the lab, where they were examined for fin clips.

Sediment samples were collected at each of the 12 sites where fish were collected and stored on ice for transport to the lab. See the work plan (Section 3.1) for a detailed description of sediment sampling techniques and decontamination procedures of sediment sampling materials.

The fish and sediment samples were delivered to Cambridge Analytical Associates for analysis of the following parameters:

Fish - Analytical Parameters

- composite mercury concentration
- composite chromium concentration
- composite PCB concentration (Aroclor 1248)
- composite lipid content
- composite dry weight

Sediment - Analytical Parameters

- mercury concentration
- chromium concentration
- PCB concentration (Aroclor 1248)
- pH
- grain size distribution
- total organic carbon

(Note: The analytical laboratory was responsible for observing the required extraction and analytical holding time specified in the current CLP statement of work.)

Samples collected for the purpose of quality assurance (replicates and duplicates) were indistinguishable from the other samples sent to the analytical lab. Additional details concerning analytical protocols are described in the work plan (Appendix A). In addition, the laboratory methodologies employed by Cambridge Analytical Associates are described in the reports submitted to ENSR (Cambridge Analytical Associates, 1989a, 1988b, 1988c, 1988d).

C.3 Results

C.3.1 Fish Population Estimates

Fish recapture rates were calculated from the data in Tables C-1 and C-2 and were used to derive population estimates. A population estimate (N) of 57,246 with a 95 percent confidence interval of 41,256 to 71,788 was determined for the Fundulus at the UOP site (composite of areas A, B and C) using the methods of Smith (1980).

C.3.2 Fish Homerange

The fin clip information recorded in the mark-recapture study was used to estimate potential home ranges for the Fundulus located at the UOP site. Homerange as defined by Smith (1980) is the area over which an animal ranges throughout the year. Defining this range will make possible an estimation of how much time the fish might spend on site. Determination of exposure time will facilitate assigning body burden concentrations attributable to the toxins at the UOP site.

Data analysis suggests a small homerange for Fundulus spp. at this site in the autumn because of a high number of

TABLE C-1

UOP: MARK-RECAPTURE STUDY

PHASE II (RECAPTURE) 11/4-7/88

Site	Fish with Clips	Total Fish in Sample
Aa	6	83
Ab	1	254
Ac	13	251
Ba	5	118
Bb	8	169
Bc	5	277
Ea*	3	61
Ca	6	280
Cb	3	420
Cc	8	380
Da	17	499
Db	9	477
Dc	5	357
Total	89	3626

1448 fish were marked in Phase I (precensus)

*Duplicate of Bc

TABLE C-2

UOP: MARK-RECAPTURE STUDY

Site	Number Marked	Recapture On-Site	Recapture Off-Site	Breakdown of Recaptures by Site Origin
A	520	3	17	3A 2B 15C
B	336	2	19	1A 2B 18C
C	592	13	4	4B
D	0	-	31	2A 4B 25C
Total Marked	1448			

recaptured fish in the mark-recapture study. This homerange did, however, include all of the test subsites and the reference area. The exact homerange of Fundulus on the UOP site is unknown. However, it was demonstrated that more than two dozen mummichogs ranged over 1,000 meters to the reference site across Berry's Creek, depending on the individual fish and on the tidal hydraulics of the channels and Berry's Creek. The field experiments show that mummichogs move with the tides. Further, in careful observations, no mummichogs were seen to burrow into the sediment on the falling tide.

C.3.3 Fish Condition Factors

Condition factors of fish were analyzed by site and size class (Table C-3). The condition factors for the larger size class are slightly higher than those of the small size class, indicating that they are slightly heavier for their length than the small fish.

Body burdens of all three contaminants in fish by site are depicted in Figures C-1 and C-2. PCBs comprise the largest body burden of the three contaminants. There was appreciable variation in PCB body burdens within a site (Figures C-3 and C-4).

The data graphed in Figures C-1 through C-4 is presented in Tables D-1 and D-2 of Appendix D.

C.3.4 Sediments

Sediment concentrations of the contaminants (Figures C-5 to C-7) showed a high degree of within-site variability (Figures C-8 to C-11). Sediment concentrations of PCBs and chromium were highest on the site, while mercury concentrations were highest in the reference area (samples Da - Dc).

Comparisons of fish to sediment concentrations for each size class (Figures C-12 to C-17) indicated a poor relationship

TABLE C-3

CONDITION FACTORS (K) FOR *FUNDULUS* SPP.

BY SITE

Site	Size Class	Average K
A	Small	1.2
B	Small	1.3
C	Small	1.4
D	Small	1.2
A	Large	1.5
B	Large	1.6
C	Large	1.6
D	Large	1.5

Note: Data from sample Ea (Large and Small) is included in sample B

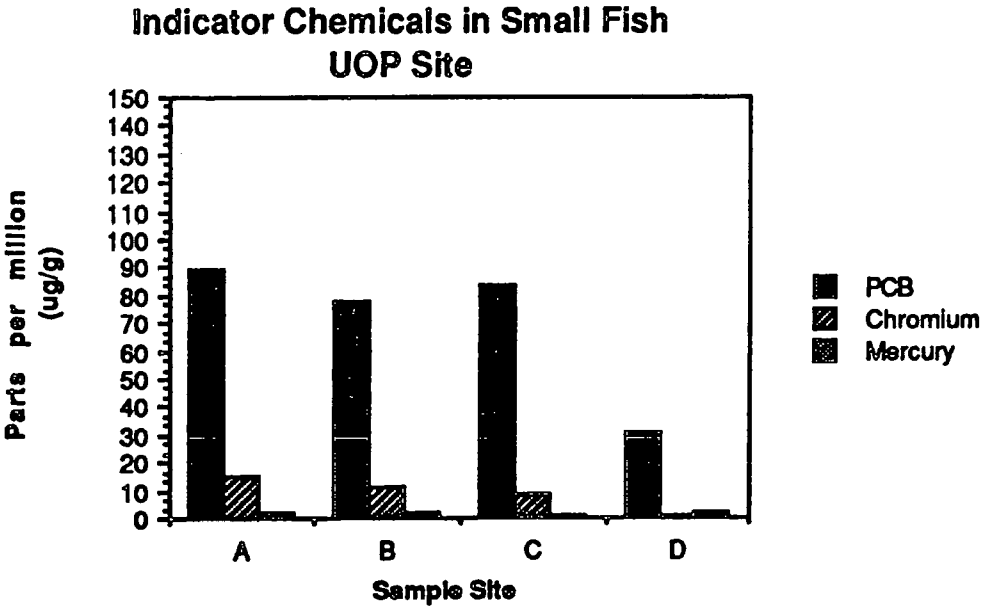

Indicator Chemicals in Small Fish
UOP Site
Parts per million (ug/g)
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
A
B
C
D
Sample Site
PCB
Chromium
Mercury

Figure C-1

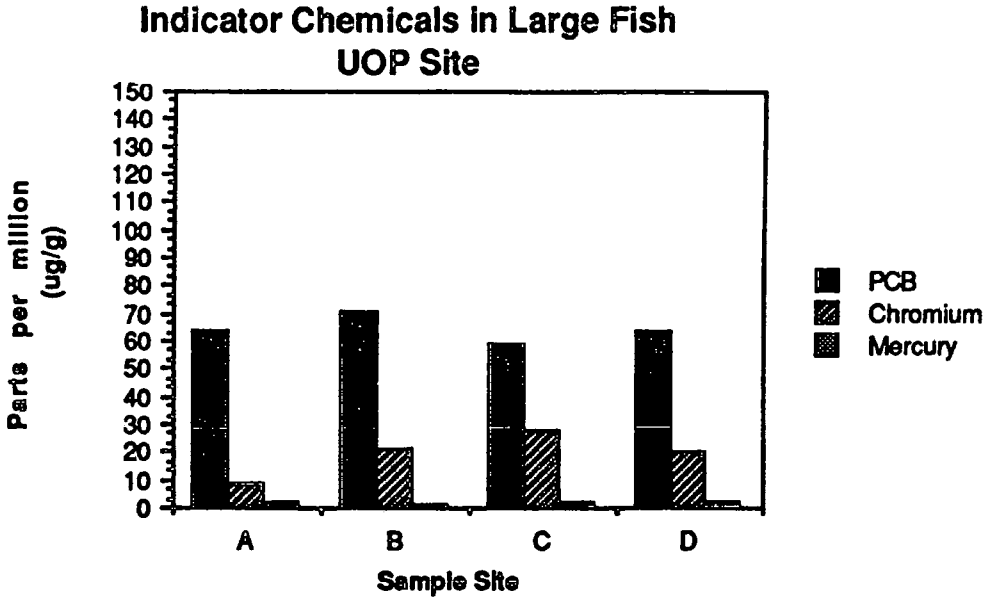

Indicator Chemicals in Large Fish
UOP Site
Parts per million (ug/g)
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
A
B
C
D
Sample Site
PCB
Chromium
Mercury

Figure C-2

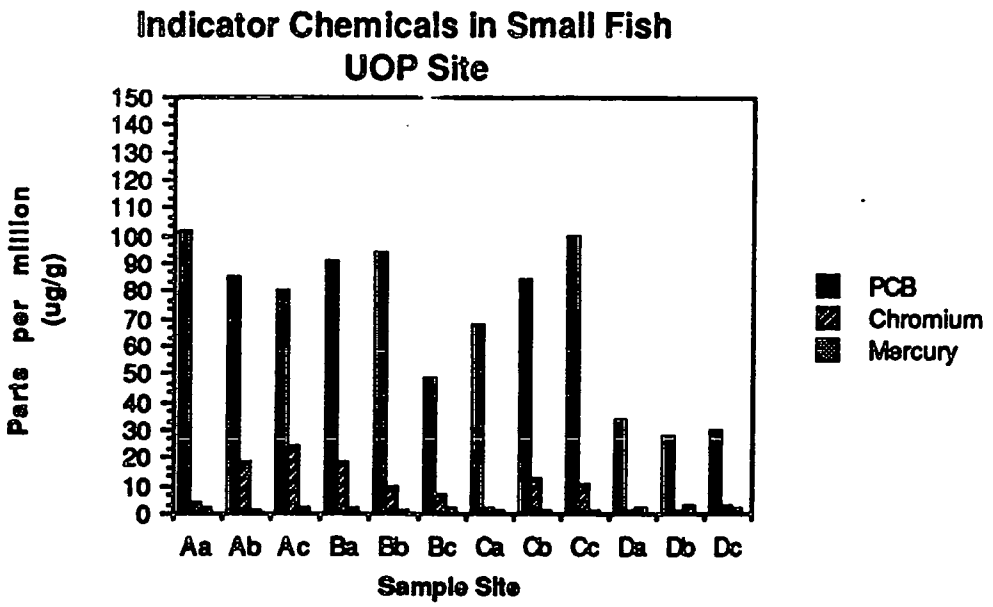

Indicator Chemicals in Small Fish
UOP Site
Parts per million (ug/g)
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
Aa Ab Ac Ba Bb Bc Ca Cb Cc Da Db Dc
Sample Site
PCB
Chromium
Mercury

Figure C-3

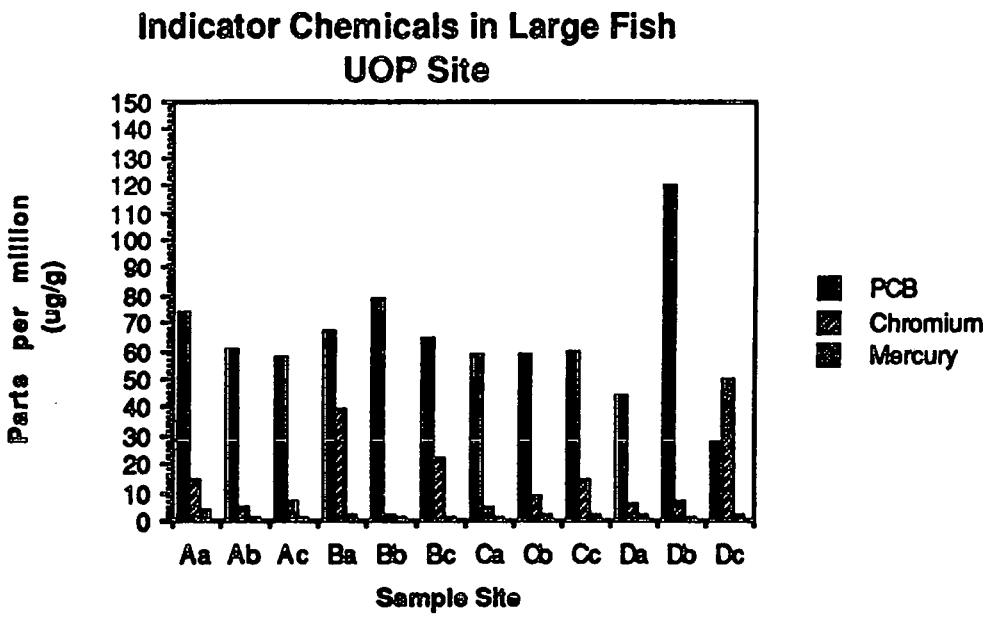

Indicator Chemicals in Large Fish
UOP Site
Parts per million (ug/g)
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
Aa Ab Ac Ba Bb Bc Ca Cb Cc Da Db Dc
Sample Site
PCB
Chromium
Mercury

Figure C-4

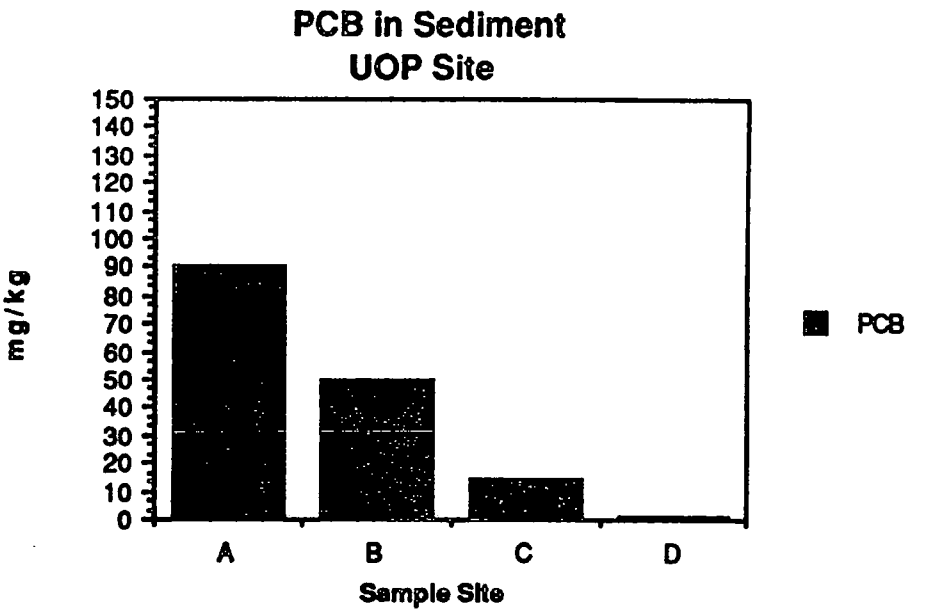

PCB in Sediment
UOP Site
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
mg/kg
A
B
C
D
Sample Site
PCB

Figure C-5

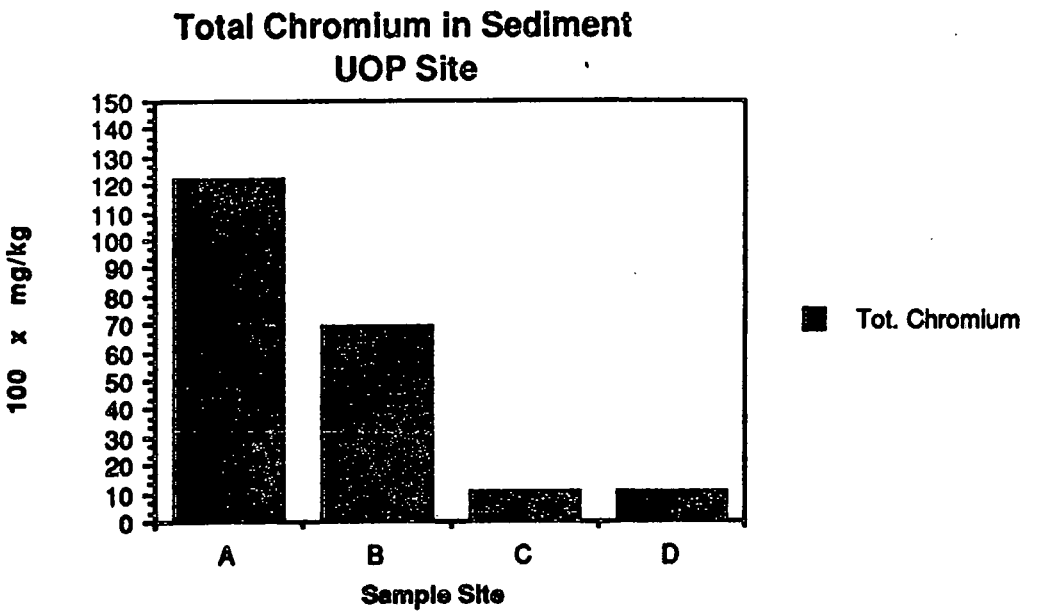

Total Chromium in Sediment
UOP Site
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
100 x mg/kg
A
B
C
D
Sample Site
Tot. Chromium

Figure C-6

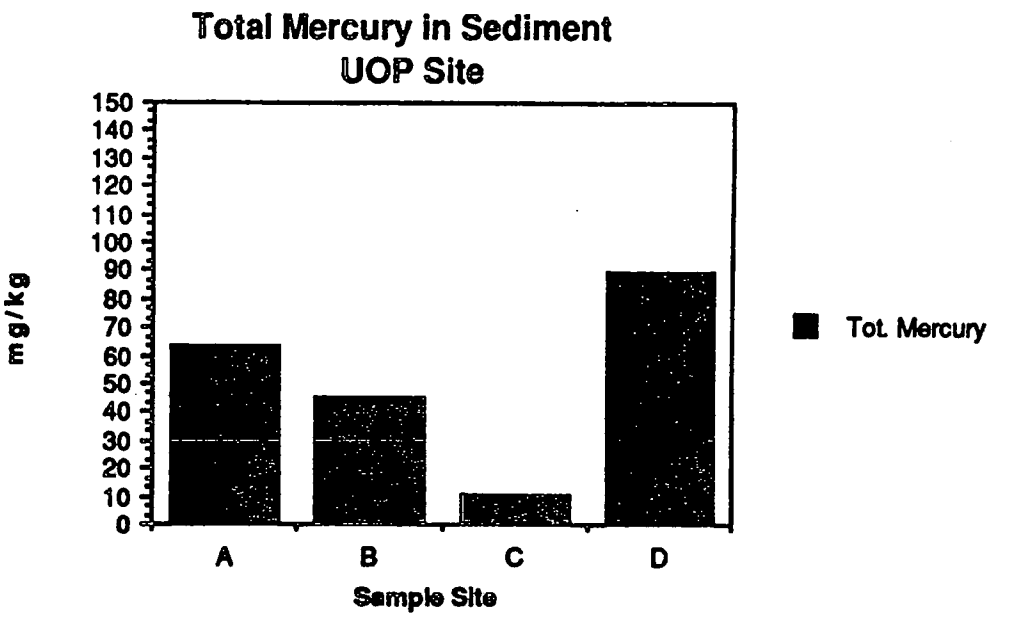

Total Mercury in Sediment
UOP Site
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
mg/kg
A
B
C
D
Sample Site
Tot. Mercury

Figure C-7

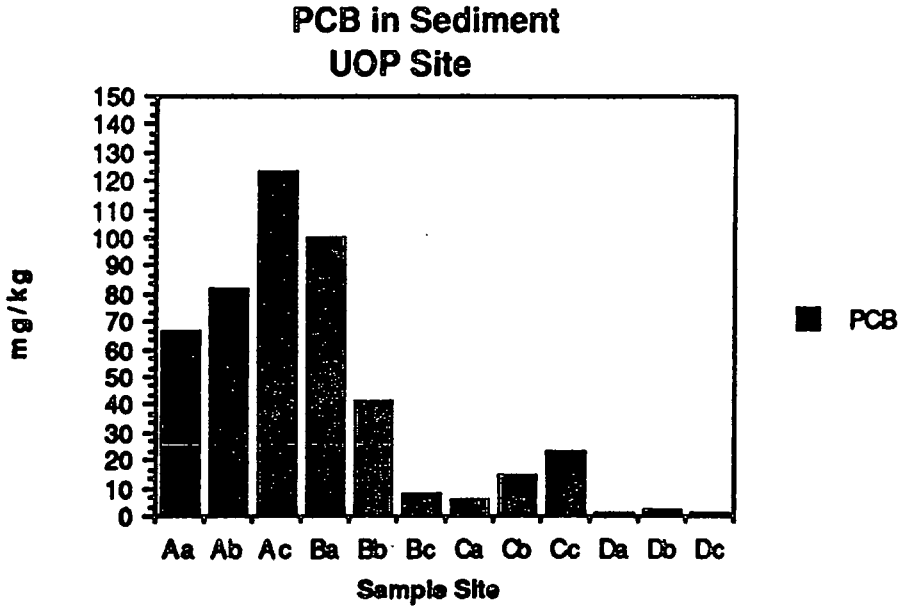

PCB in Sediment
UOP Site
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
mg/kg
Aa Ab Ac Ba Bb Bc Ca Cb Cc Da Db Dc
Sample Site
PCB

Figure C-8

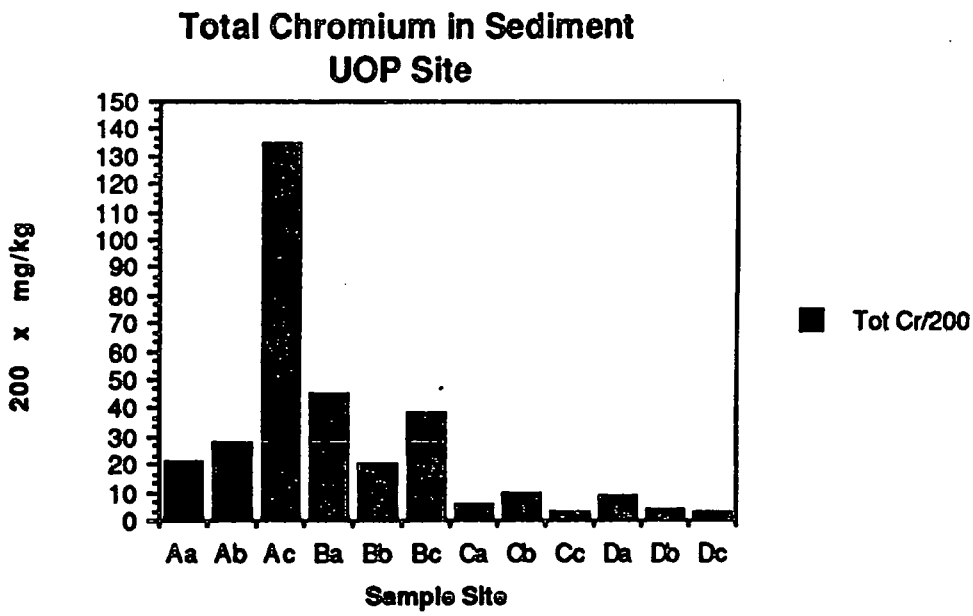

Total Chromium in Sediment
UOP Site
200 x mg/kg
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
Aa Ab Ac Ba Bb Bc Ca Cb Cc Da Db Dc
Sample Site
Tot Cr/200

Figure C-9

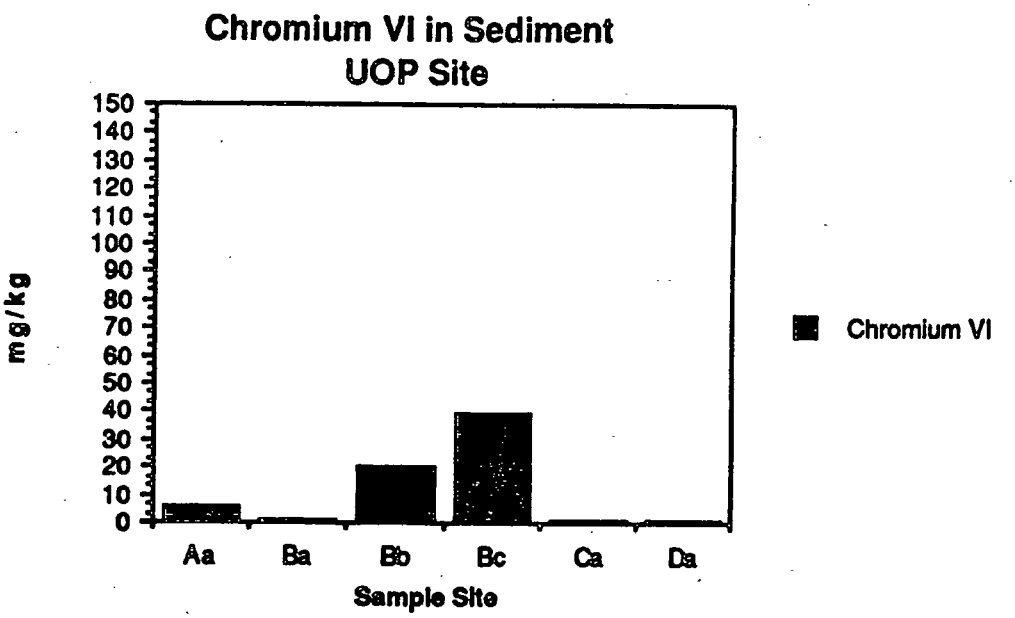

Chromium VI in Sediment
UOP Site
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
mg/kg
Aa
Ba
Bb
Bc
Ca
Da
Sample Site
Chromium VI

Figure C-10

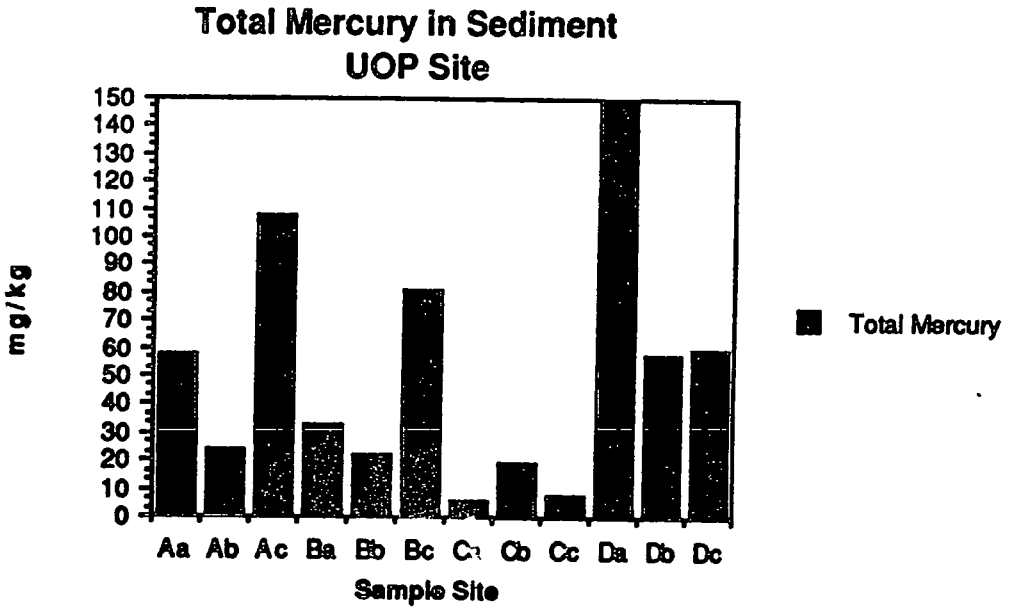
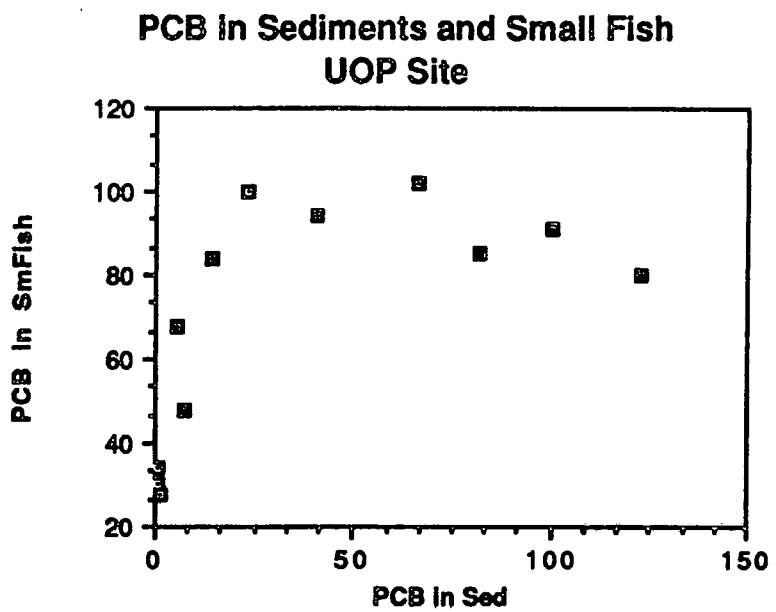

Total Mercury in Sediment
UOP Site
mg/kg
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
Aa Ab Ac Ba Bb Bc Ca Cb Cc Da Db Dc
Sample Site
Total Mercury

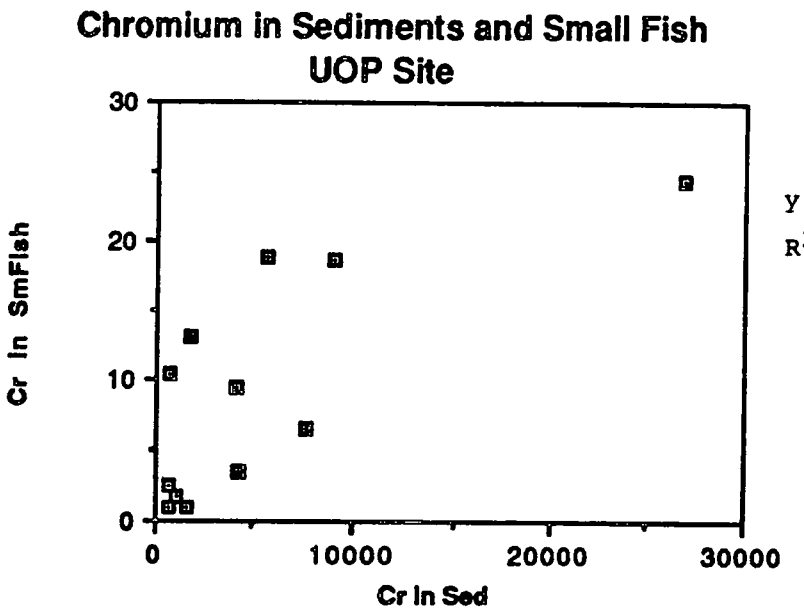
Figure C-11



$y = 37.791+30.031 * Log (X)$

$R^2 = 0.793$

Figure C-12



$y = 3.51+1.37e-3X - 2.20e-8X^2$

$R^2 = 0.53$

Figure C-13

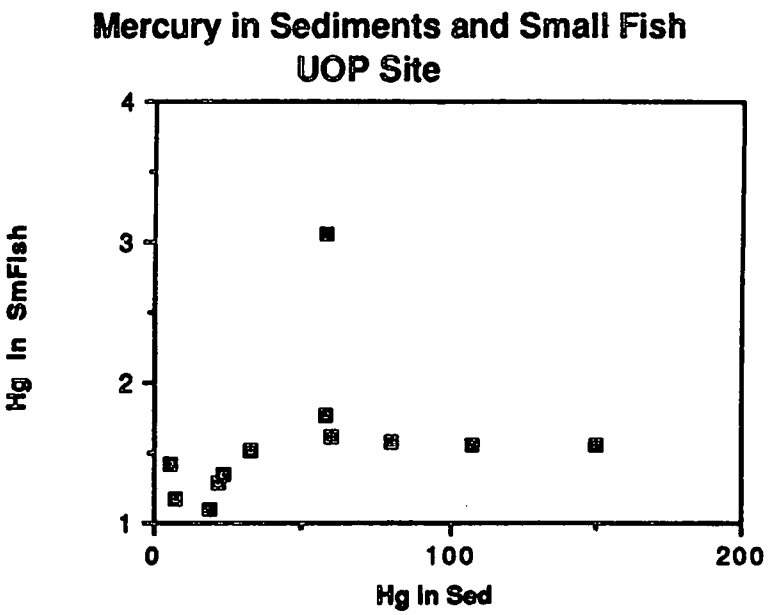

Mercury in Sediments and Small Fish
UOP Site
Hg In SmFish
4
3
2
1
0
100
200
Hg In Sed

Figure C-14

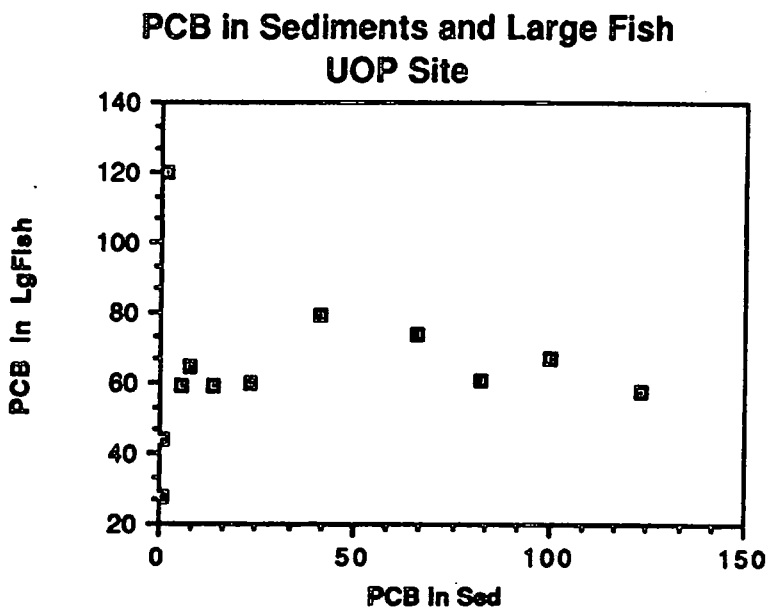

PCB in Sediments and Large Fish
UOP Site
140
120
100
80
60
40
20
PCB In LgFish
0
50
100
150
PCB In Sed

Figure C-15

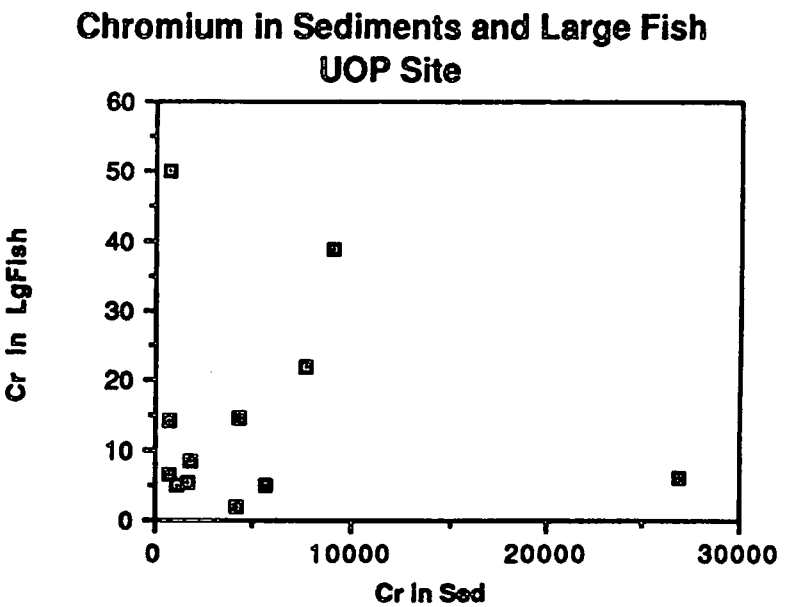

Chromium in Sediments and Large Fish
UOP Site
Cr in LgFish
60
50
40
30
20
10
0
0
10000
20000
30000
Cr in Sed

Figure C-16

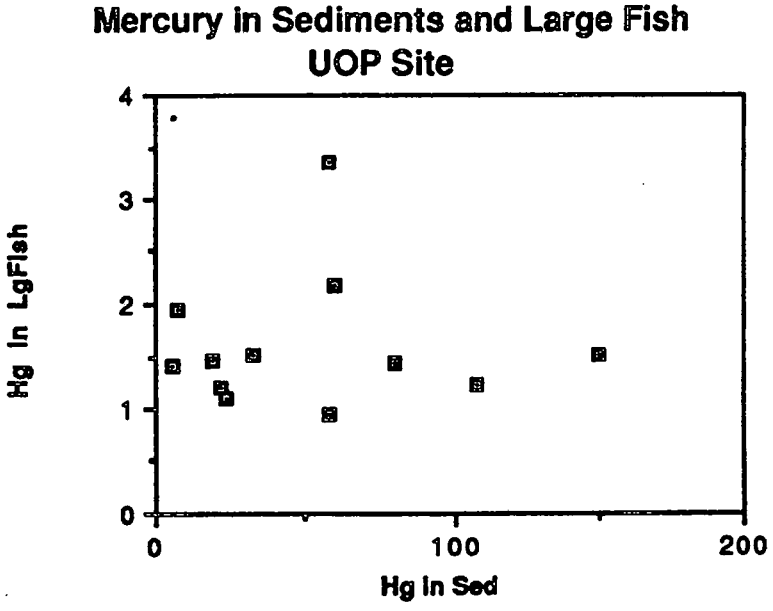

Mercury in Sediments and Large Fish
UOP Site
Hg in LgFish
Hg in Sed
0
1
2
3
4
0
100
200

Figure C-17

between the two for mercury and chromium, but a positive relationship for PCBs, especially in small fish (Figure C-12).

Grain size distribution results are shown in Figures D-1 through D-13. These graphs indicate that the majority of the sediment falls in the fine sand (0.1 to 0.5mm diameter) and, silt and clay (<0.1mm diameter) size classifications.

The data graphed in Figures C-5 through C-11 is presented in Table D-6 of Appendix D. The data graphed in Figures C-12 through C-17 and C-20 through C-22 is a compilation of the data presented in Tables D-1, D-2 and D-6 of Appendix D.

C.4 Discussion

The data from the mark-recapture study reveal movement patterns of Fundulus spp. similar to those reported at other sites by Field (1960), Butner and Brattstrom (1960), Darnell (1964) and Jeffries (1972). Fundulus spp. make use of the flood tide, moving from the main channel (Berry's Creek) into previously inaccessible drainage channels presumably to feed upon fauna and detritus. During the highest tide levels, Fundulus spp. have been observed to infiltrate marsh surface (Field, 1960; Darnell, 1964; Jeffries, 1972; Valiela et al., 1977). Butner and Brattstrom (1960) observed that upon the ebb of the tide, Fundulus would evacuate the channels and or marsh surface, returning to a narrow range in the main channel.

The recapture rate indicates a homerange of 3500 m. The overwintering burying behavior of Fundulus in soft sediments (Fritz et al., 1975) was not observed during the collection periods.

Fritz et al. (1975) determined a summer homerange of 36 m for Fundulus heteroclitus in a tidal creek in Delaware. This homerange is significantly smaller than that which we found at the UOP site. This could be due to polytypic behaviors stemming from temperature or seasonal cues or to varying definitions of the term homerange. The homerange indicated in Lotrich's study may refer to the narrow range in the main

channel to which the fish return at low tide, while in this study it refers to the longest distance traveled by individaul fish: site Aa to the reference area D (Walden Swamp).

The possibility exists that seasonal ranges may expand and decrease depending upon external cues. Seasonal migrations of Fundulus spp. have been observed to occur. Fritz et al. (1975) hypothesized photoperiod (prime initiator) and temperature (reinforcer) to be the cues initiating the move. Figure C-18 presents general trends in the seasonal migrations of Fundulus spp. as reported in the literature.

Meredith (1974) reported Fundulus populations in saline waters from late spring to early September. With the onset of autumn, both studies reported migration of fish from this area; Meredith (1974) determined that the fish move towards lower salinity waters while Lotrich (in press) only reported that the fish moved off station. Local data on salinity (Table C-4 and Figure C-19) indicate that there may be a slight salinity gradient within the UOP site (HMDC, 1986) which could be influencing movement patterns.

Studies performed during the fall and winter recorded various and contradictory migratory patterns. Childster (1920) reported that Fundulus spp. migrate to more saline waters in order to overwinter. Smith (1971) reported a migration to less saline waters. This discrepancy could be explained by a study performed by Fritz et al. (1975). It was proposed that these fish are polytypic in their migratory behavior. Fritz et al. (1975) observed that fish inhabiting low salinity waters (1-21%) during the summer migrated towards even lower salinity waters while those fish inhabiting higher salinity waters (17-29%) were reported to remain stationary throughout the year.

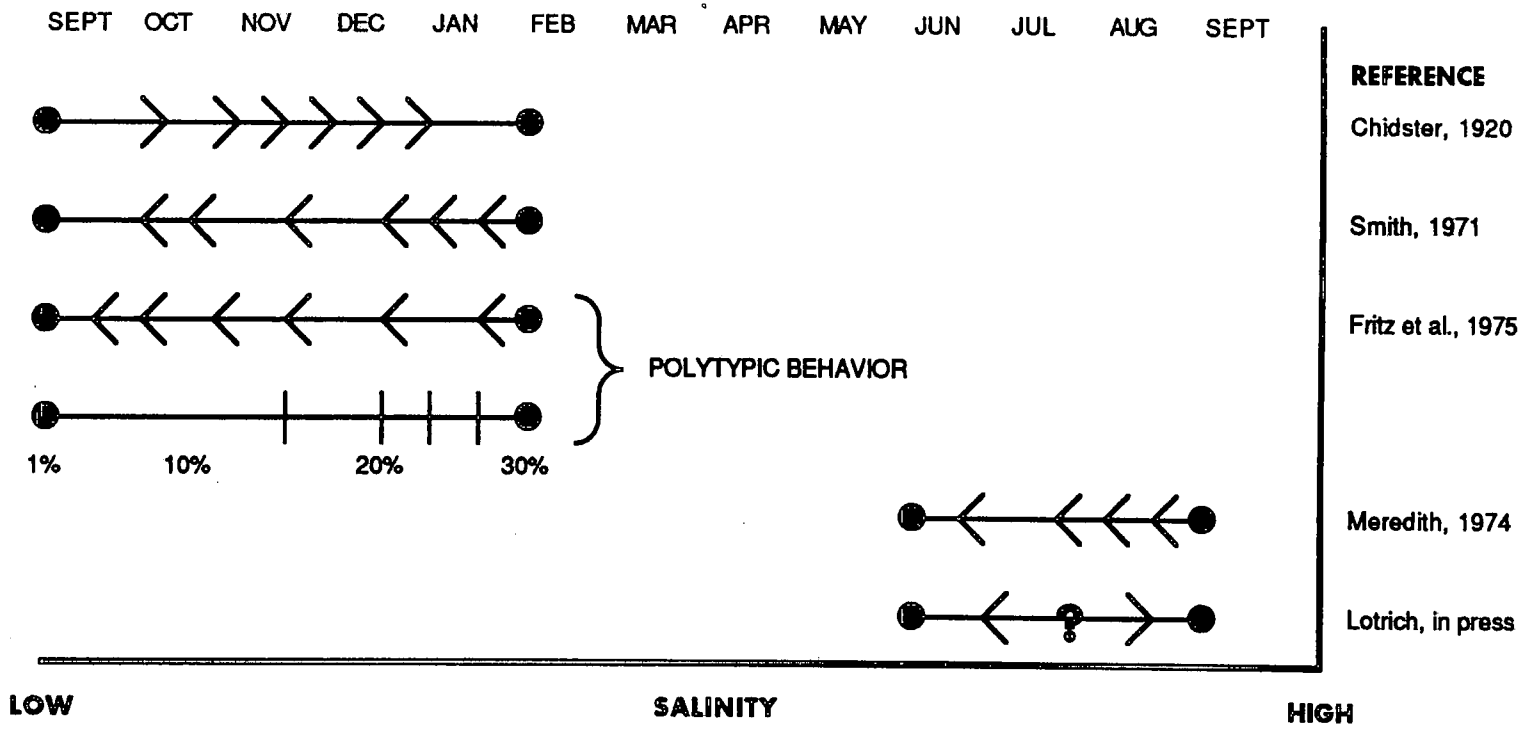
There was little information available on the condition factors of Fundulus at unrelated sites. Weis et al. (1986) at another site of similar pH and salinity reported slightly higher condition factors for Fundulus heteroclitus. The range of condition factors (K) which they reported were from approximately 1.85 to 2.0. The fish at the UOP site ranged

TABLE C-4

HACKENSACK MEADOWLANDS SUMMER WATER QUALITY PROJECT

FOUR YEAR SUMMARY 1983 - 1986

Station		Salinity (ppt) 1983	1984	1985	1986
Station 7	Min	2.3	0.0	4.2	2.5
Berry's Creek Lower	Max	10.5	8.0	9	12.0
(BCL)	High Avg	4.9	1.9	5.4	9.1
	Low Avg	3.7	1.6	5.6	4.7
	Tot Avg	4.2	2.9	5.5	6.4
	Count	18	17	17	13
Station 8	Min	0.8	0.0	2	2.0
Berry's Creek Middle	Max	7.0	5.0	20	10.0
(BCM)	High Avg	4.9	1.9	5.4	9.1
	Low Avg	3.7	1.6	5.6	4.7
	Tot Avg	4.2	2.9	5.5	6.4
	Count	18	17	17	13
Station 9	Min	0.5	0.0	0.0	1.0
Berry's Creek Upper	Max	5.2	3.5	7	8.0
(BCU)	High Avg	3.0	1.2	2.5	6.2
	Low Avg	1.2	0.7	2.9	2.8
	Tot Avg	2.3	0.87	2.7	4.1
	Count	18	17	17	13



Indicate direction of movement (i.e. towards greater or lesser salinity) during autumn and late spring

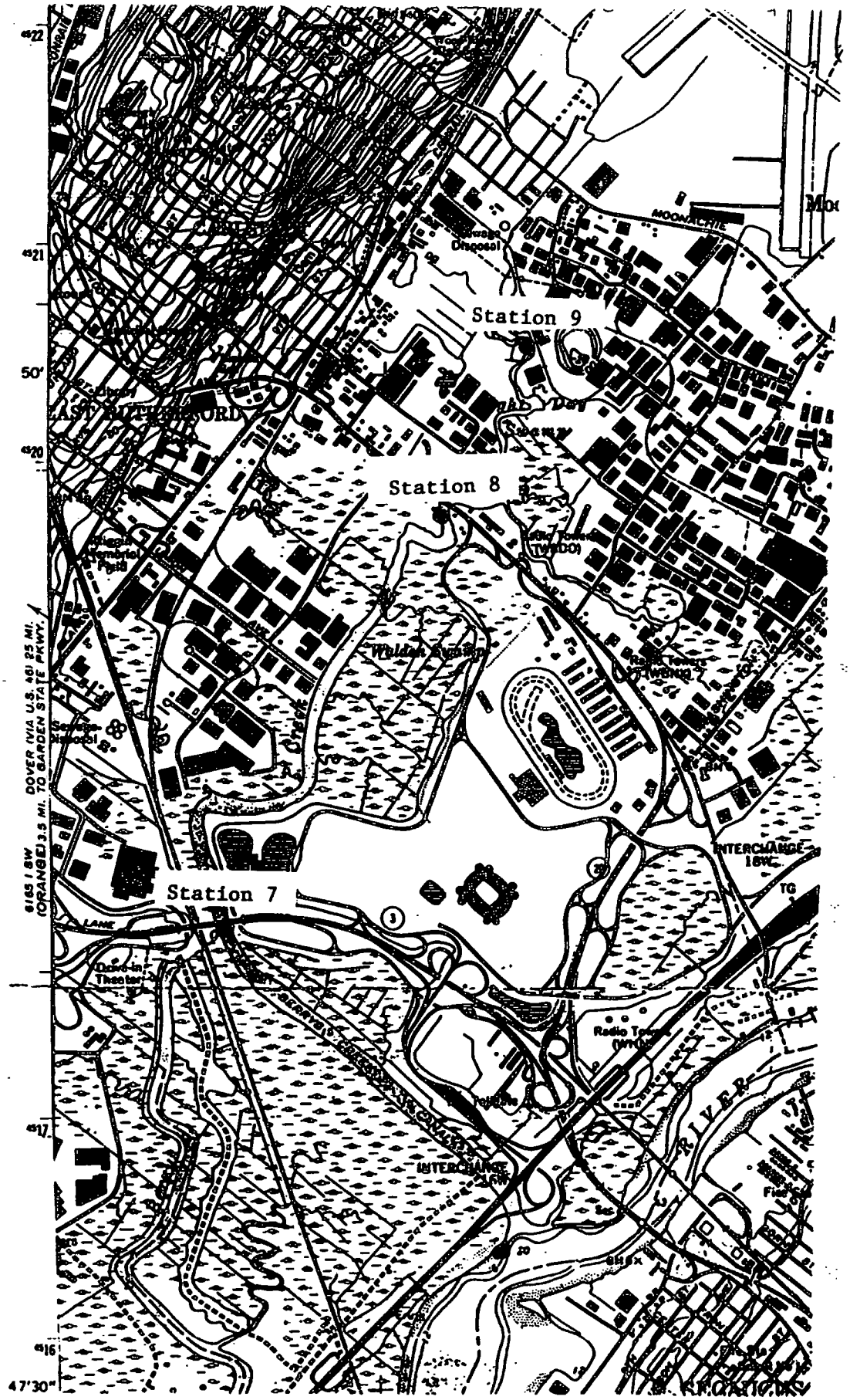
Indicates no significant movement

Indicates specimen moved off station. Destination unknown

Indicates endpoint of observation period

For a majority of each season (winter/summer) Fundulus spp. have a very limited homerange. The summer homerange determined by Lotrich (1974) was 36m. Between seasons, Fundulus appear to migrate. Frite (1975) observed migration of up to 2 km.

Figure C-18 SEASONAL MIGRATION OF FUNDULUS spp



Hackensack Meadowlands Development Commission Water Quality Project Key Map

Data Collected during the summers 1983-1986

Station 7- Berrys Creek at Route 3 Bridge
Station 8 - Berrys Creek at Paterson Plank Road
Station 9 - Berrys Creek at Starke Road

Figure C-19

from approximately 1.2 (small size class) to 1.6 (large size class). Condition factors for Fundulus were also reported from a relatively pristine lake in New Hampshire (Lutz, 1989). These Fundulus (species unknown) were reported to have extremely low background levels of contaminants and were used as the control organisms for another Berry's Creek Fundulus study (Lutz, 1989). The average K value for these New Hampshire fish was 1.2, which is lower than the averages determined in this study. One would expect that a fish from an uncontaminated site would have a high K value as a function of the "cleaner environment"; unless the individuals at the UOP site are not affected by the contaminants. Other possible explanations for the lower K value of these New Hampshire fish could be a lack of food in their environment or sensitivity to low pH (Weis et al., 1986) or a difference in species. Weisberg and Lotrich (1982) found that food availability and not crowding governed growth in the mummichog Fundulus heteroclitus.

Sediment concentrations of PCBs and chromium are highest at the most instream portion of the study site and lowest in the reference area. This sort of gradient is not evident for mercury concentrations. There appears to be a nearly uniform distribution of mercury in on-site and off-site areas. It is important to note that there is substantial within site variation in sediment concentrations of mercury, chromium, and PCBs indicating that exposure concentrations are not uniform within the stream channels.

$Log_{(10)}$ transformed sediment and fish concentrations from the same locations are not strongly correlated with the exception of PCB concentrations in small fish (Figures C-20 to C-22). The data concerning PCB body burdens indicate that small fish accumulate higher PCB levels than larger fish when compared on a body weight basis. This may be due to a difference in the ability of large and small fish to metabolize or excrete PCBs. A second possibility exists that large fish,

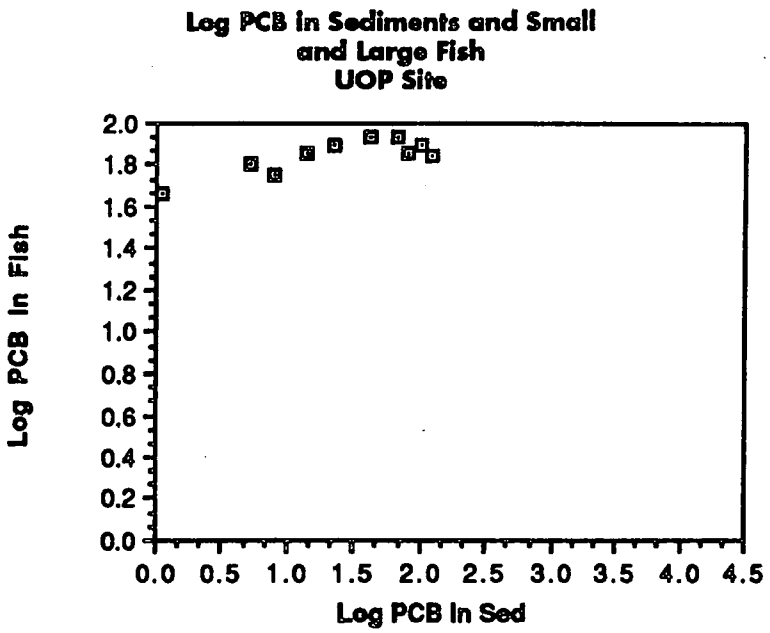

Log PCB in Sediments and Small
and Large Fish
UOP Site
Log PCB in Fish
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
0.0 0.5 1.0 1.5 2.0 2.5 3.0 3.5 4.0 4.5
Log PCB in Sed

Figure C-20

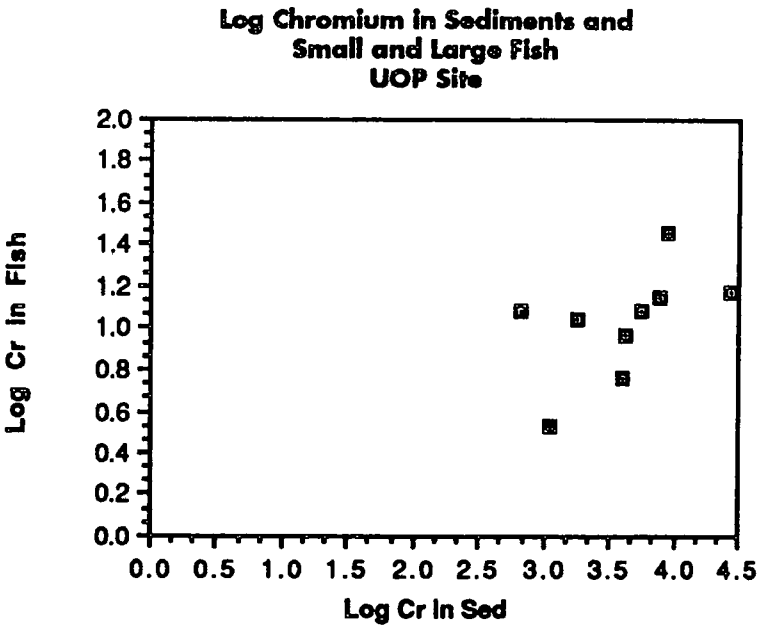

Log Chromium in Sediments and
Small and Large Fish
UOP Site
Log Cr in Fish
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
0.0 0.5 1.0 1.5 2.0 2.5 3.0 3.5 4.0 4.5
Log Cr in Sed

Figure C-21

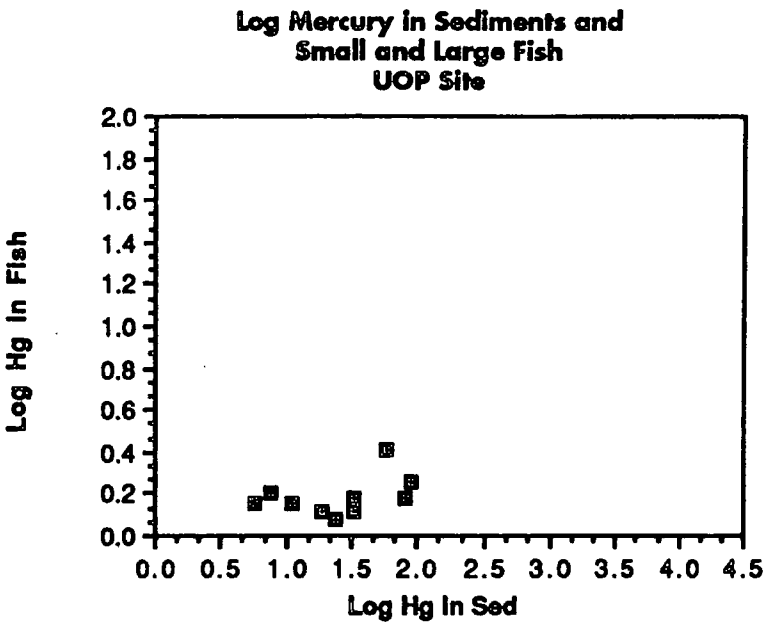

Log Mercury in Sediments and
Small and Large Fish
UOP Site
Log Hg in Fish
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
0.0 0.5 1.0 1.5 2.0 2.5 3.0 3.5 4.0 4.5
Log Hg in Sed

Figure C-22

due to their high lipid content, rapidly accumulate a burden of PCBs which leads to death. If so, the only large fish which do survive are those which have not yet accumulated high body burdens. The influence of other physiological parameters, for which data were not available, may explain this size-class/concentration relationship.

Comparing fish body burdens to channel sediment concentrations, the large fish were found to have similar PCB tissue concentrations, regardless of sampling site sediment concentrations. In small fish, however, body burden appeared to depend on sediment concentration. Explanations for this difference include the possibilities that large and small fish distinctions may reflect species and/or sex differences. Different sexes and species may have different physiological processes such as metabolic rate and ability to biotransform and excrete PCBs.

As there are so many unknowns which come into play when interpreting the data, it is not possible to explain the correlations between PCB sediment concentration, fish tissue concentration, and size class.

APPENDIX D

FOCUSED ENVIRONMENTAL ASSESSMENT DATA

APPENDIX D

FOCUSED ENVIRONMENTAL ASSESSMENT DATA

Appendix D contains focused environmental assessment data for the UOP Site in East Rutherford, New Jersey. The data presented here include concentrations of indicator chemicals in fish and sediment samples, mark and recapture study results, and physical parameters for fish and sediment samples. The data include both large and small size classes for fish.

TABLE D-1

SUMMARY OF FISH TISSUE ANALYSIS

SMALL FISH (DRY WEIGHTS)

UOP SITE, EAST RUTHERFORD, NEW JERSEY

Sample Location	PCB Aroclor 1248 (µg/g)	Total Chromium (µg/g)	Total Mercury (µg/g)	Percent Lipids	Percent Solids
Aa	102.00	3.55	1.76	3.22	22.4
Ab	85.00	18.80	1.34	2.53	23.2
Ac	80.00	24.40	1.56	3.65	24.2
Ba	91.00	18.60	1.51	2.74	21.6
Bb	94.00	9.46	1.28	3.09	24.3
Bc	65.00	3.10	1.45	3.48	27.9
Ea	31.00	10.00	1.72	3.14	24.6
Ca	68.00	1.73	1.43	2.97	28.9
Cb	84.00	13.00	1.09	3.48	26.0
Cc	100.00	10.40	1.18	3.92	26.2
Da	34.00	<1.75 (1)	1.56	2.95	28.5
Db	28.00	<2.00 (1)	3.05	2.80	24.7
Dc	30.00	2.43	1.62	2.66	24.1
Averages					
A	89.00	15.58	2.33	3.13	23.3
B	77.67	11.54	1.46	3.05	24.1
C	84.00	8.38	1.23	3.46	27.0
D	30.67	1.44	2.08	2.80	25.8

(1) One half of the detection limit was used for calculations.

TABLE D-2

SUMMARY OF FISH TISSUE ANALYSIS

LARGE FISH (DRY WEIGHTS)

UOP SITE, EAST RUTHERFORD, NEW JERSEY

Sample Location	PCB Aroclor 1248 (µg/g)	Total Chromium (µg/g)	Total Mercury (µg/g)	Percent Lipids	Percent Solids
Aa	74.00	14.50	3.37	3.78	13.4
Ab	61.00	4.87	1.09	4.68	48.7
Ac	58.00	6.33	1.23	4.10	25.2
Ba	67.00	39.00	1.51	5.46	27.5
Bb	79.00	1.77	1.21	4.53	28.2
Bc	63.00	2.13	1.26	3.07	-
Ea	66.00	42.00	1.60	3.63	25.0
Ca	59.00	5.08	1.42	4.76	31.8
Cb	59.00	8.35	1.45	4.33	32.6
Cc	60.00	14.30	1.94	3.67	24.7
Da	44.00	5.40	1.51	4.19	31.2
Db	120.00	6.35	0.94	2.96	28.4
Dc	28.00	50.00	2.17	3.40	23.2
Averages					
A	64.33	8.57	1.90	4.19	15.8
B	70.17	20.95	1.38	4.45	26.7
C	59.33	27.73	1.60	4.25	29.7
D	64.00	20.58	1.54	3.52	27.6

TABLE D-3

SUMMARY OF FUNDULUS MARK AND RECAPTURE STUDY

UOP SITE

SITE	TAG LOCATION		
	LEFT PECTORAL	RIGHT PECTORAL	CAUDAL
AA	0	0	6
AB	0	0	1
AC	3	2	8
BA	0	1	4
BB	0	1	7
BC	1	0	4
CA	0	0	6
CB	0	0	3
CC	0	4	4
DA	0	3	14
DB	1	0	8
DC	1	1	3
EA	0	0	3
total	6	12	71

N: population estimate
T: # marked (precensus)
t: # marked (recapture)
n: # trapped (recapture)

$$N = nT/t$$
$$= 55,798$$

$$S.E. = N \quad \frac{(N-T)(N-n)}{Tn(N-1)}$$
$$= 7,085$$

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TABLE D-4

SUMMARY OF PHYSICAL PARAMETERS FOR FISH SAMPLES

LARGE AND SMALL FISH

UOP SITE

SITE	SIZE	AVG. LENGTH (mm)	AVG. WEIGHT (g)	AVG. CONDITION (K)
AA	LG	56.5	2.6	1.4
AA	SM	77.5	7.2	1.5
AB	LG	60.8	2.9	1.3
AB	SM	71.5	6.7	1.8
AC	LG	63.2	3.1	1.2
AC	SM	73.6	6.4	1.6
BA	LG	58.7	2.8	1.4
BA	SM	77.5	8.5	1.8
BB	LG	59	2.9	1.4
BB	SM	80.2	8.4	1.6
BC	LG	63.3	3.3	1.3
BC	SM	78.8	7.6	1.5
CA	LG	58	2.8	1.4
CA	SM	81	8.7	1.6
CB	LG	52.8	2.3	1.5
CB	SM	73.2	6.4	1.6
CC	LG	60.1	3.1	1.4
CC	SM	77.7	7.9	1.7
DA	LG	60.8	3.1	1.4
DA	SM	76.1	6.6	1.5
DB	LG	67.4	4.2	1.4
DB	SM	75.5	6.6	1.5
DC	LG	63.9	2.9	1.1
DC	SM	76.8	6.7	1.5
EA	LG	54.8	2.3	1.4
EA	SM	76.2	7.3	1.7

TABLE D-5

MARK AND RECAPTURE AND PHYSICAL PARAMETER RAW DATA

FOR FISH SAMPLES

UOP SITE

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
AA	Sm		45	1.2	1.3
			62	3	1.3
			60	3.4	1.6
			64	4.6	1.8
			57	2.2	1.2
			58	2.4	1.2
			73	4.2	1.1
			42	1	1.3
			65	3.8	1.4
			66	4	1.4
			46	1.4	1.4
			52	2.2	1.6
			61	3.2	1.4
			45	1.2	1.3
		c	49	1.2	1.0
			47	1	1.0
			68	4	1.3
			62	2.6	1.1
			43	0.8	1.0
			48	0.8	0.7
		c	62	3	1.3
		c	40	1.2	1.9
			65	3.6	1.3
			66	3.2	1.1
			45	1.2	1.3
			73	4.2	1.1
			61	2.4	1.1
			65	3.6	1.3
			49	1.4	1.2
			44	0.8	0.9
			40	0.8	1.3
			48	1	0.9
			65	4.2	1.5
			54	1	0.6
			43	0.6	0.8
			53	2	1.3
			40	0.8	1.3
			43	1	1.3
			55	2	1.2
			70	3.8	1.1
			50	1.8	1.4
			45	1	1.1
			70	4.1	1.2
			65	3.6	1.3

TABLE D-5 (Page 2 of 29)

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			75	6.2	1.5
			63	3.6	1.4
			71	3.8	1.1
			66	4.2	1.5
			68	5	1.6
			37	0.6	1.2
			66	4	1.4
			40	1	1.6
			63	3	1.2
			52	1.8	1.3
			40	0.6	0.9
			52	1.6	1.1
			53	2.2	1.5
			53	3.2	2.1
			70	4.6	1.3
			42	0.8	1.1
			67	3.3	1.1
			48	1	0.9
			68	5.6	1.8
			72	5.8	1.6
			70	4.8	1.4
			65	4	1.5
			51	1.8	1.4
			75	5.8	1.4
AA	Lg		73	5	1.3
			73	6	1.5
			70	5.5	1.6
		c	84	9.5	1.6
		c	75	6	1.4
		c	80	7	1.4
			89	11	1.6
			84	10	1.7
			74	5.5	1.4
			73	6	1.5
			80	8	1.6
			74	6.5	1.6
			72	6	1.6
			78	6.5	1.4
			83	9.5	1.7
AB	Sm		71	4.4	1.2
			70	4.2	1.2
			53	1.6	1.1
			66	3.4	1.2
			52	1.6	1.1

TABLE D-5 (Page 3 of 29)

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			71	4.2	1.2
			63	2.6	1.0
			51	1.6	1.2
			66	3.6	1.3
			65	3.2	1.2
			70	4.4	1.3
			62	3.4	1.4
			55	2	1.2
			69	4.2	1.3
			61	2.6	1.1
			59	3.2	1.6
			60	2.8	1.3
			61	3.4	1.5
			59	3.2	1.6
			60	2.8	1.3
			61	3.4	1.5
			65	3.4	1.2
			57	2	1.1
		rp	60	3	1.4
		c	77	4.2	0.9
			67	4	1.3
			61	3.8	1.7
			62	3.2	1.3
			61	3.4	1.5
			62	3	1.3
			62	3.2	1.3
			66	3.6	1.3
			58	2.2	1.1
			57	2.8	1.5
			63	2.8	1.1
			58	2.2	1.1
			52	1.8	1.3
			57	2.2	1.2
			64	3.2	1.2
			45	1.2	1.3
			66	3.8	1.3
			61	2.4	1.1
			59	2.2	1.1
			68	3.8	1.2
			70	4.8	1.4
			48	1.6	1.4
			67	3.2	1.1
			65	3	1.1
			61	2.8	1.2

TABLE D-5 (Page 4 of 29)

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			58	1.6	0.8
			61	2.8	1.2
			61	1.8	0.8
		c	55	1.8	1.1
			52	1.2	0.9
		c	50	1.2	1.0
			41	0.8	1.2
AB	Lg		75	5.5	1.3
			72	5.5	1.5
			76	7	1.6
			72	5	1.3
			76	6	1.4
			71	6	1.7
			70	6.5	1.9
			74	5.5	1.4
			75	6	1.4
			73	5	1.3
			73	6	1.5
			70	5.5	1.6
			81	6	1.1
			74	6	1.5
			80	7	1.4
			68	5	1.6
			93	13	1.6
			77	6	1.3
			82	9	1.6
			95	11.5	1.3
			75	6.5	1.5
			85	8.5	1.4
			72	7	1.9
			81	7	1.3
			80	6.5	1.3
			74	6	1.5
			79	8	1.6
			76	6.5	1.5
			74	6	1.5
AC	Sm		64	3.6	1.4
			70	4	1.2
			61	3.4	1.5
		c	59	2.8	1.4
			67	3.8	1.3
			66	3	1.0
			73	5.6	1.4
			46	1.2	1.2

TABLE D-5 (Page 5 of 29)

SITE	SIZE	LENGTH (mm)	WEIGHT (g)	CONDITION (K)
		58	2.2	1.1
		66	3.6	1.3
		48	1.4	1.3
		69	4.4	1.3
		67	4.6	1.5
		71	4.2	1.2
		70	4.2	1.2
		70	4.4	1.3
		66	3	1.0
		65	2.8	1.0
		71	4.6	1.3
		68	4	1.3
		71	4.6	1.3
		67	4	1.3
		64	3.2	1.2
		76	4.2	1.0
		78	6.2	1.3
	c	67	3.4	1.1
		64	2.8	1.1
		52	2	1.4
		62	3	1.3
		70	4.4	1.3
		67	3.8	1.3
		68	4	1.3
		65	3.4	1.2
		65	3	1.1
	lp	65	3	1.1
		65	3.4	1.2
		67	3.6	1.2
		67	4	1.3
		65	3.2	1.2
		60	3	1.4
		58	1.8	0.9
	c	46	1.2	1.2
		70	3.8	1.1
		56	1.6	0.9
		69	3.6	1.1
		66	3.4	1.2
		63	3	1.2
		65	3.2	1.2
		57	2	1.1
		52	1.4	1.0
		67	3.6	1.2
		59	2.6	1.3

TABLE D-5 (Page 6 of 29)

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			58	2	1.0
			65	2.8	1.0
			55	1.8	1.1
			67	3.6	1.2
			45	1.4	1.5
			58	2.4	1.2
			51	1.2	0.9
			65	2.2	0.8
			60	2.2	1.0
			64	2.6	1.0
			65	3.6	1.3
			68	3.4	1.1
			65	2.6	0.9
			68	3.8	1.2
			70	3.4	1.0
			65	2.8	1.0
			74	4.6	1.1
			62	2.6	1.1
			64	3	1.1
			65	3.2	1.2
			68	3.8	1.2
			71	3.6	1.0
		c	64	4	1.5
			61	2.6	1.1
			48	1.6	1.4
			46	1.2	1.2
			67	3.6	1.2
			45	1.2	1.3
			44	0.8	0.9
AC	Lg		74	6.5	1.6
			70	6.5	1.9
			72	6	1.6
			75	8	1.9
			77	8	1.8
			74	7.5	1.9
			68	4.5	1.4
			75	6	1.4
			76	7	1.6
			77	7	1.5
			76	8	1.8
			66	4.5	1.6
			72	6.5	1.7
			78	7	1.5
		c	76	7.5	1.7

TABLE D-5 (Page 7 of 29)

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			82	8	1.5
			80	8	1.6
			75	6.5	1.5
			74	5	1.2
			82	8.5	1.5
			73	7.5	1.9
			70	4.5	1.3
			69	5	1.5
			68	5	1.6
		c	71	5.5	1.5
			78	7.5	1.6
			73	6	1.5
			71	5.5	1.5
			72	5.5	1.5
			70	5	1.5
			68	5	1.6
			73	6	1.5
BA	Sm		56	1.8	1.0
			62	2.6	1.1
			63	3.2	1.3
			69	5	1.5
			41	0.8	1.2
			58	2.6	1.3
			61	3	1.3
			66	3.8	1.3
			69	3.8	1.2
			70	4.8	1.4
			69	4.2	1.3
			62	2.8	1.2
			43	1	1.3
			68	4.2	1.3
			48	1.2	1.1
			66	4.2	1.5
			48	1.6	1.4
			63	3.6	1.4
			65	4.8	1.7
			42	1	1.3
			63	2.4	1.0
			66	2.6	0.9
			77	5.4	1.2
			58	2.6	1.3
			49	1.6	1.4
			64	3.8	1.4
			67	3.6	1.2

TABLE D-5 (Page 8 of 29)

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			62	2.2	0.9
			80	7.2	1.4
			61	3.4	1.5
			63	2.8	1.1
			62	4.2	1.8
			64	3	1.1
			65	3.4	1.2
			67	3	1.0
			69	4.2	1.3
			50	1.8	1.4
			68	4.4	1.4
			57	2.6	1.4
			61	2.6	1.1
			66	4.8	1.7
			72	3.8	1.0
			59	2.4	1.2
			49	1.4	1.2
		c	65	3.2	1.2
			54	1.4	0.9
			49	1.4	1.2
			40	0.8	1.3
			53	1.8	1.2
			46	1.6	1.6
			52	1.4	1.0
			60	3.8	1.8
			60	3	1.4
			58	3	1.5
			44	1.8	2.1
			47	1	1.0
			44	1	1.2
			40	0.8	1.3
			44	1	1.2
BA	Lg	rp	84	9	1.5
		c	76	8	1.8
			96	15.5	1.8
			97	17.5	1.9
			75	6.5	1.5
			81	8	1.5
			71	7	2.0
			65	5	1.8
			80	9.5	1.9
			75	6.5	1.5
			68	5.5	1.7
			71	6	1.7

TABLE D-5 (Page 9 of 29)

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			69	6	1.8
BB	Sm		46	1.4	1.4
		c	42	1	1.3
			42	1.2	1.6
			54	3	1.9
			66	3	1.0
			67	3.6	1.2
			63	3	1.2
			61	3.6	1.6
			63	3.6	1.4
			63	3	1.2
			65	3.8	1.4
			56	2.2	1.3
			59	2.6	1.3
		c	60	2	0.9
		c	62	2	0.8
			65	3.4	1.2
			37	0.6	1.2
			40	0.8	1.3
			53	1.8	1.2
			60	2.4	1.1
			64	3.6	1.4
			56	2	1.1
			60	2.6	1.2
			70	4.4	1.3
			68	4.4	1.4
			59	2.4	1.2
			61	3.2	1.4
			66	3.8	1.3
			39	1	1.7
			57	2.2	1.2
			62	3	1.3
			65	3.8	1.4
			64	3.6	1.4
			63	3.4	1.4
			64	3.2	1.2
			59	2.8	1.4
			72	4	1.1
			55	2	1.2
			39	0.8	1.3
			65	3.2	1.2
			54	2	1.3
			71	5.2	1.5
			51	1.8	1.4

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			71	5.2	1.5
			69	5.2	1.6
			61	3.6	1.6
			58	3	1.5
			56	2.6	1.5
			57	2.6	1.4
			60	2.4	1.1
			70	4.6	1.3
		c	72	4.8	1.3
			63	3.4	1.4
			62	3.2	1.3
			64	3.8	1.4
			61	3.8	1.7
			52	2.4	1.7
			66	5.4	1.9
			63	3.6	1.4
			61	3.2	1.4
			40	1.2	1.9
			51	2.2	1.7
		c	54	3.2	2.0
BB	Lg		71	6.5	1.8
			82	9	1.6
			86	10	1.6
			79	7	1.4
			75	7	1.7
			81	10	1.9
			88	11	1.6
			88	9.5	1.4
			75	6.5	1.5
			83	8.5	1.5
			82	9.5	1.7
			86	9.5	1.5
			76	7	1.6
			84	9	1.5
			81	7.5	1.4
			69	5	1.5
			75	6	1.4
			87	9.5	1.4
			73	6	1.5
			78	7.5	1.6
		c	74	6	1.5
			84	9.5	1.6
			83	10	1.7
			97	16	1.8

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			79	8	1.6
			77	9.5	2.1
			72	6.5	1.7
			80	9	1.8
BC	Sm	c	71	4	1.1
			76	5.6	1.3
			65	2.8	1.0
			68	3.2	1.0
			57	2	1.1
			72	4.8	1.3
			68	4.6	1.5
			53	2	1.3
			74	5.6	1.4
			37	0.8	1.6
			78	6.8	1.4
			66	3.8	1.3
			60	1.8	0.8
			65	3.4	1.2
			75	5.2	1.2
			75	5.2	1.2
			77	5.6	1.2
			46	1.4	1.4
			75	4.4	1.0
			46	1.2	1.2
			59	1.6	0.8
			75	5.6	1.3
		c	63	2.8	1.1
			60	2.2	1.0
			48	1.2	1.1
			65	2.4	0.9
			79	6	1.2
			52	1.8	1.3
			73	4.8	1.2
			68	2.8	0.9
			74	3.6	0.9
			45	1	1.1
			46	1.2	1.2
			70	4	1.2
			72	4	1.1
			71	4.2	1.2
			69	4.6	1.4
			66	4	1.4
			66	4.2	1.5
			77	6	1.3

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
		c	38	1	1.8
			46	0.8	0.8
			76	4.6	1.0
			68	3.4	1.1
			69	4	1.2
			67	4.4	1.5
			63	3.4	1.4
			63	3	1.2
			50	1.8	1.4
			62	2.6	1.1
			40	0.8	1.3
			65	3.2	1.2
			79	4.6	0.9
			58	2	1.0
			75	4.6	1.1
			53	1.4	0.9
			40	0.8	1.3
			66	3	1.0
			61	3.4	1.5
			46	1	1.0
			70	2.8	0.8
			56	1.8	1.0
			60	2.2	1.0
			76	5.8	1.3
			65	2.8	1.0
BC	Lg		84	9.5	1.6
			85	8	1.3
			84	9.5	1.6
		c	73	5.5	1.4
			68	6	1.9
			88	9.5	1.4
			82	8.5	1.5
			80	7	1.4
			79	7	1.4
			92	12.5	1.6
			81	7.5	1.4
			82	8.5	1.5
			66	5	1.7
			76	6.5	1.5
			71	5.5	1.5
			80	7	1.4
			70	6	1.7
			79	7.5	1.5
			82	7.5	1.4

TABLE D-5 (Page 13 of 29)

SITE	SIZE	LENGTH (mm)	WEIGHT (g)	CONDITION (K)
		82	9	1.6
		80	8	1.6
		81	7.5	1.4
		76	7	1.6
		77	7.5	1.6
		76	7.5	1.7
		75	6.5	1.5
CA	Sm	70	4.6	1.3
		68	4.4	1.4
		61	2.8	1.2
		59	3	1.5
		62	4.4	1.8
		61	3	1.3
		67	3.8	1.3
		68	4	1.3
		65	3.4	1.2
		66	3.6	1.3
		63	3.4	1.4
		60	3.2	1.5
		64	3.4	1.3
		66	3.6	1.3
		61	2.4	1.1
		71	4	1.1
		65	3.2	1.2
		42	1	1.3
		66	3.8	1.3
		71	4.2	1.2
		62	2.6	1.1
		63	3.2	1.3
		62	2.8	1.2
		41	1.2	1.7
		62	3.2	1.3
		65	4.6	1.7
		67	4.4	1.5
		59	2.8	1.4
		65	3.4	1.2
		63	3.6	1.4
		65	3.8	1.4
		62	3.4	1.4
		64	3.6	1.4
		63	3	1.2
		60	3.2	1.5
		65	3.6	1.3
		63	3.2	1.3

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SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			50	2	1.6
			41	0.8	1.2
			64	3.6	1.4
			63	2.8	1.1
		c	43	1.2	1.5
			46	1.4	1.4
			60	3.2	1.5
			66	4	1.4
			65	4	1.5
			62	4.2	1.8
			66	3.2	1.1
			64	3	1.1
			67	3.2	1.1
			63	4	1.6
			62	4.2	1.8
			45	1.6	1.8
			52	2	1.4
			67	4.6	1.5
			63	3.2	1.3
			41	0.8	1.2
			43	1	1.3
			44	1	1.2
			35	0.6	1.4
			42	0.8	1.1
			42	0.8	1.1
			37	0.6	1.2
			62	3	1.3
			50	1.6	1.3
			63	2.2	0.9
			64	2.8	1.1
			46	1.4	1.4
			61	2.8	1.2
			62	2.8	1.2
			44	1.2	1.4
			42	1.8	2.4
			64	3.2	1.2
		c	62	3.6	1.5
			53	1.6	1.1
		c	41	0.6	0.9
			40	1	1.6
			47	1.4	1.3
			52	2.2	1.6
			61	2.2	1.0
			58	2.6	1.3

TABLE D-5 (Page 15 of 29)

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			60	3	1.4
		c	57	2.6	1.4
			43	0.8	1.0
CA	Lg		79	7	1.4
			75	6	1.4
			74	6.2	1.5
			81	8.5	1.6
			77	7	1.5
			83	8.5	1.5
			83	8	1.4
			67	5	1.7
			79	7	1.4
			76	6.8	1.5
			78	7.4	1.6
			73	5.2	1.3
			92	13.5	1.7
			81	7.5	1.4
			84	12	2.0
			83	10	1.7
			95	14.5	1.7
			90	11.5	1.6
			90	10.5	1.4
			86	12	1.9
			89	13	1.8
			82	10	1.8
			85	9.5	1.5
			86	9	1.4
			81	8.5	1.6
			79	8.5	1.7
			74	7	1.7
			81	8.5	1.6
			77	8	1.8
			78	7.5	1.6
			78	7.5	1.6
			76	7	1.6
CB	Sm		69	4.2	1.3
			40	0.6	0.9
			54	3.4	2.2
			56	2.8	1.6
			41	0.8	1.2
			62	3.2	1.3
			41	1	1.5
			43	1.2	1.5
			33	0.6	1.7

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SITE	SIZE	LENGTH (mm)	WEIGHT (g)	CONDITION (K)
		56	3	1.7
		38	0.8	1.5
		42	1.4	1.9
		42	1.2	1.6
		51	1.6	1.2
		52	2.2	1.6
		55	3	1.8
		59	2.8	1.4
		36	0.8	1.7
		41	1.4	2.0
		60	3	1.4
		41	0.8	1.2
		66	3.2	1.1
		44	1	1.2
		54	2.2	1.4
		40	0.8	1.3
		41	0.8	1.2
		42	1	1.3
		59	2.8	1.4
		52	2	1.4
		60	2.8	1.3
		73	4	1.0
		40	0.8	1.3
		71	4.8	1.3
		65	3.2	1.2
		46	1	1.0
		44	1	1.2
		43	1	1.3
		50	1.6	1.3
		44	1	1.2
		61	3	1.3
		58	2.6	1.3
		63	4	1.6
		60	3.8	1.8
		65	3.8	1.4
		61	3	1.3
		54	2	1.3
		59	2.8	1.4
		42	0.8	1.1
		60	3.8	1.8
		63	3.8	1.5
		62	3.8	1.6
		68	4.6	1.5
		42	0.8	1.1

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			38	0.8	1.5
			41	1.4	2.0
			38	0.8	1.5
			42	1	1.3
			46	0.8	0.8
			44	0.6	0.7
			44	0.8	0.9
			63	3.4	1.4
			64	3.8	1.4
			44	1.2	1.4
			71	4.8	1.3
			45	1.4	1.5
			56	2.6	1.5
			61	1.2	0.5
			61	3.2	1.4
			52	1.8	1.3
			60	2.8	1.3
			66	3.8	1.3
			66	3.8	1.3
			40	0.8	1.3
			45	1	1.1
		c	61	3.4	1.5
			65	4.8	1.7
			63	3.6	1.4
			57	2.6	1.4
			65	3.6	1.3
			64	3.6	1.4
CB	Lg		65	4	1.5
			76	5.8	1.3
			62	3.6	1.5
			62	2.6	1.1
			69	4.8	1.5
			64	3.2	1.2
			67	4.6	1.5
			71	5.6	1.6
			69	5.6	1.7
			71	5.6	1.6
			67	5.4	1.8
			78	7.8	1.6
			77	7	1.5
			71	5	1.4
			64	4	1.5
			65	4.6	1.7
			69	4.2	1.3

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			83	10.5	1.8
			84	9	1.5
			92	12.5	1.6
			83	9.5	1.7
			72	7	1.9
			95	16	1.9
			84	10	1.7
			87	10.5	1.6
			76	7	1.6
			80	7.5	1.5
			72	6	1.6
		c	72	5	1.3
			74	5	1.2
			70	5	1.5
			80	7.8	1.5
			70	5	1.5
			71	6	1.7
			71	5.4	1.5
			63	3.4	1.4
			64	3.8	1.4
CC	Sm		57	2.2	1.2
			62	2.8	1.2
			61	3.8	1.7
			62	3.8	1.6
			46	1.6	1.6
			62	3.2	1.3
			78	3.1	0.7
			44	1.2	1.4
			42	1.2	1.6
			39	0.6	1.0
			60	3.2	1.5
			42	0.8	1.1
			61	2.6	1.1
		rp	63	2.8	1.1
			62	3.2	1.3
			56	2.6	1.5
			57	2.4	1.3
			61	2.8	1.2
			54	2.8	1.8
			61	2.8	1.2
			52	2.2	1.6
			63	3.2	1.3
			65	3.8	1.4
			61	3	1.3

SITE	SIZE	LENGTH (mm)	WEIGHT (g)	CONDITION (K)
		62	3	1.3
		59	2.2	1.1
		60	3.2	1.5
		62	3.2	1.3
		64	4.2	1.6
		66	3.2	1.1
		64	3.6	1.4
		52	2	1.4
		54	2.2	1.4
		64	3.4	1.3
		61	2.4	1.1
		62	3.6	1.5
		59	3.4	1.7
		61	2.6	1.1
		58	2.2	1.1
		56	2.6	1.5
		63	2.4	1.0
	rp	41	0.8	1.2
		41	1.6	2.3
		57	2.6	1.4
		59	2.4	1.2
		54	2.2	1.4
		68	4.6	1.5
		58	2.8	1.4
		71	4.6	1.3
		73	4.8	1.2
		69	4.4	1.3
		65	3.4	1.2
		71	4.9	1.4
		70	3.9	1.1
		66	3.4	1.2
		69	5.1	1.6
		43	1.2	1.5
		69	4.3	1.3
		62	3.4	1.4
		62	3.4	1.4
		63	3.5	1.4
		66	3.9	1.4
		68	4.1	1.3
		38	0.8	1.5
		43	1	1.3
		65	3.8	1.4
		64	3.8	1.4
		61	3.1	1.4

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SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			64	3.8	1.4
			67	4.4	1.5
			66	4.4	1.5
			42	0.8	1.1
			40	0.8	1.3
			64	3.2	1.2
			66	4	1.4
			72	6	1.6
			75	4.8	1.1
			61	3.8	1.7
			63	3.4	1.4
		c	66	3.6	1.3
			63	3.4	1.4
			64	4.4	1.7
			65	2.6	0.9
			65	4.4	1.6
			67	3.4	1.1
			65	4.3	1.6
CC	Lg		72	6	1.6
			74	6	1.5
			87	10.5	1.6
			75	6.5	1.5
			86	9	1.4
			70	5.5	1.6
			72	6	1.6
			70	6	1.7
			78	7.5	1.6
			72	6	1.6
			76	7	1.6
			91	15	2.0
			86	11	1.7
			90	12.5	1.7
			85	11	1.8
			84	10	1.7
			75	7	1.7
			77	8	1.8
			83	9.5	1.7
			78	8	1.7
			74	8.5	2.1
			76	7.5	1.7
			81	7.5	1.4
			77	6.5	1.4
			74	8	2.0
			72	7	1.9

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SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			76	5	1.1
		c	73	6	1.5
			77	7.5	1.6
			77	8.5	1.9
			70	5.5	1.6
DA	Sm		56	3.4	1.9
			65	3.6	1.3
			60	3.2	1.5
			62	3.8	1.6
			58	3	1.5
			62	3.8	1.6
			60	2.6	1.2
			63	2.8	1.1
			60	2.8	1.3
			51	1.8	1.4
			65	3.8	1.4
			64	3.8	1.4
		c	57	2.8	1.5
			69	4	1.2
			61	3.2	1.4
		c	64	3	1.1
			41	1	1.5
			62	3.4	1.4
			65	4	1.5
			74	1.2	0.3
			62	4.2	1.8
			44	1.4	1.6
			61	3	1.3
			69	4.6	1.4
			56	2.4	1.4
			59	2	1.0
			66	3.6	1.3
			58	2.4	1.2
			61	2.2	1.0
			60	3.6	1.7
			53	2.2	1.5
			60	2.4	1.1
			62	3.6	1.5
			61	2.8	1.2
			59	3.2	1.6
			64	3.8	1.4
			67	4	1.3
			59	2.6	1.3
			61	2.4	1.1

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
		c	62	3	1.3
			61	2.8	1.2
			56	2.2	1.3
			67	3.8	1.3
		c	69	4.2	1.3
			63	3.2	1.3
			69	4.4	1.3
			55	2.6	1.6
			60	2.6	1.2
			65	3.2	1.2
			61	3	1.3
			69	4.8	1.5
			66	3	1.0
			59	2.8	1.4
			70	4.6	1.3
			67	3.8	1.3
			56	2.4	1.4
			61	3	1.3
			60	3.4	1.6
			62	2.6	1.1
			63	3.8	1.5
			42	1.2	1.6
			61	3.4	1.5
			63	3	1.2
			61	3	1.3
			61	2.6	1.1
			64	3.4	1.3
		c	62	2.8	1.2
			62	5	2.1
			68	3.8	1.2
			54	1.6	1.0
			59	2.8	1.4
		c	39	0.8	1.3
			62	3.2	1.3
DA	Lg		67	5	1.7
			75	5	1.2
			71	5.5	1.5
			70	5.5	1.6
			73	5	1.3
			72	5	1.3
			71	4.5	1.3
			73	6	1.5
			74	5.5	1.4
			74	6	1.5

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			73	6	1.5
			79	7.5	1.5
			81	7.5	1.4
			89	10	1.4
			79	7	1.4
			77	6	1.3
			77	7.5	1.6
			80	7.5	1.5
			86	9.5	1.5
		c	75	7.5	1.8
			76	6	1.4
		c	70	6.5	1.9
		c	73	6	1.5
			78	6.5	1.4
			83	9.5	1.7
			78	7.5	1.6
			72	5	1.3
			77	6.5	1.4
		c	85	9.5	1.5
			74	5	1.2
			77	6.5	1.4
DB	Sm		76	7	1.6
			71	4.5	1.3
			72	4.5	1.2
			72	5	1.3
			76	7	1.6
			65	3.5	1.3
			62	2.5	1.0
			50	2	1.6
		c	63	3.5	1.4
			64	3	1.1
			65	4	1.5
			62	3.5	1.5
			73	5	1.3
			68	4.5	1.4
			62	3.5	1.5
			60	3.5	1.6
			54	3	1.9
			70	4	1.2
			71	5	1.4
			68	4	1.3
			69	4	1.2
			74	4.5	1.1
			67	4	1.3

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SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
		c	77	6.5	1.4
			68	4.5	1.4
			70	4.5	1.3
			74	5	1.2
			66	3.5	1.2
			73	5	1.3
			65	4	1.5
			71	4.5	1.3
			73	5	1.3
			75	5	1.2
			68	3.5	1.1
			67	4	1.3
			61	3.5	1.5
			68	3.5	1.1
			69	4.5	1.4
			67	3	1.0
			72	5	1.3
			61	3.5	1.5
			66	3.5	1.2
			69	5	1.5
			70	5	1.5
			64	3.5	1.3
			72	4.5	1.2
			49	2.5	2.1
DB	Lg		73	6.5	1.7
			87	9.5	1.4
			74	7	1.7
			85	10	1.6
			75	6.5	1.5
			76	7	1.6
			87	10	1.5
			70	6	1.7
			78	7	1.5
			79	7	1.4
			81	9	1.7
			77	6	1.3
			71	6.5	1.8
			76	5.5	1.3
			76	7	1.6
			73	6.5	1.7
			86	9	1.4
			67	5.5	1.8
			73	6	1.5
			77	6.5	1.4

TABLE D-5 (Page 25 of 29)

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			72	5.5	1.5
			73	6	1.5
			77	6.5	1.4
			72	5.5	1.5
			73	6.5	1.7
		c	76	6.5	1.5
			84	8	1.3
			73	5.5	1.4
			73	5.5	1.4
		c	74	7	1.7
			70	5.5	1.6
			71	5.5	1.5
			71	5	1.4
			78	6	1.3
			74	5.5	1.4
			73	5.5	1.4
			69	6	1.8
			76	5.5	1.3
DC	Sm		55	2	1.2
			57	1.8	1.0
			71	3.4	0.9
			67	3.6	1.2
			64	2.8	1.1
			66	3.8	1.3
			50	0.8	0.6
			62	2.8	1.2
			70	3	0.9
			52	1.4	1.0
			62	2.4	1.0
			66	3.6	1.3
			58	2.2	1.1
			65	2.6	0.9
			60	2	0.9
			71	4.4	1.2
			62	3	1.3
			62	3.6	1.5
			70	2.8	0.8
		lp	60	2.6	1.2
			60	2.2	1.0
			60	1.6	0.7
			57	2.2	1.2
			68	3.2	1.0
			70	3.8	1.1
			73	3.4	0.9

TABLE D-5 (Page 26 of 29)

SITE	SIZE	LENGTH (mm)	WEIGHT (g)	CONDITION (K)
		69	2.8	0.9
		65	2.6	0.9
		67	3	1.0
		64	3.2	1.2
		70	3.2	0.9
		61	2	0.9
		66	3.1	1.1
		65	2	0.7
		60	2.2	1.0
		62	2.3	1.0
		66	2.8	1.0
		53	1.8	1.2
		70	3.4	1.0
		66	3.2	1.1
		68	4	1.3
		64	2.2	0.8
		64	1.6	0.6
		59	2.4	1.2
		66	3	1.0
		67	3.4	1.1
		66	3.2	1.1
		60	2.2	1.0
		67	2.8	0.9
		72	3.8	1.0
		67	3	1.0
		65	2.6	0.9
		62	3	1.3
		75	4.4	1.0
		60	1.8	0.8
		65	3.6	1.3
		66	2.8	1.0
		66	4	1.4
		67	3.6	1.2
		65	3.2	1.2
		66	3.2	1.1
		60	1.8	0.8
		69	3.8	1.2
		70	4.2	1.2
		69	3.6	1.1
		65	3.2	1.2
		68	3.8	1.2
		46	0.8	0.8
		63	3.2	1.3
		72	4	1.1

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			62	3.6	1.5
			59	2.6	1.3
			68	3.8	1.2
			47	1.2	1.2
			72	5	1.3
			55	1.8	1.1
			64	2.8	1.1
			65	3.8	1.4
			73	4.8	1.2
			58	2.4	1.2
			65	2.8	1.0
			60	2.2	1.0
			60	2	0.9
			67	3.8	1.3
			63	3.6	1.4
			58	3.6	1.8
DC	Lg		91	12.5	1.7
			80	7	1.4
			80	7	1.4
			74	5.5	1.4
			73	5.5	1.4
			71	5.5	1.5
			76	6	1.4
		c	77	7	1.5
			84	8.5	1.4
			78	6.5	1.4
		rp	79	7.5	1.5
			80	6	1.2
			78	8	1.7
			74	5.5	1.4
		c	78	7	1.5
			81	8	1.5
			71	6	1.7
			76	6.5	1.5
			74	6.5	1.6
		c	73	5.5	1.4
			74	6	1.5
			71	5.5	1.5
			78	7	1.5
			78	7	1.5
			82	7	1.3
		lp	72	5	1.3
			71	5.5	1.5
EA	Sm		70	3.8	1.1

TABLE D-5 (Page 28 of 29)

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			71	3.6	1.0
			73	4.4	1.1
			56	3	1.7
			70	4.2	1.2
			70	5	1.5
			45	0.6	0.7
			69	3.6	1.1
			68	5.8	1.8
			66	4	1.4
			76	4.8	1.1
			68	3.8	1.2
			71	4.6	1.3
			63	2.6	1.0
		c	66	3	1.0
			76	5.2	1.2
			46	1	1.0
			74	5.2	1.3
			63	3.6	1.4
			42	1	1.3
			42	1.2	1.6
			43	0.8	1.0
			40	0.6	0.9
			53	1.8	1.2
			41	0.8	1.2
			61	2.4	1.1
			44	1.2	1.4
			41	0.6	0.9
			50	1.4	1.1
			45	1.2	1.3
			41	0.8	1.2
			44	1.2	1.4
			44	1	1.2
			41	0.6	0.9
			38	0.6	1.1
			39	0.6	1.0
			38	0.6	1.1
			42	0.6	0.8
			46	0.6	0.6
EA	Lg		92	12.5	1.6
			81	9	1.7
			76	8	1.8
			79	7	1.4
			81	10	1.9
			73	5.5	1.4

TABLE D-5 (Page 29 of 29)

SITE	SIZE		LENGTH (mm)	WEIGHT (g)	CONDITION (K)
			81	7	1.3
		c	76	7.5	1.7
			68	5	1.6
			74	7.5	1.9
			83	9.5	1.7
			80	7	1.4
			78	7	1.5
			74	5.5	1.4
		c	73	7.5	1.9
			72	7	1.9
			68	7.5	2.4
			66	4.5	1.6
			72	5	1.3

c: caudal fin
lp: left pectoral fin
rp: right pectoral fin

TABLE D-6

SUMMARY OF INDICATOR CHEMICALS IN SEDIMENT SAMPLES[1]

ECOLOGICAL RISK ASSESSMENT

UOP SITE, EAST RUTHERFORD, NEW JERSEY

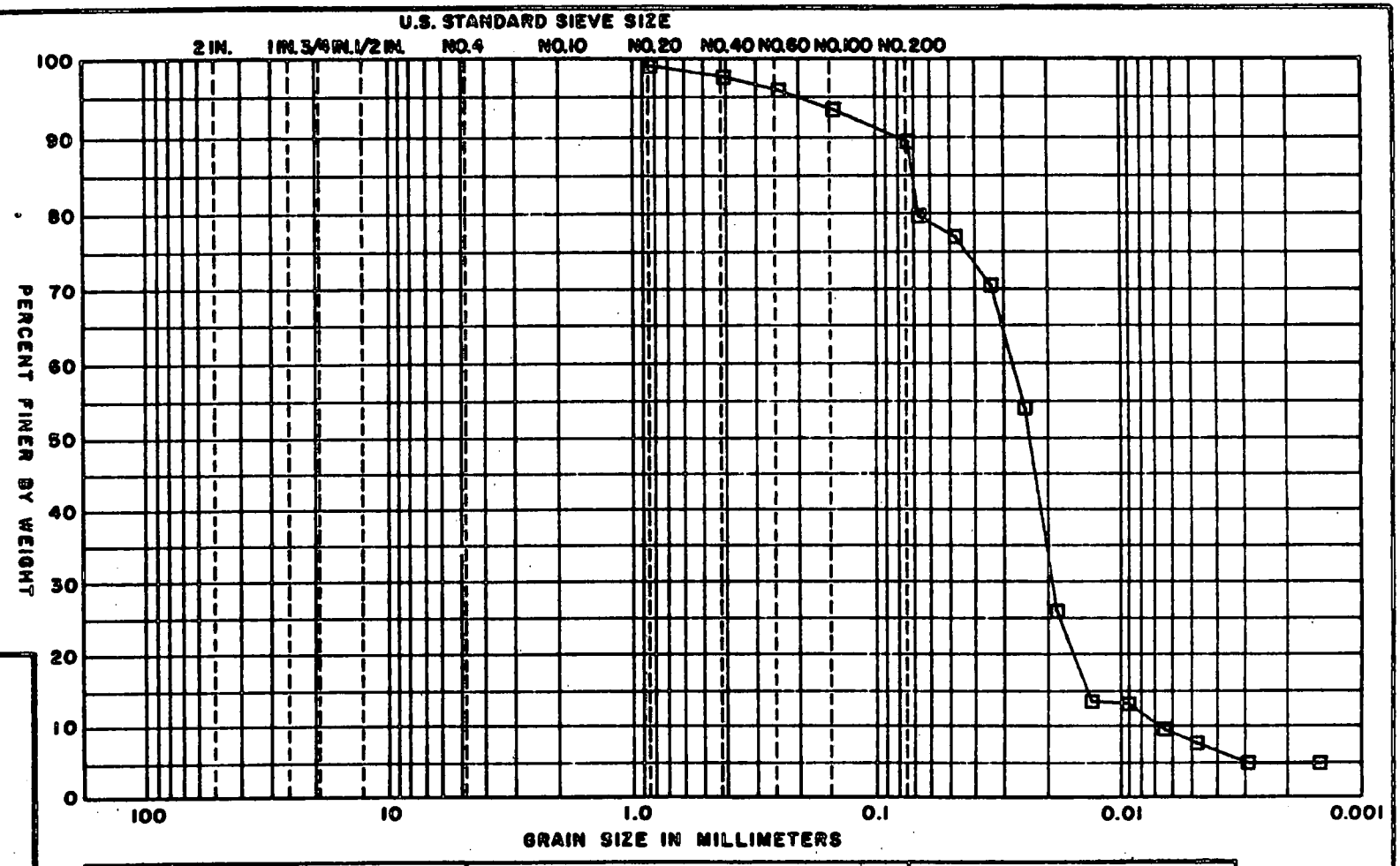
	Constituent Concentration*				Physical Parameters		
Sample Location	Aroclor® 1248*	Chromium Total*	Chromium Hexavalent*	Mercury Total*	pH	Total Organic Carbon*	Percent Solids
Aa	66	4240	60	58	6.53	55,000	25.5
Ab	82	5600	NR[3]	24	6.72	86,000	29.7
Ac	123	26900	NR	108	6.72	23,000	15.9
Ba	100	9010	1.2	33	6.74	49,000	34.9
Bb	41	4110	20	22	6.66	15,00	48.0
Bc	7.7	7400	64	86	6.74	96,000	20.0
Ea[2]	7.4	7890	14	75	6.37	60,000	18.8
Ca	5.3	1120	0.57	5.9	7.44	8,800	39.7
Cb	14	1840	NR	19	7.53	25,000	36.5
Cc	23	6.57	NR	7.75	7.08	50,000	34.4
Da	0.69	1680	1.2	150	4.15	63,000	28.4
Db	1.5	744	NR	58	5.36	70,000	25.9
Dc	1.0	644	NR	60	6.81	61,000	21.7

*mg/kg-dry weight

(1)Includes results for samples received by Cambridge Analytical Associates on November 4, 1988. Results for samples received November 7 and grain size distribution results for samples received November 4 will be reported later.

(2)Duplicate of Bc

(3)Not requested



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
S1.1	□	Sample ID.# 88-11-053-01E Aa	Black Organic SILT, little (-) fine Sand

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH. BH
REVIEWER TW

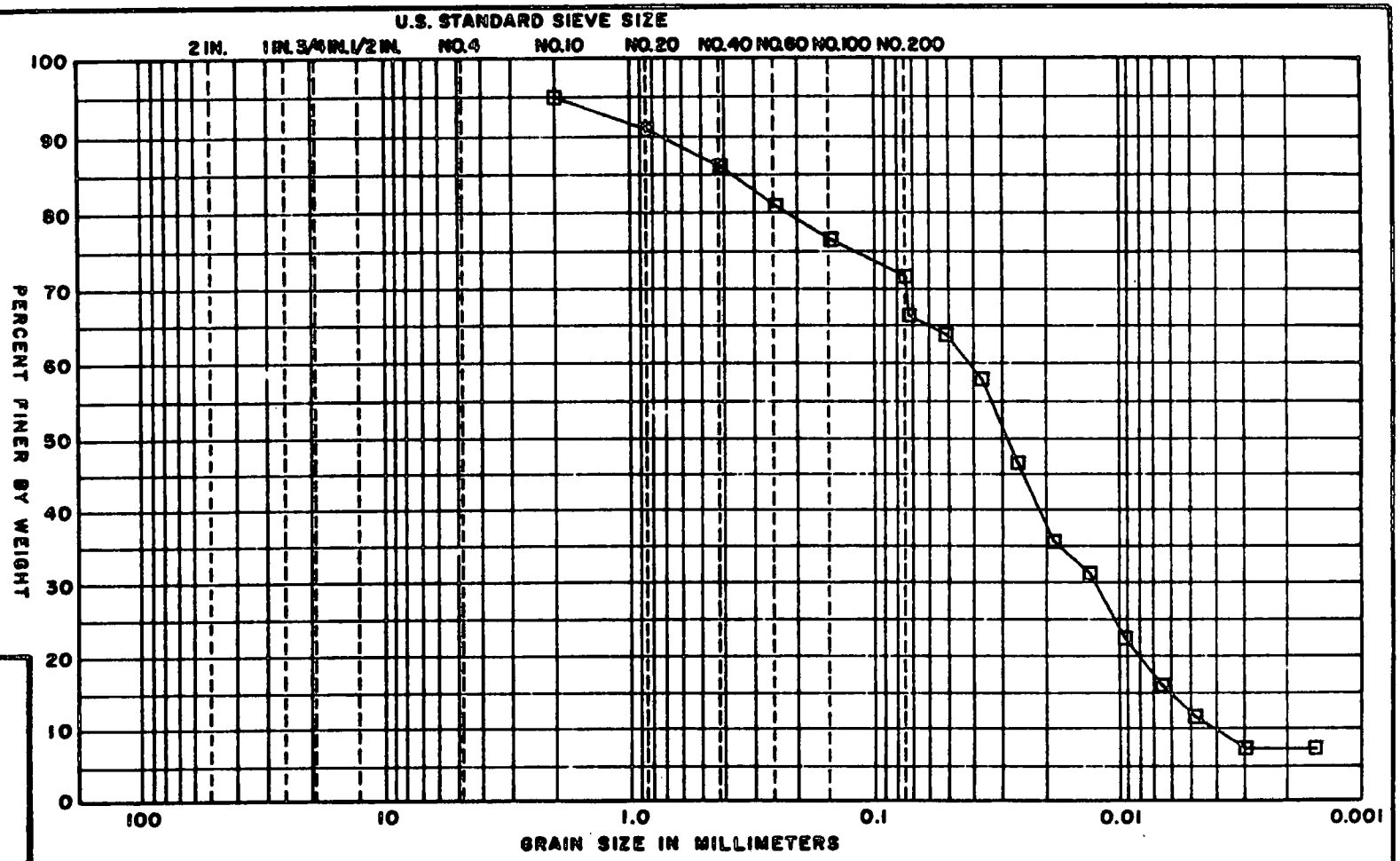
TEST SERIES 1
NO.
DATE Dec. 88
L11185

CAA 0:135.

FILE

Figure D-1

S.3



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
S2.1	□	Sample No. 88-11-053-02E Bc	Black Organic SILT, soss (+) f-m Sand

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH.
REVIEWER

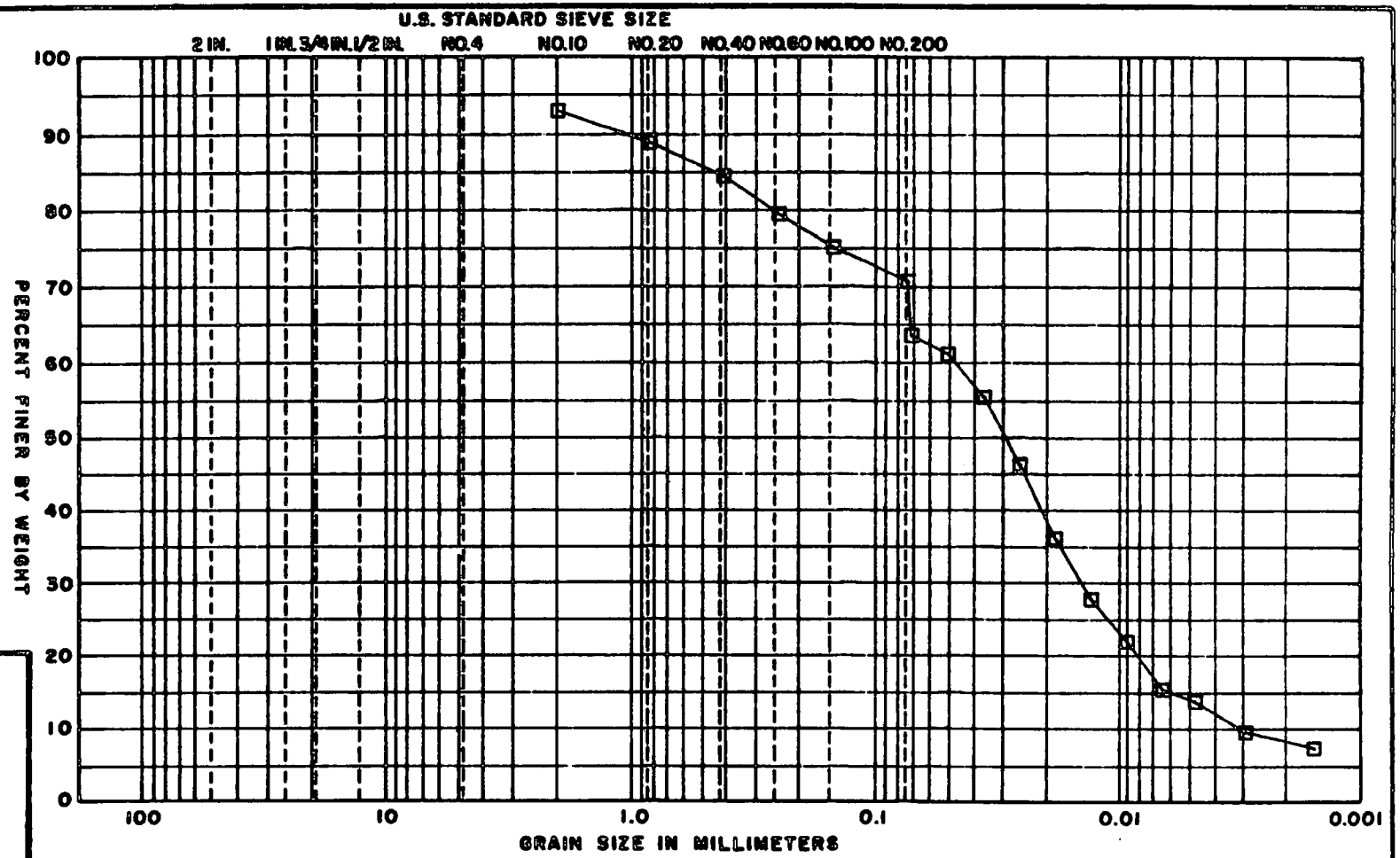
TEST SERIES NO. 2
DATE Dec. 88
L11185

CAA 0:136

FILE

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

Figure D-2



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
S3.1	□	Sample No. 88-11-053-03E Ea	Black Organic SILT and fine SAND

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH. BM
REVIEWER

TEST SERIES NO. 3
DATE Dec. 88
L11185

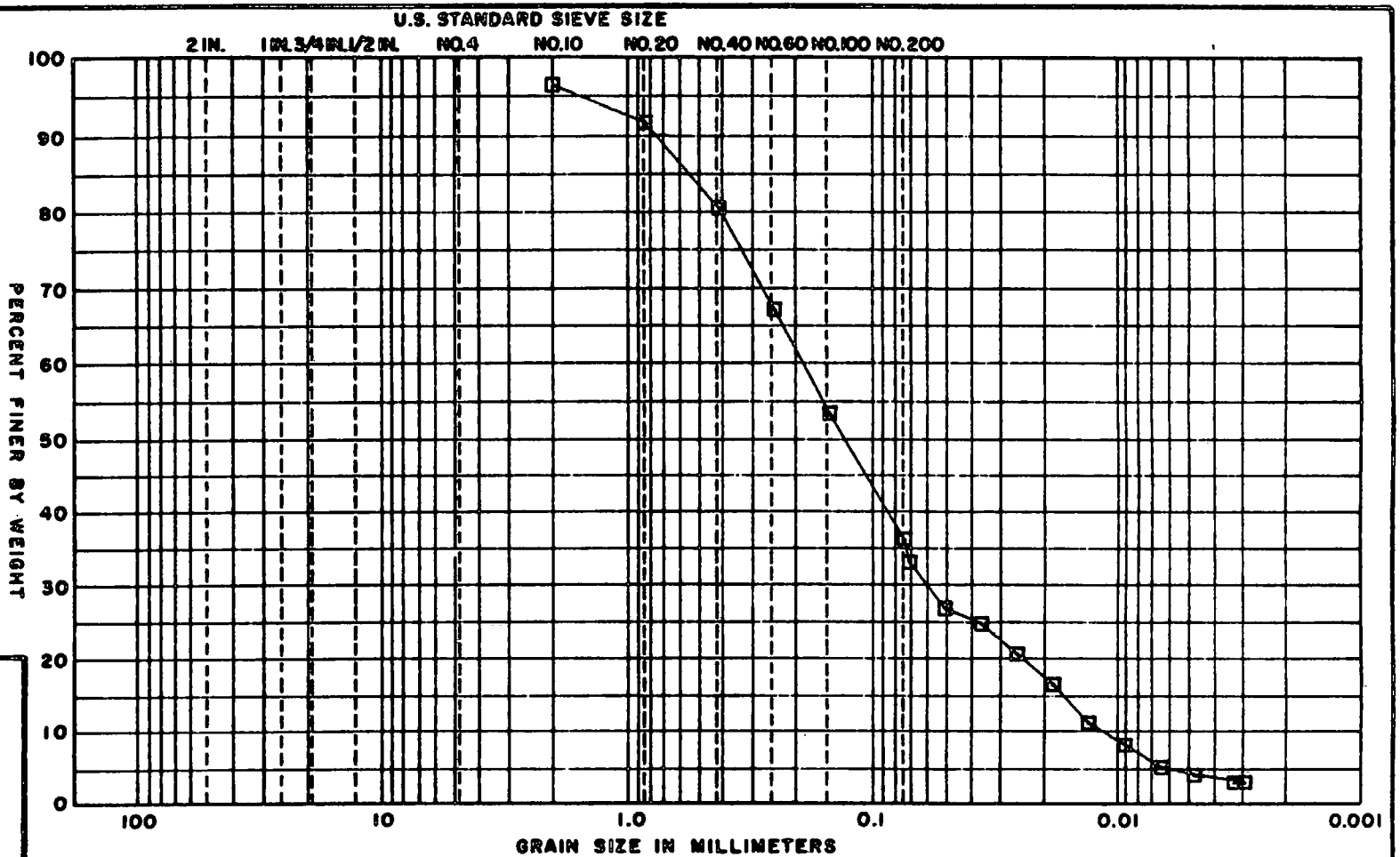
CAA

0.137

FILE

Figure D-3

S.3



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
S4.1	□	Sample ID# 88-11-054-01E Ba	Black f-m SAND, and Organic SILT

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH. BM
REVIEWER TV

TEST SERIES NO. 4
DATE Dec. 88
L11185

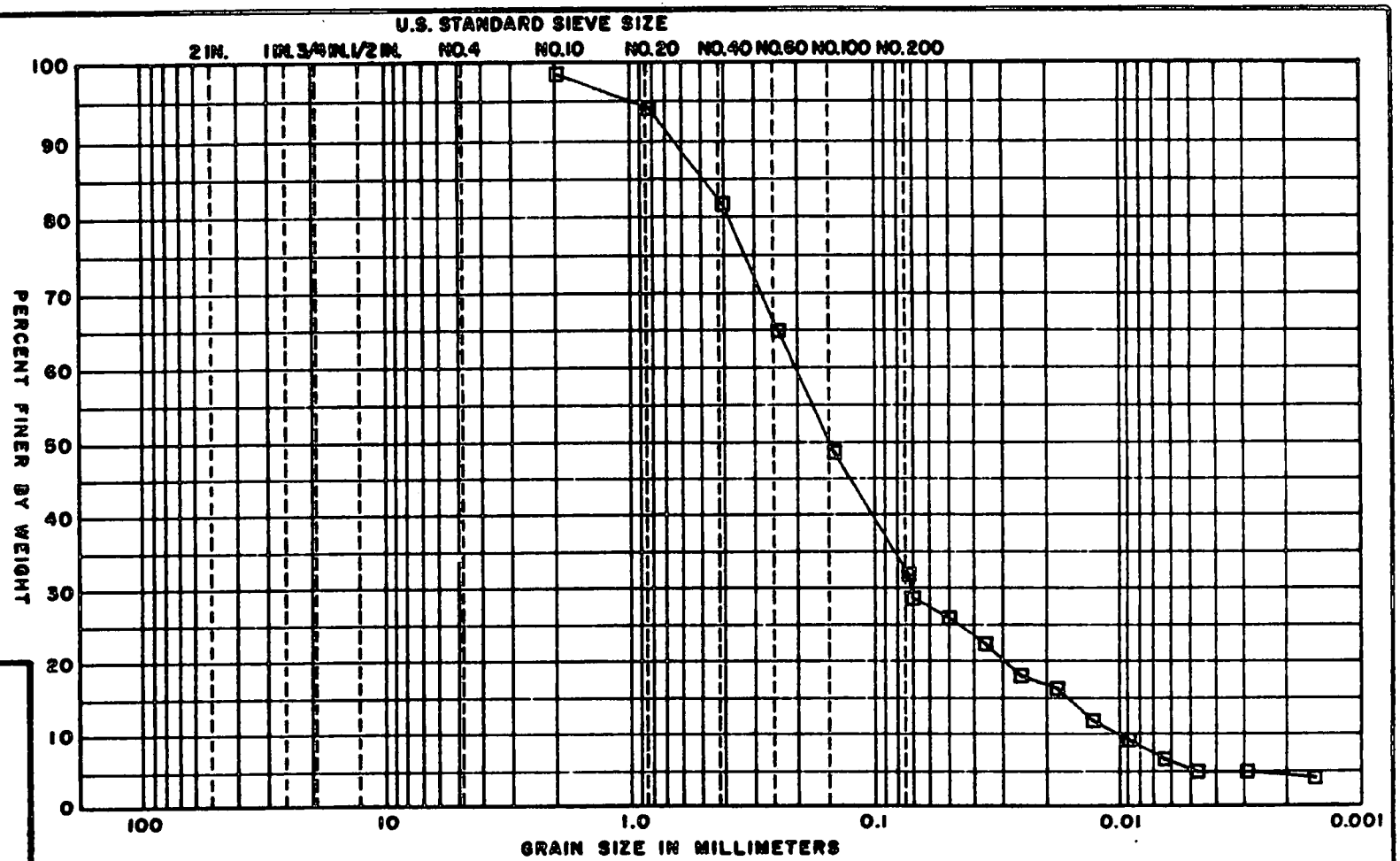
CAA

FILE

Figure D-4

S.3

8 138



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
SS. 1	□	Sample ID# 88-11-054-02E Bb	Black f-m SAND, some (+) Organic Silt

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH. BM
REVIEWER TK

TEST SERIES NO. 3
DATE Dec. 88 L11185

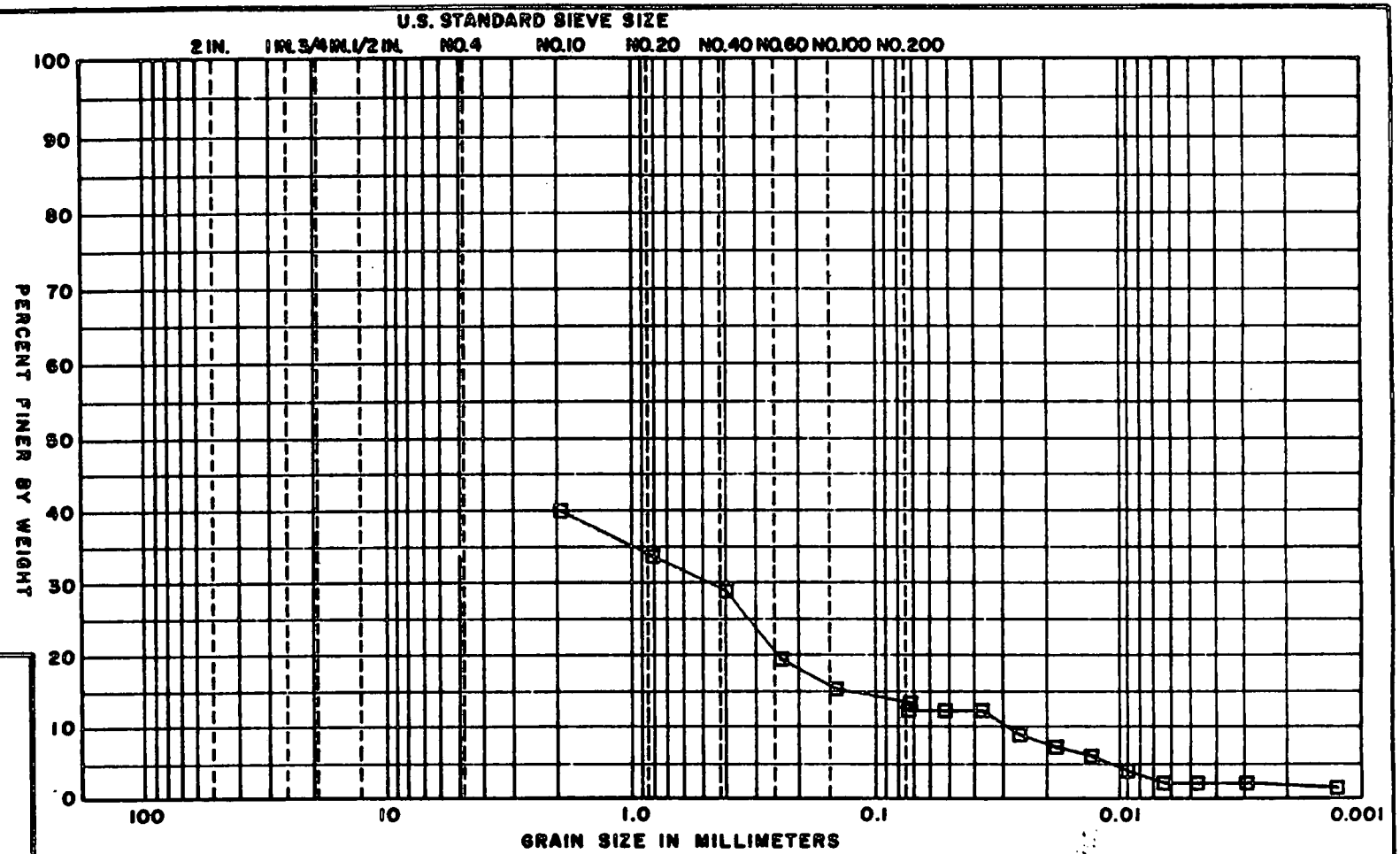
CAA

FILE

0139

Figure D-5

S.3



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
S8.1	□	Sample ID# 88-11-054-03E Ca	Black SHELLS/ shell fragments some f-m Sand, little Org. Silt

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH. BM
REVIEWER TK

TEST SERIES NO. 8
DATE Dec. 88
L11185

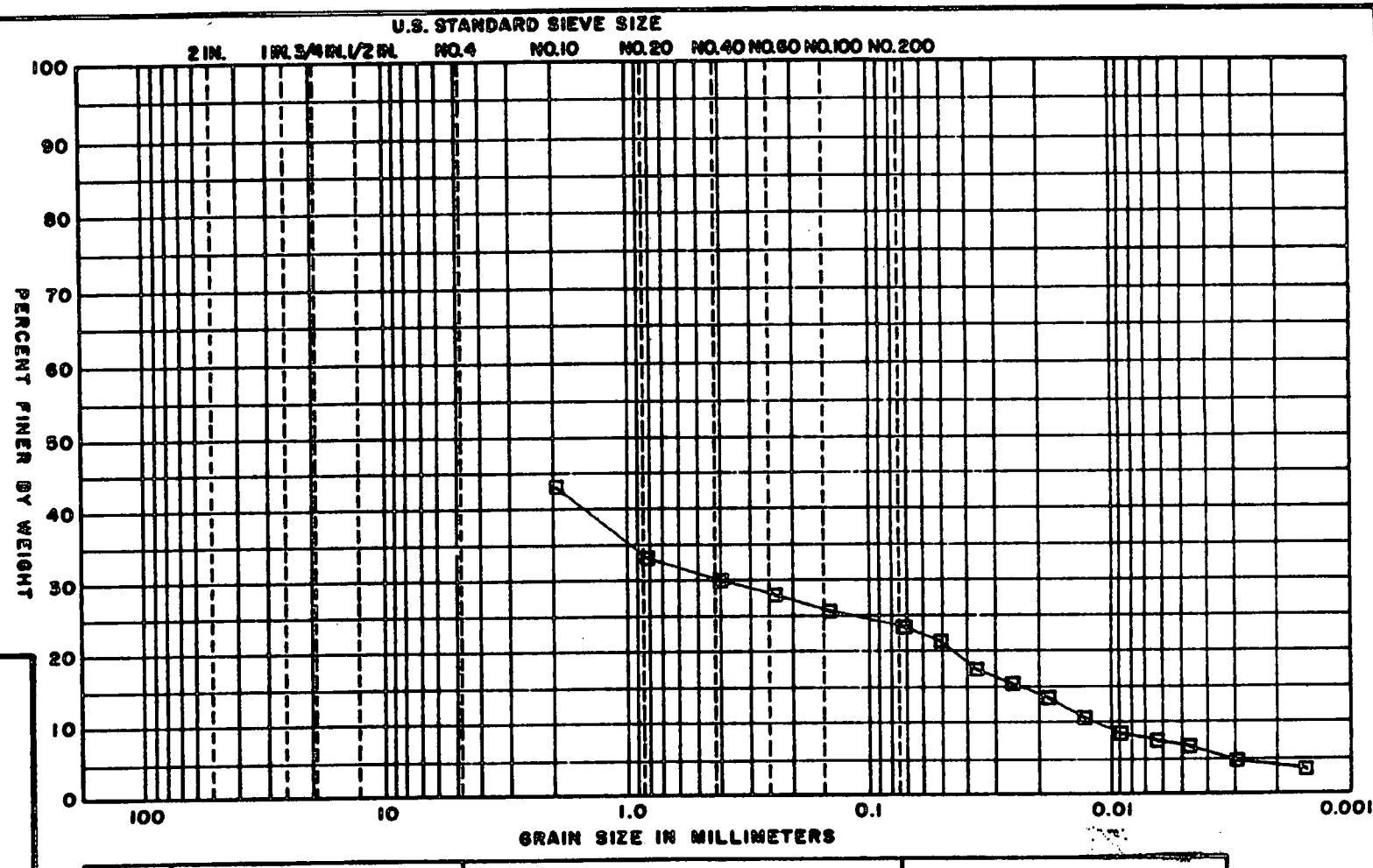
CAA

FILE

Figure D-6

0:140

S.3



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
57.1	□	Sample ID# 88-11-054-04E Cb	Black SHELLS/shell fragments, some Organic Silt, little med. Sand

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH. BH
REVIEWER TK

TEST SERIES NO.
DATE Dec. 88 1.11.89

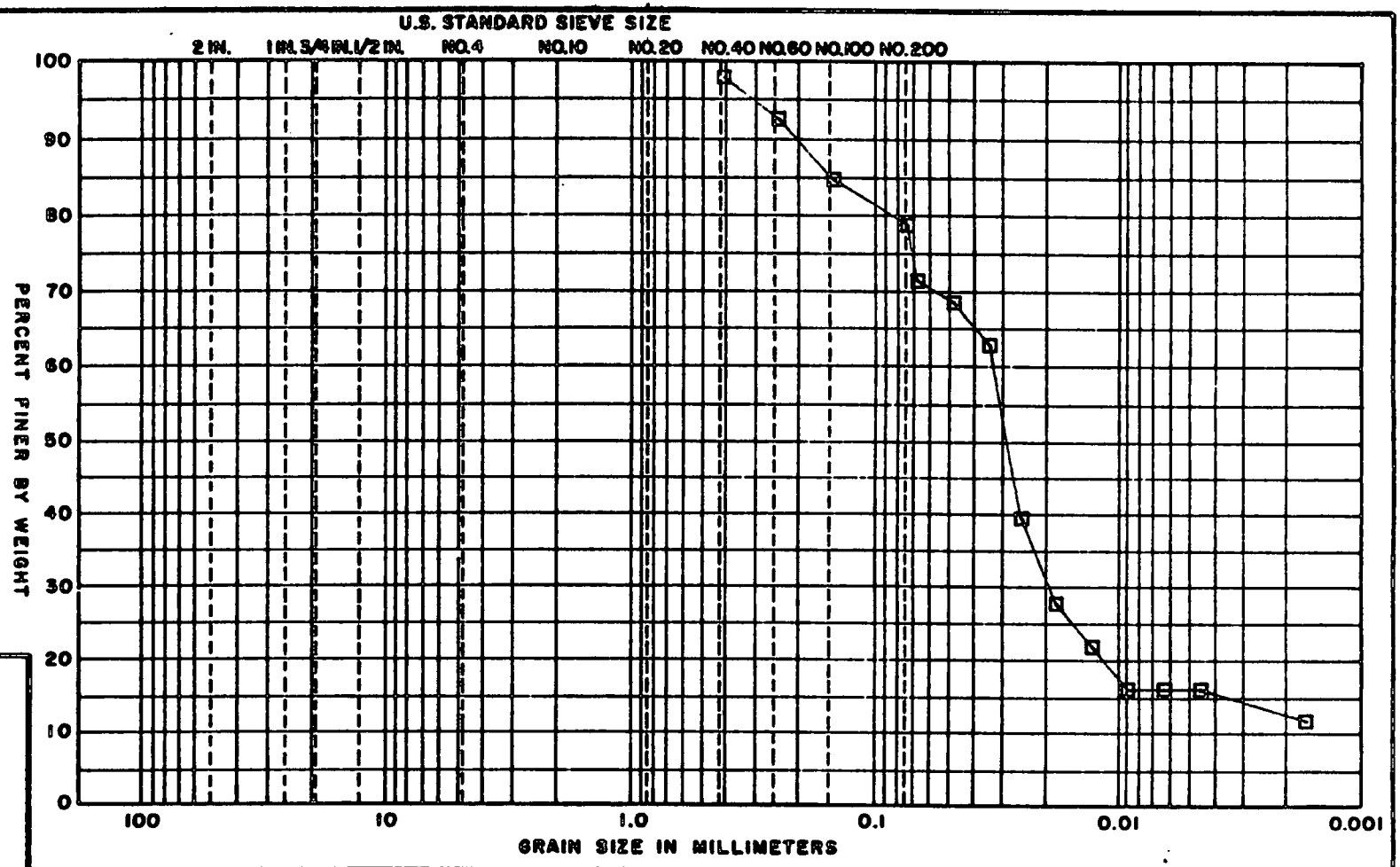
CAA

0.141

FILE

Figure D-7

S.3



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
SB. 1	□	Sample ID# 88-11-054-05E Da	Black Organic SILT, some (-) fine Sand

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH. BM
REVIEWER TK

TEST SERIES 8
NO.
DATE Dec. 88
L11183

CAA

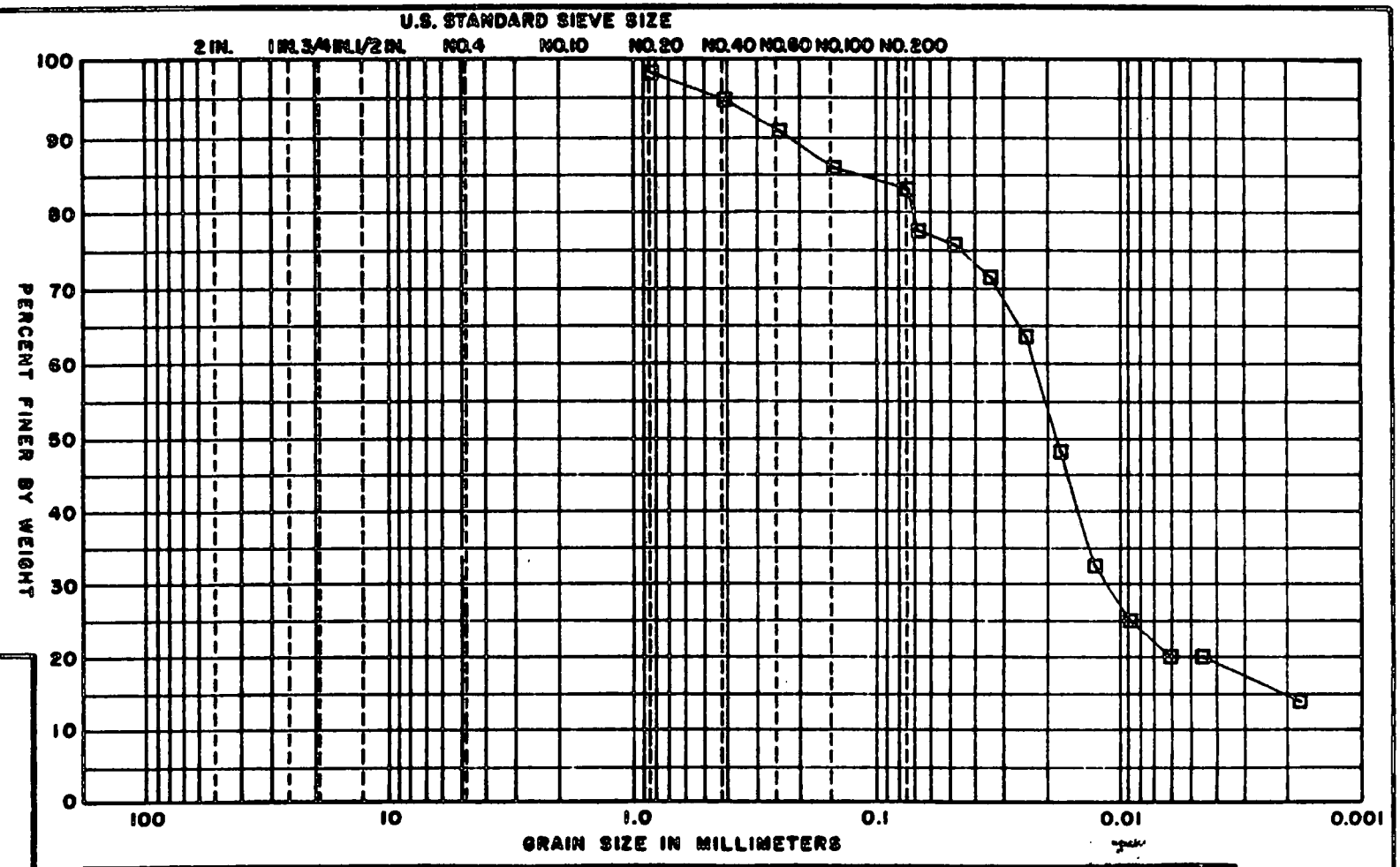
0:142

FILE

Figure D-8

S.3

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
SB. 1	□	Sample ID# 88-11-054-06E Db	Black Organic SILT, little fine Sand

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH. BM
REVIEWER TK

TEST SERIES
NO.
DATE Dec. 88
L11185

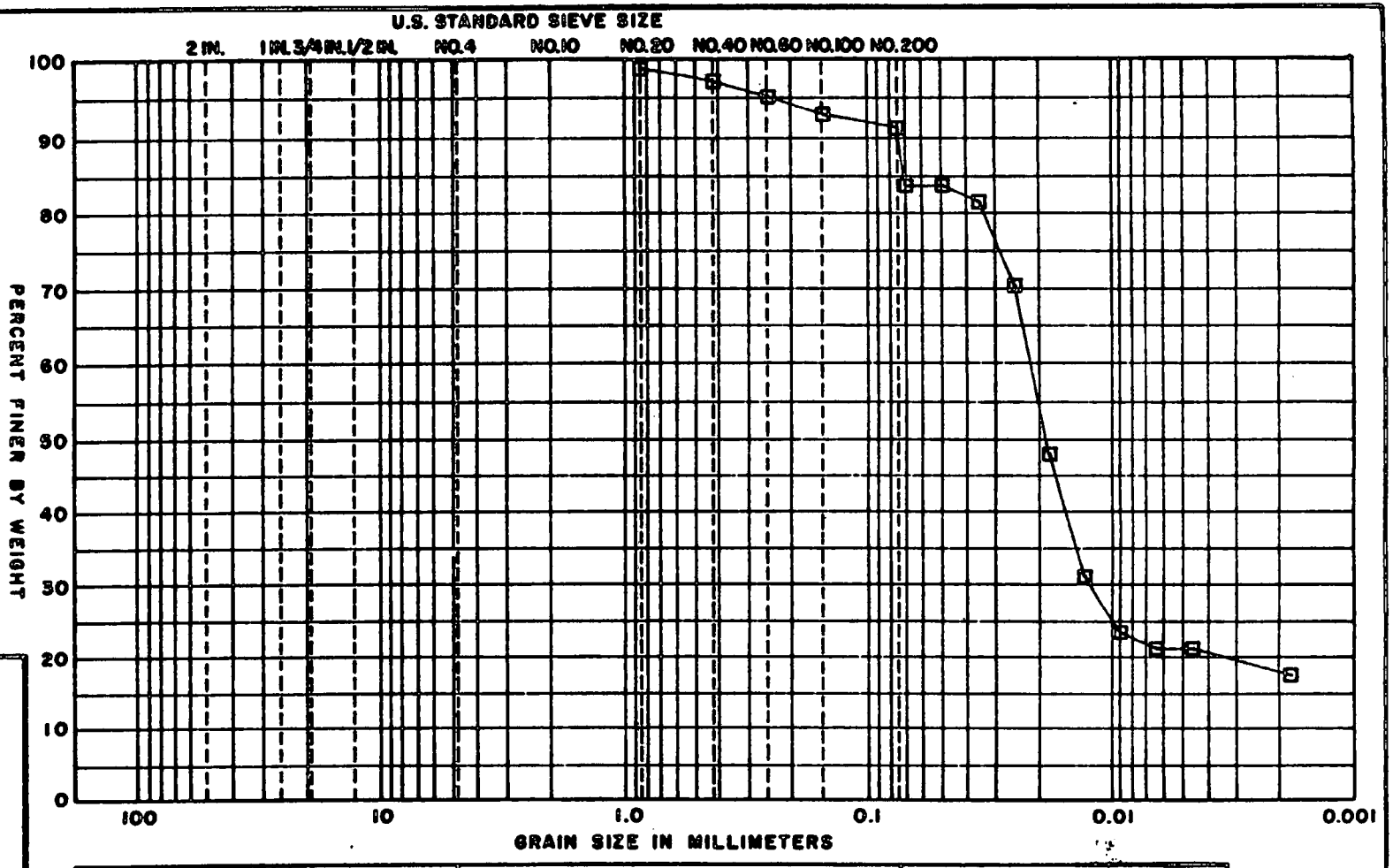
CAA

FILE

Figure D-9

S.3

D-143



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
S10.1	□	Sample ID# 88-11-054-07E Dc	Black Organic SILT, trace fine Sand

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH. BM
REVIEWER TW

TEST SERIES NO. 10
DATE Dec. 88
L11185

FILE

Figure D-10

CAA

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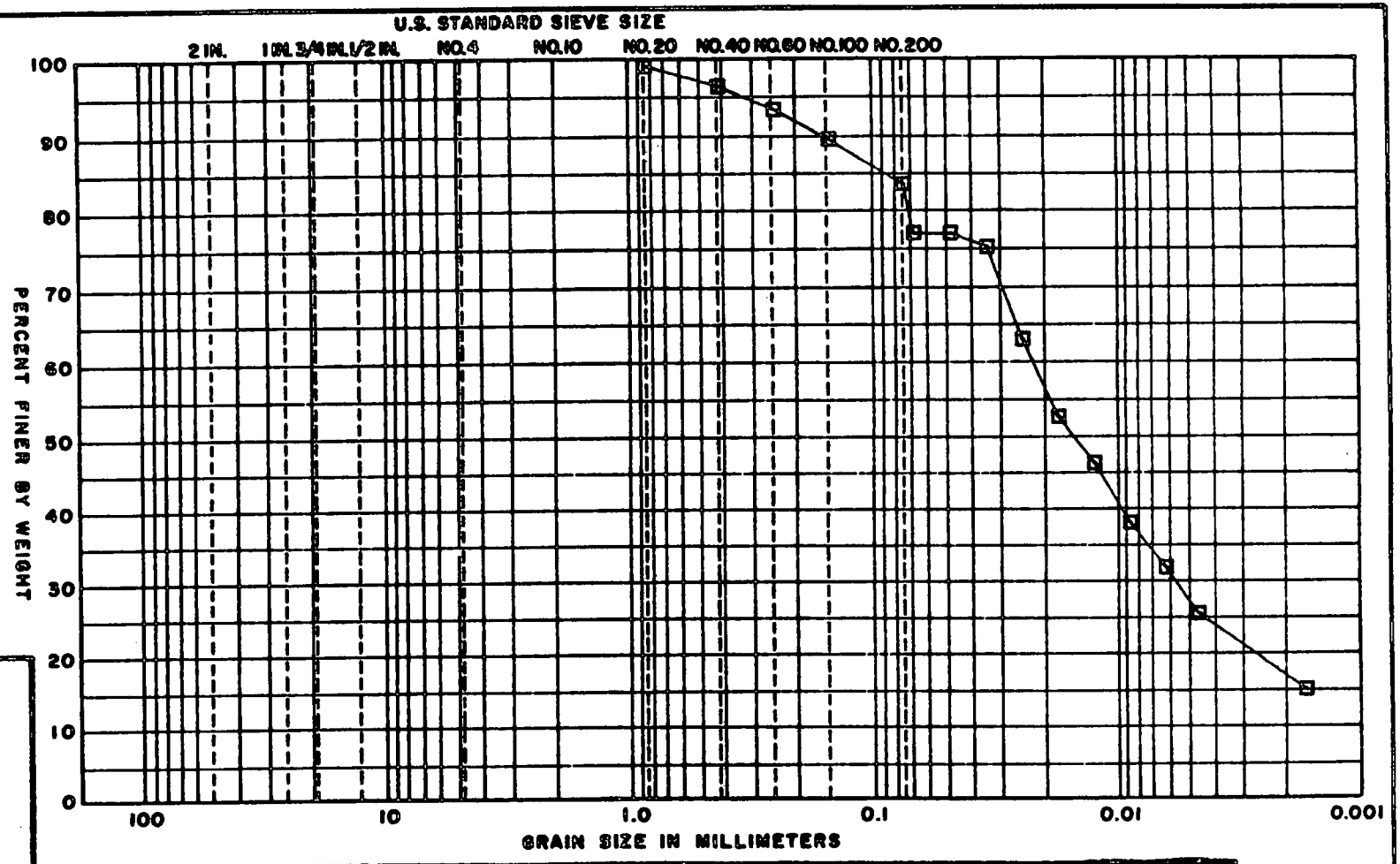
S.3



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
S11.1	□	Sample ID# 88-11-084-80 Ab	Dark Brown Organic SILT, little fine Sand

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH. BM
REVIEWER TW

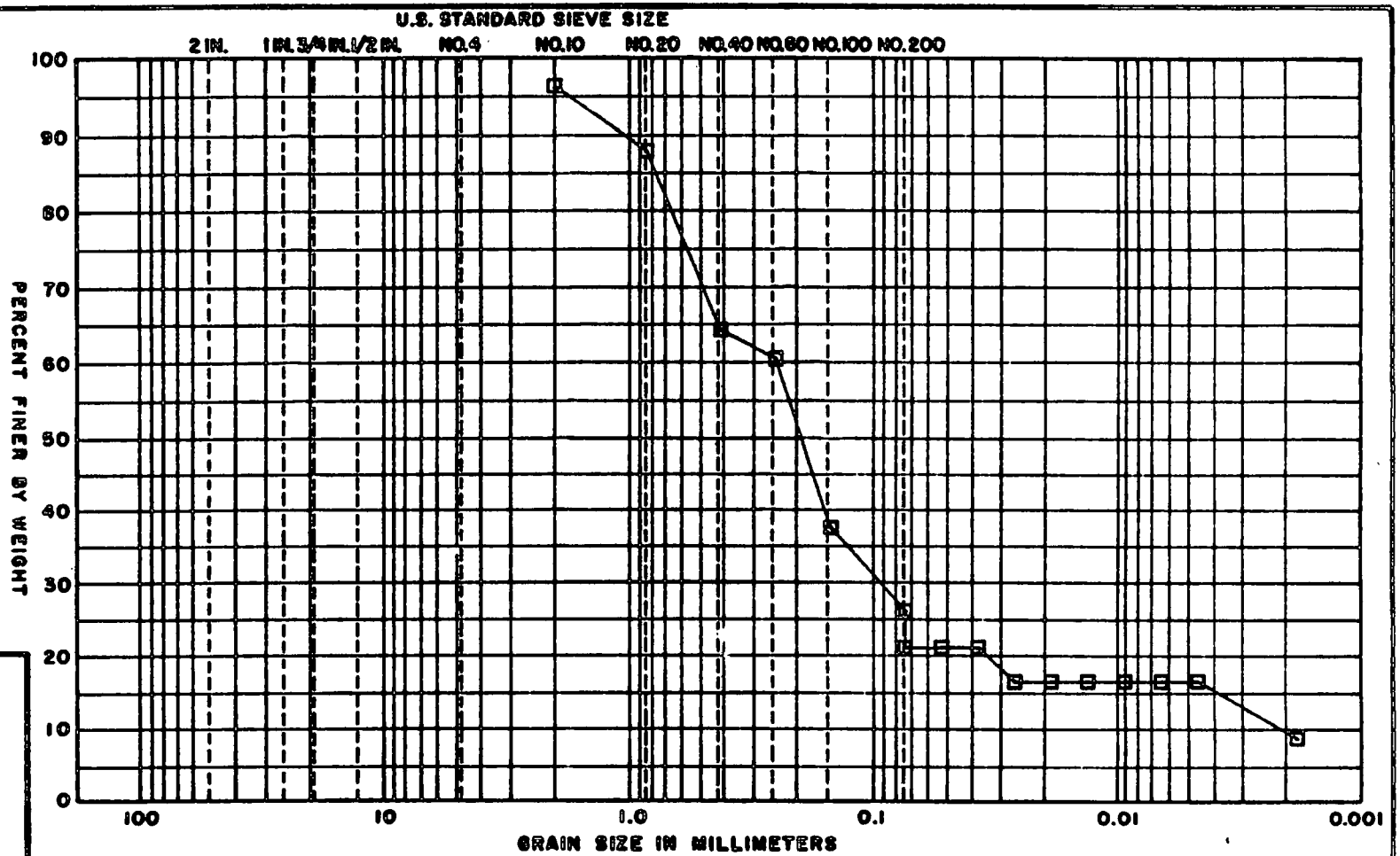
TEST SERIES 11
NO.
DATE Dec. 88
L11185

CAA

FILE

Figure D-11

S.3



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
S12.1	□	Sample ID# 88-11-084-100 Ac	Dark Brown f-m SAND, some (-) Organic Silt

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

GRADATION TESTS

BORING NO.
SAMPLE
DEPTH
TECH. EM
REVIEWER TW

TEST SERIES NO. 12
DATE Dec. 88
L11185

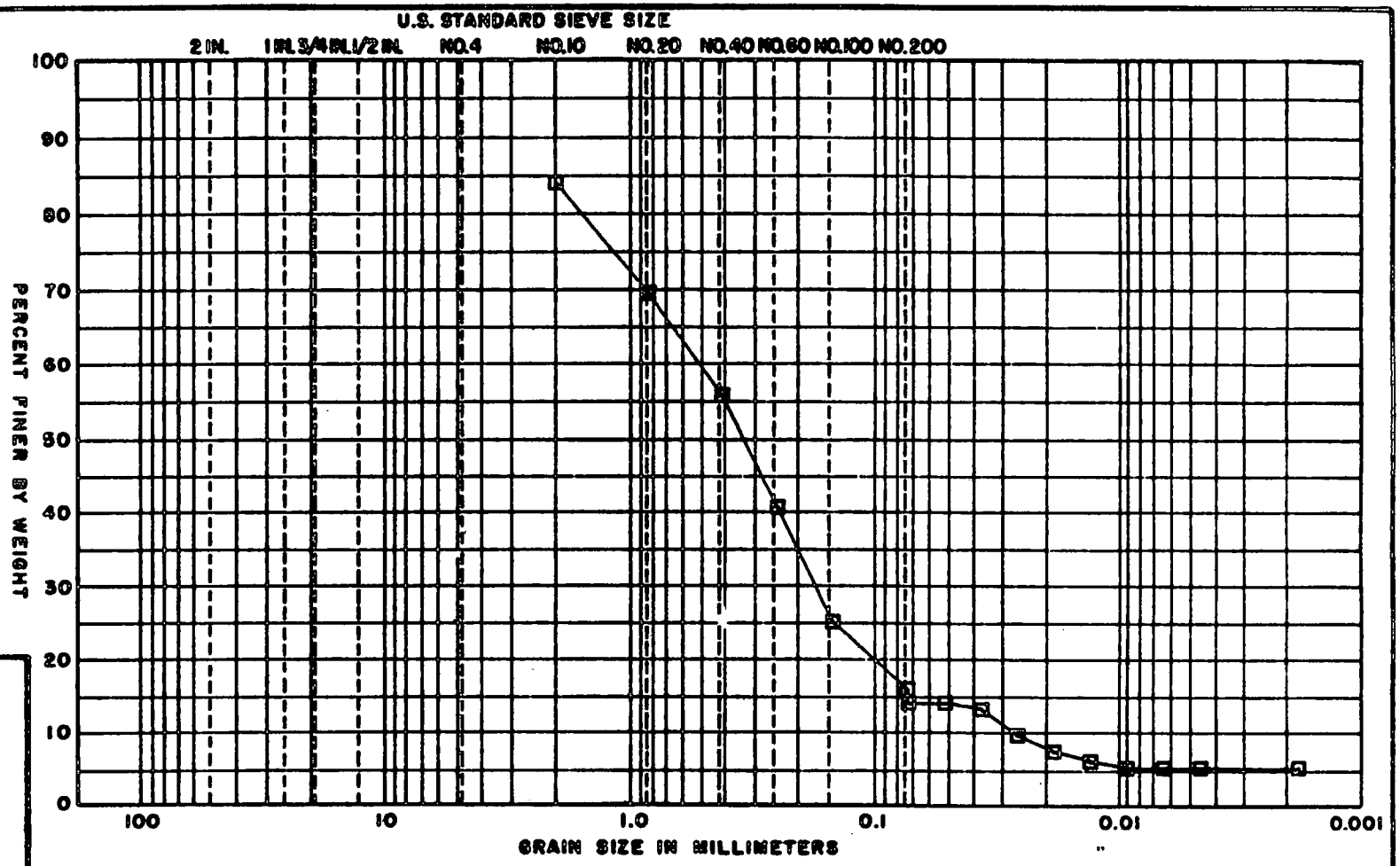
CAA

0.146

FILE

Figure D-12

S.3



COBBLES	GRAVEL		SAND			SILT OR CLAY
	COARSE	FINE	COARSE	MEDIUM	FINE	

UNIFIED SOIL CLASSIFICATION SYSTEM

TEST NO.	SYM.	MATERIAL SOURCE	REMARKS
S13.1	□	Sample ID# 88-11-084-11D Cc	Dark Brown f-m SAND, little Organic Silt

GOLDBERG-ZOINO & ASSOCIATES, INC.
GEOTECHNICAL-GEOHYDROLOGICAL CONSULTANTS

GRADATION TESTS

CAA

BORING NO.
SAMPLE
DEPTH
TECH. BM
REVIEWER "

TEST SERIES NO. 13
DATE Dec. 88
L11165

FILE

Figure D-13

S.3

0147

APPENDIX E
ECOTOXICITY PROFILES

INTRODUCTION

The toxicological profiles in this Appendix describe each Indicator Compound assessed in this report. Each profile discusses the physical and chemical properties, fate and transport mechanisms, ecotoxicological data, and regulations and standards for each chemical. Much of the ecotoxicological information contained in this Appendix comes from the Ambient Water Quality Criteria Documents prepared and published by the U.S. EPA under the Clean Water Act of 1972, as amended. The values cited here are the results of laboratory or field measurements on various test species. For those chemicals for which there is sufficient data, the U.S. EPA has adopted Ambient Water Quality Criteria (AWQC) for acute and chronic toxicity in fresh water and salt water environments. These AWQC values are not enforceable but are intended to provide scientific data and guidance on the environmental effects of chemical pollutants.

CADMIUM AND COMPOUNDS
(Inorganic)

Background Information

Cadmium is generally found in conjunction with zinc and lead ores. It typically exists in the environment as a salt of the +2 valence state or as a metal; it forms no stable organic compounds. Different cadmium salts have different water solubilities, with the oxide of cadmium being less soluble than the chloride. The abundance in the earth's crust is approximately 0.2 mg/kg. Cadmium releases are generally associated with mining, smelting, manufacturing operations, and from the disposal of alkaline batteries containing cadmium (USEPA 1981).

Chemical and Physical Properties

Property	Value	Reference
Chemical formula	Cd	Clement, 1985
Molecular weight	112.41	Clement, 1985
Boiling point	765°C	Clement, 1985
Melting point	321°C	Clement, 1985
Water solubility	insoluble	ATSDR, 1987
Vapor pressure	0.00	USEPA, 1986
Henry's Law constant	NA	USEPA, 1986
Koc	NA	
Log_{10} Kow	NA	
Kd	6.7	Baes and Sharp, 1983

Fate and Transport

Compared to other heavy metals, cadmium is relatively mobile in the aquatic surface water environment and may be

transported in solution or as organic or inorganic complexes. Sorption processes account for the removal of dissolved cadmium to bed sediments. Cadmium may also be removed from aqueous solutions by adsorption onto mineral surfaces and co-precipitation with hydrous metal oxides. Photolysis is not an important factor in determining the fate of cadmium compounds in water, and cadmium does not form volatile compounds in the aqueous environment.

Cadmium is strongly accumulated by both salt and fresh water organisms at all trophic levels. Cadmium can accumulate in tissues at levels hundreds to thousands times higher than those in the water column as shown in Table E-1. Cadmium uptake increases with increasing water temperature and decreasing salinity.

Ecotoxicology

The ecotoxicology data in this section are reported in the U.S. EPA Ambient Water Quality Criteria document on cadmium (U.S. EPA, 1984).

In general, salt water species of animals are more resistant to cadmium toxicity than are fresh water species. Among the salt water species the oligochaete worm (*Monophylephorus cuticalatus*) was the most resistant species while the fathead minnow (*Pimephales promelas*) was the most resistant fresh water species (Table E-2).

There is less information available on the chronic toxicity of cadmium. *Daphnia magna* was the most sensitive species tested, with chronic toxicity found at 0.1523 ug/L with a water hardness of 53 mg/L $CaCO_3$ (Table E-3). Insects and other invertebrates are more sensitive during molting, which usually does not occur in 96-hour tests.

While there is not an EC_{50} or LC_{50} for cadmium for the mallard (*Anas platyrhynchos*), a concentration of 200 mg/kg cadmium in food for 90 days is the maximum permissable limit to prevent adverse effects on reproduction and kidney function.

Regulations and Standards

To protect freshwater aquatic life, the four-day average concentration (in µg/L) of cadmium should not exceed the numerical valve given be $e^{(0.7852\ [\ln\ (\text{hardness})]\ -\ 3.490)}$ more than once every three years on the average and the one-hour average concentration (in µg/L) of cadmium should not exceed the numerical valve given by $e^{(1.128\ [\ln\ (\text{hardness})]\ -\ 3.828)}$ more than once every three years on the average. Therefore, at hardnesses of 50, 100, and 200 mg/L as $CaCO_3$, the four-day average concentrations of cadmium are 0.66, 1.1, and 2.0 µg/L, respectively, and the one-hour average concentrations are 1.8, 3.9, and 8.6 µg/L, respectively.

To protect salt water aquatic life, the four-day average concentration of cadmium should not exceed 9.3 ug/L more than once every three years on the average. The one-hour average concentration should not exceed 43 µg/L more than once every three years on the average.

TABLE E-1

BIOCONCENTRATION FACTORS FOR CADMIUM

Species	Tissue	Chemical	Duration (days)	Bioconcentration* Factor	Reference
FRESH WATER SPECIES					
Cladoceran, *Daphnia magna*	whole body	cadmium sulfate	2-4	320	Poldoski, 1979
Midge, Chironomidae	whole body	cadmium chloride	365	2220	Giesy et al., 1979
SALT WATER SPECIES					
Blue mussel, *Mytilus edulis*	soft parts	cadmium chloride	35	306	Phillips, 1976
Grass shrimp, *Palaemonetes vulgaris*	whole body	cadmium chloride	28	307	Nimmo et al., 1977b

*Results are based on cadmium, not the chemical compound.

TABLE E-2

ACUTE TOXICITY OF CADMIUM TO AQUATIC ORGANISMS

Species	Method	Chemical	LC_{50} or EC_{50} (µg/L)	Reference
FRESH WATER SPECIES				
Cladoceran, *Daphnia magna*	S,U	cadmium chloride	1.6	Anderson, 1948
Midge, *Chironomus sp.*	S,U	not specified	1,200	Rehwoldt et al., 1973
Banded killifish, *Fundulus diaphanus*	S,M	not specified	110	Rehwoldt et al., 1972
Fathead minnow *Pimephales promelas*	S,U	cadmium chloride	73,500	Pickering & Henderson, 1966
SALT WATER SPECIES				
Grass shrimp *Palaemonetes vulgaris*	S,U	cadmium chloride	420	Eisler, 1971
Grass shrimp *Palaemonetes vulgaris*	FT,M	cadmium chloride	760	Nimmo et al., 1977b
Blue crab (juveniles) *Callinectes sapidus*	S,U	cadmium chloride	4,700	Frank and Robertson, 1979
Blue mussel (embryo) *Mytilus edulis*	S,U	cadmium chloride	1,200	Martin, et al. 1981
Striped killifish (adult) *Fundulus majalis*	S,U	cadmium chloride	21,000	Eisler, 1971
Oligochaete worm *Monophylephorus cuticalatus*	R,U	cadmium sulfate	135,000	Chapman et al., 1982a

S = static M = measured
FT = flow through U = unmeasured
R = renewal

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TABLE E-3

CHRONIC TOXICITY OF CADMIUM TO AQUATIC ORGANISMS

Species	Chemical	Test	Limits* (µg/L)	Chronic Value (µg/L)	Reference
FRESH WATER SPECIES					
Cladoceran, *Daphnia magna*	Cadmium chloride	LC	0.08-0.29	0.15	Chapman et al. manuscript
Fathead minnow, *Pimephales promelas*	cadmium sulfate	LC	37-57	45.92	Pickering and Gast, 1972
SALT WATER SPECIES					
Mysid shrimp, *Mysidopsis bahia*	cadmium chloride	LC	6.4-10.6	8.2	Nimmo et al., 1977a
Mysid shrimp, *Mysidopsis bahia*	cadmium chloride	LC	5.1-10	7.1	Gentile et al., 1982; Lussier et al., manuscript

*Results are expressed as cadmium, not the chemical compound.
LC = Life cycle or partial life cycle

CHROMIUM

Background Information

Chromium is a heavy metal that generally exists in either a trivalent or hexavalent oxidation state which is found in low concentrations in nature systems. Problematic high levels are usually traceable to some type of manufacturing process. Chromium salts are used extensively in metal finishing as electroplating and cleaning agents, and as mordants in the textile industry. Chromium has also been extensively used in the tanning industry. Chromium is of nutritional importance to humans and animals in trace amounts.

Chemical and Physical Properties

Property	Value	Reference
Chemical formula	Cr	Clements, 1985
Molecular weight	52	USEPA, 1986
Boiling point	2675°C	Clements, 1985
Melting point	1857±20°C	Clements, 1985
Water solubility	Insoluble (Some compounds are)	Clements, 1985
Vapor pressure	0.001 mmHg	USEPA, 1986
Henry's Law constant	NA	USEPA, 1986
Koc	NA	USEPA, 1986
Log_{10} Kow	NA	USEPA, 1986
Kd	value = 37	Baes and Sharp, 1983

Fate and Transport

Hexavalent chromium (Cr VI) is soluble and quite mobile in ground water (Clements, 1985). In the presence of reducing agents, hexavalent chromium converts rapidly to trivalent chromium (Cr III) (USEPA, 1980). Hexavalent chromium is a

strong reducing agent under acidic conditions, however, it is relatively stable in most natural waters (USEPA, 1980). Trivalent chromium tends to form stable complexes with negatively charged organic and inorganic species; therefore, its toxicity and solubility vary with water hardness and alkalinity (USEPA, 1980). Hexavalent chromium is rapidly converted to trivalent chromium in the presence of large quantities of organic matter (Kabata-Pendias and Pendias, 1984). Soluble hexavalent chromium in soil is highly toxic to plants and animals.

Ecological Toxicology

The ecotoxicology data reported in Tables E-4 and E-5 were taken from the U.S. EPA Ambient Water Quality Criteria document on Chromium (U.S. EPA, 1984).

The acute toxicity of hexavalent chromium decreases as hardness and pH increase. The toxicity of trivalent chromium, in fresh water systems, is also dependent on hardness in the water. Trivalent chromium is more toxic in soft water.

Regulations and Standards

The federal Ambient Water Criteria (AWQC) for hexavalent chromium are as follows:

Fresh water (µg/L)		Salt water (µg/L)	
acute	chronic	acute	chronic
16	11	1,100	50

The federal Ambient Water Quality Criteria (AWQC) for trivalent chromium are as follows:

TABLE E-4

MEAN ACUTE VALUES FOR CHROMIUM

Hexavalent Chromium

Species	Mean Acute Value (LC_{50}) (µg/L)	Reference
FRESH WATER SPECIES		
Cladoceran *Daphnia magna*	3,490	Dowden and Bennett, 1965
Scud *Gammarus pseudolimnaeus*	67.1	Call et al. 1983
Midge *Tanytarsus dissimilis*	57,300	Call et al. 1983
Rainbow Trout *Salmo gairdneri*	69,000	Benoit, 1976
Fathead Minnow *Pimephales promelas*	17,600	Pickering and Henderson, 1966
SALT WATER SPECIES		
Brackish water clam *Rangia cuneata*	14,000	Olson and Harrel, 1973
Mysid shrimp *Mysidopsis bahia*	2,030	Lussier, et al., ms
Blue crab *Callinectes sapidus*	89,000	Frank and Robertson, 1979
Mummichog *Fundulus heteroclitus*	91,000	Eisler and Henneley, 1977

Trivalent Chromium

FRESH WATER SPECIES		
Cladoceran *Daphnia magna*	16,800	Chapman et al., ms

TABLE E-4 (Continued)

MEAN ACUTE VALUES FOR CHROMIUM

Trivalent Chromium

Species	Mean Acute Value (LC_{50}) (µg/L)	Reference
Scud *Gammarus sp.*	3,200	Rehwoldt et al., 1973
Fathead minnow *Pimephales promelas*	5,070	Pickering and Henderson, 1966
Bluegill *Lepomis macrochirus*	7,460 (hardness 20 as mg/L $CaCO_3$) 71,900 (hardness 360 as mg/L $CaCO_3$)	Pickering and Henderson, 1966
SALT WATER SPECIES		
American oyster *Crassostrea virginica*	10,300	Calabrese, 1973
Mummichog *Fundulus heteroclitus*	31,500	Dorfman, 1977

TABLE E-5

CHRONIC VALUES FOR CHROMIUM

LIFE CYCLE TESTS

Hexavalent Chromium

Species	Mean Acute Value (LC_{50}) (µg/L)	Reference
FRESH WATER SPECIES		
Fathead minnow *Pimephales promelas*	1,987	Pickering, 1980
Rainbow trout, *Salmo gairdneri*	73.18	Sauter et al., 1976
Brook trout, *Salvelinus fontinalis*	264.6	Benoit, 1976
SALT WATER SPECIES		
Polychaete worm *Neanthes arenaceodentata*	25.18	Oshida and Word, 1982
Mysid shrimp *Mysidopsis bahia*	132	Lussier et al., ms

Trivalent Chromium

FRESH WATER SPECIES		
Cladoceran *Daphnia magna*	66.11	Chapman, et al., manuscript
Fathead minnow *Pimephales promelas*	1,025	Pickering, ms
SALT WATER SPECIES		
Polychaete worm *Neanthes arenaceodentata*	>50,400	Oshida et al., 1976, 1981

Fresh water (µg/L)

acute	chronic
1,700	210

The lowest observed effect level for trivalent chromium in salt water is 10,300 µg/L for acute exposure.

1,2-DICHLOROBENZENE

Background Information

There are three structural isomers of dichlorobenzene; the focus of this profile is on 1,2-dichlorobenzene (1,2-DCB). 1,2-DCB is used primarily as a process solvent in the production of toluene disiocyanate, and in the manufacturing of dyes, herbicides, and degreasers (USEPA, 1980).

Chemical and Physical Properties

Property	Value	Reference
Chemical formula	$C_6H_{10}CL_2$	Verschueren, 1983
Molecular weight	147.01	Verschueren, 1983
Boiling point	179°C	Verschueren, 1983
Melting point	-16.7/-18°C	Verschueren, 1983
Water solubility	100mg/L at 20-30°C	USEPA, 1986
Vapor pressure	1 mmHg at 20°C	Verschueren, 1983
Henry's Law constant	1.93E-03atm-m^3/mol	USEPA, 1986
Koc	1700 mL/g	USEPA, 1986
Log_{10} Kow	3.60	USEPA, 1986

Fate and Transport

The predominant aquatic fate of 1,2-dichlorobenzene has not been determined. There is evidence that this compound is very persistent and will probably be degraded slowly by microorganisms already growing on another hydrocarbon source.

1,2-Dichlorobenzene is resistant to auto-oxidation by the peroxy radical (RO_2) in water. The compound does not easily undergo hydrolysis in water at an appreciable rate under environmental conditions. 1,2-Dichlorobenzene has a low water solubility and is volatile. Its half-life in water is 1.5 to 8.5 days. Although it is volatile, 1,2-dichlorobenzene can

also be adsorbed by sedimented organic material; the extent to which this sorption interferes with volatilization was not discussed in the available literature. As 1,2-dichlorobenzene also has a high affinity for lipids, sorption, bioaccumulation, and volatilization are expected to be competing processes.

Ecotoxicology

The ecotoxicology data in this section are reported in the U.S. EPA Ambient Water Quality Criteria document on dichlorobenzene (U.S. EPA, 1980). Rainbow trout (*Salmo gairdneri*) is the more sensitive species of the four fresh water species tested (Table E-6). Among the three salt water species tested, the mysid shrimp *Mysidopsis bahia* was the most sensitive, followed by the tidewater silverside *Medidia beryllina* and finally the sheepshead minnow *Cyprinodon variegatus* (Table E-6). Only the fathead minnow (*Pimephales promelas*) was tested under chronic conditions (Table E-7). The test was an embryo/larvae test and the reported value is 2,000 µg/L.

The available data for 1,2-dichlorobenzene indicate that acute and chronic toxicity to fresh water aquatic animal species occur at concentrations of 1,580 µg/L (*S. gairdneri*) and 2,000 µg/L (*P. promelas*), respectively. Toxicity to salt water animal species occur at concentrations of 1,970 µg/L (*M. bahia*). Toxicity could occur at lower concentrations among species more sensitive than those tested.

Regulations and Standards

There are no federal Ambient Water Quality Criteria (AWQC) for dichlorobenzene. The lowest observed effect level for fresh water is 1,120 µg/L for acute exposure and 763 µg/L for chronic exposure. The lowest observed effect level for salt water is 1,970 µg/L for acute exposure.

TABLE E-6

ACUTE TOXICITY OF 1,2-DICHLOROBENZENE TO AQUATIC LIFE

Species	Method*	Chemical	LC_{50} or EC_{50} (µg/L)	References
FRESH WATER SPECIES				
Midge, *Tanytarsus dissimilis*	S,M	1,2-DCB	11,760	U.S. EPA, 1980
Cladoceran, *Daphnia magna*	S,U	1,2-DCB	2,440	U.S. EPA, 1978
Rainbow trout *Salmo gairdneri*	FT,M	1,2-DCB	1,580	U.S. EPA, 1980
Bluegill *Lepomis macrochirus*	S,U	1,2-DCB	27,000	Dawson et al., 1977
Bluegill *Lepomis macrochirus*	S,U	1,2-DCB	5,590	U.S. EPA, 1978
SALT WATER SPECIES				
Mysid shrimp, *Mysidopsis bahia*	S,U	1,2-DCB	1,970	USEPA, 1978
Sheepshead minnow, *Cyprinodon variegatus*	S,U	1,2-DCB	9,660	USEPA, 1978
Tidewater silverside *Menidia beryllina*	S,U	1,2-DCB	7,300	Dawson et al., 1977

*S = static, U = unmeasured, M = measured

TABLE E-7

CHRONIC TOXICITY OF 1,2-DICHLOROBENZENE TO AQUATIC LIFE

Species	Test	Chemical	Limits (µg/L)	Chronic Value (µg/L)	Reference
Fathead minnow, *Pimephales promelas*	ELS	1,2-dichlorobenzene	1,600-2,500	2,000	USEPA, 1978

ELS = early life stage

Di (2-ETHYLHEXYL) PHTHALATE

Background Information

Di (2-ethylhexyl) phthalate (DEHP), also known as Bis (2-ethylhexyl) phthalate, is primarily used as a plasticizer for resins such as polyvinyl chloride, giving the plastic more flexibility. Household products which contain DEHP include rainwear, tablecloths, shower curtains, food packaging material and children's toys. DEHP is also used in tubing and containers for blood products and as a hydraulic and a dielectric fluid. (ATSDR, 1987). DEHP is present in the environment from both man-made and natural sources, and has been found in plants, animals, well water, oil, soil, and air (Mayer et al., 1972).

Chemical and Physical Properties

Property	Value	Reference
Chemical Formula	$C_{24}H_{38}O_4$	ATSDR, 1987
Molecular Weight	391	USEPA, 1980
Boiling Point	385°C	ATSDR, 1987
Melting Point	-50°C	ATSDR, 1987
Water Solubility	0.285 mg/l	ATSDR, 1987
Vapor Pressure	3.4×10^{-7} mmHg	ATSDR, 1987
Henry's Law Constant	1×10^{-4} atm - m^3/mol	ATSDR, 1987
K_{oc}	9.22 ml/g	Leo, et al. 1971
Log_{10} Kow	4.88	ATSDR, 1987

Fate and Transport

DEHP is not believed to undergo any significant direct photolysis in the aquatic environment. Indirect photolysis, which involves interaction of the hydroxyl radical with the

aromatic ring, is also considered too slow to be environmentally significant (Dorfman and Adams, 1973). DEHP is expected to hydrolyze in surface water, but at a rate which is not expected to be environmentally significant. The half-life for the hydrolysis of DEHP is expected to be on the order of years. The hydrolysis of phthalate esters, as a group, is catalyzed by both acids and bases. (Thamassi and Bruce, 1966).

The vapor pressure of DEHP is very low, indicating that volatilization is probably not a significant fate process. It has been estimated that the evaporative half-life of DEHP from water is 15 years (Branso, 1978). In addition, since DEHP is readily adsorbed onto particulates, volatilization is not considered a likely transport mechanism in natural waters.

The organic carbon partition coefficient for DEHP (9.22) indicates that it will be adsorbed onto suspended particles high in organic matter. This is supported by the observation that DEHP is commonly found in sediment samples taken from stream bottoms and the sea (Giam et al. 1978). It is also reported (Autian, 1973) that DEHP will complex with natural organic substances such as fulvic acid. The resulting DEHP fulvic acid complex is water soluble, thus mobilizing and transporting the otherwise unsoluble ester (Ogner and Schnitzer, 1970).

Ecotoxicology

The ecotoxicology data in this section are reported in the U.S. EPA Ambient Water Quality Criteria document on phthalate esters (U.S. EPA, 1980). There is only one data point reported by U.S. EPA pertaining to the acute toxicity of DEHP, although there are many toxicity endpoints for other phthalatic esters. In a static unmeasured test *Daphnia magna* suffered acute effects at a concentration of 11,100 µg/L (USEPA, 1978) (Table E-8).

In the chronic tests, two species were reported by U.S. EPA, with *Daphnia magna* having a value of <3µg/L and the

TABLE E-8

ACUTE TOXICITY OF DEHP TO AQUATIC ORGANISMS

Species	Method*	LC_{50} or EC_{50} (µg/l)	Reference
FRESH WATER SPECIES			
Cladoceram, *Daphnia magna*	S,U	11,100	USEPA, 1978
Midge, *Chironomus plumosus*		>18,000	Streufert, 1977
Scud, *Gammarus pseudolimnaeus*		>32,000	Sanders, et al., 1973

*S = static

U = unmeasured

rainbow trout (Salmo gairdneri) having a value of 8.4 µg/L (Table E-9).

DEHP, which can be bioconcentrated, has BCFs ranging from 42-113 (rainbow trout) to 54-2,680 (scud) as identified in Table E-10 for fresh water species.

Regulations and Standards

Ambient Water Quality Criteria have not been established for DEHP. The lowest observed effect level for fresh water is 940 µg/L for acute exposure and 3 µg/L for chronic exposure. The lowest observed effect level for salt water is 2,944 µg/L for acute exposure and 3.4 µg/L for chronic exposure.

TABLE E-9

CHRONIC TOXICITY OF DEHP TO AQUATIC ORGANISMS

Species	Method	Limits	Chronic Value (µg/L)	Reference
FRESH WATER SPECIES				
Cladoceran, *Daphnia magna*	LC	<3	<3	Mayer and Sanders, 1973
Rainbow Trout *Salmo gairdneri*	ELS	5-14	8.4	Mehrle and Mayer, 1976

TABLE E-10

BIOCONCENTRATION FACTORS FOR DEHP

	Tissue	Bioconcentration Factor	Duration Days	Reference
FRESH WATER SPECIES				
Rainbow trout, *Salmo gairdneri*	whole body	42-113	36	Mehrle and Mayer, 1976
Fathead minnow, *Pimephales promelas*	whole body	155-886	56	Mayer, 1976
Scud, *Gammarus pseudolimnaeus*	whole body	54-2,680	14-21	Sanders, et al. 1973

1,2-DIPHENYLHYDRAZINE

Background Information

1,2-diphenylhydrazine is the symmetrical isomer of diphenylhydrazine. 1,2-diphenylhydrazine is used in the synthesis of phenylbutazone and as an intermediate in the production of dyes (Lurie, 1964).

Chemical and Physical Data

Property	Value	Reference
Chemical Formula	$C_{12}H_{12}N_2$	Weast, 1978
Molecular Weight	184	USEPA, 1986
Boiling Point	220°C	Weast, 1978
Melting Point	131°C	Weast, 1978
Water Solubility	1.84E+03 mg/L	USEPA, 1986
Vapor pressure	2.60E-05 mm Hg	USEPA, 1986
Henry's Law Constant	3.42E-09 atm-m^3/ml	USEPA, 1986
K_{oc}	418 mL/g	USEPA, 1986
Log_{10} Kow	2.90	USEPA, 1986

Fate and Transport

1,2-diphenylhydrazine is rapidly oxidized to azobenzene in aerated solutions (Rao and Hayon, 1976) and in solutions containing environmentally common cations such as Cu (IV) and Fe (III). The oxidation reaction itself does not serve as an environmental fate, but provides a route to destruction via photochemical reactions of azobenzene. Azobenzene can be reduced by chlorophyll and its derivatives, and is therefore reduced after uptake by aquatic plants (Livingston and Pariser, 1948).

The oxidation of 1,2-diphenylhydrazine in aerated solutions is pH-dependent, and the reaction proceeds at an observable rate even in acidic solutions. The redox potential of azobenzene in water is -0.12v, indicating that the reaction is easily reversible (Rao and Hayon, 1976). However, in aerated waters azobenzene should predominate, and it will revert to 1,2-diphenylhydrazine only in the presence of reducing agents such as sulfide or hydrosulfide (Hashimoto and Sunamoto, 1966). No information indicating the occurence of further oxidation of azobenzene was found.

1,2-Diphenylhydrazine is usually resistant to hydrolysis because of the high negative charge-density of the aromatic nucleus. There are no data to suggest that hydrolysis of 1,2-diphenylhydrazine is an environmentally significant process. 1,2-Diphenylhydrazine may be subject to hydrolysis only under conditions which are obtained in the bisulfite wastes of a paper mill or coal mine (Drake, 1942).

When assessing the volatility of 1,2-diphenylhydrazine from surface waters it should be noted that it will be present predominantly as azobenzene. Azobenzene has a very low vapor pressure and is likely to be adsorbed to organic materials. For these reasons it is inferred that azobenzene has very little tendency to be transported from the aqueous environment via volatilization.

There is a strong tendency for both 1,2-diphenylhydrazine and azobenzene to be adsorbed by organic materials. In addition, azobenzene is strongly adsorbed by silicate materials (Zubareva et al., 1974). Sorption onto particles and bed sediments is probably a significant fate of both 1,2-diphenylhydrazine and azobenzene.

Ecotoxicology

U.S. EPA only reports toxicity tests with 1,2-diphenylhydrazine on two species. The acute value for *Daphnia magna* is 4100 µg/L and for the bluegill, *Lepomis*

macrochirus, the value was 270 µg/L (Table E-11). There are no acute data available for salt water organisms, nor data for chronic toxicity to fresh or salt water organisms.

Regulations and Standards

Federal Ambient Water Quality Criteria (AWQC) have not been established for 1,2-diphenylhydrazine.

TABLE E-11

ACUTE TOXICITY OF 1,2-DIPHENYLHYDRAZINE TO

AQUATIC ORGANISMS

Species	Method	LC_{50} or EC_{50} (µg/L)	Reference
Cladoceran, *Daphnia magna*	S,U	4,100	USEPA, 1980
Bluegill, *Lepomis macrochirus*	S,U	270	USEPA, 1980

S = Static

U = Unmeasured

LEAD AND COMPOUNDS
(INORGANIC)

Background Information

Lead is a metallic element found in soils, with an average abundance of approximately 15 mg/kg (ppm) in the earth's crust. It is used extensively in the manufacturing of storage batteries and was used in gasoline and paint. Lead is also a natural constituent of many soils, for which concentrations normally range from 10 to 30 milligrams lead per kilogram of soil.

Chemical and Physical Properties

Property	Value	Reference
Chemical formula	Pb	Clement, 1985
Molecular weight	207.19	Clement, 1985
Boiling point	1,740°C	Clement, 1985
Melting point	327.502°C	Clement, 1985
Water solubility	Insoluble	Weast, 1980
Vapor pressure	0.00 mm Hg	USEPA, 1986
Henry's Law constant	NA	USEPA, 1986
Koc	NA	
Log_{10} Kow	NA	
Kd	99	Baes and Sharp, 1983

Fate and Transport

The concentration of lead in waters can be reduced by sorption processes, which result in enrichment of bed sediments near the source. In severely contaminated areas, precipitation may be important in controlling the mobility of this metal. Lead tends to form complexes with naturally occuring organic materials, and this increases its adsorptive affinity for clays

and mineral surfaces. Bottom and mud dwelling (benthic) microbial organisms can methylate lead to form tetramethyl lead, which allows lead in sediments to be reintroduced into the water.

While organo-lead compounds do not undergo photolysis in water, dissolved lead can be hydrolyzed to form $Pb(OH)_2$. Most lead compounds are not volatile, with the exception of tetramethyl lead produced by benthic organisms.

The adsorption of lead to sediments is highly pH dependent. At pH = 7 and above, essentially all lead is in the solid phase. At low pH lead is negatively sorbed (repelled from the adsorbent surface) so that lead is very mobile in acidic waters.

Lead is generally not bioaccumulated or biomagnified, as the pH of most natural waters renders lead unavailable for uptake. Fish tend to accumulate very little lead in edible tissues; however, invertebrates (such as oysters and mussels) can accumulate high levels of lead. Mosquito larvae also accumulate lead.

Ecotoxicology

The acute and chronic toxicity of lead to aquatic organisms is often dependent on the hardness of the water, with species being more sensitive to lead toxicity in soft water.

The ecotoxicology data in this section are reported in the Ambient Water Quality Criteria document for lead (U.S. EPA, 1984). Some acute toxicity values are shown in Table E-12. Chronic toxicity tests demonstrate that many species of fresh water fish exhibit reproductive disorders and high embryo mortality from exposure to lead (Table E-13).

Lead will bioaccumulate, especially in the soft parts of mussels (Table E-14).

TABLE E-12

ACUTE TOXICITY OF LEAD TO AQUATIC ORGANISMS

Species	Method	Chemical	LC_{50} or EC_{50} (µg/L)	Reference
FRESH WATER SPECIES				
Cladoceran, *Daphnia magna*	R,M	Lead nitrate	447.8	Chapman et al., Manuscript
Midge, *Tanytarsus dissimilis*	FT,M	Lead nitrate	235,900	Call et al., 1983
Crayfish, *Orconectes limosus*	S,M	Lead chloride	3,300	Boutet and Chaisemarten, 1973
Fathead minnow, *Pimpehales promelas*	S,U,	Lead chloride	7,330[a] 482,000[b]	Pickering and Henderson, 1966
SALT WATER SPECIES				
Hard clam (larva), *Mercenaria mercenaria*	S,U	Lead nitrate	780	Calabrese and Nelson, 1974
Mysid shrimp, *Mysidopsis bahia*	FT, M	Lead nitrate	3,130	Lussier et al., ms

*S = static
M = measured
U = unmeasured
FT = flow-through
(a) = water hardness = 20 mg/L as $CaCO_3$
(b) = water hardness = 360 mg/L as $CaCO_3$
**results are expressed as lead, not the chemical compound.

TABLE E-13

CHRONIC TOXICITY OF LEAD TO AQUATIC ANIMALS

Species	Test*	Chemical	Limits** (µg/L)	Chronic Value (µg/L)	Reference
FRESH WATER SPECIES					
Cladoceran, *Daphnia magna*	LC	Lead nitrate	9-16.7	12.26	Chapman et al., manuscript
SALT WATER SPECIES					
Mysid shrimp, *Mysidopsis bahia*	LC	Lead nitrate	17-37	25.08	Lussier et al., manuscript

* = LC = lifecycle or partial life cycle
** results are expressed as lead, not as the chemical compound

TABLE E-14

BIOCONCENTRATION FACTORS FOR LEAD

Species	Chemical	Tissue	Bioconcentration Factor	Duration (days)	References
FRESH WATER SPECIES					
Brook trout (embryo), *Salvelinus fontinalis*	Lead nitrate	whole body	42	140	Holcombe et al., 1976
SALT WATER SPECIES					
Diatom, *Ditylum brightwelli*	Lead chloride	cells	725	14	Canterford et al, 1978
Blue mussel, *Mytilus edulis*	Lead nitrate	soft parts	650	40	Schulz-Baldes, 1974
Blue mussel, *Mytilus edulis*	Lead nitrate	soft parts	2,080	130	Schulz-Baldes, 1972

Regulations and Standards

For protection of aquatic life, the federal Ambient Water Quality Criteria (AWQC) for lead are as follows:

Fresh water (µg/L)		Salt water (µg/L)	
Acute	Chronic	Acute	Chronic
82	3.2	140	5.6

MERCURY

Background Information

Mercury is a silver white metal found as a liquid at room temperature. A major use of mercury has been as a cathode in the electrolytic preparation of chlorine and caustic soda (USEPA, 1980). Electrical apparatus such as lamps and mercury battery cells account for over 25% of the mercury used annually (USEPA, 1980). About 15% of the mercury produced annually is used in thermometers and barometers (USEPA, 1980). Other substantial uses of mercury have come from use as an antifoulant, in paints, and in formulas to control fungal diseases in seeds, bulbs, plants, and vegetation (U.S. EPA, 1980). However, the EPA no longer registers mercury for use as an antifoulant or to control diseases in those organisms listed above (U.S. EPA, 1980).

Chemical and Physical Data

Property	Value	Reference
Chemical Formula	Hg	Clement, 1985
Molecular Weight	201	USEPA, 1986
Boiling Point	356.50°C	Clement, 1985
Melting Point	-38.87°C	Clement, 1985
Water Solubility	81.3 ug/l (at 30°C)	Clement, 1985
Vapor Pressure	2.0×10^{-3} mmHg	USEPA, 1986
Henry's Law Constant	NA	Clement, 1985
K_{oc}	NA	
Log_{10} K_{ow}	NA	

Fate and Transport

Mercury has long been recognized as one of the most toxic of heavy metals. Elemental mercury is a heavy liquid at room temperature. Elemental mercury is oxidized to divalent mercury under natural conditions (Wood, 1974). Divalent mercury, whether discharged directly or produced from elemental mercury, can be methylated by both aerobic and anaerobic bacteria (National Research Council, 1978; Jernelov, 1971; and Callahan et al., 1979). Divalent mercury has also been found to undergo methylation in the slime coat and intestines of fish (Jernelov, 1968).

Methyl mercury is more water soluble and 10 to 20 times more toxic to mammals and aquatic life than other forms of mercury (Fromm, 1977; Armstrong and Scott, 1979; Jernelov, 1972; Lock et al., 1981). Even though most mercury in the water column is divalent inorganic mercury, methylmercury comprises most of the mercury residual in tissues of aquatic organisms (Hattura et al., 1978; Cappon and Smith, 1979). Sorption of mercury is strongest onto organic materials (Clement, 1985).

Ecotoxicology

The ecotoxicology data in this section are reported in the U.S. EPA Ambient Water Quality Criteria document on mercury (U.S. EPA, 1985).

There are toxicity data available for three classes of mercury: divalent inorganic mercury, methyl mercury, and other compounds (chiefly organic).

Fishes (mummichog and fathead minnow) were more resistant to inorganic mercury (up to 10 times more) than invertebrates such as the midge (*Chironomus*) and mysid shrimp (*Mysidopsis bahia*) (Table E-15). In Table E-15, the cladoceran, *Daphnia*

TABLE E-15

ACUTE TOXICITY OF MERCURY TO AQUATIC LIFE

Species	Method	Chemical	LC_{50} or EC_{50} (µg/L)	Reference
		DIVALENT INORGANIC MERCURY		
FRESH WATER SPECIES				
Cladoceran, *Daphnia magna*	S,U	mercuric chloride	2.18	Canton and Adema, 1978
Midge, *Chironomus* sp.	S,M	mercuric nitrate	20	Rehwoldt et al., 1973
Fathead minnow *Pimephales promelas*	F, M	mercuric chloride	158.7	Call et al., 1982
SALT WATER SPECIES				
Blue mussel, *Mytilus edulis*	S,U	mercuric chloride	5.8	Martin et al., 1981
Eastern oyster, *Crassostrea virginica*	S,U	mercuric chloride	5.6	Calabrese et al., 1977
Mysid, *Mysidopsis bahia*	F,M	mercuric chloride	3.5	Gentile et al., 1982, 1983
Dungeness crab, (larvae) *Cancer magister*	S,M	mercuric chloride	6.6	Glickstein, 1978
Mummichog, *Fundulus heteroclitus*	S,U	mercuric chloride	200	Dorfman, 1977
		METHYL MERCURY		
FRESH WATER SPECIES				
Rainbow trout (larva), *Salmo gairdneri*	R,U	methylmercuric chloride	24	Wobeser, 1973

TABLE E-15 (Continued)

Species	Method	Chemical	LC_{50} or EC_{50} (µg/L)	Reference
		METHYL MERCURY		
SALT WATER SPECIES				
Amphipod (adult), *Gammarus duebeni*	S,U	methylmercuric chloride	150	Lockwood and Inman, 1975
Mummichog (embryo) Fundulus heteroclitus	S,M	methylmercuric chloride	51.1	Sharp & Neff, 1982
		OTHER MERCURY COMPOUNDS		
FRESH WATER SPECIES				
Fathead minnow *Pimpehales promelas*	S,M	mercuric acetate	40	Curtis et al., 1979
SALT WATER SPECIES				
Grass shrimp (adult), *Palaemonetes pugio*	S,M	mercuric acetate	47	Curtis et al., 1979
Grass shrimp adult, *Palaemonetes pugio*	S,M	mercuric thiocyanate	76	Curtis et al., 1979
Mummichog, *Fundulus heteroclitus*	S,U	mercurous sulfate	6,800	Dorfman, 1977
Mummichog, *Fundulus heteroclitus*	S,U	mercurous sulfate	300	Dorfman, 1977

S = static, R = renewal, F = flow-through, U = unmeasured, M = measured.
Results are expressed as mercury, not as the chemical.

magna, was most sensitive to divalent inorganic mercury (value = 2.18 µg/L) followed by the mysid shrimp (value = 3.5 µg/L), eastern oyster (Crossostrea virginica) (value = 5.6 µg/L) and blue mussel (Mytilus edulis) (value = 5.8 µg/L).

There are limited data available on acute toxicity of methylmercury. However, the data indicate that fish are more resistant than invertebrates (Table E-15).

It is difficult to conclude that this same trend holds true for species exposed to other mercury compounds. However, the mummichog (values at 6,800 and 300 µg/l) is more resistant than the grass shrimp (values at 47 and 76 µg/L).

There were no great differences in toxicity among the different mercury compounds. Rather, there is a greater variation in sensitivity between species within each chemical group.

Only one salt water species, Mysidopsis bahia, was tested for chronic toxicity to any mercury compound. Divalent inorganic mercury was used and the chronic value was 1.13 ug/L. A fresh water invertebrate, Daphnia magna, had a chronic value of 1.287 µg/L (Table E-16).

Only one species, Daphnia magna, was used in chronic tests for other mercury compounds. The value was 1.46 µg/L (Table E-16).

There are various other data available for mercury compounds identified in Table E-17. Inorganic mercury has been shown to produce abnormal development of grass shrimp larva at concentrations of 10-18 µg/L. In the mummichog embryo, the number of developmental abnormalities increased with increasing concentrations of mercuric chloride. The mallard duck (Anas platyrhyncos) experiences reduced reproductive success when it receives 0.5 mg/kg methylmercuric dicyandiamide in food (Table E-17). This was a two-generation study.

When exposed to 400 µg/L methylmercuric chloride for 24 hours the mussel Mytilus edulis exhibits a 90% reduced feeding

TABLE E-16

CHRONIC TOXICITY OF MERCURY TO AQUATIC LIFE

Species	Test	Chemical	Limits (µg/l)	Chronic Value (µg/L)	Reference
			DIVALENT INORGANIC MERCURY		
FRESH WATER SPECIES					
Cladoceran, *Daphnia magna*	LC	mercuric chloride	0.91-1.82	1.287	Biesinger et al., 1982
Fathead minnow, *Pimephales promelas*	ELS	mercuric chloride	<0.23	<0.23	Call et al., 1983
SALT WATER SPECIES					
Mysid, *Mysidopsis bahia*	LC	mercuric chloride	0.8-1.6	1.13	Gentile et al., 1982, 1983
			METHYL MERCURY		
FRESH WATER SPECIES					
Cladoceran, *Daphnia maga*	LC	methylmercuric chloride	0.52-0.87	0.67	Biesinger et al., 1982
Brook trout, *Salvelinas fontinalis*	LC	methylmercuric chloride	0.29-0.93	0.52	McKim et al., 1976
			OTHER MERCURY COMPOUNDS		
FRESH WATER SPECIES					
Cladoceran, *Daphnia magna*	LC	phenylmercuric acetate	1.12-1.90	1.46	Biesinger et al., 1982

LC = life cycle, ELS = early life stage.

TABLE E-17

OTHER DATA FOR MERCURY

Species	Chemical	Duration	Effect	Result (µg/L)	Reference
			INORGANIC MERCURY		
Grass shrimp, (larva) *Palaemonetes vulgaris*	mercuric chloride	48 hours	abnormal development	10–18	Shealy and Sandifer, 1975
Mummichog, (embryo) *Fundulus heteroclitus*	mercuric chloride	3 days	many developmental abnormalities	30–40	Weis and Weis, 1977
Mummichog, (embryo) *Fundulus heteroclitus*	mercuric chloride	3 days	some developmental abnormalities	10–20	Weis and Weiss, 1977
Mallard duck, *Anas platyrhyncos*	methylmercuric dicyandiamide	2 generations	reduced reproductive success	0.5 mg/kg in food	Heinz, 1976
			METHYL MERCURY		
Blue mussel, *Mytilus edulis*	methylmercuric chloride	24 hours	90% reduced feeding rate	400	Dorn, 1976
Mummichog (adult), *Fundulus heteroclitus*	methylmercuric chloride	24 hours	disrupted osmoregulation	125	Renfro et al., 1974
Mink (adult) *Mustela vison*	methylmercuric chloride	93 days	histologic evidence of injury	1,100	Wobeser, 1973
Mink (adult) *Mustela vison*	methylmercuric chloride	93 days	LC_{50} in brain tissue	11,000	Wobeser, 1973
			OTHER MERCURY COMPOUNDS		
Eastern oyster, *Crassostrea virginica*	mercuric acetate	12 hr/day for 15 days	33% reduction in shell growth	10	Cunningham, 1976

rate. At a lower concentration of 125 µg/L (24 hr) the mummichog exhibits disrupted osmoregulation (Table E-18).

Mercury will bioaccumulate, and in the fathead minnow organic mercury has a bioconcentration factor which is 10 times that of inorganic mercury (44,130 as compared to 4,994). A similar pattern holds true for the eastern oyster (*Crassostrea viginica*) where organic mercury has a bioconcentration factor of 40,000 and inorganic has one of 10,000 (Table E-18).

Regulations and Standards

The federal Ambient Water Quality Criteria (AWQC) for mercury are as follows:

Fresh water (µg/L)		Salt water (µg/L)	
acute	chronic	acute	chronic
2.4	0.012	2.1	0.025

TABLE E-18

BIOACCUMULATION OF MERCURY BY AQUATIC ORGANISMS

Species	Tissue	Chemical	Duration (days)	Bioconcentration Factor	Reference
		DIVALENT INORGANIC MERCURY			
FRESH WATER SPECIES					
Fathead minnow, *Pimephales promelas*	whole body	mercuric	287	4,994	Snarski and Olson, 1982
SALT WATER SPEICES					
Eastern oyster, *Crassostrea virginica*	soft parts	mercuric chloride	74	10,000	Kopfler, 1974
		METHYL MERCURY			
FRESH WATER SPECIES					
Fathead minnow, *Pimephales promelas*	whole body	methylmercuric chloride	336	44,130 - 81,670	Olson et al., 1975
SALT WATER SPECIES					
Eastern oyster, *Crassostrea virginica*	soft parts	methylmercuric chloride	74	40,000	Kopfler, 1974
		OTHER MERCURY COMPOUNDS			
SALT WATER SPECIES					
Eastern oyster, *Crassostrea virginica*	soft parts	phenylmercuric chloride	74	40,000	Kopfler, 1974

PHENOL

Background Information

Phenol is a widely used industrial chemical which has a large potential for occupational exposures. It is used primarily as an intermediate component for artificial resins, medical and industrial dyes, and chemical reagents (Windholz, 1983).

Chemical and Physical Properties

Property	Value	Reference
Chemical formula	C_6H_5OH	Verschueren, 1983
Molecular weight	94.11	Verschueren, 1983
Boiling Point	182.0°C	Verschueren, 1983
Melting point	41°C	Verschueren, 1983
Water solubility	82 g/L at 15°C	Verschueren, 1983
Vapor pressure	0.2 mm at 20°C	Verschueren, 1983
Henrys Law constant	4.54E-07atm-m^3/mol	USEPA, 1986
Koc	14.2 mL/g	USEPA, 1986
Log_{10} Kow	1.46	USEPA, 1986

Fate and Transport

The destruction of phenol in the aquatic environment is determined by photoxidation, metal-catalyzed oxidation, and biodegradation.

A gradual reduction of aqueous phenol can occur through photooxidation, and hydroxylation at the C-2 position in the presence of air and iron (III) or copper (II) ions has been reported. Phenol is resistent to hydrolysis; there is little tendency for it to volatilize from water, and it is not likely to become sorbed to organic sediments.

Phenol is biodegraded in natural waters by bacteria, and increasing the aeration of a natural system enhances this removal of phenol.

Ecotoxicity

The ecotoxicology data in this section are reported in the U.S. EPA Ambient Water Quality Criteria document on phenol (U.S. EPA, 1980). Acute and chronic toxicity values are reported in Tables E-19 and E-20.

Regulations and Standards

There are no federal Ambient Water Quality Criteria (AWQC) for phenol for the protection of aquatic life. The lowest observed effect level for fresh water is 10,200 µg/L for acute exposure and 2,560 µg/L for chronic exposure. The lowest observed effect level for salt water is 5,800 µg/L for acute exposure.

TABLE E-19

ACUTE TOXICITY OF PHENOL TO AQUATIC ORGANISMS

Species	Method	LC_{50} or EC_{50} (µg/L)	References
FRESH WATER SPECIES			
Cladoceran (young), *Daphnia magna*	S,U	7,000	Dowden & Bennett, 1965
Fathead minnow (adult), *Pimephales promelas*	F,M	24,000	Ruesink and Smith, 1975
SALT WATER SPECIES			
Grass shrimp, *Palaemonetes pugio*	S,U	5,800	Tatem et al., 1978
Hard clam, *Mercenaria mercenaria*	S,U	52,630	Davis and Hidu, 1969

S = static U = unmeasured

M = measured F = flow through

TABLE E-20

CHRONIC TOXICITY OF PHENOL TO AQUATIC ORGANISMS

Freshwater Species

Species	Method	Limits (µg/L)	Chronic Value (µg/L)	References
Fathead minnow, *Pimephales promelas*	ELS	1,830-3,570	2,560	Holcombe et al., 1980

ELS = early life stage

POLYNUCLEAR AROMATIC HYDROCARBONS

Background Information

Polynuclear aromatic hydrocarbons (PAHs) are a class of compounds consisting of substituted and unsubstituted polycyclic and heterocyclic aromatic rings. PAHs are formed by the incomplete combustion of organic compounds in the presence of insufficient oxygen.

PAHs are present in the environment from both natural and man-made sources. As a group, they are widely distributed in the environment, having been found in animal and plant tissue, sediments, air and surface water (Radding et al., 1976).

Chemical and Physical Properties

Property	Value	Reference
FLUORENE		
Chemical formula	$C_{13}M_{10}$	Hawley, 1981
Molecular weight	116.15	Weast, 1977
Boiling point	293°C	Weast, 1977
Melting point	116-117°C	Weast, 1977
Water solubility	1.98 mg/L	MacKay and Shiu, 1977
Vapor pressure	7.10E-04 mmHg	USEPA, 1986
Henry's Law constant	6.42E-05 atm-m^3/mol	USEPA, 1986
Koc	7300 mL/g	USEPA, 1986
Log_{10} Kow	4.20	USEPA, 1986

Property	Value	Reference
PHENANTHRENE		
Chemical formula	$C_{14}H_{10}$	Hawley, 1981
Molecular weight	178.23	Radding et al., 1976
Boiling point	340°C	Weast, 1977
Melting point	101°C	Weast, 1977
Water solubility	1.29 mg/L	MacKay and Shiu, 1977
Vapor pressure	6.8×10^{-4} mmHg	USEPA, 1986
Henry's Law constant	1.59E-04 atm-m^3/mol	USEPA, 1986
Koc	14000	USEPA, 1986
Log_{10} Kow	4.46	USEPA, 1986
BENZO(a)PYRENE		
Chemical formula	$C_{20}H_{12}$	Hawley, 1981
Molecular weight	252	Smith et al., 1978
Boiling point	176.5°C	West, 1977
Melting point	179°C	Smith et al., 1977
Water solubility	3.8E-03 mg/L	MacKay and Shiu, 1977
Vapor pressure	5E-09 mmHg	Smith et al., 1978
Henry's Law constant	1.55E-06 atm-m^3/mol	USEPA, 1986
Koc	5,500,000	USEPA, 1986
Log_{10} Kow	6.06	USEPA, 1986
BENZO(a)ANTHRACENE		
Chemical formula		
Molecular weight	228.28	Smith et al., 1978

Property	Value	Reference
Boiling point	435°C	Weast, 1977
Melting point	155-157°C	Smith et al., 1977
Water solubility	5.70E-03	USEPA, 1986
Vapor pressure	5E-09 mmHg	Smith et al., 1978
Henry's Law constant	NA	USEPA, 1986
Koc	1,380,000 mL/g	USEPA, 1986
Log_{10} Kow	5.60	USEPA, 1986

BENZO(b)FLUORANTHENE

Chemical formula		
Molecular weight	252.32	IARC, 1973
Boiling point	-	Weast, 1977
Melting point	167-168°C	IARC, 1973
Water solubility	4.3E-03	USEPA, 1986
Vapor pressure	5.10E-07 mmHg	USEPA, 1986
Henry's Law constant	3.94E-05 atm-m^3/mol	USEPA, 1986
Koc	550,000	USEPA, 1986
Log_{10} Kow	6.06	USEPA, 1986

CHRYSENE

Chemical formula		
Molecular weight	228.28	Radding et al., 1976
Boiling point	448°C	Weast, 1977
Melting point	256°C	Radding et al., 1976
Water solubility	1.80E-03 mg/L	USEPA, 1986
Vapor pressure	6.3E-09 mmHg	USEPA, 1986
Henry's Law constant	1.05E-06 atm-m^3/mol	USEPA, 1986
Koc	200,000 mL/g	USEPA, 1986
Log_{10} Kow	5.61	USEPA, 1986

Property	Value	Reference
FLUORANTHENE		
Chemical formula	$C_{16}H_{10}$	Hawley, 1981
Molecular weight	202	USEPA, 1986
Boiling point	375°C	West, 1977
Melting point	111°C	West, 1977
Water solubility	2.06E-01 mg/L	USEPA, 1986
Vapor pressure	5.00E-06 mmHg	USEPA, 1986
Henry's Law constant	6.46E-06 atm-m^3/mol	USEPA, 1986
Koc	38,000 mL/g	USEPA, 1986
Log_{10} Kow	4.90	USEPA, 1986

Fate and Transport

There are few data available specific to individual PAH compounds, and because of this their aquatic fate is inferred from data summarized for PAH compounds in general.

Most polynuclear aromatic hydrocarbons absorb solar radiation strongly and may, therefore, undergo direct photolysis or photooxidation (Radding et al., 1976). When in the aqueous environment photolysis is rapid for benzo(a)pyrene and benzo(a)anthracene. Smith et al. (1978) reported half-lives in water of 1.2 hours and 1-2 hours, respectively. In contrast, hydrolysis is not thought to be a significant fate process, as PAH compounds do not contain groups amenable to hydrolysis (Radding et al., 1976).

Measured volatilization rates for PAHs have not been reported in the literature, so the assessment of volatilization as a transport process is only speculative. In general, the volatilization rate decreases as the vapor pressure decreases, both of which are inversely related to the number of aromatic rings. Southworth (1979) concluded that the rate of volatilization of PAHs with four or more rings will be insignificant under all conditions.

Most PAH compounds will adsorb onto particulate matter and be transported in water (Radding et al., 1976). Benzo(a)pyrene and benzo(a)anthracene show rapid partitioning onto suspended matter, and sorption onto sediments is strongly correlated with organic carbon levels in sediments.

Bacteria have been shown to use some PAHs as a sole source of carbon for growth (Radding et al., 1976). Evidence for bacterial degradation is limited, and there is no compound-specific information. Since no organisms have been isolated that are capable of using four or five ringed compounds as a sole carbon source, it is assumed that they are co-metabolized with simpler compounds.

Groenewegen and Stolp (1975) reported that phenanthrene is degraded significantly by microbial organisms. Fluorene and fluoranthene are only somewhat degraded and chrysene was not degraded to any measureable degree. The biodegradation of fluoranthene is heat sensitive, requires oxygen, and is enhanced in the presence of cyanide. Biodegradation is thought to be an important fate process for PAHs with four or fewer aromatic rings.

Ecotoxicology

The ecotoxicology data in this section are reported in the U.S. Fish and Wildlife Biological Report on Polycyclic Aromatic Hydrocarbons (Eisler, 1987).

There are few data available on the toxicity of PAHs to aquatic organisms. Among the indicator species chosen for the UOP risk assessment there is acute data available only for the grass shrimp, *Palaemonetes pugio*. Two LC_{50}s were established using both fluorene and phenanthrene. The LC_{50} (96 hr) for fluorene was 320 µg/L and the LC_{50} (24 hr) for phenanthrene was 370 µg/L. The 96-hour LC_{50} for fluorene for the sheepshead minnow (*Cyprinodon variegatus*) was 1680 µg/L

(Table E-21). These data indicate that the sheepshead minnow (Cyprinodon variegatus) is the sensitive salt water species of interest.

While there were no data available on the acute toxicity to any of the fresh water indicator species, there are data on other invertebrates and fish. The sandworm (Neanthes arenaceodentata) was the representative invertebrate tested and was most sensitive to phenanthrene (LC_{50} = 370 µg/L) (Table E-21). The LC_{50} values for benzo(a)anthracene and chrysene were both >1000 µg/L.

A variety of species were tested using fluorene and the data indicate that the fathead minnow (Pimephales promelas) is least sensitive (LC_{50} = >100,000 µg/L) and the Bluegill (Lepomis macrochirus) was most sensitive (LC_{50} = 500 µg/L).

Only phenanthrene and fluorene were used in chronic tests, and the midge (Chironomus plumosus) was more resistant to fluorene toxicity than Daphnia magna (Table E-22). Two species were used for phenanthrene tests, and Daphnia magna (95% Chronic index - 590-840 µg/L) was more resistant than the rainbow trout (95% chronic index = 10-90 µg/L). There are data available which indicate that benzo(a)pyrene will bioaccumulate. In Daphnia magna the bioconcentration factor (BCF) is 134,248 and in the eastern oyster (Crassostrea virginica) the BCF is 242 (Table E-23).

Regulations and Standards

There are no federal Ambient Water Quality Criteria available for any of the PAHs of interest. The lowest observed effect level for salt water is 300 µg/L for acute exposure.

TABLE E-21

ACUTE TOXICITY OF PAHs TO AQUATIC ORGANISMS

Species	Chemical	Concentration (µg/L)	Effect	Reference
FRESH WATER SPECIES				
Bluegill, *Lepomis macrochirus*	Benz(a)anthracene	1000	LC_{87} (6 mos)	EPA, 1980
Sandworm, *Neanthes arenaceodentata*	Benzo(a)pyrene	>1000	LC_{50} (96 h)	Neff, 1979
Sandworm, *Neanthes arenaceodentata*	Chrysene	>1000	LC_{50} (96 h)	Neff, 1979
Sandworm, *Neanthes arenaceodentata*	fluoranthene	500	LC_{50} (96 h)	Neff, 1979
Bluegill, *Lepomis macrochirus*	fluorene	500	LC_{12} (30 d)	Finger et al., 1985
Bluegill, *Lepomis macrochirus*	fluorene	910	LC_{50} (96 h)	Finger et al., 1985
FRESH WATER SPECIES				
Amphipod, *Gammarus pseudolimnaeus*	fluorene	600	LC_{50} (96 h)	Finger et al., 1985
Rainbow trout, *Salmo gairdneiri*	fluorene	820	LC_{50} (96 hr)	Finger et al., 1985
Fathead minnow, *Pimephales promelas*	fluorene	>100,000	LC_0 (96 hr)	Finger et al., 1985
Sandworm, *Neanthes arenceodentata*	phenanthrene	370	LC_{50} (24 hr)	Neff, 1979

TABLE E-21 (Continued)

Species	Chemical	Concentration (µg/L)	Effect	Reference
SALT WATER SPECIES				
Grass shrimp, *Palaemonetes pugio*	fluorene	320	LC_{50} (96 hr)	Neff, 1979
Sheepshead minnow, *Cyprinodon variegatus*	fluorene	1680	LC_{50} (96 hr)	Neff, 1979
Grass shrimp *Palaemonetes pugio*	phenanthrene	370	LC_{50} (24 hr)	Neff, 1979

TABLE E-22

CHRONIC TOXICITY OF PAHs TO AQUATIC ORGANISMS

Special	Chemical	95% chronic index (µg/L)	Reference
FRESH WATER SPECIES			
Midge, *Chironomus plumosus*	fluorene	1900–3000	Mayer and Ellersieck, 1986
Cladoceran, *Daphnia magna*	fluorene	330–550	Mayer and Ellersieck, 1986
Cladoceran, *Daphnia magna*	phenanthrene	590–840	Millman et al., 1984
Rainbow trout, *Salmo gairdeneri*	phenanthrene	10–90	Millman et al., 1984

TABLE E-23

BIOCONCENTRATION FACTORS FOR PAH

Species	Chemical	Duration	BCF	Reference
Cladoceran, *Daphnia magna*	Benzo(a)pyrene	3 days	134,248	Lu et al., 1977
Eastern oyster, *Crassostrea virginica*	Benzo(a)pyrene	14 days	242	Couch et al., In Press

POLYCHLORINATED BIPHENYLS

Background Information

Polychlorinated biphenyls (PCBs) are chlorinated derivatives of a class of aromatic organic compounds called biphenyls. Currently there is no production of PCBs in the United States as the sole manufacturer stopped production in 1974. Due to the long life of many products containing PCBs it is believed that a substantial portion of the PCBs previously manufactured are still in use, and thus represent potential future discharge into the environment (USEPA, 1980). PCBs are used in heat transfer fluids, hydraulic fluids, plasticizers, lubricants, and fluids in vacuum pumps and compressors.

PCBs consist of a mixture of chlorinated biphenyls which contain a varying number of substituted chlorine atoms on the aromatic rings. The biphenyl molecule has ten sites where chlorine substitution can occur, and 209 isomers are possible.

Chemical and Physical Properties

These data are for Aroclor® 1248.

Property	Value	Reference
Chemical formula	variable	USEPA, 1980
Molecular weight	299.5	ATSDR, 1987
Boiling point	340-375°C	ATSDR, 1987
Melting point	unknown	ATSDR, 1987
Water solubility*	0.06 mg/L @ 24°C	ATSDR, 1987
Vapor pressure	4.94×10^{-4} mmHg	ATSDR, 1987
Henry's Law constant	2.8×10^{-3} atm-m^3/mol	ATSDR, 1987
Koc	54,626 ml/g	ATSDR, 1987
Log_{10} Kow	6.2 L/kg	ATSDR, 1987

*Solubility of PCBs decreases with increasing chlorination.

Fate and Transport

Polychlorinated biphenyls are a family of compounds which vary widely in physical and chemical properties. In general, PCBs are extremely resistant to oxidation and strongly resistant to both acidic and basic hydrolysis. Some photolysis may occur on the highly chlorinated PCBs, and each may lose at least one chlorine atom annually.

Volatilization and sorption play a greater role in the fate of PCBs than do the above mentioned factors. Volatilization of PCBs is thought to account for their ubiquitous distribution. It has been suggested by Mackay and Wokloff (1973) that volatilization results from the entrainment of solute molecules in the stream of evaporating water molecules. The rate at which molecules are carried off is determined by the surface concentration of the solute and the evaporative rate of the water. Volatilization from sediments and soil is a much slower process than volatilization from water. Adsorption to sediments is the major non-destructive process affecting PCBs after introduction to the aquatic environment. PCBs have a high affinity for suspended solids, especially those high in organic carbon. Sediments in soil-water systems can serve as a reservoir for re-solution of PCBs. Release of these compounds by sediments can cause long-term pollution.

As PCBs are adsorbed strongly to organic sediments, they are also strongly bioaccumulated. Their resistance to biodegradation also makes them very persistent in organisms. The potential for bioaccumulation of PCBs is directly related to the number of chlorines. The more highly chlorinated species have a greater octanol/water partition coefficient and are also more resistant to biodegradation (Metcalf et al., 1975).

PCB compounds which have four chlorines or less can be biodegraded while those having five chlorines or more per molecule are quite resistant to biodegradation. Hydroxylation

by microbial oxidation is the mechanism of biodegradation, and the biodegradability of PCBs is a function of the number of C-H available for hydroxylation. Rates of biodegradation depend on the distribution of biota, concentration of PCBs, temperature, and availability of nutrients.

Ecotoxicology

The ecotoxicology data reported in this section comes from the U.S. EPA Ambient Water Quality Criteria document on PCBs (U.S. EPA, 1980). In toxicity tests of fresh water species it was shown that juvenile fathead minnow are less sensitive to Aroclor® 1242 than their adult counterparts (Table E-24). Adult fathead minnows (*Pimephales promelas*) were shown to have a slightly higher tolerance to Aroclor® 1242 than to Aroclor® 1254. Among the salt water species tested, the eastern oyster (*Crassostrea virginica*) was most sensitive in comparison to two species of shrimp. The oysters were equally sensitive to Aroclor® 1016, 1248, and 1254, having LC_{50} or ED_{50} values of 10.2 µg/L, 17 µg/L, and 14 µg/L, respectively . The grass shrimp (*Palaemonetes pugio*) exhibited approximately equal sensitivity to Aroclor® 1016 as did the eastern oyster.

In the chronic toxicity tests using the fathead minnow, Aroclor® 1248 was shown to be the most toxic Aroclor®, having a chronic value of 0.2 µg/L (Table E-25). The midge (*Tanytarsus dissimilis*) was also relatively sensitive, having a chronic value of 0.8 µg/L for Aroclor® 1254. *Daphnia magna* was more tolerant to Aroclor® 1254, as the chronic value was 15 µg/L.

Only one salt water species was tested, the sheepshead minnow (*Cyprinodon variegatus*). The sheepshead minnow was more sensitive to Aroclor® 1254 (value = 0.098 µg/L) than to Aroclor® 1016 (value = 7,14 µg/L). Chronic exposures to Aroclor's® produced pathological effects seen in acute tests including fungus-like lesions on the body and reduced feeding (Hansen et al., 1973). Table E-26 contains other toxicity test values reported in the literature.

TABLE E-24

ACUTE TOXICITY OF POLYCHLORINATED BIPHENYLS TO AQUATIC ORGANISMS

Species	Method	Chemical	LC_{50} or EC_{50} (µg/L)	Reference
FRESH WATER SPECIES				
Scud, *Gammarus fasciatus*	S,U	Aroclor® 1248	52	Mayer et al., 1977
Scud, *Gammarus pseudolimnaeus*	FT, M	Aroclor® 1248	29	Nebeker & Puglisi, 1974
Damselfly, *Ischnura verticalis*	FT,M	Aroclor® 1242	400	Mayer et al., 1977
Fathead minnow, *Pimephales promelas*	FT,M	Aroclor® 1242	15	Nebeker, et al., 1974
SALT WATER SPECIES				
Eastern oyster, *Crassostrea virginica*	FT,U	Aroclor® 1016	10.2	Hansen et al., 1974a
Eastern oyster, *Crassostrea virginica*	FT,U	Aroclor® 1248	17**	Lowe, undated
Eastern oyster, *Crassostrea virginica*	FT,U	Aroclor® 1254	14**	Lowe, undated
Eastern oyster, *Crassostrea virginica*	FT,U	Aroclor® 1260	60**	Lowe, undated
Grass shrimp, *Palaemonetes pugio*	FT,U	Aroclor® 1016	12.5	Hansen et al., 1974a

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S = static, FT = flow-through, U = unmeasured, M = measured

EC_{50} based on decreased growth of oysters

TABLE E-25

CHRONIC TOXICITY OF POLYCHLORINATED BIPHENYLS TO AQUATIC ORGANISMS

Species	Test	Chemical	Limits (µg/L)	Chronic Value (µg/L)	Reference
FRESH WATER SPECIES					
Cladoceran, *Daphnia magna*	LC	Aroclor® 1248	2.5-7.5	4.3	Nebeker and Puglisi, 1974
Cladoceran, *Daphnia magna*	LC	Aroclor® 1254	1.2-3.5	2.1	Nebeker and Puglisi, 1974
Midge, *Tanytarsus dissimilis*	LC	Aroclor® 1254	0.5-1.2	0.8	Nebeker and Puglisi, 1974
Scud, *Gammarus pseudolimnaeus*	LC	Aroclor® 1248	2.2-5.1	3.3	Nebeker and Puglisi, 1974
Fathead minnow, *Pimephales promelas*	LC	Aroclor® 1248	0.1-0.4	0.2	DeFoe et al., 1978
Fathead minnow, *Pimephales promelas*	LC	Aroclor® 1260	1.3-4.0	2.3	DeFoe et al., 1978
Fathead minnow, *Pimephales promelas*	LC	Aroclor® 1242	5.4-15.0	9.0	Nebeker et al., 1978
Fathead minnow, *Pimephales promelas*	LC	Aroclor® 1254	1.8-4.6	2.9	Nebeker et al., 1978
SALT WATER SPECIES					
Sheepshead minnow, *Cyprinodon variegatus*	ELS	Aroclor® 1016	3.4-15.0	7.14	Hansen et al., 1975
Sheepshead minnow, *Cyprinodon variegatus*	ELS	Aroclor® 1248	0.06-0.16	0.098	Schimmel et al., 1974

LC = life cycle or partial life cycle, ELS = early life state

TABLE E-26

OTHER DATA FOR POLYCHLORINATED BIPHENYLS

Species	Chemicals	Duration	Effect	Result (µg/L)	Reference
FRESH WATER SPECIES					
Cladoceran, *Daphnia magna*	Aroclor® 1248	2 weeks	LC_{50}	2.6	Nebeker and Puglisi, 1974
Rainbow trout, *Salmo gairdneri*	Aroclor® 1248	25 days	LC_{50}	3.4	Mayer et al., 1977
Fathead minnow, *Pimephales promelas*	Aroclor 1248	30 days	LC_{50}	4.7	DeFoe et al., 1978
Mink, *Mustela vison*	Aroclor® 1254	4 months	Reduced reproduction	1.0 µg/g	Ringer et al., 1972
Mink, *Mustela vison*	PCB residues	1 year	Depressed growth	10 µg/g	Aulerich et al., 1973
SALT WATER SPECIES					
Eastern oyster, *Crassostrea virginica*	Aroclor® 1254	24 weeks	Reduced growth	5.0	Lowe et al., 1972
Grass shrimp *Palaemonetes pugio*	Aroclor® 1254	4 days	Water efflux affected and altered metabolic state	25–45	Roesljadi et al., 1976a,b
Pink shrimp, *Penaeus duorarum*	Aroclor® 1248	48 hours	LC_{50}	32	Lowe, undated
Sheepshead minnow, (adult) *Cyprinodon variegatus*	Aroclor® 1254	28 days	Lethargy, reduced feeding fin-rot, mortality	10	Hansen et al., 1973

TABLE E-26 (Continued)

OTHER DATA FOR POLYCHLORINATED BIPHENYLS

Species	Chemicals	Duration	Effect	Result (µg/L)	Reference
Sheepshead minnow, (juvenile) *Cyprinodon variegatus*	Aroclor® 1254	21 days	Mortality	10	Schimmel et al., 1974
Sheepshead minnow, *Cyprinodon variegatus*	Aroclor® 1254	28 days	Affected reproduction*	0.14	Hansen et al., 1973

*Significantly affected hatching of embryos or the survival of fry from exposed adults.

Aquatic organisms can bioaccumulate total body concentrations of PCBs hundreds to thousands of times greater than the surrounding water (Gooch and Hamdy, 1983). Bioaccumulation can occur through ingestion of contaminated food organisms or by direct absorption through the integument. Nisbet and Sarofim (1972) reported higher concentrations of PCBs in plankton than in fish, indicating that direct absorption is the primary route of accumulation. Bioconcentration factors for fresh water and salt water species are presented in Table E-27.

Most fish lack the capacity to metabolize the higher chlorinated cogeners, and in trout they have half-lives in excess of several hundred days (Cordle, Locke, and Springer, 1982).

Regulations and Standards

The federal Ambient Water Quality Criteria (AWQC), for protection of aquatic life are as follows:

Fresh water (µg/L)		Salt water (µg/L)	
Acute	Chronic	Acute	Chronic
2.0	0.014	10	0.03

The FDA action level (marketability for human consumption) for fish and shellfish is 5000 µg/kg.

TABLE E-27

BIOCONCENTRATION FACTORS FOR POLYCLORINATED BIPHENYLS

Species	Tissue	Chemical	Bioconcentration Factor	Duration (days)	Reference
FRESH WATER SPECIES					
Cladoceran, *Daphnia magna*	Whole body	Aroclor® 1254	3,800	4	Mayer et al., 1977
Scud, *Gammarus pseudolimnaeus*	Whole body	Aroclor® 1248	108,000	60	Nebeker and Puglisi, 1974
Phantom midge, *Chaoborus punctipennis*	Whole body	Aroclor® 1254	2,700	14	Mayer et al., 1977
Fathead minnow (female) *Pimephales promelas*	Whole body	Aroclor® 1248	120,000	240	DeFoe et al., 1978
Fathead minnow (female) *Pimephales promelas*	Whole body	Aroclor® 1260	270,000	240	DeFoe et al., 1978
Fathead minnow (male) *Pimephales promelas*	Whole body	Aroclor® 1242	274,000	255	Nebeker et al., 1974
Bluegill, *Lepomis macrochirus*	Whole body	Aroclor® 1248	52,000	77	Stalling, 1971
Channel catfish, *Ictalurus punctatus*	Whole body	Aroclor® 1248	56,400	77	Mayer et al., 1977
SALT WATER SPECIES					
Eastern oyster, *Crassostrea virginica*	Edible portion	Aroclor® 1016	13,000	84	Parrish et al., 1974

TABLE E-27 (Continued)

BIOCONCENTRATION FACTORS FOR POLYCHLORINATED BIPHENYLS

Species	Tissue	Chemical	Bioconcentration Factor	Duration (days)	Reference
Eastern oyster, *Crassostrea virginica*	Edible portion	Aroclor® 1254	101,000	245	Lowe et al., 1974
Grass shrimp, *Palaemonetes pugio*	Whole body	Aroclor® 1254	27,000	16	Nimmo et al., 1974
Blue crab, *Callinectes sapidus*	Whole body	Aroclor® 1254	>230,000	Field data	Nimmo et al., 1975
Sheepshead minnow (adult) *Cyprinodon variegatus*	Whole body	Aroclor® 1016	25,000	28	Hansen et al., 1975
Sheepshead minnow (juvenile), *Cyprinodon variegatus*	Whole body	Aroclor® 1016	43,100	28	Hansen et al., 1975
Sheepshead minnow (fry), *Cyprinodon variegatus*	Whole body	Aroclor® 1016	14,400	28	Hansen et al., 1975

ZINC

Background Information

Zinc is a metallic element occuring in nature in the zero valence (metal and alloys) and +2 valance (compound) states. It is ubiquitous in the environment and is present in most foodstuffs, water, and air. Seafoods, meats, whole grains, dairy products, nuts and legumes are high in zinc content, while vegetables are lower.

Zinc is widely used in galvanizing processes, in paints, and as a wood preserver. Various zinc salts have been employed in medicinal preparations such as topic astringents, antiseptics, and antifungal agents.

Zinc is associated with relatively low toxicity and is an essential trace element for humans and other mammals (National Research Council, 1978). It is necessary for many enzyme reactions, protein synthesis, and carbohydrate metabolism.

Chemical and Physical Properties

Property	Value	Reference
Molecular weight	65	USEPA, 1986
Boiling point	907°C	Clement, 1985
Melting point	419.58°C	Clement, 1985
Water solubility	Insoluble; some salts are soluble	Clement, 1985
Vapor pressure	NA	
Henry's Law constant	NA	USEPA, 1986
Log_{10} K_{ow}	NA	
Kd	16	Baes and Sharp, 1983

Fate and Transport

There is no evidence to indicate that volatilization is an important aquatic fate of zinc. Alkyl zinc compounds are unstable in water and oxygen, therefore, volatile methylated compounds are probably not formed in aquatic environments.

The dominant fate of zinc in aquatic environments is probably sorption by hydrous metal oxides, clay minerals, and organic materials. The concentrations of zinc in suspended and bed sediments always exceed concentrations in ambient waters (USEPA, 1979). In general, the smaller grain size sediments adsorb higher concentrations of zinc than do large grain sediments.

The composition of solids in water plays a role in how zinc will be transported. Where the solids are mostly dissolved, the zinc is transported in solution as the hydrated cation or complexed species (Degroot and Allersma, 1975). Conversely, when suspended solids make a high proportion of the total solids load, most of the zinc will be transported sorbed to the suspended particles (Kubota et al., 1974).

The tendency of zinc to be adsorbed is a pH-dependent process. Below pH 6, little or no zinc is adsorbed, but when the pH exceeds 7 zinc is completely removed from solution. It has also been found that zinc is dissolved from sediments as salinity increases (Hels et al., 1975).

No evidence has been found to indicate that photolysis affects the fate of zinc.

Zinc is bioaccumulated by all organisms, and it occurs even in the absence of abnormally high zinc concentrations since it is an essential nutrient. The chemical form in which zinc occurs has a great effect on its availability for bioaccumulation. Zinc is readily accumulated from detrital organic materials, but little uptake occurs when zinc is co-precipitated with hydrous iron or manganese oxides.

Ecotoxicology

The ecotoxicological data in this section are reported in the U.S. EPA Ambient Water Quality Criteria document on zinc (U.S. EPA, 1987).

Zinc is bioaccumulated by marine and fresh water fish via both food and water. Bioconcentration factors (BCF) in aquatic organisms, listed in Table E-28, range from 51 to 23,820.

Acute toxicity tests for fresh water animals reported in the Ambient Water Quality Criteria Document Table 1 had mean values ranging from 50.7 to 88,960 µg/L. *Daphnia magna* was particularly sensitive (525 µg/L) while the banded killifish (*Fundulus diaphanus*) was less sensitive (19,200 µg/L) as shown in Table E-29. The same trend was seen among the saltwater species as invertebrates such as the mussel and mysid shrimp were more sensitive than *Fundulus*.

In the salt water species chronic test only one species, the mysid shrimp (*Mysidopsis bahia*), was reported. The chronic value of 166 µg/L is within the trend of invertebrate sensitivity (Table E-30). Table E-31 reports other toxicity testing values for aquatic organisms' exposure to zinc.

Regulations and Standards

The federal Ambient Water Quality Criteria (AWQC) for protection of aquatic life are as follows:

Fresh water (µg/L)		Salt water (µg/L)	
Acute	Chronic	Acute	Chronic
120	110	95	86

TABLE E-28

BIOCONCENTRATION FACTORS FOR ZINC

Species	Tissue	Chemical	BCF	Duration (days)	Reference
FRESH WATER SPECIES					
Stonefly, *Pteronarcys californica*	Whole body	Zinc sulfate	106	14	Nehring, 1976
Atlantic salmon, *Salma salar*	Whole body	Zinc sulfate	51	80	Farmer et al., 1979
Mayfly *Ephemerella grandis*	Whole body	Zinc sulfate	1,130	14	Nehring, 1976
SALT WATER SPECIES					
Softshell clam (adult) *Mya aronaria*	Soft tissue	Zinc chloride	135	49	Schuster and Pringle, 1968
Eastern oyster (adult) *Crassostrea virginica*	Soft tissue	Zinc chloride	23,820	126	Schuster and Pringle, 1968

TABLE E-29

ACUTE TOXICITY OF ZINC TO AQUATIC ORGANISMS

Species	Method	Chemical	Hardness (mg/L as $CaCO_3$)	LC_{50} or EC_{50} (µg/L)**	Reference
FRESH WATER SPECIES					
Cladoceran, *Daphnia magna*	S,M	Zinc Chloride	105	525	Chapman et al., manuscript
Midge, *Chironmus* sp.	-	-	50	18,200	Rehwoldt et al., 1973
Banded Killifish, *Fundulus diaphanus*	S,M	-	55	19,200	Rehwoldt et al., 1972
SALT WATER SPECIES					
Quahog clam (embryo), *Mercenaria mercenaria*	S,U	Zinc Chloride		195	Calabrese and Nelson, 1974
Mussel, *Mytilus edulis planultus*	F,M	Zinc Chloride		3,600	Ahsanullah, 1976
Mussel, *Mytilus edulis planultus*	F,M	Zinc Chloride		4,300	Ahsanullah, 1976
Mysid (juvenille) *Mysidopsis bahia*	F,M	Zinc chloride		499	Lussier et al. 1985
Green crab (adult) *Carcinus meanus*	-	Zinc Sulfate		14,500	Connor, 1972
Mummichog (adult) *Fundulus heteroclitus*	S,U	Zinc Chloride		60,000	Eisler and Hennekey, 1977

TABLE E-29 (Continued)

ACUTE TOXICITY OF ZINC TO AQUATIC ORGANISMS

Species	Method	Chemical	Hardness (mg/L as $CaCO_3$)	LC_{50} or EC_{50} (µg/L)**	Reference
Mummichog (larvae) *Fundulus heteroclitus*	S,U	Zinc Chloride		83,040	Cardin, 1985

S = static; F = flow through; M = measured; U = unmeasured.
results are expressed as zinc, not as the compound.

TABLE E-30

CHRONIC TOXICITY OF ZINC TO AQUATIC ORGANISMS

Species	Test	Chemical	Hardness (mg/L as $CaCO_3$)	Limits (µg/L)	Chronic Value (µg/L)	Reference
FRESH WATER SPECIES						
Cladoceran, *Daphnia magna*	LC	Zinc chloride	211	42-52	46.73	Chapman et al., manuscript
Fathead minnow, *Pimephales promelas*	LC	Zinc sulfate	46	78-145	106.3	Benoit and Holcombe, 1978
SALT WATER SPECIES						
Mysid shrimp, *Mysidopsis bahia*	LC	Zinc chloride	-	120-231	166.5	Lussier et al., 1985

TABLE E-31

OTHER DATA FOR ZINC

Species	Chemical	Duration	Effect	Result (µg/L)*	Reference
FRESH WATER SPECIES					
Cladoceran, *Daphnia magna*	Zinc sulfate	48 hrs	LC_{50}(15°C)	1,100	Cairns et al., 1978
Midge, *Tanytarsus dissimilis*	Zinc chloride	10 days	LC_{50}	36.8	Anderson et al., 1980
SALT WATER SPECIES					
Crab (larvae), *Rhithropanopeus harrisi*	Zinc chloride	16 days	Delayed development	50	Benijts-Claus and Benijts, 1975
Mummichog (adult), *Fundulus heteroclitus*	Zinc chloride	48 hrs	LC_{100}	157,000	Eisler, 1967
Quohog clam (larva), *Mercenaria mercenaria*	Zinc chloride	10 days	LC_{50}	195.4	Calabrese et al., 1977
Soft-shell clam (adult), *Mya arenaria*	Zinc chloride		LC_{50}	7,700	Eisler and Hennekey, 1977
Soft-shell clam (adult), *Mya arenaria*	Zinc chloride		LC_{50}	5,200	Eisler, 1977a
Oyster (larvae), *Crassostrea gigas*	Zinc sulfate	6 days	Growth inhibition	125	Brereton et al., 1973

* Results are expressed as zinc, not the compound.

APPENDIX F

PREDICTION OF SEDIMENT PORE WATER AND CHANNEL WATER CONTAMINANT CONCENTRATIONS AT THE UOP SITE IN EAST RUTHERFORD, NJ

APPENDIX F

PREDICTION OF SEDIMENT PORE WATER AND CHANNEL WATER CONTAMINANT CONCENTRATIONS AT THE UOP SITE IN EAST RUTHERFORD, NJ

F.1 Introduction

Numerous data have been collected on sediment and channel water contaminant concentrations in the tidal channels at the UOP site (see Table G-1). These data were collected at various points within the channels, and so, taken as an aggregate, they yield a reasonable survey of the contaminant concentrations at the site. The purpose of this analysis is to develop data in media where the existing database is insufficient. This is the case for pore water contaminant concentrations in the sediments. No data exist, but data are needed for the risk assessment. This analysis also estimates channel water contaminant concentrations for which data do exist but the database consists of mostly non-detected values for which the detection level is too high to be useful in the assessment. After developing the above mentioned databases, this section provides estimates of the mean and standard deviation for the indicative organisms in each of the three media: sediment, pore water, and channel water.

The first step of the analysis was the generation of an areally averaged sediment concentration. A Monte Carlo algorithm was then used to arrive at a probability distribution for pore water contaminant concentration. An analytic algorithm was employed to estimate the channel water concentration that arises from the mixing of sediment pore water with the over-lying water. [In the case of organic contaminants, suspension of bed sediments is also considered in the estimation of channel water contamination.]

F.2 Calculation of Average Sediment Concentration

Three different sediment sampling programs have taken place at the UOP site in the years 1985 through 1988. Of the many parameters analyzed, attention in this effort was focused on seven organic and seven inorganic contaminants. The site of each sample within any sampling program was assumed to be representative of the channel area adjacent to the sampling point. Thus, the area between two sampling points was divided equally between the two points until the entire channel system was partitioned. This allocation was done separately for the three sampling programs. The area associated with each sample was measured on site maps using a planimeter. The weight given each location is listed in Table F-1.

In order to estimate the channel-wide average sediment concentration, the measured concentration at a sampling station was multiplied by the fraction of the total channel area assumed to be represented by that point in the channel. The resulting products were summed to arrive at an areal average. The standard deviation about this mean was calculated as described in Mendenhall et al. (1981). Samples that yielded concentrations below the analytic detection limit were discarded from the analysis as outliers unless the detection limit was within one order of magnitude of the range of detectable samples. In that case the concentration was assumed to be one-tenth of the detection limit. Samples taken across the channel width (1986 data) were similar and their average was used to represent that point along the channel length. The resulting distributions were, based on visual inspection, $\log_{10}$ normally distributed. Results of this portion of the analysis are presented in Table F-2. For Aroclor® 1248, total chromium, and mercury, data were collected under all three sampling programs. The data from each sampling program are presented separately in this effort and are expressed in terms of dry weight. Total PCB was the analyzed parameter in the first sampling round, but was assumed to be Aroclor® 1248.

TABLE F-1

WEIGHTS ASSIGNED EACH SAMPLING STATION IN THE CALCULATION OF AREAL MEAN SEDIMENT CONCENTRATIONS AT THE UOP SITE IN EAST RUTHERFORD, NJ

STATION	WEIGHTS GIVEN EACH STATION FOR CONTAMINANTS LISTED(*)														
	B(a)A	B(a)P	CHRYSENE	AROCLOR 1248	1,2-DICHLOROBENZENE	1,2-DIPHENYLHYDRAZINE	PHENOL	DEHP	ARSENIC	TOTAL CHROMIUM	CHROMIUM 6+	LEAD	ZINC	CADMIUM	MERCURY
PHASE 2															
SS1	0.00	0.00	0.00	0.00	0.09	0.00	0.09	0.00	0.09	0.09		0.09	0.09	0.09	0.00
SS2	0.23	0.23	0.23	0.20	0.16	0.19	0.16	0.23	0.16	0.16		0.16	0.16	0.16	0.32
SS3	0.38	0.38	0.38	0.00	0.19	0.34	0.19	0.38	0.19	0.19		0.19	0.19	0.19	0.00
SS4	0.00	0.00	0.00	0.00	0.12	0.00	0.12	0.00	0.12	0.12		0.12	0.12	0.12	0.00
SS7	0.00	0.00	0.00	0.09	0.05	0.08	0.05	0.00	0.05	0.05		0.05	0.05	0.05	0.09
SS8	0.03	0.03	0.03	0.34	0.03	0.03	0.03	0.03	0.03	0.03		0.03	0.03	0.03	0.22
SS10	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20		0.20	0.20	0.20	0.20
SS11	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16		0.16	0.16	0.16	0.16
FALL 1986															
4BT				0.12						0.12					0.12
5B				0.07						0.07					0.07
6BT				0.13						0.13					0.13
15BT				0.68						0.68					0.68
FALL 1988															
Aa				0.04					0.04	0.04	0.17				0.04
Ab				0.03					0.03	0.03	0.00				0.03
Ac				0.09					0.09	0.09	0.00				0.09
Ba				0.13					0.13	0.13	0.24				0.13
Bb				0.06					0.06	0.06	0.06				0.06
Bc				0.04					0.04	0.04	0.03				0.04
Ca				0.48					0.48	0.48	0.50				0.48
Cb				0.04					0.04	0.04	0.00				0.04
Cc				0.08					0.08	0.08	0.00				0.08

(*) NOTE: ONLY PARAMETERS COLLECTED IN A GIVEN SAMPLING PROGRAM ARE DESCRIBED.
SAMPLING STATIONS DISCARDED FROM THE ANALYSIS, OR FOR WHICH SAMPLES WERE NOT ANALYZED, HAVE WEIGHTS LISTED AS ZERO AND THEIR WEIGHTS WERE DIVIDED AMONGST NEIGHBORING STATIONS.

TABLE F-2

AREALLY AVERAGED CHANNEL SEDIMENT CONCENTRATION AT THE UOP SITE IN EAST RUTHERFORD, NJ

Concentration Distributions are log(10)normal
and are presented in ppm.

Contaminant	Sediment Concentration		
	Mean of the log Transformed Data	10^Mean (ppm)	Standard Deviation of log Data
Benzo(a)anthracene	-1.09	8.10E-02	0.42
Benzo(a)pyrene	-0.67	2.10E-01	0.39
Chrysene	-0.64	2.30E-01	0.44
Arochlor 1248 (1) (*)	1.38	2.42E+01	1.08
Arochlor 1248 (2)	2.27	1.88E+02	0.11
Arochlor 1248 (3)	1.24	1.73E+01	0.56
1-2-Dichlorobenzene	-0.38	4.20E-01	1.14
1-2-DiPhenylhydrazine	-0.94	1.10E-01	0.66
Phenol	-0.29	5.10E-01	0.75
DEHP	0.66	4.57E+00	0.47
Arsenic	1.33	2.15E+01	0.59
Total Chromium (1)	2.54	3.47E+02	0.58
Total Chromium (2)	2.83	6.79E+02	0.82
Total Chromium (3)	3.40	2.52E+03	0.50
Chromium 6+	0.16	1.40E+00	0.54
Lead	1.69	4.90E+01	0.22
Zinc	2.33	2.14E+02	0.27
Cadmium	0.15	1.40E+00	0.38
Mercury (1)	1.16	1.43E+01	0.34
Mercury (2)	1.04	1.09E+01	0.60
Mercury (3)	1.16	1.44E+01	0.46

* Note: Estimates are presented based upon three sampling programs.
1: Work reported under Phase 2 investigation.
2: Sampling done in the Fall of 1986.
3: Sampling done in the Fall of 1988.

F.3 Calculation of Average Pore Water Concentration

The equilibrium relationship between sediment contaminant concentration and pore water concentration is described by the distribution coefficient (K_d) such that:

$$C_s/C_p = K_d, \qquad (1)$$

where C_s is sediment concentration and C_p is pore water concentration. For the purposes of this analysis, equilibrium will be assumed between sediment solids and pore water. This assumption results in the prediction of the highest possible pore water concentration and so is conservative.

For risk assessment, the average pore water concentration is of interest. In order to estimate this value, a Monte Carlo simulation (as described in Marr and Canale, 1988 and USEPA, 1985 and as used in the water quality model DYNTOX (LimnoTech, 1985)) was pursued using the probability distributions of sediment concentration and contaminant K_d. The simulation randomly picked values of sediment concentration and K_d from their respective distributions and used these values to estimate pore water concentration. This process was repeated 900 times until a stable probability distribution was generated for the mean pore water concentration.

Means and standard deviations of distribution coefficients for the inorganic species are available from Baes and Sharp (1983) and Rai and Zachara (1984). The distribution coefficient for organic species is commonly defined in terms of the sediment fraction organic carbon (f_{oc}) and the contaminant's organic carbon partition coefficient (K_{oc}) such that:

$$K_d = K_{oc} * f_{oc}. \qquad (2)$$

Values of K_{oc} for the organic contaminants of interest are available from Mabey et al. (1982). Sediment fraction organic

carbon was measured at a number of points in the site's channels (see Table D-3) and an areally normalized concentration distribution was generated for this parameter as described above. Probability distributions for K_d for the organic compounds were generated by the multiplication of the f_{oc} distributions by the contaminant's K_{oc}.

The distributions of pore water concentrations resulting from this process were $\log_{10}$ normally distributed and their means and standard deviations are presented in Table F-3.

F.4 Prediction of Channel Water Concentration

The sediments at the UOP site act as a source of contamination for the overlying channel water. For the purposes of modeling, the pore water concentration will be treated as a constant boundary condition at the bottom of the channel. This assumes that the sediment contamination will not change. In all probability, the concentration will decline over time although the decline may be tempered by the contribution of contaminants by groundwater discharge and soil erosion from upland areas of the site. These contributions will be evaluated in the Feasibility Study of Areas 1, 1A, 2 and 5.

Such an analytic algorithm is the one described by Mill et al. (1985) for the prediction of water contamination by the dissolution of a denser than water spill in a river. This algorithm assumes that the spill takes the form of a slick on the bottom of the river and that the boundary between the contaminant slick and the river water is formed by a layer of water saturated with the organic contaminant. Since the contaminants of interest are not present in a pure form, the assumption of a saturated boundary layer is unreasonable. For the purposes of this analysis, the boundary layer will be assumed to be the pore water concentration rather than the

TABLE F-3

ESTIMATED AVERAGE SEDIMENT PORE WATER CONCENTRATION AT THE UOP SITE IN EAST RUTHERFORD, NJ

Concentration Distributions are log(10)normal
and are presented in ppm.

Contaminant	Estimated Pore Water Concentration		
	Mean of the log Transformed Data	10^Mean (ppm)	Standard Deviation of log Data
Benzo(a)anthracene	-5.59	2.60E-06	0.58
Benzo(a)pyrene	-5.79	1.60E-06	0.58
Chrysene	-4.25	5.60E-05	0.60
Arochlor 1248 (1) (*)	-2.49	3.20E-03	1.01
Arochlor 1248 (2)	-1.60	2.50E-02	0.33
Arochlor 1248 (3)	-2.52	3.00E-03	0.71
1-2-Dichlorobenzene	-2.05	9.00E-03	1.17
1-2-DiPhenylhydrazine	-1.92	1.20E-02	0.79
Phenol	0.24	1.70E+00	0.83
DEHP	-7.02	9.53E-08	0.63
Arsenic	0.50	3.20E+00	0.65
Total Chromium (1)	-0.79	1.60E-01	0.75
Total Chromium (2)	-0.51	3.10E-01	0.97
Total Chromium (3)	0.07	1.20E+00	0.75
Chromium 6+	-1.38	4.20E-02	1.09
Lead	-0.35	4.40E-01	0.76
Zinc	1.17	1.49E+01	0.85
Cadmium	-0.68	2.10E-01	0.53
Mercury (1)	-0.96	1.10E-01	0.44
Mercury (2)	-1.09	8.20E-02	0.65
Mercury (3)	-0.97	1.10E-01	0.56

* Note: Estimates are presented based upon three sampling programs.
1: Work reported under Phase 2 investigation.
2: Sampling done in the Fall of 1986.
3: Sampling done in the Fall of 1988.

saturation concentration. The equation describing the water concentration after passage over the entire length of contaminated sediment is described by:

$$C_w = [(C_o - C_p)*\exp(-D_{cw}*L_s/B*H*U)] + C_p \quad (3)$$

Where C_w, C_o, and C_p are the post-spill river water, pre-spill river water, pore water concentrations, respectively. D_{cw} is the diffusion coefficient of the contaminant in water and B is the boundary layer thickness, a function of the hydraulic radius of the river, Manning's coefficient, water's viscosity and the water velocity. U is defined as the river water velocity, H is the water depth over the contamination source, and L_s is the length of the contamination source along the river's length.

While this algorithm was meant to describe river flow, it can be applied to the tidal channels at the UOP site. It can be assumed that the tidal channels are, in effect, rivers whose direction of flow changes periodically. The average channel water was assumed to travel one-half the channel length or the in-flowing tide. Thus, L_s is the channel length.

Equation 3 assumes that the water flowing into the channels is contaminated with a concentration C_o . Since the channels retain some portion of their water at low tide, C_o is a function of the channel concentration during the last tidal cycle. To account for this, equation 3 was modified so that C_o of the current tidal cycle is a fraction of the channel concentration in the previous tidal cycle. Assignment of a value to this fraction will be discussed below. The mixing equation then takes the form of:

$$C_{w(i+1)} = [(r*C_{w(i)} - C_p)*\exp(-D_{cw}*L_s/B*H*U)] + C_p, \quad (4)$$

where $C_{w(i+1)}$ is channel water concentration in the current tidal cycle, $C_{w(i)}$ is the channel water concentration during the previous cycle, and r is the proportion of the channel

volume retained between tidal cycles. This equation was solved iteratively until $C_{w(i+1)}$ reaches a steady-state value.

Selection of parameter values was governed by knowledge of the site's hydrology and the behavior of the contaminants. The thickness of the boundary layer is a function of the viscosity of water, Manning's roughness coefficient, current velocity, and hydraulic radius. Values for the first two parameters were obtained from Chow (1964). The water velocity in the channel was assumed to 1.5 feet per second consistent with observations in the field. The total surface area of the tidal channels is 150,000 square feet and the total length of the channels is 6,600 feet (both as measured by planimetry of site maps). These two values were used to calculate an average channel width (23 feet). The average depth of the channels is assumed to be 4.5 feet at mean tide. From the average width and depth a hydraulic radius of 2.7 feet is computed (assumes the channel is rectangular). The average depth of the channels is 4.5 feet; this approximates the mean tide condition and, again, is consistent with field observation. The average length of the channels was 2,300 feet as measured by planimetry.

The diffusion coefficient of the contaminant in water was calculated as described in Lyman et al. (1982). This method depends upon the viscosity of water and the molar volume of the compound. For organic compounds, the molar volume was calculated by summing the incremental volume of the constituent atoms as described by Lyman et al. (1982). For the inorganic chemicals, the likely environmental form of the metal was identified in Stumm and Morgan (1981) and the molar volume was calculated using that form's molecular weight and density as described in CRC (1981).

The value of the retention factor (r) used in equation 4 was estimated assuming that one-half of the channels retained 2 feet of water between tidal cycles. While this suggests that the channels retain approximately 15 percent of their water between cycles, a conservative value of 25 percent was employed in order to account for any contaminant that might be returned

from the creeks down the estuary. This is conservative since the creeks that the channels drain into have relatively high flow rates accounting for significant dilution factors.

The use of equation 4 amounts to a linear transformation of the pore water concentration (C_p). Given this, the pore water concentrations developed in the Monte Carlo simulations were multiplied by a constant to give channel water concentrations. These channel water concentrations were also found to be $\log_{10}$ normally distributed and means and standard deviations of the log values were calculated. The results of this portion of the analysis are presented in Table F-4. It should be noted that the predictions developed in this analysis are consistent with data available on channel water inorganic contaminant concentration.

[It should also be noted that the analysis described above ignores sorption of organic compounds to suspended sediment in the channels. This mechanism is addressed below.]

F.5 Consideration of Sorption to Suspended Sediments

In the analysis described in Section F.4, only dissolved species were considered. This ignores the contribution of suspended bed sediments to the total concentration observable in the channel. Because this process could be important to very hydrophobic compounds and because no organic compounds were detected in channel water to serve as calibration for the above model, efforts were made to include the effects of sorption for organic compounds. This was done by assuming that: 1) the total suspended sediment concentration in the channels is equal to the 5-year average value (28 ppm) observed at a nearby location in Berry's Creek (Konsevich, 1989), 2) that all of the suspended sediment is derived from the bed sediment and, so, 3) the suspended sediment is contaminated the same extent as the bed sediment. Thus,

$$C_{WS} = C_S * TSS$$

TABLE F-4

ESTIMATES OF AVERAGE CHANNEL WATER CONCENTRATION
AT THE UOP SITE IN EAST RUTHERFORD, NJ

Concentration Distributions are log(10)normal
and are presented in ppm.

Contaminant	Estimated Channel Water Concentration		
	Mean of the log Transformed Data	10^Mean (ppm)	Standard Deviation of log Data
Benzo(a)anthracene	-8.27	5.40E-09	0.58
Benzo(a)pyrene	-8.49	3.30E-09	0.58
Chrysene	-6.93	1.20E-07	0.60
Arochlor 1248 (1) (*)	-5.15	7.10E-06	1.01
Arochlor 1248 (2)	-4.26	5.50E-05	0.33
Arochlor 1248 (3)	-5.18	6.60E-06	0.71
1-2-Dichlorobenzene	-4.59	2.60E-05	1.17
1-2-DiPhenylhydrazine	-4.58	2.60E-05	0.79
Phenol	-2.22	6.00E-03	0.83
DEHP	-9.91	1.24E-10	0.63
Arsenic	-1.79	1.60E-02	0.65
Total Chromium (1)	-3.05	8.80E-04	0.75
Total Chromium (2)	-2.78	1.70E-03	0.97
Total Chromium (3)	-2.20	6.40E-03	0.75
Chromium 6+	-3.62	2.40E-04	1.09
Lead	-2.58	2.60E-03	0.76
Zinc	-0.89	1.30E-01	0.85
Cadmium	-2.94	1.20E-03	0.53
Mercury (1)	-3.44	3.60E-04	0.44
Mercury (2)	-3.57	2.70E-04	0.65
Mercury (3)	-3.46	3.50E-04	0.56

* Note: Estimates are presented based upon three sampling programs.
1: Work reported under Phase 2 investigation.
2: Sampling done in the Fall of 1986.
3: Sampling done in the Fall of 1988.

where C_{ws} is the channel water concentration contributed by suspended sediment sorbed material, C_s is the bed sediment concentration (as above), and TSS is the suspended sediment concentration. As described above, C_{ws} was estimated by a Monte Carlo routine from a probability distribution about C_s.

The total channel water contamination by organic compounds (C_T) was estimated by summing C_{ws} and the estimated dissolved contaminant concentration (C_w from Table F-3) or:

$$C_T = C_{ws} + C_w \tag{6}$$

These results are presented in Table F-5 along with the unchanged estimates of channel water inorganic contaminant concentration from Table F-4.

F.6 Sources of Uncertainity

While the Monte Carlo analysis was undertaken in order to include an estimate of uncertainity, only sediment concentration and K_d were assumed to contribute to the uncertainity of the concentration estimates. There are other uncertain parameters in this analysis, but the nature of their probability density functions are not known.

The hydrologic conditions assumed at the site are average ones. This ignores variation in tidal height, tidal velocity, and increased bed scour associated with storms. Since the risk assessment process is based on long-term exposures, episodic changes in tidal characteristics might be ignored.

The value of the retention factor employed in equation 4 was selected with few data regarding the tidal dynamics of the estuary system. As such, every effort was made to select a conservative value.

Finally, in the estimation of suspended sediment contribution to stream contamination by organic compounds, uncertainties arise. Most importantly, the mechanism modeled ignores the sorption of contaminants to very small material

TABLE F-5

ESTIMATES OF AVERAGE CHANNEL WATER CONCENTRATION AT THE UOP SITE IN EAST RUTHERFORD, NJ

(Includes compounds dissolved in channel water and organic compounds adsorbed to suspended solids)

Contaminant	Estimated Channel Water Concentration		
	Mean of the log Transformed Data	10^Mean (ppm)	Standard Deviation of log Data
Benzo(a)anthracene	-5.64	2.30E-06	0.41
Benzo(a)pyrene	-5.24	5.81E-06	0.40
Chrysene	-5.16	6.90E-06	0.43
Arochlor 1248 (1) (*)	-3.11	7.74E-04	1.10
Arochlor 1248 (2)	-2.27	5.38E-03	0.11
Arochlor 1248 (3)	-3.28	5.26E-04	0.59
1-2-Dichlorobenzene	-4.41	3.91E-05	1.13
1-2-DiPhenylhydrazine	-4.51	3.08E-05	0.76
Phenol	-2.22	6.03E-03	0.83
DEHP	-3.91	1.24E-04	0.47
Arsenic	-1.79	1.64E-02	0.65
Total Chromium (1)	-3.05	8.83E-04	0.75
Total Chromium (2)	-2.78	1.67E-03	0.97
Total Chromium (3)	-2.20	6.38E-03	0.75
Chromium 6+	-3.62	2.39E-04	1.09
Lead	-2.58	2.62E-03	0.76
Zinc	-0.89	1.29E-01	0.85
Cadmium	-2.94	1.16E-03	0.53
Mercury (1)	-3.44	3.62E-04	0.44
Mercury (2)	-3.57	2.70E-04	0.65
Mercury (3)	-3.46	3.51E-04	0.56

* Note: Estimates are presented based upon three sampling programs.
1: Work reported under Phase 2 investigation.
2: Sampling done in the Fall of 1986.
3: Sampling done in the Fall of 1988.

(e.g., colloids, humic acids, etc.) not measured in TSS. The concentration of these agents is not available, but it is felt that the conservative assumption that all TSS comes from the contaminated bed sediment more than accounts for their potential contribution.

APPENDIX G

SELECTION OF THE INDICATOR COMPOUNDS FOR THE CHANNELS AT THE SITE

APPENDIX G

SELECTION OF THE INDICATOR COMPOUNDS FOR THE CHANNELS AT THE SITE

G.1 Background Concentrations

Before selecting Indicator Compounds for the site, it is important (i) to compare the concentrations of the organic compounds to concentrations of the same compounds found in other industrial areas and (ii) to compare the concentrations of the heavy metals both to the concentrations of the same metals found in other industrial areas and to the concentration of the metal in the crustal abundance.

For the sediments, Aroclor® 1248 is enriched in comparison to other urban waterways (Charles Menzie, personal communication), while the PAHs are found in concentrations similar to those in other urban waterways (GRI, 1988). For the sediments, these concentrations are useful in terms of comparisons to crustal abundance (east of the Mississippi River):

	Crustal Abundance	
Heavy Metal	Average Concentration (ppm)	Concentration Range (ppm)
Chromium	33	1 to 1,000
Mercury	0.081	0.01 to 3.4
Lead	14	<10 to 300
Zinc	40	<5 to 2,900
Cadmium	0.2	(NA)

The first four values come from the U.S. Geological Survey (1984), while the last value comes from Taylor (1964). Viewed against these data for natural abundance, the concentrations of

chromium and mercury are most elevated in the sediments on the site.

G.2 Selection of the Indicator Compounds

The eleven indicator compounds from the human health risk assessment (Rev. 2, Appendix B) were reviewed as candidates for the ecological risk assessment. Eight of the eleven compounds were retained for the ecological risk assessment based on frequency of detection and concentrations detected in the surface water and sediments. Three compounds: 1,1,2,2-tetrachloroethane, benzene and arsenic were deleted because they are not detected or are detected at very low concentrations and frequencies in channels A, B and C. These three compounds were selected for the human health risk assessment because they are present in the North Ditch, which is not included in the ecological assessment.

Other compounds in the sediments and water of Channels A, B and C were reviewed for their potential toxicity to wildlife. Based on this review, three additional indicator compounds were selected: cadmium, phenol and di(2-ethylhexyl) phthalate.

PAHs associated with carcinogenic health effects were evaluated in the human health assessment. For the ecological assessment, the sum of all PAHs are evaluated.

The final list of eleven Indicator Compounds for the ecological study of the channels is:

Heavy Metals (n = 5)

- cadmium
- chromium
- lead
- mercury
- zinc

Organic Compounds (n = 6)

1,2-dichlorobenzene
phenol
Aroclor® 1248, a polychlorinated biphenyl (PCB)
1,2-diphenylhydrazine
di(2-ethylhexyl) phthalate (DEHP)
polycyclic aromatic hydrocarbons (PAHs)

G.3 Data Summary

Table G-1 lists the indicator compound results for analyses of sediments and surface water in the channels at the UOP site in East Rutherford, NJ. These are the raw data used as input in risk analyses through this report. The top two sections of the Table G-1 list the results for analyses of sediments and surface water samples taken in 1983 and 1985, grouped according to the zone (Channel A, B, C, or Control Area as shown on Figure G-1) at the site. The bottom section lists the results for analyses of sediment samples taken in 1988, also grouped according to the zone at the site. All water and sediment sampling locations are shown on Figure G-2 (in back pocket). As shown in Figure G-1, Channel A is the most upstream channel, Channel B receives water from Channel A, and Channel C drains the flow from the first two channels into Berry's Creek. The Control zone, referred to as D, is across Berry's Creek in Walden Swamp.

In Table G-1, all measurements are reported in parts per million (ppm) for each medium. In sediments and tissues, one part per million is equivalent to one milligram per kilogram, and in surface water one part per million is equivalent to one milligram per liter. Throughout the table, values reported above the limit of detection are shown in scientific notation, while values reported below the limit of detection are shown in decimal notation (for the limit of detection for that compound) preceded by a "less than" (<) symbol.

Table G-1
Monitoring Results for Sediments, Surface Water, and Fish Tissues from the Channels

zone -->	a	a	a	a	a	a	a	a	a	a	a	b	b	b
SEDIMENT	SS-1		SS-2	SS-2	6BTRB	6BTCTR	6BTLB	SS-7	4BTRB	4BTCTR	5BRB			
1983 & 1985	11/83		11/83	1/85	1/85	1/85	1/85	1/85	11/83	11/83	11/83			
					0-15"	0-15"	0-15"		0-15"	0-15"	0-15"			
	(mg/kg)		(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)			
Aroclor-1016					<2	<2	<3		<5	<5	<5			
Aroclor-1221					<2	<2	<3		<5	<5	<5			
Aroclor-1232					<2	<2	<3		<5	<5	<5			
Aroclor-1242					<2	<2	<3		<5	<5	<5			
Aroclor-1248					1.0E+2	1.4E+2	5.7E+2		7.6E+1	1.8E+2	3.7E+2			
Aroclor-1254					<2	<2	<3		<5	<5	<5			
Aroclor-1260					<2	<2	<3		<5	<5	<5			
Aroclor-1262					<2	<2	<3		<5	<5	<5			
Total PCBs														
Acenaphthylene	<20		<0.5	<0.2				<2						
Anthracene	<20		6.0E-1	<0.2				<2						
Benzo(g,h,i)perylene	<20		<0.5	<0.4				<4						
Fluoranthene	<20		1.0E+0	1.8E-1				2.4E+0						
Phenanthrene	<20		6.0E-1	<0.2				1.3E+0						
Acenaphthene	<20		<0.5	<0.2				<2						
Pyrene	<20		1.1E+0	1.3E-1				2.2E+0						
Naphthalene	<20		<0.5	<0.2				<2						
3,4-Benzofluoranthene	<20		7.0E-1	6.2E-2				<4						
2-Chloronaphthalene	<20		<0.2	<0.2				<2						
Dibenzo(a,h)anthracene	<20		<0.5	<0.4				<4						
Benzo(a)pyrene	<20		5.0E-1	9.1E-2				<4						
Benzo(a)anthracene	<20		6.0E-1	1.1E-1				<2						
Indeno(1,2,3-c,d)pyrene	<20		<0.5	<0.4				<4						
Total PAHs														
bis(2-Ethylhexyl)phthalate	<20		1.0E+1	9.5E+0				2.2E+1						
1,2-Dichlorobenzene	3.8E+1		6.0E-1	2.3E-1				1.6E+0						
1,2-Diphenylhydrazine	<20		<0.5	2.2E+0				<4						
Phenol	1.7E+1		1.6E+0	1.3E+0				2.7E+0						
Cadmium	6.4E-1		2.4E+0	2.7E+0				2.1E+0						
Chromium (total)	6.6E+2		4.0E+3	4.1E+3	2.5E+4	2.4E+4	2.4E+4	4.8E+2	2.7E+3	4.0E+3	1.1E+4			
Lead	8.8E+1		8.0E+1	9.0E+1				1.3E+2						
Mercury				4.0E+1	2.2E+2	1.7E+2	9.1E+1	2.0E+1	4.5E+0	2.3E+1	1.4E+2			
Zinc	3.0E+2		5.6E+2	4.3E+2				2.5E+2						

SURFACE WATER	ST3*	ST3
1983 & 1985	11/83	1/85
	(mg/l)	(mg/l)
Total PCBs		<0.05
Acenaphthylene	<0.01	<0.01
Anthracene	<0.01	<0.01
Benzo(g,h,i)perylene	<0.02	<0.02
Fluoranthene	<0.01	<0.01
Phenanthrene	<0.01	<0.01
Acenaphthene	<0.01	<0.01
Pyrene	<0.01	<0.01
Naphthalene	<0.01	<0.01
3,4-Benzofluoranthene	<0.02	<0.02
2-Chloronaphthalene	<0.01	<0.01
Dibenzo(a,h)anthracene	<0.02	<0.02
Benzo(a)pyrene	<0.02	<0.02
Benzo(a)anthracene	<0.01	<0.01
Indeno(1,2,3-c,d)pyrene	<0.02	<0.02
Benzo(k)fluoranthene	<0.02	<0.02
Fluorene	<0.01	<0.01
2-Methylnaphthalene	<0.02	
Total PAHs		
bis(2-Ethylhexyl)phthalate	<0.01	<0.01
1,2-Dichlorobenzene	<0.02	1.9E-1
1,2-Diphenylhydrazine	<0.02	4.5E-2
Phenol	<0.01	
Cadmium	4.0E-3	2.0E-3
Chromium (total)	<0.005	<0.02
Lead	8.3E-2	4.8E-2
Mercury		<0.0005
Zinc	7.0E-2	4.0E-2

SEDIMENT	Ac	Aa	Ab	Ba	Bb	Bc*
1988	88.0	88.0	88.0	88.0	88.0	88.0
(dry weight)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
Aroclor-1248	1.2E+2	6.6E+1	8.2E+1	1.0E+2	4.1E+1	7.7E+0
Chromium (total)	2.7E+4	4.2E+3	5.6E+3	9.0E+3	4.1E+3	7.8E+3
Mercury	1.1E+2	5.8E+1	2.4E+1	3.3E+1	2.2E+1	8.1E+1

* Average Of Duplicates
** Average Of Subsites

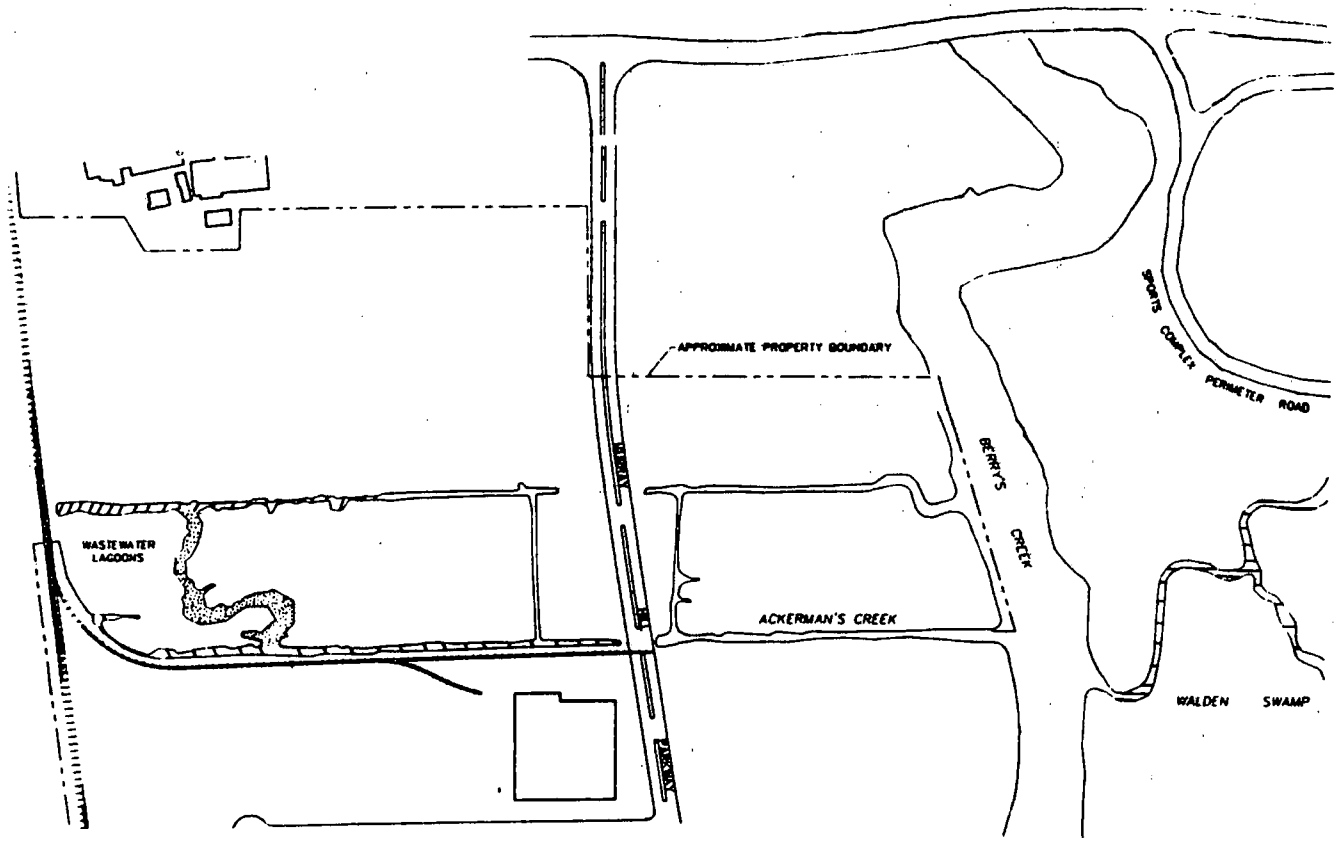
Table G-1
Monitoring Results for Sediments, Surface Water, and Fish Tissues from the Channels

zone -->	c	c	c	c	c	c	c	c	d	e
---	---	---	---	---	---	---	---	---	---	---
SEDIMENT	SS-8	SS-3		SS-4	SS-10	15BTRB	15BTCTR	15BTLB	CNTRL	SS-11
1983 & 1985	1/85	11/83		11/83	1/85	1/85	1/85	1/85		1/85
						0-15"	0-15"	0-15"		
	(mg/kg)	(mg/kg)		(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
---	---	---		---	---	---	---	---	---	---
Aroclor-1016						<0.01	<0.01	<0.05		
Aroclor-1221						<0.01	<0.01	<0.05		
Aroclor-1232						<0.01	<0.01	<0.05		
Aroclor-1242						<0.01	<0.01	<0.05		
Aroclor-1248						<0.01	1.6E+1	5.0E+0		
Aroclor-1254						<0.01	<0.01	<0.05		
Aroclor-1260						<0.01	<0.01	<0.05		
Aroclor-1262						<0.01	<0.01	<0.05		
Total PCBs										
Acenaphthylene	<0.2	<0.5		<20	<0.2					<0.2
Anthracene	1.7E-1	<0.5		<20	<0.2					<0.2
Benzo(g,h,i)perylene	<0.4	<0.5		<20	<0.4					<0.4
Fluoranthene	<0.2	5.0E-1		<20	1.6E-1					2.2E-1
Phenanthrene	<0.2	<0.5		<20	<0.2					<0.2
Acenaphthene	<0.2	<0.5		<20	<0.2					<0.2
Pyrene	<0.2	6.0E-1		<20	1.2E-1					<0.2
Naphthalene	<0.2	<0.5		<20	<0.2					<0.2
3,4-Benzofluoranthene	2.1E-1	5.0E-1		<20	2.0E-1					<0.4
2-Chloronaphthalene	<0.2	<0.5		<20	<0.2					<0.2
Dibenzo(a,h)anthracene	<0.4	<0.5		<20	<0.4					<0.4
Benzo(a)pyrene	1.2E-1	5.0E-1		<20	1.2E-1					<0.4
Benzo(a)anthracene	1.6E-1	<0.5		<20	1.0E-1					<0.2
Indeno(1,2,3-c,d)pyrene	<0.4	<0.5		<20	<0.4					<0.4
Total PAHs										
bis(2-Ethylhexyl)phthalate	3.5E+0	1.6E+1		<20	1.0E+0					6.3E+0
1,2-Dichlorobenzene	1.3E-1	1.5E+0		1.1E+1	<0.2					<0.2
1,2-Diphenylhydrazine	7.8E+0	<0.5		<20	<0.5					<0.4
Phenol	1.0E+0	8.0E-1		9.0E-1	<0.4					
Cadmium	1.2E+0	1.3E+0		2.0E-1	1.6E+0					4.3E+0
Chromium (total)	5.2E+3	1.6E+2		1.3E+2	2.2E+2	2.1E+2	3.4E+1	3.4E+2		1.0E+2
Lead	7.5E+1	2.5E+1		2.5E+1	5.0E+1					5.6E+1
Mercury	6.7E+0				8.6E+0	4.5E+0	8.9E-1	4.3E+0		8.1E+0
Zinc	2.1E+2	1.5E+2		5.3E+1	3.0E+2					2.3E+2

---	---	---	---	---
SURFACE WATER	ST7*	ST6	ST6	ST8
1983 & 1985	1/85	11/83	1/85	1/85
	(mg/l)	(mg/l)	(mg/l)	(mg/l)
---	---	---	---	---
Total PCBs	<0.05		<0.05	<0.05
Acenaphthylene	<0.01	<0.01	<0.01	<0.01
Anthracene	<0.01	<0.01	<0.01	<0.01
Benzo(g,h,i)perylene	<0.02	<0.02	<0.02	<0.02
Fluoranthene	<0.01	<0.01	<0.01	<0.01
Phenanthrene	<0.01	<0.01	<0.01	<0.01
Acenaphthene	<0.01	<0.01	<0.01	<0.01
Pyrene	<0.01	<0.01	<0.01	<0.01
Naphthalene	<0.01	<0.01	<0.01	<0.01
3,4-Benzofluoranthene	<0.02	<0.02	<0.02	<0.02
2-Chloronaphthalene	<0.01	<0.01	<0.01	<0.01
Dibenzo(a,h)anthracene	<0.02	<0.02	<0.02	<0.02
Benzo(a)pyrene	<0.02	<0.02	<0.02	<0.02
Benzo(a)anthracene	<0.01	<0.01	<0.01	<0.01
Indeno(1,2,3-c,d)pyrene	<0.02	<0.02	<0.02	<0.02
Benzo(k)fluoranthene	<0.02	<0.02	<0.02	<0.02
Fluorene	<0.01	<0.01	<0.01	<0.01
2-Methylnaphthalene				
Total PAHs				
bis(2-Ethylhexyl)phthalate	<0.01	<0.01	<0.01	
1,2-Dichlorobenzene	<0.01	<0.01	<0.01	<0.01
1,2-Diphenylhydrazine	<0.02	<0.02	<0.02	<0.02
Phenol		<0.01	<0.01	
Cadmium	<0.001	8.0E-3	<0.001	<0.001
Chromium (total)	<0.02	<0.005	<0.02	<0.02
Lead	1.2E-2	7.0E-2	1.2E-2	1.6E-2
Mercury	<0.0005		<0.0005	<0.0005
Zinc	1.2E-1	1.3E-2	9.0E-2	7.0E-2

---	---	---	---	---
SEDIMENT	Ca	Cc	Cb	D**
1988	88.0	88.0	88.0	88.0
(dry weight)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
---	---	---	---	---
Aroclor-1248	5.3E+0	2.3E+1	1.4E+1	1.1E+0
Chromium (total)	1.1E+3	6.6E+2	1.8E+3	1.0E+3
Mercury	5.9E+0	7.7E+0	1.9E+1	8.9E+1

* Average Of Duplicates
** Average Of Subsites



LEGEND

- AREA A
- AREA B
- AREA C
- AREA D

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SITE AREAS
UOP SITE
EAST RUTHERFORD, NEW JERSEY

Figure G-1

While the water and fish tissue concentrations do not show a trend in concentration as a function of distance, presumably because the tides thoroughly mix the water and the pelagic organisms, the concentrations in the sediments show a trend, decreasing as a function of distance from Channel A to Channel C.

APPENDIX H

ANALYSIS OF EXTRAPOLATION ERROR

APPENDIX H
ANALYSIS OF EXTRAPOLATION ERROR

H.1 Introduction

This appendix has three purposes. The first purpose is to explain the analysis of extrapolation error (AEE) method in greater detail than is done in the main body of the report. The second is to provide an example of the use of the method, using data summary tables in the format used here for the actual analyses. The third purpose is to provide the data summary tables for analyses presented in Chapter 6.

H.2 Explanation of the Method

The following explanation and example is extracted from Analysis of Extrapolation Error (Suter et al. 1986). Analysis of Extrapolation Error (AEE) is a method of calculating the probability that expected environmental concentrations will exceed toxicological benchmarks. The method has two components and is implemented in five steps. The first component is an extrapolation used to estimate the endpoint for the species and chemical from available test data. The second component is a risk component that estimates the probability of exceeding the endpoint from results of the extrapolations.

The first step in implementing the method is to define the endpoint in terms of a toxicological endpoint. The endpoint for the present application is the probability of exceeding maximum acceptable toxicant concentration (MATC) for the Indicator Species exposed to the Indicator Compounds. The second step is to identify the toxicological datum most closely related to the desired endpoint. The relationship between the endpoint and the identified datum is then broken into logical steps to get to the endpoint.

There are three conceivable relationships between the desired endpoint and the nearest toxicological datum. Suppose

that the needed endpoint is a chronic effect for chemical A on brook trout. The first possibility is that the needed datum is available. In that case, there will generally be a concentration associated with an observation (e.g., lowest observed effect level) but no measure of the uncertainty associated with that concentration. In this case, Suter et al. (1986) have a table showing pooled variances of log(10) transformed LC50, EC50, and MATC values from replicate tests. Table values can be used as estimates of variances for either fish (Osteichthyes) or Daphnia. No other taxonomic groups are tabulated.

If the needed datum is not available, there may be a result from an acute test for the same species. In that case, there may be an LC50 for brook trout and chemical A. To make this extrapolation, Suter et al. (1986) provide a table of 18 acute-chronic extrapolations based on linear regression using an errors-in-variables model. Using the linear regression formula $y = a + bx$, one can enter the table for any x and find slope and intercept values for calculating y. For calculating the estimated variance associated with the predicted value, the formula for estimated variance has been reduced to an expression in two variables, F1 and F2 that are included on the table. In extrapolating from a brook trout LC50 to a brook trout MATC for chemical A, based on tests used in the regression, constants are found in Suter et al.'s (1986) Table 4.3, equation 5. The formula is

$$\log(10)(\text{brook trout MATC}) = -1.51 + (1.07) \times (\log(10)(\text{brook trout}))$$

The variance for a predicted Y based on an intital value of X, X_o, is found from:

$$\text{var}(Y\ X_o) = F1 + F2(X_o - \text{Xbar})^2,$$

where F1, F2, and Xbar are found in the tables provided by Suter et al. (1986).

The third situation is that the nearest datum is not for the same species and not for the desired endpoint, and a double extrapolation is needed. For example, the desired endpoint is MATC for brook trout exposed to chemical A, and the nearest datum is an LC_{50} for rainbow trout exposed to the same chemical. Here the double extrapolation is first from taxon X to taxon Y (rainbow trout to brook trout) and the from an acute LC50 result to MATC. For this, Suter et al. (1986) have tabulated constants for taxonomic extrapolations based on linear regression using an errors-in-variables model. Once again, using the linear regression formula $y = a + bx$, one can enter the table for any x and find slope and intercept values for calculating y. Values of F1 and F2 are also included for estimating variance. Suppose that the nearest datum is an LC_{50} of 5300 µg/L for rainbow trout (using the example in Suter et al., 1986). Then the LC_{50} for brook trout is found from Suter et al.'s Table 4.1. There is no taxonomic extrapolation from rainbow trout (Salmo trutta) to brook trout (Salvelinus fontinalis) on the table, so the next higher or genus to genus extrapolation (Salmo-Salvelinus) is used.

$$\log(10)(\text{Salvelinus LC50}) = -0.33 + (1.1) \times (\log(10)(\text{Salmo } LC_{50}))$$

Then the extrapolated Salvelinus result is used as the X value in the acute-chronic extrapolation previously described. Variance at each extrapolation step is calculated using the tabulated values of F1 and F2. The linear regression formula for the second extrapolation is $z = c + dy$, using y from the first extrapolation. The total variance for the two extrapolations is:

$$\text{var}\ (z|X_o) = \text{var}(Z|Y_o) + d^2(\text{var}(Y|X_o))$$

If we assume that the expected environmental concentration of chemical A and the estimated end point are independent and

log-normally distributed, the risk can be calculated and the cumulative probability obtained from a statistical z table. The probability that extrapolated chronic benchmark concentration is less than the expected environmental concentration (and so, the species is protected) is one minus the tabulated z value. Multiplying by one hundred gives percent risk.

The following tables show calculations. Tables H-1 to H-16 show extrapolations to indicator species chronic values (the first component of the method). Tables H-17 and H-18 show calculations of the probability that environmental concentrations exceed chronic endpoints.

TABLE H-1

FILE: AE2CR3FU.WR1

REGRESSION CALCULATION FOR AEE METHOD: FUNDULUS & CR (III)

FROM: FUNDULUS LC50
AS: LC50
TO: FUNDULUS MATC
AS: MATC
VALUE (V) = 31500 ug/l BY: EQ.7, CLASS = M

Y = BX + A

X (LOG(V)) = 4.498
SLOPE= 0.730
INTERCEPT= -0.700

F1= 0.370
F2= 0.020
XBAR= 3.250
Xo= 4.498

CALCULATED VALUES:

Y = 2.584
VAR(Y|Xo) = 0.401

ANTILOG Y = 383.5011
ANTILOG VAR 2.518637

TABLE H-2

FILE: AE2CR3BL.WR1

REGRESSION CALCULATION FOR AEE METHOD: DECAPODS & CR(III)

FROM: GAMMARUS SP. LC50
AS: AMPHIPODA
TO: DECAPODA LC50
AS: DECAPODA
VALUE (V) = 3200 ug/l

Y = BX + A

X (LOG(V)) =	3.505
SLOPE=	1.670
INTERCEPT=	0.650
F1=	2.730
F2=	0.250
XBAR=	0.890
Xo=	3.505

CALCULATED VALUES:

Y =	6.504
VAR(Y|Xo) =	4.440

ANTILOG Y = 3188603.
ANTILOG VAR 27526.58

FROM: DECAPODA LC50
TO: DECAPODA MATC
BY: EQ.7, CLASS = M

Z = DY + C

Y=	6.504
SLOPE(D)	0.730
INTERCEPT=	-0.700
F1=	0.370
F2=	0.020
YBAR=	3.250
Yo=	6.504

CALCULATED VALUES:

Z =	4.048
VAR(Z|Yo) =	0.582

ANTILOG Z = 11159.07
ANTILOG VAR 3.816966

VARIANCE FOR 2-STEP
2.948

ANTILOG VAR 886.4665

TABLE H-3

FILE: AE2CR6FU.WR1

REGRESSION CALCULATION FOR AEE METHOD: FUNDULUS & CR (VI)

FROM: FUNDULUS LC50
AS: LC50
TO: FUNDULUS MATC
AS: MATC
VALUE (V) = 74010 BY: EQ.7, CLASS = M

Y = BX + A

X (LOG(V)) = 4.869
SLOPE= 0.730
INTERCEPT= -0.700

F1= 0.370
F2= 0.020
XBAR= 3.250
Xo= 4.869

CALCULATED VALUES:

Y = 2.855
VAR(Y|Xo) = 0.422

ANTILOG Y = 715.4544
ANTILOG VAR 2.645099

TABLE H-4

FILE: AE2CR6BL.WR1

REGRESSION CALCULATION FOR AEE METHOD: BLUE CRABS & CR(VI)

FROM: BLUE CRAB LC50
AS: LC50
TO: BLUE CRAB MATC
AS: MATC
VALUE (V) = 93390 ug/l BY: EQ.7, CLASS = M

Y = BX + A

X (LOG(V)) =	4.970
SLOPE=	0.730
INTERCEPT=	-0.700
F1=	0.370
F2=	0.020
XBAR=	3.250
Xo=	4.970

CALCULATED VALUES:

Y =	2.928
VAR(Y|Xo) =	0.429

ANTILOG Y = 847.8504
ANTILOG VAR 2.686511

TABLE H-5

FILE: AE2CR6DE.WR1

REGRESSION CALCULATION FOR AEE METHOD: SHRIMP & CR(VI)

FROM: GAMMARUS PSEUDOL. LC50	FROM: DECAPODA LC50
AS: AMPHIPODA	
TO: DECAPODA LC50	TO: DECAPODA MATC
AS: DECAPODA	
VALUE (V) = 67.1 ug/l	BY: EQ.7, CLASS = M
Y = BX + A	Z = DY + C
X (LOG(V)) = 1.827	Y= 3.701
SLOPE= 1.670	SLOPE(D) 0.730
INTERCEPT= 0.650	INTERCEPT= -0.700
F1= 2.730	F1= 0.370
F2= 0.250	F2= 0.020
XBAR= 0.890	YBAR= 3.250
Xo= 1.827	Yo= 3.701
CALCULATED VALUES:	CALCULATED VALUES:
Y = 3.701	Z = 2.001
VAR(Y|Xo) = 2.949	VAR(Z|Yo) = 0.374
ANTILOG Y = 5019.108	ANTILOG Z = 100.3361
ANTILOG VAR 889.9431	ANTILOG VAR 2.366253

VARIANCE FOR 2-STEP
1.946

ANTILOG VAR 88.26254

TABLE H-6

FILE: AE2HGFUN.WR1

REGRESSION CALCULATION FOR AEE METHOD: FUNDULUS & INORGANIC Hg

FROM: FUNDULUS LC50
AS: LC50
TO: FUNDULUS MATC
AS: MATC
VALUE (V) = 67.4 ug/l BY: EQ.7, CLASS = M

Y = BX + A

X (LOG(V)) =	1.829
SLOPE=	0.730
INTERCEPT=	-0.700
F1=	0.370
F2=	0.020
XBAR=	3.250
Xo=	1.829

CALCULATED VALUES:

Y =	0.635
VAR(Y|Xo) =	0.410

ANTILOG Y = 4.314413
ANTILOG VAR 2.572788

TABLE H-7

FILE: AE2HGBLU.WR1

REGRESSION CALCULATION FOR AEE METHOD: BLUE CRAB & INORGANIC Hg

FROM: CARCINUS MAENAS LC50
AS: SURROGATE LC50
TO: BLUE CRAB MATC
AS: MATC
VALUE (V) = 14 ug/l BY: EQ.7, CLASS = M

Y = BX + A

X (LOG(V)) =	1.146
SLOPE=	0.730
INTERCEPT=	-0.700
F1=	0.370
F2=	0.020
XBAR=	3.250
Xo=	1.146

CALCULATED VALUES:

Y =	0.137
VAR(Y|Xo) =	0.459

ANTILOG Y = 1.369851
ANTILOG VAR 2.874256

TABLE H-8

FILE: AE2HG2DC.WR1

REGRESSION CALCULATION FOR AEE METHOD: SHRIMP & INORGANIC Hg

FROM: WHITE SHRIMP LC50
AS: SURROGATE
TO: GRASS SHRIMP MATC
AS: MATC
VALUE (V) = 17 ug/l BY: EQ. 7, CLASS = METALS

Y = BX + A

X (LOG(V)) =	1.230
SLOPE=	0.730
INTERCEPT=	-0.700
F1=	0.370
F2=	0.020
XBAR=	3.250
Xo=	1.230

CALCULATED VALUES:

Y =	0.198
VAR(Y|Xo) =	0.452

ANTILOG Y = 1.578438
ANTILOG VAR 2.828601

TABLE H-9

FILE: AE2ME2FU.WR1

REGRESSION CALCULATION FOR AEE METHOD: FUNDULUS & METHYL Hg

FROM: FUNDULUS LC50
AS: LC50
TO: FUNDULUS MATC
AS: MATC
VALUE (V) = 51.1 ug/l BY: EQ.7, CLASS = M

Y = BX + A

X (LOG(V)) =	1.708
SLOPE=	0.730
INTERCEPT=	-0.700
F1=	0.370
F2=	0.020
XBAR=	3.250
Xo=	1.708

CALCULATED VALUES:

Y =	0.547
VAR(Y|Xo) =	0.418

ANTILOG Y = 3.524903
ANTILOG VAR 2.615347

TABLE H-10

FILE: AE2MEBLU.WR1

REGRESSION CALCULATION FOR AEE METHOD: DECAPODS & METHYL Hg

FROM: GAMMARUS DUEBENI LC50
AS: AMPHIPODA
TO: DECAPOD LC50
AS: DECAPODA
VALUE (V) = 150 ug/l

Y = BX + A

X (LOG(V)) =	2.176
SLOPE=	1.670
INTERCEPT=	0.650
F1=	2.730
F2=	0.250
XBAR=	0.890
Xo=	2.176

CALCULATED VALUES:

Y =	4.284
VAR(Y|Xo) =	3.144

ANTILOG Y = 19234.12
ANTILOG VAR 1391.578

FROM: DECAPODA LC50
TO: DECAPODA MATC
BY: EQ.7, CLASS = M

Z = DY + C

Y=	4.284
SLOPE(D)	0.730
INTERCEPT=	-0.700
F1=	0.370
F2=	0.020
YBAR=	3.250
Yo=	4.284

CALCULATED VALUES:

Z =	2.427
VAR(Z|Yo) =	0.391

ANTILOG Z = 267.5302
ANTILOG VAR 2.462555

VARIANCE FOR 2-STEP
2.067

ANTILOG VAR 116.5631

TABLE H-11

FILE: AE2ZNFUN.WR1

REGRESSION CALCULATION FOR AEE METHOD: FUNDULUS & ZINC

FROM: FATHEAD LIFECYCLE
AS: FM MATC
TO: FUNDULUS MATC
AS: ALL FISH MATC
VALUE (V) = 106.3

Y = BX + A

X (LOG(V)) =	2.027
SLOPE=	0.790
INTERCEPT=	-0.040
F1=	0.330
F2=	0.010
XBAR=	1.800
Xo=	2.027

CALCULATED VALUES:

Y =	1.561
VAR(Y|Xo) =	0.331

ANTILOG Y = 36.38825
ANTILOG VAR 2.140489

TABLE H-12

FILE: AE2ZNBLU.WR1

REGRESSION CALCULATION FOR AEE METHOD: BLUE CRAB & ZINC

FROM: CARCINUS MAENAS LC50
AS: SURROGATE
TO: BLUE CRAB MATC
AS: MATC
VALUE (V) = 1000 BY: EQ. 7, CLASS = METALS

Y = BX + A

X (LOG(V)) =	3.000
SLOPE=	0.730
INTERCEPT=	-0.700
F1=	0.370
F2=	0.020
XBAR=	3.250
Xo=	3.000

CALCULATED VALUES:

Y =	1.490
VAR(Y|Xo) =	0.371

ANTILOG Y = 30.90295
ANTILOG VAR 2.350985

TABLE H-13

FILE: AE2ZNDEC.WR1

REGRESSION CALCULATION FOR AEE METHOD: SHRIMP & ZINC

FROM: COROPHIUM VOL. LC50
AS: AMPHIPODA
TO: DECAPOD LC50
AS: DECAPODA
VALUE (V) = 4683 ug/l

Y = BX + A

X (LOG(V)) =	3.671
SLOPE=	1.670
INTERCEPT=	0.650
F1=	2.730
F2=	0.250
XBAR=	0.890
Xo=	3.671

CALCULATED VALUES:

Y =	6.780
VAR(Y|Xo) =	4.663

ANTILOG Y = 6022479.
ANTILOG VAR 46007.50

FROM: DECAPODA LC50
TO: DECAPODA MATC
BY: EQ.7, CLASS = M

Z = DY + C

Y=	6.780
SLOPE(D)	0.730
INTERCEPT=	-0.700
F1=	0.370
F2=	0.020
YBAR=	3.250
Yo=	6.780

CALCULATED VALUES:

Z =	4.249
VAR(Z|Yo) =	0.619

ANTILOG Z = 17751.53
ANTILOG VAR 4.160890

VARIANCE FOR 2-STEP
3.104

ANTILOG VAR 1270.596

TABLE H-14

FILE: AE242FUN.WR1

REGRESSION CALCULATION FOR AEE METHOD: FUNDULUS & PCB AROCLOR 1242

FROM: FATHEAD LIFECYCLE
AS: FM MATC
TO: FUNDULUS MATC
AS: ALL FISH MATC
VALUE (V) = 9 ug/l

Y = BX + A

X (LOG(V)) =	0.954
SLOPE=	0.790
INTERCEPT=	-0.040
F1=	0.330
F2=	0.010
XBAR=	1.800
Xo=	0.954

CALCULATED VALUES:

Y =	0.714
VAR(Y|Xo) =	0.337

ANTILOG Y = 5.174299
ANTILOG VAR 2.173467

TABLE H-15

FILE: AE254FUN.WR1

REGRESSION CALCULATION FOR AEE METHOD: FUNDULUS & PCB AROCLOR 1254

FROM: FATHEAD LIFECYCLE
AS: FM MATC
TO: FUNDULUS MATC
AS: ALL FISH MATC
VALUE (V) = 2.9 ug/l

Y = BX + A

X (LOG(V)) =	0.462
SLOPE=	0.790
INTERCEPT=	-0.040
F1=	0.330
F2=	0.010
XBAR=	1.800
Xo=	0.462

CALCULATED VALUES:

Y =	0.325
VAR(Y|Xo) =	0.348

ANTILOG Y = 2.114922
ANTILOG VAR 2.227879

TABLE H-16

FILE: AE2PCBLU.WR1

REGRESSION CALCULATION FOR AEE METHOD: DECAPODS & PCBs

FROM: GAMMARUS PSEUSOL. LC50
AS: AMPHIPODA
TO: DECAPOD LC50
AS: DECAPODA
VALUE (V) = 29 ug/l

Y = BX + A

X (LOG(V)) =	1.462
SLOPE=	1.670
INTERCEPT=	0.650
F1=	2.730
F2=	0.250
XBAR=	0.890
Xo=	1.462

CALCULATED VALUES:

Y =	3.092
VAR(Y|Xo) =	2.812

ANTILOG Y = 1236.529
ANTILOG VAR 648.4998

FROM: DECAPODA LC50
TO: DECAPODA MATC
BY: EQ. 5, CLASS = ALL

Z = DY + C

Y=	3.092
SLOPE(D)	1.070
INTERCEPT=	-1.510
F1=	0.590
F2=	0.000
YBAR=	3.130
Yo=	3.092

CALCULATED VALUES:

Z =	1.799
VAR(Z|Yo) =	0.590

ANTILOG Z = 62.90120
ANTILOG VAR 3.890451

VARIANCE FOR 2-STEP
3.809

ANTILOG VAR 6446.969

TABLE H-17

FILE: AEESUM52.WR1

Probability calculations for AEE method.
Entries are means and variances of EEC and extrapolated MATCs.

SOURCE	XBAR	VAR	Z	Cum. Prob.	100*(1 - Cum Prob)
CHROMIUM (III)***					
MODEL EEC**	0.80	0.563			
CRAB	4.05	2.948	-1.73353	0.042	95.8
SHRIMP	4.05	2.948	-1.73353	0.042	95.8
FUNDULUS	2.58	0.401	-1.81748	0.034	96.6
CHROMIUM (VI)					
MODEL EEC**	-0.62	1.188			
CRAB	2.93	0.429	-2.79007	0.003	99.7
SHRIMP	2.00	1.946	-1.48051	0.069	93.1
FUNDULUS	2.86	0.422	-2.73860	0.003	99.7
MERCURY, INORGANIC***					
MODEL EEC**	-0.46	0.314			
CRAB	0.14	0.459	-0.67920	0.248	75.2
SHRIMP	0.20	0.452	-0.75201	0.227	77.3
FUNDULUS	0.64	0.410	-1.28726	0.099	90.1

* From Suter et al. (1986) Eq. 4.22
** From Appendix Table F-4. Data are Log(10) transformed and expressed in units of ug/l.
*** 1988 data only.

TABLE H-18

FILE: AEESUM51.WR1

Probability calculations for AEE method.
Entries are means and variances of EEC and extrapolated MATCs.

SOURCE	XBAR	VAR		Z	Cum. Prob.	100*(1 - Cum Prob)
MERCURY, METHYL***						
MODEL EEC**	-0.46	0.314				
CRAB	2.43	2.067		-1.87113	0.031	96.9
SHRIMP	2.43	2.067		-1.87113	0.031	96.9
FUNDULUS	0.55	0.418		-1.17731	0.119	88.1
ZINC						
MODEL EEC**	2.11	0.722				
CRAB	1.49	0.371	****	0.59290	0.278	27.8
SHRIMP	4.25	3.104		-1.09348	0.138	86.2
FUNDULUS	1.56	0.331	****	0.53488	0.298	29.8
PCB***						
MODEL EEC**	-0.28	0.348				
CRAB	1.80	3.809		-1.01967	0.154	84.6
SHRIMP	1.80	3.809		-1.01967	0.154	84.6
FUNDULUS, 1242	0.71	0.337		-1.20091	0.115	88.5
FUNDULUS, 1254	0.33	0.348		-0.72514	0.233	76.7

* From Suter et al. (1986) Eq. 4.22
** From Appendix Table F-4. Data are Log(10) transformed and expressed in units of ug/l.
*** 1988 data only.
**** Mean Expected Environmental Concentration exceeds

APPENDIX I

REFERENCES

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REFERENCES

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PART II

HUMAN HEALTH FOOD CHAIN ASSESSMENT

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1. INTRODUCTION

A human health risk assessment for the terrestrial portions of the UOP Site was submitted to the NJDEP in 1987. At the request of NJDEP, the aquatic portion of the site known as Area 4 or Ackerman's Creek, was incorporated into Revision 2 of the risk assessment report (hereafter referred to as Rev. 2), which was submitted to the NJDEP in October 1988.

Appendix B of Rev. 2 assessed direct exposure pathways, i.e. dermal contact with sediments and surface water. This document addresses indirect exposure pathways, including potential ingestion of blue crab from Ackerman's Creek and edible fish from Berry's Creek. It is organized in a similar manner to Rev. 2, Appendix B, and consists of 9 sections and 2 appendices. Following Section 1, Introduction, the remaining sections are:

2. Selection of Indicator Compounds
3. Toxicity Assessment
4. Identification of Exposure Pathways
5. Calculation of Crab and Fish Tissue Concentrations
6. Calculation of Dose
7. Estimation of Risk
8. Presentation of Risks and Uncertainties
9. Summary of Risks associated with Direct and Indirect Exposure Pathways in Area 4

This document draws heavily from the information presented in the Rev. 2 text and Appendix B. Much of the material supplied in the earlier document is referred to rather than repeated here. This document also relies heavily on the information provided by the ecological risk assessment (Part I, herein).

2. SELECTION OF INDICATOR COMPOUNDS

The selection of indicator compounds is performed to identify a subset of chemicals that are the most toxic, mobile, persistent, and prevalent chemicals present on the site and, therefore, are those which potentially pose the greatest adverse health effects due to exposure in Area 4. Eleven indicator chemicals for Area 4 were selected in Rev. 2, Appendix B, Section B.2:

- Arsenic
- Benzene
- Chromium
- 1,2-Dichlorobenzene
- 1,2-Diphenylhydrazine
- Lead
- Mercury
- Carcinogenic Polynuclear Aromatic Hydrocarbons (PAHs), consisting of three PAH compounds detected in Ackerman's Creek sediment or surface water:
    - Benzo(a)anthracene
    - Benzo(a)pyrene
    - Chrysene
- Polychlorinated Biphenyls (PCBs)
- 1,1,2,2-Tetrachloroethane
- Zinc

These chemicals will be used in this document to assess the potential risks associated with indirect exposure pathways.

3. TOXICITY ASSESSMENT

Toxicity profiles for the eleven Indicator Compounds were provided in Section 2 of Rev. 2, and will not be repeated here. However, the updated EPA-derived dose-response values which will be used to calculate risks for this report are presented in Table 3-1.

TABLE 3-1

DOSE-RESPONSE DATA FOR INDICATOR CHEMICALS SELECTED

FOR HEALTH RISK ASSESSMENT, AREA 4

UOP SITE

Chemical	RfD[a] (mg/kg/day)	Source	Potential Oral Carcinogenic Potency (mg/kg/day)$^{-1}$	Source
Arsenic	0.001	U.S. EPA, 1989	0.175[b]	IRIS, 1989
Benzene	-	-	0.029	U.S. EPA, 1989
Chromium III	1	IRIS, 1989	-	-
Chromium VI	0.005	U.S. EPA, 1989	-	-
1,2-Dichlorobenzene	0.4	U.S. EPA, 1989	-	-
1,2-Diphenylhydrazine	-	-	0.8	U.S. EPA, 1989
Lead	0.0014	U.S. EPA, 1986	-	-
Mercury	0.0003	U.S. EPA, 1989	-	-
Carcinogenic PAH	-	-	11.5	U.S. EPA, 1986
PCB	-	-	7.7	U.S. EPA, 1989
1,1,2,2-Tetrachloroethane	-	-	0.2	IRIS, 1989
Zinc	0.2	U.S. EPA, 1989	-	-

Notes:

(a) RfD = Risk Reference Dose
(b) Value includes modifying factor of 10, as indicated in IRIS.

Sources:

U.S. EPA (1986). Superfund Public Health Evaluation Manual. EPA 540/1-86/060.
IRIS (1989). The U.S. EPA Integrated Risk Information System (on-line database).
U.S. EPA (1989). Health Effects Assessment Summary Tables. First Quarter FY 1989. March 1989.

4. IDENTIFICATION OF EXPOSURE PATHWAYS

At any site humans may potentially be exposed to chemicals in a variety of media such as soil, sediment, or water. The potential receptor may be exposed directly by ingestion, inhalation, or dermal absorption. Indirect exposure through food chains, such as the consumption of contaminated biota, are also potential exposure pathways. Direct exposures to chemicals in sediments and surface water in the stream channels were examined in Rev. 2, Appendix B. Potential indirect exposures due to ingestion of contaminated blue crab or edible fish are assessed here. Of the other indicator organisms, only duck are also eaten by humans (the mussel species, Congeria, is not consumed by people). Duck ingestion was not assessed in this document, however, since ingestion of duck is likely to be a minor portion of an individual's diet, and would occur infrequently and to few individuals.

4.1 Ingestion of Blue Crab

It is possible that blue crab live in Ackerman's Creek. Although it is unlikely that many people, if any, crab directly in Ackerman's Creek (due to difficult access to the area), it was assumed as a hypothetical, conservative case, that blue crab from the stream channels are caught and ingested by local fisherman.

4.2 Ingestion of Edible Fish

It is highly unlikely that edible fish live in Ackerman's Creek. However, a survey of organisms performed by the Hackensack Meadowlands Development Commission (HMDC, 1987) indicates that both white perch and striped bass live in Berry's Creek. These organisms may become contaminated from ingestion of organisms from Ackerman's Creek or contact with water that washes out at low tide.

During numerous visits to the site, ENSR staff have not observed fishermen in Berry's Creek in the vicinity of Ackerman's Creek. The low or no incidence of fishing is thought to be attributed to access difficulty to Berry's Creek near Ackerman's Creek and to general public knowledge of the areawide mercury contamination. Therefore, fish ingestion is not thought to be a viable exposure scenario, but is included in the assessment as a hypothetical case.

5. CALCULATION OF CRAB AND FISH CONCENTRATIONS

5.1 Quantitative Approach

No data are available on actual crab or edible fish tissue concentrations in Ackerman's or Berry's Creek. Therefore, crab or edible fish tissue concentrations were estimated using the method described in the Superfund Exposure Assessment Manual (U.S. EPA, 1988), i.e.:

$$C_{c,f} = BCF \times C_w$$

Where:

$C_{c,f}$ = Concentration in crab or fish tissue
BCF = Bioconcentration factor
C_w = Concentration in channel water (modelled mean, see Part I, Appendix F)

A chemical-specific BCF reflects the tendency of the compound to accumulate in fish tissues directly from the water. BCFs differ from compound to compound, by species, and with environmental conditions. Bioconcentration factors for this assessment were obtained from the available literature. An attempt was made to use BCFs from the same species or from a taxonomically similar species, where available, and to find BCFs from field studies, where conditions were similar to those found at the site. For several chemicals, no BCFs were found for species that are taxonomically similar to the blue crab. This does not imply, however, that these chemicals will not potentially be found in the tissues of blue crab. Although the chemicals lacking BCFs for blue crab were not quantified, they were assessed qualitatively (see Appendix C). Table 5-1 lists the BCFs that were used for edible fish and blue crab in this assessment. Tables 5-2 and 5-3 present the actual calculations and resulting crab and fish concentrations.

Data from NYDEC (1981) indicates that PCBs strongly partition into the crab hepatopancreas (which is seldom eaten) rather than in the edible muscle tissue; 1979-80 Hudson River

TABLE 5-1

BIOCONCENTRATION FACTORS USED FOR CALCULATION OF FISH AND CRAB TISSUE CONCENTRATIONS

	FISH			CRAB		
Chemical	BCF (l/kg fish)	Species[a]	Source[b]	BCF (l/kg crab)	Species[a]	Source[b]
Benzene	5.21	estimated	AWQC	—	—	—
1,2-Dichlorobenzene	89	BS	AWQC, Lyman	—	—	—
1,2-Diphenylhydrazine	24.9	estimated	AWQC	—	—	—
Carcinogenic PAH	2630	FM	Lyman	—	—	—
PCB	3,500	LMB	AWQC	230,000[c]	BC	AWQC
1,1,2,2-Tetrachloroethane	14	BS	Lyman	—	—	—
Arsenic	44	?	SPHEM	—	—	—
Chromium VI	2.8	RT	AWQC	440	C	AWQC
Chromium III	2.8	RT	AWQC	440	C	AWQC
Lead	49	?	SPHEM	—	—	—
Mercury	4994	FM	AWQC	129	AL	AWQC
Zinc	432	FF	AWQC	8800	BC	AWQC

Notes:

[a]Species: BS = Bluegill sunfish; FM = Fathead Minnow; RT = Rainbow Trout; FF = Flagfish; BC = Blue Crab; C = Crustacean; AL = American Lobster; LMB = Large Mouth Bass

[b]Sources: AWQC = Ambient Water Quality Criteria, U.S. EPA 1980; Lyman = Lyman et al., 1982; SPHEM = Superfund Public Health Evaluation Manual, U.S. EPA 1986.

[c]Value derived for Aroclor 1254.

— No BCF was found for a taxonomically similar species

TABLE 5-2
CALCULATION OF CRAB TISSUE CONCENTRATION

CHEMICAL	MODELED MEAN CHANNEL WATER CONCENTRATION (MG/L)	BCF (L/KG)	MODELED MEAN CRAB TISSUE CONCENTRATION (MG/KG)
ARSENIC	1.64E-02	*	0.00E+00
BENZENE	2.00E-03	*	0.00E+00
CHROMIUM III	6.14E-03	440	2.70E+00
CHROMIUM VI	2.39E-04	440	1.05E-01
1,2-DICHLOROBENZENE	3.91E-05	*	0.00E+00
1,2-DIPHENYLHYDRAZINE	3.08E-05	*	0.00E+00
LEAD	2.62E-03	*	0.00E+00
MERCURY	3.51E-04	129	4.53E-02
CARCINOGENIC PAH	1.50E-05	*	0.00E+00
PCB	5.26E-04	230000	1.21E+02
1,1,2,2-TETRACHLORETHANE	5.50E-03	*	0.00E+00
ZINC	1.29E-01	8800	1.14E+03

UOPCRAB5.WK1

NOTE:

* = No BCF for taxonomically similar species was found for this chemical in the available literature; therefore it is not assessed

TABLE 5-3
CALCULATION OF FISH TISSUE CONCENTRATION

CHEMICAL	MODELED MEAN CHANNEL WATER CONCENTRATION (MG/L)	BCF (L/KG)	MODELED MEAN FISH TISSUE CONCENTRATION (MG/KG)
ARSENIC	1.64E-02	44	7.22E-01
BENZENE	2.00E-03	5.21	1.04E-02
CHROMIUM III	6.14E-03	2.8	1.72E-02
CHROMIUM VI	2.39E-04	2.8	6.69E-04
1,2-DICHLOROBENZENE	3.91E-05	89	3.48E-03
1,2-DIPHENYLHYDRAZINE	3.08E-05	24.9	7.67E-04
LEAD	2.62E-03	49	1.28E-01
MERCURY	3.51E-04	4994	1.75E+00
CARCINOGENIC PAH	1.50E-05	2630	3.94E-02
PCB	5.26E-04	3500	1.84E+00
1,1,2,2-TETRACHLORETHANE	5.50E-03	14	7.70E-02
ZINC	1.29E-01	432	5.57E+01

UOPFISH4.wk1

survey data indicate that only 1-10% of the total PCBs in blue crab are in the muscle tissue (depending on date and location of sample collection).

The BCF for PCBs used in this assessment was derived for blue crab whole body under field conditions (Nimmo et al., 1975). Because the experimental research was conducted in the field, the BCF derived reflects exposure to all pathways that naturally occur in the field, such as column water, pore water, sediments and food chain exposure routes. Therefore, the average PCB concentration in the edible muscle tissue is estimated through the following discussion.

According to Wiedow (1981), the average blue crab weighs 212 grams, much of which is water (Engle, personal communication). Assuming that the shell accounts for approximately 25% of the total crab weight, the weight of the shell can be calculated as:

212 grams x 25% = 53 grams

Wiedow (1981) further states that 60-70% of the total weight of the soft body parts is muscle, and 20% is hepatopancreas. The mean weight of the hepatopancreas is 19 grams; therefore, the mean weight of the soft body parts is:

19 grams x 5 = 95 grams

The total weight of the crab, excluding water, is then approximately:

95 grams + 53 grams = 148 grams

Using the NYDEC (1981) data, we can assume that 90% of the PCBs partitions into the hepatopancreas, and 10% partitions equally into the shell and soft body parts other than hepatopancreas. Therefore, the PCB concentration in shell and soft body parts (including edible muscle tissue) can be calculated as follows:

121 mg/kg (whole body PCBs concentration; see Table 5-2) x 148 grams (Total crab weight, excluding water) x $\frac{1\ kg}{1000\ g}$ = 17.91 mg PCBs per crab

17.91 mg PCBs x 0.10 (Portion in shell and soft body parts excluding hepatopancreas) = 1.79 mg PCB

1.79 mg PCBs ÷ 129 grams (Weight of shell and soft body parts excluding hepatopancreas) x $\frac{1000\ grams}{1\ kg}$ = 13.9 mg/kg

The PCBs concentration which will be used to estimate risks from crab ingestion is therefore 13.9 mg PCB/kg crab.

6. CALCULATION OF DOSE

Once the concentration of each chemical in fish or crab has been determined, these values are used in the following equation to estimate human ingestion intake dose:

Dose due to fish or crab ingestion (mg/kg/day) = Fish or crab concentration (mg/kg fish or crab) x gastrointestinal (GI) absorption adjustment factor (unitless) x daily fish ingestion rate (kg fish/day) x local intake factor x proportion remaining after cooking (unitless)/body weight (kg)

An extensive literature search was undertaken to obtain the latest scientific data on GI absorption for the Indicator Compounds. The absorption adjustment factors that were obtained as a result of this search are listed in Table 6-1; the studies that these factors were obtained from are described in Appendix A.

Surveys of fish and shellfish consumption that examined age-specific and region-specific ingestion rates were undertaken by Rupp et al. (1979). Data in the surveys were statistically analyzed to determine percentage distributions of individuals eating different quantities of marine and freshwater fish and shellfish. Fish and crab consumption rates used in this assessment are based on the average reported by Rupp et al. for persons residing in mid-Atlantic states; these are listed in Table 6-2. Assessment of edible fish utilized the freshwater finfish consumption rates, and blue crab consumption was based on the shellfish consumption rates. It was conservatively assumed that once per year an individual's daily shellfish intake will consist of blue crab from Ackerman's Creek. Similarly, it was assumed that once per year an individual will get their daily intake of fish from the most contaminated portion of Berry's Creek, i.e., at the mouth of Ackerman's Creek.

TABLE 6-1

GI ABSORPTION ADJUSTMENT FACTORS

Chemical	GI AAF
Arsenic	1
Benzene	1
Chromium III	0.01
Chromium VI	0.15
1,2-Dichlorobenzene	1
1,2-Diphenylhydrazine	1
Lead: Child	0.5
Adult	0.15
Mercury	0.0001
Carcinogenic PAH	0.9
PCB	1
1,1,2,2-Tetrachloroethane	1
Zinc	0.3

TABLE 6-2

FISH CONSUMPTION PATTERNS FOR THE MID-ATLANTIC REGION

	Average Consumption (g/day)	
	Freshwater Finfish	Shellfish
Young Children (1-11 years)	0.33	0.74
Older Children (12-18 years)	0.30	1.56
Adults (19-70 years)	0.96	3.92

Source: Rupp *et al*. (1980)

Several papers have reported that, at a minimum, 25% of the PCBs in fillets are lost during cooking (Puffer and Gossett, 1983; Zabik et al., 1979). This assessment conservatively assumes that 25% of all organic Indicator Compounds are lost during cooking, although it is likely that the smaller, lower molecular weight organics with high vapor pressures are lost to a greater degree.

Body weights for the three age groups examined were determined from data provided in Anderson et al. (1985). These age-specific body weights are:

- 1-11 years: 20 kg
- 12-18 years: 51 kg
- 19-70 years: 70 kg

All intake calculations are provided in, Appendix B. Tables 6-3 and 6-4 summarize age-specific and total daily and lifetime intake doses calculated for crab and fish ingestion.

Table 6-3

INTAKE DOSES BASED ON CONSUMPTION OF BLUE CRAB
POTENTIALLY CAUGHT IN ACKERMAN'S CREEK

CHEMICAL	CRAB CONCENTRATIONS (mg/kg)	YOUNGER CHILD AVERAGE DAILY INTAKE (mg/kg/day)	YOUNGER CHILD LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)	OLDER CHILD AVERAGE DAILY INTAKE (mg/kg/day)	OLDER CHILD LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)	ADULT AVERAGE DAILY INTAKE (mg/kg/day)	ADULT LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)	TOTAL AVERAGE DAILY INTAKE	TOTAL LIFETIME DAILY INTAKE
Arsenic	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Benzene	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Chromium III	2.70E+00	2.74E-09	4.30E-10	2.27E-09	1.94E-10	4.14E-09	3.13E-09	9.14E-09	3.76E-09
Chromium VI	1.05E-01	1.60E-09	2.51E-10	1.32E-09	1.13E-10	2.42E-09	1.83E-09	5.33E-09	2.19E-09
1,2-Dichlorobenzene	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
1,2-Diphenylhydrazine	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Lead	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mercury	4.53E-02	4.59E-13	7.21E-14	3.80E-13	3.26E-14	6.95E-13	5.26E-13	1.53E-12	6.31E-13
Carcinogenic PAH	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
PCB	1.39E+01	1.06E-06	1.66E-07	8.75E-07	7.50E-08	1.60E-06	1.21E-06	3.53E-06	1.45E-06
1,1,2,2-Tetrachloroethane	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Zinc	1.14E+03	3.47E-05	5.45E-06	2.87E-05	2.46E-06	5.24E-05	3.97E-05	1.16E-04	4.76E-05

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SOURCE: ENSR 1989

Table 6-4

INTAKE DOSES BASED ON CONSUMPTION OF EDIBLE FISH
CAUGHT IN BERRY'S CREEK

CHEMICAL	FISH CONCENTRATIONS (mg/kg)	YOUNGER CHILD AVERAGE DAILY INTAKE (mg/kg/day)	YOUNGER CHILD LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)	OLDER CHILD AVERAGE DAILY INTAKE (mg/kg/day)	OLDER CHILD LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)	ADULT AVERAGE DAILY INTAKE (mg/kg/day)	ADULT LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)	TOTAL AVERAGE DAILY INTAKE	TOTAL LIFETIME DAILY INTAKE
Arsenic	7.22E-01	3.25E-08	5.11E-09	1.17E-08	1.00E-09	2.71E-08	2.05E-08	7.13E-08	2.66E-08
Benzene	1.04E-02	3.51E-10	5.52E-11	1.26E-10	1.08E-11	2.93E-10	2.22E-10	7.70E-10	2.88E-10
Chromium III	1.72E-02	7.75E-12	1.22E-12	2.78E-12	2.39E-13	6.46E-12	4.89E-12	1.70E-11	6.34E-12
Chromium VI	6.69E-04	4.52E-12	7.10E-13	1.62E-12	1.39E-13	3.77E-12	2.85E-12	9.91E-12	3.70E-12
1,2-Dichlorobenzene	3.48E-03	1.18E-10	1.85E-11	4.23E-11	3.62E-12	9.80E-11	7.42E-11	2.58E-10	9.63E-11
1,2-Diphenylhydrazine	7.67E-04	2.59E-11	4.07E-12	9.31E-12	7.98E-13	2.16E-11	1.63E-11	5.68E-11	2.12E-11
Lead	1.28E-01	2.88E-09	4.53E-10	1.04E-09	8.88E-11	7.21E-10	5.46E-10	4.64E-09	1.09E-09
Mercury	1.75E+00	7.88E-12	1.24E-12	2.83E-12	2.43E-13	6.57E-12	4.97E-12	1.73E-11	6.45E-12
Carcinogenic PAH	3.94E-02	1.20E-09	1.88E-10	4.31E-10	3.69E-11	9.98E-10	7.56E-10	2.63E-09	9.81E-10
PCB	1.84E+00	6.22E-08	9.77E-09	2.23E-08	1.92E-09	5.18E-08	3.92E-08	1.36E-07	5.09E-08
1,1,2,2-Tetrachloroethane	7.70E-02	2.60E-09	4.09E-10	9.35E-10	8.01E-11	2.17E-09	1.64E-09	5.70E-09	2.13E-09
Zinc	5.57E+01	7.53E-07	1.18E-07	2.71E-07	2.32E-08	6.27E-07	4.75E-07	1.65E-06	6.16E-07

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7. ESTIMATION OF RISK

Two types of risk will be evaluated in this document: noncarcinogenic and carcinogenic. Noncarcinogenic risk estimates are determined by dividing total daily intake doses (calculated as described in Section 6) for each appropriate indicator chemical by its RfD (See Table 3-1). This results in a ratio, or Hazard Index (HI). If the HI is less than or equal to one, the associated body dose level is likely to be without significant adverse noncarcinogenic health risk to humans. If the ratio is greater than one, the predicted intake dose could potentially result in adverse health effects. HIs for individual chemicals are summed to give a total HI. HIs should only be summed for chemicals with similar toxic endpoints, e.g. HIs for chemicals that produce liver effects should not be added to HIs for chemicals that produce only primary irritation. However, all HIs can be summed as a screening tool. If the resulting total HI is less than one, then no adverse health effects would be anticipated. However, if the total HI exceeds one, further analysis should be performed.

Lifetime incremental carcinogenic risk estimates are determined by multiplying the lifetime intake levels by cancer potency slopes, which results in a cancer probability. To put these incremental risk levels into perspective, they should be evaluated against a reference (no-effect) level. However, for carcinogens, it is current USEPA policy that cancer induction is a non-threshold phenomenon. Therefore, any exposure is presumed to pose some probability of causing cancer, and a "safe" (i.e. no-effect) level cannot be determined. USEPA guidelines indicate that the target total individual lifetime incremental carcinogenic risk resulting from exposures at a Superfund site may range anywhere from 10^{-4} (that is, a dose corresponding to one excess chance in 10,000) to 10^{-7} (one excess chance in 10,000,000). Remedial alternatives being considered should be able to reduce total potential cancer risks to individuals to levels within this range (U.S. EPA, 1986).

8. PRESENTATION OF RISKS AND UNCERTAINTIES

Potential risks associated with ingestion of blue crab from Ackerman's Creek and fish from Berry's Creek are presented in Tables 8-1 and 8-2. It can be seen from these tables that no noncarcinogenic adverse chronic health effects would be anticipated from ingestion of either species. The summary hazard index for crab ingestion is 0.00058 and for fish ingestion is 0.0000829.

Total carcinogenic risk for both fish and crab were within the EPA target range for site clean-up. Total potential carcinogenic risk for crab ingestion is $1x10^{-5}$ and for fish ingestion is $4x10^{-7}$.

Uncertainties in the risk assessment process come from a variety of sources. Most of these were discussed in detail in Section 10 of Rev. 2, and will not be repeated here. An uncertainty associated specifically with the scenarios presented here concerns the calculation of crab and fish tissue concentration using BCFs. It is not known whether the BCFs used reflect actual site conditions for the species that are being studied; except for PCBs, where the BCF for crab tissue was derived under field conditions. Use of these BCFs could overestimate or underestimate actual fish and crab tissue concentrations at the site. For crab, because the BCF for PCBs was derived in the field, the estimate of crab tissue concentration of PCBs is likely to be realistic. The BCFs for crab tissue for several organic compounds and metals are unknown. Thus, the quantitative contribution to human health risks from these compounds is uncertain. For fish, it is much more likely that these BCFs overestimate tissue concentrations since they were derived for whole body rather than fillet.

Table 8-1

POTENTIAL RISKS ASSOCIATED WITH INGESTING CRAB FROM ACKERMAN'S CREEK

The hazard index for potential chronic effects and the 95% upper bound excess lifetime cancer risk are shown for a person consuming crab caught at the UOP Site
Scenarios include loss of organic chemicals through cooking processes.

CHEMICAL	Crab Concentration (mg/kg)	Potential Average Daily Intake (mg/kg/day)	Potential Lifetime Average Daily Intake (mg/kg/day)	Potential Threshold Effects: Reference Dose (mg/kg/day)	Potential Threshold Effects: Hazard Index	Potential Cancer Risk: Carcinogenic Potency Factor (day-kg/mg)	Potential Cancer Risk: Excess Lifetime Cancer Risk
Arsenic	0.00E+00	0.00E+00	0.00E+00	1.00E-03	0.00E+00	1.75E-01	0.00E+00
Benzene	0.00E+00	0.00E+00	0.00E+00	(a)	(a)	2.90E-02	0.00E+00
Chromium III	2.70E+00	9.14E-09	3.76E-09	1.00E+00	9.14E-09	(b)	(b)
Chromium VI	1.05E-01	5.33E-09	2.19E-09	5.00E-03	1.07E-06	(b)	(b)
1,2-Dichlorobenzene	0.00E+00	0.00E+00	0.00E+00	4.00E-01	0.00E+00	(b)	(b)
1,2-Diphenylhydrazine	0.00E+00	0.00E+00	0.00E+00	(a)	(a)	8.00E-01	0.00E+00
Lead	0.00E+00	0.00E+00	0.00E+00	1.40E-03	0.00E+00	(b)	(b)
Mercury	4.53E-02	1.53E-12	6.31E-13	3.00E-04	5.11E-09	(b)	(b)
Carcinogenic PAH	0.00E+00	0.00E+00	0.00E+00	(a)	(a)	1.15E+01	0.00E+00
PCB	1.39E+01	3.53E-06	1.45E-06	(a)	(a)	7.70E+00	1.12E-05
1,1,2,2-Tetrachloroethane	0.00E+00	0.00E+00	0.00E+00	(a)	(a)	2.00E-01	0.00E+00
Zinc	1.14E+03	1.16E-04	4.76E-05	2.00E-01	5.79E-04	(b)	(b)
				Summed Index	5.80E-04	Summed Risk	1.12E-05

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(a): The U.S. EPA has not derived an AIC for this compound.
(b): This compound is not thought to be carcinogenic.

SOURCE: ENSR 1989

Table 8-2

POTENTIAL RISKS ASSOCIATED WITH INGESTING FISH FROM BERRY'S CREEK

The hazard index for potential chronic effects and the 95% upper bound excess lifetime cancer risk are shown for a person consuming fish caught at the UOP Site
Scenarios include loss of organic chemicals through cooking processes.

				Potential Threshold Effects		Potential Cancer Risk	
CHEMICAL	Fish Concentration (mg/kg)	Potential Average Daily Intake (mg/kg/day)	Potential Lifetime Average Daily Intake (mg/kg/day)	Reference Dose (mg/kg/day)	Hazard Index	Carcinogenic Potency Factor (day-kg/mg)	Excess Lifetime Cancer Risk
Arsenic	7.22E-01	7.13E-08	2.66E-08	1.00E-03	7.13E-05	1.75E-01	4.66E-09
Benzene	1.04E-02	7.70E-10	2.88E-10	(a)	(a)	2.90E-02	8.34E-12
Chromium III	1.72E-02	1.70E-11	6.34E-12	1.00E+00	1.70E-11	(b)	(b)
Chromium VI	6.69E-04	9.91E-12	3.70E-12	5.00E-03	1.98E-09	(b)	(b)
1,2-Dichlorobenzene	3.48E-03	2.58E-10	9.63E-11	4.00E-01	6.44E-10	(b)	(b)
1,2-Diphenylhydrazine	7.67E-04	5.68E-11	2.12E-11	(a)	(a)	8.00E-01	1.70E-11
Lead	1.28E-01	4.64E-09	1.09E-09	1.40E-03	3.31E-06	(b)	(b)
Mercury	1.75E+00	1.73E-11	6.45E-12	3.00E-04	5.76E-08	(b)	(b)
Carcinogenic PAH	3.94E-02	2.63E-09	9.81E-10	(a)	(a)	1.15E+01	1.13E-08
PCB	1.84E+00	1.36E-07	5.09E-08	(a)	(a)	7.70E+00	3.92E-07
1,1,2,2-Tetrachloroethane	7.70E-02	5.70E-09	2.13E-09	(a)	(a)	2.00E-01	4.26E-10
Zinc	5.57E+01	1.65E-06	6.16E-07	2.00E-01	8.25E-06	(b)	(b)
				Summed Index	8.29E-05	Summed Risk	4.08E-07

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(a): The U.S. EPA has not derived an AIC for this compound.
(b): This compound is not thought to be carcinogenic.

SOURCE: ENSR 1989

9. SUMMARY OF RISKS ASSOCIATED WITH DIRECT AND INDIRECT EXPOSURE PATHWAYS IN AREA 4

Risks associated with both direct and indirect exposure pathways in Area 4 are summarized in Table 9-1. Direct contact with sediments and surface water were assessed in Rev. 2, Appendix B. Hazard indices for all direct exposure scenarios are much less than one, indicating that the associated body dose levels would not be expected to cause chronic noncarcinogenic adverse health effects. Total carcinogenic risks associated with direct exposure scenarios are also very low. The total risks for sediments range from 8×10^{-8} to 4×10^{-7}, and for surface water range from 1×10^{-9} to 1×10^{-8}. All carcinogenic risks are therefore below the U.S. EPA target risk range for site clean-up considerations. It can be concluded that direct contact with sediments and surface water in the Area 4 stream channels would not pose a threat to human health under reasonable worst case conditions.

Risks associated with indirect exposure due to fish and crab ingestion were assessed in this document. Noncarcinogenic adverse chronic health effects would not be anticipated from ingestion of either species. The summary hazard index for crab ingestion is 0.00058 and for fish ingestion is 0.0000829. Total carcinogenic risks for both fish and crab were within the EPA target range for site clean-up considerations. Total potential carcinogenic risk for crab ingestion is 1×10^{-5}. Although certain chemicals were not quantitatively assessed in the crab ingestion scenario, using worst-possible-case assumptions for the organic compounds, the calculated potential cancer risk will still be in the 10^{-5} range (see Appendix C). Total potential carcinogenic risk for fish ingestion is 4×10^{-7}.

TABLE 9-1

AREA 4 RISK SUMMARY TABLE

UOP SITE, EAST RUTHERFORD, NJ

Medium	Total Hazard Index		Total Cancer Risk	
	Maximum	Average	Maximum	Average
Sediments	0.0019	0.00036	4×10^{-7}	8×10^{-8}
Surface Water	0.00038	0.00015	1×10^{-8}	1×10^{-9}
Fish	0.0000829		4×10^{-7}	
Crab	0.00058		1×10^{-5}	

APPENDIX A

DERIVATION OF GASTROINTESTINAL ABSORPTION ADJUSTMENT FACTORS

APPENDIX A

DERIVATION OF GASTROINTESTINAL ABSORPTION ADJUSTMENT FACTOR

The gastrointestinal (GI) absorption adjustment factors (AAF) used throughout this risk assessment were derived after an extensive literature search of the relevant pharmacokinetic data for the indicator chemicals. Brief summaries of the experimental sources of the AAFs are described below. In many cases these AAFs will overestimate actual absorption since most of the data were derived using pure or diluted chemicals, ignoring the potential for matrix attenuation.

Arsenic

A GI AAF of 1.0 was used for arsenic based upon a reported >95% GI absorption in humans (USEPA, 1984).

Benzene

ATSDR (1987k) reports that case studies of accidental or intentional poisoning of humans have indicated that benzene is readily and rapidly absorbed by the oral route. In addition, Parke and Williams (1953a) reported that approximately 90% of the administered dose of ^{14}C-labeled benzene administered orally to rabbits (0.34 to 0.5 g/kg) was accounted for in the total radioactivity eliminated in exhaled air and urine. Based on these findings, a GI AAF of 1.0 was used for Benzene.

Chromium III

Donaldson and Barreras (1966) reported that, in humans, approximately 0.4% of the ^{51}Cr from an oral dose of $^{51}CrCl_3$ was absorbed. Therefore, a GI AAF of 0.01 (1%) was conservatively chosen for Chromium III.

Chromium VI

Donaldson and Barreras (1966) found that ~ 10.6% of the ^{51}Cr from an oral dose of $^{51}CrNa_2CrO_4$ was absorbed in humans. Based on this finding, a GI AAF of 0.15 (15%) was conservatively chosen for Chromium VI.

1,2-Dichlorobenzene

No quantitative data could be found on the gastrointestinal absorption of 1,2-dichlorobenzene in humans or animals. A GI AAF of 1.0 was used as a worst-case estimate.

1,2-Diphenylhydrazine

No quantitative data could be found on the GI absorption of 1,2-diphenylhydrazine in humans or animals. A GI AAF of 1.0 was used as a worst-case estimate.

Lead

GI absorption of lead from the diet in children is reported to be about 50% while the GI absorption factor in adults ranges from 8% (Hammond, 1982) to 15% (Chamberlain et al., 1978). Based on these data, the GI AAF used is 0.50 for children and 0.15 for adults.

Mercury

A GI AAF of 0.0001 was used for mercury based on Doull et al.'s (1980) report of poor GI absorption of probably less than 0.01%.

Carcinogenic PAH

Benzo(a)pyrene (B(a)P) was used to represent carcinogenic PAH. A GI AAF of 0.9 was derived for B(a)P based upon a study by Hecht et al. (1979). Hecht et al. (1979) administered ^{14}C-labeled B(a)P (0.04 µmol, 0.4µmol and 4.0 µmol) dissolved in peanut oil to rats by gavage. The total excretion averaged 74 to 79% from 0 to 48 hours and 85% from 0 to 168 hours.

PCB

A GI AAF of 1.0 was used for PCB based upon a study by Albro and Fishbein (1972) in which greater than 90% of orally administered PCB was retained (fed minus excreted) by rats.

1,1,2,2-Tetrachloroethane

Human studies involving absorption of 1,1,2,2-tetrachloroethane following oral exposure were not available in the literature. Milman et al. (1984) reported that rats and mice absorbed most of the dose of 1,1,2,2-tetrachloroethane following oral exposure but the specifics of the experiment were not available. Mitoma et al. (1985) reported that rats and mice metabolized 70% of an oral dose of 1,1,2,2-tetrachloroethane within 48 hours indicating at least that much absorption. Based on the relatively high absorption in rats and mice and the lack of human absorption data, a GI AAF of 1.0 was used as a worst case estimate.

Zinc

ATSDR (1989) reports that approximately 20-30% of all zinc ingested is absorbed in those with adequate physiological levels of zinc. Based on these data, a GI AAF of 0.30 was used in this risk assessment.

APPENDIX B

CALCULATION OF EXPOSURE AND RISK DUE TO FISH AND CRAB INGESTION

CALCULATION OF EXPOSURE AND RISK FROM INGESTING BLUE CRAB

(Data from Modeled Mean Channel Water Concentrations)

9160H 6020-006-270

UOP Site - [illegible]man Health Risk Assessment
Exposure and Risk from Ingesting Blue Crab
Data from Modelled Mean
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05/04/89

Prepared by: LKW
QA/QC by: RDC/ETH

INTAKE ASSUMPTIONS

CRAB INGESTION INTAKE

	Assumed Value
*Shellfish Consumption Rate (Young Children)(kg shellfish/day)	7.40E-04
*Shellfish Consumption Rate (Older Children)(kg shellfish/day)	1.56E-03
*Shellfish Consumption Rate (Adults)(kg shellfish/day)	3.92E-03
*Body Weight (Younger Children)(kg)	20
*Body Weight (Older Children)(kg)	51
*Body Weight (Adults)(kg)	70
*Exposure period (Younger Children)(yr/yr)	11/70
*Exposure period (Older Children)(yr/yr)	6/70
*Exposure period (Adults)(yr/yr)	53/70
*Local Consumption Factor	0.0027

CHEMICAL	PARAMETER					
	Crab Conc. (mg/kg)	Cooking Factor	GI AAF (Adult)	GI AAF (Child)	Accep. Intake Chronic (AIC)	Carc. Poten. Factor (CPF)
Arsenic	0.00E+00	1	1	1	1.00E-03	1.75E-01
Benzene	0.00E+00	0.75	1	1		2.90E-02
Chromium III	2.70E+00	1	0.01	0.01	1.00E+00	
Chromium VI	1.05E-01	1	0.15	0.15	5.00E-03	
1,2-Dichlorobenzene	0.00E+00	0.75	1	1	4.00E-01	
1,2-Diphenylhydrazine	0.00E+00	0.75	1	1		8.00E-01
Lead	0.00E+00	1	0.15	0.5	1.40E-03	
Mercury	4.53E-02	1	0.0001	0.0001	3.00E-04	
Carcinogenic PAH	0.00E+00	0.75	0.9	0.9		1.15E+01
PCB	1.39E+01	0.75	1	1		7.70E+00
1,1,2,2-Tetrachloroethane	0.00E+00	0.75	1	1		2.00E-01
Zinc	1.14E+03	1	0.3	0.3	2.00E-01	

SHELLFISH INGESTION INTAKE

YOUNGER CHILDREN

CHEMICAL	Crab Concentration (mg/kg)	Crab Consumption Rate (kg crab/day)	Body Weight (kg)	Cooking Factor	GI Absorption Adjustment Factor	Local Consumption Factor	Average Daily Intake (mg/kg/day)	Exposure Period (yr/yr)	LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)
Arsenic	0.00E+00	7.40E-04	20	1	1	0.002739726	0.00E+00	1.57E-01	0.00E+00
Benzene	0.00E+00	7.40E-04	20	0.75	1	0.002739726	0.00E+00	1.57E-01	0.00E+00
Chromium III	2.70E+00	7.40E-04	20	1	0.01	0.002739726	2.74E-09	1.57E-01	4.30E-10
Chromium VI	1.05E-01	7.40E-04	20	1	0.15	0.002739726	1.60E-09	1.57E-01	2.51E-10
1,2-Dichlorobenzene	0.00E+00	7.40E-04	20	0.75	1	0.002739726	0.00E+00	1.57E-01	0.00E+00
1,2-Diphenylhydrazine	0.00E+00	7.40E-04	20	0.75	1	0.002739726	0.00E+00	1.57E-01	0.00E+00
Lead	0.00E+00	7.40E-04	20	1	0.5	0.002739726	0.00E+00	1.57E-01	0.00E+00
Mercury	4.53E-02	7.40E-04	20	1	0.0001	0.002739726	4.59E-13	1.57E-01	7.21E-14
Carcinogenic PAH	0.00E+00	7.40E-04	20	0.75	0.9	0.002739726	0.00E+00	1.57E-01	0.00E+00
PCB	1.39E+01	7.40E-04	20	0.75	1	0.002739726	1.06E-06	1.57E-01	1.66E-07
1,1,2,2-Tetrachloroethane	0.00E+00	7.40E-04	20	0.75	1	0.002739726	0.00E+00	1.57E-01	0.00E+00
Zinc	1.14E+03	7.40E-04	20	1	0.3	0.002739726	3.47E-05	1.57E-01	5.45E-06

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OLDER CHILDREN

CHEMICAL	Crab Concentration (mg/kg)	Crab Consumption Rate (kg crab/day)	Body Weight (kg)	Cooking Factor	GI Absorption Adjustment Factor	Local Consumption Factor	Average Daily Intake (mg/kg/day)	Exposure Period (yr/yr)	LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)
Arsenic	0.00E+00	1.56E-03	51	1	1	0.002739726	0.00E+00	8.57E-02	0.00E+00
Benzene	0.00E+00	1.56E-03	51	0.75	1	0.002739726	0.00E+00	8.57E-02	0.00E+00
Chromium III	2.70E+00	1.56E-03	51	1	0.01	0.002739726	2.27E-09	8.57E-02	1.94E-10
Chromium VI	1.05E-01	1.56E-03	51	1	0.15	0.002739726	1.32E-09	8.57E-02	1.13E-10
1,2-Dichlorobenzene	0.00E+00	1.56E-03	51	0.75	1	0.002739726	0.00E+00	8.57E-02	0.00E+00
1,2-Diphenylhydrazine	0.00E+00	1.56E-03	51	0.75	1	0.002739726	0.00E+00	8.57E-02	0.00E+00
Lead	0.00E+00	1.56E-03	51	1	0.5	0.002739726	0.00E+00	8.57E-02	0.00E+00
Mercury	4.53E-02	1.56E-03	51	1	0.0001	0.002739726	3.80E-13	8.57E-02	3.26E-14
Carcinogenic PAH	0.00E+00	1.56E-03	51	0.75	0.9	0.002739726	0.00E+00	8.57E-02	0.00E+00
PCB	1.39E+01	1.56E-03	51	0.75	1	0.002739726	8.75E-07	8.57E-02	7.50E-08
1,1,2,2-Tetrachloroethane	0.00E+00	1.56E-03	51	0.75	1	0.002739726	0.00E+00	8.57E-02	0.00E+00
Zinc	1.14E+03	1.56E-03	51	1	0.3	0.002739726	2.87E-05	8.57E-02	2.46E-06

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ADULTS

CHEMICAL	Crab Concentration (mg/kg)	Crab Consumption Rate (kg crab/day)	Body Weight (kg)	Cooking Factor	GI Absorption Adjustment Factor	Local Consumption Factor	Average Daily Intake (mg/kg/day)	Exposure Period (yr/yr)	LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)
Arsenic	0.00E+00	3.92E-03	70	1	1	0.002739726	0.00E+00	7.57E-01	0.00E+00
Benzene	0.00E+00	3.92E-03	70	0.75	1	0.002739726	0.00E+00	7.57E-01	0.00E+00
Chromium III	2.70E+00	3.92E-03	70	1	0.01	0.002739726	4.14E-09	7.57E-01	3.13E-09
Chromium VI	1.05E-01	3.92E-03	70	1	0.15	0.002739726	2.42E-09	7.57E-01	1.83E-09
1,2-Dichlorobenzene	0.00E+00	3.92E-03	70	0.75	1	0.002739726	0.00E+00	7.57E-01	0.00E+00
1,2-Diphenylhydrazine	0.00E+00	3.92E-03	70	0.75	1	0.002739726	0.00E+00	7.57E-01	0.00E+00
Lead	0.00E+00	3.92E-03	70	1	0.15	0.002739726	0.00E+00	7.57E-01	0.00E+00
Mercury	4.53E-02	3.92E-03	70	1	0.0001	0.002739726	6.95E-13	7.57E-01	5.26E-13
Carcinogenic PAH	0.00E+00	3.92E-03	70	0.75	0.9	0.002739726	0.00E+00	7.57E-01	0.00E+00
PCB	1.39E+01	3.92E-03	70	0.75	1	0.002739726	1.60E-06	7.57E-01	1.21E-06
1,1,2,2-Tetrachloroethane	0.00E+00	3.92E-03	70	0.75	1	0.002739726	0.00E+00	7.57E-01	0.00E+00
Zinc	1.14E+03	3.92E-03	70	1	0.3	0.002739726	5.24E-05	7.57E-01	3.97E-05

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CALCULATION OF EXPOSURE AND RISK FROM INGESTING FISH

(Data From Modeled Mean Channel Water Concentrations)

9160H 6020-006-270

UOP Site - Human Health Risk Assessment
Exposure and Risk from Ingesting Fish
Data from Modelled Mean
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23-May-89

Prepared by: LKW
QA/QC by: RDC/ETH

INTAKE ASSUMPTIONS	Assumed Value
FISH INGESTION INTAKE	
*Freshwater Finfish Consumption Rate (Young Children)(kg fish/day)	3.29E-04
*Freshwater Finfish Consumption Rate (Older Children)(kg fish/day)	3.01E-04
*Freshwater Finfish Consumption Rate (Adults)(kg fish/day)	9.59E-04
*Body Weight (Younger Children)(kg)	20
*Body Weight (Older Children)(kg)	51
*Body Weight (Adults)(kg)	70
*Exposure period (Younger Children)(yr/yr)	11/70
*Exposure period (Older Children)(yr/yr)	6/70
*Exposure period (Adults)(yr/yr)	53/70
*Local Consumption Factor	0.0027

CHEMICAL	PARAMETER					
	Fish Conc. (mg/kg)	Cooking Factor	GI AAF (Adult)	GI AAF (Child)	Accep. Intake Chronic (AIC)	Carc. Poten. Factor (CPF)
Arsenic	7.22E-01	1	1	1	1.00E-03	1.75E-01
Benzene	1.04E-02	0.75	1	1		2.90E-02
Chromium III	1.72E-02	1	0.01	0.01	1.00E+00	
Chromium VI	6.69E-04	1	0.15	0.15	5.00E-03	
1,2-Dichlorobenzene	3.48E-03	0.75	1	1	4.00E-01	
1,2-Diphenylhydrazine	7.67E-04	0.75	1	1		8.00E-01
Lead	1.28E-01	1	0.15	0.5	1.40E-03	
Mercury	1.75E+00	1	0.0001	0.0001	3.00E-04	
Carcinogenic PAH	3.94E-02	0.75	0.9	0.9		1.15E+01
PCB	1.84E+00	0.75	1	1		7.70E+00
1,1,2,2-Tetrachloroethane	7.70E-02	0.75	1	1		2.00E-01
Zinc	5.57E+01	1	0.3	0.3	2.00E-01	

FRESHWATER FISH INGESTION INTAKE

YOUNGER CHILDREN

CHEMICAL	Fish Concentration (mg/kg)	Fish Consumption Rate (kg fish/day)	Body Weight (kg)	Cooking Factor	GI Absorption Adjustment Factor	Local Consumption Factor	Average Daily Intake (mg/kg/day)	Exposure Period (yr/yr)	LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)
Arsenic	7.22E-01	3.29E-04	20	1	1	0.002739726	3.25E-08	1.57E-01	5.11E-09
Benzene	1.04E-02	3.29E-04	20	0.75	1	0.002739726	3.51E-10	1.57E-01	5.52E-11
Chromium III	1.72E-02	3.29E-04	20	1	0.01	0.002739726	7.75E-12	1.57E-01	1.22E-12
Chromium VI	6.69E-04	3.29E-04	20	1	0.15	0.002739726	4.52E-12	1.57E-01	7.10E-13
1,2-Dichlorobenzene	3.48E-03	3.29E-04	20	0.75	1	0.002739726	1.18E-10	1.57E-01	1.85E-11
1,2-Diphenylhydrazine	7.67E-04	3.29E-04	20	0.75	1	0.002739726	2.59E-11	1.57E-01	4.07E-12
Lead	1.28E-01	3.29E-04	20	1	0.5	0.002739726	2.88E-09	1.57E-01	4.53E-10
Mercury	1.75E+00	3.29E-04	20	1	0.0001	0.002739726	7.88E-12	1.57E-01	1.24E-12
Carcinogenic PAH	3.94E-02	3.29E-04	20	0.75	0.9	0.002739726	1.20E-09	1.57E-01	1.88E-10
PCB	1.84E+00	3.29E-04	20	0.75	1	0.002739726	6.22E-08	1.57E-01	9.77E-09
1,1,2,2-Tetrachloroethane	7.70E-02	3.29E-04	20	0.75	1	0.002739726	2.60E-09	1.57E-01	4.09E-10
Zinc	5.57E+01	3.29E-04	20	1	0.3	0.002739726	7.53E-07	1.57E-01	1.18E-07

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OLDER CHILDREN

CHEMICAL	Fish Concentration (mg/kg)	Fish Consumption Rate (kg fish/day)	Body Weight (kg)	Cooking Factor	GI Absorption Adjustment Factor	Local Consumption Factor	Average Daily Intake (mg/kg/day)	Exposure Period (yr/yr)	LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)
Arsenic	7.22E-01	3.01E-04	51	1	1	0.002739726	1.17E-08	8.57E-02	1.00E-09
Benzene	1.04E-02	3.01E-04	51	0.75	1	0.002739726	1.26E-10	8.57E-02	1.08E-11
Chromium III	1.72E-02	3.01E-04	51	1	0.01	0.002739726	2.78E-12	8.57E-02	2.39E-13
Chromium VI	6.69E-04	3.01E-04	51	1	0.15	0.002739726	1.62E-12	8.57E-02	1.39E-13
1,2-Dichlorobenzene	3.48E-03	3.01E-04	51	0.75	1	0.002739726	4.23E-11	8.57E-02	3.62E-12
1,2-Diphenylhydrazine	7.67E-04	3.01E-04	51	0.75	1	0.002739726	9.31E-12	8.57E-02	7.98E-13
Lead	1.28E-01	3.01E-04	51	1	0.5	0.002739726	1.04E-09	8.57E-02	8.88E-11
Mercury	1.75E+00	3.01E-04	51	1	0.0001	0.002739726	2.83E-12	8.57E-02	2.43E-13
Carcinogenic PAH	3.94E-02	3.01E-04	51	0.75	0.9	0.002739726	4.31E-10	8.57E-02	3.69E-11
PCB	1.84E+00	3.01E-04	51	0.75	1	0.002739726	2.23E-08	8.57E-02	1.92E-09
1,1,2,2-Tetrachloroethane	7.70E-02	3.01E-04	51	0.75	1	0.002739726	9.35E-10	8.57E-02	8.01E-11
Zinc	5.57E+01	3.01E-04	51	1	0.3	0.002739726	2.71E-07	8.57E-02	2.32E-08

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ADULTS

CHEMICAL	Fish Concentration (mg/kg)	Fish Consumption Rate (kg fish/day)	Body Weight (kg)	Cooking Factor	GI Absorption Adjustment Factor	Local Consumption Factor	Average Daily Intake (mg/kg/day)	Exposure Period (yr/yr)	LIFETIME AVERAGE DAILY INTAKE (mg/kg/day)
Arsenic	7.22E-01	9.59E-04	70	1	1	0.002739726	2.71E-08	7.57E-01	2.05E-08
Benzene	1.04E-02	9.59E-04	70	0.75	1	0.002739726	2.93E-10	7.57E-01	2.22E-10
Chromium III	1.72E-02	9.59E-04	70	1	0.01	0.002739726	6.46E-12	7.57E-01	4.89E-12
Chromium VI	6.69E-04	9.59E-04	70	1	0.15	0.002739726	3.77E-12	7.57E-01	2.85E-12
1,2-Dichlorobenzene	3.48E-03	9.59E-04	70	0.75	1	0.002739726	9.80E-11	7.57E-01	7.42E-11
1,2-Diphenylhydrazine	7.67E-04	9.59E-04	70	0.75	1	0.002739726	2.16E-11	7.57E-01	1.63E-11
Lead	1.28E-01	9.59E-04	70	1	0.15	0.002739726	7.21E-10	7.57E-01	5.46E-10
Mercury	1.75E+00	9.59E-04	70	1	0.0001	0.002739726	6.57E-12	7.57E-01	4.97E-12
Carcinogenic PAH	3.94E-02	9.59E-04	70	0.75	0.9	0.002739726	9.98E-10	7.57E-01	7.56E-10
PCB	1.84E+00	9.59E-04	70	0.75	1	0.002739726	5.18E-08	7.57E-01	3.92E-08
1,1,2,2-Tetrachloroethane	7.70E-02	9.59E-04	70	0.75	1	0.002739726	2.17E-09	7.57E-01	1.64E-09
Zinc	5.57E+01	9.59E-04	70	1	0.3	0.002739726	6.27E-07	7.57E-01	4.75E-07

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APPENDIX C

QUALITATIVE ASSESSMENT OF RISKS FROM COMPOUNDS LACKING CRAB BCFs

INTRODUCTION

Human health risks associated with the consumption of crabs from Ackerman's Creek cannot be quantified for chemicals for which no BCFs in crab tissue are reported in the literature. Crab BCFs have not been calculated for: arsenic, lead, benzene, 1,2-dichlorobenzene, 1,2-diphenylhydrazine, carcinogenic PAH, and 1,1,2,2-tetrachloroethane. The potential human health risks from these chemicals, however, can be qualitatively examined.

Because bioconcentration depends on both species-specific and chemical-specific characteristics, BCFs cannot be estimated using a comparative approach between taxonomically unrelated species. Thus, available information on the bioconcentration of various chemicals by fish cannot be used to quantitatively predict bioconcentration of those same chemicals in crabs. The approach described in this appendix qualitatively ranks crab bioconcentration of chemicals on the basis of the chemical-specific characteristics of solubility. This approach is used for organic chemicals, but cannot be applied to metals, because the bioconcentration of metals depends only in part on solubility. When fish tissue BCFs for metals were compared, it was observed that they do not correlate with K_{ow}, nor do they correlate with the crab tissue BCFs. Thus, crab BCFs for arsenic and lead are neither quantitatively nor qualitatively assessed and the effect of these chemicals on humans through consumption of crabs is not estimated.

BCF Relative Ranking

For organic compounds, the degree to which a compound bioaccumulates is inversely related to a compound's water solubility (Oliver and Niimi, 1985). In other words, the more water-soluble a compound, the less likely it is to bioaccumulate. By ranking the compounds according to their octanol-water partition coefficient (K_{ow}), which is related

to the compound's solubility, a relative ranking of bioconcentration can be established for the organic Indicator Compounds (benzene, 1,2-dichlorobenzene, 1,2-diphenylhydrazine, carcinogenic PAH, PCB, and 1,1,2,2-tetrachloroethane). Table C-1 shows the relative ranking of K_{ow} versus BCF for fish tissue and the predicted relative ranking of K_{ow} versus BCF for crab tissue. Trends in the ratio of K_{ow} to BCF for fish tissues indicate that the highly lipophilic compounds (as indicated by high K_{ow}), such as PCBs, bioconcentrate to a much greater degree than do the less lipophilic compounds.

In addition to ranking the compounds according to their predicted BCFs, the relative magnitude of the BCFs in crab tissue is also estimated. This was done by assuming that the BCF for PCB is 1, and that the BCF for other compounds is equal to the K_{ow} of that compound divided by the K_{ow} of PCB. The relative magnitudes of BCFs (Table C-1) should not be used to quantify bioconcentration of the chemicals in blue crabs, but can be used for predicting effects on a relative basis.

Crab Tissue Concentrations and Potential Dose to Humans

Using the relative magnitudes of BCFs for organic Indicator Compounds in crab tissue (see Table C-1), general statements can be made about the anticipated relative crab tissue concentrations. Relative crab tissue concentration is estimated by multiplying the predicted water concentration of the organic Indicator Compound by its respective BCF which is relative to the BCF for PCBs. Because the predicted water concentrations of 1,2-diphenylhydrazine, 1,2-dichlorobenzene, and carcinogenic PAH are all lower than the predicted water concentration of PCBs and the BCFs for these three chemicals are assumed to be lower than the BCF for PCBs (Table C-1), it follows that crab tissue concentrations for these chemicals will be lower than those estimated for PCBs. Even though the predicted water concentrations of benzene and 1,1,2,2-tetrachloroethane are an order of magnitude

TABLE C-1

RELATIVE RANKINGS OF BCFs FOR ORGANIC CHEMICALS

IN FISH AND CRAB BASED ON K_{ow}

Chemical	Ranking	K_{ow}**	BCFs for Fish (from Table 5-3)	BCFs for Crab (from Table 5-2)	Predicted Magnitude of BCFs Relative to K_{ow} for PCB
Benzene	Lowest	131.8	5.21	*	0.0001
1,1,2,2-Tetrachloroethane		245.5	14	*	0.0002
1,2-Diphenylhydrazine		794.3	24.9	*	0.0007
1,2-Dichlorobenzene		3,981.1	89	*	0.004
Carcinogenic PAH		651,213.7	2,630	*	0.59
PCB	Highest	1,096,478.2	3,500	230,000	1.0

Notes:

*BCF for a taxonomically similar species was not found in the available literature.

**Source: U.S. EPA, 1986

higher than that estimated for PCBs, the BCFs are assumed to be two or three orders of magnitude lower than the BCF for PCBs (Table C-1). Thus, the crab tissue concentration of benzene and 1,1,2,2-tetrachloroethane is assumed to be one to two orders of magnitude lower than the concentration of PCBs.

Once crab tissue concentrations of the organic Indicator Compounds relative to PCB are known, the potential dose to someone eating crabs needs to be estimated in order to estimate any potential human health risks. Of the parameters used to convert crab tissue concentration to potential human dose, only one, the gastrointestinal absorption factor (GI AAF), has the potential to affect the estimated relative magnitudes of the potential dose, because it is not constant for all the organic Indicator Compounds. But, because the GI AAF is assumed to be 1 for all organic Indicator Compounds except potentially carcinogenic PAH, for which it is assumed to be 0.9, the effect of GI AAF on the relative ranking of potential dose to people ingesting crabs is minor and the relative magnitude of potential dose is the same as the relative ranking of crab tissue concentration of the organic Indicator Compound.

Ranking of Potential Human Health Risks

Although the potential risks cannot be quantified for the organic Indicator Compounds for which there were no crab tissue BCFs, some general conclusions can be drawn about the risks associated with these chemicals relative to the potential risks posed by PCBs. Because potential lifetime incremental cancer risk is estimated by multiplying predicted dose by cancer potency, an estimate of the relative magnitude of potential risks relative to PCBs can be established by multiplying the estimated relative doses by the cancer potency of each organic Indicator Compound. It must be stressed that this is not a quantitative estimate of potential risk from these organic Indicator Compounds because actual BCF values are not available. This relative ranking of potential risk can be

represented as Relative Risk Ranking = [(water conc. * relative BCF) * GI AAF] * cancer potency slope. Table C-2 shows the relative risk rankings for the potentially carcinogenic organic, Indicator Compounds.

It can be seen from the relative risk rankings in Table C-2 that the highest relative risk ranking is associated with PCBs. The next highest relative risk ranking is for PAH, but it is nearly two orders of magnitude less than that for PCBs. The relative risks for the other compounds are smaller still. Because the actual total risk (in the 10^{-15} range) computed in Section 8 is based solely on PCBs and the relative risk from the other compounds is demonstrated in this section to be orders of magnitude less, the conclusions of the human health food chain assessment would not be affected by computing the added risk from the other organic compounds.

TABLE C-2

POTENTIAL RELATIVE RISK RANKINGS FOR ORGANICS FOR WHICH THERE WERE NO CRAB TISSUE BCFs

(Note: These are not lifetime incidental cancer risk calculations)

Chemical	Predicted Water Conc. (mg/l)	BCF Relative to K_{ow} for BCF	GI AAF	Cancer Potency Slope	Relative Risk Ranking*
Benzene	2.00E-3	0.0001	1	2.90E-2	5.8E-9
1,1,2,2,-Tetrachloroethane	5.50E-3	0.0002	1	2.00E-1	2.2E-7
1,2-Diphenylhydrazine	3.08E-5	0.0007	1	8.00E-1	1.7E-8
1,2-Dichlorobenzene	3.91E-5	0.004	1	**	**
Carcinogenic PAH	1.50E-5	0.59	0.9	1.15E+1	9.2E-5
PCB	5.26E-4	1.0	1	7.70E+0	4.1E-3

*This is not a calculation of lifetime incidental cancer risk. It is a relative ranking of risks.

**This compound is not thought to be carcinogenic.

APPENDIX D
REFERENCES

APPENDIX D
REFERENCES

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