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Chesapeake Bay Program

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Final Report

A Probabilistic Ecological Risk Assessment of Zinc in Surface Waters of the Chesapeake Bay Watershed

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ABSTRACT

The goal of this study was to conduct a screening level probabilistic ecological risk assessment for zinc in the Chesapeake Bay watershed by using the following distinct phases: problem formulation, analysis and risk characterization. This probabilistic ecological risk assessment characterized risk by comparing the probability distributions of environmental exposure concentrations with the probability distributions of species response data determined from laboratory toxicity studies. The overlap of these distributions was a measure of risk to aquatic life. Comparative risk from zinc exposure was determined for various basins in the Chesapeake Bay watershed.

Zinc exposure data were available from 116 stations in 19 basins in the Chesapeake Bay watershed from 1985 through 1996. Highest environmental concentrations of zinc (based on 90th percentiles) were reported in selected locations in the Middle River, Potomac River, Choptank River and Nanticoke River. Sources of zinc responsible for these exposures can not be identified with certainty but human activities associated with urban runoff, industrial/municipal effluents, antifouling paints and non-point source runoff (fertilizers) are likely candidates. As expected, the lowest concentrations of zinc were reported in areas with the least amount of direct human activity such as the lower mainstem Chesapeake Bay, Sassafras River and York River.

The ecological effects data used for this risk assessment were derived from zinc acute laboratory toxicity tests conducted in both fresh and salt water. Freshwater acute toxicity data for zinc were standardized to a hardness of 50 mg/L to allow for accurate rankings of species sensitivity. The 10th percentile (concentration protecting 90% of the species) for all species derived from the freshwater acute zinc toxicity data base was 142 ug/L. Within the acute freshwater zinc data base, a 10th percentile of 212 ug/L was reported for the most sensitive trophic group (benthos) containing

data from at least eight species. For acute saltwater zinc toxicity data, the acute 10th percentile for all species was 79 ug/L. The lowest 10th percentile for the most sensitive trophic group within the saltwater acute zinc data base was 10 ug/L for plants. The acute toxicity benchmarks described above, with at least 8 data points by trophic group, were used to characterize ecological risks for zinc in the 19 basins where exposure data were available.

Highest potential ecological risk from zinc water column exposures based on saltwater acute effects for all species and the most sensitive trophic group (plants) was reported in the Middle River area of the northern Chesapeake Bay watershed. Potential ecological risk from zinc exposure in the Wye River was reported to be low when all species were considered but somewhat higher risk was suggested when using the plant 10th percentile of 10 ug/L. However, based on the documented recovery potential of plant populations to episodic stressors this risk is still judged to be low. Potential ecological risk from zinc water column exposure in the other 17 basins was either low or data were lacking to assess ecological risk.

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SECTION 1
INTRODUCTION

The Chesapeake Bay Basinwide Toxics Reduction Strategy is a critical component of the 1987 Chesapeake Bay Agreement that contains various commitments in areas such as research, monitoring and toxic substance management that are directed to overall chemical reduction in the Chesapeake Bay watershed (Chesapeake Bay Executive Council, 1988). A specific commitment in the Toxic Reduction Strategy is the creation of a Toxics of Concern List (TOC) for the Chesapeake Bay. This TOC list was designed to: (1) prioritize over 1000 chemicals that may be impacting aquatic life or human health in Chesapeake Bay by using a risk based ranking system and (2) direct future research efforts and management.

The first TOC list was completed in 1990 and was recently revised in 1996 (U. S. EPA, 1991; U. S. EPA, 1996a). The proposed revised list is currently under review. The proposed revised TOC list was developed using a chemical ranking system that incorporates sources, fate, exposure and effects of chemicals on Chesapeake Bay living resources and human health (Battelle, 1989). The TOC list contains both a list of primary toxics of concern as well as a secondary list (chemicals of potential concern). For both the 1990 and 1996 TOC lists, zinc was identified as a toxic of potential concern. Zinc is found naturally in the aquatic environment at low concentrations and is an essential micronutrient for all living organisms. This metal enters the aquatic environment from both point and non-point sources. Possible anthropogenic sources of zinc in the Chesapeake Bay watershed include electroplaters, smelting and ore processors, mine drainage, domestic sewage, industrial effluents, combustion of solid wastes, fossils fuels (e. g. coal fired power plants), road surface runoff, corrosion of zinc alloys and galvanized surfaces, antifouling paints, pesticides, fertilizers and erosion of agricultural soils (Eisler, 1997).

Although zinc has been identified as a Toxics of Concern in the Chesapeake Bay watershed, a quantitative probabilistic ecological risk assessment has not been conducted for this metal. The objectives of this study were to (1) quantify the probability and significance of potential ecological effects from zinc water column exposure in the Chesapeake Bay watershed and (2) rank basins in the Chesapeake Bay watershed from high to low probability of ecological risks based on zinc exposures. Procedures described in the following documents were used for this risk assessment: Report of the *Aquatic Risk Assessment and Mitigation Dialogue Group* (SETAC, 1994), the *EPA Framework for Ecological Risk Assessment* (U. S. EPA, 1992), a paper entitled "*An Ecological Risk Assessment of Atrazine in North American Surface Waters*" (Solomon et al., 1996) and a recent report entitled "*A Screening Level Probabilistic Ecological Risk Assessment of Copper and Cadmium in the Chespeake Bay Watershed*" (Hall et al., 1997b).

1.1 Problem Formulation

The following distinct phases are included in this ecological risk assessment: Problem Formulation, Analysis and Risk Characterization (Figure 1). The problem formulation phase identifies major issues to be addressed in the risk assessment and describes how analysis will be conducted. The analysis phase reviews existing zinc data on exposure (environmental monitoring) and ecological effects (laboratory toxicity studies). The risk characterization phase involves estimation of the probability of adverse effects on aquatic populations and communities in potentially impacted areas of the Chesapeake Bay watershed.

The problem formulation phase of this risk assessment identified the following major issues to be addressed: stressor characteristics, exposure data, ecological effects data, risk characterization, endpoints, stressors impacting aquatic communities, and a conceptual model for risk assessment.

1.1.1 Stressor Characteristics

The chemical and physical properties of zinc are described in detail in the Exposure section (Section 2) of this report. In the problem formulation phase of this risk assessment, the solubility, persistence in water and sediment and bioconcentration potential were considered critical.

Zinc is a bluish-white metal that dissolves readily in strong acids and occurs in nature as a sulfide, oxide or carbonate. Zinc and its salts are soluble in water, persistent and may bind to particulates. Zinc mobility in aquatic systems is a function of the following factors: composition of suspended and bed sediments, dissolved and particulate iron and manganese concentrations, pH, salinity, concentrations of complexing ligands and the concentration of zinc (U. S, EPA, 1987). Aquatic biota bioconcentrate zinc in their tissues. Bioconcentration factors (BCFs) as high as 1,130 for freshwater insects and 4,000,000 for saltwater scallops have been reported (Eisler, 1997).

1.1.2 Analysis of Exposure Data

Determining estimation of exposures to aquatic biota is an important part of the risk assessment process for ecosystems. Environmental exposures (ECs) may be determined by using actual measured concentrations from monitoring studies, derivations from highest exposure scenarios and reasonable-high-exposure computer simulations (SETAC, 1994). For deterministic ecological risk assessments, the EC is expressed as a single value but the quantitative likelihood is unknown because a probabilistic approach has not been used to determine the variability of measured environmental concentrations. In recent years, an approach has been endorsed that estimates exposures by taking natural variation into account by using distributions of ECs rather that single point values (SETAC, 1994, Giddings et al., 1997). These probabilistic EC distributions can be used to estimate how frequently concentrations of a contaminant (e. g. zinc) exceed a given toxicity benchmark (threshold) in the environment.

The zinc exposure data used in this risk assessment were obtained from surface water monitoring studies from 19 different data sources in the Chesapeake Bay watershed from 1985 to 1996 (116 stations). Most of the exposure data were collected from Maryland waters of the Chesapeake Bay watershed (Figure 2).

1.1.3 Analysis of Ecological Effects Data

An analysis of toxicity data for risk assessment should cover the range of sensitivity of species to the contaminant being evaluted. This is particularly true for contaminants that have receptor-mediated modes of toxicity. Receptor-mediated modes of toxicity usually result in high toxicity to organisms that possess the receptor system and lower toxicity in non-receptor organisms (e. g. plants - receptor species - are more sensitive to herbicides than non-receptor species such as animals). Toxicity data from sensitive and non-sensitive species should be used in the characterization. However, calculation of 10th percentiles for the most sensitive trophic group is useful for conservative determinations of risk and this conservative determination assumes that protecting the most sensitive species (taxonomic group) will also protect non-sensitive species (Giddings et al., 1997).

A comprehensive review and synthesis of the zinc aquatic toxicity literature was conducted by using literature searches (AQUIRE etc.through 1997) and various review documents such as the U. S. EPA water quality criteria reports (U. S. EPA, 1987) and a recent review of zinc by Eisler (1997). These data were used to determine the distribution of sensitivity of aquatic species to zinc. Limited mesocosm data were also reviewed to address issues of ecological interaction and population recovery.

1.1.4 Risk Characterization

This probabilistic risk assessment characterizes risk by comparing probability distributions of environmental exposure concentrations with the probability distributions of species toxicity data from laboratory studies (SETAC, 1994). The overlap of these distributions is a measure of potential risk to aquatic life in the Chesapeake Bay watershed. The probabilistic approach for characterizing effects and exposure has been suggested as a way to account for the range of species sensitivities to many contaminants (SETAC, 1994). This approach has a number of advantages over a quotient method (comparing the most sensitive species with the highest environmental concentrations) because it allows, if not exact quantification, a least a strong sense for the magnitude and likelihood of potential ecosystem effects of zinc in Chesapeake Bay. An implied assumption of this approach is that protecting a large percentage of species will also preserve ecosystem structure and function. The final result of the risk characterization is expressed as the probability that exposure concentrations of zinc (within a defined spatial and temporal range) will exceed concentrations protective of aquatic life in the Chesapeake Bay watershed.

1.1.5 Endpoints

Endpoints are critical measures used in ecological risk assessment. Two types of endpoints defined in the *Framework for Ecological Risk Assessment* are assessment endpoints and measurement endpoints (U. S. EPA, 1992). Assessment endpoints have recognized societial value and are the actual environmental values that are to be protected (e. g. fish populations). Measurement endpoints are the measured responses to a stressor that can be correlated with or used to protect assessment endpoints (Suter, 1990). With each higher level of testing, measurement endpoints differ while assessment endpoints remain the same.

The assessment endpoints for this risk assessment are the long term viability of aquatic communities in the Chesapeake Bay (fish, benthos etc.). Specifically, the protection of at least 90% of the species 90% of the time (10th percentile from species susceptibility distributions) from acute zinc exposure is the defined assessment endpoint. Measurement endpoints include all acute toxicity data (survival, growth and reproduction) generated from freshwater and saltwater laboratory toxicity studies.

1.1.6 Stressors Potentially Impacting Aquatic Communities

When assessing the potential impact of zinc on aquatic communities in the Chesapeake Bay watershed it is important to remember that both biotic (food quality and quantity) and abiotic factors (water quality, other contaminants, physical habitat alteration) influence the status of biological communities. Zinc is an example of a metal often measured in the Chesapeake Bay environment concurrently with other metals such as cadmium and copper (Hall, 1985; Hall et al., 1986,1987,1989,1991b,1992b). Co-occurance of zinc with cadmium and copper may therefore induce additive or antagonistic toxicity (Eisler, 1997).

In ecological risk assessment, it is important to remember that individuals are part of the food web and somewhat expendable - either consumed or being consumed. Individuals within the various biological communities are more sensitive to contaminant stress than the community as a whole. Therefore, individual losses due to a stressors such as zinc may or may not impact the viability (persistence, abundance, distribution) of the population depending on all the factors influencing the population.

1.1.7 Conceptual Model

Problem formulation is completed with the development of a conceptual model where a preliminary analysis of the ecosystem at risk, stressor characteristics, exposure pathways and ecological effects are used to define the possible exposure and effects scenarios. The goal is to develop a working hypothesis to determine how the stressor might affect exposed ecosystems. The conceptual model is based on information about the ecosystem at risk and the relationship between the measurement and assessment endpoints. Professional judgement is used in the selection of a risk hypothesis. The conceptual model describes the approach that will be used for the analysis phase and the types of data and analytical tools that will be needed. Specific data gaps and areas of uncertainty will be described later in this report.

The hypothesis considered in this risk assessment was:

Zinc may cause permanent reductions at the species and community level for fish, benthos, zooplankton or plants in the Chesapeake Bay watershed and these reductions may adversely impact community structure and function.

SECTION 2
EXPOSURE CHARACTERIZATION

2.1 Introduction

An important component of a probabilistic ecological risk assessment for zinc is the potential exposure of aquatic organisms. Exposure data are used in conjunction with effects data (see next section) to conduct a risk characterization. The exposure analysis for zinc considers use rates, sources, loadings, chemical properties and spatial/temporal scale of measured concentrations (data sources, sampling regimes, analytical methods and data analysis).

2.2 Zinc Loading in the Chesapeake Bay Watershed

Unlike various pesticides, the sources for trace metals such as zinc are often difficult to identify because zinc is found naturally in the aquatic environment and numerous point and non-point sources exist. Anthropogenic activities that contribute to zinc loading in Chesapeake Bay watershed are electroplaters, smelting and ore processors, mine drainage, domestic sewage, industrial effluents, combustion of solid wastes, fossils fuels (e. g. coal fired power plants), road surface runoff, corrosion of zinc alloys and galvanized surfaces, pesticides, fertilizer, erosion of agricultural soils, industrial effluents and atmospheric deposition (Eisler, 1997). Zinc is employed in the following major types of industries that are located in the Chesapeake Bay watershed: paper mills, organic chemical/petroleum, alkalis-chlorine-inorganic chemicals, fertilizers, petroleum refining, basic steel works foundries, basic non-ferrous metal works foundries and steam generating power plants (Dean et al., 1972).

The estimated total basinwide annual loading of zinc and zinc compounds to the Chesapeake Bay was 482,500 pounds based on data collected from 1987 to 1992 (U. S. EPA, 1994). The annual load of zinc from February 1994 through January 1995 for the Susquehanna River, Maryland (the major source of freshwater in the Chesapeake Bay) was 438 metric tons (U. S. EPA, 1996b).

2.3 Chemical Properties of Zinc

Zinc always has an oxidation state of $^{+}2$ in aqueous solution and therefore has a strong tendency to react with acidic, alkaline and inorganic compounds (Merian, 1991). Due to its amphoteric properties, zinc forms a variety of salts. Zinc chlorate, zinc chloride, the sulfates, and the nitrates are readily soluble in water whereas the oxide, carbonate, phosphates, silicates, sulfides and organic complexes have limited solubility in water (Merian, 1991). Zinc is one of the most mobile of the heavy metals. Complexes of zinc with common ligands of surface waters are soluble in neutral and acidic solutions, so that zinc is readily transported in most natural waters.

In natural waters and sediments zinc occurs in many forms. For example, at a pH = 6 in freshwater, the dominant forms of dissolved zinc are the free ion (98%) and zinc sulfate (2%), whereas at pH = 9, the dominant forms are the mon-hydroxide ion (78%) , zinc carbonate (16%) and the free ion (6%) (Turner et al., 1981). In seawater at pH = 8.1, the dominant species of soluble zinc are zinc hydroxide (62%), the free ion (17%), the mono-chloride (6.4%) and zinc carbonate (5.8%) (Zirino and Yamamoto, 1972). The percentage of dissolved zinc present in sea water as the free ion increases to 50% at a pH of 7.0. The major fraction of dissolved zinc is in the form of zinc-organic complexes in the presence of dissolved organic material such as humic acids (Lu and Chen, 1977).

Most of the zinc introduced into aquatic environments is sorbed onto hydrous iron and manganese oxides, clay materials, and organic materials where it is eventually partitioned into sediments (U. S. EPA, 1987). Zinc can be present in sediments is several forms, including precipitated $Zn(OH)_2$, precipitates with ferric and manganic oxyhydroxides, insoluble organic complexes, insoluble sulfides, and residual forms (Patrick et al., 1977). Zinc is mobilized and released in a soluble form as sediments change from a reduced to an oxidized state (Lu and Chen, 1977). Benthic organisms play an important role in partitioning zinc between the water column and sediment (U. S. EPA, 1987). Aquatic biota have a moderate to high potential to bioconcentrate zinc depending on the species. Bioconcentration factors (BCFs) as high as 4,000,000 have been reported in scallops (Eisler, 1997).

The potential for sediment-bound zinc to cause risk to sediment dwelling aquatic biota exists; however, the focus of this risk assessment was an evaluation of risk to aquatic biota from exposures to surface water concentrations. Probabilistic risk assessment techniques for assessing risk of aquatic species to sediment exposures is still developmental and contains a higher degree of uncertainty than water column exposures. By using surface water concentrations in this risk assessment, the results can be more closely related to regulatory issues such as the U. S. Environmental Protection Agency's water quality criteria (U. S. EPA, 1987).

2.4 Measured Concentrations of Zinc in the Chesapeake Bay Watershed

2.4.1 Data Sources and Sampling Regimes

Dissolved zinc exposure data from 19 data sources were available from 1985 to 1996 at 116 stations (19 basins) in freshwater and saltwater tributaries and mainstem areas of the Chesapeake Bay watershed (Figure 2, Tables 1 and 2). The zinc data sources are described below.

Ambient Toxicity Testing Program (Hall et al., 1991a, 1992a, 1994a, 1996, 1997a)

These data were collected over a period of five years (1990 - 1995) on a limited temporal scale (August through October and April 1993) at the following locations: Elizabeth River, Potomac River, Wye River and Patapsco River in 1990; Patapsco River, Potomac River, Wye River in 1991;

Middle River, Nanticoke River and Wye River in 1992-3; Patapsco River (Baltimore Harbor), Magothy River, Sassafras River and Severn River in 1994 and James and York Rivers in 1995.

Fall Line Monitoring Data (MDE, 1993, 1995)

These data were collected at one station each in the Susquehanna and James Rivers monthly from 1990 to 1993.

NOAA Data (Riedel et al., in press)

These data were collected quarterly (May, August, November and February) at 15 stations during 1995 and 1996 in the Patuxent River.

Striped Bass Data (Hall, 1985; Hall et al., 1986, 1987, 1989, 1991b and 1992b)

Zinc was measured from 1985 through 1990 in following tributaries or mainstem areas during April and May as part of an in-situ striped bass contaminant study: Chesapeake and Delaware (C and D) Canal in 1985; Potomac River in 1986; Choptank River and C and D Canal in 1987; Potomac River in 1988; Potomac River and Upper Chesapeake Bay in 1989 and Potomac River and Upper Chesapeake Bay in 1990.

Maryland Coastal Plain Stream Data (Hall et al., 1994b, 1995)

Data were collected at 24 Maryland coastal plains stream stations at five different sampling periods over a two year period (1992-93). Streams from the following basins were sampled: Nanticoke, Choptank, Chester, West Chesapeake, Patuxent and Potomac.

Interstate Commission on the Potomac River Basin (Velinisky et al., 1994)

Zinc data were collected from four sites (one or two samples per site) in the Anacostia River during September of 1992).

District of Columbia Environmental Regulation Commission (Gruessner et al., 1997)

A total of 36 zinc measurements were reported from two sites in the Anacostia River from September 1995 to September 1996.

University of Delaware Data (Culberson and Church, 1988)

Data were collected at 20 stations in mainstem Chesapeake Bay from the mouth of the Bay in Virginia to the northern section in Maryland during August of 1985.

2.4.2 Methods of Zinc Analysis

Zinc data reported during the **Ambient Toxicity Testing Program** were collected from subsurface depth integrated grab samples (a composite of bottom, mid-depth and surface samples). All samples were filtered using a 0.40 um polycarbonate membrane and preserved in ultrex grade nitric acid. Zinc was analyzed using an atomic absorption-furnace (AA-F) method as outlined in U. S. EPA (1979). The limit of detection ranged from 2 to 10 ug/L.

Zinc from the **Fall Line Monitoring Program** was measured from grab samples at the James River and Susquehanna River stations using ultra clean sampling procedures. Dissolved concentrations of zinc were measured using an Inductively coupled plasma mass spectrometer (ICP-MS) method as described in Fishman and Friedman (1989). The detection limit was 0.14 ug/L.

In the **NOAA/COASTES study**, zinc was measured from surface water grab samples using an ultra-clean technique. All samples were filtered using 0.45 um polypropylene capsule filters and preserved using 0.2% ultrex grade hydrochloric acid. Zinc analysis was conducted by using an AA-F method as described in Bruland et al. (1979). The detection limit was <0.3 ug/L.

The zinc data from the **Striped Bass Study** were collected from both subsurface grab samples and composite samples (usually 24 h in duration). All samples were filtered using 0.40 um polycarbonate membranes and preserved using ultrex grade nitric acid. Zinc was analysed using an atomic absorption furnace (AA-F) method as outlined in U. S. EPA (1979). Detection limits for zinc ranged from 3 to 20 ug/L (for five of the six studies detection limits were < 10 ug/L).

For the **Maryland Coastal Plain Stream Data Base,** zinc was measured from grab samples taken seasonally. All samples were filtered using 0.40 um polycarbonate membranes and preserved in ultrex grade nitric acid. Zinc was analysed using an AA-F method (U. S. EPA, 1979). The detection limit was 3 ug/L.

Zinc data from the **Interstate Commission on the Potomac River Basin** were collected from grab samples. All samples were filtered through pre-cleaned and tared 0.4 um Nuclepore filters. Filtered water samples were acidified with double-distilled quartz HCL (0.04% volume/volume) and kept frozen until analysis. Zinc was analysed by an Atomic Absorption Spectrometer with an HGA graphite furnace. The detection limit was 0.14 ug/L.

Zinc measurements from the **District of Columbia Environmental Regulation Administration** were from grab samples. These samples were filtered through 0.4 um Nuclepore membranes. All samples were acidified with Ultrex grade nitric acid and kept refrigerated until analysis. Zinc was analysed by chelation ion chromatography using the method described by Long and Martin (1992). The detection limit was <0.8 ug/L.

Zinc measurements from the **University of Delaware Data Base** were taken from discrete water column depths in the mainstem Chesapeake Bay. All samples were filtered with 0.4 um acid cleaned nuclepore membranes, acidified to pH<2 and frozen until analysis. Zinc was analysed using an AA-F method as described in Danielsson et al.(1978). The detection limit was <0.7 ug/L.

2.4.3 Methods of Data Analysis

Approaches for handling values below the detection limits include assigning these values as zero, one-half the detection limit or the detection limit (MacBean and Rovers, 1984; Giddings et al. 1997). For this risk assessment, zinc values below the detection limit were assumed to be log-normally distributed. The distribution of exposure data was calculated based on the measured values and the concentrations of the non-detects were assumed to be distributed along a lower extension of this distribution. For example, if 80 out of 100 samples were reported as non-detects, the 20 measured values were assigned ranks from 81 to 100 and the frequency distribution was calculated from these 20 values. In some cases in these data sets, actual concentrations were reported even though they were below the detection limits. When this occurred, the concentrations were used in the analysis. For cases where more than one value was available at the same time and station, the highest value was used in the frequency distribution.

For data sets arranged by basin or station with four or more values above the detection limit, log-normal distributions of exposure concentration were determined as follows. The observations in each data set were ranked by concentration and for each observation the percentile ranking was calculated as n/(N+1) where n is the rank sum of the observation and N is the total number of observations including the non-detects. Percentile rankings were converted to probabilities and a linear regression was performed using the logarithm of concentration as the independent variable and normalized rank percentile as the dependent variable. Although non-detects observations were not included in the regression analysis, they were included in the calculation of the observation ranks. The 90th percentile concentrations (exceedence of a given value only 10% of the time) were calculated for sampling stations (or basins) based on the calculated log-normal concentration distributions.

2.5 Measured Concentrations by Basin

The 90th percentile values for zinc in 19 basins presented in Table 2 showed that values ranged from a high of 140 ug/L in the Middle River to 5.2 ug/L for the lower Bay (Table 2, Figure 3). Due to concentrations below the detection limit, 90th percentile values could not be calculated for Baltimore Harbor, Magothy River, Sassafras River, Severn River and the York River . The high 90th percentile value in the Middle River was likely related to anthropogenic activities near marina areas and/or urban runoff as copper and cadmium concentrations above background have also been reported in this basin (Hall et al., 1997b). The second highest zinc 90th percentile of 70 ug/L was reported in the Potomac River was likely related to the proximity of these sampling stations near point source discharges from facilities such as Quantico Marine Base, the Possum Point Power Plant or the Indian Head Military Facility. Elevated cadmium concentrations were also reported at these stations (Hall et al., 1997b). The lower 90th percentiles in Lower Chesapeake Bay, Magothy River, Sassafras River, Severn River and York River were likely related to less anthropogenic activity.

2.6 Temporal Trends

The NOAA data from the Patuxent River (quarterly sampling in 1995 and 1996) and the Fall Line Monitoring Data from the James and Susquehanna River (monthly sampling in 1990 to 1993) were used to examine temporal trends in zinc over single or multiple years (Riedel et al. in press, MDE, 1993,1995). These were the only data sets that were appropriate for temporal analysis.

2.6.1 Patuxent River

The quarterly mean zinc concentrations (May, August, November - 1995 and February - 1996) from the 15 pooled stations in the Patuxent River showed that concentrations were elevated during February (Figure 4). The mean zinc concentration for the February time period (2.57 ug/L) was approximately twice as high the other three sampling periods (0.99 to 1.34 ug/L). The highest zinc concentrations (10.8 ug/L) in this data set was also reported in February. The elevated concentrations of zinc during the winter were likely related to the increased flow during this time period.

2.6.2 James and Susquehanna Rivers

Monthly measurements of zinc in the James River over 4 years (1990 to 1993) ranged from below the detection limit to 30 ug/L (Figure 5). The highest value of 30 ug/L occurred in October of 1990. Other peak values of 15 and 13 ug/L were reported May of 1992 and December of 1992, respectively. There appears to be no consistent temporal trends of zinc concentrations in the James River.

Monthly measurements of zinc in the Susquehanna River during 1990 through 1993 ranged from below the detection limit to 22 ug/L (Figure 6). The four peak concentrations of zinc (~ 20 ug/L) were reported in June of 1990, January of 1991, April of 1992 and September of 1993. There is no apparent temporal trend with zinc exposure data from the Susquehanna River.

2.7 Summary of Exposure Data

Highest environmental concentrations of zinc (based on 90th percentiles) in the Chesapeake Bay watershed were reported in the Middle River, Potomac River, Choptank River and Nanticoke River. Sources of zinc responsible for these exposures can not be identified with certainty but human activities associated with urban runoff and marina facilities (Middle River), industrial effluents (Potomac River), fertilizers (Choptank and Nanticoke Rivers) and a power plant (Nanticoke River) are likely candidates. Natural sources of zinc in some of these areas may also be a source. As expected the lowest concentrations of zinc were generally reported in areas with the least amount of direct human activity such lower mainstem Chesapeake Bay, Sassafras River and York River. It is noteworthy that Baltimore Harbor, a highly industralized area, had low concentrations of zinc in the water column. However, sediment concentrations of zinc for the various stations sampled for water column measurement were relatively high and in some cases exceeded the Long et al. (1995) Effects Range Median values (Hall et al., 1996). Quarterly measurements of zinc in the Patuxent River showed somewhat elevated concentrations of zinc during the winter months that were related to increased flow. There were no apparent temporal trends in zinc concentrations from monthly measurements (1990-1993) in the James and Susquehanna Rivers.

SECTION 3
ECOLOGICAL EFFECTS

3.1 Mode of Toxicity

Zinc is an essential micronutrient for all living organisms and is ubiquitous in the tissues of plants and animals (U. S. EPA, 1987). Zinc is particularly critical for normal growth and reproduction in aquatic biota. Numerous different enzymes require zinc for maximum catalytic activity, including carbonic anhydrase, alkaline phosphate, alcohol dehydrogenase, acid phosphatase, lactic dehydrogenase, carboxypeptidase and superoxide dismutase (Eisler, 1997). Zinc is critical in controlling zinc-dependent enzymes that regulate the biosynthesis and catabolic rate of RNA and DNA (Prasad, 1979). Zinc deficiency effects have been reported in aquatic organisms at concentrations between 0.65 and 6.5 ug/L (Eisler, 1997).

Bioavailability of zinc is important to consider when assessing the toxicity to aquatic biota. Zinc toxicity to aquatic biota is influenced by the chemical and physical forms of zinc, the toxicity of each form, and the degree of interconversion for each form. In most cases aquatic fish and plants are unaffected by suspended zinc; however, many invertebrates and some fish may be impacted if zinc-containing particulates are ingested (U. S. EPA, 1987). Zinc adversely impacts fish by causing mortality, growth retardation, tissue alteration (destroys gill epithelium), respiratory and cardiac changes and inhibition of spawning (Sorenson, 1991). Inhibition of photosynthesis and disruption of plant growth resulting from impairment of enzyme systems are suspected to be the major adverse effects from excessive zinc exposure in plants.

3.2 Methods of Toxicity Data Analysis

Hardness (concentrations of calcium and magnesium) is one water quality variable that significantly influences the toxicity of zinc in freshwater. As hardness increases, the toxicity of zinc to biota generally decreases due to reduced bioavailability of the metal or alteration of the osmoregulatory capacity of the organism. The U. S. Environmental Protection Agency addresses the influence of hardness on zinc toxicity in their development of freshwater water quality criteria (U. S. EPA, 1987). For the zinc toxicity data used in this risk assessment, hardness was also considered in the ranking of sensitivities of various freshwater species. In order to realistically compare freshwater toxicity data among species, all data were standardized to a hardness of 50 mg/L $CaCO_3$. Fifty mg/L was selected because it is the mean hardness value of 24 coastal plains streams sampled five times over a two year period in 1992-93 (Hall et al., 1994b; 1995). The following equation was used to hardness adjust the freshwater acute and chronic toxicity data:

$\ln LC_{50\ standardized} = \ln LC_{50\ observed} - (b[1] \ln hardness_{observed} - \ln_{hardnessstandardized})$

$_{hardnessstandardized} = 50$ mg/L as $CaCO_3$

Slope = b[1] = 0.8473 for zinc acute and chronic toxicity data (U. S. EPA, 1987)

It is also important to note that other water quality parameters such as pH and dissolved organic carbon also influence zinc toxicity. These and other parameters have been the basis for the Biotic Ligand model the U. S Enviromental Protection Agency is considering for use in revising water quality criteria for metals (Andrew Green, personal communication, International Lead and Zinc Research Organization).

The 10th percentile of species sensitivity (protection of 90% of the species) from acute exposures was the primary benchmark used for this risk assessment. The implied assumption when using this benchmark is that protecting a large percentage of the species assemblage will preserve ecosystem structure and function. This level of species protection is not universally accepted, especially if the unprotected 10% are keystone species and have commercial or recreational significance. However, protection of 90% of the species 90% of the time (10th percentile) has been recommended by the Society of Environmental Toxicology and Chemistry (SETAC, 1994) and others (Solomon et al., 1996). Recent mesocosm studies have reported that this level of protection is conservative (Solomon et al., 1996; Giddings, 1992).

Zinc toxicity data were analyzed as a distribution on the assumption that the data represented all species in the Chesapeake Bay ecosystem. An approximation was made since it is not possible to test all species in the Chesapeake Bay. This approximation assumes that the number of species tested (N) is one less than the number in the Chesapeake Bay. To obtain graphical distributions for smaller data sets that are symmetrical (normal distributions) percentages were calculated from the formula (100 x n/(N + 1)) where n is the rank number of the datum point and N is the total number of data points in the set (Parkhurst et al., 1994). This formula compensates for the size of the data sets as small (uncertain) data sets will give a flatter distribution with more chance of overlap than larger (more certain) data sets. In cases where there were multiple data points for a given species, the lowest value was used in the regression analysis of the distribution. When data were available for multiple life stages of a species, the lowest values were generally reported for early life stage. Using the lowest value therefore provides a conservative approach for protecting the most sensitive life stage of a species. Data were plotted using Sigma Plot (Jandel Corporation, 1992).

3.3 Effects of Zinc from Laboratory Toxicity Tests

Acute and chronic zinc toxicity data used in this risk assessment were obtained from the AQUIRE database through 1998, U. S. EPA water quality criteria document (U. S. EPA, 1987), a recent review of zinc (Eisler, 1997) and manual searches of grey literature from academia, industry and government sources. Zinc acute and chronic toxicity data by water type (freshwater and saltwater) are discussed below.

3.3.1 Acute Toxicity of Zinc

Acute freshwater zinc toxicity data were available for 101 species, primarily fish and benthos, as shown in Table 3 and Figure 7. Hardness data were available for approximately half of these species (n=55) and these data were used for the analysis of species sensitivity distribution and calculation of 10th percentiles. The range of acute toxicity values was 32 ug/L for *Ceriodaphnia* to 260,000 ug/L for the climbing perch (Table 3). The acute 10th percentile for all freshwater species was 142 ug/L (Table 4). This value is approximately twice as high as the U. S. EPA freshwater water quality criteria (5th percentile) of 65 ug/L at 50 mg/L hardness (U. S. EPA, 1987). The order of sensitivity from most to least sensitive trophic group using 10th percentiles was as follows: zooplankton (4.3 ug/L), benthos (212 ug/L), fish (216 ug/L), amphibians (629 ug/L) and plants (789 ug/L). Data for amphibians (n=2) zooplankton (n=5), and plants (n=2) were limited and were therefore not used for assessing risk to the most sensitive trophic. The 10th percentiles for fish (216 ug/L) and benthos were similar (212 ug/L).

Acute zinc saltwater toxicity data were available for 82 species as shown in Table 5 and Figure 8. As reported above for freshwater acute data, most of the saltwater acute toxicity data were with fish and benthos. Zinc toxicity ranged from 19 ug/L for a diatom to 119,300 ug/L for a fish species (Table 5). The acute 10th percentile for all species was 79 ug/L (Table 4). This value is similar to the U. S. EPA water quality criteria (5th percentile) of 95 ug/L (U. S. EPA, 1987). The order of sensitivity from most to least sensitive trophic group using 10th percentiles was plants (10 ug/L), zooplankton (46 ug/L), fish (69 ug/L) and benthos (102 ug/L). The plant 10th percentile of 10 ug/L was substantially lower than the 10th percentiles for other trophic groups.

3.3.2 Chronic Toxicity of Zinc

Chronic zinc toxicity data were available for 14 freshwater species (Table 6 and Figure 9). Hardness was reported for 12 of the 14 species tested (data used for calculation of 10th percentiles). Chronic values ranged from 25 ug/L for a cladoceran to > 5,243 ug/L for a caddisfly for non-hardness adjusted data. A hardness adjusted value of 5.5 ug/L was reported for a cladoceran. The chronic 10th percentile for all freshwater species was 11 ug/L. This value is substantially lower than the U. S. EPA chronic freshwater criteria (5th percentile) of 59 ug/L at a hardness of 50 mg/L (U. S. EPA, 1987). The order of sensitivity from most to least sensitive trophic groups based on 10th percentiles was zooplankton (0.8 ug/L), fish (56 ug/L) and benthos (74 ug/L).

Saltwater chronic toxicity data were limited to six species and actual chronic values were only reported for the Pacific oyster (30 ug/L) and two mysid species (152 ug/L) (Table 7 and Figure 10). The 10th percentile for the saltwater chronic toxicity data was 8.7 ug/L (Table 4). This 10th percentile is much lower that the U. S. EPA saltwater chronic criteria (5th percentile) of 86 ug/L (U. S. EPA, 1987). However, value is similar to the freshwater chronic value of 11 ug/L reported above.

3.4 Mesocosm/Microcosm Studies

Zinc mesocosm studies with reported MATC (maximum acceptable toxicant concentrations), LOEC (lowest observed effect concentrations) or NOEC (no observed effect concentrations) values were very limited. Genter et al. (1987) exposed algal communities to zinc concentrations of 0 (control), 50, 500 and 1,000 ug/L for 30 days in outdoor flow-through stream mesocosms. Treatments as low as 50 ug/L were reported to significantly change community composition from diatoms to green or blue green-algae. However, this species shift does not necessarily imply that the functions of aquatic communities have been impaired. In another study, Genter et al. (1989) exposed algal and protozoa communities to zinc concentration ranging from 0 to 10,000 ug/L for 7 days. Results from this study showed that none of the algal or protozoan species had reduced biovolume density in high zinc concentrations (10,000 ug/L) even though the total number of protozoan species decreased.

3.5 Summary of Effects Data

The 10th percentile for all species derived from the freshwater acute zinc toxicity data base was 142 ug/L. Most of the data used for the calculation of this 10th percentile were from toxicity studies with fish and benthos. Cladocerns were reported to be the most sensitive freshwater species to zinc exposure (LC50s of 35 and 21 (hardness adjusted) ug/L). A ranking of sensitivity among trophic groups from most to least sensitive showed the following order: zooplankton, benthos, fish, amphibians and plants. The 10th percentiles for zooplankton, amphibians and plants were not used for risk characterization for the most sensitive trophic group because these data were very limited. The freshwater chronic 10th percentile for all species was 11 ug/L. As reported above for acute freshwater data, zooplankton were the most sensitive trophic group subjected to chronic zinc exposures.

The saltwater acute zinc 10th percentile for all species was 79 ug/L. Diatoms (phytoplankton) were reported to be the most sensitive species (EC50s of 19 to 26 ug/L). The ranking of sensitivity among trophic groups from most to least sensitive was as follows based on acute 10th percentiles: plants, zooplankton, fish and benthos. The acute saltwater 10th percentile for plants (10 ug/L) was lower than for other trophic groups. The chronic 10th percentile for all saltwater species (8.7 ug/L) was based on a limited set of toxicity data from six benthic species.

SECTION 4
RISK CHARACTERIZATION

4.1 Characterizating Risks

Risk quotients are one simple and commonly used method for characterizing risks to aquatic biota. Risk quotients are simple ratios of exposure and effects concentrations where the susceptibility of the most sensitive species is compared with the median, mean or highest environmental exposure concentration. Safety factors such as the division of the effect concentration by a number ranging from one to 100 are often applied to allow for unquantified uncertainty in effect and exposure concentrations. If the exposure concentration equals or exceeds the effects concentration in the risk quotient approach then an ecological risk is suspected. The quotient method is a valuable first tier assessment that allows a determination of a worst case effects and exposure scenario for a particular contaminant. However, some of the major limitations of the quotient method for ecological risk assessment are that it fails to consider variability of exposures among individuals in a population, ranges of sensitivity among species in the aquatic ecosystem and the ecological function of these individual species. The probabilistic approach addresses these various concerns as it expresses the results of an exposure or effects characterization as a distribution of values rather than a single point estimate. Quantitative expressions of risks to aquatic communities are therefore determined by using all relevant single species toxicity data in conjunction with exposure distributions. A detailed presentation of the principles used in a probabilistic ecological risk assessment are presented by Solomon et al. (1996).

The following sections will summarize the results of the risk characterization phase of this probabilistic ecological risk assessment of zinc in the Chesapeake Bay watershed. The toxicity benchmark used for the risk characterization will be either the freshwater or saltwater acute 10th percentile, depending on whether freshwater or saltwater is present within the basin. The acute 10th percentile was selected for the following reasons: (1) based on laboratory experimental data, dissolved and bioavailable zinc are only in the water column of the aquatic environment for short periods of time (due to complexation with natural organic particulates) which are more closely related to acute exposures that chronic exposures; (2) the low acute to chronic ratio (~2) reported for zinc by the U. S. EPA (1987) suggests that exposure duration does not significantly increase toxicity and (3) toxicity data are much more numerous and represent a wider range of trophic groups for acute studies than chronic studies. In addition to using the acute 10th percentile for all species in freshwater or saltwater, the trophic group with the lowest acute 10th percentile with at least 8 data points (8 species) was also used as an additional benchmark (more conservative approach) to assess possible ecological risk. The U.S. Environmental Protection Agency uses a minimum value of 8 species for development of acute numeric water quality criteria (Stephan et al., 1985).

4.2 Risk Characterization of Zinc in the Chesapeake Bay Watershed

Potential ecological risk from zinc exposure was characterized by using freshwater acute effects data for freshwater areas and saltwater effects data for saltwater areas (Table 8, Appendix A). There were five saltwater and nine freshwater basins where data were sufficient for characterizing risk. The highest potential ecological risk area for zinc exposures in the Chesapeake Bay watershed was reported in the Middle River (Table 8). The percent probability of exceeding the acute saltwater 10th percentile for all species was 21%. For the most sensitive trophic group (based on acute saltwater exposures), the probability of exceeding the 10th percentile for plants was even higher

(79%). The Middle River was the only basin where the potential ecological risk from zinc exposure was judged to be significant. The second highest risk area for zinc exposures in the Bay watershed was the Wye River (Table 8). The probability of exceeding the 10th percentile for all species and the probability of exceeding the 10th percentile of the most sensitive trophic group with at least eight species (based on acute saltwater exposures) was 2.7 and 42%, respectively. The 2.7% exceedence for all species in the Wye River basin is relatively low risk. The 42% exceedence for plant species does suggest a somewhat higher potential risk to this trophic group although 90% of the plants species would not be at risk based on the probabilistic risk analysis used. The third highest risk area for zinc exposures was the Potomac River. The percent probability of exceeding the 10th percentile for all species and most sensitive trophic group with at least eight species (benthos = 212 ug/L) was 3.3 and 1.5 %, respectively. The percent exceedence for both of these benchmarks or overall ecological risk in the Potomac River is low. For all other 11 basins in Table 8, ecological risk from zinc exposure was generally low using either the acute saltwater or freshwater 10th percentiles.

4.3 Uncertainty in Ecological Risk Assessment

All scientific endeavors have uncertainty and ecological risk assessment is no exception. Development of exposure benchmarks, such as the 90th percentile for environmental concentrations, or toxicity benchmarks, such as the 10th percentile for species susceptibility, may seem to be exact. However, these values involve uncertainty when extrapolating risks from laboratory data to aquatic ecosystems. Uncertainty plays a particularly important role in ecological risk assessment as it impacts problem formulation, analysis of exposure and effects data and risk characterization. Evaluation of uncertainty in this risk assessment was critical in determining data gaps (research needs) as described in the final section of the report. Addressing these various research needs in future efforts will reduce uncertainty.

Uncertainty associated with metals risk assessment such as zinc have some fundamental differences when compared to pesticides (European Commission, 1996). The following differences exist:

(1) Unlike most organics, metals such as zinc (micronutrients) and some organometallic compounds (e. g. methylmercury) are a class of chemicals that occur naturally in the environment. Therefore, natural background concentrations and exposure to these concentrations should be factored in risk characterization.

(2) The availability of zinc for uptake by organisms under field conditions is limited, will vary from site to site and is highly dependent on the speciation of the metal. Exposure and effects data should therefore be based on similar levels of availability (in this case disssolved concentrations).

(3) The same toxic form of zinc can originate from a variety of different substances (e. g. Zn^{+2} from $ZnSO_4$, $ZnCl_2$ etc.). Therefore, it is necessary to take into account all metal species that are emitted to the environment which may result in concentrations of the toxic form.

Uncertainty in ecological risk assessment has three basic sources: (1) lack of knowledge in areas that should be known; (2) systematic errors resulting from human or analytical error and (3) non-systematic errors resulting from the random nature of the ecosystem (e.g. Chesapeake Bay watershed). The following sections will address specific uncertainty from the above three sources as associated with exposure data, effects data and risk characterization.

4.3.1 Uncertainty Associated with Exposure Characterization

Zinc exposure data used for this risk assessment were obtained from 19 different data sources from 1985 to 1996 as described in Section 2. The spatial scale of these data (116 stations in 19 basins/mainstem areas) was somewhat limited considering that there are at least 50 major rivers and numerous smaller tributaries that discharge into the Chesapeake Bay. Exposure data from basins in Virginia waters of Chesapeake Bay were particularly limited as only the James River, York River and the lower mainstem Bay were represented. The temporal scale (sampling frequency) of the available data for the Bay watershed was even more limited. In many cases there were only a few measurements made for these metals at various stations. Rain event sampling for these metals in tributaries and streams was generally not considered in the sampling designs of the various monitoring studies. Although rain event sampling is more relevant for pesticides that are applied on agricultural crops and enter aquatic systems during runoff, such events may be important for zinc loading resulting from fertilizer (chicken manure based fertilizer) used on crops or zinc loading from urban stormwater discharges or municipal/ industrial overflow. Roman-Mas et al. (1994) have recommended a sampling interval of 5% of the duration of the storm flow as adequate to characterize pesticide concentration distributions in runoff with an error of less than 5% (for example during an event with storm flow lasting 100 h sampling should be every 5 h). The sampling frequency of the present exposure data for zinc is clearly inadequate for rain event sampling.

The zinc analysis associated with the various laboratories introduces uncertainty because analytical procedures differed among the laboratories (see Section 2). For example, samples were collected for analysis using either grab, depth integrated or composite techniques. In all cases samples were filtered with either 0.4 or 0.45 um membranes but the membranes were made of different material (polycarbonate, polypropylene or nucleopore). The method of metal analysis was somewhat consistent among laboratories as an Atomic Absorption - Furnace method (AA-F or HGA) was used for all data sets except for the Fall Line Data (Inductively Coupled Plasma Mass

Spectrometer - ICP-MS) and the District of Columbia Environmental Regulation Administration data (chelation ion chromatography). The detection limits varied among the different laboratories (generally 0.14 to 10 ug/L; for one study 20 ug/L was used).

4.3.2 Uncertainty Associated with Ecological Effects Data

There is uncertainty when extrapolating laboratory toxicity data to responses of natural taxa found in the Chesapeake Bay watershed due to the relatively small number of species that can be cultured and tested in laboratory toxicity studies. In the case of zinc in the Chesapeake Bay watershed, freshwater and saltwater acute toxicity were available for 55 (hardness adjusted data) and 82 species, respectively, for use in the calculation of the 10th percentile. Although these data seem adequate for all species, the distribution among the various trophic groups was weighted more with fish and benthos. Acute zinc toxicity data were particularly limited for plants (phytoplankton and macrophytes), zooplankton and amphibians in freshwater. Chronic data, although not used in this risk assessment, were limited for both types of water but particularly for saltwater species (n = 6). In addition to more data with an expanded list of species, more ecologically relevant zinc toxicity data are needed to reduce uncertainty and address comparisons of laboratory and field data. Metal speciation, dissolved organic carbon, suspended particulates and bedded sediments should be considered with laboratory to field extrapolations.

Variability in the results of toxicity tests for a given species tested in different experiments or by different authors is a potential source of random and systematic errors. In this assessment, the most conservative (lowest) effect value was used when multiple data points were available for a given species. The range of toxicity data among trophic groups differed for each water type. For example, the acute zinc freshwater 10th percentile values among trophic groups ranged from 4.3 ug/L (zooplantkton) to 789 ug/L (plants) - a factor of 183x. Acute saltwater 10th percentiles by trophic group ranged from 10 ug/L (plants) to 102 ug/L (benthos). This 10 fold difference for saltwater species is much less than the factor of 183x reported above for freshwater 10th percentiles. Using the distribution of species susceptibility accounts for this range of data points. Distributions will be flatter, with greater chance of overlap with exposure distributions, when the range is large.

Acute freshwater and saltwater zinc toxicity data were primarily used in the risk characterization as previously discussed. The use of acute data for predicting ecosystem effects is often questioned and assumed to be an area of significant uncertainty. However, Slooff et al. (1986) in their review of single species and ecosystem toxicity for various chemical compounds, have reported that there is no solid evidence that predictions of ecosystem level effects from acute tests are unreliable. The result of Slooff et al. (1986) coupled with the use of a distribution of acute toxicity data reduces some of the uncertainty associated with using acute data.

Although single species laboratory toxicity tests are valuable in risk assessment, microcosm and mesocosm data provide the following useful information for assessing the impact of a stressor on aquatic communities in an ecosystem: (1) aggregate responses of multiple species; (2) observation of population and community recovery after exposure and (3) indirect effects resulting from changes in food supply. Unfortunately, microcosm and mesocosm studies that determined No Observed Effect Concentrations (NOEC) were limited. The lack of these type data, where the interaction of biotic communities have been assessed under zinc exposure, was a source of uncertainty in this risk assessment since microcosm/mesocosm toxicity benchmarks were not available for risk characterization.

4.3.3 Uncertainty Associated with Risk Characterization

The Society of Environmental Toxicology and Chemistry (SETAC, 1994) reported that many of the uncertainties associated with the variability in the exposure and effects characterizations discussed above are incorporated in the probabilistic approach used in this risk assessment. A distribution of exposure and effects data are used for quantitative analysis of risks.

Ecological uncertainty includes the effects of confounding stressors such as other contaminants (e. g. zinc often occurs concurrently with other metals such as copper and cadmium) and the ecological redundancy of the functions of affected species. In the Chesapeake Bay watershed, numerous contaminants may be present simultaneously in the same aquatic habitats; therefore, "joint toxicity"may occur. For zinc, additive toxicity is likely if other metals such as copper and cadmium are present. The concurrent presence of various contaminants along with zinc makes it difficult to determine the risk of zinc in isolation.

Ecological redundancy is known to occur in aquatic systems. Field studies have shown that resistant taxa tend to replace more sensitive species under stressful environmental conditions (Solomon et al., 1996; Giddings, 1992) The resistant species may replace the sensitive species if it is functionally equivalent in the aquatic ecosystem and the impact on overall ecosystem function is reduced by these species shifts. For this risk assessment, information on the ecological interactions among species would help to reduce this area of uncertainty.

SECTION 5
CONCLUSIONS AND RESEARCH NEEDS

Potential ecological risk from zinc water column exposure was higher in the Middle River than any of the other basins in the Chesapeake Bay watershed. Ecological risk from zinc exposure in the Wye River was insignificant when all acute species data were used but the most sensitive trophic group (plants) suggested that some risk may occur with the most sensitive 10% of plant species. Based on the documented recovery of plant populations to episodic stressors, however, the zinc exposure to plant populations is still judged to be low in the Wye River since 90% of the plant species would not be affected. Ecological risk from zinc water column exposure was judged to be low or data were lacking for assessing risks in the other 17 basins.

The following research is recommended to supplement existing data for assessing the ecological risks of zinc in the Chesapeake Bay watershed:

(1) A probabilistic ecological risk assessment for zinc exposure in sediment is recommended to complement this water column risk assessment. Most of the zinc introduced into the aquatic environment is sorbed onto hydrous iron, manganese oxides, clay materials and organic materials where it is eventually partitioned into sediments. Therefore, assessing risk of sediment dwelling organisms exposed to zinc would expand our knowledge on the potential ecological risk of this metal in the environment.

(2) Exposure assessments for zinc using randomly selected stations are needed on a broad spatial and temporal scale in the Chesapeake Bay watershed. On a spatial scale, zinc data are needed for the

major rivers (tributaries) and representative freshwater streams where these data are lacking, particularly in Virginia waters of the Chesapeake Bay watershed (e.g. Rappahannock River and lower eastern shore). Exposure assessments with increased sampling frequency covering all seasons of the year at representative locations in the Bay watershed (including some of the basins in this report where data are lacking) are also needed to improve our ability to determine risk of aquatic biota to zinc. Specifically, rain event sampling (e.g. samples every 2 to 4 h during the duration of the event) and subsequent measurement of metals in streams or tributaries near known sources of zinc are needed (agricultural fields using chicken manure based fertilizer). These data may provide insight on why zinc concentration were higher than ambient concentrations in agricultural areas such as the Choptank, Nanticoke and Wye Rivers. All exposure assessments of zinc should be conducted by laboratories using the most updated analytical methods (with documented and approved Quality Assurance/Quality Control procedures) with detection limits below the toxicity thresholds for the most sensitive species.

(3) An extensive spatial and temporal exposure assessment of zinc (including rain event sampling) is recommended in the Middle River area over multiple years. Since the Middle River was the highest risk area for zinc based on limited data collected in 1993, the obvious question is whether this area still has concentrations that may pose a risk to aquatic biota. Biological communities should also be sampled in the Middle River area to see if they are impaired when compared to communities in similar habitats.

(4) Acute zinc toxicity data for various trophic groups in freshwater and saltwater are needed for

improving the present toxicity data base. Specifically, acute freshwater and saltwater toxicity data for zinc (with measured concentrations) are needed with plants such as phytoplankton and aquatic macrophytes. Acute freshwater data are also needed for amphibians and zooplankton. Chronic data for all trophic groups would also be useful.

(5) Microcosm/mesocosm toxicity data that include the calculation of NOEC, LOEC and chronic values for zinc in freshwater and saltwater environments are needed to provide insight on the interaction of aggregate species assemblages during zinc exposure, recovery potential of exposed species and possible indirect effects on higher trophic groups. These studies should be designed to simulate environmentally realistic pulsed exposures of these zinc concentrations documented to occur in the environment.

(6) Assessments of biological communities (Index of Biotic Integrity for fish, invertebrates etc.) in aquatic systems that receive the highest exposures of zinc are recommended to determine if the predicted ecological risk (impaired biological communities) from this metal in the water column can be confirmed with actual field data.

(7) Investigations are needed to determine how to incorporate the essentiality of relevant metals (such as zinc) for aquatic organisms into the risk assessment process.

SECTION 6
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TABLES

Table 1. Summary of the 19 zinc exposure data sources used for this risk assessment.

Reference	Data ID	Total # samples	Sample Period	Detection Limit (μg/L Zn)
Hall et al., 1991a	AMBTOX90	12	Aug-Sep 1990	5
Hall et al., 1992a	AMBTOX91	13	Aug-Sep 1991	5
Hall et al., 1994a	AMBTOX93	14	Oct 1992 & Apr1993	2
Hall et al., 1996	AMBTOX94	12	Oct 1994	10
Hall et al., 1997a	AMBTOX95	8	Oct 1995	10
MDE, 1993, 1995	Fall Line Monitoring	164	monthly 1990-93	0.14
Reidel et al. in press	NOAA/COASTES	60	quarterly 1995-96	<0.3
Hall, 1985	Striped Bass Study '85	51	Apr 1985	10
Hall et al., 1986	Striped Bass Study '86	39	Apr 1986	20
Hall et al., 1987	Striped Bass Study '87	40	Apr 1987	10
Hall et al., 1989	Striped Bass Study '88	49	Apr-May 1988	3
Hall et al., 1991b	Striped Bass Study '89	71	Apr-May 1989	10
Hall et al., 1992b	Striped Bass Study '90	36	Apr-May 1990	10
Hall et al., 1994b, 1995	MD Coastal Plain (CPS)	120	Apr, Jun, Oct 1992-93	3
Velinsky et al., 1994	ICPRB	7	September 1992	1
Gruessner et al., 1997	DC ERA	36	Sept 1995-Sep 1996	<0.8
Culberson & Church, 1988	UDE	20	Aug 1985	<0.7

Table 2. Summary of zinc exposure data for all basins and stations. Maximum and 90th percentile concentrations (minimum of 4 detected concentrations) are presented by station and basin.

Basin				Concentration (µg/L)	
Data ID	Station	# Samples	# Detections	Maximum	90th percentile
Baltimore Harbor					
AMBTOX90,91	Patapsco R.	5	0	BLD	-
AMBTOX94	Bear Creek	1	0	BLD	-
AMBTOX94	Curtis Bay	1	0	BLD	-
AMBTOX94	Middle Branch	1	0	BLD	-
AMBTOX94	Northwest Harbor	1	0	BLD	-
AMBTOX94	Outer Harbor	1	0	BLD	-
AMBTOX94	Sparrows Point	1	0	BLD	-
Baltimore Harbor	Stations combined	11	0	BLD	-
C&D Canal					
Striped Bass Studies	Chesapeake City	37	19	80	49
Striped Bass Studies	Delaware City	16	16	55	78
Striped Bass Studies	Courthouse Pt.	18	18	38	46
C&D Canal	Stations combined	71	53	80	53
Chester					
CPS	URL	2	2	23	-
CPS	USE	2	2	30	-
Chester	Stations combined	4	4	30	35
Choptank					
Striped Bass Studies	Martinak	20	20	80	70
CPS	KGC	2	2	10	-
CPS	UTK	2	2	9	-
Choptank	Stations combined	24	24	80	66
James					
AMBTOX90	Elizabeth River	2	2	15	-
AMBTOX95	JRANN	1	0	BLD	-
AMBTOX95	JRBNN	1	0	BLD	-
AMBTOX95	Willoughby Bay	1	0	BLD	-
AMBTOX95	Lynnhaven River	1	0	BLD	-
Fall Line Monitoring	02035000	71	24	30	8
James	Stations combined	77	26	30	8
Lower Bay Mainstem					
UDE	CB1	1	1	1.7	-
UDE	CB2	1	1	2.8	-
UDE	CB3	1	1	0.8	-
UDE	CB5	1	1	3.6	-
UDE	CB6	1	1	0.7	-
UDE	CB7	1	1	2.8	-
UDE	CB8	1	1	1.9	-
Lower Bay Mainstem	Stations combined	7	7	3.6	5.2
Middle Bay Mainstem					
UDE	CB9	1	1	3.7	-
UDE	CB10	1	1	6.4	-
UDE	CB11	1	1	2.2	-
UDE	CB12	1	1	3.6	-

Basin Data ID	Station	# Samples	# Detections	Concentration (µg/L) Maximum	90th percentile
UDE	CB13	1	1	9.2	-
UDE	CB14	1	1	2.8	-
UDE	CR1D	1	1	9.6	-
Middle Bay Mainstem	Stations combined	7	7	9.6	12
Upper Bay Mainstem					
Striped Bass Studies	Grove	19	18	31	19
Striped Bass Studies	Howell	18	18	16	16
Striped Bass Studies	Spesutie	19	17	69	32
Striped Bass Studies	Elkton	6	6	24	26
Striped Bass Studies	Kentmore	5	5	28	34
Striped Bass Studies	Havre de Grace	6	5	13	16
UDE	CB15	1	1	5.6	-
UDE	CB16	1	1	1.0	-
UDE	CB17	1	1	29	-
UDE	CB18	1	1	3.4	-
UDE	CB19	1	1	1.5	-
UDE	CB20	1	1	26	-
Upper Bay Mainstem	Stations combined	79	75	69	23
Magothy					
AMBTOX94	Gibson Island	1	1	BLD	-
AMBTOX94	South Ferry	1	1	BLD	-
Magothy	Stations combined	2	2	BLD	-
Middle					
AMBTOX93	Frog Mortar	3	3	38	-
AMBTOX93	Wilson Point	3	3	134	-
Middle	Stations combined	6	6	134	140
Nanticoke					
AMBTOX93	Bivalve	2	2	23	-
AMBTOX93	Sandy Hill Beach	2	2	48	-
CPS	DMP	2	2	49	-
CPS	FBB	2	2	21	-
CPS	FBI	2	2	19	-
CPS	NDB	2	2	29	-
CPS	TLB	2	2	23	-
CPS	TWM	2	2	36	-
CPS	UMH	2	2	41	-
Nanticoke	Stations combined	18	18	49	56
Patuxent					
CPS	CAB	2	2	24	-
CPS	LYC	2	2	13	-
CPS	SEW	2	2	9.9	-
NOAA/COASTES	LPXT0173	4	3	0.79	-
NOAA/COASTES	PTXCF8747	4	4	1.1	1.6
NOAA/COASTES	PTXCF9575	4	4	1.1	1.3
NOAA/COASTES	PTXDE2792	4	2	0.98	-
NOAA/COASTES	PTXDE5339	4	2	0.46	-
NOAA/COASTES	PTXDE9401	4	3	0.98	-

Basin Data ID	Station	# Samples	# Detections	Concentration (µg/L) Maximum	 90th percentile
NOAA/COASTES	PTXDF0407	4	3	1.1	-
NOAA/COASTES	PTXED4892	4	4	2.4	3.4
NOAA/COASTES	PTXED9490	4	4	1.8	2.5
NOAA/COASTES	PXT0402	4	4	2.0	2.9
NOAA/COASTES	PXT0494	4	4	6.6	11
NOAA/COASTES	PXT0603	4	4	7.7	8.5
NOAA/COASTES	PXT0809	4	3	1.6	-
NOAA/COASTES	PXT0972	4	3	2.1	-
NOAA/COASTES	WBPXT0045	4	4	11	21
Patuxent Basin	Stations combined	66	57	24	7.1
Potomac					
AMBTOX90	Freestone Point	1	1	27	-
AMBTOX90	Indian Head	1	1	22	-
AMBTOX90	Morgantown	5	0	BLD	-
AMBTOX90	Possum Point	1	1	20	-
AMBTOX90	Dahlgren	5	1	7.4	-
CPS	BTM	2	2	26	-
CPS	CHP	2	2	8.9	-
CPS	COF	2	2	6.9	-
CPS	DYN	2	2	29	-
CPS	FOR	2	2	16	-
CPS	MTW	2	2	23	-
DC ERA	Anacostia 01649500	18	18	21	26
DC ERA	Anacostia 01651000	18	18	16	14
ICPRB	Anacostia T120	2	2	3.5	-
ICPRB	Anacostia T800	2	2	4.8	-
ICPRB	Anacostia T500	2	2	2.3	-
ICPRB	Anacostia T1100	1	1	2.1	-
Striped Bass Studies	Cherry Hill	13	13	110	73
Striped Bass Studies	Maryland	25	25	310	86
Striped Bass Studies	Mid	26	26	220	73
Striped Bass Studies	Virginia	32	32	184	96
Striped Bass Studies	Quantico	13	13	90	75
Striped Bass Studies	Widewater	13	12	120	142
Potomac	Stations combined	190	180	310	70
Sassafras					
AMBTOX94	Betterton	1	0	BLD	-
AMBTOX94	Turners Creek	1	0	BLD	-
CPS	MLC	2	2	15	-
Sassafras	Stations combined	7	2	15	-
Susquehanna					
Fall Line Monitoring	01578310	93	59	22	9.3
Severn					
AMBTOX94	Junction Rt. 50	1	1	BLD	-
AMBTOX94	Annapolis	1	1	BLD	-
Severn	Stations combined	2	2	BLD	-

Basin Data ID	Station	# Samples	# Detections	Concentration (µg/L) Maximum	90th percentile
West Chesapeake					
CPS	BEB	2	2	29	-
CPS	BRB	2	2	29	-
CPS	NRV	2	2	23	-
West Chesapeake	Stations combined	6	6	29	35
Wye					
AMBTOX90,91,93	Manor House	7	2	28	-
AMBTOX93	Quarter Creek	2	2	29	-
Wye	Stations combined	9	4	29	36
York					
AMBTOX95	YRACA	1	0	BLD	-
AMBTOX95	YRBCA	1	0	BLD	-
AMBTOX95	PRAWP	1	0	BLD	-
AMBTOX95	PRBWP	1	0	BLD	-
York	Stations combined	4	0	BLD	-

Table 3. Freshwater acute zinc toxicity data presented in order from most to least sensitive species. Symbols used include: *NR=not reported, S=static test, N=nominal concentration, F=flow-thru test, M=measured concentration, R= renewal test

Species	Method	Chemical	Hardness (mg/L as $CaCO_3$)	LC50 (ug/L)	Hard adj. LC50 (ug/L)	Duration & Effect	Reference
Cladoceran, *Ceriodaphnia reticulata*	S,N	zinc chloride	45	32	34.99	48 hr LC50	Carlson & Roush 1985
Green algae, *Selenastrum capricornutum*	S,N	zinc sulfate	*NR	50		72 hr EC50, GRO	Vasseur et al. 1988
Cladoceran, *Daphnia magna*	S,N	zinc	45	68	74.35	48 hr LC50	Mount & Norberg 1984
Chinook salmon, *Onchorhynchus tshawytscha*	F,M	zinc sulfate	21	84	175.2	96 hr LC50	Finlayson & Verrue 1982
Cutthroat trout, *Salmo clarki*	S,M	zinc sulfate	*NR	90		96 hr TLm	Rabe & Sappington 1970
Rainbow trout, *Oncorhynchus mykiss*	F,M	zinc chloride	23	93	179.6	96 hr LC50	Chapman 1975
Cladoceran, *Ceriodaphnia dubia*	S, M	zinc sulfate	289.8 280-300	95	21.4	48 hr LC50	Schubauer-Berigan et al. 1993
Cladoceran, *Daphnia pulex*	S,N	zinc	45	107	117.0	48 hr LC50	Mount & Norberg 1984
Arctic grayling, *Thymallus arcticus*	S	zinc chloride	soft*	112		96 hr LC50	Buhl & Hamilton 1990

Fathead minnow, *Pimephales promelas*	R	zinc sulfate	46.5	204	216.9	96 hr LC50	Norberg-King 1989
Loach, *Noemacheilus montanus*	S		*NR	140		96 hr LC50	Joshi & Chamoli 1987
Cladoceran, *Daphnia similis*	S		*NR	165		96 hr LC50	Soundrapandian & Venkataraman 1990
Ciliate, *Chilodonella uncinata*	S	zinc chloride	*NR	<170		24 hr LC50	Madoni et al. 1996
Big claw river shrimp, *Macrobrachium carcinus*	S	zinc sulfate	*NR	200		96 hr LC50	Correa 1987
Mussel, *Anodonta imbecillis*	S	zinc sulfate	39	268	330.8	96 hr LC50	Keller & Zam 1991
Snail, *Bellamya bengalensis*	R	zinc sulfate	*NR	280		96 hr LC50	Rao & Jayasree 1987
Scud, *Hyalella azteca*	S, M	zinc sulfate	290	290	65.4	96 hr LC50	Schubauer-Berigan et al. 1993
Cyprinid fish, *Barilius bendelisis*	S	zinc	*NR	400		*NR	Deoray & Wagh 1987
Striped Bass, *Morone saxatilis*	S,N	zinc chloride	285	430	98.4	96 hr LC50	Palawski et al. 1985

Snail, *Physa heterostropha*	S,N	zinc sulfate	100 (10.6 C)	434	241.2	96 hr LC50	Wurtz 1962
Cladoceran, *Daphnia lumholzi*	R	zinc	*NR	437.5		96 hr LC50	Vardia et al. 1988
Shrimp, *Paratya compressa*	S	zinc	*NR	722		48 hr LC50	Hatakeyama & Sugaya 1989
Atlantic salmon, *Salmo salar*	F,M	zinc sulfate	14	740	2,176		Carson & Carson 1972
Sockeye salmon, *Onchorhynchus nerka*	F,M	zinc chloride	22	749	1,502	96 hr LC50	Chapman 1975,1978a
Longfin dace, *Agosia chrysogaster*	R,M	zinc sulfate	217	790	227.8	96 hr LC50	Lewis 1978
Coho salmon, *Oncorhynchus kisutch*	F,M	zinc chloride	25	905	1,628	96 hr LC50	Chapman & Stevens 1978
Nematode, *Caenorhabditis elegans*	S	zinc chloride	*NR	1000		96 hr LC50	Williams & Dusenbery 1990
Snail, *Helisoma campanulatum*	S,N	zinc sulfate	100 (22.8 C)	1,270	705.9	96 hr LC50	Wurtz 1962
Snail, *Physa gyrina*	F,M	zinc chloride	36	1,274	1,683	96 hr LC50	Nebeker et al. 1986

Diatom, *Navicula seminulum*			58	1,320	1,164	120 hr EC50	Academy of Natural Sciences 1960
Rotifer, *Brachionus calyciflorus*	S	zinc chloride	36.2	1,320	1,735	24 hr LC50	Couillard et al. 1989
Water flea, *Moina macrocopa*	S,N	zinc sulfate	*NR	1,320		48 hr LC50	Hatakeyama & Sugaya 1989
Flagfish, *Jordanella floridae*	F,M	zinc sulfate	44	1,500	1,672	96 hr LC50	Spehar 1976a,b
Brook trout, *Salvelinus fontinalis*	F,M	zinc sulfate	46.8	1,550	1,639	96 hr LC50	Holcombe & Andrew 1978
Mozambique tilapia, *Tilapia mossambica*	S,N	zinc chloride	115	1,600	790	96 hr LC50	Qureshi & Saksena 1980
Pond snail, *Lymnaea luteola*	R	zinc sulfate	200.6 175-230	1,680	518	96 hr LC50	Khangarot & Ray 1987b
Colorado squawfish, *Ptychocheilus lucius*	S	zinc chloride	191.2 182-201	1,700	546	96 hr LC50	Hamilton 1995
Guppy, *Poecilia reticulata*	S,M	zinc sulfate	30	1,740	2,682		Pierson 1981
Ciliate, *Colpidium campylum*	S	zinc chloride	*NR	1,850		24 hr EC50, IMM	Madoni et al. 1992
Leech, *Erpobdella octoculata*	R	zinc sulfate	*NR	2,050		96 hr LC50	Willis 1989

White Sucker, *Catostomus commersoni*	F,M	zinc chloride	18	2,200	5,228	96 hr LC50	Duncan & Klaverkamp 1983
Scud, *Gammarus lacustris*	F	zinc	*NR	2,240		96 hr LC50	De March 1988
Tubificid worm, *Limnodrilus hoffmeisteri*	S,N	zinc sulfate	100	>2,274	>1,264	11 day, LC50	Wurtz and Bridges 1961
Ciliate, *Euplotes* sp.	S	zinc chloride	*NR	2,390		24 hr LC50	Madoni et al. 1996
Protozoa, *Aspidisca cicada*	S	zinc chloride	*NR	2,400		24 hr EC50	Madoni et al. 1992
Green algae, *Chlorella vulgaris*		zinc sulfate	*NR	2,400		96 hr EC50	Rachlin and Farran 1974
Ciliate, *Paramecium caudatum*	S	zinc chloride	*NR	2,500		24 hr EC50, IMM	Madoni et al. 1992
Protozoa, *Uronema nigricans*	S	zinc chloride	*NR	2,900		24 hr EC50, IMM	Madoni et al. 1992
Razorback sucker, *Xyrauchen texanus*	S	zinc chloride	199.4 196-203	2,920	904	96 hr LC50	Buhl & Hamilton 1996
Protozoa, *Euplotes affinis*	S	zinc chloride	*NR	3,100		24 hr EC50 IMM	Madoni et al. 1992

Common Carp, *Cyprinus carpio*	R,N	zinc sulfate	19	3,120	7,083	96 hr LC50	Khangarot et al. 1983
River Limpet, *Ancylus fluviatilis*	S	zinc sulfate	*NR	3,200		96 hr LC50	Willis 1988
Cyclopoid copepod, *Cyclops* sp.	S	zinc	*NR	3,310		48 hr LC50	Abbasi et al. 1988
Bluegill, *Lepomis macrochirus*	F,M	zinc chloride	45	3,314	3,623	96 hr TLm	Cairns & Scheier 1959
Northern squawfish, *Ptychochellus Oregonensis*	F,M	zinc chloride	24.4 20-30	3,498	6,423	96 hr LC50	Andros & Garton 1980
Clawed toad, *Xenopus laevis*	R	zinc sulfate	100	3,600	2,001	96 hr EC50, ABN	Dawson et al. 1988
Ciliate, *Vorticella convallaria*	S	zinc chloride	*NR	3,790		24 hr LC50	Madoni et al. 1996
Diatom, *Nitzschia linearis*	S	zinc chloride	294.6	4,300	957	120 hr LC50	Patrick et al. 1968
Bryozoan, *Pectinatella magnifica*	S,N	zinc	204.4 190-220	4,310	1,307	96 hr LC50	Pardue & Wood 1980
Fish, *Lepidocephalichthyes guntea*	S	zinc sulfate	*NR	4400		96 hr LC50	Bengeri & Patil 1987

Bonytail, *Gila elegans*	S	zinc chloride	191.2 182-201	4,800	1,540	96 hr LC50	Hamilton, S.J. 1995
Bryozoan, *Plumatella emarginata*	S,N	zinc	204.4 190-220	5,300	1,607	96 hr LC50	Pardue & Wood 1980
Bryozoan, *Lophopodella carteri*	S,N	zinc	204.4 190-220	5,630	1,707	96 hr LC50	Pardue & Wood 1980
Golden shiner, *Notemigonus crysoleucas*	S,N	zinc sulfate	50	6,000	6,000	96 hr TLm	Cairns et al. 1969
Asiatic clam, *Corbicula fluminea*	S,M	zinc sulfate	64	6,040	4,900	96 hr LC50	Cherry et al. 1980 Rodgers et al. 1980
Worm, oligochaete *Lumbriculus variegatus*	S,N	zinc chloride	30	6,300	9,712	9 day, LC50	Bailey & Liu 1980
Green alga, *Chlorella saccharophila*	S,M	zinc sulfate	*NR	7,100		96 hr EC50	Rachlin et al. 1982
Goldfish, *Carassius auratus*	S,N	zinc sulfate	50	7,500	7,500	96 hr TLm	Cairns et al. 1969
Amphipod, *Gammarus sp.*	S,M	zinc	50	8,100	8,100	96 hr TLm	Rehwoldt et al. 1973
Ostracod, *Cypris subglobosa*	R	zinc	*NR	8,352		96 hr LC50	Vardia et al. 1988
Isopod, *Lirceus alabamae*	F,M	zinc sulfate	152	8,375	3,265		Bosnak & Morgan 1981

Isopod, *Asellus communis*	S,N	zinc sulfate	100	8,755	4,866	96 hr LC50	Wurtz & Bridges 1961
Diatom, *Navicula incerta*	S,M	zinc chloride	*NR	10,000		96 hr EC50	Rachlin et al. 1983
Duckweed, *Lemna minor*			*NR	10,000		96 hr EC50	Wang 1986
Zebra danio, zebrafish *Brachydanio rerio*	R	zinc chloride	*NR	10,000		168 hr LC50	Van Leeuwen et al. 1990
Southern platyfish, *Xiphophorus maculatus*	S,N	zinc sulfate	166	12,000	4,341		Rachlin & Perlmutter 1968
Tilapia, *Tilapia zillii*	S	zinc sulfate	*NR	13,000		96 hr LC50	Hilmy et al. 1987
Snail, *Amnicola sp.*	S,M	zinc	50	14,000	14,000	96 hr TLm	Rehwoldt et al. 1973
White perch, *Morone americana*	S,M	zinc nitrate	55	14,400	13,280	96 hr TLm	Rehwoldt et al. 1972
American eel, *Anguilla rostrata*	S,N	zinc nitrate	55	14,500	13,380	96 hr TLm	Rehwoldt et al. 1972
Tubificid worm, *Tubifex tubifex*	R	zinc sulfate	*NR	17,780		96 hr EC50, IMM	Khangarot 1991
Worm, *Nais sp.*	S,M	zinc	50	18,400	18,400	96 hr TLm	Rehwoldt et al. 1973

Banded killifish, *Fundulus diaphanus*	S,M	zinc nitrate	55	19,200	17,710	96 hr TLm	Rehwoldt et al. 1972
Amphipod, *Crangonyx psuedogracilis*	R,N	zinc sulfate	50	19,800	19,800	96 hr EC50, IMM	Martin & Holdich 1986
Common indian toad, *Bufo melanostictus*	S	zinc sulfate	188.3 165-215	19,860	6,457	96 hr LC50	Khangarot & Ray 1987a
Pumpkinseed, *Lepomis gibbosus*	S,M	zinc nitrate	55	20,100	18,540	96 hr TLm	Rehwoldt et al. 1972
Isopod, *Asellus bicrenata*	F,M	zinc sulfate	220	20,110	5,731		Bosnak & Morgan 1981
Frog, *Microhyla ornata*	R	zinc sulfate	*NR	22,410		96 hr LC50	Rao & Madhyastha 1987
Catfish, *Clarias lazera*	S	zinc sulfate	*NR	26,000	NR	96 hr LC50	Hilmy et al. 1987
Damselfly, *Argia sp*	S,N	zinc sulfate	20	40,930	88,960	120 hr LC50	Wurtz & Bridges 1961
Harlequinfish, red rasbora, *Rasbora heteromorpha*	S	zinc salt	*NR	46,400		48 hr LC50	Svobodova & Vykusova 1988
Ciliate, *Aspidisca lynceus*	S	zinc chloride	*NR	50,000		24 hr LC50	Madoni et al. 1996
Protozoa, *Euplotes patella*	S	zinc chloride	*NR	50,000		24 hr EC50, IMM	Madoni et al. 1992

Bullfrog, *Rana catesbeiana*		zinc	*NR	70,000	96 hr LC50	Zhang et al. 1992
Indian catfish, *Heteropneustes fossilis*	R	zinc chloride	*NR	75,000	96 hr LC50	Hemalatha & Banerjee 1993
Jaguar guapote, *Cichlasoma managuense*		zinc	*NR	77,980	96 hr LC50	Zhang et al. 1992
Snake-head catfish, *Channa punctatus*	R	zinc chloride	*NR	80,000	96 hr LC50	Dalal & Bhattacharya 1994
Green alga, *Chlorella pyrenoidosa*		zinc sulfate	*NR	>200,000	96 hr LC50	Wong et al. 1979
Green alga, *Chlorella salina*		zinc sulfate	*NR	>200,000	96 hr LC50	Wong et al. 1979
Green alga, *Scenedesmus quadricauda*		zinc sulfaste	*NR	>200,000	96 hr LC50	Wong et al. 1979
Climbing perch, *Anabas testudineus*	S	zinc chloride	*NR	260,000	24 hr LC50	Banerjee & Kumari 1988

Table 4. The 10^{th} percentile intercepts for freshwater and saltwater zinc toxicity data by test duration and trophic group. These values represent protection of 90% of the test species.

Water type	Acute or Chronic	Trophic Group	n	10^{th} Percentile (µg/L)
Freshwater*	acute	All species	55	142
		amphibians	2	629
		zooplankton	5	4.3
		benthos	20	212
		fish	26	216
		plants	2	789
Freshwater*	chronic	All species	12	11
		zooplankton	3	0.8
		benthos	2	74
		fish	7	56
Saltwater	acute	All species	82	79
		zooplankton	10	46
		benthos	51	102
		fish	12	69
		plants	9	10
Saltwater	chronic	All species	6	8.7

* Hardness adjusted values are used (50 mg/L).

Table 5. Saltwater acute zinc toxicity data presented in order from most to least sensitive species. Symbols used include: *NR=not reported, S=static test, N=nominal concentration, F=flow-thru test, M=measured concentration, R=renewal test

Species	Method	Chemical	LC50 (ug/L)	Duration & Effect	Reference
Diatom, *Schroederella schroederi*		zinc sulfate	19.01	96 hr EC50 GRO	Kayser 1977
Purple sea urchin, *Strongylocentrotus purpurat*	S	zinc chloride	23	120 hr EC50, DVP	Dinnel et al. 1989
Diatom, *Thalassiosira rotula*		zinc sulfate	25.8	120 hr EC50 GRO	Kayser 1977
Sand dollar, *Dendraster excentricus*	S	zinc chloride	28	1.3 hr EC50, REP	Dinnel et al. 1989
Inland silverside, *Menidia beryllina*	R	zinc chloride	30	96 hr LC50	Lewis 1993
Red abalone, *Haliotis rufescens*	F,U	zinc sulfate	37	48 hr EC50	Conroy et al. 1996
Calanoid copepod, *Temora stylifera*	S,N	zinc sulfate	40	48 hr LC50	Nipper et al. 1993
Mysid, *Acanthomysis costata*	R, M	zinc sulfate	46	168 hr LC50	Martin et al. 1989
Brine shrimp, *Artemia franchiscana*	S	zinc sulfate	65	72 hr LC50	MacRae & Pandey 1991

Coccolithophorid, *Cricosphaera carterae*		zinc sulfate	76.69	96 hr EC50 GRO	Stillwell 1977
Mediterranean mussel, *Mytilus galloprovincialis*	S	zinc sulfate	145	48 hr EC50, DVP	Pavicic et al. 1994
Lobster, *Homarus americanus*	S,N	zinc chloride	175	96 hr LC50	Johnson 1985
Cabezon, *Scorpaenichthys marmoratus*	S	zinc chloride	191	96h EC50, IMM	Dinnel et al. 1989
Quahog clam, *Mercenaria mercenaria*	S,N	zinc chloride	195	48 hr LC50, MOR	Calabrese & Nelson 1974
Eastern oyster, *Crassostrea virginica*	S,N	zinc chloride	205.7	48 hr EC50, DVP	MacInnes & Calabrese 1978
Pacific oyster, *Crassostrea gigas*	S,M	zinc chloride	206.5		Dinnel et al. 1983
Diatom, *Nitzschia closterium*		zinc sulfate	271	96 hr EC50 GRO	Rosko and Rachlin 1975
Copepod, *Acartia tonsa*	S,N	zinc chloride	294.2	96 hr LC50	Lussier & Cardin 1985
Fleshy prawn, *Penaeus chinensis*	S	zinc sulfate	300	96 hr LC50	Wu & Chen 1988

Opossum shrimp, *Mysidopsis bahia*	S	zinc chloride	303	96 hr LC50	Cripe 1994
Rea sea urchin, *Strongylocentrotus francisc*	S	zinc chloride	313	1.3 hr EC50, REP	Dinnel et al. 1989
Dinoflagellate, *Procentrum micans*		zinc sulfate	319.1	96 hr EC50 GRO	Kayser 1977
Calanoid copepod, *Acartia lilljeborgi*	S,N	zinc sulfate	320	48 hr LC50	Nipper et al. 1993
Shrimp, *Mysidopsis juniae*	S,N	zinc sulfate	340	96 hr LC50	Nipper et al. 1993
Green sea urchin, *Strongylocentrotus droebach*	S	zinc chloride	383	1.3 hr EC50, REP	Dinnel et al. 1989
Hermit Crab, *Pagurus longicarpus*	S,N	zinc chloride	400	96 hr LC50	Eisler & Hennekey 1977
Harpacticoid copepod, *Tisbe holothuriae*	S	zinc sulfate	421	48 hr LC50	Verriopoulos & Moraitou-Apostolopoulou 1989
Striped bass, *Morone saxatilis*	S,N	zinc chloride	430	96 hr LC50	Palawski et al. 1985
Mysid, *Mysidopsis bahia*	F,M	zinc chloride	499	96 hr LC50	Lussier et al. 1985
Red tongue sole, *Cynoglossus joyneri*	S	zinc sulfate	500	96 hr LC50	Cui et al. 1987

Scud, *Allorchestes compressa*	R,M	zinc chloride	580	96 hr LC50	Ahsanullah 1976
Dungeness crab, *Cancer magister*	S	zinc chloride	586	96 hr EC50, IMM	Dinnel et al. 1989
Mysid, *Mysidopsis bigelowi*	S,M	zinc chloride	591.3	96 hr LC50	Lussier & Gentile 1985
Diatom, *Thalassiosira pseudonana*	S	zinc chloride	823.1	72 hr EC50 GRO	Fisher et al. 1984
Harpacticoid copepod, *Nitocra spinipes*	F	zinc chloride	850	96 hr LC50	Bengtsson & Bergstrom 1987
Polychaete worm, *Neanthes arenaceodentata*	S,N	zinc sulfate	900	96 hr LC50	Reish et al. 1976
Amphipod, *Corophium volutator*	S,N	zinc sulfate	1,000	96 hr LC50	Bryant et al. 1985
Green crab, *Carcinus maenas*	S,N	zinc sulfate	1,000	48 hr LC50	Connor 1972
Pink shrimp, *Penaeus duorarum*	S	zinc chloride	1,050	96 hr LC50	Cripe 1994
Sheepshead minnow, *Cyprinodon variegatus*	R	zinc chloride	1,000-10,000	96 hr LC50	Lewis 1993
Polychaete worm, *Ophryotrocha diadema*	S,N	zinc sulfate	1,400	96 hr LC50	Reish & Carr 1978

Polychaete worm, *Nereis diversicolor*	R,N	zinc sulfate	1,500	96 hr LC50	Bryan & Hummerstone 1973
Copepod, *Acartia clausi*	S,N	zinc chloride	1,507	72 hr LC50	Lussier & Cardin 1985
Brine shrimp, *Artemia salina*	*NR	zinc	1,700	48 hr LC50	Govindarajan et al. 1993
Polychaete worm, *Capitella capitata*	S,N	zinc sulfate	1,700	96 hr LC50	Reish et al. 1976
Polychaete worm, *Neanthes grubei*	S	zinc chloride	1,800	96 hr LC50	Reish & LeMay 1991
Polychaete worm, *Ophryotrocha labronica*	S	zinc chloride	1,800	96 hr LC50	Reish & LeMay 1991
Scud, *Corophium insidiosum*	*NR	zinc chloride	1,900	96 hr LC50	Reish 1993
Squid, *Loligo opalescens*	S	zinc chloride	>1,920	96 hr EC50, IMM	Dinnel et al. 1989
Indian prawn, *Penaeus indicus*	*NR	zinc	1,990	48 hr LC50	Govindarajan et al. 1993
Bay scallop, *Argopecten irradians*	R	zinc chloride	2,250	96 hr LC50	Nelson et al. 1988
Blue mussel, *Mytilus edulis planulatus*	R,M	zinc chloride	2,500	96 hr LC50	Ahsanullah 1976
Atlantic silverside, *Menidia menidia*	S,N	zinc chloride	2,728		Cardin 1985

Cone worm, *Pectinaria californiensis*	S	zinc chloride	2,800	96 hr LC50	Reish & LeMay 1991
Surf clam, *Spisula solidissima*	R	zinc chloride	2,950	96 hr LC50	Nelson et al. 1988
Green mussel, *Perna viridis*	*NR	zinc	3,100	24 hr LC50	Govindarajan et al. 1993
Dinoflagellate, *Gymnodinium splendens*		zinc sulfate	3,716	96 hr EC50 GRO	Kayser 1977
Copepod, *Eurytemora affinis*	S,N	zinc chloride	4,074		Lussier & Cardin 1985
Rotifer, *Brachionus plicatilis*	*NR	zinc	>4,800	24 hr LC50	Snell et al. 1991b
Winter flounder, *Pseudopleuronectes americanus*	S,N	zinc chloride	4,922		Cardin 1985
Mollusk, *Neotrigonia margaritacea*	R,M	zinc chloride	>5,000	96 hr LC50	Ahsanullah 1976
Soft-shell clam, *Mya arenaria*	S,N	zinc chloride	5,200	96 hr LC50	Eisler 1977a
Polychaete worm, *Neanthes vaali*	R,M	zinc chloride	5,500	96 hr LC50	Ahsanullah 1976
Tidewater silverside, *Menidia peninsulae*	S,N	zinc sulfate	5,600		Hansen 1983

Nematode, *Monhystera disjuncta*	S	zinc	5,700	96 hr EC50, DVP	Vranken et al. 1988
Hirame flounder, *Paralichthys olivaceus*	*NR	zinc	6,700	48 hr LC50	Wu et al. 1990
Polychaete worm, *Ctenodrilus serratus*	S,N	zinc sulfate	7,100	96 hr LC50	Reish & Carr 1978
Polychaete worm, *Nereis virens*	S,N	zinc chloride	8,100	96 hr LC50	Eisler & Hennekey 1977
Santa Domingo falsemussel, *Mytilopsis sallei*	R	zinc sulfate	8,360	96 hr LC50	Uma Devi 1995
Starfish, *Patiriella exigua*	R,M	zinc chloride	>10,000	96 hr LC50	Ahsanullah 1976
Diatom, *Navicula incerta*		zinc chloride	10,100	96 hr EC50 GRO	Rachlin et al. 1983
Crab, *Paragrapsus quadridentatus*	F,M	zinc chloride	10,500	120 hr LC50	Ahsanullah 1976
Daggerblade grass shrimp, *Palaemonetes pugio*	S	zinc chloride	11,300	48 hr LC50	Burton & Fisher 1990
Green alga, *Dunaliella tertiolecta*	S	zinc chloride	13,000	72 hr EC50 GRO	Fisher et al. 1984

Mummichog, *Fundulus heteroclitus*	S,N	zinc chloride	17,500	96 hr TLm	Dorfman 1977
Fiddler crab, *Uca annulipes*	R	zinc sulfate	31,930	96 hr LC50	Devi 1987
Spot, *Leiostomus xanthurus*	S,N	zinc sulfate	38,000		Hansen 1983
Starfish, *Asterias forbesil*	S,N	zinc chloride	39,000	96 hr LC50	Eisler & Hennekey 1977
Fiddler crab, *Uca triangularis*	R	zinc sulfate	39,050	96 hr LC50	Devi 1987
Mud snail, *Nassarius obsoletus*	S,N	zinc chloride	50,000	96 hr LC50	Eisler & Hennekey 1977
Clam, *Macoma balthica*	S,N	zinc sulfate	60,000	96 hr LC50	Bryant et al. 1985
Rivulus, *Rivulus marmoratus*	F	zinc sulfate	119,300	96 hr LC50	Lin & Dunson 1993

Table 6. Freshwater chronic zinc toxicity data presented in order from most to least sensitive species. Symbols used include: LC=life cycle test, ELS= early life stage test, R=renewal test, S=static test, F=flow-thru test, M=measured concentration, N=nominal concentration, REP=reproduction endpoint, GRO=growth endpoint, MOR=mortality endpoint, HAT=hatching endpoint, DVP=development endpoint

Species	Method	Chemical	Hardness (mg/L as $CaCO_3$)	Chronic Value* (ug/L)	Hardness adjusted Chronic value (ug/L)	Duration & Effect	Reference
Cladoceran, *Moina irrasa*	R	Zinc chloride	<5	NOEC 25	176	3 week, REP	Zou 1997.
Flagfish, *Jordanella floridae*	LC	zinc sulfate	44	LOEC 51	57	30 day, GRO	Spehar 1976a,b
Cladoceran, *Daphnia magna*	LC	zinc chloride	211	46.73	14		Chapman et al. manuscript
Snail, *Potamopyrgus jenkinsi*	R	zinc chloride	*NR	59.7		77-112 day, GRO	Dorgelo, J. et al. 1995
Fathead minnow, *Pimephales promelas*	LC	zinc sulfate	46	106.3	114	8 wk, MOR	Benoit & Holcombe 1978
Guppy, *Poecilia reticulata*	LC	zinc sulfate	30	<173	267	134 day GRO	Pierson 1981
Sockeye salmon, *Onchorhynchus nerka*	ELS	zinc chloride	32-37	>242	332	21 month MOR	Chapman 1978
Rainbow trout, *Salmo gairdneri*	ELS, F	zinc sulfate	25	191	344	21 month MOR	Sinley et al. 1974

Cladoceran, *Ceriodaphnia reticulata*	R,M	zinc	372.5	221.4	5.5	7 day, MOR	Carlson & Roush 1985
Chinook salmon, *Oncorhynchus tshawytscha*	ELS, F	zinc chloride	25	371.1	668	4 month, MOR	Chapman 1975
Snail, *Physa gyrina*	F, M	zinc chloride	36	NOEC 570	753	30 day MOR	Nebeker et al. 1986
Brook trout, *Salvelinus fontinalis*	LC, F	zinc sulfate	45.9	854.7	919	4-27 month, MOR	Holcombe et al. 1979
Zebra danio zebrafish, *Brachydanio rerio*	R		*NR	1,102		< 16 day, HAT	Dave et al. 1987
Caddisfly, *Clistoronia magnifica*	LC	zinc chloride	31	>5,243	>7,860	76 day DVP	Nebecker et al. 1984

* If chronic value was not reported a NOEC or LOEC was used.

Table 7. Saltwater chronic zinc toxicity data presented in order from most to least sensitive species . Symbols used include: LC=life cycle test, ELS= early life stage test, R=renewal test, S=static test, F=flow-thru test, M=measured concentration, N=nominal concentration, REP=reproduction endpoint, GRO=growth endpoint, MOR=mortality endpoint, HAT=hatching endpoint, DVP=development endpoint

Species	Method	Chemical	Chronic Value* (ug/L)	Duration & Effect	Reference
Mysid, *Acanthomysis costata*	R,M	zinc sulfate	NOEC 18	7 day, MOR	Martin et al. 1989
Red abalone, *Haliotis rufescens*	F	zinc sulfate	NOEC 19	9 day, DVP	Hunt & Anderson 1989
Pacific oyster, *Crassostrea gigas*	S, N	zinc	30	6 day EC50, DVP	Watling 1983
Scud, *Allorchestes compressa*	F	zinc sulfate	LOEC 99	28 day, MOR	Ahsanullah & Williams 1991
Mysid, *Mysidopsis bahia*	R,M	zinc sulfate	152 MATC	7 day, MOR	Harmon and Langdon 1996
Mysid, *Mysidopsis intii*	R,M	zinc sulfate	152 MATC	7 day, MOR	Harmon and Langdon 1996

* If chronic value was not reported a NOEC or LOEC was used.

Table 8. The percent probability of exceeding the zinc acute freshwater (FW) or saltwater (SW)10th percentile for all species and the percent probability of exceeding the acute 10th percentile for the most sensitive trophic group with n>8.

Location	Acute 10^{th} Percentile ($\mu g/L$) ( ) - most sensitive trophic group 10th percentile		% Probability > 10th percentile	
Middle River (SW)	79	(10 - plants)	21	(79)
Potomac River (FW)	142	(212 - benthos)	3.3	(1.6)
Wye River (SW)	79	(10 - plants)	2.7	(42)
Choptank River (FW)	142	(212 - benthos)	1.2	(0.3)
Nanticoke River (FW)	142	(212 - benthos)	0.8	(0.2)
C and D Canal (FW)	142	(212 - benthos)	0.6	(0.1)
James River (FW)	142	(212 - benthos)	0.3	(0.2)
Patuxent River (SW)	79	(10 - plants)	0.3	(7.6)
Upper mainstem Bay (FW)	142	(212 - benthos)	<0.1	(<0.1)
Chester River (FW)	142	(212 - benthos)	<0.1	(<0.1)
Susquehanna River (FW)	142	(212 - benthos)	<0.1	(<0.1)
Lower mainstem Bay (SW)	79	(10 - plants)	<0.1	(1.4)
West Chesapeake (FW)	142	(212 - benthos)	<0.1	(<0.1)
Middle mainstem Bay (SW)	79	(10 - plants)	<0.1	(16)

FIGURES

Figure 1. Ecological risk assessment approach.

PROBLEM FORMULATION

- Stressor Characteristics
- Exposure Data
- Ecological Effects Data
- Risk Characterization
- Endpoints
- Stressors Potentially Impacting Aquatic Communities
- Conceptual Model

↓

ANALYSIS

Characterization of Exposure: Water column monitoring data on zinc in the Chesapeake Bay watershed.

Characterization of Ecological Effects: Laboratory toxicity studies.

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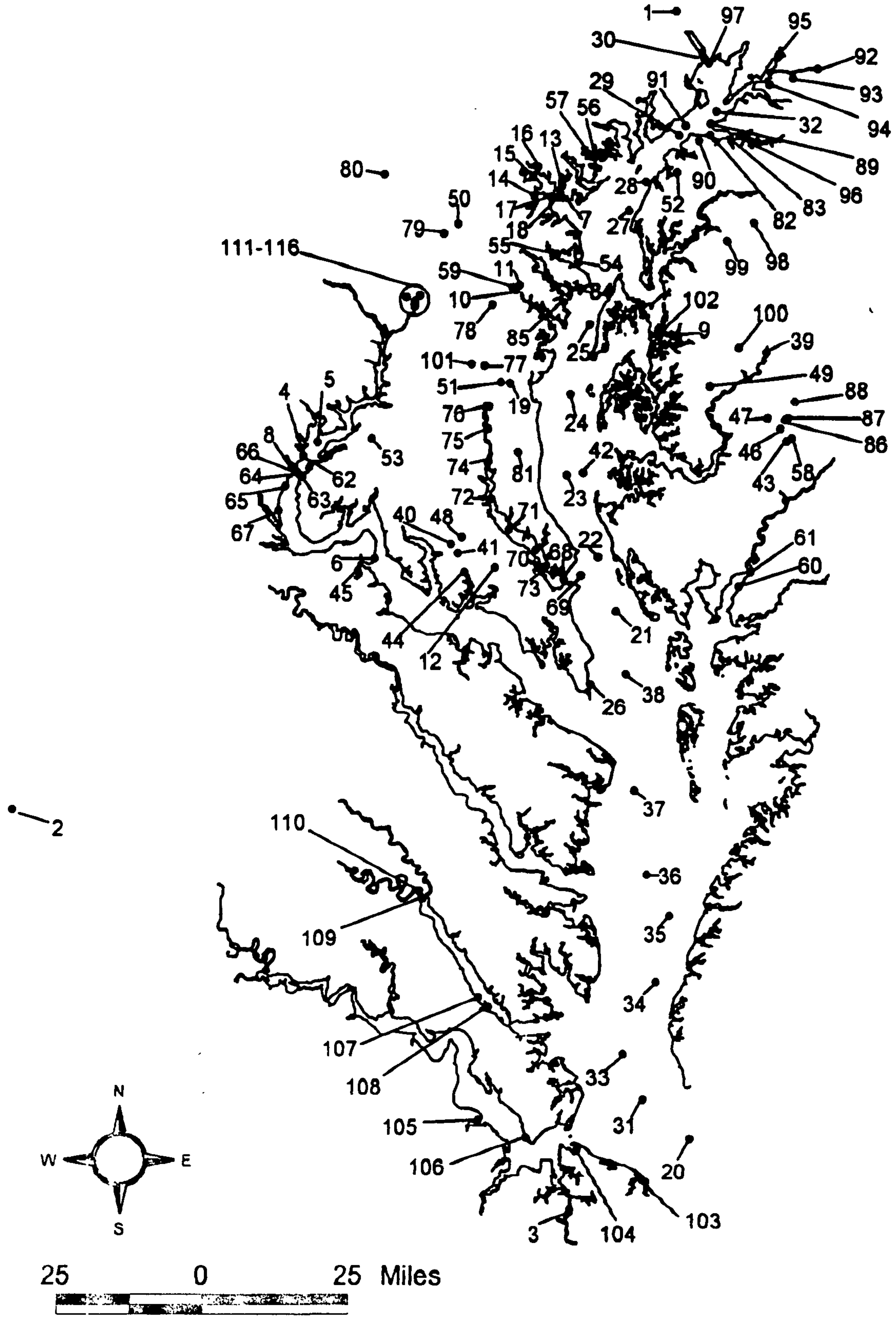
RISK CHARACTERIZATION

Probabilistic comparison of species sensitivity and surface water exposure distributions.

Figure 2. Location of the 116 stations where zinc concentrations were measured from 1985 to 1996. See key to map where stations are described.



Key to map for Figure 2 Stations where zinc was measured from 1985 to 1996. Latitude and longitude coordinates are given in decimal degrees. Station number corresponds to station location on Figure 2. Abbreviated station names are in parentheses.

Station number	Station	Latitude	Longitude
1	Susquehanna River Fall Line (1578310)	39.6586	76.1744
2	James River Fall Line (2035000)	37.6708	78.0861
3	Elizabeth River	36.8081	76.2933
4	Freestone Point	38.5833	77.2667
5	Indian Head	38.6000	77.2167
6	Morgantown	38.3337	77.0157
7	Patapsco River	39.2167	76.5000
8	Possum Point	38.5362	77.2920
9	Wye River (Manor House)	38.9028	76.1298
10	Bell Branch (BEB)	38.9917	76.6333
11	Bacon Ridge Branch (BRB)	38.9992	76.6136
12	Burnt Mill Creek (BTM)	38.3322	76.6369
13	Bear Creek	39.2358	76.4961
14	Curtis Bay	39.2064	76.5803
15	Middle Branch	39.2528	76.5883
16	North West Harbor	39.2767	76.5742
17	Outer Harbor	39.2089	76.5247
18	Sparrows Point	39.2081	76.5075
19	Cabin Branch (CAB)	38.7694	76.6528
20	CB1	36.9950	75.9467
21	CB10	38.2467	76.2617
22	CB11	38.3717	76.3233
23	CB12	38.5633	76.4317
24	CB13	38.7517	76.4350
25	CB14	38.9183	76.3883
26	CB15	38.0717	76.3233
27	CB16	39.1883	76.2883
28	CB17	39.2567	76.2400
29	CB18	39.3683	76.1433
30	CB19	39.5500	76.0800
31	CB2	37.0833	76.0950
32	CB20	39.4300	76.0333
33	CB3	37.1883	76.1633
34	CB5	37.3650	76.0750
35	CB6	37.5267	76.0433
36	CB7	37.6200	76.1200
37	CB8	37.8217	76.1750
38	CB9	38.1000	76.2200
39	Martinak	38.8750	75.8417
40	Chaptico Creek (CHP)	38.3817	76.7822
41	Coffee Hill (COF)	38.3614	76.7578
42	CR1D	38.5700	76.3833
43	Davis Millpond (DMP)	38.6708	75.7639
44	Dynards Run (DYN)	38.3164	76.7344
45	Dahlgren	38.3012	77.0660
46	Faulkners Branch - Bradley Rd. (FBB)	38.6989	75.7853

Station number	Station	Latitude	Longitude
47	Faulkners Branch - Ischer Rd. (FBI)	38.7214	75.8261
48	Forest Hall (FOR)	38.3989	76.7492
49	Kings Creek (KGC)	38.7897	76.0094
50	LPXT0173	39.1333	76.8183
51	Lyons Creek (LYC)	38.7689	76.6239
52	Mill Creek (MLC)	39.2825	76.1436
53	Mattawoman Creek (MTW)	38.6161	77.0486
54	Gibson Island	39.0600	76.4350
55	South Ferry	39.0767	76.5014
56	Frog Mortar	39.3083	76.4028
57	Wilson Point	39.3083	76.4125
58	North Davis Branch (NDB)	38.6783	75.7478
59	North River (NRV)	38.9878	76.6233
60	Bivalve	38.3214	75.8894
61	Sandy Hill Beach	38.3567	75.8558
62	Cherry Hill	38.5667	77.2583
63	Maryland	38.5167	77.2583
64	Mid	38.5222	77.2667
65	Virginia	38.4917	77.3083
66	Quantico	38.5278	77.2750
67	Widewater	38.4333	77.3250
68	PTXCF8747	38.3133	76.4222
69	PTXCF9575	38.3265	76.3713
70	PTXDE2792	38.3800	76.5150
71	PTXDE5339	38.4243	76.6008
72	PTXDE9401	38.4940	76.6645
73	PTXDF0407	38.3413	76.4858
74	PTXED4892	38.5828	76.6783
75	PTXED9490	38.6582	76.6845
76	PXT0402	38.7118	76.6858
77	PXT0494	38.8062	76.7075
78	PXT0603	38.9500	76.6950
79	PXT0809	39.1083	76.8617
80	PXT0972	39.2350	77.0583
81	Sewell Branch (SEW)	38.6083	76.5867
82	Betterton	39.3742	76.0503
83	Turners Creek	39.3631	75.9842
84	Junction Rt. 50	39.0056	76.5067
85	Annapolis	38.9669	76.4717
86	Tull Branch (TLB)	38.7194	75.7719
87	Twiford Meadow (TWM)	38.7236	75.7625
88	Trib. to Marshyhope Creek (UMH)	38.7631	75.7431
89	Grove	39.4000	76.0500
90	Howell	39.3583	76.0833
91	Spesutie	39.3917	76.1250
92	Delaware City	39.5417	75.7250
93	Chesapeake City	39.5167	75.8000
94	Courthouse Point	39.5000	75.8750
95	Elkton	39.5667	75.8500
96	Kentmore	39.3750	75.9583

Station number	Station	Latitude	Longitude
97	Havre de Grace	39.5417	76.0667
98	Trib. to Red Lion Branch (URL)	39.1767	75.8992
99	Trib. to Southeast Creek (USE)	39.1308	75.9794
100	Trib. to Tuckahoe Creek (UTK)	38.8831	75.9269
101	WBPXT0045	38.8085	76.7507
102	Quarter Creek	38.9167	76.1667
103	Lynnhaven River	36.8972	76.0886
104	Willoughby Bay	36.9528	76.2819
105	James River above Newport News (JRANN)	37.0103	76.5883
106	James R. below Newport News (JRBNN)	36.9758	76.4389
107	York River above Cheatham Annex (YRACA)	37.3050	76.6104
108	York R. below Cheatham Annex (YRBCA)	37.2833	76.5767
109	Pamunkey River below West Point (PRBWP)	37.5319	76.8036
110	Pamunkey R. above West Point (PRAWP)	37.5464	76.8122
111	Anacostia River (T800)	38.9322	76.9394
112	Anacostia River (T500)	38.9378	76.9419
113	Anacostia River (T1100)	38.9453	76.9406
114	Anacostia River (T120)	38.9350	76.9397
115	Anacostia River (01649500)	38.9603	76.9261
116	Anacostia River (01651000)	38.9525	76.9667

Figure 3. The zinc 90th percentiles determined for basins with at least 4 detected concentrations.

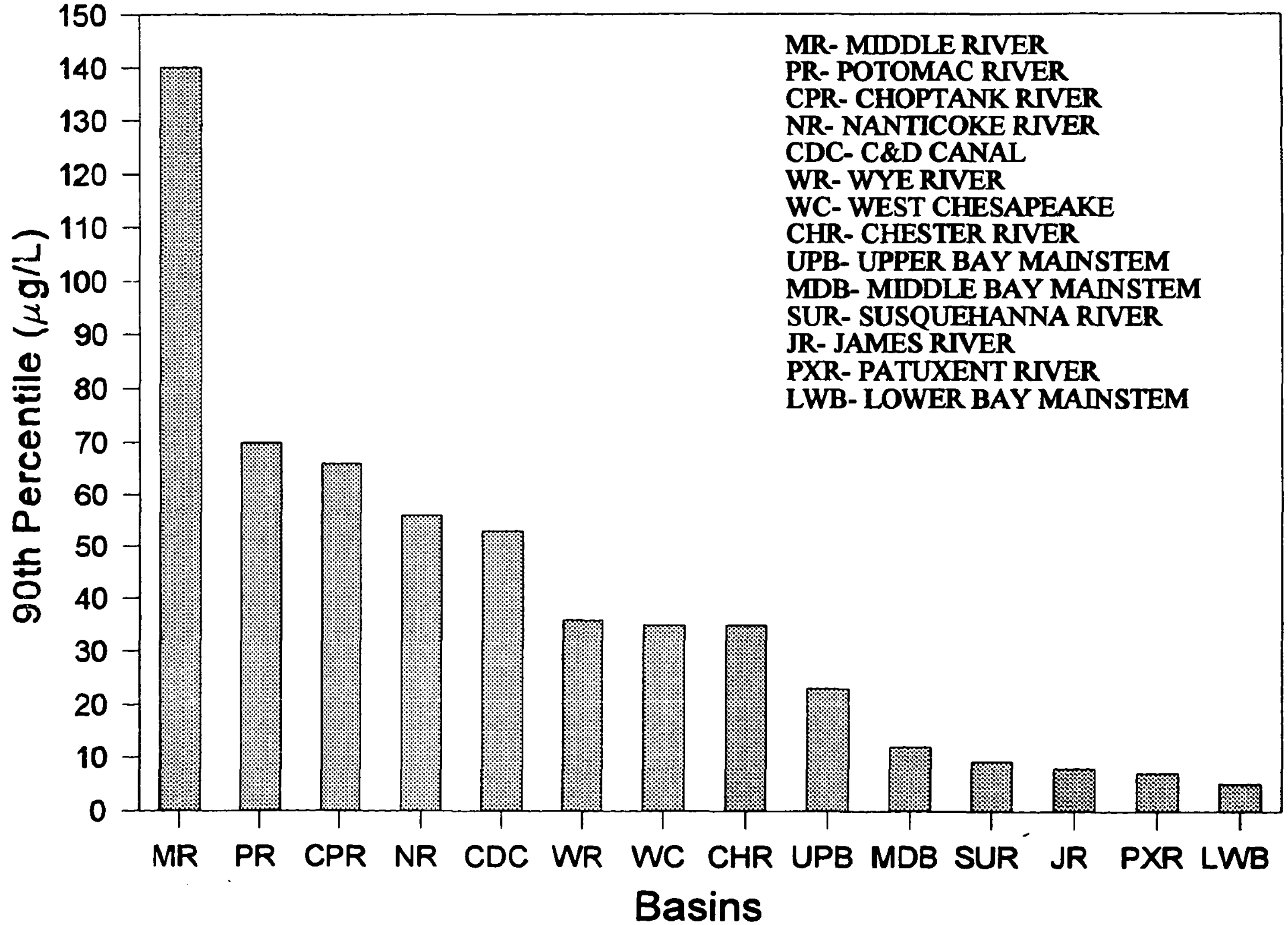


Figure 4. Seasonal pooled mean zinc concentrations and ranges (μg/L) from 15 stations during Patuxent River sampling (May 1995-February 1996).

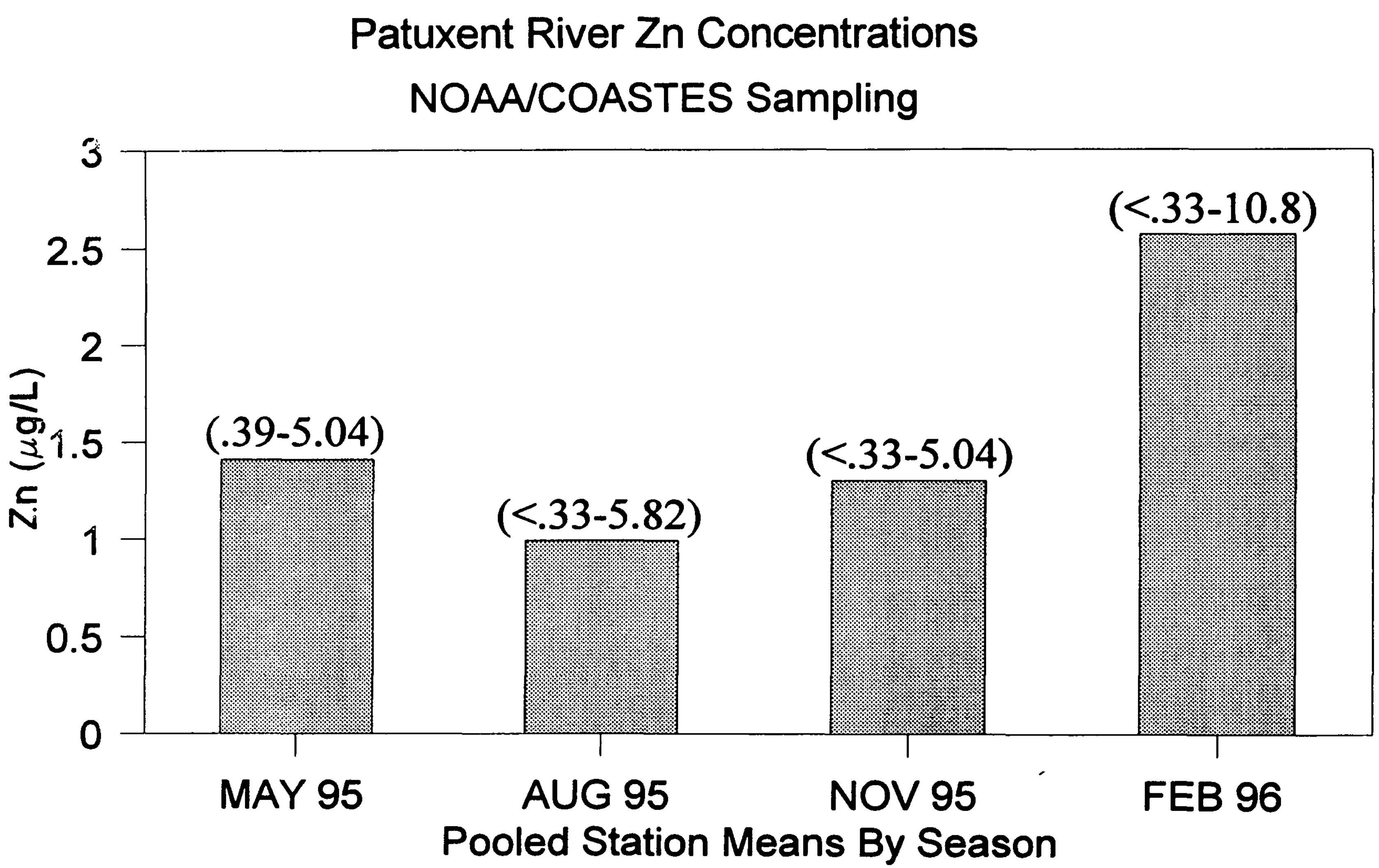


Figure 5. Zinc concentrations from the James River (1990 - 1993).

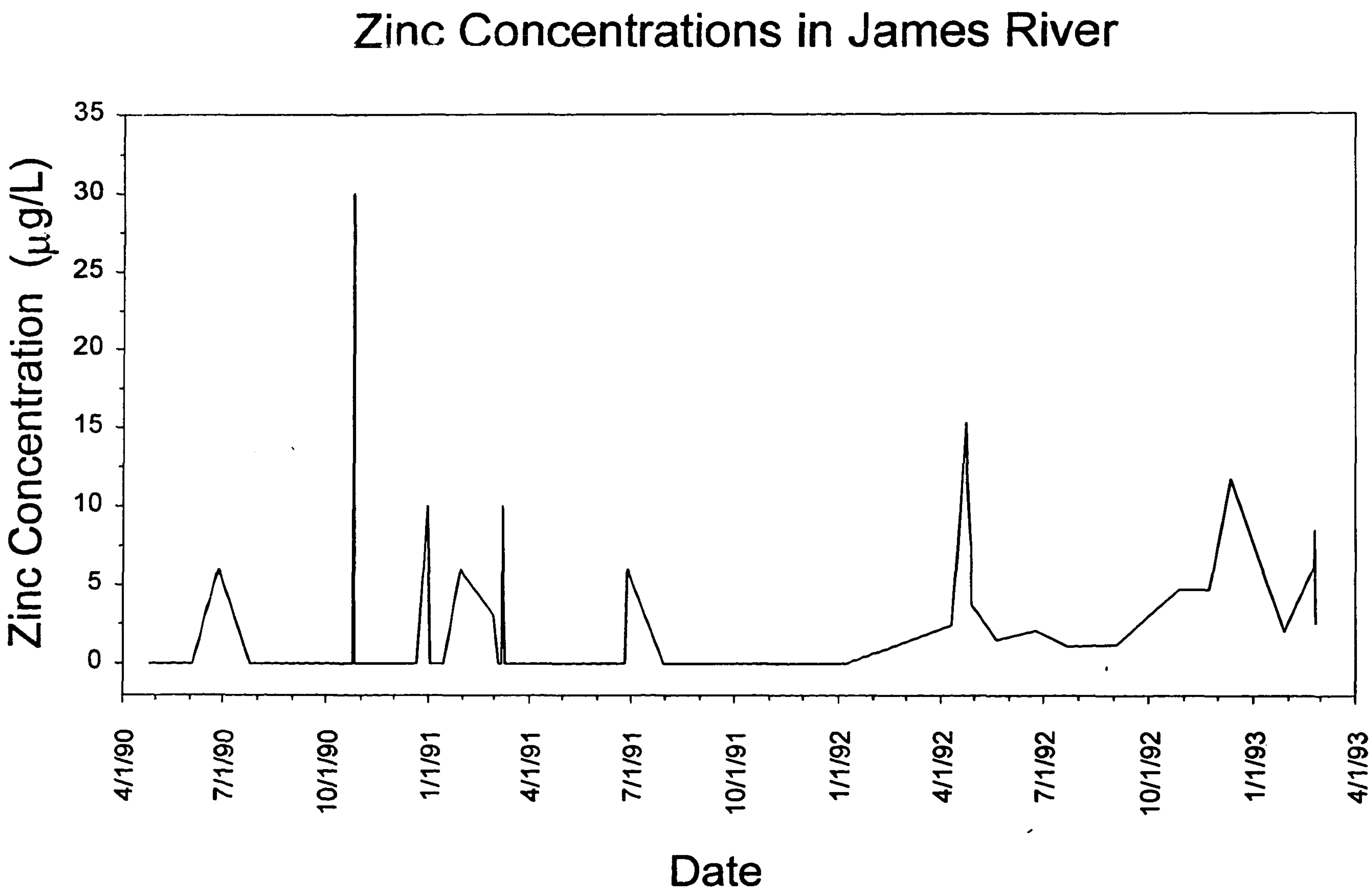


Figure 6. Zinc concentrations from the Susquehanna River (1990 -1993).

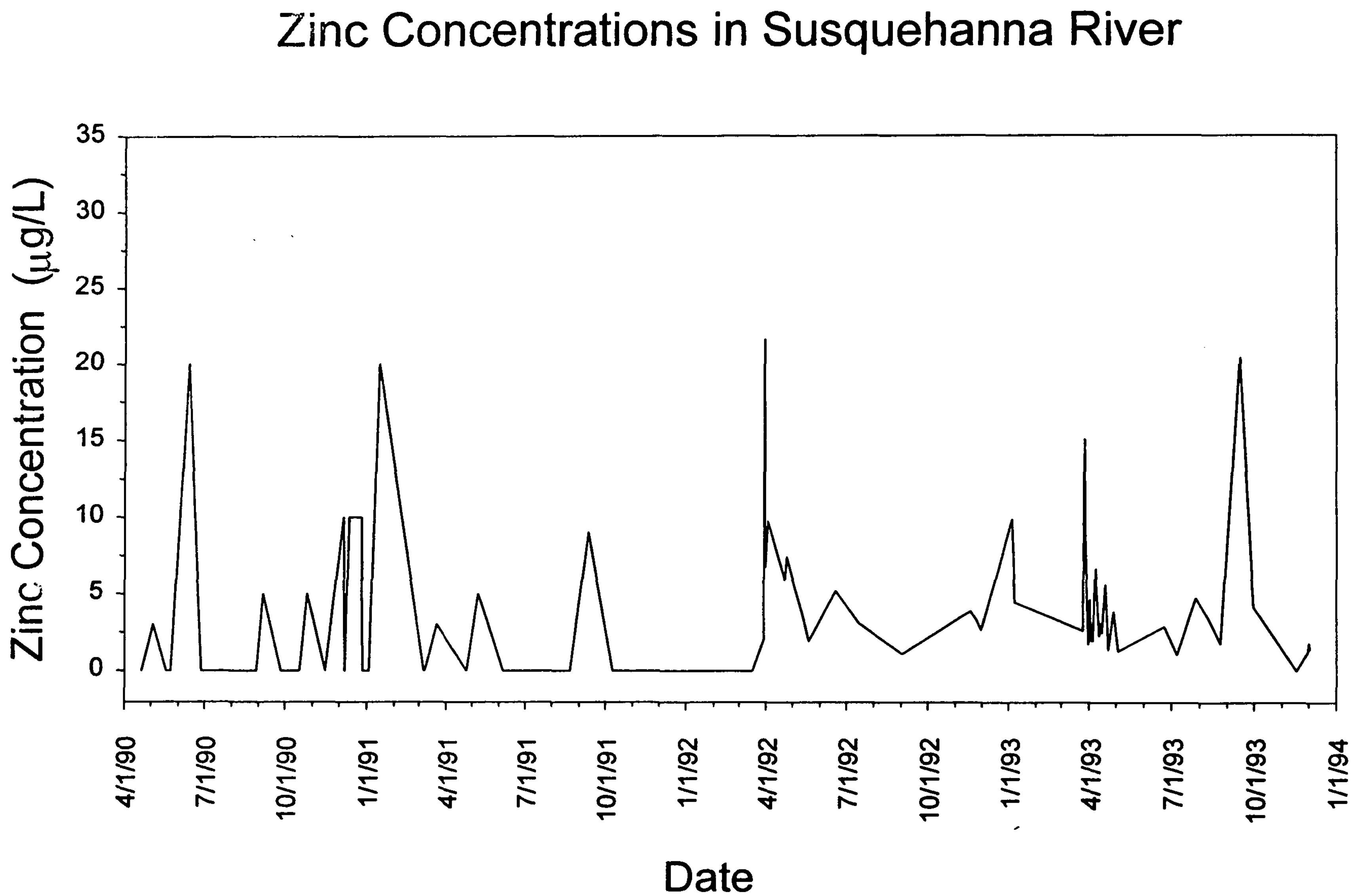


Figure 7. Distribution of acute zinc toxicity data (LC / EC50s) for freshwater species.

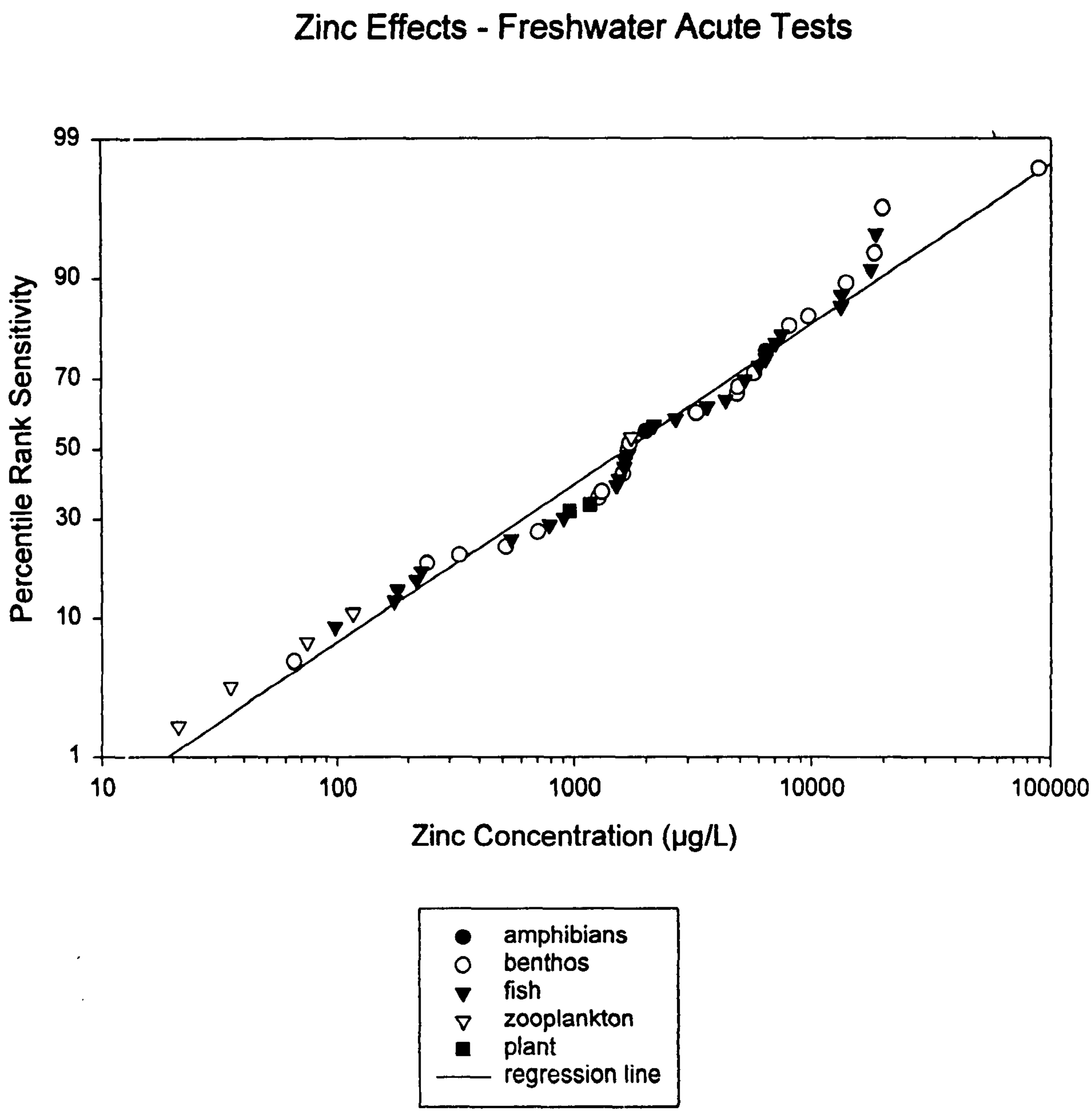


Figure 8. Distribution of acute zinc toxicity data (LC / EC50s) for saltwater species.

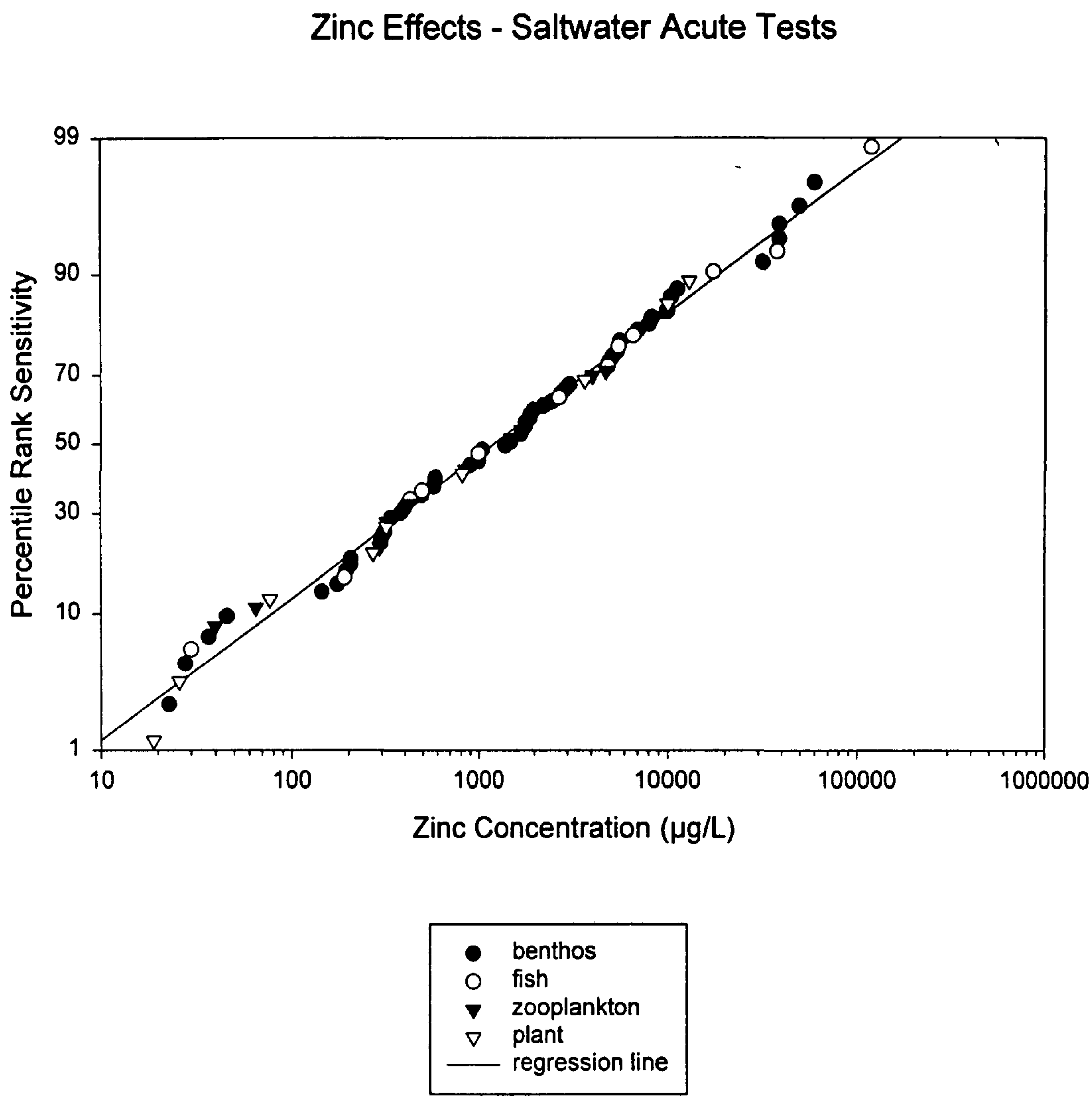


Figure 9. Distribution of chronic zinc toxicity data for freshwater species.

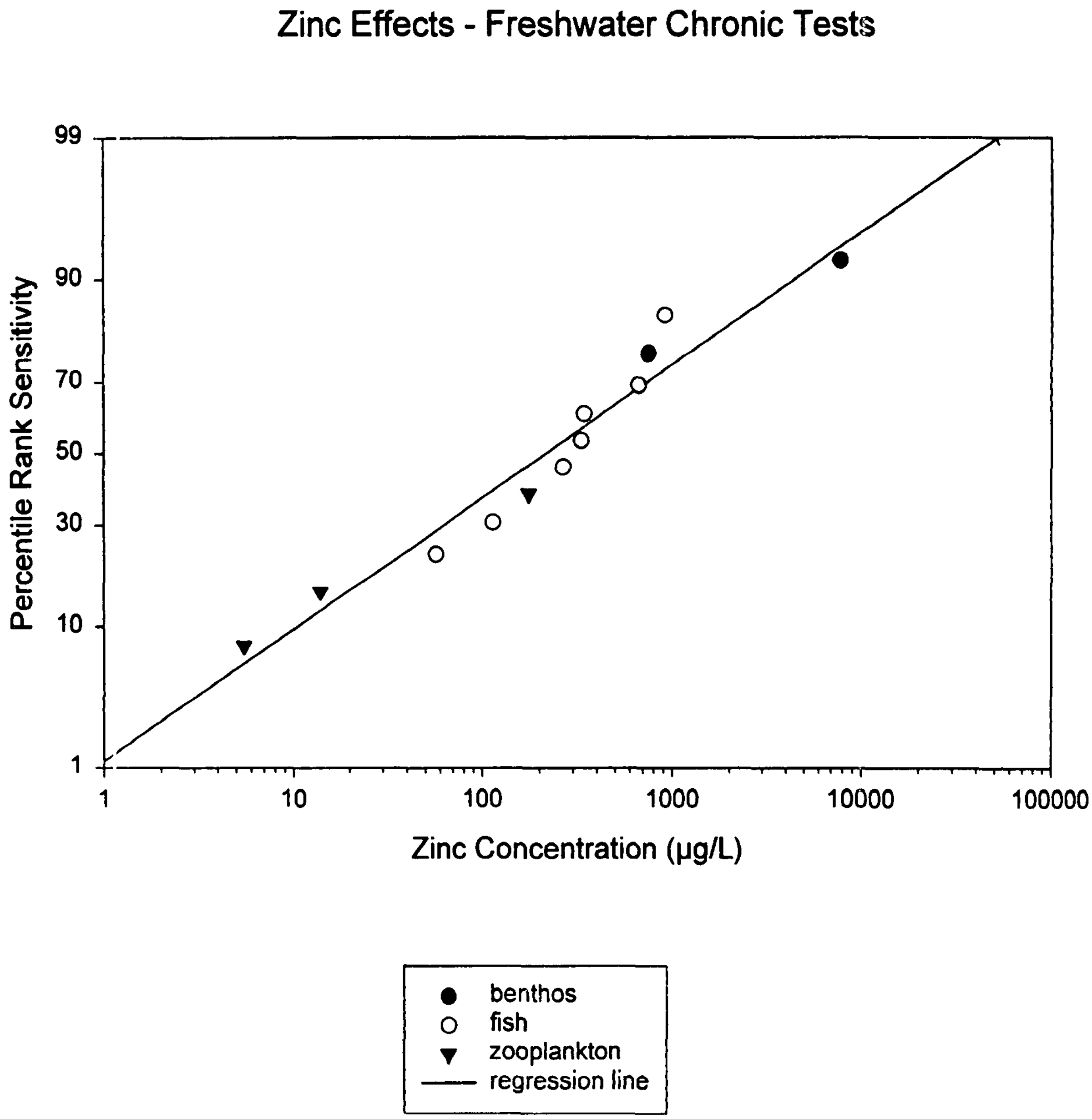
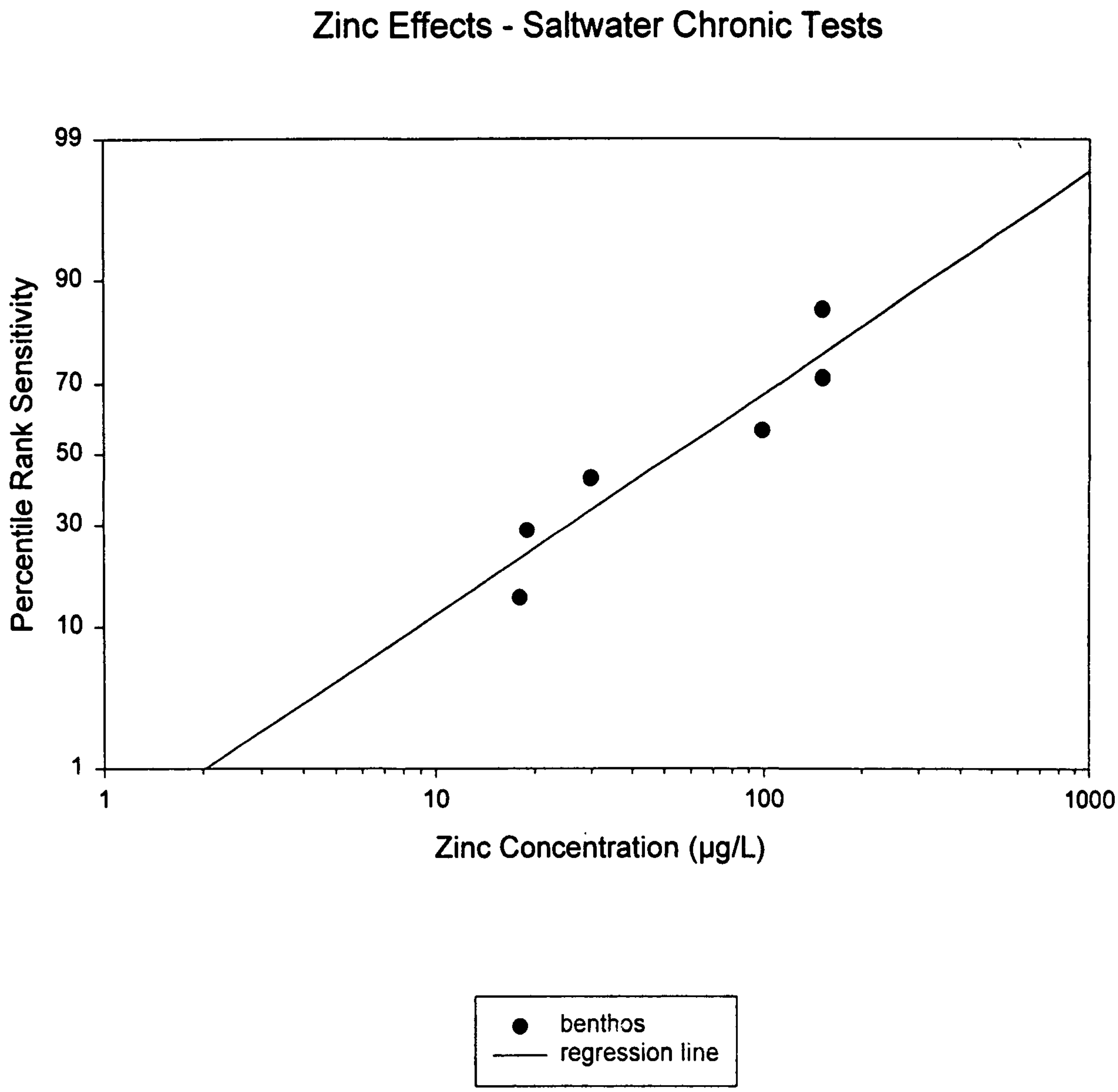


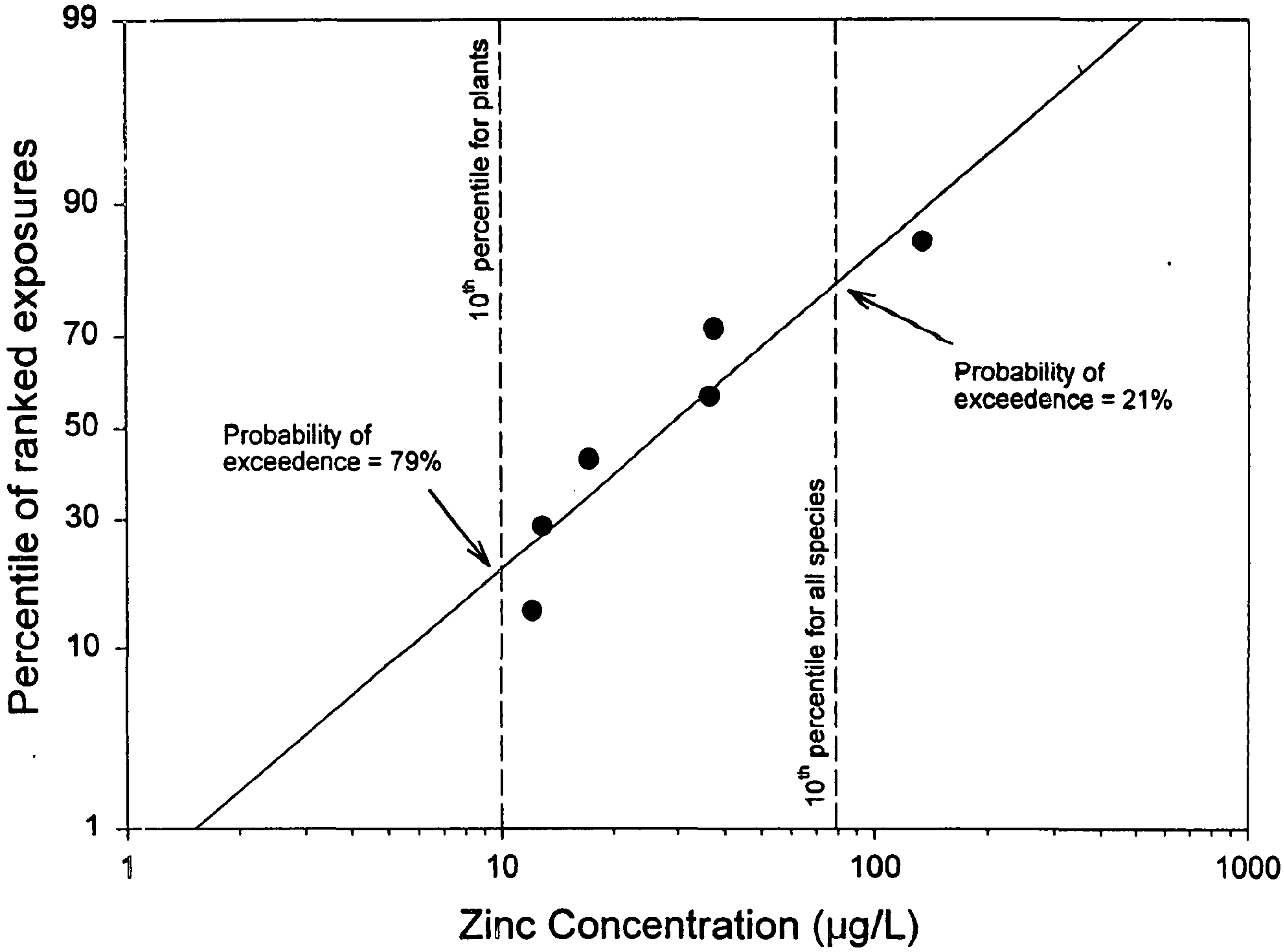
Figure 10. Distribution of chronic zinc toxicity data for saltwater species.

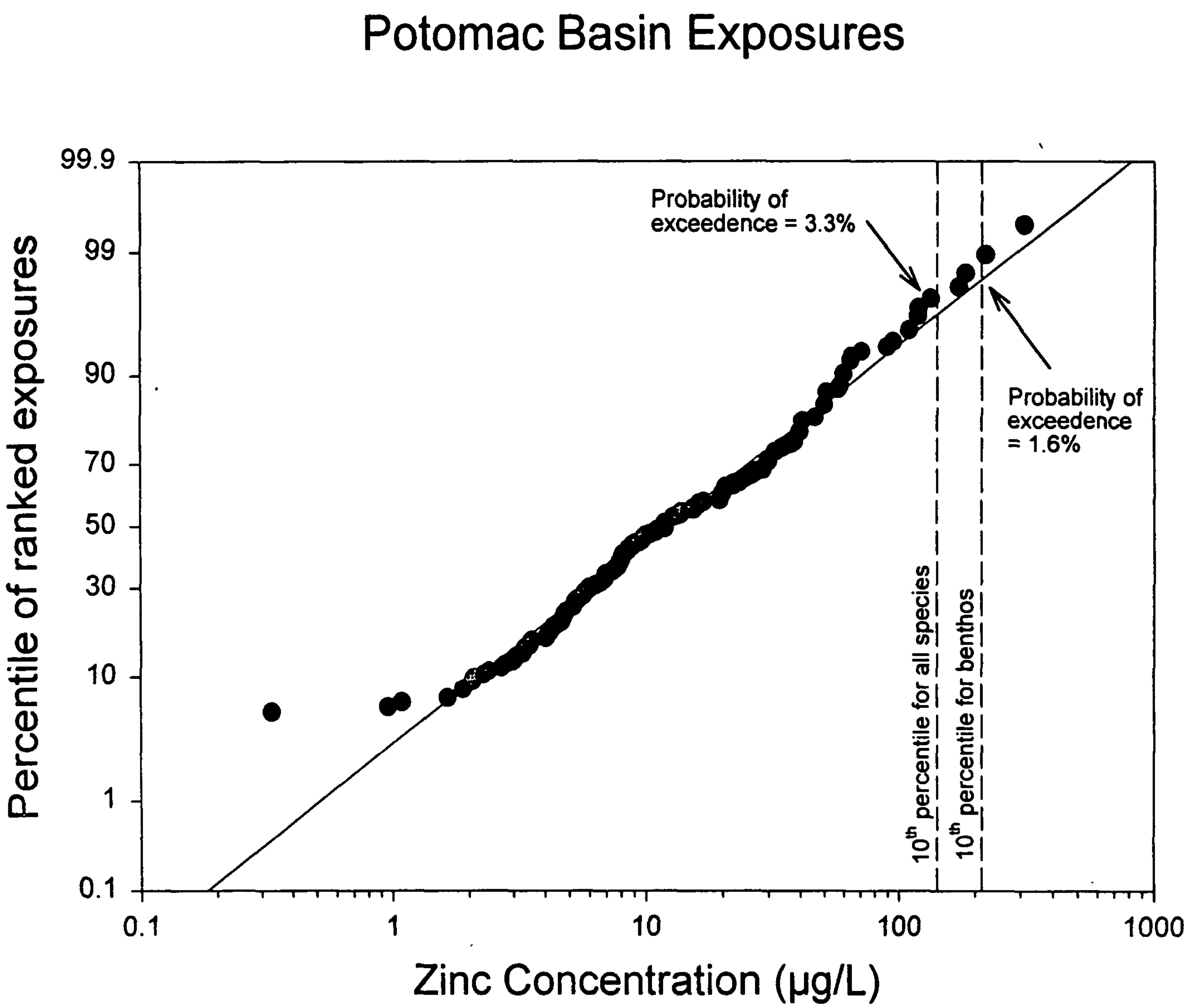


APPENDIX A

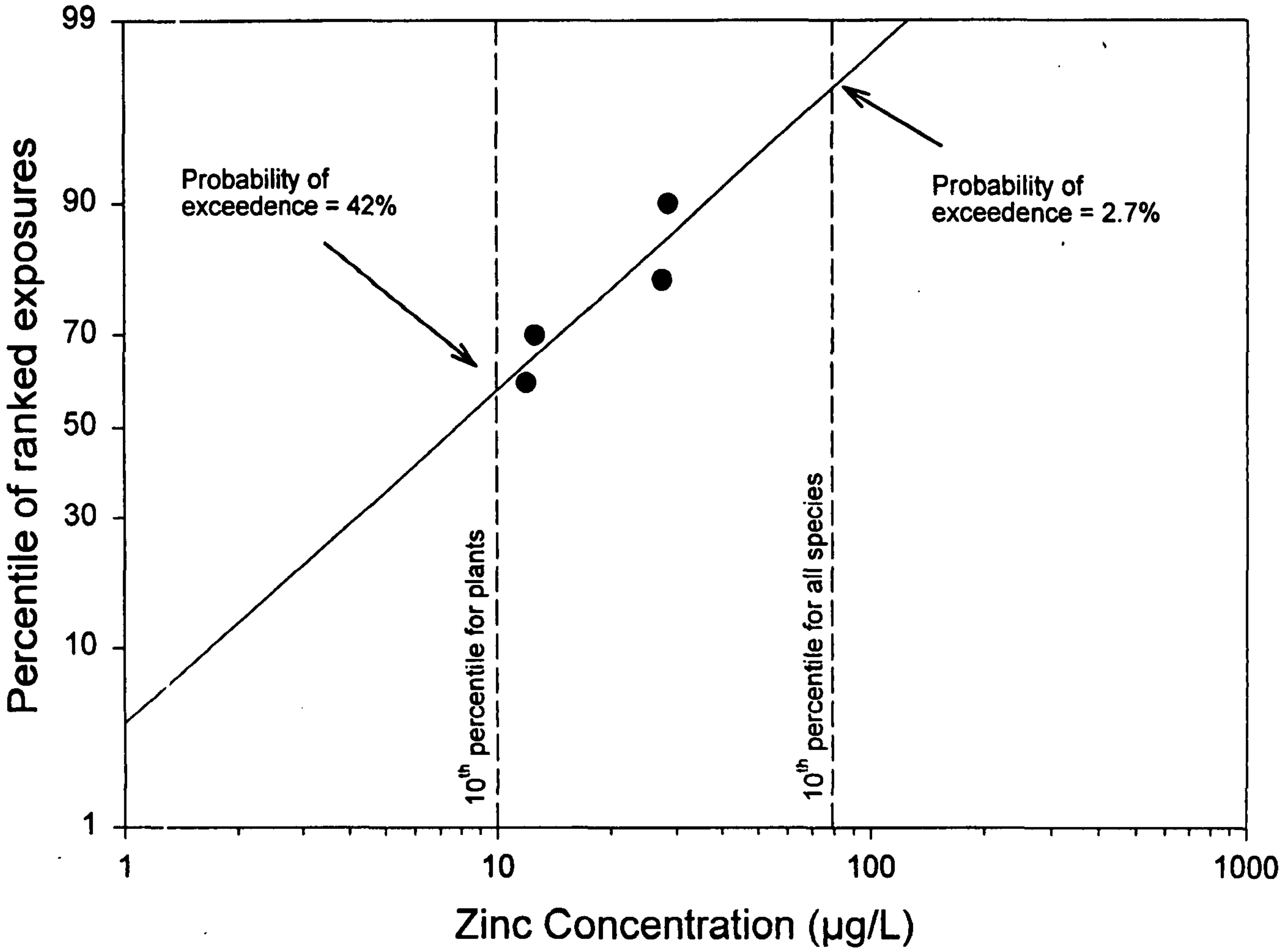
Zinc risk characterization by basin

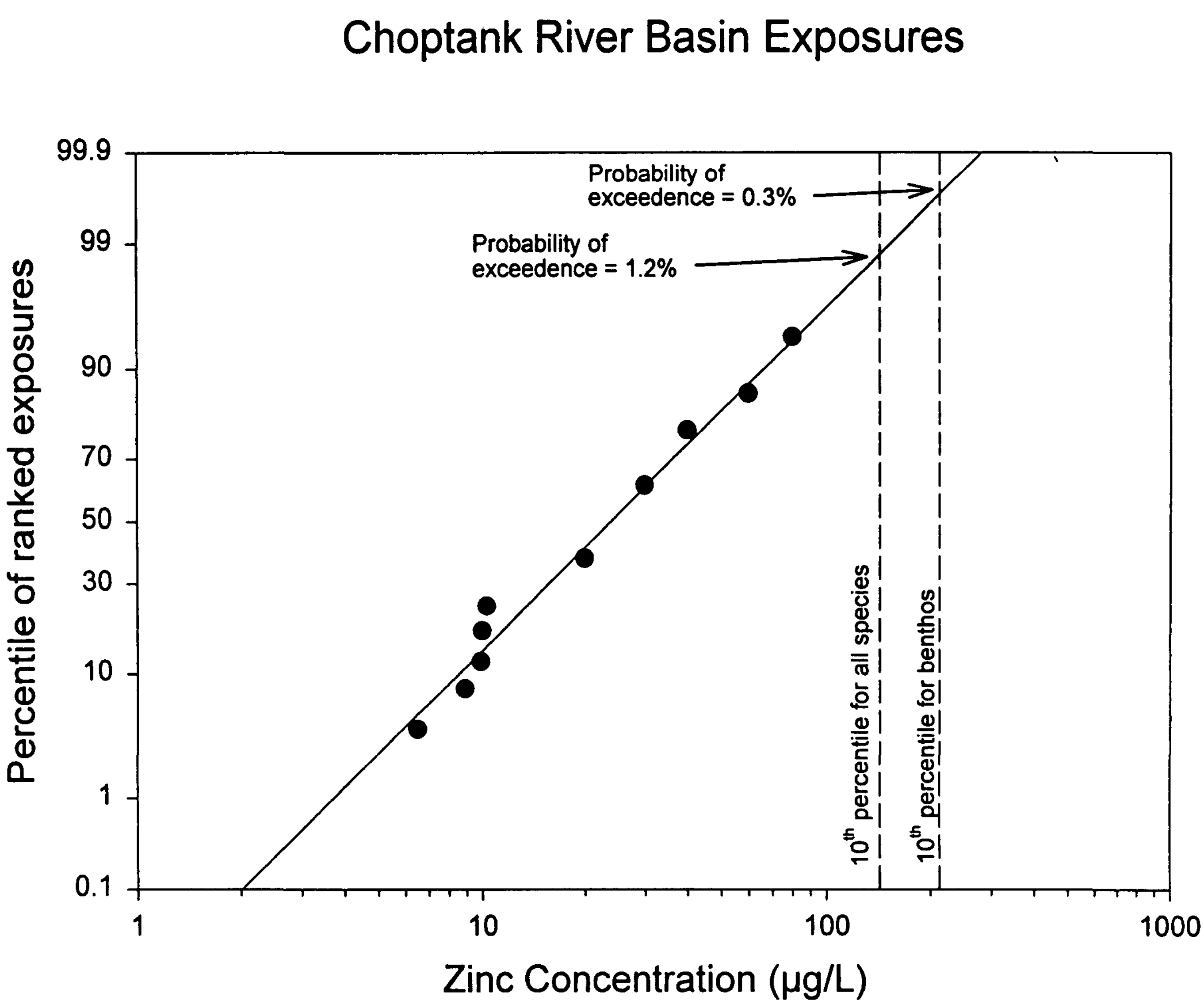
Middle River Basin Exposures

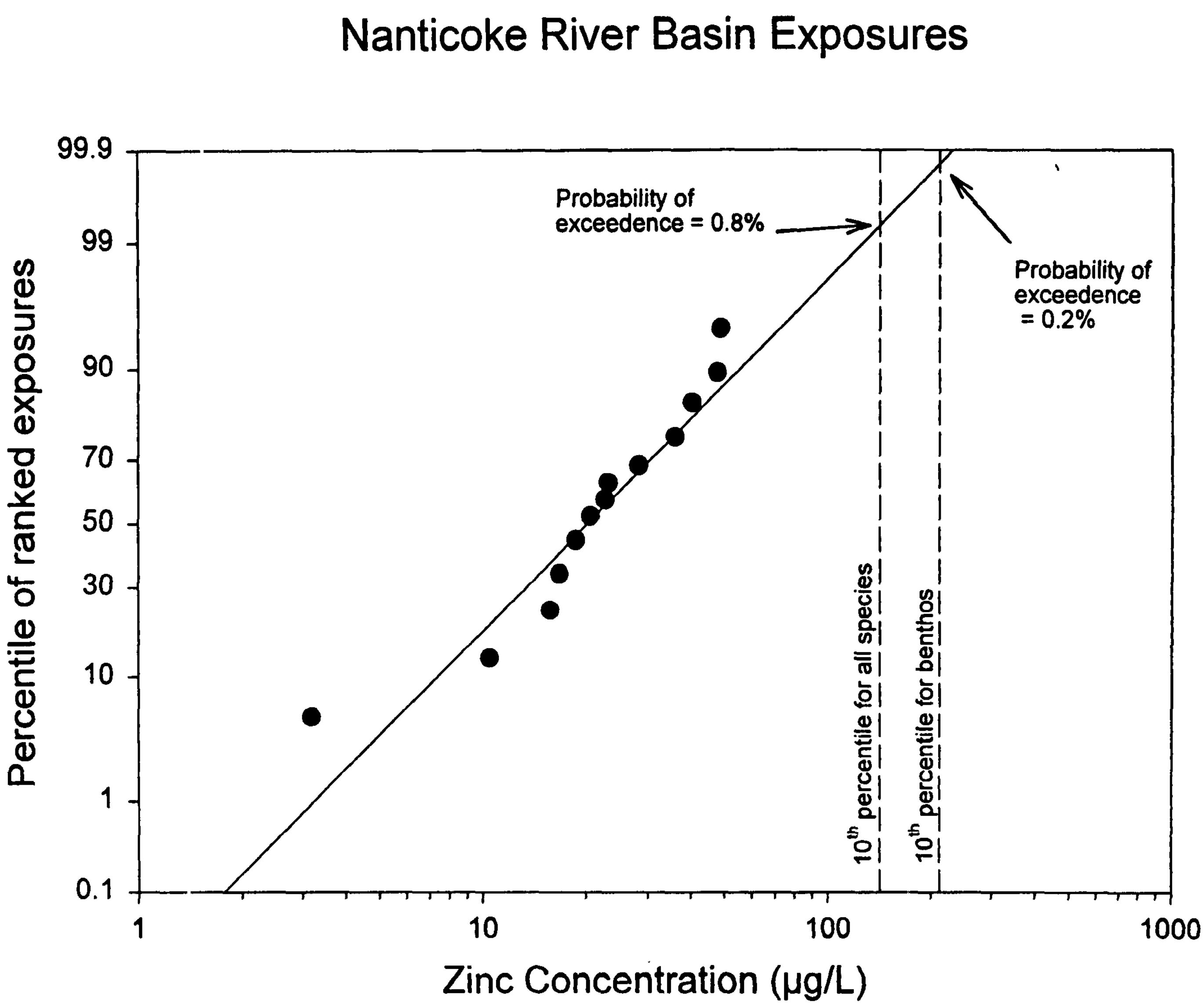



Potomac Basin Exposures
Percentile of ranked exposures
99.9
99
90
70
50
30
10
1
0.1
0.1
1
10
100
1000
Zinc Concentration (µg/L)
Probability of exceedence = 3.3%
Probability of exceedence = 1.6%
10th percentile for all species
10th percentile for benthos

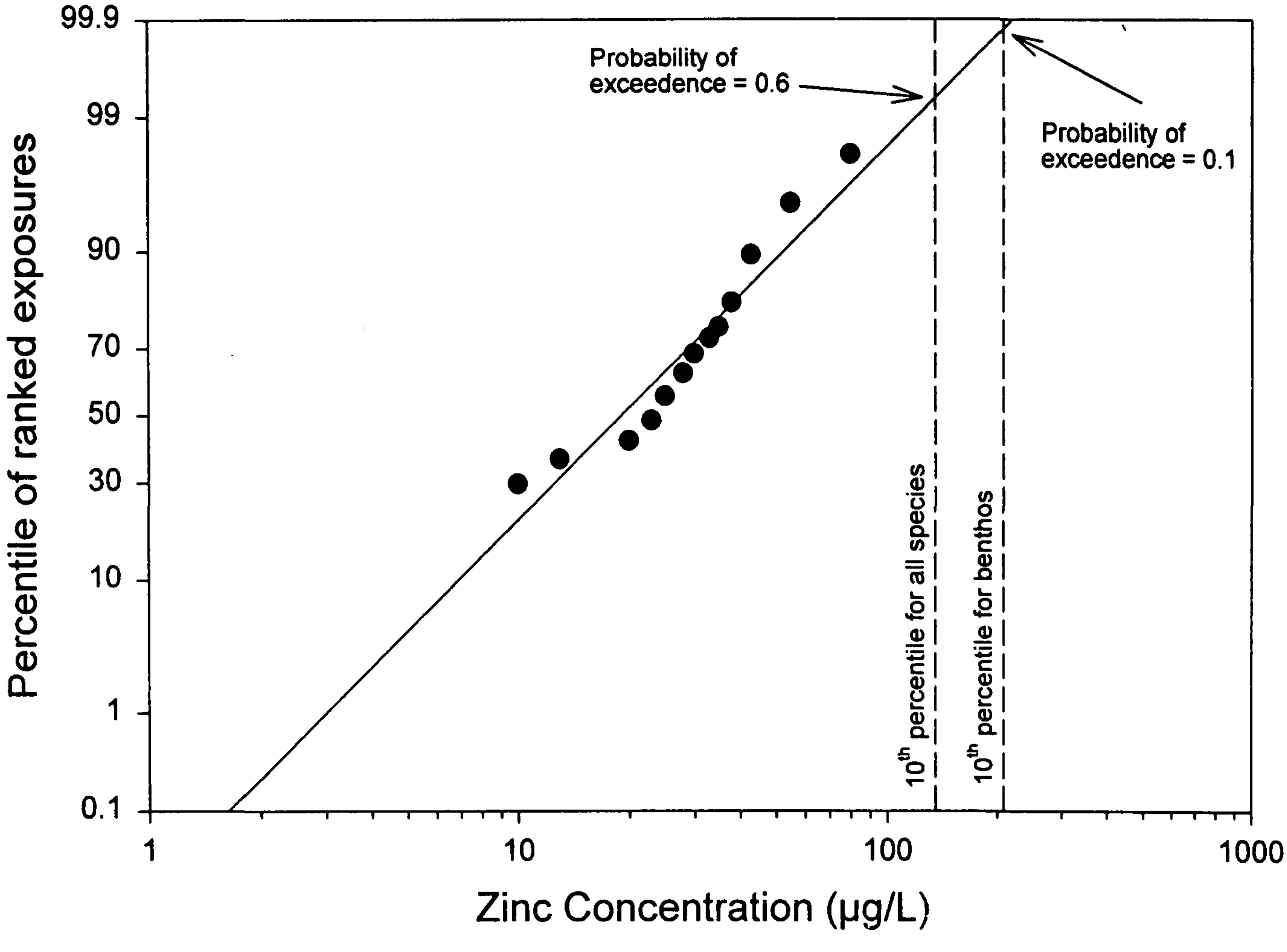
Wye River Basin Exposures

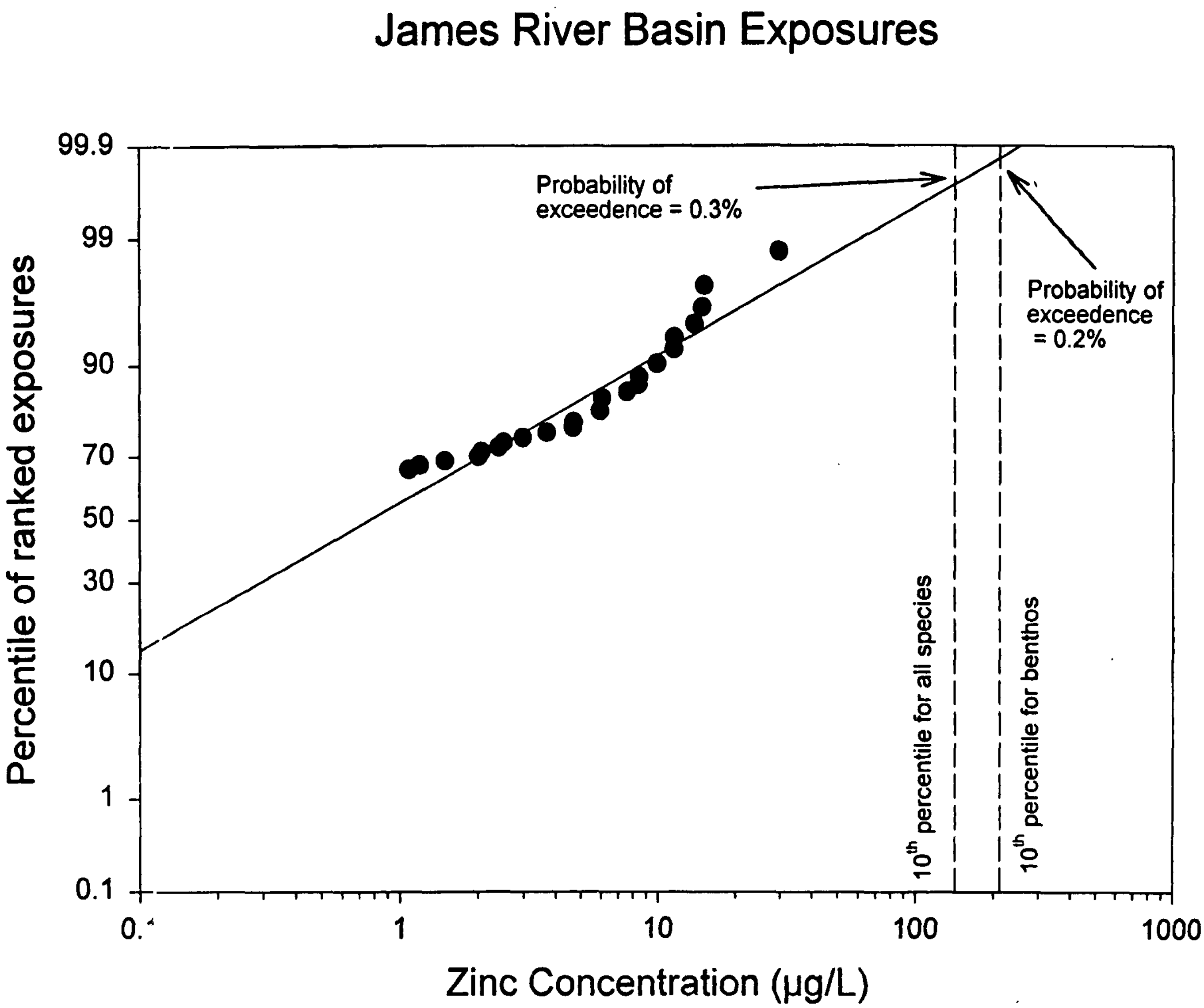



Choptank River Basin Exposures
Percentile of ranked exposures
99.9
99
90
70
50
30
10
1
0.1
Probability of exceedence = 0.3%
Probability of exceedence = 1.2%
10th percentile for all species
10th percentile for benthos
1
10
100
1000
Zinc Concentration (µg/L)

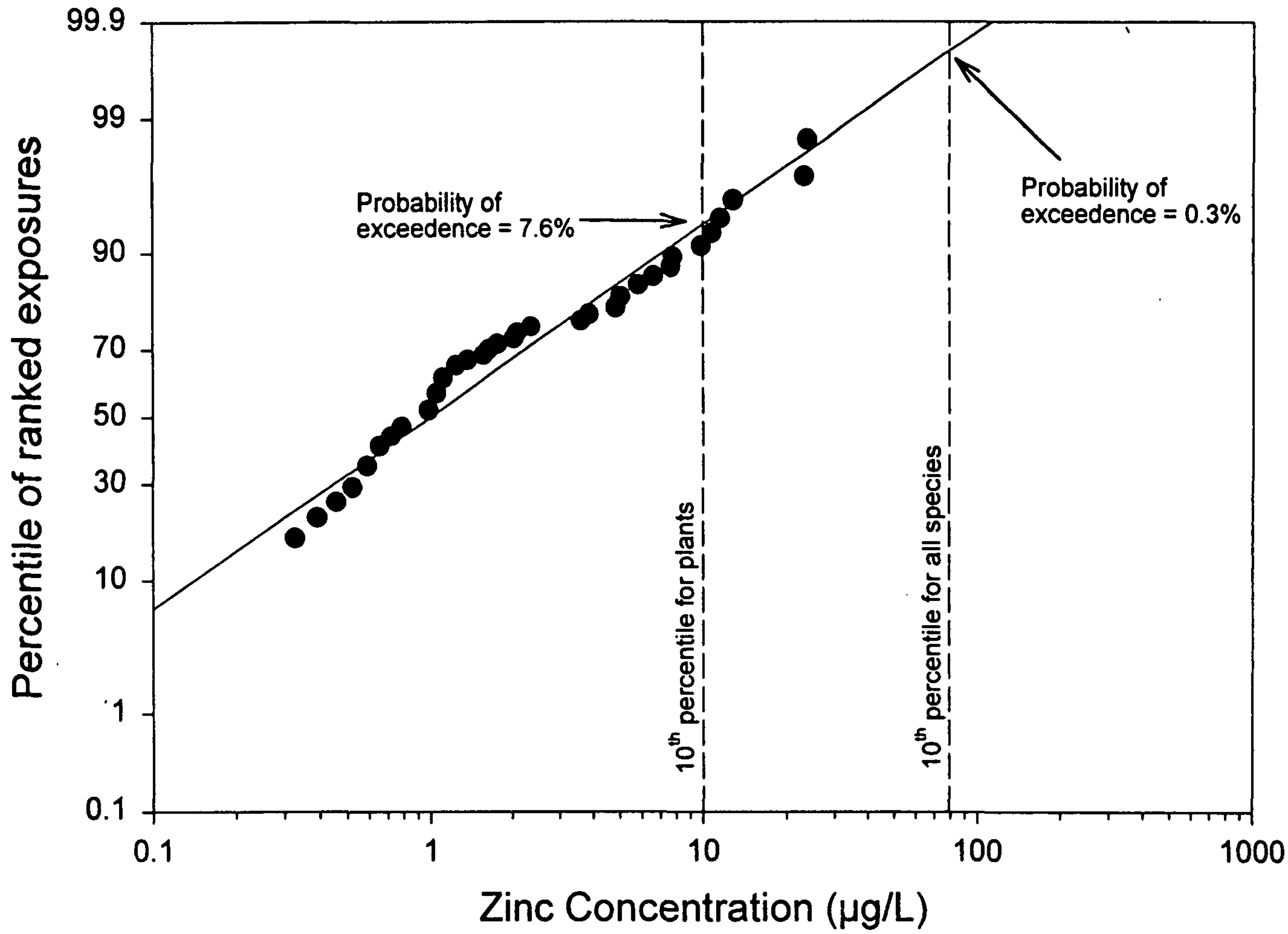

Nanticoke River Basin Exposures
Probability of exceedence = 0.8%
Probability of exceedence = 0.2%
10th percentile for all species
10th percentile for benthos
99.9
99
90
70
50
30
10
1
0.1
1
10
100
1000
Percentile of ranked exposures
Zinc Concentration (μg/L)

C&D Canal Exposures




James River Basin Exposures
Percentile of ranked exposures
99.9
99
90
70
50
30
10
1
0.1
Probability of
exceedence = 0.3%
Probability of
exceedence
= 0.2%
10th percentile for all species
10th percentile for benthos
1
10
100
1000
Zinc Concentration (µg/L)

Patuxent River Basin Exposures



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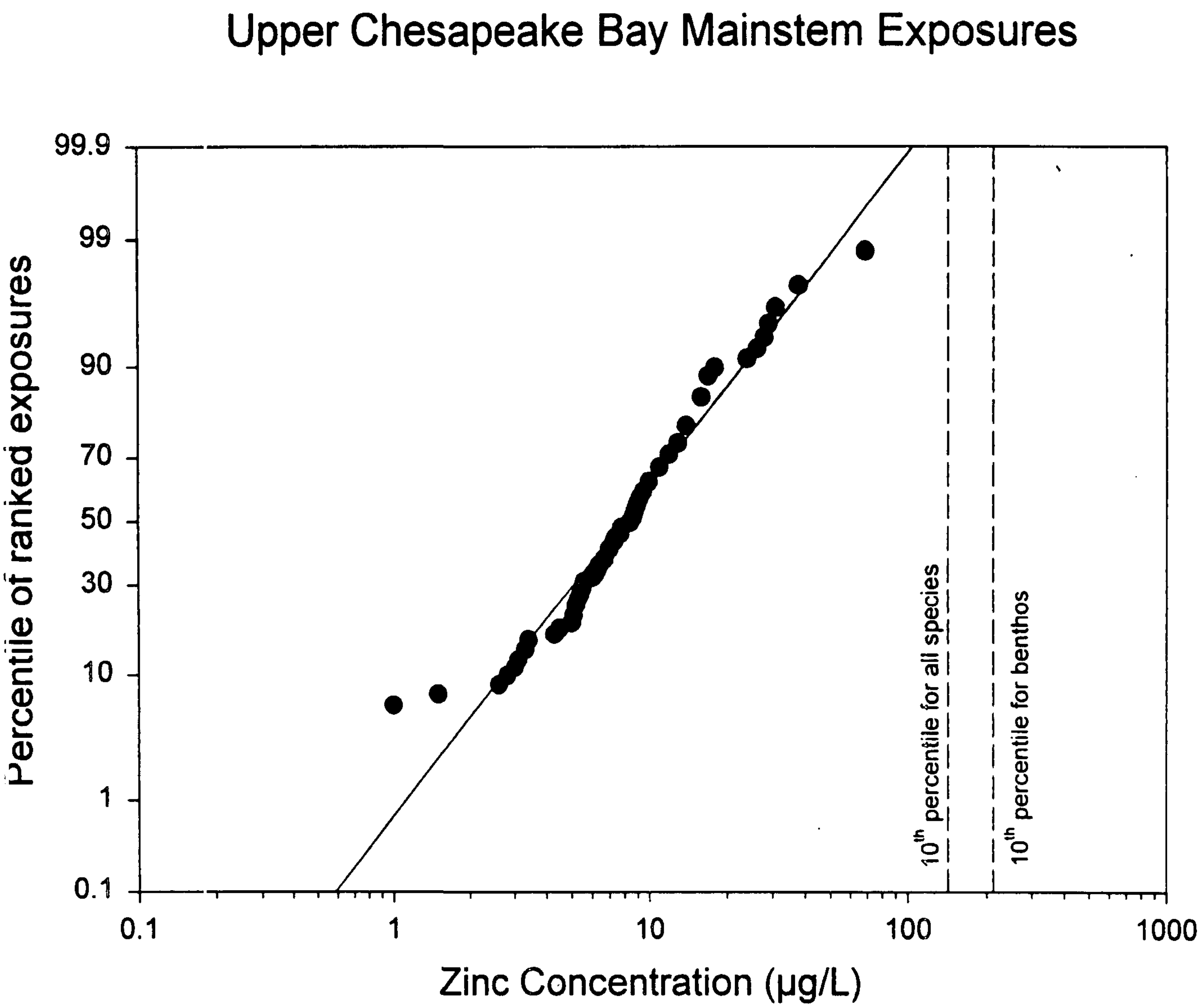
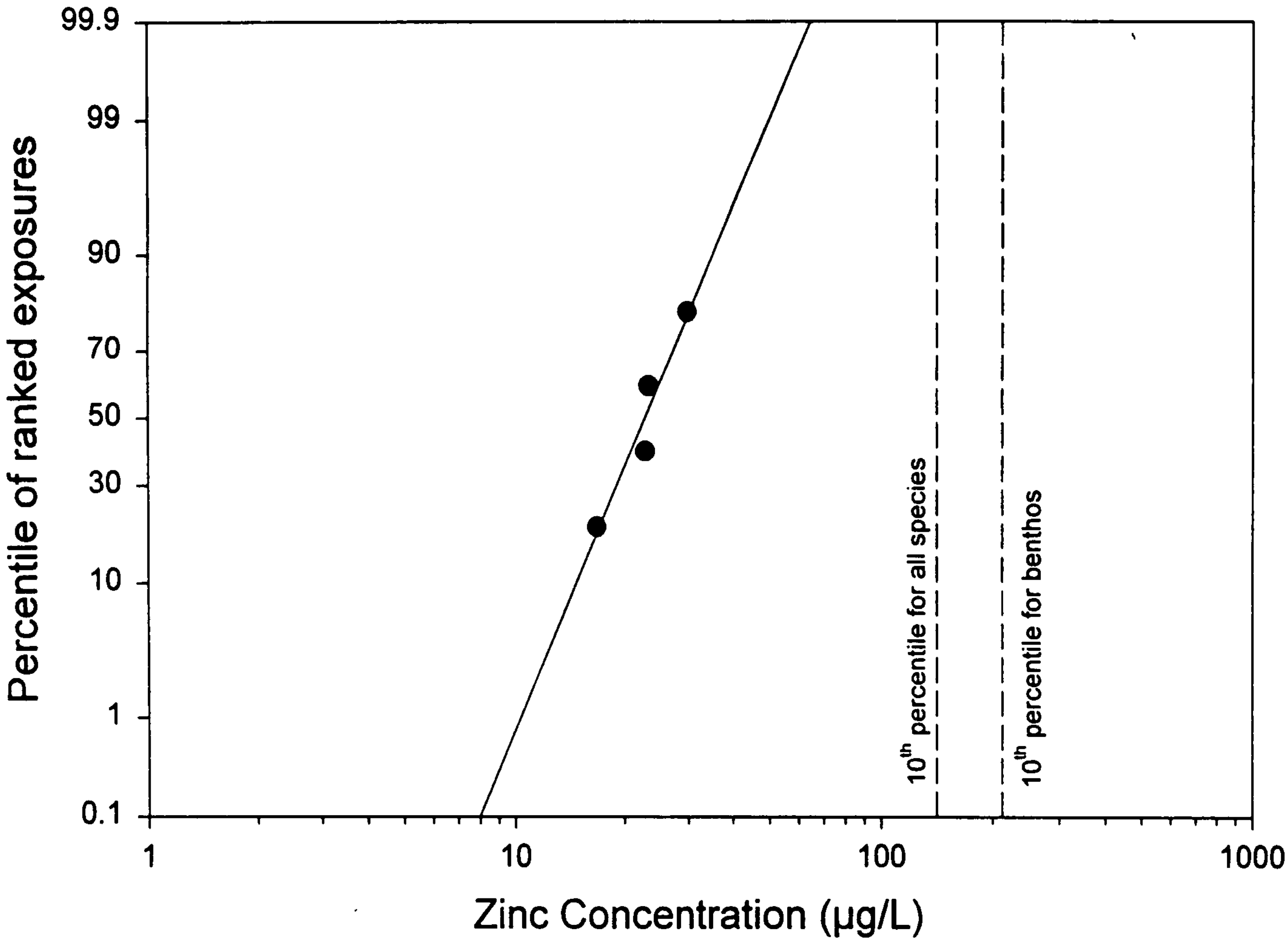
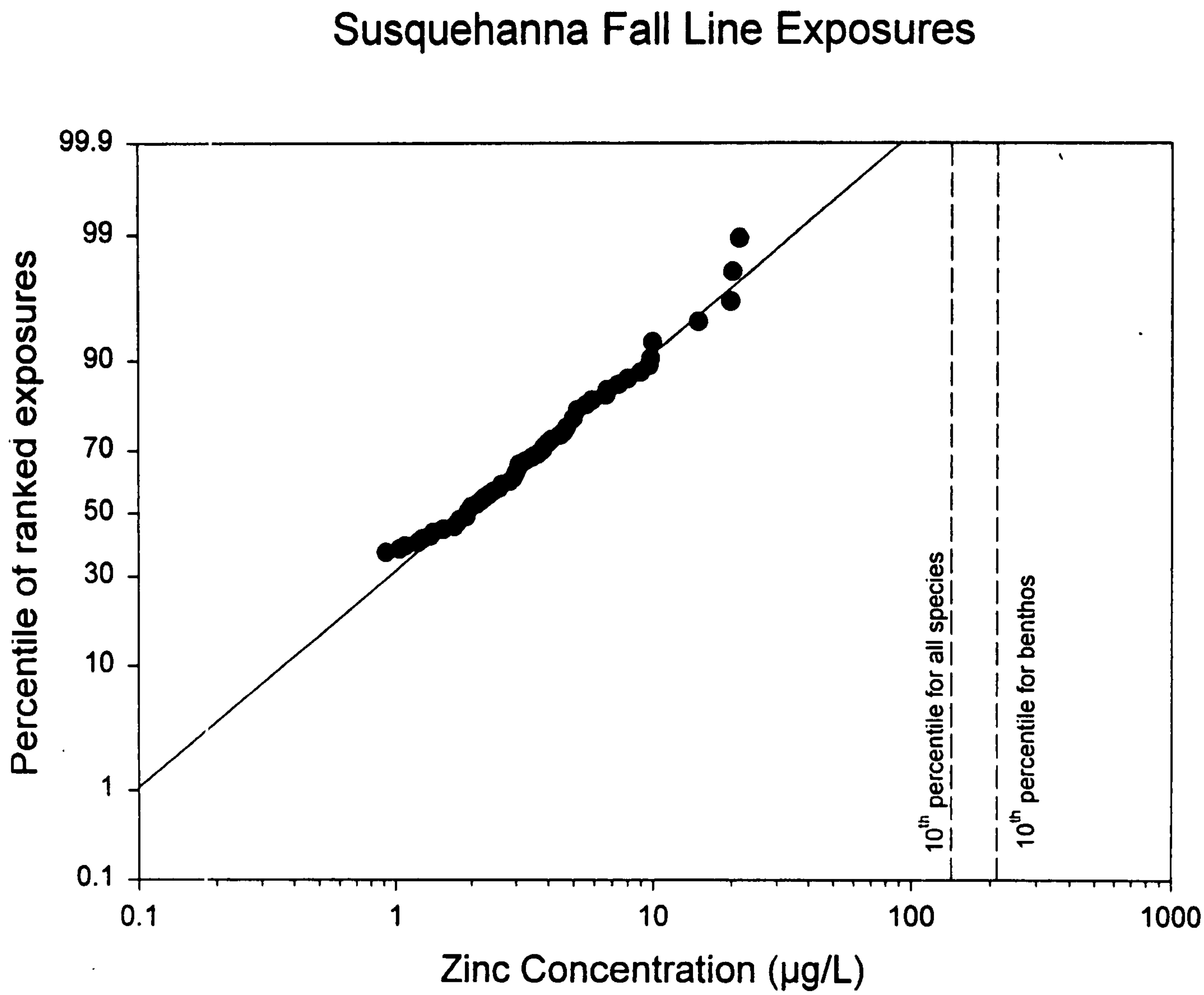


Upper Chesapeake Bay Mainstem Exposures
Percentile of ranked exposures
99.9
99
90
70
50
30
10
1
0.1
0.1
1
10
100
1000
Zinc Concentration (µg/L)
10th percentile for all species
10th percentile for benthos

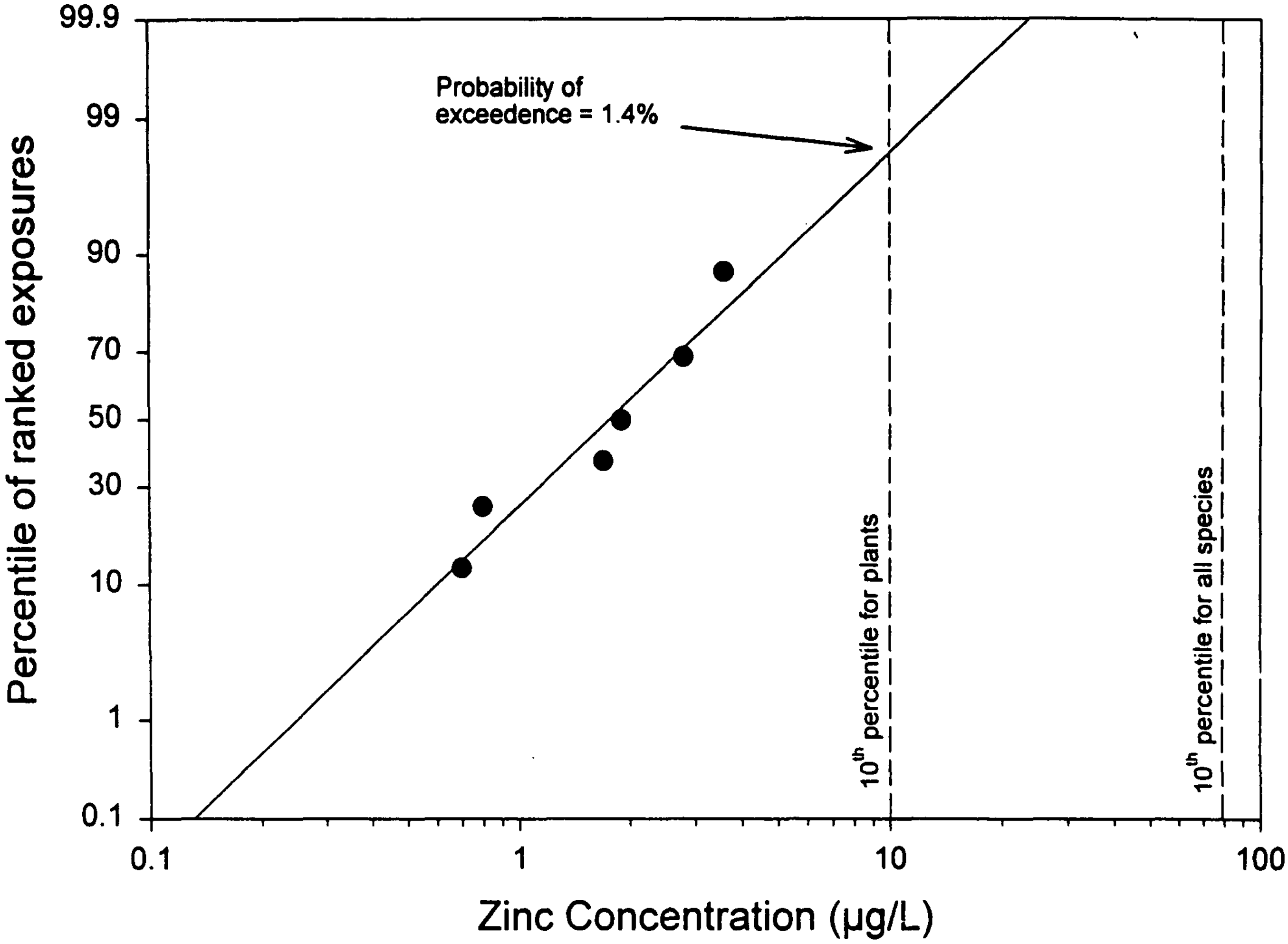
Chester River Basin Exposures

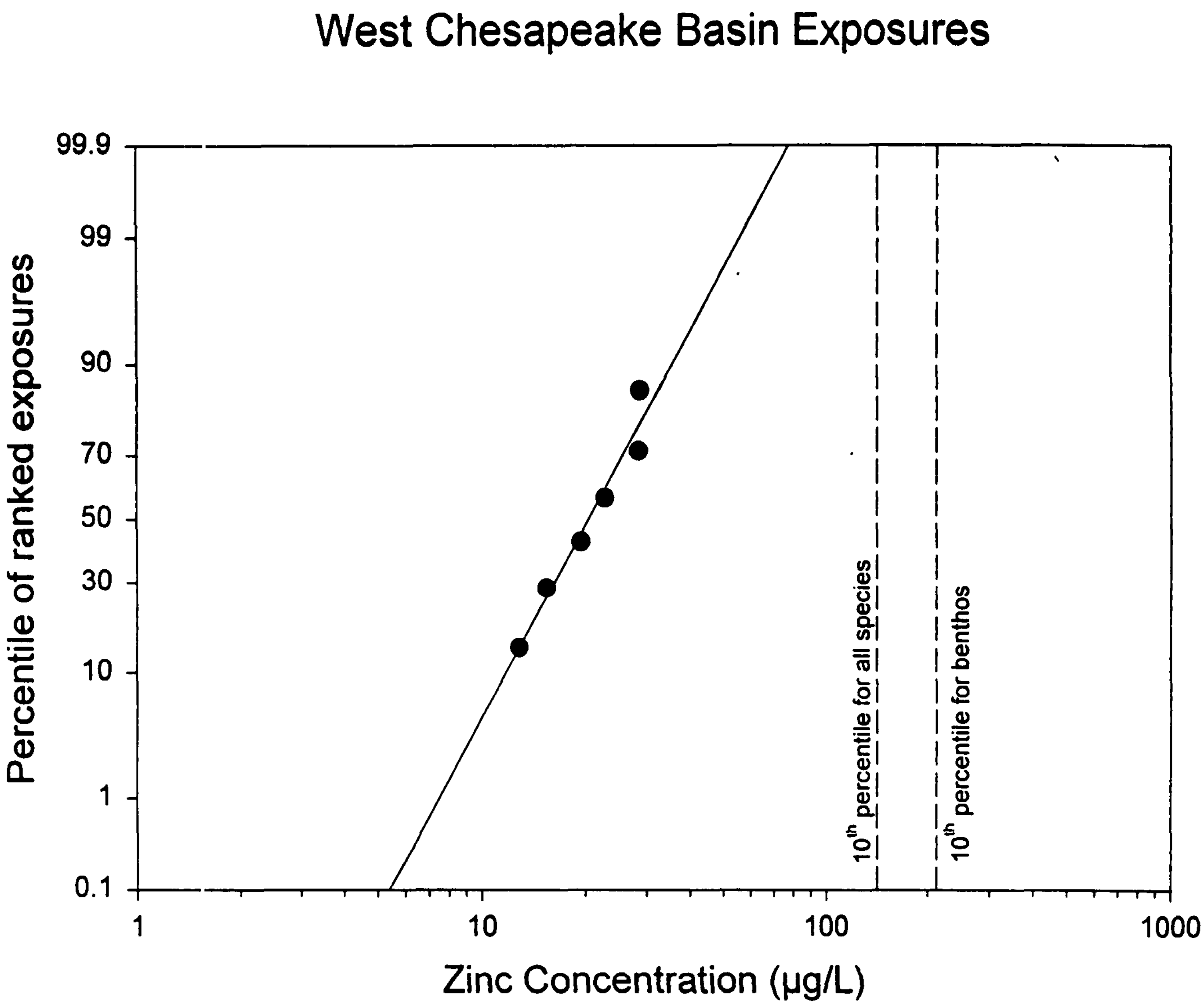




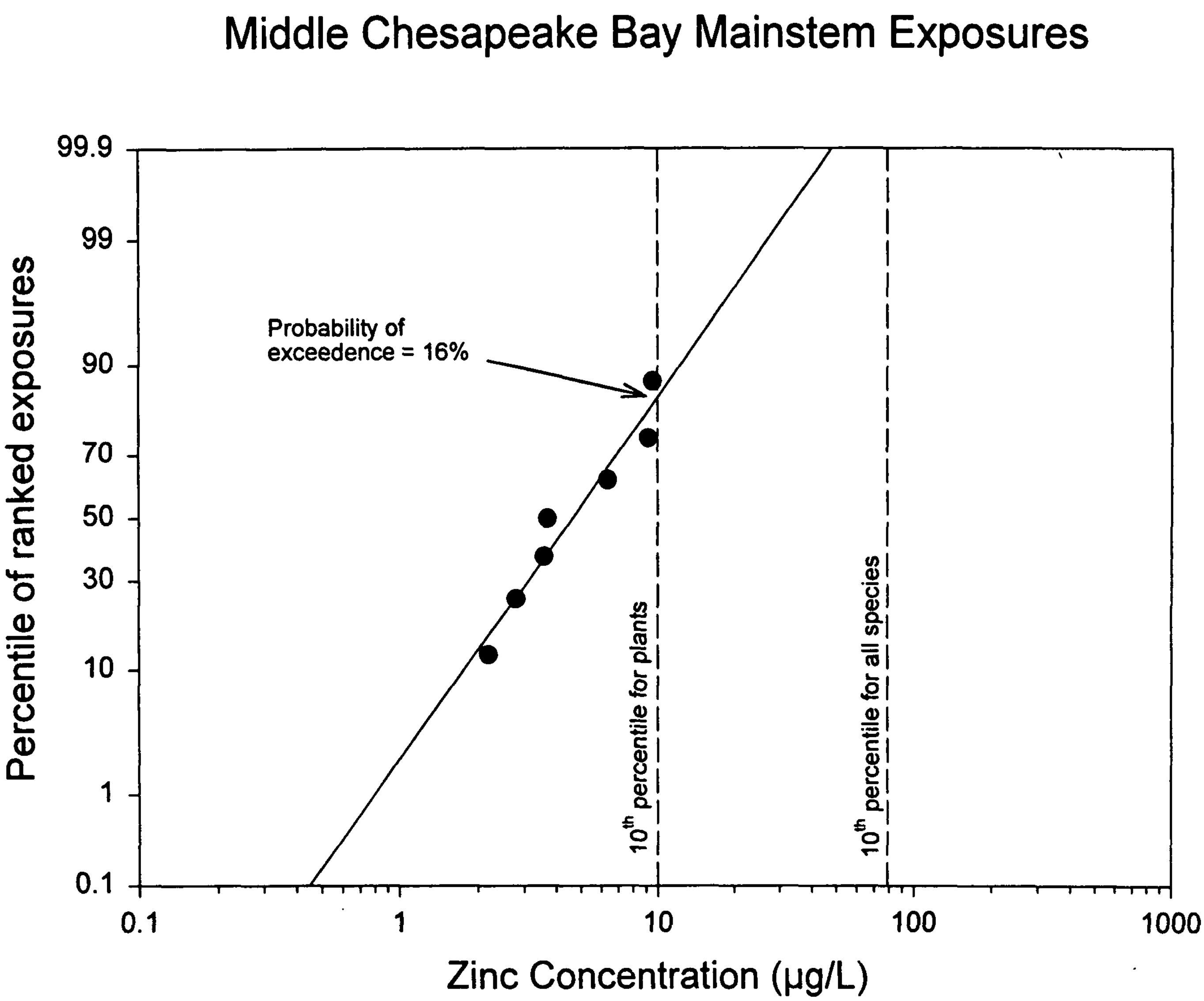
Susquehanna Fall Line Exposures
99.9
99
90
70
50
30
10
1
0.1
Percentile of ranked exposures
0.1
1
10
100
1000
Zinc Concentration (µg/L)
10th percentile for all species
10th percentile for benthos

Lower Chesapeake Bay Mainstem Exposures





West Chesapeake Basin Exposures
Percentile of ranked exposures
99.9
99
90
70
50
30
10
1
0.1
1
10
100
1000
Zinc Concentration (µg/L)
10th percentile for all species
10th percentile for benthos


Middle Chesapeake Bay Mainstem Exposures
Percentile of ranked exposures
99.9
99
90
70
50
30
10
1
0.1
Probability of exceedence = 16%
10th percentile for plants
10th percentile for all species
0.1
1
10
100
1000
Zinc Concentration (µg/L)