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BASELINE ECOLOGICAL RISK ASSESSMENT NEW BEDFORD HARBOR SITE FEASIBILITY STUDY

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NEW BEDFORD HARBOR, MASSACHUSETTS
ECOLOGICAL RISK ASSESSMENT

TABLE OF CONTENTS

NEW BEDFORD HARBOR, MASSACHUSETTS
ECOLOGICAL RISK ASSESSMENT

TABLE OF CONTENTS

NEW BEDFORD HARBOR, MASSACHUSETTS
ECOLOGICAL RISK ASSESSMENT

TABLE OF CONTENTS

NEW BEDFORD HARBOR, MASSACHUSETTS
ECOLOGICAL RISK ASSESSMENT

LIST OF FIGURES

NEW BEDFORD HARBOR, MASSACHUSETTS
ECOLOGICAL RISK ASSESSMENT

LIST OF FIGURES

NEW BEDFORD HARBOR, MASSACHUSETTS
ECOLOGICAL RISK ASSESSMENT

LIST OF TABLES

EXECUTIVE SUMMARY

New Bedford Harbor is a tidal estuary on the western shore of Buzzards Bay, Massachusetts. Studies of the harbor conducted in the mid-1970s indicated widespread polychlorinated biphenyl (PCB) and heavy metals contamination. Large areas of the harbor were subsequently closed to fishing to reduce the potential for human exposure to PCBs. The New Bedford Harbor site was added to the U.S. Environmental Protection Agency (EPA) Interim National Priorities List in July 1982; shortly thereafter, EPA initiated a more comprehensive assessment of the extent of the PCB contamination problem. These and other studies have confirmed extensive PCB contamination of water, sediments, and biota in the harbor, with sediment concentrations reported in excess of 100,000 parts per million (ppm) in the area of maximum contamination. Concentrations in biota in many areas exceed the U.S. Food and Drug Administration tolerance level of 2 ppm.

Under authority of the Comprehensive Environmental Response, Compensation, and Liability Act (or Superfund), EPA is responsible for conducting a Remedial Investigation and Feasibility Study (RI/FS) to support the need for and extent of remediation in New Bedford Harbor. This baseline ecological risk assessment, as part of the RI/FS process, presents and quantifies risks to aquatic organisms due to exposure to PCBs and heavy metals in New Bedford Harbor. Based on current conditions in the harbor, it will serve as a benchmark against which the effectiveness of various remedial options may be evaluated.

The ecological risk assessment is based on data collected by several investigations, but draws most heavily on information generated by Battelle (Battelle Pacific Northwest Laboratories, Richland, Washington; and Battelle Ocean Sciences, Duxbury, Massachusetts) in conjunction with the development of a numerical hydrodynamic/sediment-transport model of the harbor. Risk to aquatic biota was evaluated using a joint probability analysis in which two probability distributions, one representing contaminant levels in various zones of the harbor and the second representing the sensitivity of biota to contaminants, were combined to present a comprehensive probabilistic evaluation of risk. The joint probability analysis was supplemented by comparison of PCB levels in the harbor to EPA water quality criteria, evaluation of site-specific toxicity tests, and examination of data on the structure of faunal communities in the harbor.

Results of these various approaches to evaluating risk, both together and independently, support the conclusion that aquatic organisms are at significant risk due to exposure to

PCBs in New Bedford Harbor. Some risk due to exposure to metals was also identified; however, it was negligible compared to the risk due to PCBs.

Concentrations of dissolved PCBs in the area of maximum contamination (i.e., the Hot Spot) and in all areas of the Inner Harbor (i.e., inside the Hurricane Barrier) were sufficiently elevated to result in a significant likelihood of chronic effects to indigenous biota. PCB concentrations in sediment and sediment pore water in many areas of the harbor were found to be highly toxic to at least some members of all major taxonomic groups of organisms. In the Upper Estuary, the probability of these sediments being toxic to marine fish, the most sensitive taxonomic group investigated, approached certainty. These conclusions were found to be consistent with the reported results of laboratory experiments conducted using New Bedford Harbor sediments and with available data on faunal community structure. EPA ambient water quality criteria and interim sediment quality criteria were exceeded in many areas of the Inner Harbor.

Potential community or ecosystem level impacts due to PCBs in New Bedford Harbor cannot be evaluated fully by assessing impacts to individual species or taxonomic groups. However, the state of development of ecological risk assessment methodology does not allow quantification of impacts or risk at these higher levels. Nonetheless, the results of numerous site-specific and laboratory studies, including this risk assessment, indicate that New Bedford Harbor is an ecosystem under stress and there is a high probability that PCBs are a significant contributing factor to the integrity of the harbor as an integrated functioning ecosystem.

1.0 INTRODUCTION

1.1 NEW BEDFORD HARBOR ECOSYSTEM

New Bedford Harbor is a tidal estuary on the western shore of Buzzards Bay, Massachusetts, situated between the City of New Bedford on the west and the towns of Fairhaven and Acushnet on the east. The area contains approximately six square miles of open water, tidal creeks, salt marshes, and wetlands. The major freshwater inflow to this area is the Acushnet River, a small stream with mean annual flow of approximately 1 cubic meter per second. As a result, the system does not fit the traditional definition of an estuary; salinities throughout the harbor are high and the strong horizontal and vertical salinity gradients that control patterns of faunal distribution in estuaries are absent. Nonetheless, the system does provide habitats for a wide variety of aquatic organisms that use this area for spawning, foraging, and overwintering.

The topographical characteristics of New Bedford Harbor have been adequately described in several other reports generated as a result of studies undertaken to provide information for the Remedial Investigation/Feasibility Study (RI/FS) process and will not be repeated herein. However, several features of the area have importance for understanding the ecological risk assessment. The estuary and harbor may be conveniently divided into subareas by bridges and other manmade structures that also represent logical divisions between zones of ecological similarity. Therefore, the Coggeshall Street Bridge represents not only a convenient boundary for the area defined in these studies as the Upper Estuary, but also separates an area of shallow water with predominantly organic silts and clays with silty sands poorly sorted muddy to the north from deeper water with silty sands to the south (Figure 1-1). At the State Route 6 Bridge (Popes Island), depths generally increase, with water depths in most of the area south of the bridge maintained by dredging. This area of New Bedford Harbor is also the most heavily impacted by industrialization, with considerable shoreline development and ship traffic related to the fishing industry.

The Lower Harbor ends at the Hurricane Barrier, which separates the comparatively low-energy silty sediment of the harbor from the high-energy sands typical of littoral areas in Buzzards Bay. The Hurricane Barrier represents a significant feature of importance for the current regime in the harbor, and the jet effect created by the narrow opening dominates patterns of mixing.

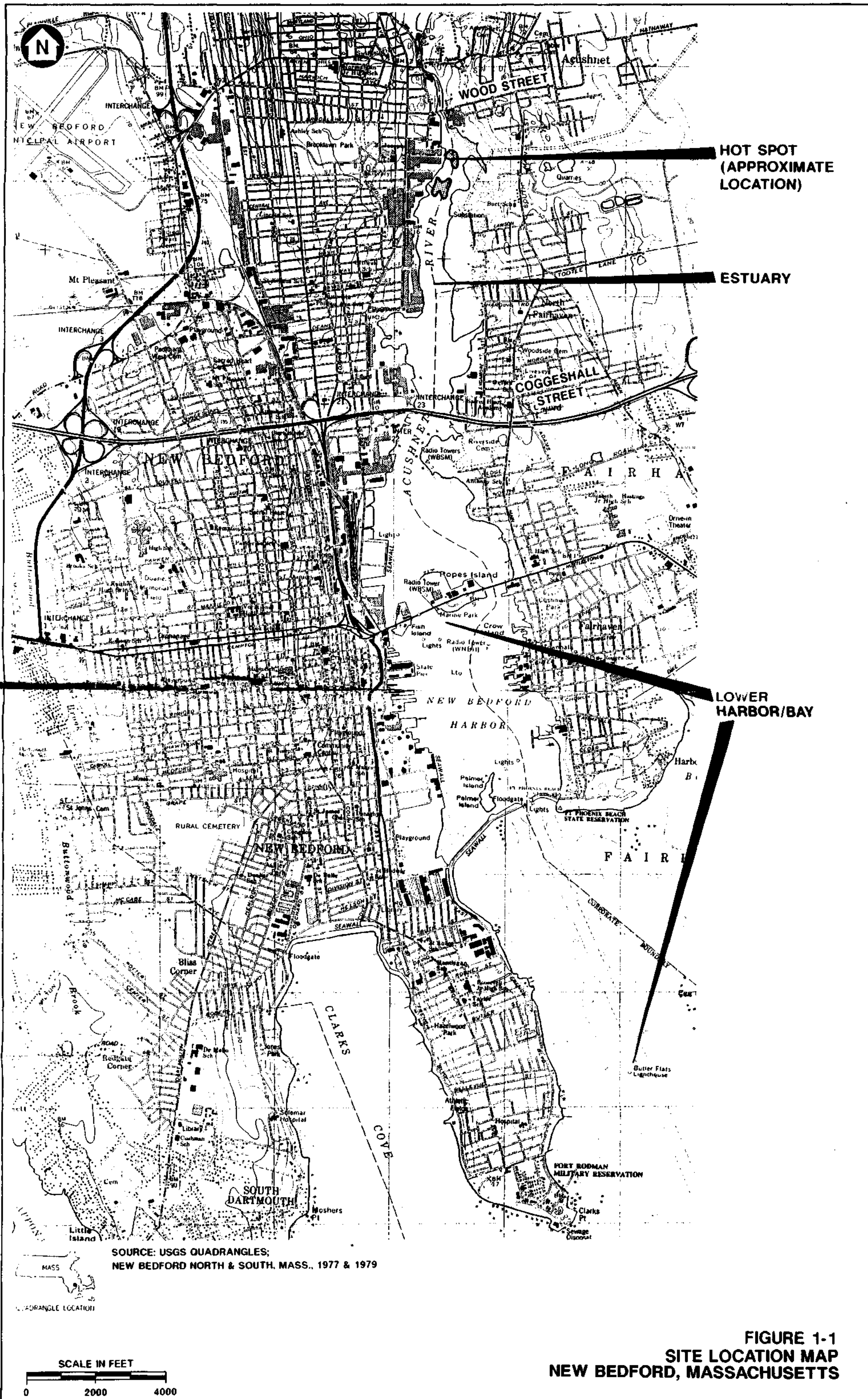


FIGURE 1-1
SITE LOCATION MAP
NEW BEDFORD, MASSACHUSETTS

1.2 SITE HISTORY

Between 1974 and 1982, a number of environmental studies were conducted to assess the magnitude and distribution of polychlorinated biphenyl (PCB) and, to a lesser extent, heavy metals contamination in New Bedford Harbor. Results of these studies revealed that sediment north of the Hurricane Barrier contain elevated levels of PCBs and heavy metals. Additional investigations revealed that PCBs had been discharged into the surface waters of New Bedford Harbor, causing significantly elevated PCB concentrations in sediment, water, fish, and shellfish.

To reduce the potential for human exposure to PCBs, the Massachusetts Department of Public Health closed much of the New Bedford Harbor area to fishing. Three closure areas were established on September 25, 1979 (Figure 1-2). Area 1 (New Bedford Harbor) is closed to the taking of all finfish, shellfish, and lobster. Area 2 (Hurricane Barrier to a line extending from Ricketson Point to Wilbur Point) is closed to the taking of lobster and bottom-feeding fish (eel, scup, flounder, and tautog). Area 3 (from Area 2 out to a line from Mishaum Point, Negro Ledge, and Rock Point) is closed to the taking of lobster.

In July 1982, the U.S. Environmental Protection Agency (EPA) placed New Bedford Harbor on the Interim National Priorities List (NPL). The final NPL was promulgated in September 1984. The site, as listed, includes the Upper Estuary of Acushnet River, New Bedford Harbor, and portions of Buzzards Bay. Following the NPL listing, EPA Region I initiated a comprehensive assessment of the PCB problem in the New Bedford Harbor area, including an areawide ambient air monitoring program, sediment sampling in the Acushnet River and New Bedford Harbor, and biota sampling in the estuary and harbor.

As a result of these studies, the extent of PCB contamination is better understood. The entire harbor north of the Hurricane Barrier, an area of 985 acres, is underlain by sediment containing elevated levels of PCBs and heavy metals. PCB concentrations in this area range from a few parts per million (ppm) to more than 100,000 ppm. Portions of western Buzzards Bay sediment are also contaminated, with PCB concentrations occasionally exceeding 50 ppm. The water column in New Bedford Harbor has been measured to contain PCBs in excess of the EPA 30-parts-per-trillion ambient water quality criterion

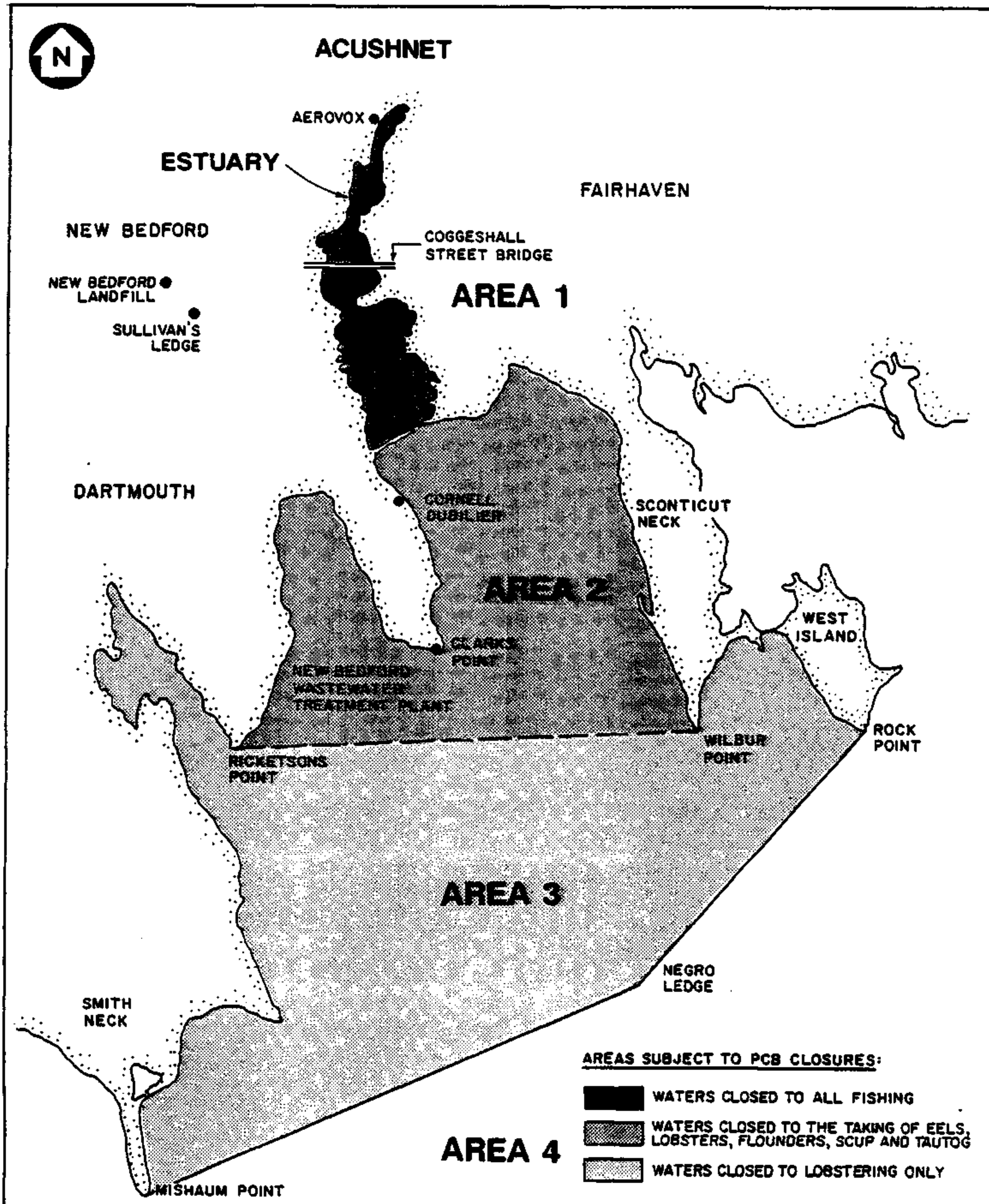


FIGURE 1-2
FISHERY CLOSURE AREAS
NEW BEDFORD, MASSACHUSETTS

NOT TO SCALE

(AWQC). Concentrations of PCBs in edible portions of locally caught fish have been measured in excess of the U.S. Food and Drug Administration (FDA) 2-ppm tolerance level for PCBs.

In 1984, EPA conducted an initial FS of the highly contaminated mudflats and sediment in the Upper Estuary of Acushnet River (NUS, 1984a and 1984b). Five clean-up options were presented in that report. EPA received extensive comments on these options from other federal, state, and local officials, potentially responsible parties, and the public. Many of the comments expressed concern regarding the proposed dredging techniques and potential impacts of dredging on the harbor, and potential leachate from the proposed unlined disposal sites.

In responding to these comments, EPA elected to conduct additional studies before choosing a clean-up alternative for the Upper Estuary. Concurrent with these studies, EPA conducted additional surveys to better define the extent of PCB contamination throughout the overall harbor and bay. Through these efforts, clean-up options for the site are being developed.

1.3 OBJECTIVES AND LIMITATIONS OF THIS REPORT

EPA Region I is responsible for the cleanup of the New Bedford Harbor site under authority of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) as amended by the Superfund Amendments and Reauthorization Act of 1986. Pursuant to this charter, EPA has direct responsibility for conducting the appropriate studies for this site to support the need for and extent of remediation. In accordance with the National Contingency Plan, these studies form the basis of the RI/FS for the site.

This ecological risk assessment presents and quantifies risks to aquatic organisms due to exposure to PCBs and selected heavy metals (i.e., copper, cadium, and lead) in the New Bedford Harbor area under baseline (i.e., existing) conditions. The baseline assessment is the first of a series of risk evaluations that will provide the basis for evaluating the need for and extent of remediation. It is based on existing conditions in New Bedford Harbor only; the potential natural decrease in contaminant mass and concentration in the harbor due to transport and degradation through time is not considered. Subsequent evaluations will examine the relative effectiveness of various remedial alternatives against current conditions using results of the numerical simulation model for PCBs.

EPA defines ecological risk resulting from toxic contaminants to include both direct risks to the growth, reproduction, or survival of the ecological receptor species, as well as the resource value of any species being reduced as a result of contaminant body burdens. Although both aspects of risk will be considered to some extent in this document, the former (direct) risk is the major concern of the assessment.

Ecological risks in New Bedford Harbor were determined by a mathematical evaluation and combination of two factors: (1) the degree of exposure to contaminants at the site, and (2) the ecotoxicity of PCBs and the three metals to aquatic organisms. Ecological risk was then quantified as the probability of impact to specific taxonomic groups representing the major ecotypes present in the harbor. Future evaluation of remedial alternatives via this method will require only repeating the exposure section of the assessment to reflect the new exposure conditions as determined by the numerical modeling results, and then using the previously derived (and unchanged) ecotoxicity calculations to determine new risk probabilities.

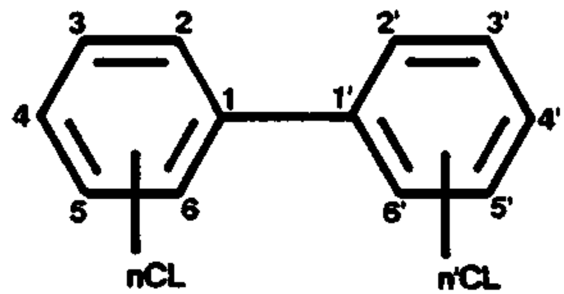
Following this strategy, this report consists of three sections. The first section is the exposure assessment, in which a representative subset of the organisms residing in the New Bedford Harbor area is identified, the routes of exposure are defined, and the degree of exposure is quantified. The second section, the ecotoxicity assessment, describes the acute and chronic toxic effects associated with PCB and metals exposure for each identified group. In addition, existing standards and criteria for PCBs and metals are discussed. The final section, the risk evaluation, combines the information presented in the two preceding sections to describe and quantify potential adverse effects on the New Bedford Harbor ecosystem resulting from the presence of these contaminants.

Both PCBs and metals are discussed in this report; however, PCBs were the primary focus of this study. Therefore, only the tables and figures for PCBs are included with the text. The tables and figures associated with the metals discussion are presented in Appendices A, B, and C.

The development of methodologies for determination of ecological risk is a relatively new and rapidly advancing field; the consensus among professionals concerning the most applicable methods at a particular site is limited. In addition, there are particular difficulties in determining risk due to PCBs in New Bedford Harbor because of the peculiar characteristics of PCBs as an environmental contaminant. PCBs are often treated as a single chemical or a small group of chemicals with similar properties; however, they actually consist of a group of 209 distinctly different chemical congeners. PCBs are relatively inert and, therefore, persistent compounds, with low vapor pressures, low water solubility, and high octanol/water partition coefficients. Although perhaps only half the potential congeners have actually been found to occur in the environment, they nonetheless consist of a diverse group of chemical species with widely varying physical, chemical, and biochemical properties.

In the manufacturing process, PCBs were formed by the addition of chlorine to the biphenyl molecule, and the number and types of PCB congeners formed in this process were not precisely determinable (Figure 1-3). Because PCBs were desirable primarily for their physical properties, which are largely related to the amount of chlorine substitution on the two rings, it was not necessary to know or control the exact congener mix; rather, only the percent of substituted chlorine in the final PCB mixture.

Most PCBs used in the U.S. were marketed as a mix of congeners under the name of Aroclor, a trade name of the Monsanto Company. Different Aroclors were designated by a four-digit code number (e.g., 1242 and 1254), with the last two digits signifying the amount of chlorine substitution as a weight percentage of the total mixture (e.g., Aroclor 1242 is 42 percent chlorine by weight). The sole exception to this numbering scheme is Aroclor 1016, which is approximately 41 percent chlorine. Aroclors 1016, 1242, and 1254 were most commonly used by the electrical component manufacturers in New Bedford. Because the desired properties of the Aroclors were determined by the overall amount of chlorination rather than the specific mix of congeners, it is probable that the actual congeners in a particular Aroclor varied among manufacturing batches. Reference Aroclors were subsequently established for analytical purposes; however, the relation of the reference Aroclors to the actual production batches is not clear.



(WHERE n AND n' MAY VARY FROM 0 TO 5)

FIGURE 1-3
CHEMICAL STRUCTURE OF A PCB MOLECULE
NEW BEDFORD, MASSACHUSETTS

After PCBs in the form of Aroclors are introduced into the environment, they begin to "weather," thereby changing and further complicating the problem of determining the actual mixture of components present. Lighter (i.e., less chlorinated) congeners are generally more volatile and soluble; therefore, they are (1) transported farther from the source before deposition, (2) less easily deposited into sediment, and (3) more easily mobilized and transported out of the original zone of deposition. More saturated congeners would demonstrate generally opposite behavior. In addition, differential rates of biochemical degradation, uptake, and depuration by biota, not easily related to level-of-chlorination but also determined by the actual pattern of chlorine substitution, would further serve to make the actual congener mix at any location different from the mixture originally released.

Although work is still ongoing to develop better analytical methods, it is possible to analyze environmental samples for many of the actual PCB congeners present; however, few congener-specific data are available because of the considerably greater analytical cost of the procedure. Most early studies reported PCBs as a "total" concentration or as the concentration of one or more Aroclors. Due to these problems, both methods produce less than completely satisfactory results. For the field sampling program conducted by Battelle Ocean Sciences (BOS) to produce calibration/validation data for the physical/chemical model (the source of much of the data used in this risk assessment), the analyses were reported in terms of "level-of-chlorination" homologs. This type of analysis provides valuable additional information, and because physical behavior determining fate and transport of PCBs is relatively similar for each homolog group, quantification (and subsequent numerical modeling) by homologs was deemed a reasonable cost-effective analytical goal for the modeling program. It was later decided to model only total PCBs, and the modeling program data were subsequently converted into total PCBs for risk assessment purposes by summing all homolog groups. Because the modeling and any remedial activities will be determined solely on the basis of total PCBs and, because of the lack of homolog-specific toxicity data, the risk assessment was conducted using total PCBs only.

The unique properties of PCBs and the problems with analysis described previously present considerable difficulties for determination of ecological (or public health) risk. Without analysis for specific congeners, it is not possible in most cases to know the actual congener

mix at a particular site, even if the exact congener composition of the PCBs introduced to the site were known, which is essentially never the case. Even if the mix of congeners were determined, the analysis would be valid only for the specific sample, and in an area such as New Bedford Harbor, the changing concentrations and mixture of congeners would present a complicated mosaic of spatial and temporal change. Therefore, the first step in conducting a risk assessment (i.e., determining the concentration of the contaminant(s) of interest at the specified site) is not possible for PCBs at the same level of detail as for other environmental contaminants.
Most analytical difficulties and uncertainties associated with determining PCB concentrations in the environment apply equally to any toxicological studies conducted with PCBs. A synthesis of the results of these studies is the second fundamental step in risk assessment and, because work to date has been conducted with contaminant concentrations reported as total PCBs or as one or more Aroclors, it is difficult to combine and use all data sources equally. Accordingly, various assumptions and simplifications were necessary at several points in the risk assessment so that the limited available data on PCB toxicity would not be unnecessarily reduced.

Recent work indicated substantial variability among congeners with regard to toxicity to aquatic organisms (Dill et al., 1982). Some toxicological properties are believed related to the configuration the two phenyl rings assume relative to each other which is, in turn, controlled by the position of the chlorines on the molecule. Fully ortho-substituted congeners do not assume a co-planar structure and are believed, in general, to be the least toxic. Conversely, non-ortho-substituted congeners are free to assume a co-planar configuration and are believed to be more toxic in general.

Site-specific water and sediment toxicity testing is perhaps the best solution to this problem; however, limited work has been conducted on New Bedford Harbor water and sediment. Although the availability of more data would have been valuable in that it would enable evaluation of the toxicity of the actual weathered PCB mixtures in New Bedford Harbor, it cannot prove that any effects measured are in fact due to the PCBs present rather than another contaminant. Therefore, both laboratory data on the toxicity of "pure" Aroclors and the limited data on actual toxicity of New Bedford Harbor environmental media must be used in combination to provide the "weight of evidence" for ecological risk.

The combination of these factors necessarily limits to some degree confidence in the accuracy of the risk probabilities for PCBs generated in this assessment, in the same way that confidence is decreased in using a statistical test to calculate probabilities when all assumptions for the test are not strictly satisfied. In some cases, it was possible to quantify the degree of uncertainty of some of the parameters and develop a quantitative estimate of overall uncertainty. For other issues, such as the question of congener-specific toxicity, it is not possible to approach the issue in a quantitative sense. However, because most toxicity studies have used congener mixtures, it is probable that a wide variety of toxicities is represented in both the test mixtures and the mixture occurring in New Bedford Harbor. The use of the risk probabilities in a relative sense (i.e., to compare the efficacy of different remedial alternatives against a no-action alternative) would have considerably greater validity, even if the absolute risk probabilities were questionable. It is this latter use that is important for the risk assessment.

Determination of risk due to heavy metals was not affected by the problems described previously for PCBs; however, other concerns became apparent during the analysis. Chief among these was the considerably smaller data set available for the three metals (particularly cadmium) and the probability that sampling for metals was concentrated in areas of suspected high concentrations, thereby biasing the data set. In addition, analysis of metals was deleted from the Battelle physical/chemical model and it was therefore not possible to work from the initial conditions established for each model cell, as was done for PCBs. This latter procedure would have largely corrected for the sampling bias. It was decided finally to use the available metals data exactly as provided thereby providing, to the extent that the data are biased toward higher concentrations, a more conservative estimate of risk.

1.4 PROGRAM DATA BASE

At most CERCLA sites, the ecological risk assessment would be based on findings of the RI report. However, because of the many studies conducted as part of the New Bedford Harbor project, numerous reports have been produced which obviate the need for a separate RI document. Therefore, this risk assessment is based primarily on the sampling data contained in the New Bedford Harbor data base, aspects of modeling efforts by HydroQual, Inc. (Hydroqual) and Battelle Pacific Northwest Laboratories (PNL), various site investigation reports, the Greater New Bedford Health Effects Study, and the U.S. Army Corps of Engineers (USACE) Pilot Dredging Study and Wetlands Assessment. An extensive data base generated between 1981 and 1986 provides an accurate description of the current extent and level of contamination within most of the New Bedford Harbor area.

1.4.1 PCB Concentrations in Sediments

Data on distribution of PCBs in sediment and overlying waters of New Bedford Harbor and the Acushnet River Estuary were provided by PNL and BOS. For consistency with other aspects of the RI/FS process at the New Bedford Harbor site, the ecological risk assessment for PCBs was based primarily on a data set developed as the initial conditions for the physical/chemical transport model. Initial conditions were established by PNL using information on PCBs in the harbor obtained from three sources: (1) data collected by BOS (Duxbury, Massachusetts) specifically for the calibration and validation of the model; (2) a data base compiled by GCA Corporation (now Alliance Technologies Corporation [Alliance]) from various historical sources; and (3) a detailed survey of PCBs in the harbor conducted by NUS Corporation (NUS). These three data sets were subsequently combined into the central New Bedford Harbor data base by BOS. An additional intensive sampling of the Hot Spot provided the data used to establish concentrations in Hot Spot sediment.

1.4.1.1 BOS Calibration/Validation Data

From 1985 through 1986, BOS conducted four samplings of water, sediment, and biota in the Acushnet River Estuary, New Bedford Harbor, and adjacent areas of Buzzards Bay to provide data for calibration and validation of the physical/chemical transport model and food-chain model. Twenty-five stations were established and sampled on each of three surveys; the remaining survey was limited to eight stations and was conducted immediately following a storm event. Although the samples obtained during these surveys were collected and analyzed under rigorous quality control procedures, the data were intended for use primarily for model calibration/validation. The usefulness for determining patterns of contaminant distribution in New Bedford Harbor is limited by the relatively sparse spatial distribution.

1.4.1.2 Alliance Data Base

This previously compiled data base summarizing several of diverse field investigations in New Bedford Harbor represents an important source of data and was used extensively to set initial conditions for the model. The data base was originally constructed for EPA by Metcalf & Eddy, Inc., in 1983 and was transferred to Alliance in 1986. Alliance began to expand the data base and converted it to run under dBASE III, a personal computer data base management software package. This work was never completed, and the data base was subsequently provided to BOS for quality assurance checks and subsequent incorporation into the central New Bedford Harbor data base. The Alliance data base was provided to PNL by E.C. Jordan Co. (Jordan) as part of the data base PNL used to establish initial conditions for the physical/chemical transport model.

1.4.1.3 NUS Data Base

The NUS data base was provided to PNL in digital form by BOS. The data base was apparently complete and contained data for PCBs expressed as the concentrations of various Aroclors for samples obtained on a regular grid. The NUS data proved to be valuable because concentration data for the entire study area was provided. Data in the Alliance data base, for example, were concentrated at the Hot Spot and around various wastewater or combined sewer overflow discharges.

Details of the data selection, conversions, and manipulations conducted by PNL to establish the initial sediment PCB concentrations for the physical/chemical model will be discussed in the final modeling report currently in preparation (Battelle, 1990). In the remainder of this section, aspects of this process that are important for understanding this risk assessment are reviewed.

1.4.1.4 Selection of Data

Sediment PCB data from the BOS and NUS data sets were complete and easily interpretable, and were used as received. The Alliance data base contained a wide variety of contaminant measurements and included samples of air, water, wastewater, sediment, and biota from the general vicinity of New Bedford Harbor. In addition to data on PCBs and metals, the data base included data on water quality parameters and other organic and inorganic contaminants, most of which were irrelevant for establishing initial PCB concentrations for the modeling. PCB data were retrieved from the Alliance data base via a series of FORTRAN programs written by PNL.

1.4.1.5 Sample Depths

The BOS data base contained various combinations of samples taken at a number of different horizons in the sediment, gross (bulk) samples, and samples of different size fractions (i.e., sand, silt, and clay). Only gross (bulk) sediment samples from the upper stratum (5 centimeters) were retained for subsequent evaluation. The NUS data included samples taken from the upper stratum (6 inches), depths of 12 to 18 inches, and at specified greater depths. Only samples from the upper 6-inch stratum were retained.

Reflecting its multiple data sources, the Alliance data base included a wide variety of sampling horizons. The data records were divided into two categories: (1) surface samples obtained with a grab sampling device or collected as subsamples from the upper 8 inches of a sediment core; and (2) deep samples, for which any part of the subsample was taken from 8 inches or deeper below the sediment water interface. Only the surface samples were used in subsequent data analysis.

1.4.1.6 Data Conversions

The data sets used by PNL to establish the initial conditions for the modeling included PCB data in various forms. The most variation was encountered in the Alliance data base, in which PCBs were reported most commonly as Aroclors 1242, 1254, and 1242/1016, and non-specific PCBs. Some samples included data on level-of-chlorination homologs. The desired final measure, total PCBs, was obtained for each sample by summing the concentrations of all quantified Aroclors. Any samples reported on a wet-weight basis were converted to dry weight using an average water content of 55 percent.

PCB concentrations in the NUS data base were reported as Aroclor 1242, Aroclor 1248, or Aroclor 1254 in units of micrograms per kilogram, and assumed to be dry weight. Typically, only one or two Aroclor concentrations were reported for each sample. All reported Aroclor concentrations were summed and converted to units of micrograms per gram (ug/g), equivalent to ppm dry weight.

The BOS data base reported PCB concentrations by level-of-chlorination homolog in units of ug/g dry weight. These concentrations were summed to produce an estimate of total PCB concentration.

Values below specified detection limits occurred in all three data bases and were used in determining the initial conditions; values reported as zero were not used. Data reported below detection limits were assigned a value equal to approximately 0.1 times the specified detection limit of the analytical procedure and were placed in a separate file. When detection limits were not reported, concentrations of zero were assigned values of approximately 0.1 times the lowest reported value. These somewhat arbitrary assignments were necessary because the data were later log-transformed and values of zero would have been unacceptable.

1.4.1.7 Data Processing and Analysis

Standard univariate statistics were calculated by PNL for the raw and log-transformed data. The log-transformed data produced near-normal distributions around the mean value for each data set.

Contour plots of the surface sediment PCB concentrations were prepared at PNL and delivered to Jordan in November 1987. Initial PCB concentrations were calculated by PNL on a 100-by-100-foot grid and subsequently transferred to the larger i,j physical/chemical model grid by calculating an arithmetic average of all 100-foot grid data within each model grid element. The initial values for the i,j model grid, provided to Jordan by PNL in April 1989, were used for all subsequent analyses conducted for the ecological risk assessment, with one modification at the Hot Spot. Following the final assignment of initial conditions for the model, USACE funded an additional intensive survey of PCB concentrations in the Hot Spot. Three model grid cell concentrations were changed from initial condition assignments to reflect the updated information.

1.4.2 PCB Water Concentrations

PCB concentrations in the water column for the risk assessment were also based on values used for the physical/chemical transport model. However, unlike sediment concentrations, the use of initial conditions is not appropriate because preliminary model runs indicated that concentrations in the water column are determined largely by the assigned sediment concentrations following a brief "spin-up" period of approximately 90 days simulation. Accordingly, PNL did not determine initial conditions for the water column in a manner similar to that previously described for sediment; rather, it assigned initial conditions generally consistent with the field data and then allowed the model to produce its own "starting conditions" based on the assigned sediment concentrations. These starting conditions in the water column were averaged vertically for each cell in the i,j grid and provided to Jordan with the initial sediment conditions.

1.4.3 Metals Concentrations

Because metals were not included in the Battelle physical/chemical modeling effort, it was not possible to use model initial conditions for the calculation of exposure estimates at the New Bedford Harbor site. Metals data were obtained from the program data base maintained by BOS. All data for the three metals in water and sediment were requested and received via magnetic disk. Data characterized as "rejected" in the data validation were removed from the data set and not used in the risk assessment. The data set contained numerous "non-detects," which were entered into the analysis as half the lowest reported concentration for the particular metal. All remaining data were used as received.

1.5 OVERVIEW OF METHOD FOR THE ECOLOGICAL RISK ASSESSMENT

A joint probability model was used in the risk assessment to quantitatively evaluate potential impacts to New Bedford Harbor biota for each contaminant. The basic components of the model are two probability distributions, one representing the expected distribution of contaminant levels in the environment, and the second representing the probability distribution of some benchmark concentration for a particular group of potential receptors over a range of contaminant levels. The joint probability model is used to determine the likelihood that a typical species (which displays a particular biological effect at the benchmark concentration) will encounter an environmental concentration sufficient to elicit the particular effect.

In Subsection 2.1.2, development of the expected distribution of environmental levels is discussed. These distributions are termed expected environmental concentration (EEC) probability curves. The development of the probability density function that relates contaminant concentration to a biological benchmark is discussed in Subsection 3.2. Finally, the joint probability model is used to determine quantitative risk estimates in Section 4.0.

2.0 EXPOSURE ASSESSMENT

The environmental exposure assessment was performed to identify representative organisms within New Bedford Harbor that may be exposed to PCBs and metals. The assessment included identification of ecological receptors and exposure routes, with the goal of selecting a subset of species to represent the wide variety of potential aquatic receptors at the site. These species were used to identify the principal routes of exposure and describe contaminant exposure within the New Bedford Harbor area.

For the purposes of accumulating results at various (simulated) points in time, the Battelle transport model divides the estuary and harbor into the following five zones, based in part on natural and manmade structures and on the initial contaminant concentrations detected in the sediment (Figure 2-1):

- Zone 1: the area between the Wood Street Bridge and the southern boundary of the Hot Spot
- Zone 2: from the southern boundary of the Hot Spot to the Coggeshall Street Bridge
- Zone 3: the area between the Coggeshall Street Bridge and Popes Island (State Route 6 Bridge)
- Zone 4: the area between Popes Island (State Route 6 Bridge) and the Hurricane Barrier
- Zone 5: from the Hurricane Barrier out to the limit of the modeling grid, roughly delineated by the line from Ricketsons Point to Wilbur Point

Different systems of dividing New Bedford Harbor into zones have been used at various times for specific purposes. The zone definition used in this report for the purpose of the ecological risk assessment is identical to the zonation being used for the physical/chemical transport modeling. The risk assessment is based primarily on both the input to and output from the model, and use of the same zones simplified inclusion of the data from modeling runs. Therefore, slightly different divisions of the harbor were used for the HydroQual food-chain model, the public health risk assessment, and the draft ecological risk assessment.

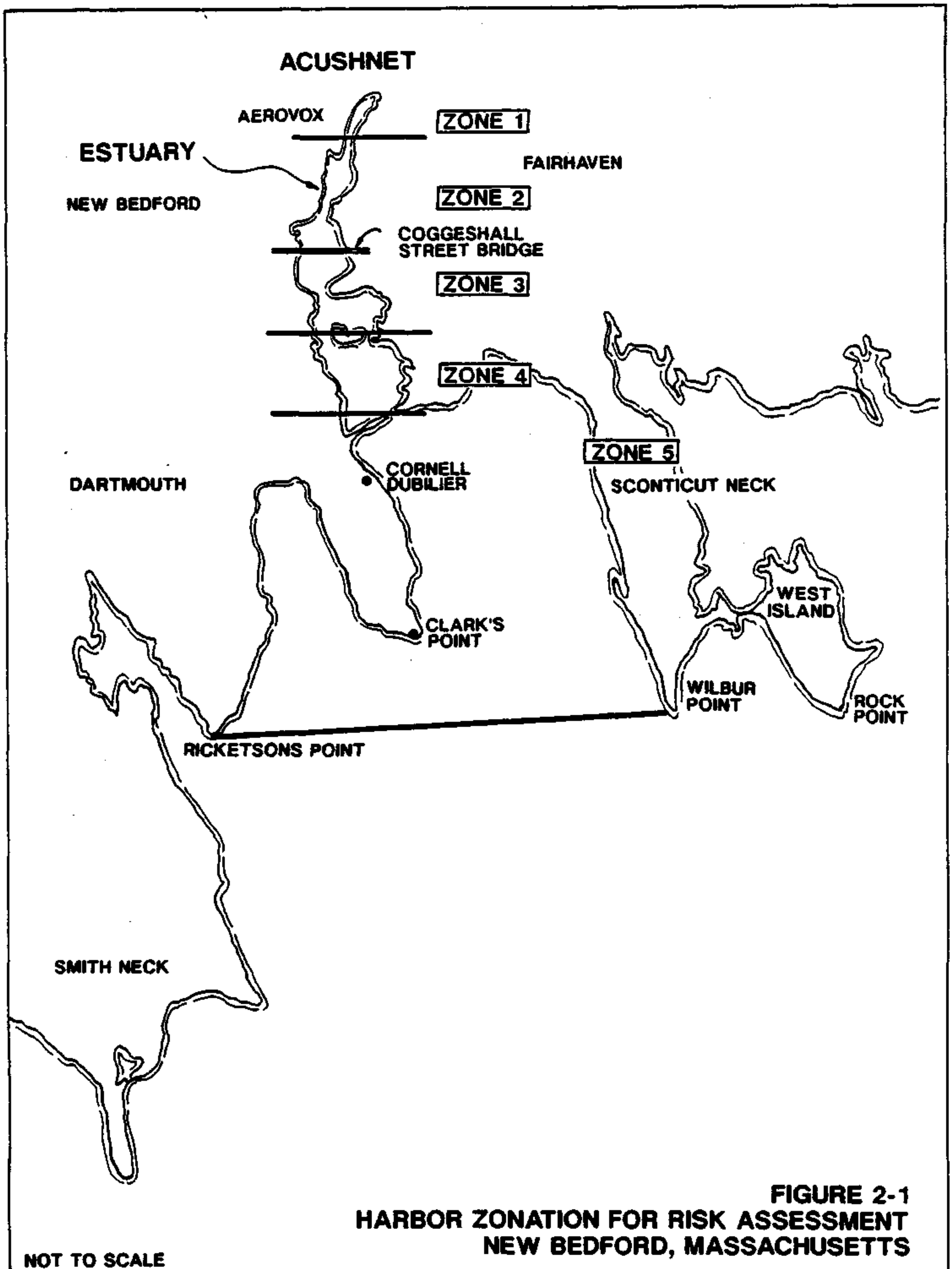


FIGURE 2-1
HARBOR ZONATION FOR RISK ASSESSMENT
NEW BEDFORD, MASSACHUSETTS

Although all these divisions correspond in some areas to the various fishery closure zones, none is exactly the same.

2.1 RECEPTOR IDENTIFICATION

2.1.1 Exposed Species Analysis

Many organisms in New Bedford Harbor are potentially at risk as a result of exposure to PCBs and heavy metals. The four primary routes of exposure include (1) direct contact with the water in the water column, (2) direct contact with or ingestion of sediment, (3) direct contact with sediment pore water, and (4) ingestion of contaminated food. The route of exposure can also be defined by the method of obtaining food (e.g., herbivore, carnivore, suspension feeder, deposit feeder, and scavenger). To describe how aquatic organisms may be exposed to contaminants at the New Bedford Harbor site, a representative subset of the species known to inhabit this area was identified. The basis of the selection was defined by the possible routes of exposure for the organisms in question.

To evaluate the level of effects due to exposure and for risk characterization, the organisms in New Bedford Harbor were separated into ecotypes, which also correspond to taxonomic groups. Five groups of organisms, corresponding to the major aquatic organisms present in the harbor and also representative of the range of exposure routes, were developed: marine fish, crustaceans, mollusks, polychaetes, and algae. The rationale for these groupings and typical representative species for each in New Bedford Harbor are presented in Section 3.0. Lack of toxicological data for marine polychaetes precluded separate analysis of potential contaminant effects on this group. However, these organisms are considered relatively insensitive to organic contamination in sediment and are widely used for bioaccumulation studies for this reason. In the determination of risk in Section 4.0, it is assumed that a typical polycheate would be no more sensitive than a typical mollusk, and the benchmark distribution for mollusks will be used conservatively to assess risk to polychaetes as well.

Although most organisms can be exposed to environmental contaminants via all media, for purposes of assessing exposure in this risk assessment, the various habitat locations (i.e., benthic or pelagic), lifestages (i.e., egg, larvae, and adult), and feeding method (e.g., filter feeder, deposit feeder, or carnivore) of typical members of each group were used to define the primary routes of exposure for the group. Based on habitat, direct contact with dissolved or particulate contaminants in the water column was considered the primary route of exposure for pelagic fish, bivalves, and plankton. An important secondary route of exposure for most species is consumption of biota that have bioaccumulated contaminants. For benthic infaunal invertebrates, it was determined that direct contact with and ingestion of contaminated sediment and food organisms were the primary routes of exposure. Direct contact with the water column was determined to be a secondary route of exposure, although it can also be the primary exposure route for planktonic lifestages of infaunal adults.

2.1.2 Species of Concern

Species of concern inhabiting the New Bedford Harbor area were identified based on the biological surveys conducted by IEP, Inc., for USACE (USACE, 1988b); Sanford Ecological Services for USACE (USACE, 1986); Camp, Dresser and McKee (Camp, Dresser and McKee, 1979); and historical data reported in Bigelow and Schroeder (Bigelow and Schroeder, 1953).

A subset of receptor species was selected from these data based on the following criteria: distribution within the study area, trophic level (i.e., producer, primary, secondary, or tertiary consumer); commercial and/or recreational use; and availability of biological and ecological information.

Criteria such as habitat location, trophic level, and reproductive potential are important factors that may influence the ways in which each species may be exposed to contaminants in the New Bedford Harbor area and the potential effects of contaminant exposure. The commercial and/or recreational value of a resource species is a key factor for species selection because the loss and limitation of use of such species may have economic significance.

Twenty-eight species of various trophic levels and habitat types representing the five taxonomic groups of aquatic organisms discussed previously (i.e., finfish, crustaceans, mollusks, annelids, and plankton) were selected as typical aquatic receptors for the New Bedford Harbor site. Distribution of these species within the Acushnet River/Buzzards Bay area is shown in Table 2-1.

TABLE 2-1
DISTRIBUTION OF THE 28 SELECTED SPECIES OF CONCERN IN NEW BEDFORD HARBOR

NEW BEDFORD HARBOR

ALL ZONES	ZONE 1 (AREA 1)	ZONE 2 (AREA 1)	ZONE 3 (AREA 1)	ZONE 4 (AREA 1)	ZONE 5 (AREA 2)
Fish					
Herring Flounder Silverside Mummichog	American Eel	American Eel	Scup Tautog American Eel	Scup Tautog Mackeral	Scup Tautog Mackeral
Crustaceans					
	Isopod	Blue Crab Fiddler Crab Green Crab Amphipod	Blue Crab Green Crab Lobster Fiddler Crab Amphipod Grass Shrimp	Green Crab Lobster Grass Shrimp	Lobster Amphipod
Mollusks					
Quahog Ribbed Mussel	Mud Nasa Soft-shell Clam	Mud Nasa Soft-shell Clam Blue Mussel Quahog	Blue Mussel Slipper Shell Bay Scallop Soft-shell Clam Eastern Oyster Quahog	Blue Mussel Slipper Shell Eastern Oyster Quahog	Quahog
Plankton					
Diatoms		Copepod	Copepod	Copepod	Copepod
Annelids					
Clam Worm Mud Worm Thread Worm					

NOTE:

Zones correspond to Figure 2-1; areas correspond to Figure 1-2.

2.2 EXPOSURE LEVELS FOR RECEPTORS

2.2.1 Introduction

The amount of contaminant exposure experienced by an aquatic organism is a function of the type(s) of contaminated media to which the organism is exposed, contaminant concentrations in the media, and the mechanisms by which contaminants are taken up from each medium. Each factor was considered and, to the extent possible, quantified, in determining exposure levels for the five organism groups used for the risk assessment.

PCB contamination in New Bedford Harbor has been documented in all environmental media (i.e., water, sediment, and biota) throughout the harbor; however, it varies considerably in concentration, generally decreasing with distance from the Hot Spot in the Upper Estuary. Metals contamination is similarly ubiquitous; however, the area of highest metals concentrations is found in Zone 3 between the Coggeshall Street and Popes Island bridges. Organisms residing in New Bedford Harbor for all or part of their lives may be exposed to these contaminants as a result of direct contact with and/or ingestion of contaminated food, water, and sediment. Migration from the harbor of prey species with elevated PCB and metals tissue burdens expands the potential area of exposure for predators. Uptake of contaminants from water, sediment, or food into the tissues of organisms ultimately occurs by either passive diffusion, active transport, or facilitated transport across the membranes of the gills, gastrointestinal lining, mouth lining, and body wall (Swartz and Lee, 1980).

Terms such as bioconcentration and bioaccumulation relate to the source and specific outcomes of exposure to contaminants. Bioconcentration refers to the net uptake of dissolved chemicals into an organism from water. Another directly related term, bioconcentration factor (BCF), is the ratio of concentration found in the tissue of an organism to the concentration in the water to which the organism was exposed (Schimmel and Garnas, 1985). The term bioaccumulation refers to the net uptake of a contaminant by an organism from all sources, including ingestion of and/or contact with water, food, and sediment (Menzer and Nelson, 1986). Biomagnification is generally used to refer to the concentration of a contaminant between trophic levels in a food chain.

2.2.2 Methods

PCB concentrations in the water column (i.e., dissolved concentration), pore water, and sediment developed as initial conditions for the modeling program were the primary sources of exposure data for the ecological risk assessment. The source and development of the initial condition concentrations are discussed in Subsection 1.4. For the Upper Estuary Hot Spot, the initial conditions data were supplemented with concentrations obtained from the USACE data set for this area (USACE, 1988c).

The modeling program PCB data were provided as total bed sediment concentrations and vertically averaged water column concentrations for each element in the i,j grid used for the physical/chemical model. Each data point was weighted equally for subsequent analysis; however, there is some variation in the size and, therefore, the amount of the harbor represented by each model grid element. Hot Spot concentrations, assumed to represent the range of concentrations present in the Hot Spot, were also weighted equally.

All data were log-transformed and assigned to one of six groups representing the Hot Spot and each of the five zones of the harbor discussed previously (see Figure 2-1). Simple descriptive statistics (mean and variance) were calculated for each zone and used to generate an EEC probability function for each zone. EECs are cumulative frequency distributions that quantify the likelihood that the actual environmental concentration at any location in a zone will be equal to or less than a particular value.

Because the joint probability model used to estimate risks in Section 4.0 presumes that the EEC and the effects distributions are normally distributed, the log-transformed PCB concentration data for each harbor zone were examined for deviations from normality using the Kolmogorov-Smirnov test (i.e., a=0.05). In most cases, results indicated that the transformed concentration data are not normally distributed. No other transformations were attempted to rectify this problem, because the toxicological data used in development of effects curves are log-normally distributed, and the same scales must be used for both the EEC and effects distributions to determine a joint probability risk estimate. Also, examination of the moment statistics for EEC distributions indicated that the major reason distributions are not normally distributed is due to leptokurtosis rather than skewness. In contrast with skewed distributions, the distributions are symmetrical around the mean value, and deviations from normality are less problematical.

Data reduction and analysis for metals was conducted following procedures essentially similar to those described previously for PCBs, the primary difference being that raw data from the program data base maintained by BOS were used in place of initial conditions for the physical/chemical model.

2.2.3 Exposure to Water Column Contamination

2.2.3.1 Species and Mechanisms

Organisms exposed to contaminants primarily via the water column include pelagic or planktonic species that live suspended or swimming in the water column, and demersal finfish that may have some contact with the bottom but receive most exposure from the water. Representative pelagic and demersal fish found in the New Bedford Harbor area include winter flounder (*Pseudopleuronectes americanus*), bluefish (*Pomatomus saltatrix*), blueback herring (*Alosa aestivalis*), and Atlantic silverside (*Menidia menidia*).

Phytoplankton and zooplankton are also exposed nearly exclusively via contaminants in the water column. Although effects on holozooplankton and phytoplankton are usually not of direct concern, their importance for higher trophic levels can be significant. Representative plankton in New Bedford Harbor include the copepods (*Acartia tonsa*) and two diatoms (*Rhizosolenia alata* and *Skeletonema costatum*). The opossum shrimp (*Neomysis americana*) is generally considered epibenthic rather than planktonic; however, for the purposes of the risk assessment, its behavior is sufficiently similar to planktonic organisms that it can be considered part of the planktonic group.

Bivalve mollusks, although seemingly species that would be exposed via sediment, are primarily exposed to waterborne contaminants due to the filtering of large amounts of water to extract food. In addition, bivalve mollusks have planktonic larval stages that are also exposed to contaminants in the water column. Representative bivalves in New Bedford Harbor include the Atlantic ribbed mussel (*Geukensia demissa*), the blue mussel (*Mytilus edulis*), the Atlantic bay scallop (*Aequipecten irradians*), and the Eastern oyster (*Crassostrea virginica*).

For all these organisms, the epithelial tissue of the gills is usually the primary site of contaminant uptake because of its structure and function. Uptake of contaminants from water can also occur across the linings of the mouth and gastrointestinal tract, the sensory organs, and even the viscera if they are perfused with water, as in some mollusks. Waterborne contaminants can also become adsorbed onto exposed surfaces such as the skin, where they may disrupt the function of some tissues but do not generally contribute to systemic toxicity.

2.2.3.2 PCB Exposure Concentrations in Water

Exposure levels in the water column are for the disssolved concentrations of PCBs. The dissolved component in the water column, as opposed to total concentrations, was used because most data about toxicological effects of PCBs on organisms are based on dissolved concentrations. Therefore, assessing the impact of dissolved concentrations of the contaminant more directly relates to the toxicological data. The concentration is the average for the entire water column. The mean, standard deviation, and variance for each zone are listed in Table 2-2. Cumulative probability plots for the water column exposure levels, presented in Figure 2-2, are based on a random sample of 100 data points from distributions with the calculated parameters (see Table 2-2). As shown in Table 2-2, the mean water column PCB levels decrease with increasing distance from the Hot Spot in Zone 1. Despite the large difference in the number of grid elements for the various zones, the variances associated with the different zones are similar. Mean values for Zone 1 and the Hot Spot are 2.55 and 3.10 micrograms per liter (ug/L), respectively, decreasing to 0.02 ug/L in Zone 5.

Because of the similarity in the variances associated with the environmental concentration data, the shape of the resulting EEC curves are similar, differing mainly in location along the PCB concentration axis (see Figure 2-2).

2.2.3.3 Metals Exposure Concentrations in Water

The exposure levels in the water column for all metals are for the dissolved concentrations of the metals. As in the case of PCBs, the dissolved component was used rather than the total concentration because most of the data about toxicological effects of metals are based on dissolved concentrations. The geometric mean, standard deviation, and variance for each zone are in Appendix A; that is, Table A-1 for copper, Table A-2 for cadmium, and Table A-3 for lead. The cumulative EEC probability plots for all zones for copper, cadmium, and lead are presented in Figures A-1, A-2, and A-3, respectively.

There is little indication of any relationship between the concentrations of copper and cadmium, and distance from

TABLE 2-2
EXPECTED EXPOSURE CONCENTRATIONS FOR PCBS (1)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

HARBOR ZONE	MEAN (ug/l)	TRANSFORMED VALUES (2)		
		MEAN	STANDARD DEVIATION	VARIANCE
Hot Spot, Water Column	3.097	0.491	0.128	0.016
1. Water Column	2.559	0.408	0.139	0.019
2. Water Column	1.074	0.031	0.272	0.074
3. Water Column	0.157	-0.804	0.250	0.063
4. Water Column	0.065	-1.185	0.099	0.010
5. Water Column	0.023	-1.639	0.255	0.065
Hot Spot, Pore Water	73.114	1.864	0.642	0.767
1. Pore Water	38.282	1.583	0.302	0.091
2. Pore Water	4.406	0.644	0.954	0.910
3. Pore Water	0.277	-0.558	0.393	0.154
4. Pore Water	0.075	-1.125	0.708	0.502
5. Pore Water	1.000	-1.320	0.551	0.303

NOTES:

1. All data developed using initial conditions for Battelle numerical model. Expected pore water concentrations derived from initial sediment concentrations times model mass-transfer coefficient.

2. Log (base 10) transformed values, with standard deviations and variances.

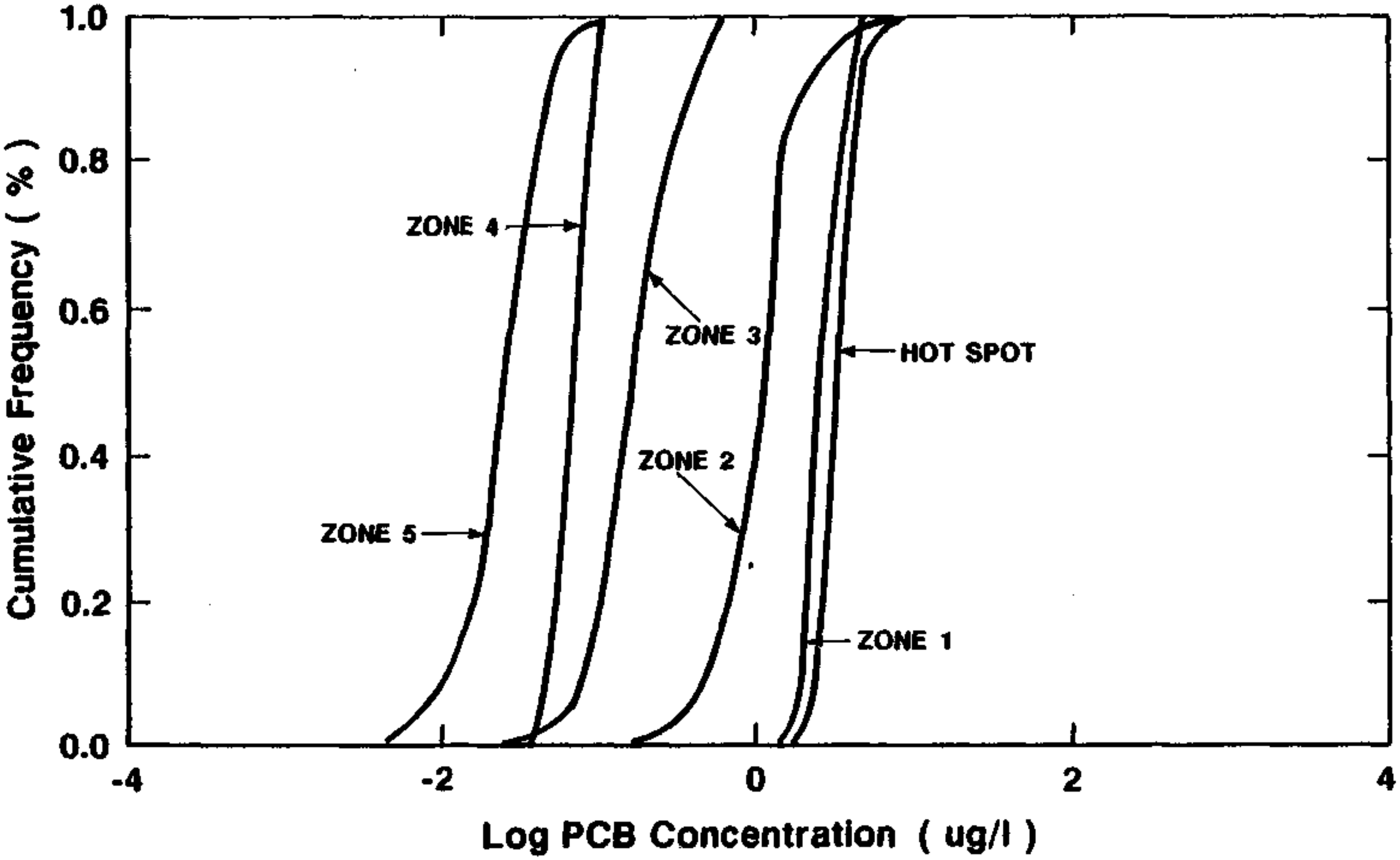


FIGURE 2-2
EECs BY ZONE FOR PCBs, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

the Upper Estuary, as was found with PCBs. However, there is a noticeable decrease in lead concentrations with increasing distance from Zone 1; within zones, lead concentrations were more variable than copper and cadmium concentrations.

2.2.4 Exposure to Sediment Contamination

2.2.4.1 Species and Mechanisms

Direct contact with and ingestion of contaminated sediment and its associated pore water are the primary routes of exposure for benthic infauna that live in close association with or are buried in the sediment. Exposure of epifaunal benthic organisms is more difficult to quantify because they are exposed to both sediment and the overlying water; for these species, exposure primarily to sediment can be used as a conservative worst case. Typical benthic invertebrates in New Bedford Harbor include the American lobster (*Homarus americanus*), amphipod (*Ampelisca vadorum*), tubificid worm (*Tubificoides* sp.), slipper shell (*Crepidula fornicata*), and mud snail (*Ilyanassa obsoleta*).

In the environment, sediment usually provides the most concentrated pool of contaminants, as evidenced at the New Bedford Harbor site (Larsson, 1985). For most of the contaminated sediment in the harbor, PCBs and metals are continually being released into the interstitial or pore water, from which uptake by benthic organisms occurs. Resuspension of sediment also increases total contaminant concentrations in the water column, but these particulate-bound contaminants are not directly available for uptake as are the dissolved-phase contaminants.

Sediment-bound contaminants are also taken up directly from the sediment by aquatic organisms (O'Donnel et al., 1985). Deposit-feeding organisms that feed by ingesting sediment also ingest any contaminants bound to the sediment. Contaminants strongly bound to sediment are less likely to desorb from sediment particles, and are absorbed in the gut less than the more weakly bound contaminants. Uptake may also occur as a result of equilibrium partitioning of contaminants between the body surfaces of the organism and surface coatings of the sediment (Swartz and Lee, 1980).

Although these various modes of uptake have all been documented, a quantitative assessment of risk incorporating all the mechanisms is not possible because of the lack of sufficient relevant toxicological data. Therefore, risk for benthic organisms was defined as risk due to exposure to contaminants dissolved in pore water. By assessing risk in this form, it is possible to draw on the body of toxicological data that has largely been developed using dissolved contaminants.

2.2.4.2 PCB Exposure Concentrations in Sediment Pore Water

PCB concentrations in pore water were calculated from the initial conditions sediment concentration data for the physical/chemical model via partition coefficients (K_d). Because of the properties of PCBs discussed in Subsection 1.3, partitioning is a complex phenomenon that varies over several orders of magnitude according to specific PCB congeners. Because the PCBs present in New Bedford Harbor represent a mixture of congeners, no single K_d can fully describe the partitioning that is occurring.

Values for site-specific apparent K_d in New Bedford Harbor are available from experiments conducted by BOS as part of the modeling program, and from the literature (Brownawell and Farrington, 1986). The K_ds ultimately selected were numerically equivalent to the mass transfer K_ds used in the physical/chemical model to approximate diffusion of dissolved PCBs from bed sediment, and are generally comparable to K_ds determined empirically by BOS, and consistent with the range of values reported in other studies (Brownawell and Farrington, 1986; and Pavlou and Dexter, 1979).

For areas above the Coggeshall Street Bridge (i.e., Zones 1 and 2), the K_d used was $5x10^5$; below the Coggeshall Street Bridge (i.e., Zones 3, 4, and 5), the K_d used was $2x10^5$. The K_ds were applied to the original data and the results log-transformed. Descriptive statistics were calculated as described for water concentrations, and the results are summarized in Table 2-2. As with the water column data, estimated pore-water PCB concentrations are highest in the Hot Spot, decreasing with distance from this area. Mean values for Zone 1 and the Hot Spot are 38.28 and 73.11 ug/L, respectively, decreasing to 0.05 ug/L in Zone 5. As was the case with data for water column PCB levels, variances associated with estimated pore water levels for the different zones are comparable, resulting in similarly shaped EEC curves (Figure 2-3).

2.2.4.3 Metals Exposure Concentrations in Sediment Pore Water

Exposure levels for metals in the pore water were calculated from the sediment concentrations via K_ds.

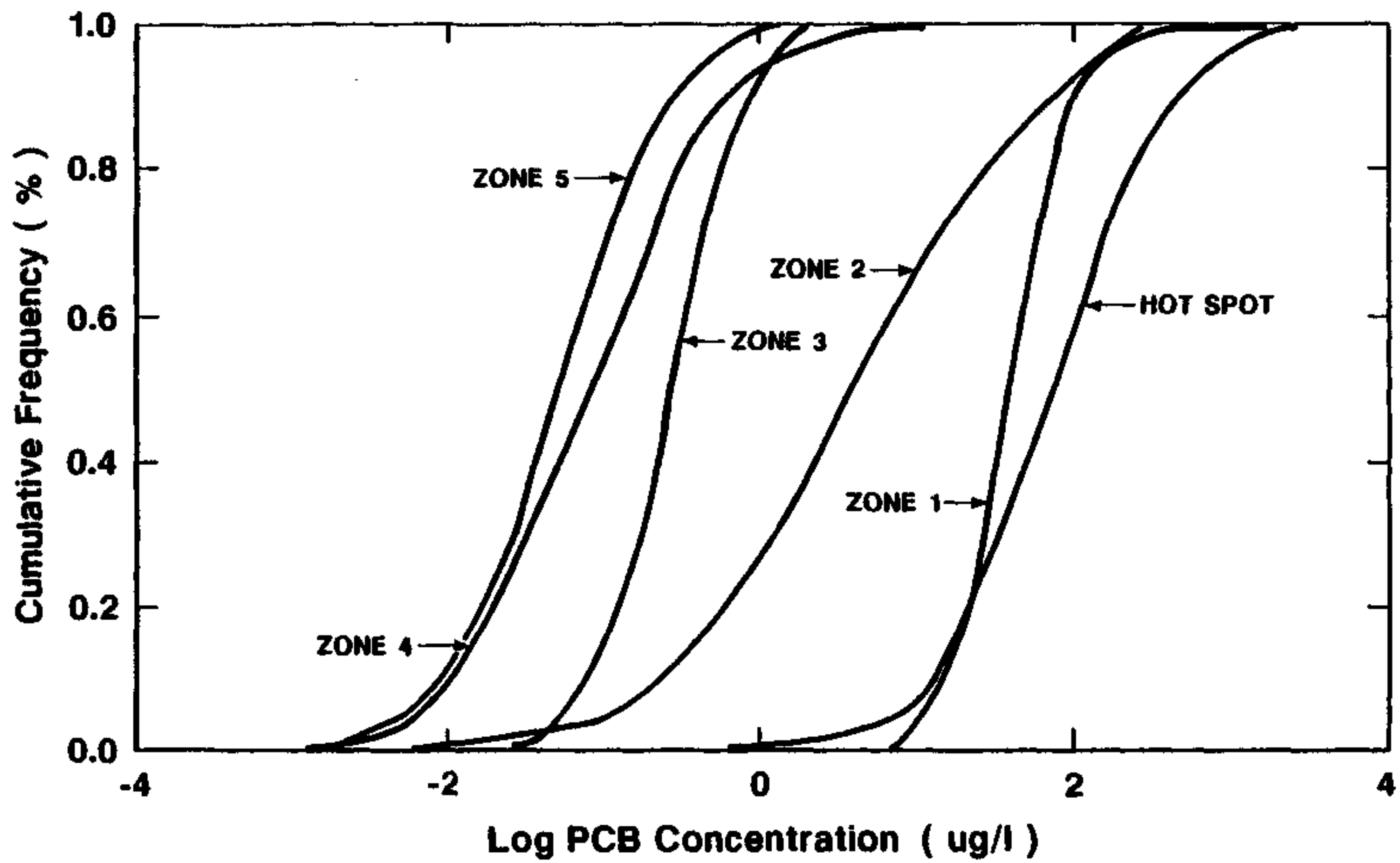


FIGURE 2-3
EECs BY ZONE FOR PCBs, PORE WATER
NEW BEDFORD, MASSACHUSETTS

The K_ds used were based on field measurements made throughout the New Bedford Harbor site, provided by Damian Shea from BOS (unpublished masters thesis). The K_ds used were $8x10^5$ for copper, $4x10^4$ for cadmium, and $2x10^5$ for lead.

The mean, standard deviation, and variance for each zone are presented in Table A-1 for copper, Table A-2 for cadmium, and Table A-3 for lead. The cumulative EEC probability plots for all zones for copper, cadmium, and lead are presented in Figures A-4, A-5, and A-6, respectively.

Calculated pore water concentrations of copper and cadmium were the lowest in Zone 5 and the highest in Zones 1 and 3 (Figures A-4 and A-5). Lead concentrations in the pore water were the lowest in Zone 4 and the highest in Zones 1 and 3. For all metals, the highest variance was associated with Zone 2. As with the water column concentrations, a decrease in concentrations with increasing distance from the PCB Hot Spot is not as well defined as for PCB concentrations, although a weak trend can be observed.

2.2.5 Exposure to Contaminated Food

Allotrophic organisms in New Bedford Harbor are exposed to PCBs and metals via ingestion of contaminated food. Lipophilic organic compounds (e.g., PCBs) transfer efficiently across the gut membranes because of the relatively long contact time between food and membranes. The consumption of contaminated food is of concern if dietary intake directly results in toxicity, and/or if the chemical is subject to food-chain transfer resulting in tissue burdens that may potentially be toxic.

A food-chain model is being developed for the New Bedford Harbor site by HydroQual. The transfer and fate of PCBs and metals are being assessed with the model for two different food chains, culminating in American lobster (Homarus americanus) and winter flounder (Pseudopleuronectes americanus), respectively (Figures 2-4 and 2-5).

The HydroQual model consists of a series of differential equations that numerically simulate the various processes that determine the residue value, or amount of a contaminant that remains in the tissues of the organism over time. Processes simulated in the model include surface sorption, transfer across the gills, ingestion of

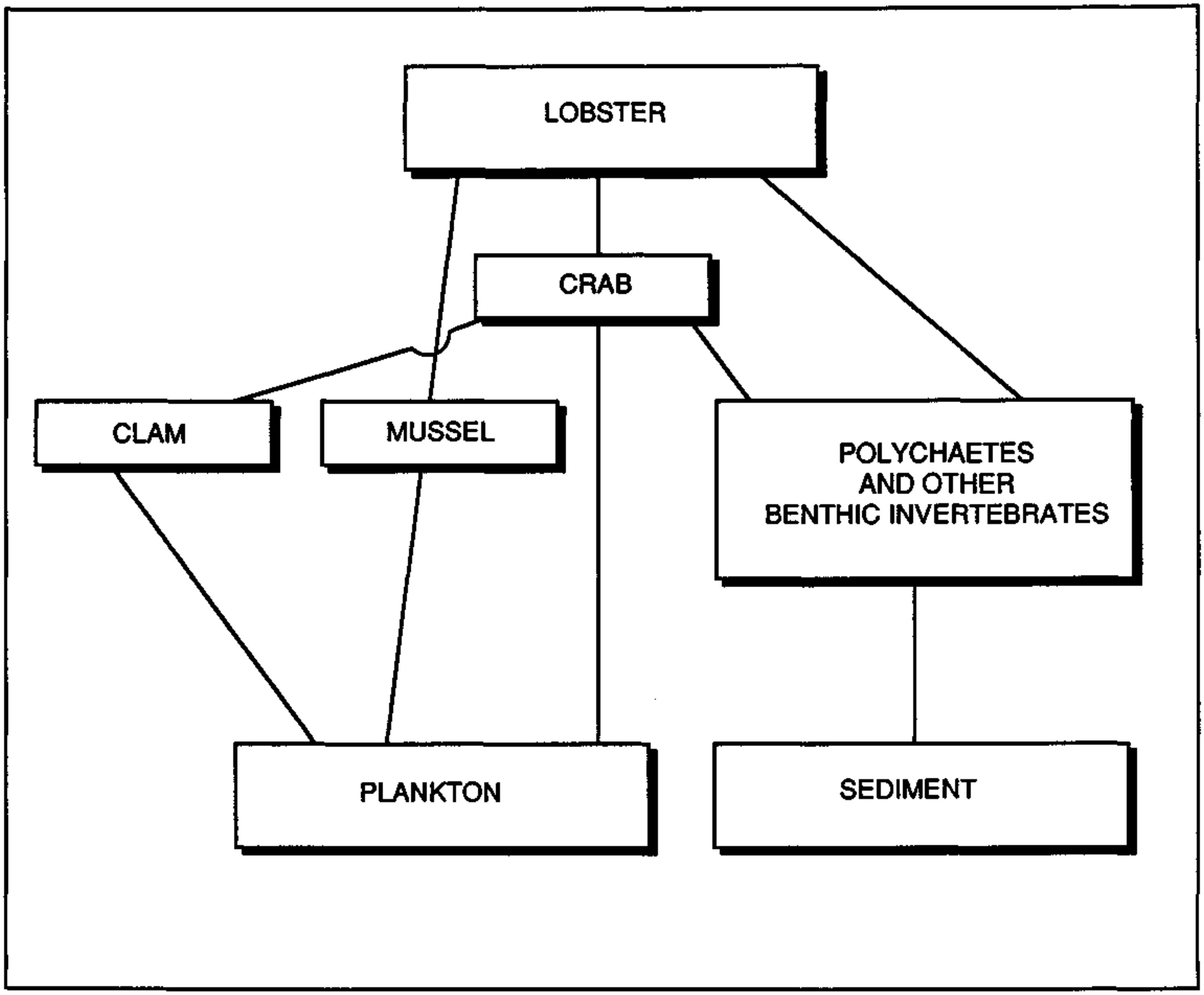


FIGURE 2- 4
LOBSTER FOOD CHAIN
NEW BEDFORD, MASSACHUSETTS

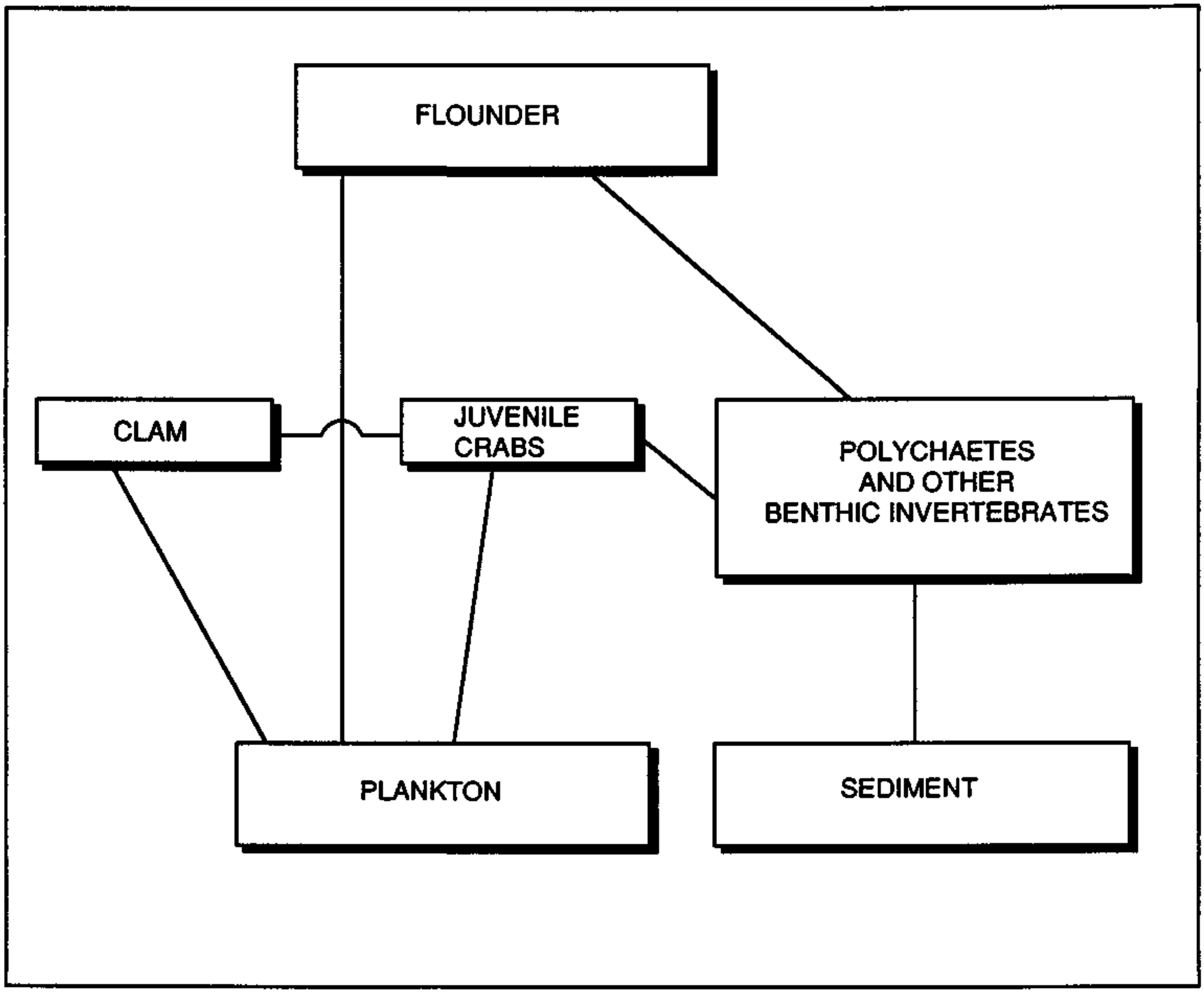


FIGURE 2- 5
FLOUNDER FOOD CHAIN
NEW BEDFORD, MASSACHUSETTS

contaminated food, desorption, metabolism, excretion, and growth. These processes are regulated by the physical/chemical characteristics of PCBs and by the physiological processes of the biota.

The food-chain model is designed to predict residue concentrations in species consumed by humans; therefore, it is a component of the public health risk assessment, as well as the ecological risk assessment. Because there are relatively few data available on the effect of residue values on aquatic biota, it is not possible to use the model results directly in the ecological risk assessment. The model does not include provisions for modifying any of the physiological processes as the organisms become stressed due to increasing body burdens of contaminants. However, it is necessary to consider toxic effects due to residue values as part of the risk assessment (see Section 4.0).

Also of importance for the risk assessment is the observation, based on calibration and validation of the food-chain model, that consumption of PCB-contaminated food may account for the majority (up to 95 percent) of PCB residue concentrations in aquatic species in New Bedford Harbor, although other investigators consider this figure unreasonably high for all but top predators (Hansen, 1990). Therefore, although there are insufficient data to evaluate this pathway quantitatively, it must be considered in some way if the risk assessment is to reflect actual effects on aquatic biota in New Bedford Harbor. This aspect of ecological risk is discussed in Section 4.0.

The mean levels (and ranges) of PCB tissue concentration found in organisms in the New Bedford Harbor area are summarized in Table 2-3, which is based on levels found in samples collected during the Battelle cruises of 1984, 1985, and 1986. These data indicate that PCB tissue residue concentrations are correlated with the levels of PCBs found in the New Bedford Harbor sediment and water column. For the six species comprising varied trophic levels and habitat preferences, highest tissue burdens were found in organisms collected from the inner harbor; levels decreased in successive areas in the outer harbor. The highest tissue levels were observed in polychaete worms, which are in direct and continuous contact with highly contaminated sediment. Winter flounder (*Pseudopleuronectes americanus*) also had relatively high whole-body tissue levels, perhaps reflecting its position in the marine food web and its habit of lying partially covered by bottom sediments.

TABLE 2-3
WHOLE-BODY CONCENTRATIONS OF TOTAL PCBS (PPM) IN ORGANISMS
COLLECTED FROM NEW BEDFORD HARBOR

NEW BEDFORD HARBOR

	LOCATION[1]			
SPECIES	AREA 1	AREA 2	AREA 3	AREA 4
American Lobster				
Minimum	---	0.195	0.042	0.017
Mean	1.131[2]	0.568	0.213	0.064
Maximum	---	1.235	0.351	0.176
Winter Flounder				
Minimum	3.138	0.926	0.515	0.123
Mean	7.992	2.853	2.138	0.777
Maximum	20.230	8.067	6.349	2.616
Mussel				
Minimum	1.467	1.461	0.254	0.008
Mean	2.262	3.874	0.266	0.023
Maximum	2.962	6.204	0.278	0.039
Quahog				
Minimum	0.200	0.010	0.026	0.200
Mean	5.300	1.777	1.200	0.300
Maximum	2.121	1.182	0.478	0.137
Green Crab				
Minimum	0.071	0.067	0.624	0.020
Mean	0.398	0.184	0.976	0.048
Maximum	0.725	0.301	1.329	0.077
Polychaetes				
Minimum	---	---	0.096	0.182
Mean	12.972[2]	1.654[2]	0.392	0.486
Maximum	---	---	0.689	0.790

NOTES:

[1] Locations correspond to Fishing Closure Areas (see Figure 1-2).
[2] Only one value available.

SOURCE: New Bedford Harbor Data Base

Table 2-4 summarizes the ranges of whole-body metals concentrations detected in organisms in the New Bedford Harbor area. The tissue residue levels of metals did not show general trends in contaminant concentrations between areas or between species. Overall, cadmium was detected at concentrations lower than either copper or lead. Copper concentrations were highest in crustaceans (i.e., crabs and lobsters), which probably reflects their copper-based heme system.

TABLE 2-4
RANGE [1] OF TOTAL WHOLE-BODY METALS IN NEW BEDFORD HARBOR BIOTA

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

ORGANISM	CADMIUM (ppm)	n[3]	COPPER (ppm)	n[3]	LEAD (ppm)	n[3]
Lobster	0.002NC	2	0.11-24.9	2	0.223-1.29	2
	0.002-0.703	16	20.778-46.814	16	0.106-3.034	16
	0.001-0.538	14	17.997-50.945	14	0.021-1.124	14
	0.002-0.588	21	15.788-62.663	21	0.029-0.842	21
Winter Flounder	0.004-0.014	23	0.692-11.147	23	0.215-3.336	22
	0.002-0.019	27	0.618-19.847	27	0.154-4.523	27
	0.002-0.012	17	0.691-51.642	17	0.099-2.728	17
	0.003-0.099	22	0.480-43.9	22	0.089-6.84	22
Mussel	0.242-0.326	9	1.948-2.49	9	0.293-1.41	9
	0.229-0.271	9	1.895-2.779	9	0.237-1.17	9
	0.326-0.397	6	0.726-0.841	6	0.367-0.647	6
	0.145-0.209	6	0.727-1.081	6	0.134-0.308	6
Quahog	0.087-0.356	18	3.727-8.302	18	0.58-1.901	18
	0.209-0.329	18	1.47-4.055	18	0.488-0.981	18
	0.12-0.381	18	1.302-2.713	18	0.208-3.463	18
	0.119-0.495	10	1.225-2.239	10	0.098-1.720	10
Green Crab	0.075-0.105	5	53.418-262.475	5	4.292-29.768	5
	0.027-0.095	4	12.1-52.897	4	1.45-6.908	4
	0.081[2]	1	201[2]	1	30.6[2]	1
	0.057	3	180.231[2]	3	13.824	3
Polychaetes	NA		NA		NA	
	NA		NA		NA	
	0.065-0.188	6	2.36-6.37	6	0.467-3.979	6
	0.111[2]	3	7.708[2]	3	1.076[2]	3

NOTES:

1 Each value represents the mean of several organisms within one size class
2 Only one value available
3 Total number of organisms sampled in each area
4 Areas correspond to Fisheries Closure Areas
NA = Not Available

3.0 ECOTOXICITY ASSESSMENT

The ecotoxicity assessment is a two-step process consisting of a compilation and evaluation of available toxicological information, and a synthesis of the information to provide a quantitative assessment of concentration/response data. Available toxicological information, some of which is presented herein, strongly supports the conclusion that PCBs in the marine environment represent a potential threat to biota, and provides additional information necessary to determine the nature and severity of actual or potential adverse effects associated with exposure. Although additional toxicological studies would be useful, the data available are sufficient to allow a quantitative estimation of the risk from contaminant exposure for four of the five groups discussed in Section 2.0. For the remaining group, the polychaete worms, the lack of available data precludes development of good quantitative concentration/response relationships. The concentration/response relationships developed herein will be combined with the exposure concentrations from Section 2.0 to provide the quantitative estimate of risk.

3.1 ECOTOXICITY PROFILES

3.1.1 PCBs

PCBs belong to a class of chemically stable, multi-use industrial chemicals that have been widely distributed in the New Bedford Harbor ecosystem. Electrical component manufacturers in New Bedford used PCBs in transformers and capacitors as dielectric insulating fluids resistant to fire. Discharge of PCBs into the harbor has resulted in contamination of the sediment, water, and biota in the area. Aspects of the structure, fate, and transport of PCBs with importance for determination of ecological risk are discussed in Subsection 1.3.

Adsorption to organic material in sediment is probably the major fate in the marine and estuarine environments of at least the more heavily chlorinated PCBs. Once bound, PCBs may persist for years, with slow desorption providing continuous exposure to the surrounding environment. Because PCBs are persistent in the environment and are lipophilic compounds, they are bioaccumulated (EPA, 1980b). The potential for bioaccumulation of an Aroclor mixture, as with other aspects of the biochemical behavior of PCBs, is related to the percentage of chlorine, with the BCF value generally increasing with higher chlorine content (Callahan et al., 1979). PCBs may be degraded by microorganisms (mainly the mono-, di-, and tri-chlorinated congeners) and by photolysis by ultraviolet light (mainly PCBs with five or more chlorines). Biodegradation rates and mechanisms appear to be specific to individual isomers and it is impossible to generalize about the overall rate for complex mixtures, except that many Aroclors persist for years or decades in the environment. Photolysis is extremely slow, but it may be a significant degradation pathway (EPA, 1980b).

EPA derived an AWQC for the protection of marine organisms for PCBs of 0.03 ug/L (parts per billion [ppb]). This value is based on laboratory-derived BCFs and was established to ensure that PCB burden in edible fish tissue (i.e., the final residue value [FRV]) would not exceed the former FDA tolerance level of 5.0 milligrams per kilogram (mg/kg) and not necessarily to protect ecological receptor organisms (EPA, 1980c). A recalculation of the criteria based on the new tolerance level value of 2.0 mg/kg would establish the new criterion at 0.012 ug/L (ppb); however, this change has not yet been made.

FDA tolerance levels are set to be protective of public health, but are based in part on economical and technical considerations. However, data from acute and chronic toxicity tests using Aroclors indicate that neither acute nor chronic toxicity should occur at the AWQC of 0.03 ug/L.

Marine AWQC, based on final toxicity values, are established to be protective of 95 percent of saltwater species. For PCBs, the AWQC document does not derive final acute or chronic values because determination of acute toxicity concentrations is problematic for PCBs (acute values are often in excess of maximum solubilities); minimum data criteria are not satisfied; and differing toxicities are demonstrated by the various PCB Aroclors and congeners (EPA, 1980b). Therefore, the saltwater AWQC for PCBs is based on the FRV, and is intended to protect the use of marine species as seafood rather than the species themselves, although it is considered sufficiently protective of the organisms as well. As such, these criteria serve as a tool to make general comparisons between the observed water column concentrations in New Bedford Harbor and toxicity information. However, site-specific ecotoxicity data provide a more definitive measure of the potential adverse effects of PCBs to marine organisms in New Bedford Harbor.

Tables B-1, B-2, and B-3 in Appendix B summarize available PCB ecotoxicity data, including acute and chronic toxicity data, as well as bioconcentration data for saltwater species discussed in the toxicological evaluation. Although PCBs have been shown to be acutely toxic to aquatic organisms, the actual exposure concentrations are unknown because the reported concentrations for the acute toxicity tests exceeded solubilities for some portion of PCB isomers, and the complex physical behavior of PCB mixtures makes cross-study comparisons difficult.

Based on the summarized acute and chronic toxicity data on PCBs, marine fish as a group are sensitive to the effects of PCB exposure. Chronic effects observed for marine fish include reduced hatching of embryos, reduced survivorship of fry, lethargy, fin rot, and decreased feeding, as well as mortality. Crustaceans are also quite sensitive, with acute effects being observed at exposures as low as 1 ug/L. The observed effects after chronic exposure for crustaceans include molt inhibition, dispersion of melanin in shells, altered metabolic state, and avoidance (Table B-2). Mortality has also been observed for crustaceans after chronic exposure.

Mollusks as a group are generally not as sensitive to PCB exposure as marine fish and crustaceans; however, reduced growth was observed at an exposure of 5 ug/L. Reduced growth rates are also observed in alga exposed to PCBs. Reduced cell division, reduced carbon dioxide uptake, and even no growth have been observed in alga after chronic exposure to PCBs. When populations of more than one algae species are exposed to PCBs, changes in species ratios and decreased diversity in the communities are observed. Overall PCB toxic effects are varied and at low concentrations. Toxic effects have been reported at concentrations of PCBs higher than the solubilities of the compounds.

BCFs for marine organisms are relatively high, ranging from 800 to greater than 670,000 (EPA, 1980b). Field and Dexter summarized available data for bioaccumulation from PCB-contaminated sediment with ratios ranging to 20 (Field and Dexter, 1988). These high factors would be predictable based on the lipophilic nature of PCBs. BCFs vary depending on several factors, including the level of total organic carbon (TOC) in the sediment and the length of exposure. BCFs vary among species and for different congeners. In general, the factors will be higher for species with greater amounts of fatty tissue. For congeners, the highest factors appear to occur among the congeners with five and six chlorine atoms; the lowest among those with eight and nine atoms (Lake et al., 1989).

3.1.2 Copper

Copper is a necessary nutrient for plants and animals; however, it is toxic at higher concentrations (EPA, 1985a). The copper ion is highly reactive and complexes with many inorganic and organic constituents of natural waters (EPA, 1985a). Hydrous iron and manganese oxides can effectively remove almost all free copper from the water column (Lee, 1975); and sediment/clay complexes, carbonates, and organic acids are all similarly effective under particular conditions. Most organic and inorganic copper complexes and precipitates appear to be much less toxic than free cupric ion.

Relatively few marine toxicological data are available for copper. However, mollusks and phytoplankton appear to be most sensitive to copper. Tables B-4 and B-5 in Appendix B summarize the toxicity data available for marine organisms. Copper has been shown to be acutely toxic to embryos of the blue mussel (Mytilus edulis) at 5.8 ug/L (Martin et al., 1977), and several diatom and marine alga species are sensitive to copper in the 1-to-10-ppb range. In fact, copper has been historically used as an aquatic herbicide and as a molluscicide to control schistosomiasis. Mean lethal concentration (LC_{50}) values for tests on winter flounder embryos (Pseudopleuronectes americanus) and the American lobster (Homarus americanus) were 130 and 69 ug/L, respectively (EPA, 1985a).

The only chronic data available for marine organisms are for Mysidopsis bahia; EPA established a chronic value of 54 ug/L based on lifecycle tests with this species. Various phytoplankton, polychaete worms, and mollusks have been shown to bioaccumulate copper with BCF values ranging from less than 100 to over 20,000. The marine chronic AWQC was established by EPA at 2.9 ug/L (ppb).

3.1.3 Cadmium

Although cadmium is insoluble in water, its chloride and sulphate salts readily solubilize. Humic acids and, to a lesser extent, hydrous iron and manganese oxides, appear to be primarily responsible for determining the extent of adsorption to sediment, while increased acidity and oxygenation tends to amplify desorption rates and subsequent bioavailability (Eisler, 1985; and Forstner, 1983). In addition, increasing salinity appears to mitigate the toxicological impact of this contaminant (EPA, 1985b). Tables B-6 and B-7 in Appendix B summarize the available saltwater ecotoxicity data for cadmium.

In general, freshwater species are considerably more sensitive to cadmium poisoning than marine species (Eisler, 1985). Among marine organisms, invertebrates are most sensitive to cadmium toxicity, with acute test results ranging from 41 to 135,000 ug/L for Mysidopsis bahia and an oligochaete worm, Monophylephorus cuticalcatus, respectively (EPA, 1985b).

Sublethal effects, including growth retardation, physiological disruptions, and alteration of oxygen consumption and respiratory rates, have been observed in marine organisms exposed to ambient cadmium concentrations on the order of 0.5 to 10 ug/L (Eisler, 1985).

Marine organisms can readily bioconcentrate cadmium, and BCF values over 2,000 have been recorded in some polychaete worms and mollusks (EPA, 1985b). However, reported BCFs for the lobster (Homarus americanus) and a marine fish, Fundulus heteroclitus, were 21 and 15, respectively (Eisler, 1985). EPA derived a chronic AWQC of 9.3 ug/L for the protection of marine organisms for cadmium.

3.1.4 Lead

Lead is most soluble under aqueous conditions characterized by low pH, low organic content, low particulate matter, and low concentrations of the salts of calcium, cadmium, iron, manganese, and zinc (Eisler, 1988). Most lead entering aquatic environments is quickly precipitated to bed sediments, and is released only under specific conditions (Demayo et al., 1982).

Relatively few toxicological data for marine species are available, with chronic-level effects observed in some organisms, particulary phytoplankton, in the 1-to-10-ug/L range. The plaice, Pleoronectes platessa, was acutely sensitive to tetramethyl lead at 50 ug/L (Eisler, 1988); a lifelong maximum acceptable toxicant concentration (MATC) between 17 and 37 ug/L was calculated for Mysidopsis bahia.

BCFs for lead in marine organisms ranged from 17.5 to 2,570 for the quahog (Mercenaria mercenaria) and the blue mussel (Mytilus edulis), respectively (EPA, 1980b). However, there is no evidence to indicate that lead is transferred through aquatic food chains (Eisler, 1988).

Tables B-8 and B-9 in Appendix B summarize available ecotoxicological data specific to the effects of lead exposure to marine organisms. Based on these data, EPA derived a chronic AWQC of 5.6 ug/L for the protection of marine organisms for lead.

3.2 EFFECTS EVALUATION

3.2.1 Methods

PCB and metals effects curves were constructed for the four taxonomic groups (i.e., marine fish, crustaceans, mollusks, and alga) for which ecotoxicity data were available. Data on benchmark effects were summarized, and the mean and variance of these data were used in the joint probability analysis to estimate risk, and to generate cumulative frequency probability curves. The curves provide an evaluation of probability of effect at various contaminant concentrations.

The standard acute benchmark for evaluating the acute response of an aquatic organism to the environmental concentration of a toxic contaminant is the 96-hour median LC_{50} (EPA, 1982; and ASTM, 1984). However, for purposes of risk assessment, the acute benchmark is not appropriate because the organisms are assumed to be exposed for periods longer than 96 hours. A more appropriate benchmark is the MATC, which is the threshold for significant effects on growth, reproduction, or survival (EPA, 1982; and ASTM, 1984). The benchmark is based on the most sensitive response of the organism to the contaminant in question.

Few MATC data are available for marine organisms, and the research that has been performed is limited with respect to both contaminant type and test organisms used. There are insufficient MATC data for PCBs to generate distributions for any of the taxonomic groups of interest. For this risk assessment, MATCs for the four taxonomic groups were developed using a method described by Suter and Rosen (Suter et al., 1986; and Suter and Rosen, 1986). This method uses an errors-in-variables regression model to predict a toxicological endpoint (in this case, the MATC) based on an extrapolation from existing endpoints for similar organisms. The regression equations used were established based on several large aquatic toxicological data bases (Suter and Rosen, 1986). For example, the model allows extrapolation from the LC_{50} of one species to the LC_{50} of another; similar extrapolations can be performed between LC_{50}s and MATCs. Therefore, a regression equation can be developed that has a coefficient (slope) and constant (intercept) that characterizes a between-taxon LC_{50} relationship or a within-taxon relationship between LC_{50}s and MATCs.

The errors-in-variables approach considers the following characteristics of toxicity data that a linear least-squared model would not address: (1) the observed values of both the independent (X) and dependent (Y) variables have inherent variability and are subject to measurement error; (2) the independent variable is not a controlled variable; and (3) the values assumed by (X) and (Y) are open-ended and non-normally distributed (Ricker, 1973). This method allows for quantification of uncertainty from interspecific differences in sensitivity, and the variability of the relationship between acute and chronic effects of contaminants. The uncertainty is quantified in the variances that result from the extrapolation. This variance is then applied in the joint probability analysis, which uses the estimated toxicological benchmark value and its variance, along with an EEC and its variance to estimate risk of chronic effects to a particular group of organisms. The final risk estimate is interpreted as the probability of an adverse effect being realized in a typical member of the group in question, given the variability in contaminant levels.

This model and its application are discussed in more detail in Section 4.0. MATCs for four groups of organisms (i.e., marine fish, crustaceans, mollusks, and alga) representative of the range of organisms found in New Bedford Harbor were developed using this approach. The taxonomic groupings were necessary to facilitate the application of the errors-in-variables methodology, because extrapolations are within or between taxonomic levels. A comparable analysis by strict trophic and/or habitat classification by this method would not have been possible because multiple taxa groups would be a part of such an analysis. However, these groups generally also define a primary means of exposure (e.g., via water or sediment) and, therefore, allow consistency with respect to applying exposure concentrations to provide a risk estimate.

For marine fish, crustaceans, and mollusks, MATCs were developed using the errors-in-variables methodology. For the algae, a chronic effect concentration was developed based on the existing toxicological data. The data used for the overall MATC development for alga and mollusks came from the AWQC and Eisler documents (EPA, 1980a, 1980b, and 1980c; and Eisler, 1986). These data sets were also used as the source of the LC_{50} for the sheepshead minnow and the MATC for Daphnia magna used in extrapolations for marine fish and crustacean MATCs.

All data used for the regressions were log-transformed. Test results reported as greater than or less than a particular value were not used. When replicate data were available for a chemical-species pair, the geometric mean for the species was used. Use of the geometric rather than the arithmetic mean for replicate tests is consistent with EPA methods for AWQC development (EPA, 1982).

3.2.2 Application and Results

3.2.2.1 Marine Fish

Development of the MATCs for marine fish was based on previously reported relationships. Suter and Rosen performed extrapolations between the LC_{50}s for sheepshead minnow (*Cyprinodon variegatus*) and LC_{50}s for marine species, as well as derivation of the errors-in-variables relationship between marine fish LC_{50} and marine fish MATCs (Suter and Rosen, 1986). The slope, intercept, and variance from these extrapolations used in the MATC development and risk assessment for marine fish in New Bedford Harbor are presented in Table 3-1.

The overall marine fish MATC for PCBs was created by a double extrapolation: first from the sheepshead minnow chronic LC_{50} for PCBs (0.93 ug/L) to a typical marine fish LC_{50} for PCBs

TABLE 3-1
PCB MATC ESTIMATES FOR ORGANISMS AT NEW BEDFORD HARBOR

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

TAXON	SLOPE	INTERCEPT	MATC	TOTAL VARIANCE
Marine Fish	0.97	0.03		
	0.98	-0.6	-0.601	1.021
Crustaceans	0.95	0.0	0.668	0.956
Mollusks	1.577	-0.456		
	0.98	-0.6	1.358	3.024
Algae			0.987	4.907

NOTES:

1. The basic regression equation that defines the extrapolation is Y = Intercept + (X * Slope), where X is the acute toxicological estimate and Y the extrapolated MATC value.

2. No extrapolation was done for algae; rather, chronic data were used to estimate the benchmark value for the taxon.

3. In cases where two sets of slope and intercept values are listed, the first set is for a LC50-to-LC50 extrapolation, and the second for the final LC50-to-MATC extrapolation.

4. All units expressed as Log (base 10) ug/L.

(0.99 ug/L), then to a marine fish MATC of 0.25 ug/L. The chronic LC_{50} value used as the starting point for these extrapolations was an early life stage test using Aroclor 1254. Similar testing with Aroclor 1016 produced similar responses only at concentrations above 10 ug/L. Other Aroclors are expected to fall generally within this range, and the lower value for Aroclor 1254 provides a conservative estimate of the toxicity of the actual mix of PCB congeners in New Bedford Harbor. The effect curve, which is a cumulative probability plot based on the MATC value and its variance, is shown in Figure 3-1.

Approximately 95 percent of the calculated MATC values for marine fish falls within a range of four orders of magnitude; chronic values in the literature, most of which are based on one of three species, span approximately half this range. This difference is largely a result of the procedure that uses the actual data as a sample from the universe of MATCs and generates a probability plot for all marine species in the taxon of interest. The actual range for species residing in New Bedford Harbor may well be smaller; however, there is no way of developing such a site- specific MATC with the available data.

The metal MATC values for marine fish were extrapolated using a relationship between the MATCs of the mysid, Mysidopsis bahia and the MATCs of fish developed by Suter and Rosen (Suter and Rosen, 1986). The extrapolations were from the mysid MATCs of 54, 5.5, and 25 ug/L for copper, cadmium, and lead, respectively. The MATCs derived for marine fish were 329, 32, and 150 ug/L for copper, cadmium, and lead, respectively.

The MATC effects curves are shown in Figures B-1, B-2, and B-3 in Appendix B. The slope, intercept, and variance from these extrapolations used in the MATC development and risk assessment for metals and marine fish in New Bedford Harbor are presented in Tables B-10, B-11, and B-12.

3.2.2.2 Crustaceans

The PCB MATC for crustaceans was obtained from the association between the MATC for the cladoceran (Daphnia magna) and MATCs for marine crustaceans developed by Suter and Rosen (Suter and Rosen, 1986). The slope, intercept, and variance developed in this errors-in-variables model are presented in Table 3-1. One extrapolation from the cladoceran MATC (5.14 ug/L) was required to derive the typical marine crustacean MATC of 4.66 ug/L. The MATC probability curve for crustaceans is shown in Figure 3-1.

A single extrapolation was required to develop the metal MATCs for crustaceans. These MATC values were extrapolated using a relationship between the MATCs of the mysid, Mysidopsis bahia, and the MATCs of crustaceans developed by Suter and Rosen (Suter

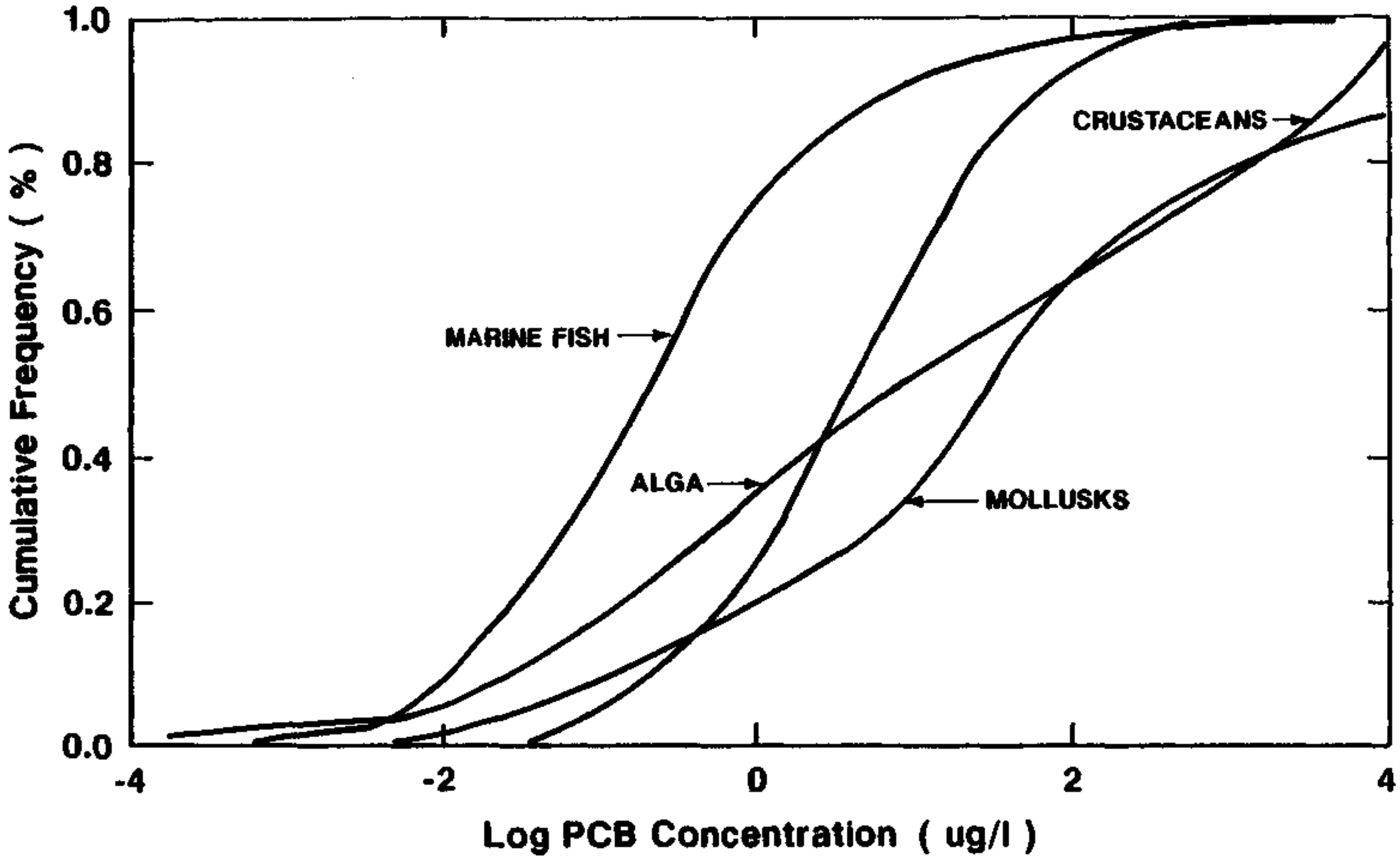


FIGURE 3-1
MATC CURVES FOR PCBs
NEW BEDFORD, MASSACHUSETTS

and Rosen, 1986). The extrapolations were from the mysid MATC values of 54, 5.5, and 25 ug/L for copper, cadmium, and lead, respectively. The extrapolated MATCs developed for crustaceans were 65.5, 10.5, and 35.3 ug/L for copper, cadmium, and lead, respectively. The slope, intercept, and variance from these models are shown in Tables B-10, B-11, and B-12 in Appendix B. The MATC curves for copper, cadmium, and lead are shown in Figures B-1, B-2, and B-3, respectively.

3.2.2.3 Mollusks

To develop the PCB MATC for mollusks, two extrapolations were needed. First, a relationship between the LC_{50}s for the mysid, Mysidopsis bahia, and LC_{50}s of mollusks was developed. The relationship between these species was used because the greatest number of matches between chemical-species pairs was available and, although there is no close taxonomic relationship, the mysid is a standard test species. Because there are no MATC data available for mollusks, an estimate of the MATC was performed by using the relationship between marine fish LC_{50}s and MATCs, on the assumption that the ratios between acute and chronic effects for marine fish and mollusks are similar. The slopes, intercepts, and variances used in this MATC development are shown in Table 3-1.

The mollusk LC_{50} of 99.61 ug/L was obtained by forward extrapolation from the mysid LC_{50} (36.0 ug/L). The estimated mollusk LC_{50} was then used to estimate the typical mollusk MATC (22.82 ug/L) based on the LC_{50}/MATC relationship for marine fish. The effects curve is shown in Figure 3-1. There is a large variance associated with this MATC due to the double extrapolation. Large variances were observed by Suter and Rosen for similar extrapolations between higher level taxonomic groups (Suter et al., 1986; and Suter and Rosen, 1986). Because the variance for the extrapolation from LC_{50} to MATC for marine fish is small, its use in this application may result in an underestimation of the variance associated with the MATC for mollusks.

As in the case of PCBs, limited data are available on metal MATCs for mollusks. To develop MATCs for mollusks, the same marine fish LC_{50}-to-MATC relationship was used as for PCBs, assuming that the ratios between acute and chronic effects for marine fish and mollusks are similar. The LC_{50}s used in this extrapolation were developed from values reported in the AWQC and Eisler documents (EPA, 1980a, 1980b, and 1980c; and Eisler 1985 and 1986). These data are compiled in Tables B-4 through B-9 in Appendix B. For each metal, the mollusk LC_{50} value used in the extrapolation is a geometric mean of the values reported for all mollusks.

The metal MATCs for mollusks were derived from the mollusk LC_{50} values of 72.4, 2,666, and 1,244 ug/L for copper, cadmium, and lead, respectively. The single forward extrapolation for each metal estimated the mollusk MATCs to be 16.7, 571, and 271 ug/L for copper, cadmium, and lead, respectively. The effects curves for the MATCs are presented in Figures B-1, B-2, and B-3 in Appendix B. The slope, intercept, and variance from these extrapolations are presented in Tables B-10, B-11, and B-12.

3.2.2.4 Polychaetes

There were sufficient acute toxicological data for the three metals to develop MATC estimates for polychaetes, using the crustacean LC_{50} and MATC extrapolation developed by Suter and Rosen (Suter and Rosen, 1986). In this case, it was assumed that the ratios between acute and chronic effects for crustaceans and polychaetes are similar. The LC_{50}s used in this extrapolation were developed from values reported in the AWQC and Eisler documents (EPA, 1980a, 1980b, and 1980c; and Eisler 1985 and 1986). Tables B-4 through B-9 in Appendix B summarize of the toxicological data used to develop MATC estimates for polychaetes. The polychaete LC_{50} for each metal is a geometric mean of the values reported for all polychaetes and oligochaetes.

The metal MATCs for polychaetes were derived from the polychaete LC_{50} values of 199, 9,682, and 10,691 ug/L for copper, cadmium, and lead, respectively. A single forward extrapolation for each metal was necessary to estimate the polychaete MATCs as 30.2, 1,276, and 1,409 ug/L for copper, cadmium, and lead, respectively. MATC curves for copper, cadmium, and lead are shown in Figures B-1, B-2, and B-3, respectively. The slope, intercept, and variance from these individual extrapolations are presented in Tables B-10, B-11, and B-12.

3.2.2.5 Algae

For the algal species at the New Bedford Harbor site, a benchmark concentration was developed using the geometric mean of the results from chronic tests as presented in the AWQC and Eisler documents (EPA, 1980; and Eisler, 1986). Although this value is not an MATC by definition, it is a reasonable best estimate of chronic toxicological effects of PCBs on algal species based on the limited data available. The benchmark concentration of 9.71 ug/L has a high amount of variance (4.44); this is due to the large amount of variability in reported responses to PCBs. The effects curve is shown in Figure 3-1.

For the metals, a geometric mean was developed from chronic effects data presented in the AWQC and Eisler documents (EPA, 1980a and 1980c; and Eisler, 1985 and 1988). The benchmark values derived were 12, 99.3, and 234 ug/L for copper, cadmium, and lead, respectively. The effects curves for the MATCs are shown in Figures B-1, B-2, and B-3 in Appendix B. Summary statistics for these benchmark concentrations are in Tables B-10, B-11, and B-12.

3.2.3 Evaluation of MATCs

Because of the limited amount of data available about the effects of PCBs and metals on marine organisms, the estimates of MATC or chronic effect benchmarks as used in this risk assessment have some uncertainty, which was quantified to some extent by the variances from the errors-in-variables extrapolations. The relative effect of this source of uncertainty may be observed graphically by comparison of the slope of the probability function for the MATC of each group in Figure 3-1. This uncertainty is also evident in the effect of the variance on results of the analysis of extrapolation error model used for risk characterization in Section 4.0. In all cases, the variance in the estimates for metal MATC values was not as high as for PCBs, primarily due to the fact that only one extrapolation was necessary.

Another area of uncertainty for these MATC estimates results from the need to perform extrapolations from a single species to a taxonomic group consisting of many species, some of which may be only distantly related. If the single species used in the extrapolation happens to be particularly sensitive to contaminants, the final estimate of the group MATC may be overly conservative. This is probably the case for the extrapolation from the sheepshead minnow to marine fish in general. The PCB LC_{50} for the sheepshead minnow (0.93 ug/L), the species used to develop most of the available data, is quite low, driving the marine fish MATC to a lower value than may be the case. However, other marine fish tested also have low LC_{50}s for PCBs.

4.0 RISK CHARACTERIZATION

Risk to marine organisms in New Bedford Harbor was evaluated for exposure to waterborne and sediment-bound PCBs and metals, as well as for consumption of PCB-contaminated food. Risk estimates for each environmental medium were evaluated by taxonomic group for each harbor zone described in Section 1.0, and overall ecosystem risk was assessed qualitatively from the individual risk estimates.

A quantitative uncertainty (or joint probability) analysis was performed by combining results of the analyses of exposure and ecotoxicity presented in the two preceding sections to develop probabilistic estimates of risk in New Bedford Harbor. In addition, risk to organisms exposed to dissolved contaminants in the water and directly to PCB-contaminated sediment was evaluated by comparing analytical data on existing contaminant levels with appropriate water and sediment criteria, and by examining the results of site-specific bioassays. Risk due to ingestion of PCB-contaminated food was evaluated by comparing the tissue burden levels detected in New Bedford Harbor biota to effect levels associated with reproductive impairment and pathological effects in marine fish.

4.1 JOINT PROBABILITY ANALYSIS

4.1.1 PCB Water Column Contamination

The probability functions for chronic effects due to dissolved PCBs in the water column for each of the four taxonomic groups with sufficient toxicological data to perform the analysis are shown co-plotted with the EEC probability functions for the Hot Spot and Zones 1 through 5 in Figures 4-1 through 4-4. Results of the joint probability analysis for each group using these two sets of curves are presented in Table 4-1. For the algae (see Figure 4-1), potential impacts are projected for each zone, particularly areas north of the Coggeshall Street Bridge (Zones 1 and 2, and the Hot Spot), where there is a 30 percent or greater probability that the average dissolved PCB concentration encountered by a typical marine algal species would exceed the respective chronic benchmark. Another way of expressing this effect would be as an impact on the most sensitive 30 percent of the various algal species used for the toxicity studies upon which the chronic effects curve was based and, therefore, are representative of taxa that might occur in the area. For Zones 3 and 4, the average concentration encountered would potentially impact 20 percent or less of the algal species; however, essentially the entire harbor north of the Hurricane Barrier has a high probability of impacting more than 5 percent of the algal species (i.e., a benchmark used by EPA in determining water quality criteria). Because of the wide range of sensitivities

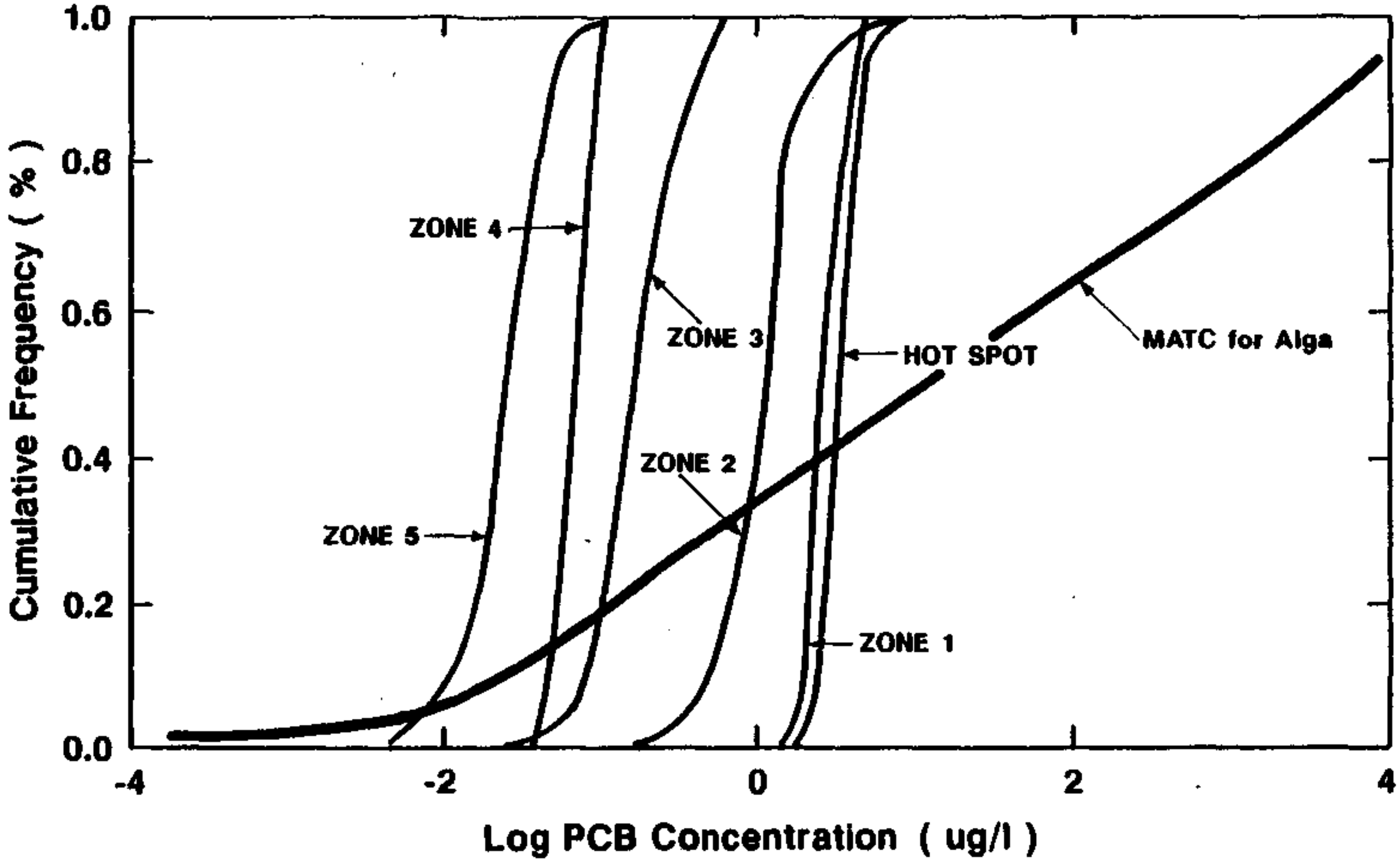


FIGURE 4-1
MATC FOR ALGA AND EECs FOR ALL ZONES, PCBs, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

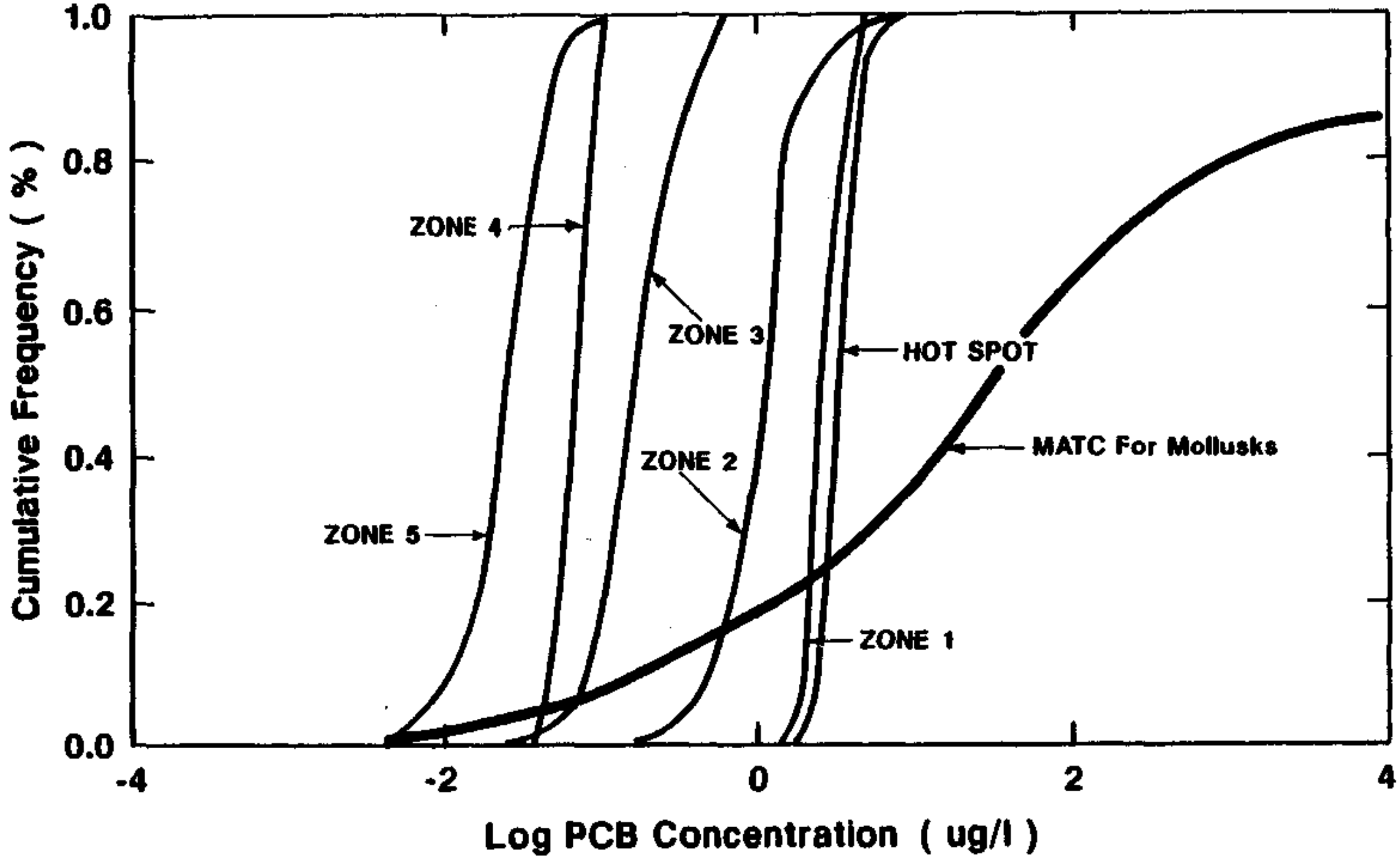


FIGURE 4-2
MATC FOR MOLLUSKS AND EECs FOR ALL ZONES, PCBs, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

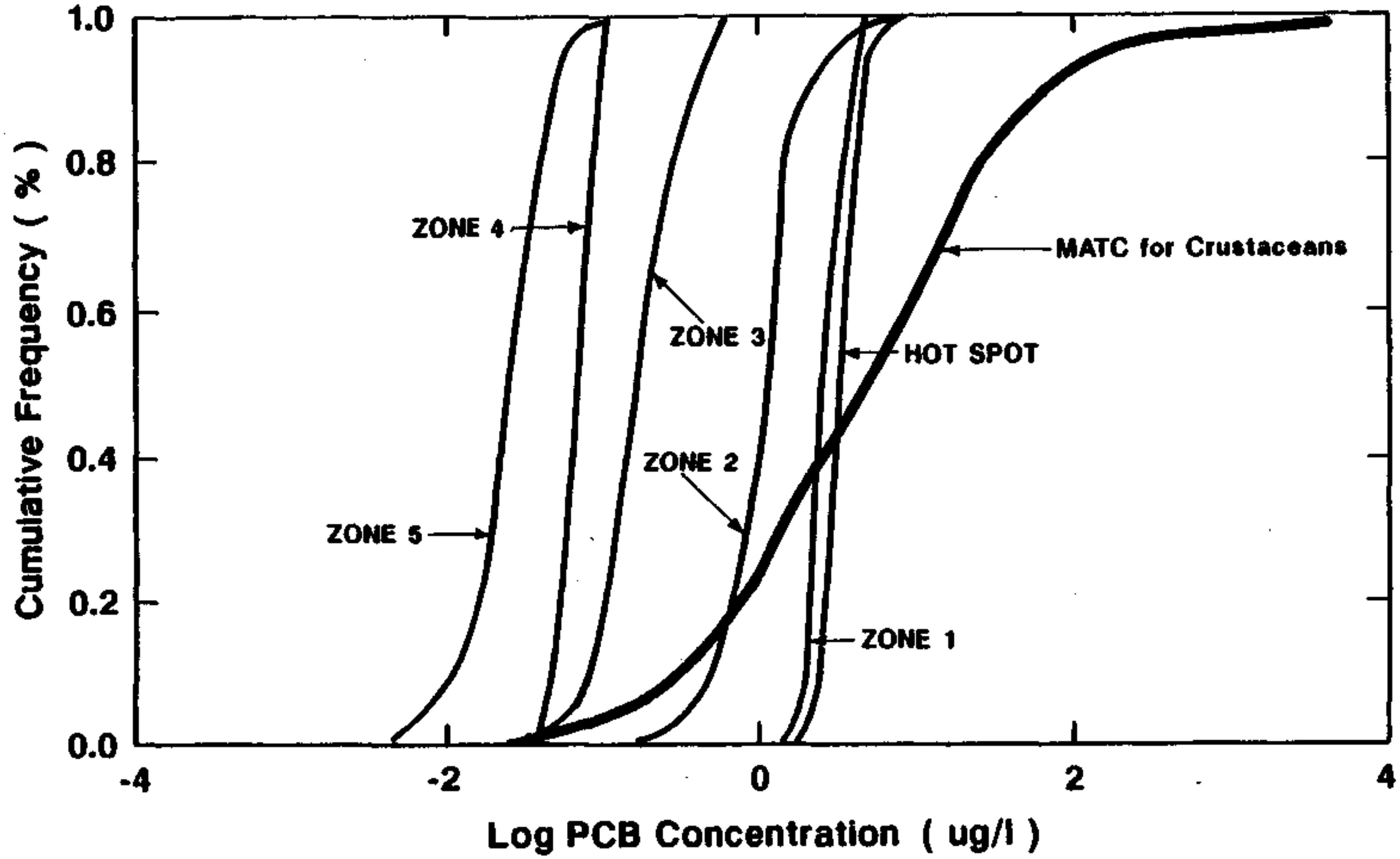


FIGURE 4-3
MATC FOR CRUSTACEANS AND EECs FOR ALL ZONES, PCBs, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

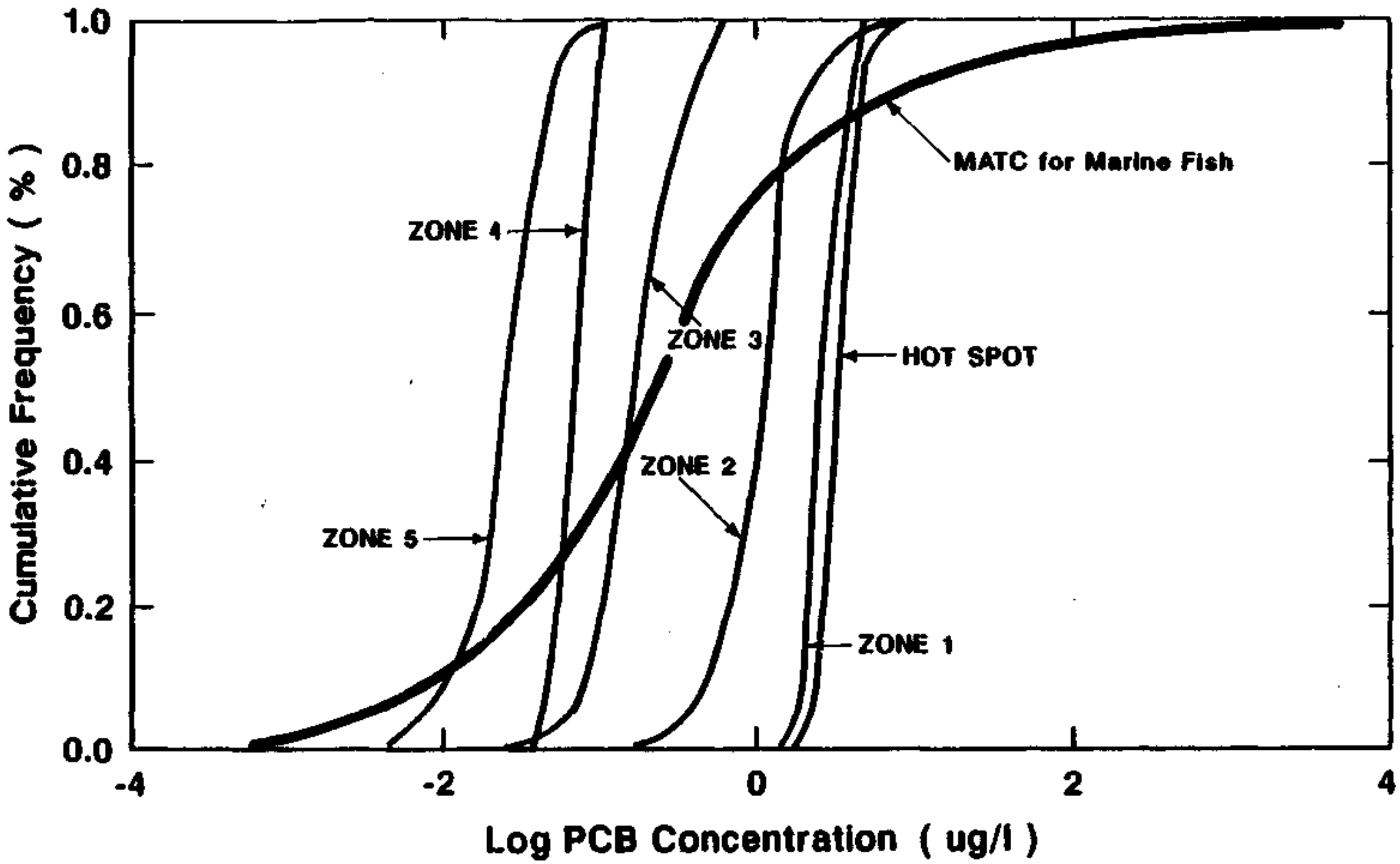


FIGURE 4-4
MATC FOR MARINE FISH AND EECs FOR ALL ZONES, PCBs, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

TABLE 4-1
CUMULATIVE PROBABILITY THAT THE EXPECTED EXPOSURE CONCENTRATION WILL EXCEED THE PCB MATC FOR THE PARTICULAR TAXON

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

HARBOR ZONE	MARINE FISH	CRUSTACEANS	MOLLUSKS	ALGAE
Hot Spot, Water Column	0.86	0.43	0.31	0.41
1. Water Column	0.84	0.40	0.29	0.40
2. Water Column	0.73	0.26	0.23	0.33
3. Water Column	0.42	0.07	0.11	0.21
4. Water Column	0.28	0.03	0.07	0.16
5. Water Column	0.16	0.01	0.04	0.12
Hot Spot, Pore Water	0.97	0.82	0.60	0.64
1. Pore Water	0.98	0.81	0.55	0.61
2. Pore Water	0.82	0.49	0.36	0.44
3. Pore Water	0.52	0.12	0.14	0.25
4. Pore Water	0.33	0.07	0.09	0.18
5. Pore Water	0.24	0.04	0.07	0.16

NOTES:

Probabilities calculated as the area under a normally distributed curve defined by a particular Z score, where Z = (Mean EEC - BM) / (Var EEC + Var BM)^2. Source: Suter et al., 1986.

EEC = Expected Environmental Concentration

BM = Benchmark, which in this application are the MATCs developed by extrapolation, in the case of Marine Fish, Crustaceans, and Mollusks. For Algae, the benchmark was based on available chronic toxicity data demonstrated by this taxonomic group (indicated by the slope of the chronic effects function), even the highest concentrations seen at the Hot Spot would not impact the least sensitive 50 percent of algal species.

Because of the similarity between the chronic effects probability curves, the effects for algal species generally are true for mollusks (see Figure 4-2). PCB concentrations above the Coggeshall Street Bridge would be expected to impact approximately 20 percent of the molluscan species; however, concentrations in the remainder of the harbor would not be expected to pose as great a threat to this group, and would likely impact less than 10 percent of the species.

The pattern of risk for crustaceans (see Figure 4-3) is markedly different from the preceding two groups because of the generally narrower range of sensitivities to PCB exposure, as indicated by the steeper slope of the MATC function. For the crustaceans, there is approximately a 40 percent likelihood that the typical PCB concentrations encountered in the Hot Spot and Zone 1 would be expected to exceed the MATC value of the typical crustacean. The slightly lower concentrations in Zone 2 would have a smaller yet still serious impact. Outside the Coggeshall Street Bridge, anticipated impacts on crustaceans are small, with concentrations projected to impact less than 5 percent of the species.

Because of their much greater sensitivity to dissolved PCBs, marine fish are the most heavily impacted group (see Figure 4-4). For this group, typical concentrations in the Upper Estuary are projected to impact more than 80 percent of the fish species, and even the tenth-percentile concentration would have nearly as large an effect. In Zones 3 and 4, the impact remains high, with concentrations projected to impact approximately 30 percent of the marine fish. This analysis indicates that marine fish are at high risk of impact due to chronic exposure to dissolved PCBs for the entire area inside the Hurricane Barrier.

The mean total PCB concentration in Zone 5 was below concentrations shown in laboratory studies to produce toxic effects. In addition, the exceedance probabilities for all taxonomic groups were in the 5- to 15-percent range, indicating that potential impacts of PCB contamination in this zone would be expected to be much less than the remainder of the study area, although still significant.

Figures 4-5 through 4-8 show the areal extent of the probability that chronic effects will be observed due to water column exposure to PCBs for the various taxonomic groups, based on the initial conditions concentration for each grid cell. The probability contours shown on these maps indicate general trends within each zone and should not be used to assess localized differences of chronic effects.

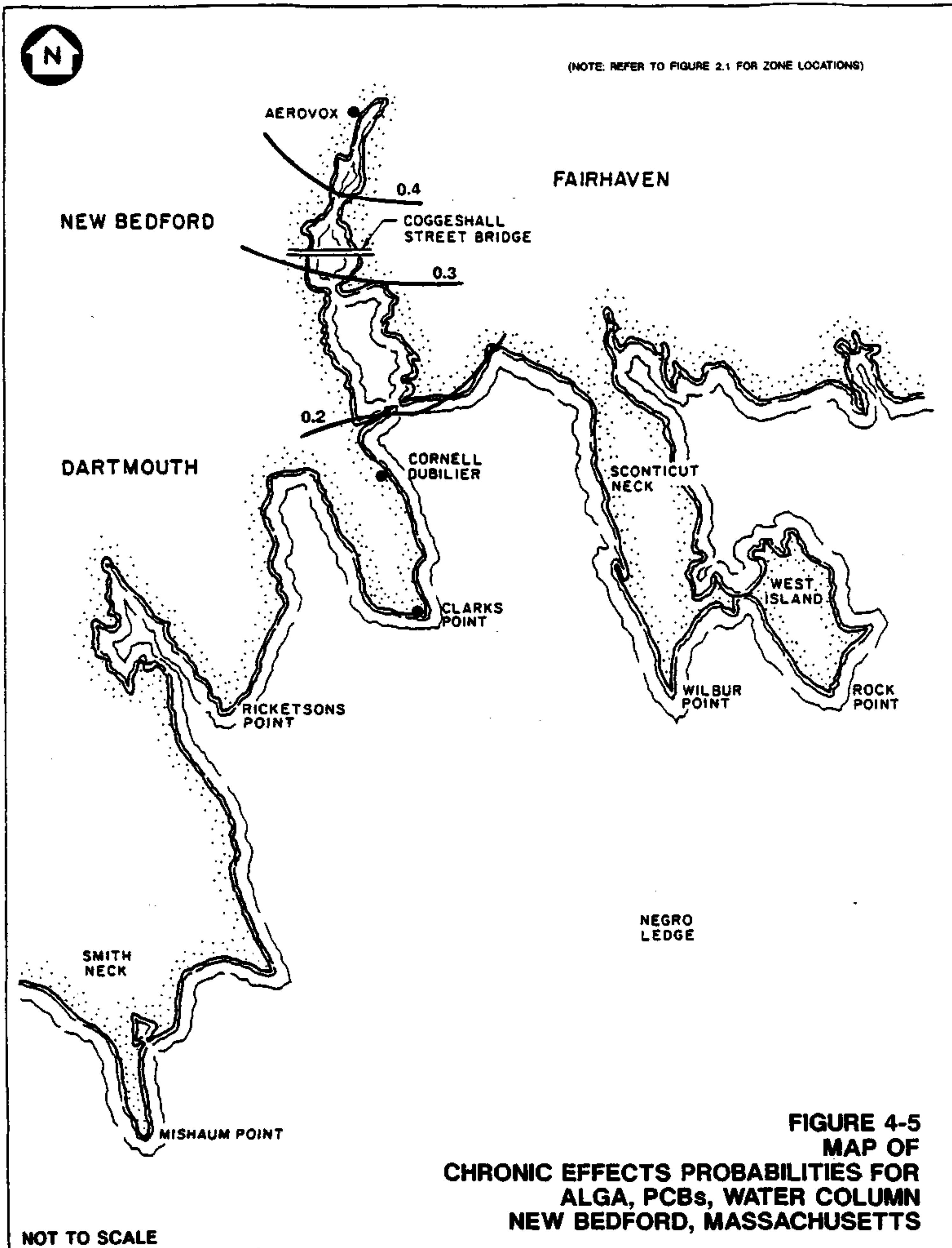


FIGURE 4-5
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
ALGA, PCBs, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

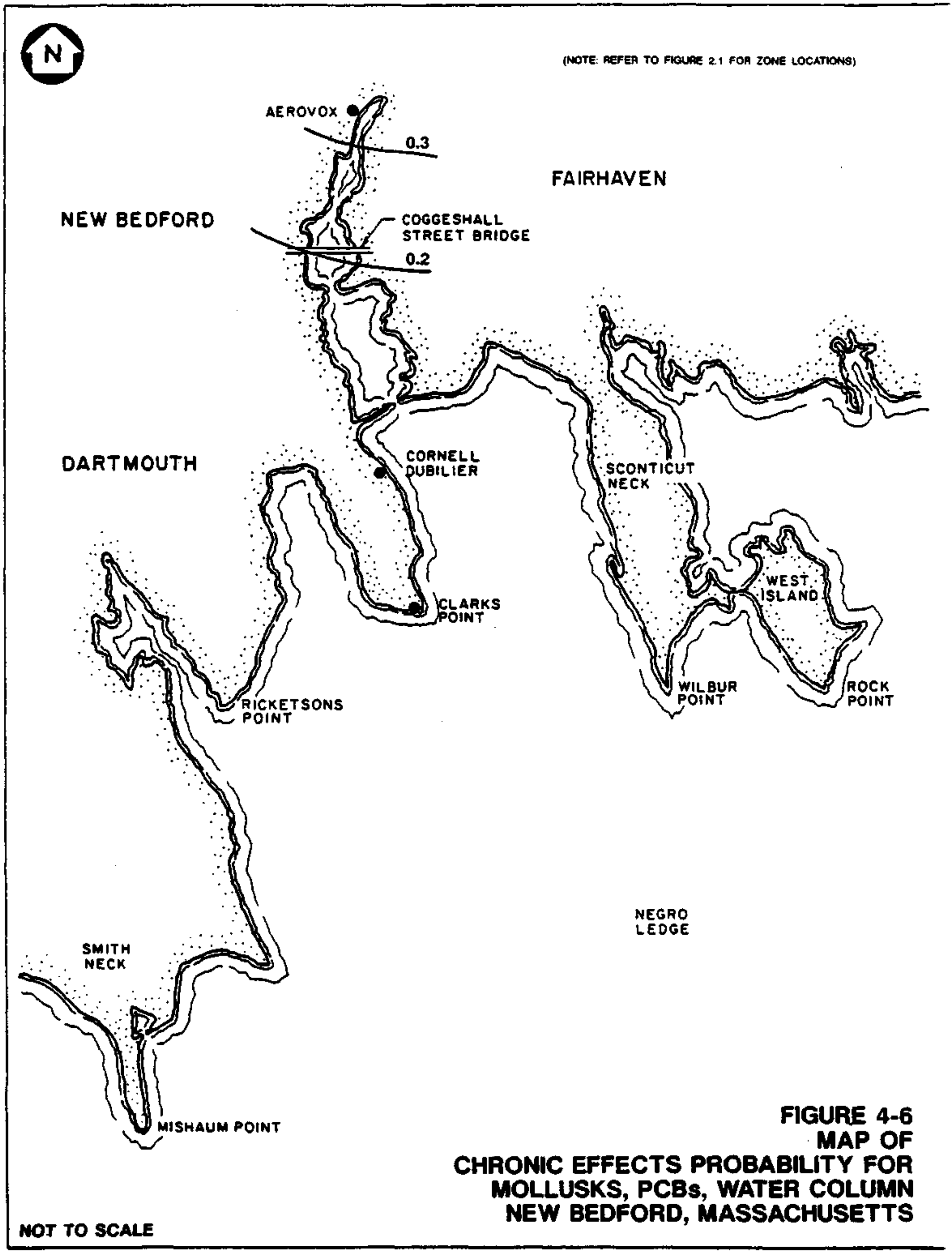


FIGURE 4-6
MAP OF
CHRONIC EFFECTS PROBABILITY FOR
MOLLUSKS, PCBs, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

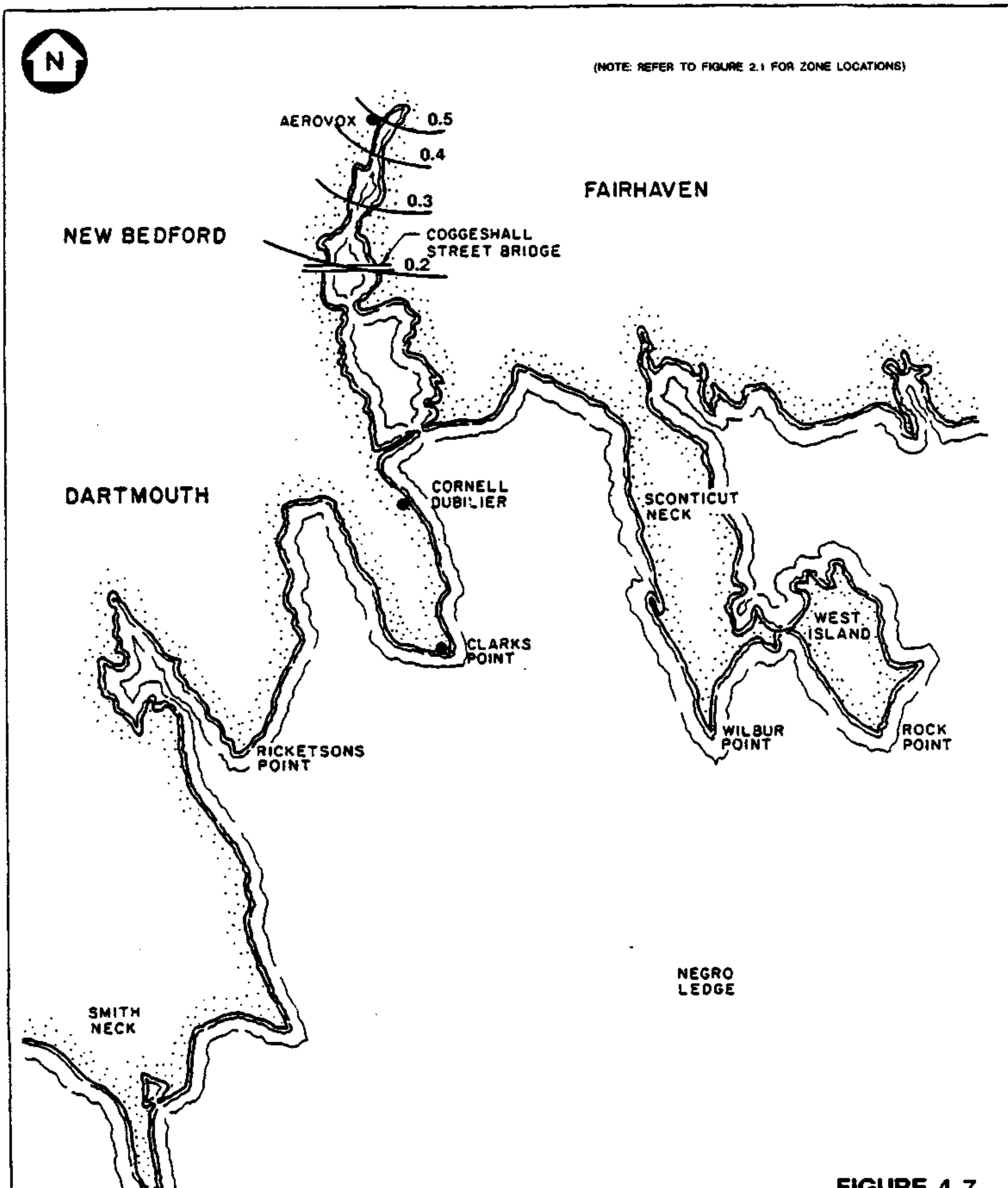


FIGURE 4-7
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR CRUSTACEANS, PCBs, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

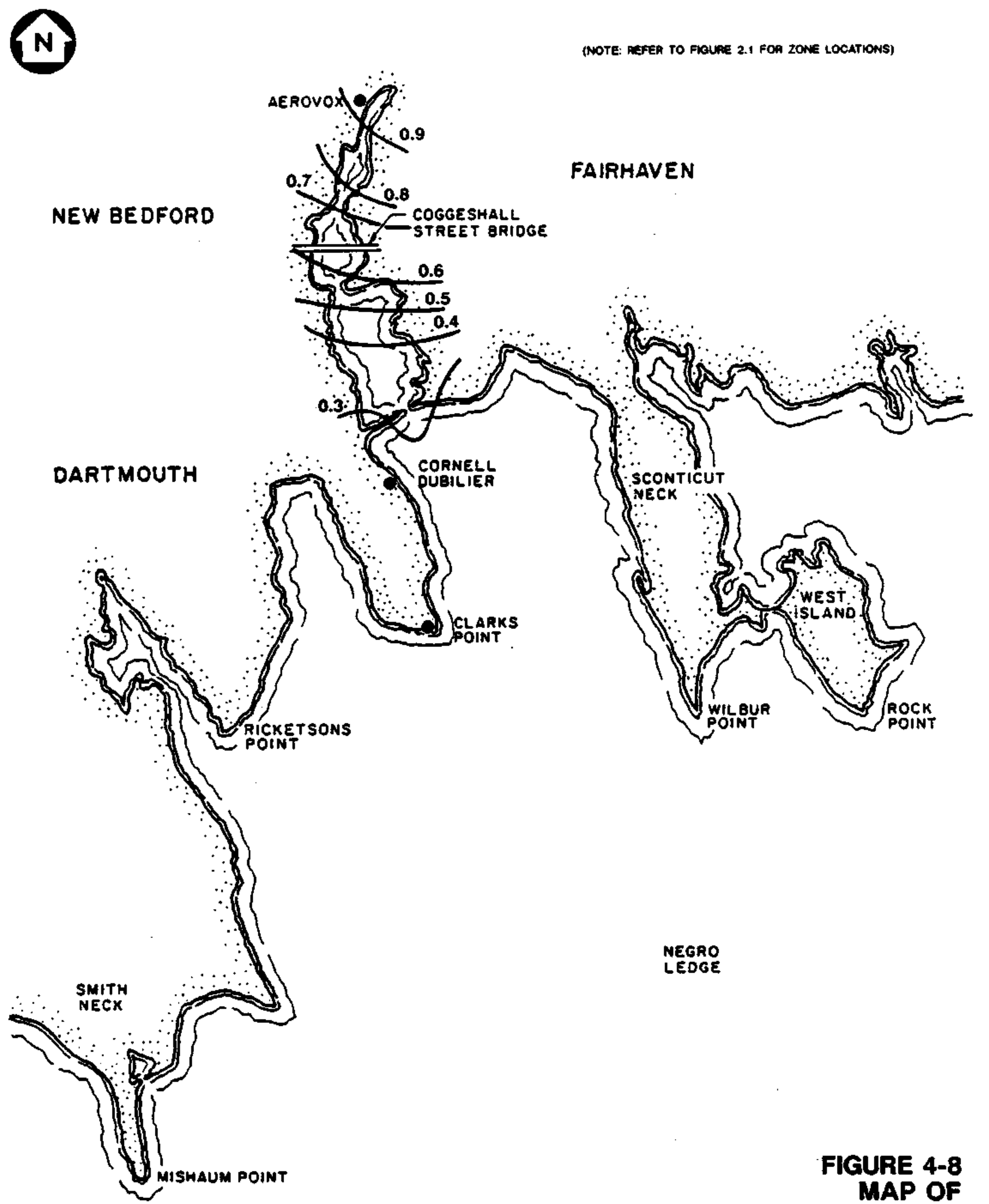


FIGURE 4-8
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR MARINE FISH, PCBs, WATER COLUMN NEW BEDFORD, MASSACHUSETTS

4.1.2 PCB Sediment Contamination

The risks previously discussed caused by water column contamination with dissolved PCBs occur ultimately as a result of contaminated bed sediment in the harbor and estuary, which provide a reservoir of PCBs that are desorbed and resuspended into the water column. Therefore, all risks in the system may be thought of as due to sediment contamination. However, throughout the risk assessment risks due to contaminated sediment are meant to include those risks that result from direct exposure to the sediment and its associated pore water, and not to overlying water contaminated from the sediment.

The exposure curves developed for the various harbor zones in this analysis represent the expected distribution of PCB contaminant levels in the pore water. Considerable effort has been devoted in the New Bedford Harbor project to the question of pore water concentrations as part of the modeling effort; however, no site-specific calculation of pore water PCB concentrations from sediment-bound concentrations has been developed. As discussed in Subsection 2.2.2.2, the mass transfer coefficients developed for calibration of the physical/chemical model were used as apparent K_ds to calculate pore water concentrations for this risk assessment. This approach results in pore water concentrations that are generally higher than the overlying water column concentrations.

In development of the food-chain model, pore water was assumed to be in equilibrium with the overlying water column; therefore, the water column concentrations were also used as pore water concentrations. It is probable that the actual concentrations experienced by benthic and demersal organisms will be between these two extremes; consequently, the developed exposure curves probably overestimate the actual exposure concentrations experienced by most species. As such, the risk probabilities should be considered conservative; however, in the absence of more specific data, a conservative approach is necessary.

MATC curves and EEC sediment (i.e., sediment pore water) curves are co-plotted for mollusks, crustaceans, and marine fish in Figures 4-9 through 4-11. Because they would not be expected to be exposed to sediment pore water, the evaluation was not conducted for algae. There is considerable variability in behavior and habitat preference among the species comprising all three taxonomic groups, and some species (e.g., pelagic fish, mussels, and copepods) would not be expected to have any direct contact with sediment pore water. However, insufficient data were available to construct separate MATC curves based on life history and, on the assumption that sensitivity to PCBs would not be expected to vary between benthic and pelagic members of a taxonomic group, the single MATC curve was used for each group. Consequently, chronic effects distributions for these three

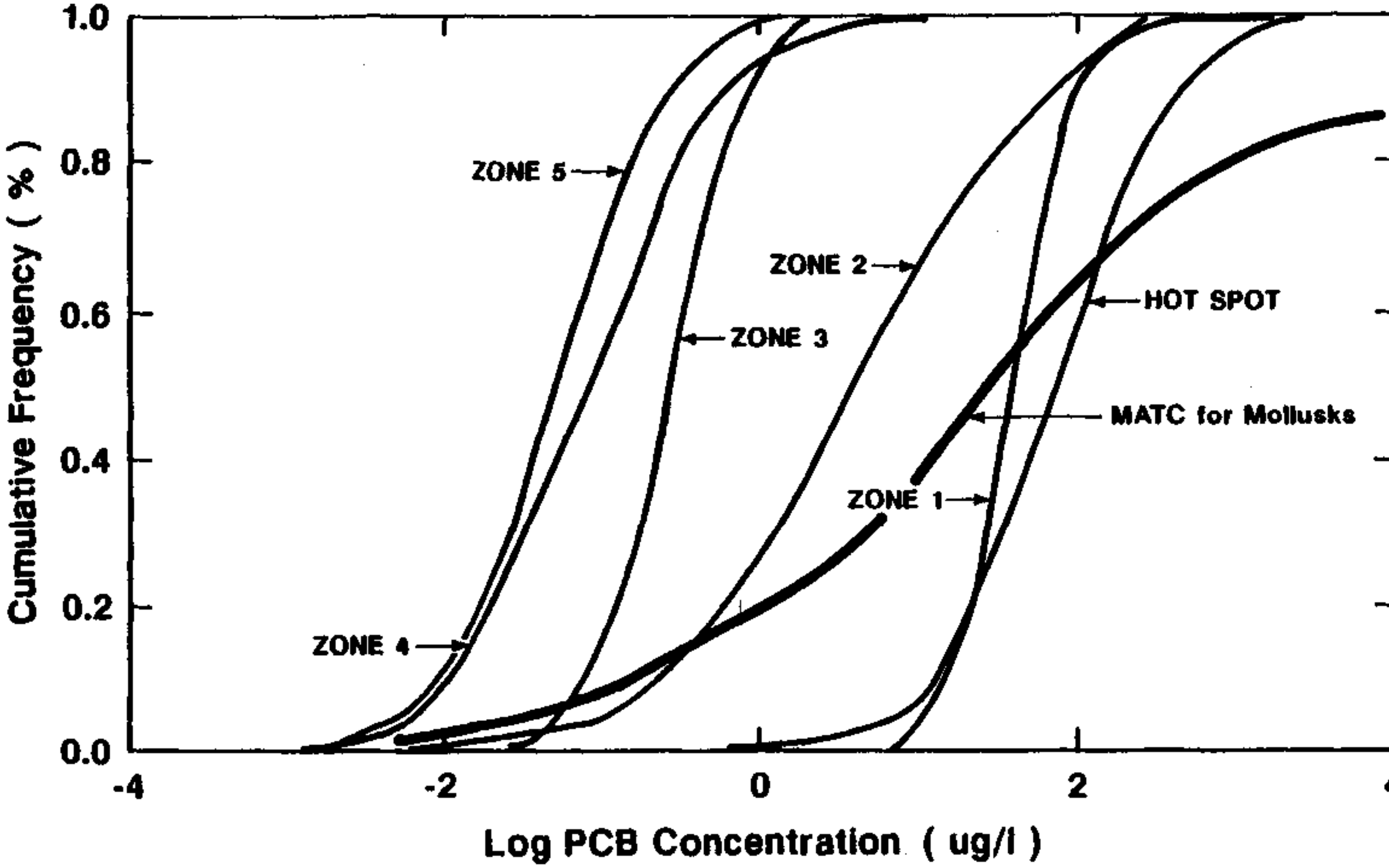


FIGURE 4-9
MATC FOR MOLLUSKS AND EECs FOR ALL ZONES, PCBs, PORE WATER
NEW BEDFORD, MASSACHUSETTS

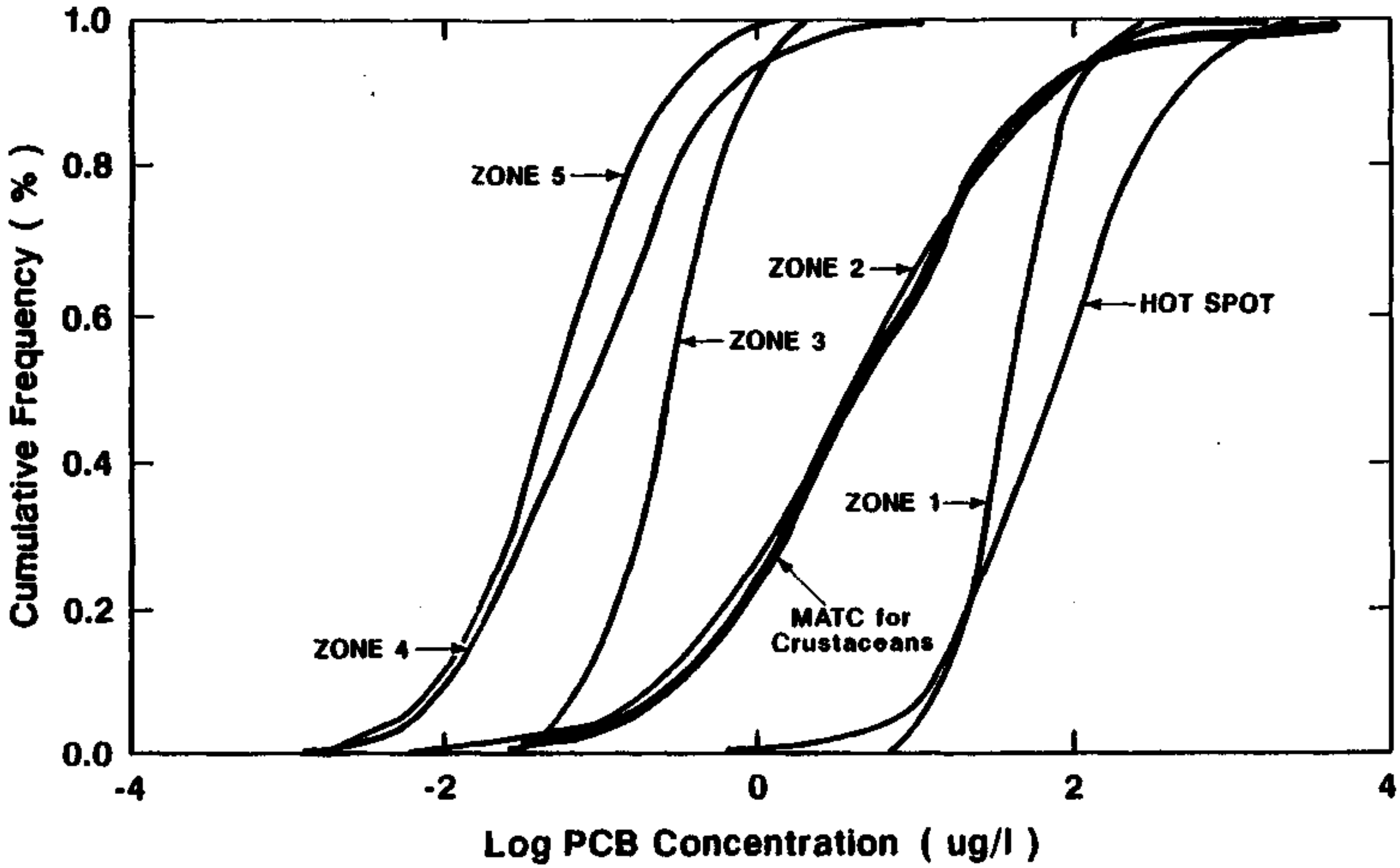


FIGURE 4-10
MATC FOR CRUSTACEANS AND EECs FOR ALL ZONES, PCBs, PORE WATER
NEW BEDFORD, MASSACHUSETTS

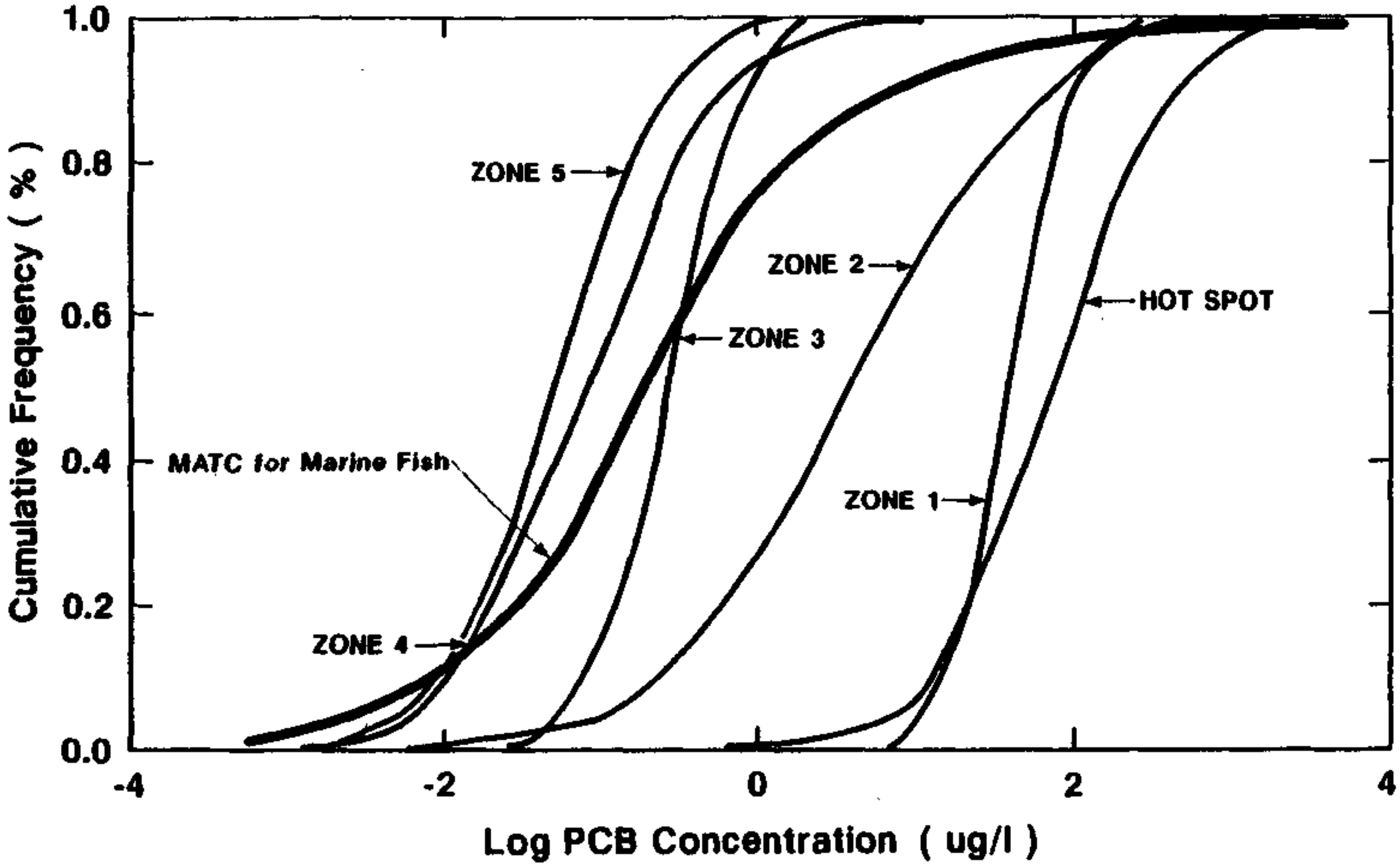


FIGURE 4-11
MATC FOR MARINE FISH AND EECs FOR ALL ZONES, PCBs, PORE WATER
NEW BEDFORD, MASSACHUSETTS

groups are the same as used in the joint probability analysis for the water column exposure.

These results are summarized in Table 4-1 as the percent probability of the median sediment concentration resulting in risk to each group. Exceedance probabilities in the Hot Spot and Zone 1 are 81 and 55 percent for crustaceans and mollusks, respectively, declining with increasing distance from the Upper Estuary. In Zone 4, the probability that a typical member of either group would experience contaminant levels likely to result in chronic effects is predicted to be less than 10 percent.

Based on available toxicological data, the probability that fish exposed to pore water PCB concentrations in Zone 1 and the Hot Spot, specifically, will experience chronic effects is close to a certainty. This likelihood is approximately 82 percent in Zone 2, declining to 24 percent in Zone 5. It is unlikely that any fish will be continually exposed to dissolved PCB concentrations similar to those found in the pore water; to the extent that this is not the case, the actual risks experienced would be considerably lower.

Figures 4-12 through 4-14 show the areal extent of the probability that chronic effects will be observed due to pore water exposure to PCBs for the various taxonomic groups, based on initial conditions for each grid cell.

4.1.3 Water Column Metals Contamination

The chronic effects probability functions for each of the five taxonomic groups are shown in Appendix C, co-plotted with the EEC probability functions for Zones 1 through 5 in Figures C-1 through C-5, Figures C-6 through C-10, and Figures C-11 through C-15, for copper, cadmium, and lead, respectively. Tables C-1 through C-3 present results of the joint probability analysis for each group.

Compared with results discussed previously for PCBs, there is less indication that aquatic organisms are at risk due to the metals contamination in New Bedford Harbor. This analysis would predict that crustaceans, as a group, are most likely to experience deleterious effects from copper, cadmium, and lead contamination. However, even in the most contaminated zones, impacts are predicted for less than 20 percent of these sensitive organisms. The other four taxonomic groups are at little discernable risk due to metals contamination in the water column, except for mollusks exposed to dissolved copper in Zones 1, 2, and 3 (see Figure C-3). In this case, this analysis would predict that levels of dissolved copper in the water column could have some impact on the most sensitive 10 to 15 percent of mollusk species in New Bedford Harbor. Although these potential

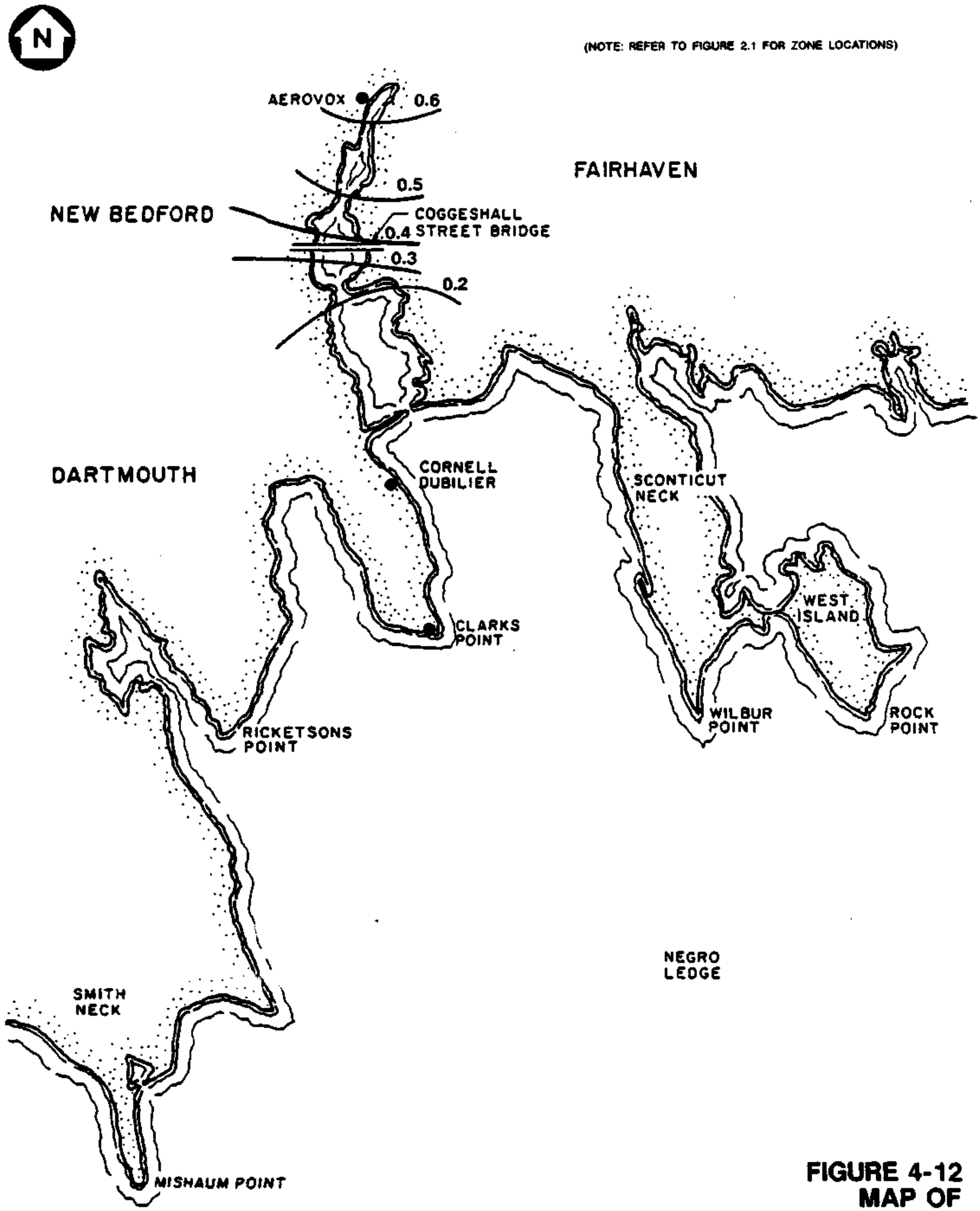


FIGURE 4-12
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
MOLLUSKS, PCBs, PORE WATER
NEW BEDFORD, MASSACHUSETTS

NOT TO SCALE

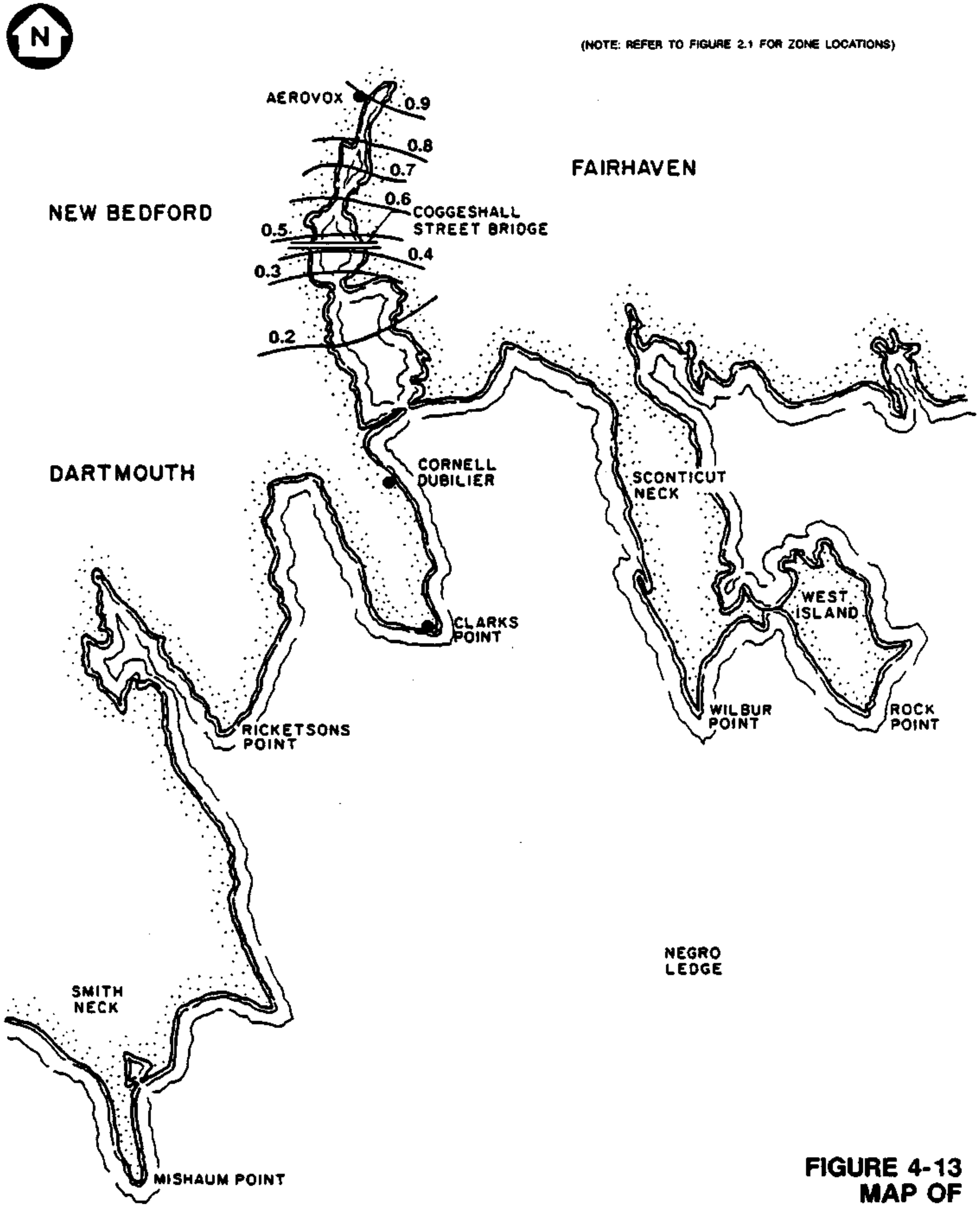


FIGURE 4-13
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR CRUSTACEANS, PCBs, PORE WATER
NEW BEDFORD, MASSACHUSETTS

NOT TO SCALE

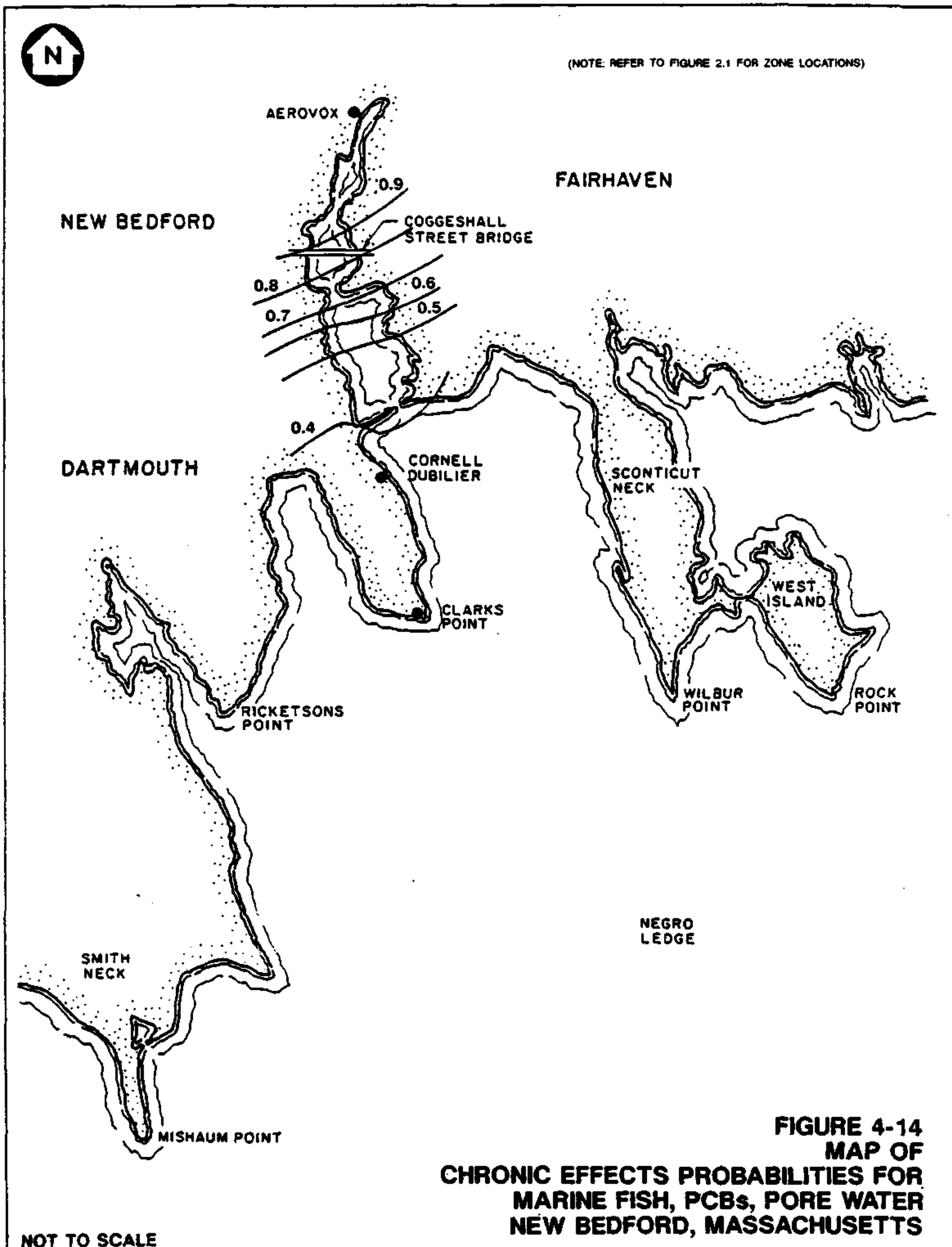


FIGURE 4-14
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
MARINE FISH, PCBs, PORE WATER
NEW BEDFORD, MASSACHUSETTS

risks are significant, they are not of the same magnitude as those described previously for PCBs.

Figures C-16 through C-30 show the areal extent of the probability that chronic effects will be observed due to water column exposure to metals for the various taxonomic groups.

4.1.4 Sediment Metals Contamination

MATC curves and EEC pore water curves are co-plotted for all taxonomic groups except algae in Appendix C, Figures C-31 through C-34, Figures C-35 through C-38, and Figures C-39 through C-42 for copper, cadmium, and lead, respectively. As for PCBs, the same chronic effects distributions were used for comparison with sediment pore water concentrations as with water column concentrations.

These results are summarized in Tables C-1 through C-3 as the percent probability of the mean sediment concentration resulting in risk to each group for the three metals of concern. In general, the exceedance probabilities are similar to those determined for water column exposures to these metals. Crustaceans are predicted to be most likely impacted by sediment contamination, with risk estimates of a much lower magnitude to those calculated for PCB contamination in these same areas (i.e., Zones 1, 2, and 3).

The other three taxonomic groups are predicted to be minimally impacted by the levels of these three contaminants in sediment, with probabilities ranging from 5 percent to virtually zero probability of exceeding the respective chronic effects thresholds.

Figures C-43 through C-46, Figures C-47 through C-50, and Figures C-51 through C-54, present the areal extent of the probabilities that chronic effects will be observed due to pore water exposure to copper, cadmium, and lead (respectively) for the various taxonomic groups.

4.2 COMPARISON WITH AMBIENT WATER QUALITY CRITERIA

4.2.1 Water Column Concentrations

The chronic PCB AWQC for the protection of marine life and its uses is 0.03 ug/L. There is no 1-hour marine acute criterion for PCBs; however, the AWQC document indicates that acute effects to aquatic organisms from PCB exposure may be probable at concentrations greater than 10 ug/L (EPA, 1980b).

Because the intent of the baseline risk assessment is to provide a benchmark against which results of numerical modeling of remedial alternatives may be compared, the model start-up conditions were used for risk comparisons. The start-up conditions reflect both the initial sediment conditions, which are based on available data for the area, and the dynamics of the physical/chemical model. The vertically averaged start-up conditions in each zone were believed to accurately represent chronic exposure in the harbor.

The maximum concentrations observed were considered to be reflective of potential short-term exposures. Consequently, for each zone, maximum PCB concentration values were compared to the 10-ug/L benchmark, and mean concentration data to the chronic AWQC, to generate a measure of potential risks to aquatic organisms. Simple statistics summarizing the concentration data by zone are presented in Table C-1 in Appendix C. The acute benchmark concentration of 10 ug/L was not exceeded by the maximum concentration in the start-up conditions data in any zone at the New Bedford Harbor site. Based on this comparison, potential risks associated with short-term exposure to PCBs dissolved in the water column are expected to be slight.

However, the chronic AWQC is exceeded by the mean PCB concentration in all zones except Zone 5. Therefore, aquatic organisms are potentially at risk of experiencing effects due to chronic exposure to PCB contamination in all areas of New Bedford Harbor north of the Hurricane Barrier. Because the chronic AWQC of 0.03 ug/L for PCBs is not based solely on toxicity information (EPA, 1980b), it does not necessarily reflect a level protective of aquatic life, but rather of aquatic life and its uses, and may be considered a conservative standard against which to evaluate risk.

Although the chronic marine AWQC for copper (2.9 ug/L) was exceeded by the mean water column concentrations in both Zones 2 and 3 (see Table 2-3), the exceedence was slight. Ratios of the mean copper concentration to the chronic criterion were only 1.17 and 1.2 for Zones 2 and 3, respectively. Although some potential exists for adverse impacts due to dissolved copper in the water column in these areas, these ratios suggest that any effects would not be severe. The chronic criteria for cadmium and lead were not exceeded in any zone in New Bedford Harbor.

4.2.2 Sediment Concentrations

An interim Sediment Quality Criterion (SQC) is available for PCBs (Aroclor 1254); no SQC have been developed for metals. As is the case for the AWQC, the interim SQC developed by EPA (EPA, 1988) is residue-based; that is, it is intended to be a value that will not result in commercially harvested species having PCB body burdens exceeding the original FDA action level of 5 ppm. SQC are not currently considered to be ARARs for Superfund programs. The SQC was derived from the AWQC by applying a partitioning coefficient (K_{oc}) that varies with the amount of organic carbon in the sediment. The upper and lower 95 percent confidence intervals (CIs) for the SQC are based on the variance of K_{oc} and represent the range within which the actual sediment criterion value is expected to fall. The lower CI is assumed to represent the concentration which, with 97.5 percent certainty, will result in body burdens in resident commercial species remaining below 5 ppm.

The mean sediment concentrations in each zone were compared to the lower 95 percent CI; the maximum concentrations were compared to the SQC. TOC values for sediments in the area of interest vary from less than 1 percent to nearly 10 percent, but are generally higher in the Acushnet River Estuary where values near 5 percent are typical. For simplicity, a value of 1 percent TOC was assumed for all areas, providing a conservative estimate of sediment toxicity in the estuary. Assuming an average TOC of 1 percent, the carbon-normalized SQC is 0.418 ug/g (ppm), with a lower 95 percent CI of 0.083 ug/g. These results indicate that virtually all areas of the harbor, including most adjacent areas of the Outer Harbor and even some areas well out into Buzzards Bay, pose a risk to at least some aquatic organisms. Even assuming a TOC of 10 percent, which would reduce the amount of PCB available for uptake by biota by an order of magnitude, essentially all areas of the harbor would exceed the lower 95 percent CI of 0.829 ug/g.

4.3 SITE-SPECIFIC TOXICITY TESTS

Several toxicity tests have been performed with New Bedford Harbor sediment, and the results provide the most realistic indication of the degree of toxicity posed by contaminated sediment in the harbor. Although these studies provide the most direct indication of toxicity, it is difficult to separate effects due to PCBs from effects due to metals and other contaminants that may be present in the sediment. In addition, it is difficult to evaluate how closely the laboratory conditions simulated actual harbor conditions in the various tests. Despite these limitations, site-specific data permit an independent verification of the reasonableness and accuracy of the more theoretically based predictions discussed previously.

In a solid-phase bioassay, Hansen exposed the sheepshead minnow (*Cyprinodon variegatus*) and amphipod (*Ampelisca abdita*) to New Bedford Harbor sediment (Hansen, 1986). The toxicological endpoints examined were mortality, fish embryo survival, and hatched fish survival. Other sublethal effects theoretically included in the joint probability and AWQC evaluations may also have been occurring but were not evaluated. In addition, it is not possible to identify the specific contaminants responsible for these effects.

The reported results of Hansen's study were as follows (Hansen, 1986):

- o significant reduction in survival of adult sheepshead minnows exposed for 29 days to sediment (i.e., to water contaminated by contact with contaminated sediment) collected from Zones 1 and 2 (zero and 72 percent, respectively)

- o significant reduction in survival of progeny (i.e., embryos and/or hatched fish) of adult minnows exposed to sediment collected from Zones 1, 2, and 3

- o 10-day amphipod mortality correlated with the spatial gradient of contaminants in harbor sediment, with mortality rates of 100 and 92.2 percent in amphipods exposed to sediment from Zones 1 and 2, respectively, compared to 13.3 percent in the reference area

- o mortality rates of 11.1 to 73.3 percent in amphipods exposed for 10 days to sediment obtained from Zones 4 and 3, respectively

Results of these sediment toxicity tests indicate that New Bedford Harbor sediment is toxic to certain aquatic organisms. Based on these data, it appears that sediment obtained from within the inner harbor (north of the Popes Island/State Route 6 Bridge) poses a risk to resident aquatic invertebrates and to the survival and reproduction of resident fish. Measurable but less severe adverse effects were observed in fish and amphipods exposed to sediment obtained from Zone 4, which contained 10 ppm total PCBs (Hansen, 1986).

In general, the toxicity of New Bedford Harbor sediment to amphipods and fish decreases from the Upper Estuary toward the Hurricane Barrier. Toxic effects have been observed in sediment from Zone 4; however, these effects are not statistically significant when compared to a reference sediment collected from central Long Island Sound.

In 1988, the National Oceanic and Atmospheric Administration developed sediment target levels for PCBs that were considered protective of aquatic life. The recommended range, 0.1 to 1.0 ppm PCBs, is based on information showing that concentrations of PCBs in aquatic organisms residing in contaminated areas are equal to or exceed the PCB concentrations found in the sediment (Field and Dexter, 1988). This relationship is generally true for xenobiotic compounds (e.g., PCBs) that are persistent in the environment, readily bioaccumulated by aquatic organisms, and slowly biotransformed and excreted by fish (Lech and Peterson, 1983). In addition, toxicological effects were observed in fish with tissue concentrations of PCBs less than 0.1 ppm (see Subsection 4.4).

4.4 RISK DUE TO BIOACCUMULATION OF PCBS

Bioaccumulation of PCBs by exposed organisms results in high tissue burden levels of these compounds. There is evidence suggesting that PCBs are also biomagnified in the food chain (Shaw and Connell, 1982; Thomann, 1978; and Thomann and Connolly, 1984). The bioaccumulation of PCBs may result in elevated tissue levels that may be toxic to the organism directly, or indirectly as a result of modified behavior with consequent increased exposure to predators.

Food-chain transfer of PCBs is considered likely for organisms within the New Bedford Harbor area, because elevated PCB concentrations were detected in prey organisms. Mean PCB concentrations in polychaetes, clams, mussels, and crabs in the harbor are 12.9, 5.3, 2.6, and 0.4 ppm, respectively (see Figure 4-2). These organisms are all constituents of the diet of winter flounder, striped bass, and bluefish.

PCB tissue concentrations resulting from dietary exposure in upper level carnivores have been shown to produce the following effects in marine fish:

- Concentrations of 11 to 98 mg/kg caused liver abnormalities in the tomcod (Klauda et al., 1981).
- Concentrations greater than 24 mg/kg caused reproductive failure in the cyprinid minnow (Bengtsson, 1980).
- Concentrations greater than 7.0 mg/kg caused reduced survival of sheepshead minnow embryos (Hansen, 1973).
- Concentrations of 0.12 mg/kg caused inhibited reproduction in the Baltic flounder (Spies, 1985).
- Concentrations of 0.2 mg/kg reduced reproductive success in the starry flounder (Spies, 1985).
- Concentrations of 1.4 mg/kg caused reproductive impairment in the striped bass (Ray et al., 1984).
- Concentrations from 0.005 to 0.05 mg/kg caused histological changes in the Atlantic cod (Freeman et al., 1982).

PCB tissue levels in winter flounder from the New Bedford Harbor area were compared to available toxicity data for similar species. To allow comparisons between the New Bedford Harbor whole-body concentrations and organ-specific toxicity data, the whole-body PCB concentrations were adjusted using an edible:whole-body ratio derived by BOS for winter flounder collected to provide calibration data for the food-chain model (Battelle, 1987). Whole-body concentrations for winter flounder in the modeling program data base were multiplied by 0.13 to produce edible-tissue concentrations, which were then adjusted based on the results using striped bass to produce concentrations in the gonads (Ray et al., 1984). Ray found that fish tend to accumulate PCBs in the gonadal tissues, with the ratio of muscle to gonad PCB concentrations ranging from 1:1 to 10:1 (Ray et al., 1984). Estimates of the PCB concentration in the gonads of winter flounder are listed in Table 4-2.

Limited data are available on the effects of PCB concentrations in gonads of winter flounder. Toxicity data for two similar species (Baltic and starry flounder) were used to qualitatively assess the potential risks associated with PCB tissue burdens. These data indicate that concentrations as low as 0.12 and 0.2 ppm PCBs in the ovaries of these species can inhibit reproduction (Spies, 1985; and Von Westernhagen et al., 1981). The range of estimated PCB concentrations in the gonads of the winter flounder exceed 0.2 ppm PCBs in all areas except Area 4, where the mean estimated gonad concentration was 0.1 ppm.

Because of the assumptions used to derive these concentrations, conclusions concerning the potential risk to these organisms cannot be made. However, these data do indicate the potential for the accumulation of PCBs in reproductive organs of species inhabiting New Bedford Harbor to levels that have been shown to cause reproductive effects.

Reproductive effects in winter flounder exposed to surface water from New Bedford Harbor have been observed by Black (Black, et al., 1986). Gravid female flounder were collected from New Bedford Harbor (Zone 5), and the collected progeny were reared under uncontaminated conditions. Elevated PCB concentrations were observed in the eggs of winter flounder from the New Bedford Harbor area. Larvae hatched from these eggs were significantly smaller in length and lower in weight than the eggs and larvae from the reference area near Fox Island in lower Narragansett Bay. PCB tissue concentrations in the adult winter flounder were not reported; therefore, direct relationships between PCB body burdens and reproductive effects cannot be made. At larval metamorphosis, the differences between locations had disappeared. However, in a competitive and stressful natural environment, it is likely that even transient differences in size would result in significant differences in juvenile survivorship.

TABLE 4-2
CONVERSION OF WHOLE-BODY WINTER FLOUNDER PCB
TISSUE CONCENTRATIONS TO EXPECTED GONAD CONCENTRATIONS

NEW BEDFORD, MASSACHUSETTS

Winter Flounder	Whole-body PCB Concentration (mg/kg)	Edible-tissue PCB Concentration[1] (mg/kg)	Expected Range of PCB-gonad Concentration[2] (mg/kg)
Area 1 MAXIMUM	20.23	2.63	2.63 - 26.30
MEAN	7.99	1.039	1.039 - 10.39
Area 2 MAXIMUM	8.07	1.05	1.05 - 10.5
MEAN	2.85	0.371	0.371 - 3.71
Area 3 MAXIMUM	6.35	0.83	0.83 - 8.3
MEAN	2.14	0.278	0.278 - 2.78
Area 4 MAXIMUM	2.62	0.34	0.34 - 3.4
MEAN	0.78	0.101	0.101 - 1.01

NOTES:

[1] These values are based on an edible-muscle-to-whole-body ratio of 0.13.
[2] These values are based on muscle-to-gonad ratios ranging from 1:1 to 10:1.

Thurberg examined the effects of high PCB body residues in American lobster, Homarus americanus, on egg-hatching success, larval growth and survival, molting success, and the duration of the larval period (Thurberg, 1985). Despite the elevated levels of PCBs in the eggs and larvae of New Bedford Harbor lobsters, there were no discernable differences in any of the biological response variables.

Capuzzo investigated the effects of PCB uptake and accumulation on growth, energetics, and reproductive potential of the mollusk (Mytilis edulis) (Capuzzo, 1986). Mussels were placed in screened cages at various locations in Buzzards Bay and Nantucket Sound where in situ physiological measurements relating to energetic partitioning were taken. Mussels transplanted to the Hurricane Barrier (Zone 4) showed considerable uptake of PCBs initially, followed by a gradual stabilization, and experienced a lower growth potential, relative to the stations in Nantucket Sound and at Cleveland Ledge. This effect was due to a decrease in the amount of carbon ingested and assimilated, as well as to increased respiratory expenditures. These individuals also made the lowest reproductive effort (measured as the amount of energy allocated to reproduction relative to the total amount of energy assimilated to growth and respiration during the spawning period) of the three stations.

The studies cited previously have shown that:

- PCBs accumulate in certain aquatic organisms (Capuzzo, 1986).
- PCBs concentrate in the gonads of fish (Ray et al., 1984).
- PCB concentrations greater than 0.1 ppm in the gonads of flounder have been shown to cause reproductive effects (Spies, 1985 and Van Westernhagen et al., 1981).
- Eggs from winter flounder in the New Bedford Harbor area had elevated levels of PCBs (Black et al., 1986).
- Larvae hatched from eggs containing elevated PCB levels were smaller in length and lower in weight.
- Reproductive effects (measured as the amount of energy allotted to reproduction) were lower in the mussels exposed to surface water from the New Bedford Harbor area.

The body of toxicity data described indicate that biota at the New Bedford Harbor site are at potential risk due to the consequences of PCB accumulation; this is supported by the site-specific data generated by Black and Capuzzo (Black et al., 1986; and Capuzzo, 1986).

Because no toxicity data associated with PCB tissue burdens could be identified for other species (e.g., lobsters, clams, crabs, and polychaetes), a discussion of risk to these species is not possible. However, PCBs are lipophilic, are known to accumulate in fatty tissues, and have been detected in all biota in New Bedford Harbor. Although there is considerable variation in tolerance to PCBs across species, some species would be expected to be at least as sensitive to PCBs as the species for which data are available, and would therefore be expected to be impacted by the observed body burdens.

4.5 BENTHIC SURVEYS

Several infaunal surveys have been performed at the New Bedford Harbor site. Although many ecological factors in addition to chemical contamination can contribute to areal differences in the numbers and kinds of organisms, these results generally support the conclusions reached previously in this report.

An extensive benthic sampling program was conducted for USACE (USACE, 1988a). The 26 sampling locations spanned all areas of New Bedford Harbor discussed in this report. Significant correlations between the level of PCB contamination in the harbor and several measures of community, including the number of species, and diversity and evenness indices were found. Due to differences in the sampling methodology used during the program, there is some concern regarding comparability of the sampling data. However, overall trends relating benthic community descriptors to PCB levels appear to be consistent. The basic pattern observed was a domination in the Upper Estuary by the polychaete, Streblospio benedicti; another polychaete, Tharyx acutus, was dominant in the rest of the inner harbor. Outside the Hurricane Barrier, bivalves and gastropods became the most common organisms. Associated with these taxonomic differences were an increase in the species diversity of the infaunal community and a more equal representation of individual species from the Upper Estuary into the outer harbor.

A comparative study of this nature suffers from the gross differences in habitat between different locations. It is possible that physical factors (e.g., sediment characteristics and turbidity) are the primary determinants of the community patterns observed. However, these results do not contradict previous conclusions regarding risks associated with different zones. Many polychaetes are generally less sensitive to sediment contamination than other taxa, and their general domination of the most highly contaminated sediments in the harbor suggests the impact that PCBs and other chemicals may be having on this ecosystem (Rubinstein, 1989).

A wetland study compared chemical and biological data from six wetland areas in the harbor and from a relatively unpolluted reference area in Buzzards Bay (USACE, 1988b). The study found a depressed benthic community in the Zone 1 wetland. In addition, comparison of the biological data between a Zone 2 wetland and the reference area indicated significant differences in species diversity and evenness, particularly among polychaetes, amphipods, and mollusks. However, habitat differences complicate any attempt to relate differences in benthic community patterns to variation in the PCB contamination between these locations.

4.6 SUMMARY OF RISK CHARACTERIZATION

As part of the ecological risk assessment for the New Bedford Harbor site, a joint probability analysis was used to develop probabilistic risk estimates for the effects of PCBs and heavy metals (i.e., copper, cadmium, and lead) contamination on marine organisms. The expected distribution of a taxonomic group response to a contaminant was estimated by extrapolating the responses observed in individual organisms to larger groups. This methodology involved the summarization of the available toxicological data using errors-in-variables regression models and the quantification of uncertainty as the combining of variances through the various extrapolations.

Separate estimates were developed for the major taxonomic groups in New Bedford Harbor to provide more detailed information on how contamination is affecting specific components of the harbor ecosystem. This permits the risk assessment process to isolate the most sensitive groups of organisms, as well as quantifying the likelihood of impact for all groups. Presentation of the risk analysis in probabilistic terms will provide a more complete representation of the impacts of the various remedial alternatives on potentially affected organisms. In addition to this approach, PCB and metals concentrations in the harbor were compared to sediment and water criteria, and the results of various site-specific bioassays and benthic surveys were evaluated with respect to potential risk. Results of these different approaches are summarized in the following paragraphs; risks are discussed in view of these findings.

Aquatic organisms (particularly marine fish) are at risk due to exposure to waterborne PCBs in New Bedford Harbor. The mean PCB concentrations in the Hot Spot and Zones 1 through 4 exceed the chronic AWQC, and the joint probability analysis indicates that there is significant likelihood that chronic effects will be realized in at least some species inhabiting New Bedford Harbor. These risks are most severe in Zones 1 and 2 and the Hot Spot; however, potential risk is evident for all zones within the Hurricane Barrier.

The pore water PCB concentrations in the sediment are highly toxic to at least some members of all major taxonomic groups. In the Upper Estuary, the likelihood that chronic effects would be observed in a typical marine fish species exposed to PCBs in pore water is close to 100 percent; risk is substantial for mollusks and crustaceans as well. The risk probabilities for all groups decline toward the outer harbor; however, marine fish may still be substantially impacted in Zone 5. However, in Zone 4, the likelihood that chronic effects would be realized in typical crustaceans and mollusks is predicted to be less than 10 percent. The SQC, carbon-normalized to 1 percent TOC, is exceeded in Zones 1 and 2, and the lower 95 percent confidence level for the SQC is exceeded in all zones. Finally, results of various sediment bioassays support the conclusions based on laboratory-generated toxicological data and comparisons with interim SQC. Sediment from the inner harbor has been demonstrated to be toxic to both benthic invertebrates and fish; the degree of toxicity is correlated with PCB levels in test sediments.

Many marine organisms from New Bedford Harbor have been shown to be contaminated with elevated tissue levels of PCBs. PCB levels in gonadal tissue of winter flounder collected from Zones 1, 2, and 3 exceed levels shown to result in reproductive impairment and other effects in marine fish. Levels in organisms from lower trophic levels may either induce toxicological effects or impact predator species.

Risk due to exposure to PCBs is also largely dependent on location of the organisms in the harbor, and may be a function of migratory behavior or reproductive habits. Organisms such as American eels, which reside mostly in the Upper Estuary (i.e., Zones 1 and 2) in close contact with the sediment, are likely to be at greater risk of toxic effects from exposure to PCB contamination than organisms that only migrate periodically into this area (e.g., blueback herring) and remain in the water column. In addition, juvenile aquatic organisms using the Upper Estuary/Hot Spot area as a nursery ground may be at an elevated risk of contaminant exposure, given that this lifestage is generally more sensitive to chemical insult than the adult stage. Foraging behavior and prey preferences can also influence the degree of exposure encountered by a particular organism.

With regard to potential risks due to heavy metals, both the joint probability analysis and a comparison with AWQC indicate some possibility for impacts on marine biota in New Bedford Harbor. Based on comparisons with AWQC, concentrations of copper in the water column represent some potential for concern, with crustaceans determined to be the taxon most likely at risk. Results of this analysis suggest that, although metals may be having some impact on the harbor ecosystem, the effects attributable to these contaminants are overshadowed by the presence of PCBs at much more harmful levels.

Potential impacts due to the presence of PCBs or heavy metals in New Bedford Harbor cannot be adequately defined by assessing risk to a single species or taxonomic group or by exposure to a single medium. Chemical stresses placed on aquatic organisms are multilayered. An organism in New Bedford Harbor is simultaneously exposed to many contaminants in addition to those evaluated in this risk assessment. However, based on available data, it appears that the four contaminants chosen (i.e., PCBs, copper, cadmium, and lead) constitute the most significant risk to organisms in the harbor. It is impossible to quantify the effects of multiple exposures to a mixture of contaminants. Furthermore, member species in an ecological community interact and depend on other species to satisfy many essential biological needs. Because of the interdependence of ecological units that comprise an ecosystem, seemingly minor disturbances affecting components of the system can have significant ramifications on the stability and functioning of the overall system. In view of the inherent complexity involved in attempting to assess the impacts of chemical stress on overall ecosystem integrity, only a qualitative approach is typically feasible.

The effects of chemical stress on an ecosystem can potentially affect such interspecific ecological interactions as competition, predation, and disease resistance. These effects can alter a population's birth and death rates resulting in long-term changes in numerical abundance (Ricklefs, 1979). The elimination of commercial harvesting of finfish, shellfish, and lobsters since 1979 further complicates the evaluation of large-scale effects in New Bedford Harbor.

Numerous site-specific and laboratory studies indicate that New Bedford Harbor is an ecosystem under stress due to PCBs and other chemical contamination. This stress can be manifested in many ways that are perceived as having negative consequences from a human perspective. There are many potentially affected species for which changes in population dynamics or marketability are of interest, including various shellfish and fish harvested from New Bedford Harbor before the closure enactment. On another level, however, the health of the overall harbor is of concern, in that anthropogenic effects can alter the resource value of the harbor (i.e., recreational, food, and esthetics). The issue is whether the stability and functioning of the harbor ecosystem has been or will be impacted by the described contamination, stability being defined as the intrinsic ability of a system to withstand or recover from externally caused change (Ricklefs, 1979). Overall stability may be affected by various changes related to chemical contamination in the harbor, including population size, species diversity or evenness, and physiological or behavioral changes that impact interactions between species.

In conclusion, all approaches used to assess risk associated with PCB contamination in New Bedford Harbor indicate that levels in Zones 1, 2, and 3 have the potential to strongly impact individual biota in the harbor, as well as the overall integrity of the harbor as an integrated functioning unit. This impact may take the form of numerical changes at the population level, changes in community composition, and ultimately ecosystem stability. Ecosystem level disruptions are less strongly indicated in Zone 4 but still are probable.

GLOSSARY OF ACRONYMS AND ABBREVIATIONS

AWQC	Ambient Water Quality Criteria
BCF	bioconcentration factor
BOS	Battelle Ocean Sciences
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CI	confidence interval
EEC	expected environmental concentration
EPA	U.S. Environmental Protection Agency
FDA	U.S. Food and Drug Administration
FRV	final residue value
FS	Feasibility Study
K_d	partition coefficient
K_{oc}	partitioning coefficient
MATC	maximum acceptable toxicant concentration
mg/kg	milligrams per kilogram
NPL	National Priorities List
NUS	NUS Corporation
PCB	polychlorinated biphenyl
PNL	Pacific Northwest Laboratories (Battelle)
ppb	parts per billion
ppm	parts per million
RI	Remedial Investigation
SQC	Sediment Quality Criterion
TOC	total organic carbon
ug/g	micrograms per gram
ug/L	micrograms per liter
USACE	U.S. Army Corps of Engineers

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APPENDIX A

EXPECTED EXPOSURE CONCENTRATIONS

FOR

COPPER, CADMIUM, AND LEAD

TABLE A-1
EXPECTED EXPOSURE CONCENTRATIONS FOR COPPER (1)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

HARBOR ZONE	MEAN (ug/l)	TRANSFORMED VALUES (2) MEAN	ST. DEV.	VARIANCE
1, Water Column	2.218	0.346	0.067	0.004
2, Water Column	3.406	0.532	0.134	0.018
3, Water Column	3.486	0.542	0.131	0.017
4, Water Column	2.180	0.338	0.247	0.061
5, Water Column	0.710	-0.149	0.340	0.115
1, Pore Water	0.317	-0.499	0.836	0.698
2, Pore Water	0.112	-0.953	1.137	1.129
3, Pore Water	0.340	-0.468	0.818	0.670
4, Pore Water	0.191	-0.719	0.695	0.483
5, Pore Water	0.047	-1.327	0.687	0.472

Notes:

(1) Estimates derived from the program data base maintained by Battelle Ocean Sciences.

(2) Log (base 10) transformed values, with standard deviations and variances.

TABLE A-2
EXPECTED EXPOSURE CONCENTRATIONS FOR CADMIUM (1)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

HARBOR ZONE	MEAN (ug/l)	TRANSFORMED VALUES (2)		
		MEAN	ST. DEV.	VARIANCE
1, Water Column	2.460	-0.709	0.391	0.153
2, Water Column	2.404	-0.508	0.381	0.145
3, Water Column	1.560	-0.735	0.193	0.037
4, Water Column	2.198	-0.971	0.342	0.117
5, Water Column	2.477	-1.359	0.394	0.155
1, Pore Water	2.985	-0.694	0.475	0.226
2, Pore Water	8.810	-0.866	0.945	0.893
3, Pore Water	2.924	-0.907	0.466	0.217
4, Pore Water	3.597	-1.281	0.556	0.309
5, Pore Water	5.957	-1.963	0.775	0.601

Notes:

(1) Estimates derived from the program data base maintained by Battelle Ocean Sciences.

(2) Log (base 10) transformed values, with standard deviations and variances.

TABLE A-3
EXPECTED EXPOSURE CONCENTRATIONS FOR LEAD (1)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

HARBOR ZONE	MEAN (ug/l)	TRANSFORMED VALUES (2) MEAN	ST. DEV.	VARIANCE
1, Water Column	1.259	0.100	0.412	0.170
2, Water Column	1.183	0.073	0.088	0.008
3, Water Column	0.560	-0.251	0.482	0.233
4, Water Column	0.212	-0.673	0.520	0.270
5, Water Column	0.052	-1.280	0.957	0.916
1, Pore Water	1.005	0.002	0.785	0.617
2, Pore Water	0.287	-0.541	1.009	1.018
3, Pore Water	0.583	-0.235	0.677	0.458
4, Pore Water	0.103	-0.988	0.577	0.333
5, Pore Water	0.245	-0.611	0.675	0.456

Notes:

(1) Estimates derived from the program data base maintained by Battelle Ocean Sciences.

(2) Log (base 10) transformed values, with standard deviations and variances.

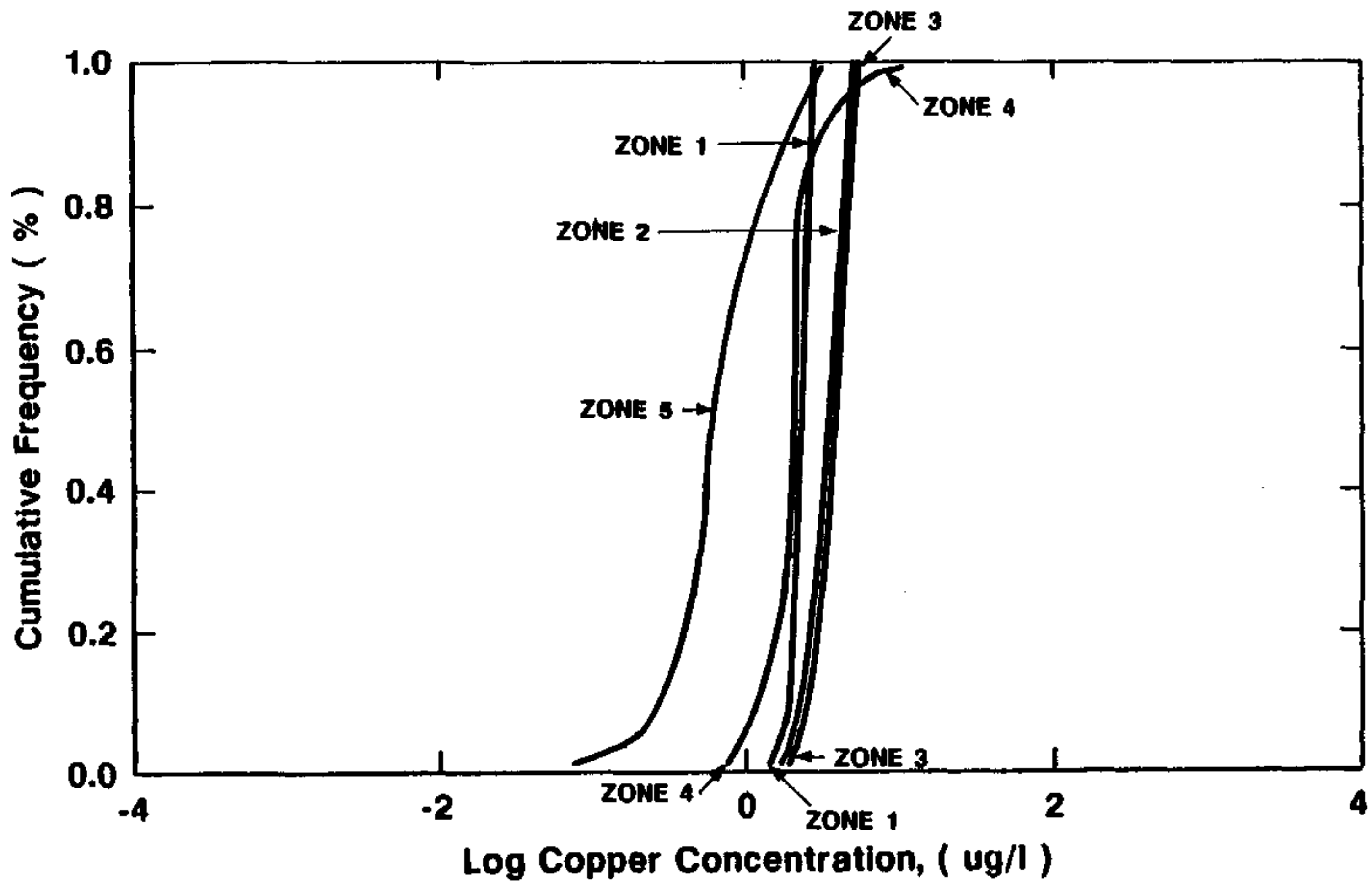

FIGURE A-1
EECs BY ZONE FOR COPPER, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

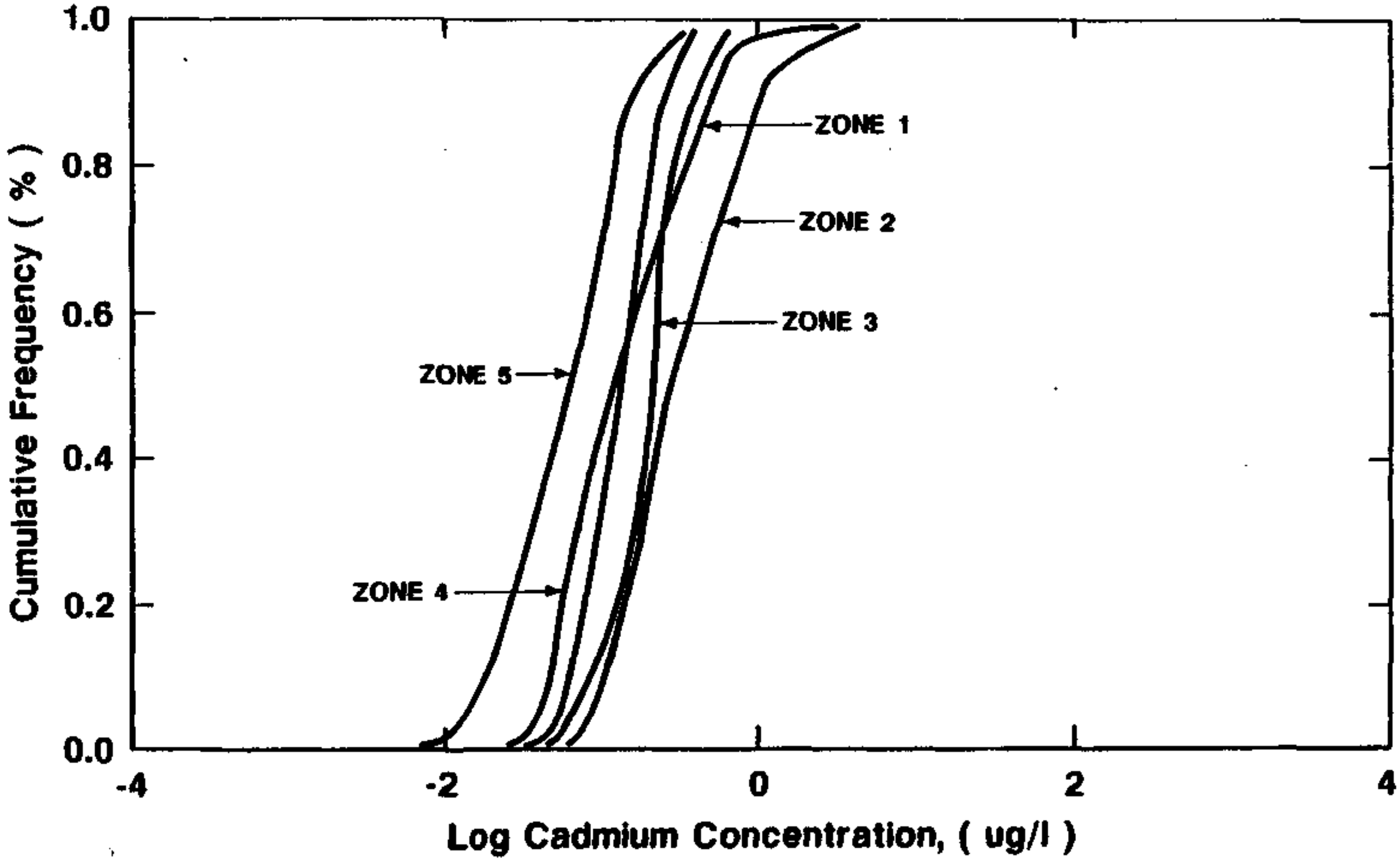

FIGURE A-2
EECs BY ZONE FOR CADMIUM, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

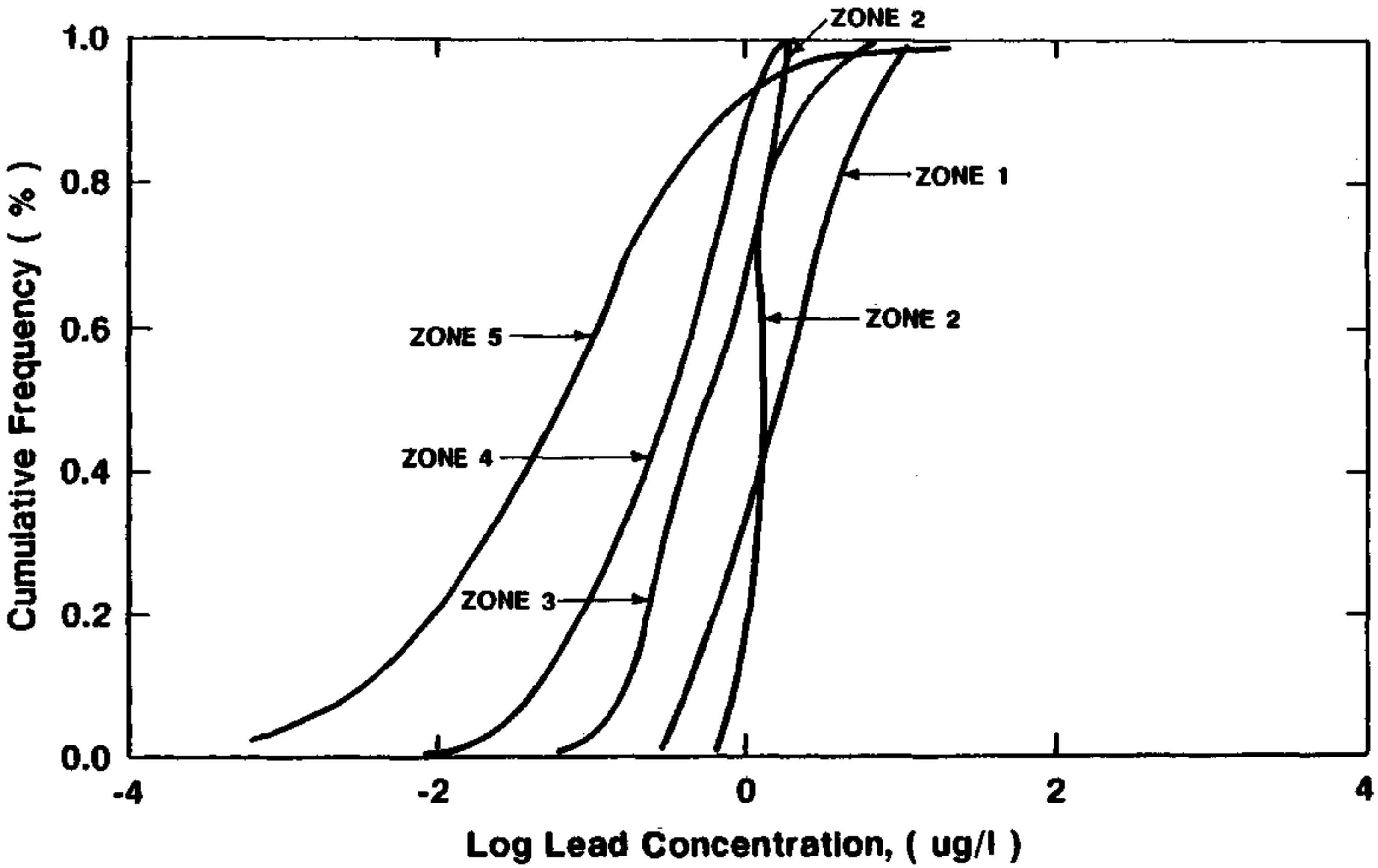

FIGURE A-3
EECs BY ZONE FOR LEAD, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

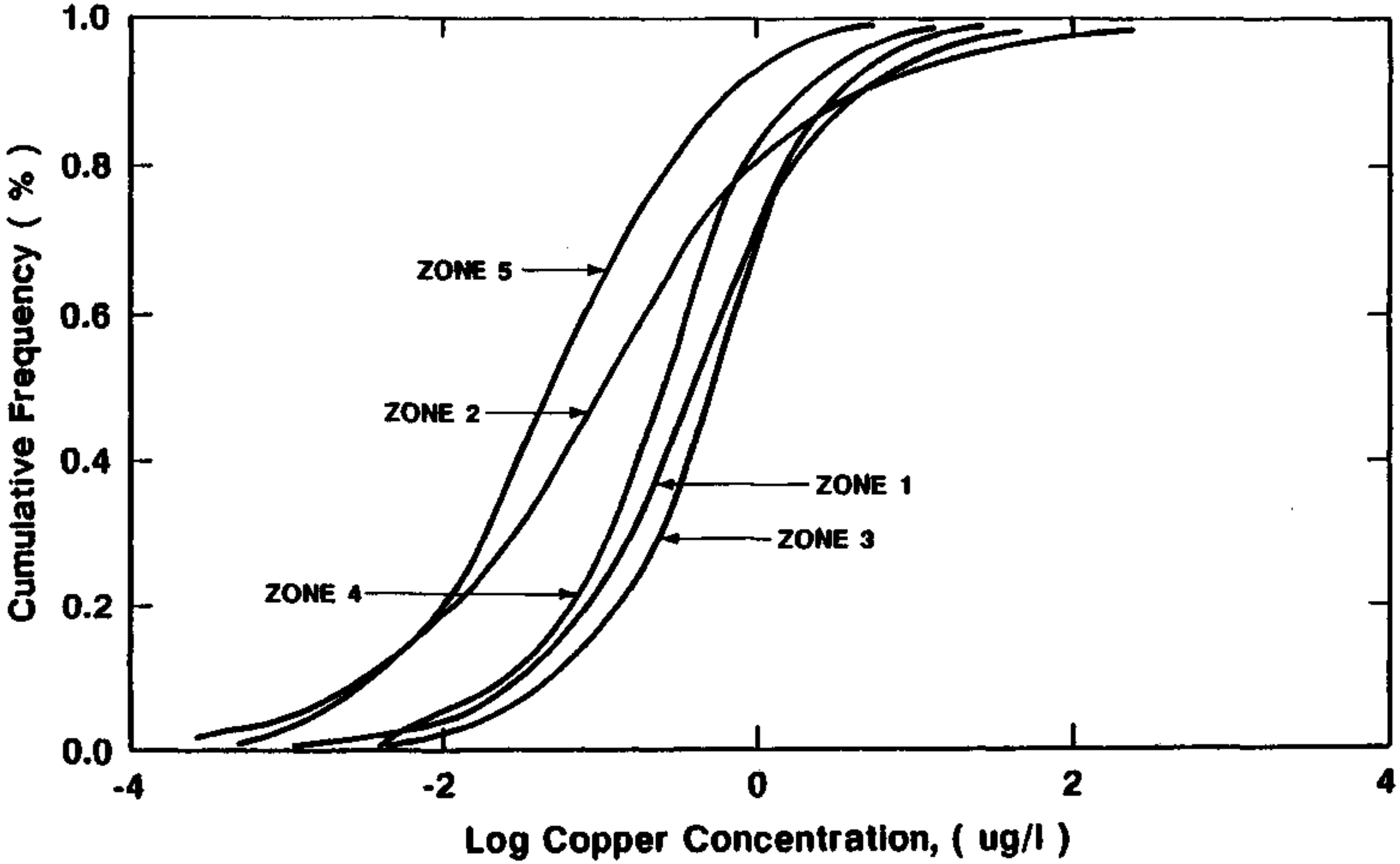

FIGURE A-4
EECs BY ZONE FOR COPPER, PORE WATER
NEW BEDFORD, MASSACHUSETTS

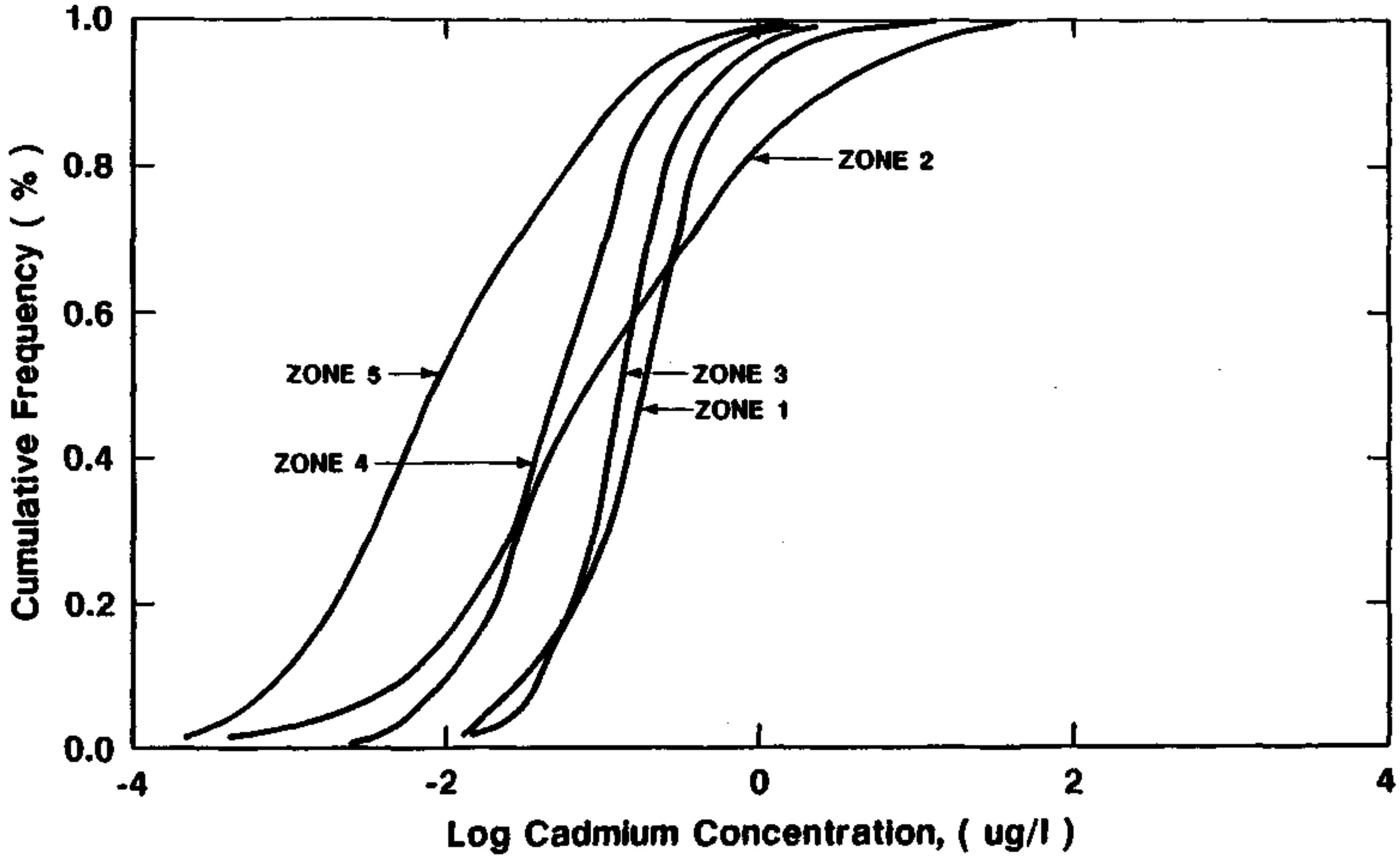

FIGURE A-5
EECs BY ZONE FOR CADMIUM, PORE WATER
NEW BEDFORD, MASSACHUSETTS

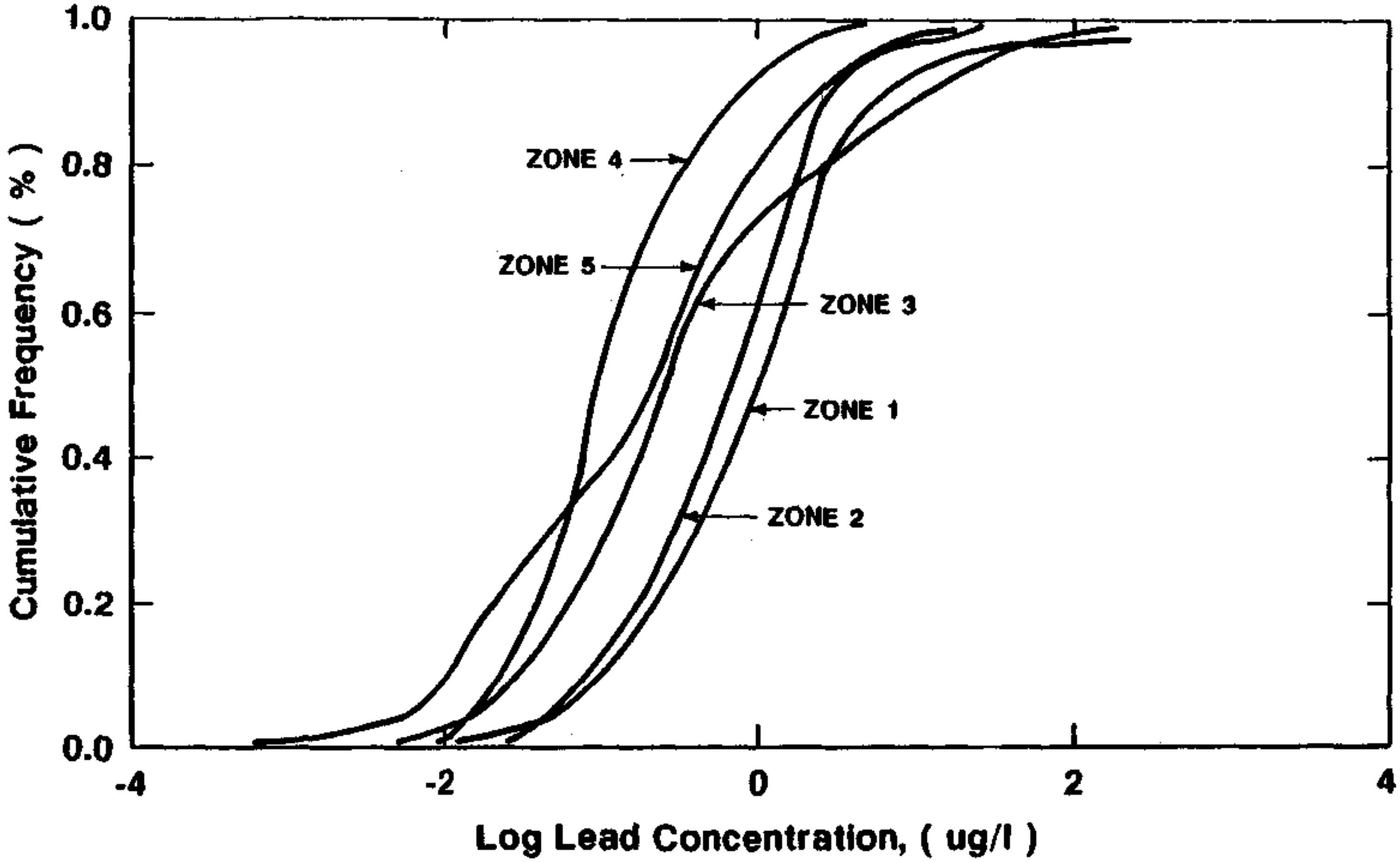

FIGURE A-6
EECs BY ZONE FOR LEAD, PORE WATER
NEW BEDFORD, MASSACHUSETTS

APPENDIX B

TOXICITY DATA

FOR

PCBs, COPPER, CADMIUM, AND LEAD

TABLE B-1
PCB ACUTE TOXICITY DATA FOR MARINE ORGANISMS

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

SPECIES	CHEMICAL	LC50 or EC50 (ug/l)	REFERENCE	HABITAT GROUP
Brown shrimp Penaeus aztecus	Aroclor 1016	10.5	Hansen et al., 1974a	Demersal
Grass shrimp Palaemonetes pugio	Aroclor 1016	12.5	Hansen et al., 1974a	Demersal
Grass shrimp Palaemonetes pugio	Aroclor 1254	6.1 to 7.8	Ernst, 1984	Demersal
Pink shrimp Penaeus duorarum	Aroclor 1248	32	Lowe, undated	Demersal
Pink shrimp Penaeus duorarum	Aroclor 1254	1	Nimmo & Bahner, 1976	Demersal
Pink shrimp Penaeus duorarum	Aroclor 1254	32	Lowe, undated	Demersal
Shrimp, Crangon septemspinosa	Aroclor 1242	13	McLeese & Metcalf, 1980	Demersal
Shrimp, Crangon septemspinosa	Aroclor 1254	12	McLeese & Metcalf, 1980	Demersal
Sheepshead minnow (embyros and fry) Cyprinodon variegatus	Aroclor 1254	0.93	Schimmel et al., 1974	Demersal
Sheepshead minnow (fry) Cyprinodon variegatus	Aroclor 1254	0.1 to 0.32	Ernst, 1984	Demersal
Eastern oyster Crassostrea virginica	Aroclor 1016	10.2	Hansen et al., 1974a	Benthic
Eastern oyster Crassostrea virginica	Aroclor 1248	17	Lowe, undated	Benthic
Eastern oyster Crassostrea virginica	Aroclor 1260	60	Lowe, undated	Benthic
Eastern oyster Crassostrea virginica	Aroclor 1254	14	Lowe, undated	Benthic
Pinfish Lagodon rhomboides	Aroclor 1254	0.5	Ernst, 1984	Demersal

TABLE B-1
PCB ACUTE TOXICITY DATA FOR MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Spot Leiostomus xanthurus	Aroclor 1254	0.5	Ernst, 1984	Demersal
Cladoceran, Daphnia magna	Aroclor 1254	1.8	Nebeker & Puglisi, 1974	Nekton/Plankton
Cladoceran, Daphnia magna	Aroclor 1254	1.3	Nebeker & Puglisi, 1974	Nekton/Plankton
Cladoceran, Daphnia magna	Aroclor 1254	24	Maki & Johnson, 1975	Nekton/Plankton
Cladoceran, Daphnia magna	Aroclor 1248	2.6	Nebeker & Puglisi, 1974	Nekton/Plankton
Cladoceran, Daphnia magna	Aroclor 1221	180	Nebeker & Puglisi, 1974	Nekton/Plankton
Cladoceran, Daphnia magna	Aroclor 1232	72	Nebeker & Puglisi, 1974	Nekton/Plankton
Cladoceran, Daphnia magna	Aroclor 1242	67	Nebeker & Puglisi, 1974	Nekton/Plankton
Cladoceran, Daphnia magna	Aroclor 1260	36	Nebeker & Puglisi, 1974	Nekton/Plankton

Table taken from USEPA, 1980, and Eisler, 1985.

Toxicity data for the cladoceran, Daphnia magna, are included because these values were used during the extrapolation process.

TABLE B-2
OTHER DATA ON EFFECTS OF PCBs ON MARINE ORGANISMS

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

SPECIES	CHEMICAL	DURATION	EFFECT	RESULT (ug/l)	REFERENCE	HABITAT GROUP
Chlorophyceae Dunallella tertiolecta	Aroclor 1254	-	Increased cell division	100	Harding & Phillips, 1978	Nekton/Plankton
Chrysophyceae Monochrysis lutheri	Aroclor 1254	-	Reduced cell division	10	Harding & Phillips, 1978	Nekton/Plankton
Diatom Thalassiosira pseudonana	Aroclor 1254	-	Reduced cell division	1	Harding & Phillips, 1978	Nekton/Plankton
Diatom Skeletonema costatum	Aroclor 1254	-	Reduced growth	10	Mosser et al.,1972a	Nekton/Plankton
Diatom Rhizosolenia setiger	Aroclor 1254	48 hours	No growth in 48	0.1	Fisher & Wurster, 1973	Nekton/Plankton
Diatom Thalassiosira pseudonana	Aroclor 1254	-	Reduced growth	25 to 100	Mosser et al.,1972b	Nekton/Plankton
Diatom Nitzschia lonsissima	Aroclor 1254	-	No effect on cell	100	Harding & Phillips, 1978	Nekton/Plankton
Diatom Skeletoma costatum	Aroclor 1254	-	Reduced cell division	10	Harding & Phillips, 1978	Nekton/Plankton
Diatom Cylindortheca closterium	Aroclor 1254	-	Reduced growth	100	Kell et al., 1971	Nekton/Plankton
Diatom, Thalassiosira pseudonana and green alga	Aroclor 1254	-	Species ratio change	1	Mosser et al.,1972a	Nekton/Plankton
Haptophyceae Isochrysisgalbana	Aroclor 1254	-	Reduced cell division	1	Harding & Phillips, 1978	Nekton/Plankton
Natural phytoplankton community	Aroclor 1254	-	Decreased diversity,	100	Laird, 1973	Nekton/Plankton
Phytoplankton populations	Aroclor 1254	-	Toxicity in 24 hours	6.5	Moore & Hariss, 1972	Nekton/Plankton
Phytoplankton populations	Aroclor 1254	-	Toxicity in 24 hours	15	Moore & Hariss, 1972	
Diatoms Thalassiosira pseudonana and Skeletomema costatum	Aroclor 1254	-	Reduced growth and carbon	10	Fisher et al., 1973	Nekton/Plankton

TABLE B-2
OTHER DATA ON EFFECTS OF PCBs ON MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Diatom, Thalassiosira pseudonana and green alga Dunaliella tertiolecta	Aroclor 1254	-	Species ratio change	0.1	Fisher et al., 1974	Nekton/Plankton
Diatom Chaetoceros socialis	Aroclor 1254	-	Reduced cell divsion	10	Harding & Phillips, 1978	Nekton/Plankton
Eastern Oyster Crassostrea virginica	Aroclor 1254	24 weeks	Reduced growth	5	Lowe et al., 1972	Benthic
Amphipod, Gammarus oceanicus	Aroclor 1254	30 days	Mortality	>= 10	Wildish, 1970	Benthic
Grass shrimp, Palaemonetes pugio	Aroclor 1254	1 hour	Avoidance	10	Hansen et al., 1974b	Demersal
Grass shrimp, Palaemonetes pugio	Aroclor 1254	4 days	Water efflux affected and altered metabolic state	25 to 45	Roesljadl et al.,1976a,b	Demersal
Grass shrimp, Palaemonetes pugio	Aroclor 1254	96 hours	LC50	6.1 to 7.8	Ernst, 1984	Demersal
Pink shrimp, Penaeus deorarum	Aroclor 1248	48 hours	LC50	32	Lowe, undated	Demersal
Pink shrimp, Penaeus deorarum	Aroclor 1254	48 hours	LC50	32	Lowe, undated	Demersal
Pink shrimp, Penaeus deorarum	Arclor 1254	48 hours	51% Mortality	0.94	Nimmo et al., 1971	Demersal
Pink shrimp, Penaeus deorarum	Aroclor 1254	48 hours	LC50	1	Nimmo & Bahner, 1976	Demersal
Ciliate protozoans, Tetrahymena pyriformis	Aroclor 1248	96 hours	Reduced growth	1000	Cooley et al., 1973	Nekton/Plankton
Ciliate protozoans, Tetrahymena pyriformis	Aroclor 1254	96 hours	Reduced growth	1	Cooley et al., 1973	Nekton/Plankton
Ciliate protozoans, Tetrahymena pyriformis	Aroclor 1260	96 hours	Reduced growth	1000	Cooley et al., 1973	Nekton/Plankton
Fiddler crab, Uca pugilator	Aroclor 1254	38 days	Inhibited molting	8	Fingerman & Fingerman, 1977	Benthic
Fiddler crab, Uca pugilator	Aroclor 1242	4 days	Greater dispersion of melanin	2000	Fingerman & Fingerman, 1978	Benthic
Communities of organisms	Aroclor 1254	4 months	Affected composition	0.6	Hansen, 1974	

TABLE B-2
OTHER DATA ON EFFECTS OF PCBs ON MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Spot, Leiostomus xanthurus	Aroclor 1254	-	Liver pathogenesis	5	Nimmo et al., 1975	Demersal
Spot, Leiostomus xanthurus	Aroclor 1254	20 to 45 days	51 to 62% mortality	5	Hansen et al., 1971	Demersal
Spot, Leiostomus xanthurus	Aroclor 1254	96 hours	LC50	0.5	Ernst, 1984	Demersal
Pinfish, Lagodon rhomboides	Aroclor 1254	1 hour	Avoidance	10	Hansen et al., 1974b	Demersal
Pinfish, Lagodon rhomboides	Aroclor 1254	96 hours	LC50	0.5	Ernst, 1984	Demersal
Pinfish, Lagodon rhomboides	Aroclor 1254	14 to 35 days	41 to 66% mortality	5	Hansen et al., 1971	Demersal
Pinfish, Lagodon rhomboides	Aroclor 1016	42 days	50% mortality	21	Hansen et al., 1974a	Demersal
Sheepshead minnow (adult) Cyprinodon variegatus	Aroclor 1254	28 days	Lethaargy, reduced feeding, fin rot, mortality	10	Hansen et al., 1973	Demersal
Sheepshead minnow (juvenile) Cyprinodon variegatus	Aroclor 1254	21 days	Mortality	10	Schimmel et al., 1974	Demersal
Sheepshead minnow (embryos and fry) Cyprinodon variegatus	Aroclor 1254	21 days	LC50	0.93	Schimmel et al., 1974	Demersal
Sheepshead minnow (fry) Cyprinodon variegatus	Aroclor 1254	21 days	LC50	0.1 to 0.32	Ernst, 1984	Demersal
Sheepshead minnow Cyprinodon variegatus	Aroclor 1254	28 days	Significantly affected hatching of embryos or the survival of fry	0.14	Hansen et al., 1973	Demersal
Sheepshead minnow Cyprinodon variegatus	Aroclor 1016	-	Chronic value	3.4 to 15.0	Hansen et al., 1975	Demersal
Sheepshead minnow Cyprinodon variegatus	Aroclor 1254	-	Chronic value	0.06 to 0.16	Hansen et al., 1974	Demersal
Atlantic cod, Sadus morhua	Aroclor 1254	-	Impaired bone development and abnormalities in	0.4	Sangalang et al., 1981	Nekton/Plankton

Table taken from USEPA, 1980.

TABLE 8-3
BIOCONCENTRATION DATA FOR PCBs - MARINE ORGANISMS

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

SPECIES	TISSUE	LIPID %	CHEMICAL	BIOCONCENTRATION FACTOR	DURATION (days)	REFERENCE	HABITAT GROUP
Diatom, Cylindrotheca closterium	Whole organism	-	Aroclor 1242	1,000	14	Kell et al., 1971	Nekton/Plankton
Polychaete, Nereis diversicolor	Whole body	-	Pnenochlor DP-5	800	14	Fowler, et al., 1978	Benthic
Eastern oyster, Crassostrea virginica	Edible portion	-	Aroclor 1016	13,000	84	Parrish et al., 1974	Demersal
Eastern oyster, Crassostrea virginica	Edible portion	-	Aroclor 1254	101,000	245	Lowe et al., 1972	Demersal
Eastern oyster, Crassostrea virginica	Edible portion	-	Aroclor 1254	>100,000	Field data	Duke et al., 1979; Nimmo et al., 1975	Demersal
Grass shrimp, Palaemonetes pugio	Whole body	-	Aroclor 1254	27,000	16	Nimmo et al., 1974	Demersal
Blue crab, Callinectes sapidus	Whole body	-	Aroclor 1254	>230,000	Field data	Nimmo et al., 1975	Demersal
Spot, Leiostomus xanthurum	Whole body	1.1	Aroclor 1254	37,000	28	Hansen et al., 1971	Demersal
Sheepshead minnow (adult) Cyprinodon variegatus	Whole body	3.6	Aroclor 1016	25,000	28	Hansen et al., 1975	Nekton/Plankton
Sheepshead minnow (juvenile) Cyprinodon variegatus	Whole body	-	Aroclor 1016	43,100	28	Hansen et al., 1975	Nekton/Plankton
Sheepshead minnow (fry) Cyprinodon variegatus	Whole body	-	Aroclor 1016	14,400	28	Hansen et al., 1975	Nekton/Plankton
Sheepshead minnow (adult) Cyprinodon variegatus	Whole body	3.6	Aroclor 1254	30,000	28	Hansen et al., 1973	Nekton/Plankton
Pinfish, Lagodon rhomboides	Whole body	-	Aroclor 1016	17,000	21-28	Hansen et al., 1974a	Nekton/Plankton
Speckled trout, Cynoscion nebulosus	Whole body	-	Aroclor 1254	>670,000	Field data	Duke et al., 1970; Nimmo et al., 1975	Nekton/Plankton
Fishes	Whole body	-	Aroclor 1254	>133,000	Field data	Nimmo et al., 1975	
Invertebrates	Whole body	-	Aroclor 1254	>27,000	Field data	Nimmo et al., 1975	

Table taken from USEPA, 1980.

TABLE B-4
COPPER ACUTE TOXICITY DATA FOR MARINE ORGANISMS

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

SPECIES	METHOD	CHEMICAL	LC50 or EC50 (ug/l)	SPECIES MEAN ACUTE VALUE (ug/l)	REFERENCE	HABITAT GROUP
Polychaete worm, Phyllodoce maculata	S, U	Copper sulfate	120.00	120.00	McLusky & Phillips, 1975	Benthic
Polychaete worm, Neanthes arenaceodentata	FT, M	Copper nitrate	77.00	-	Pesch & Morgan, 1978	Benthic
Polychaete worm, Neanthes arenaceodentata	FT, M	Copper nitrate	200.00	-	Pesch & Morgan, 1978	Benthic
Polychaete worm, Neanthes arenaceodentata	FT, M	Copper nitrate	222.00	150.60	Pesch & Hoffman, 1982	Benthic
Polychaete worm, Nereis diversicolor	S, U	Copper sulfate	200.00	-	Jones et al., 1976	Benthic
Polychaete worm, Nereis diversicolor	S, U	Copper sulfate	445.00	-	Jones et al., 1976	Benthic
Polychaete worm, Nereis diversicolor	S, U	Copper sulfate	480.00	-	Jones et al., 1976	Benthic
Polychaete worm, Nereis diversicolor	S, U	Copper sulfate	410.00	363.80	Jones et al., 1976	Benthic
Blue mussel (embryo) Mytilus edulis	S, U	Copper sulfate	5.80	5.80	Martin et al., 1981	Benthic
Pacific oyster (embryo), Crassostrea gigas	S, U	Copper sulfate	5.30	-	Martin et al., 1981	Benthic
Pacific oyster (embryo), Crassostrea gigas	S, U	Copper sulfate	11.50	-	Cogilanese & Martin, 1981	Benthic
Pacific oyster (adult), Crassostrea gigas	FT, M	Copper sulfate	560.00	7.80	Okazaki, 1976	Benthic
Eastern oyster (embryo), Crassostrea virginica	S, U	Copper chloride	128.00	-	Calabrese et al., 1973	Benthic
Eastern oyster (embryo), Crassostrea virginica	S, U	Copper chloride	15.10	-	Macinnes & Calabrese, 1978	Benthic
Eastern oyster (embryo), Crassostrea virginica	S, U	Copper chloride	18.70	-	Macinnes & Calabrese, 1978	Benthic
Eastern oyster (embryo), Crassostrea virginica	S, U	Copper chloride	18.30	28.52	Macinnes & Calabrese, 1978	Benthic

TABLE B-4
COPPER ACUTE TOXICITY DATA FOR MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Wedge clam, Rangia cuneata	S, U	-	8000.00	-	Olson & Harrel, 1973	Benthic
Wedge clam, Rangia cuneata	S, U	-	7400.00	7694.00	Olson & Harrel, 1973	Benthic
Soft-shelled clam, Mya arenaria	S, U	Copper chloride	39.00	39.00	Eisler, 1977	Benthic
Copepod, Pseudodiaptomus coronatus	S, U	Copper chloride	138.00	138.00	Gentile, 1982	Nekton/Plankton
Copepod, Eurytemora affinis	S, U	Copper chloride	526.00	526.00	Gentile, 1982	Nekton/Plankton
Copepod, Acartia clausi	S, U	Copper chloride	52.00	52.00	Gentile, 1982	Nekton/Plankton
Copepod, Acartia tonsa	S, U	Copper chloride	17.00	-	Sosnowski & Gentile, 1978	Nekton/Plankton
Copepod, Acartia tonsa	S, U	Copper chloride	55.00	-	Sosnowski & Gentile, 1978	Nekton/Plankton
Copepod, Acartia tonsa	S, U	Copper chloride	31.00	30.72	Sosnowski & Gentile, 1978	Nekton/Plankton
Mysid, Mysidopsis bahia	FT, M	Copper nitrate	181.00	181.00	Lussler et al., Manuscript	Demersal
Mysid, Mysidopsis bigelowi	FT, M	Copper nitrate	141.00	141.00	Gentile, 1982	Demersal
American lobster (larva), Homarus americanus	S, U	Copper sulfate	48.00	-	Johnson & Gentile, 1979	Demersal
American lobster (adult), Homarus americanus	S, U	Copper sulfate	100.00	69.28	McLeese, 1974	Demersal
Dungeness crab (larva), Cancer magister	S, U	Copper sulfate	49.00	49.00	Martin, et al., 1981	Demersal
Green crab (larva), Carcinus maenas	S, U	Copper nitrate	600.00	600.00	Conner, 1972	Demersal
Sheepshead minnow, Cyprinondon variegatus	S, U	Copper nitrate	280.00	280.00	Hansen, 1983	Demersal
Atlantic silverside (larva), Menidia menidia	FT, M	Copper nitrate	66.60	-	Cardin, 1982	Demersal

TABLE B-4
COPPER ACUTE TOXICITY DATA FOR MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Atlantic silverside (larva), Menidia menidia	FT, M	Copper nitrate	216.50	-	Cardin, 1982	Demersal
Atlantic silverside (larva), Menidia menidia	FT, M	Copper nitrate	101.80	-	Cardin, 1982	Demersal
Atlantic silverside (larva), Menidia menidia	FT, M	Copper nitrate	97.60	-	Cardin, 1982	Demersal
Atlantic silverside (larva), Menidia menidia	FT, M	Copper nitrate	155.90	-	Cardin, 1982	Demersal
Atlantic silverside (larva), Menidia menidia	FT, M	Copper nitrate	197.60	-	Cardin, 1982	Demersal
Atlantic silverside (larva), Menidia menidia	FT, M	Copper nitrate	190.90	135.60	Cardin, 1982	Demersal
Tidewater silverside, Menidia peninsulae	S, U	Copper nitrate	140.00	140.00	Hansen, 1983	Demersal
Florida pompano Trachinotus carolinus	S, U	Copper sulfate	360.00	-	Birdsong & Avavit, 1971	Nekton/Plankton
Florida pompano Trachinotus carolinus	S, U	Copper sulfate	380.00	-	Birdsong & Avavit, 1971	Nekton/Plankton
Florida pompano Trachinotus carolinus	S, U	Copper sulfate	510.00	411.70	Birdsong & Avavit, 1971	Nekton/Plankton
Summer flounder (embryo), Paralichthys dentatus	FT, M	Copper nitrate	16.30	-	Cardin, 1982	Demersal
Summer flounder (embryo), Paralichthys dentatus	FT, M	Copper nitrate	11.90	-	Cardin, 1982	Demersal
Summer flounder (embryo), Paralichthys dentatus	FT, M	Copper chloride	111.80	13.93	Cardin, 1982	Demersal
Winter flounder (embryo), Pseudopleuronectes americanus	FT, M	Copper nitrate	77.50	-	Cardin, 1982	Demersal
Winter flounder (embryo), Pseudopleuronectes americanus	FT, M	Copper nitrate	167.30	-	Cardin, 1982	Demersal
Winter flounder (embryo), Pseudopleuronectes americanus	FT, M	Copper nitrate	52.70	-	Cardin, 1982	Demersal
Winter flounder (embryo), Pseudopleuronectes americanus	FT, M	Copper nitrate	158.00	-	Cardin, 1982	Demersal

TABLE B-4
COPPER ACUTE TOXICITY DATA FOR MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Winter flounder (embryo), Pseudopleuronectes americanus	FT, M	Copper chloride	173.70	-	Cardin, 1982	Demersal
Winter flounder (embryo), Pseudopleuronectes americanus	FT, M	Copper nitrate	271.00	-	Cardin, 1982	Demersal
Winter flounder (embryo), Pseudopleuronectes americanus	FT, M	Copper chloride	132.80	-	Cardin, 1982	Demersal
Winter flounder (embryo), Pseudopleuronectes americanus	FT, M	Copper nitrate	148.20	-	Cardin, 1982	Demersal
Winter flounder (embryo), Pseudopleuronectes americanus	FT, M	Copper nitrate	98.20	128.90	Cardin, 1982	Demersal

Table taken from USEPA, 1985b.

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

TABLE B-5
OTHER DATA ON EFFECTS OF COPPER ON MARINE ORGANISMS

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

SPECIES	CHEMICAL	DURATION	EFFECT	RESULT (ug/l)	REFERENCE	HABITAT GROUP
Alga, Amphidinium carteri	Copper	14 Days	EC50 Growth rate	<50	Erickson et al., 1970	Nekton/Plankton
Diatom, Skeletonema costatum	Copper	14 day EC50	Growth rate	5.00	Erickson et al., 1970	Nekton/Plankton
Diatom, Thalassiosira aestevallis	Copper	-	Reduced chlorophyll a	19.00	Hollibaugh et al., 1980	Nekton/Plankton
Diatom, Thalassiosira aestevallis	Copper	3 day EC50	Growth rate	5.00	Erickson, 1972	Nekton/Plankton
Diatom, Asterionella japonica	Copper	3 day EC50	Growth rate	12.70	Fisher & Jones, 1981	Nekton/Plankton
Alga, Olisthodiscus luteus	Copper	14 days	EC50 Growth rate	<50	Erickson et al., 1970	Nekton/Plankton
Alga, Nitschia closterium	Copper	4 days	EC50 Growth rate	33.00	Rosko & Rachlin, 1975	Nekton/Plankton
Alga, Scrippsiella faeroense	Copper	5 days	EC50 Growth rate	5.00	Saifullah, 1978	Nekton/Plankton
Alga, Prorocentrum micans	Copper	5 days	EC50 Growth rate	10.00	Saifullah, 1978	Nekton/Plankton
Alga, Gymnodinium splendons	Copper	5 days	EC50 Growth rate	20.00	Saifullah, 1978	Nekton/Plankton
Red alga, Champia parvula	Copper	-	Reduced tetrasporophyte growth	4.60	Steele & Thursby, 1983	Nekton/Plankton
Red alga, Champia parvula	Copper	-	Reduced tetrasporangia production	13.30	Steele & Thursby, 1983	Nekton/Plankton
Red alga, Champia parvula	Copper	-	Reduced female growth	4.70	Steele & Thursby, 1983	Nekton/Plankton
Red alga, Champia parvula	Copper	-	Stopped sexual reproduction	7.30	Steele & Thursby, 1983	Nekton/Plankton
Natural phytoplankton population	Copper	5 days	Reduced chlorophyll a	19.00	Hollibaugh et al., 1980	Nekton/Plankton
Natural phytoplankton population	Copper	4 days	Reduced biomass	6.40	Hollibaugh et al., 1980	Nekton/Plankton

TABLE B-5
OTHER DATA ON EFFECTS OF COPPER ON MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Hydroid, Campanularia flexuosa	Copper	11 days	Growth rate inhibition	10-13	Stebbing, 1976	Benthic
Hydroid, Campanularia flexuosa	Copper	-	Enzyme inhibition	1.43	Moore & Stebbing, 1976	Benthic
Hydromedusa, Phalalldium spp.	Copper	1 day	LC50	36.00	Reeve et al., 1976	Benthic
Ctenophore, Pleurobrachia pileus	Copper	1 day	LC50	33.00	Reeve et al., 1976	Nekton/Plankton
Ctenophore, Mnemiopsis mccrdayl	Copper	1 day	LC50	17-29	Reeve et al., 1976	Nekton/Plankton
Rotifer, Brachionus plicatillis	Copper	1 day	LC50	100.00	Reeve et al., 1976	Nekton/Plankton
Polychaete worm, Phyllodoce maculata	Copper	9 days	LC50	80.00	McLusky & Phillips, 1975	Benthic
Polychaete worm, Neanthes arenaceodentata	Copper	28 days	LC50	44.00	Pesch & Morgan, 1978	Benthic
Polychaete worm, Neanthes arenaceodentata	Copper	28 days	LC50	100.00	Pesch & Morgan, 1978	Benthic
Polychaete worm, Neanthes arenaceodentata	Copper	7 days	LC50	137.00	Pesch & Morgan, 1982	Benthic
Polychaete worm, Neanthes arenaceodentata	Copper	10 days	LC50	98.00	Pesch & Morgan, 1982	Benthic
Polychaete worm, Neanthes arenaceodentata	Copper	28 days	LC50	56.00	Pesch & Morgan, 1982	Benthic
Polychaete worm, Cirriformia spirabranchia	Copper	26 days	LC50	40.00	Milanovich et al., 1976	Benthic
Larval annelids, Mixed species	Copper	1 day	LC50	89.00	Reeve et al., 1976	Benthic
Channeled whelk, Busycon canaliculatum	Copper	77 days	LC50	470.00	Betzer & Yevich, 1975	Benthic
Mud snail, Nassarius obsoletus	Copper	3 days	Decrease in oxygen consumption	100.00	MacInnes & Thurberg, 1973	Demersal
Blue mussel, Mytilus edulis	Copper	7 days	LC50	200.00	Scott & Major, 1972	Benthic

TABLE B-5
OTHER DATA ON EFFECTS OF COPPER ON MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Bay scallop, Argopecton irradians	Copper	42 days	EC50 (growth)	5.80	Pesch et al., 1979	Benthic
Bay scallop, Argopecton irradians	Copper	119 days	100% mortality	5.00	Zaroogian & Johnson, 1983	Benthic
Eastern oyster, Crassostrea virginica	Copper	12 days	LC50	46.00	Calabrese et al., 1977	Benthic
Wedge clam, Rangia cuneata	Copper	4 days	LC50 (<1 g/kg salinity)	210.00	Olson & Harrel, 1973	Benthic
Clam, Macoma inquinata	Copper	30 days	LC50	15.70	Crecellus et al., 1982	Benthic
Clam, Macoma inquinata	Copper	30 days	LC50	20.70	Crecellus et al., 1982	Benthic
Quahog clam (larva), Mercenaria mercenaria	Copper	8-10 days	LC50	30.00	Calabrese et al., 1977	Benthic
Quahog clam (larva), Mercenaria mercenaria	Copper	77 days	LC50	25.00	Shuster & Pringle, 1968	Benthic
Common Pacific littleneck, Protothaca staminea	Copper	17 days	LC50	39.00	Roesijadi, 1980	Benthic
Soft-shelled clam, Mya arenaria	Copper	7 days	LC50	35.00	Eisler, 1977	Benthic
Copepod, Undinula vulgaris	Copper	1 day	LC50	192.00	Reeve et al., 1976	Nekton/Plankton
Copepod, Euchaeta marina	Copper	1 day	LC50	188.00	Reeve et al., 1976	Nekton/Plankton
Copepod, Metridia pacifica	Copper	1 day	LC50	176.00	Reeve et al., 1976	Nekton/Plankton
Copepod, Labidocera scotti	Copper	1 day	LC50	132.00	Reeve et al., 1976	Nekton/Plankton
Copepod, Acartia clausi	Copper	2 days	LC50	34-82	Moraltou-Apostolopoulou, 1978	Nekton/Plankton
Copepod, Acartia tonsa	Copper	6 days	LC50	9-73	Sosnowski et al., 1979	Nekton/Plankton

Table taken from USEPA, 1985b.

TABLE B-6
CADMIUM ACUTE TOXICITY DATA FOR MARINE ORGANISMS

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

SPECIES	METHOD	CHEMICAL	LC50 or EC50 (ug/l)	SPECIES MEAN ACUTE VALUE (ug/l)	REFERENCE	HABITAT GROUP
Polychaete worm (adult), Neanthes arenaceodentata	S, U	Cadmium chloride	12000	-	Reish, et al., 1976	Benthic
Polychaete worm (juvenile), Neanthes arenaceodentata	S, U	Cadmium chloride	12500	12250	Reish, et al., 1976	Benthic
Sand Worm Nereis virens	S, U	Cadmium chloride	9300	-	Eisler & Hennekey, 1977	Benthic
Polychaete worm Nereis virens	S, U	Cadmium chloride	11000	10110	Eisler, 1971	Benthic
Polychaete worm (adult) Capitella capitella	S, U	Cadmium chloride	7500	-	Reish, et al., 1976	Benthic
Polychaete worm (larvae) Capitella capitella	S, U	Cadmium chloride	200	200	Reish, et al., 1976	Benthic
Oligochaete worm Limnodriloides verrucosus	R, U	Cadmium sulfate	10000	10000	Chapman, et al., 1982a	Benthic
Oligochaete worm Monophylephorus cuticalatus	R,U	Cadmium sulfate	135000	135000	Chapman, et al., 1982a	Benthic
Oligochaete worm Tubificoides gabriellae	R, U	Cadmium sulfate	24000	24000	Chapman, et al., 1982a	Benthic
Oyster drill Urosalpinx cinerea	S, U	Cadmium chloride	6600	6600	Eisler, 1971	Benthic
Mud snail Nassarius oboletus	S, U	Cadmium chloride	35000	-	Eisler & Hennekey, 1977	Benthic
Mud snail Nassarius oboletus	S, U	Cadmium chloride	10500	19170	Eisler, 1971	Benthic
Blue mussel Mytilus edulis	S, U	Cadmium chloride	25000	-	Eisler, 1971	Benthic
Blue mussel (embryo), Mytilus edulis	S, U	Cadmium chloride	1200	-	Martin, et al., 1981	Benthic
Blue mussel Mytilus edulis	S, M	Cadmium chloride	1620	-	Ahsanullah, 1976	Benthic
Blue mussel Mytilus edulis	S, M	Cadmium chloride	3600	-	Ahsanullah, 1976	Benthic

TABLE B-6
CADMIUM ACUTE TOXICITY DATA FOR MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Blue mussel Mytilus edulis	FT, M	Cadmium chloride	4300	3934	Ahsanullah, 1976	Benthic
Bay Scallop (juvenile) Argopecten irradiens	FT, M	Cadmium chloride	1480	1480	Nelson, et al., 1976	Benthic
Pacific Oyster Crassostrea gigas	S, U	Cadmium chloride	611	-	Martin, et al., 1981	Benthic
Pacific Oyster Crassostrea gigas	S, U	Cadmium chloride	85	227.9	Watling, 1982	Benthic
Atlantic Oyster Crassostrea virginica	S, U	Cadmium chloride	3800	3800	Calabrese, et al., 1973	Benthic
Soft-shell clam Mya arenaria	S, U	Cadmium chloride	2500	-	Eisler & Hennekey, 1977	Benthic
Soft-shell clam Mya arenaria	S, U	Cadmium chloride	2200	-	Eisler, 1971	Benthic
Soft-shell clam, Mya arenaria	S, U	Cadmium chloride	850	1672	Eisler, 1977	Benthic
Copepod, Pseudolaptomus coronatus	S, U	Cadmium chloride	1708	1708	Gentile, 1982	Nekton/Plankton
Copepod, Eurytemora affinis	S, U	Cadmium chloride	1080	-	Gentile, 1982	Nekton/Plankton
Copepod (nauplius), Eurytemora affinis	S, U	Cadmium chloride	147.7	399.4	Sullivan et al., 1983	Nekton/Plankton
Copepod, Acartia clausi	S, U	Cadmium chloride	144	144	Gentile, 1982	Nekton/Plankton
Copepod, Acarti tonsa	S, U	Cadmium chloride	90	-	Sosnowski & Gentile, 1978	Nekton/Plankton
Copepod, Acarti tonsa	S, U	Cadmium chloride	122	-	Sosnowski & Gentile, 1978	Nekton/Plankton
Copepod, Acarti tonsa	S, U	Cadmium chloride	220	-	Sosnowski & Gentile, 1978	Nekton/Plankton
Copepod, Acarti tonsa	S, U	Cadmium chloride	337	168.9	Sosnowski & Gentile, 1978	Nekton/Plankton
Copepod, Nitocra spinipes	S, U	Cadmium chloride	1800	1800	Bengtsson, 1978	Nekton/Plankton

TABLE B-6
CADMIUM ACUTE TOXICITY DATA FOR MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Mysid, Mysidopsis bahia	FT, M	Cadmium chloride	15.5	-	Nimmo, et at., 1977a	Demersal
Mysid, Mysidopsis bahia	FT, M	Cadmium chloride	110	41.29	Gentile, et al., 1982 Lussler, et al., Manuscript	Demersal
Mysid, Mysidopsis bigelowi	FT, M	Cadmium chloride	110	110	Gentile, et al., 1982	Demersal
Amphipod (adult), Ampeilsca abdita	S, M	Cadmium chloride	2900	2900	Scott, et al., Manuscript	Benthic
Amphipod (young), Marinogammarus obtusatus	S, M	Cadmium chloride	3500	-	Wright & Frain, 1981	Benthic
Amphipod (adult), Marinogammarus obtusatus	S, M	Cadmium chloride	13000	3500	Wright & Frain, 1981	Benthic
Pink Shrimp Penaeus duorarum	FT, M	Cadmium chloride	3500	3500	Nimmo, et at., 1977b	Demersal
Grass shrimp, Palaemonetes vulgaris	S, U	Cadmium chloride	420	-	Eisler, 1971	Demersal
Grass shrimp, Palaemonetes vulgaris	FT, M	Cadmium chloride	760	760	Nimmo, et at., 1977b	Demersal
Sand shrimp, Crangon septemspinosa	S, U	Cadmium chloride	320	320	Eisler, 1971	Benthic
American lobster (larvae), Homarus americanus	S, U	Cadmium chloride	78	78	Johnson & Gentile, 1979	Demersal
Hermit crab, Pagurus longicarpus	S, U	Cadmium chloride	320	-	Eisler, 1971	Benthic
Hermit crab, Pagurus longicarpus	S, U	Cadmium chloride	1300	645	Eisler & Hennekey, 1977	Benthic
Rock crab (zoea), Cancer irroratus	FT, M	Cadmium chloride	250	250	Johns & Miller, 1982	Demersal
Dungeness crab (zoea), Cancer magister	S, U	Cadmium chloride	247	247	Martin, et al., 1981	Demersal
Blue crab (juvenile), Callinectes sapidus	S, U	Cadmium chloride	11600	-	Frank & Robertson, 1979	Demersal
Blue crab (juvenile), Callinectes sapidus	S, U	Cadmium chloride	4700	7384	Frank & Robertson, 1979	Demersal

TABLE B-6
CADMIUM ACUTE TOXICITY DATA FOR MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Green crab, *Carcinus maenas*	S, U	Cadmium chloride	4100	4100	Eisler, 1971	Demersal
Fiddler crab, *Uca pugilator*	S, U	Cadmium chloride	46600	-	O'Hara, 1973a	Benthic
Fiddler crab, *Uca pugilator*	S, U	Cadmium chloride	37000	-	O'Hara, 1973a	Benthic
Fiddler crab, *Uca pugilator*	S, U	Cadmium chloride	32300	-	O'Hara, 1973a	Benthic
Fiddler crab, *Uca pugilator*	S, U	Cadmium chloride	23300	-	O'Hara, 1973a	Benthic
Fiddler crab, *Uca pugilator*	S, U	Cadmium chloride	10400	-	O'Hara, 1973a	Benthic
Fiddler crab, *Uca pugilator*	S, U	Cadmium chloride	6800	21240	O'Hara, 1973a	Benthic
Starfish, *Asterias forbesi*	S, U	Cadmium chloride	7100	-	Eisler & Hennekey, 1977	Benthic
Starfish, *Asterias forbesi*	S, U	Cadmium chloride	820	2413	Eisler, 1971	Benthic
Sheepshead minnow, *Cyperindon variegatus*	S, U	Cadmium chloride	50000	50000	Eisler, 1971	Demersal
Mummichog (adult), *Fundulus heteroclitus*	S, U	Cadmium chloride	49000	-	Eisler, 1971	Demersal
Mummichog (adult), *Fundulus heteroclitus*	S, U	Cadmium chloride	22000	-	Eisler & Hennekey, 1977	Demersal
Mummichog (juvenile), *Fundulus heteroclitus*	S, U	Cadmium chloride	114000	-	Voyer, 1975	Demersal
Mummichog (juvenile), *Fundulus heteroclitus*	S, U	Cadmium chloride	92000	-	Voyer, 1975	Demersal
Mummichog (juvenile), *Fundulus heteroclitus*	S, U	Cadmium chloride	78000	-	Voyer, 1975	Demersal
Mummichog (juvenile), *Fundulus heteroclitus*	S, U	Cadmium chloride	73000	-	Voyer, 1975	Demersal
Mummichog (juvenile), *Fundulus heteroclitus*	S, U	Cadmium chloride	63000	-	Voyer, 1975	Demersal

TABLE B-6
CADMIUM ACUTE TOXICITY DATA FOR MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Mummichog (juvenile), Fundulus heteroclitus	S, U	Cadmium chloride	31000	-	Voyer, 1975	Demersal
Mummichog (juvenile), Fundulus heteroclitus	S, U	Cadmium chloride	30000	-	Voyer, 1975	Demersal
Mummichog (juvenile), Fundulus heteroclitus	S, U	Cadmium chloride	29000	50570	Voyer, 1975	Demersal
Stripped killfish (adult), Fundelus majallis	S, U	Cadmium chloride	21000	21000	Eisler, 1971	Demersal
Atlantic silverside, Menidia menidia	S, U	Cadmium chloride	2032	-	Cardin, 1982	Demersal
Atlantic silverside, Menidia menidia	S, U	Cadmium chloride	28532	-	Cardin, 1982	Demersal
Atlantic silverside, Menidia menidia	S, U	Cadmium chloride	13652	-	Cardin, 1982	Demersal
Atlantic silverside (larvae), Menidia menidia	S, U	Cadmium chloride	1054	-	Cardin, 1982	Demersal
Atlantic silverside (larvae), Menidia menidia	S, U	Cadmium chloride	577	779.8	Cardin, 1982	Demersal
Winter flounder (larvae), Pseudopleuronectes americanus	S, U	Cadmium chloride	602	-	Cardin, 1982	Benthic
Winter flounder (larvae), Pseudopleuronectes americanus	S, U	Cadmium chloride	14297	14297	Cardin, 1982	Benthic

Table taken from USEPA, 1985a.

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

TABLE B-7
OTHER DATA ON EFFECTS OF CADMIUM ON MARINE ORGANISMS

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

SPECIES	CHEMICAL	DURATION	EFFECT	RESULT (ug/l)	REFERENCE	HABITAT GROUP
Natural phytoplankton population	Cadmium chloride	4 days	Reduced biomass	112	Hollibaugh et al., 1980	Nekton/Plankton
Diatom, Asterionella japonica	Cadmium chloride	3 days	72-hr EC50 growth rate	224.8	Fisher & Jones, 1981	Nekton/Plankton
Diatom, Ditylum brightwelli	Cadmium chloride	5 Days	EC50 Growth	60	Centerford & Centerford, 1980	Nekton/Plankton
Diatom, Thalassiosira pseudonana	Cadmium chloride	4 days	EC50 Growth rate	160	Gentile & Johnson, 1982	Nekton/Plankton
Diatom, Skeletoma costatum	Cadmium chloride	4 days	EC50 Growth rate	175	Gentile & Johnson, 1982	Nekton/Plankton
Red alga, Champia parvula	Cadmium chloride	-	Reduced tetrasporophyte growth	24.9	Steele & Thursby, 1983	Benthic
Red alga, Champia parvula	Cadmium chloride	-	Reduced tetrasporangia production	>189	Steele & Thursby, 1983	Benthic
Red alga, Champia parvula	Cadmium chloride	-	Reduced female growth	22.8	Steele & Thursby, 1983	Benthic
Red alga, Champia parvula	Cadmium chloride	-	Stopped sexual reproduction	22.8	Steele & Thursby, 1983	Benthic
Hydroid, Campanularia flexuosa	-	-	Enzyme inhibition	40-75	Moore & Stebbling, 1976	Benthic
Hydroid, Campanularia flexuosa	-	11 days	Growth Rate	110-280	Stebbling, 1976	Benthic
Polychaete worm, Neanthes arenaceodentata	Cadmium chloride	28 days	LC50	3000	Reish et al., 1976	Benthic
Polychaete worm, Capitella capitata	Cadmium chloride	28 days	LC50	630	Reish et al., 1976	Benthic
Polychaete worm, Capitella capitata	Cadmium chloride	28 days	LC50	700	Reish et al., 1976	Benthic
Blue mussel Mytilus edulis	Cadmium EDTA	28 days	BCF=252	-	George & Coombs, 1977	Benthic
Blue mussel Mytilus edulis	Cadmium alginate	28 days	BCF=252	-	George & Coombs, 1977	Benthic

TABLE B-7
OTHER DATA ON EFFECTS OF CADMIUM ON MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Blue mussel Mytilus edulis	Cadmium humate	28 days	BCF=252	-	George & Coambs, 1977	Benthic
Blue mussel Mytilus edulis	Cadmium pectate	28 days	BCF=252	-	George & Coambs, 1977	Benthic
Blue mussel Mytilus edulis	Cadmium chloride	21 days	BCF=710	-	Janssen & Scholz, 1979	Benthic
Bay scallop, Argopecton irradians	Cadmium chloride	42 days	EC50 (growth reduction)	78	Pesch & Stewart, 1980	Benthic
Bay scallop, Argopecton irradians	Cadmium chloride	21 days	BCF=168		Eisler et al., 1972	Benthic
Eastern oyster, Crassostrea virginica	Cadmium iodide	40 days	BCF=677		Kerfoot & Jacobs, 1976	Benthic
Eastern oyster, Crassostrea virginica	Cadmium chloride	21 days	BCF=149		Eisler et al., 1972	Benthic
Eastern oyster, Crassostrea virginica	Cadmium chloride	2 days	Reduction in embryonic development	15	Zarooglan & Morrison, 1981	Benthic
Pacific oyster, Crassostrea gigas	Cadmium chloride	6 days	50% reduction in settlement	20-25	Watling, 1983b	Benthic
Pacific oyster, Crassostrea gigas	Cadmium chloride	14 days	Growth reduction	10	Watling, 1983b	Benthic
Pacific oyster, Crassostrea gigas	Cadmium chloride	23 days	LC50	50	Watling, 1983b	Benthic
Soft-shell clam, Mya arenaria	Cadmium chloride	7 days	LC50	150	Eisler, 1977	Benthic
Soft-shell clam, Mya arenaria	Cadmium chloride	7 days	LC50	700	Eisler & Hennekey, 1977	Benthic
Copepod (naupilus), Eurytemora affinis	Cadmium chloride	1 day	Reduction in swimming speed	130	Sulivan et al., 1983	Nekton/Plankton
Copepod, (naupilus), Eurytemora affinis	Cadmium chloride	2 days	Reduction in development rate	116	Sulivan et al., 1983	Nekton/Plankton
Copepod, Tisbe holothuriae	Cadmium chloride	2 days	LC50	970	Moraitou-Apostolopoulou & Verriopoulos, 1982	Nekton/Plankton
Mysid, Mysidopsis bahia	-	17 days	LC50 (15-23 g/kg salinity)	11	Nimmo et al., 1977a	Demersal

TABLE B-7
OTHER DATA ON EFFECTS OF CADMIUM ON MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Mysid, Mysidopsis bahia	Cadmium chloride	16 days	LC50 (30 g/kg salinity)	28	Gentile et al., 1982	Demersal
Mysid, Mysidopsis bahia	Cadmium chloride	8 days	LC50	60	Gentile et al., 1982	Demersal
Mysid, Mysidopsis bigelowi	Cadmium chloride	8 days	LC50	70	Gentile et al., 1982	Demersal
Mysid, Mysidopsis bigelowi	Cadmium chloride	28 days	LC50	18	Gentile et al., 1982	Demersal
Isopod, Idotea baltica	Cadmium sulfate	5 days	LC50 (3 g/kg salinity)	10000	Jones, 1975	Benthic
Isopod, Idotea baltica	Cadmium sulfate	3 days	LC50 (21 g/kg salintiy)	10000	Jones, 1975	Benthic
Isopod, Idotea baltica	Cadmium sulfate	1.5 days	LC50 (14 g/kg salintiy)	10000	Jones, 1975	Benthic
Pink shrimp, Penaeus duorarum	Cadmium chloride	30 days	LC50	720	Nimmo et al., 1977b	Demersal
Grass shrimp, Palaemonetes pugio	Cadmium chloride	42 days	LC50	300	Pesch & Stewart, 1980	Demersal
Grass shrimp, Palaemonetes pugio	Cadmium chloride	21 days	LC25 (5 g/kg salinity)	50	Vernberg et al., 1977	Demersal
Grass shrimp, Palaemonetes pugio	Cadmium chloride	21 days	LC10 (10 g/kg salinity)	50	Vernberg et al., 1977	Demersal
Grass shrimp, Palaemonetes pugio	Cadmium chloride	21 days	LC5 (20 g/kg salinity)	50	Vernberg et al., 1977	Demersal
Grass shrimp, Palaemonetes pugio	Cadmium chloride	6 days	LC75 (10 g/kg salintiy)	300	Middaugh & Floyd, 1978	Demersal
Grass shrimp, Palaemonetes pugio	Cadmium chloride	6 days	LC50 (15 g/kg salinity)	300	Middaugh & Floyd, 1978	Demersal
Grass shrimp, Palaemonetes pugio	Cadmium chloride	6 days	LC25 (30 g/kg salinity)	300	Middaugh & Floyd, 1978	Demersal
Grass shrimp, Palaemonetes pugio	Cadmium chloride	21 days	BCF=140	-	Vernberg et al., 1977	Demersal
Grass shrimp, Palaemonetes vulgaris	Cadmium chloride	29 days	LC50	120	Nimmo et al., 1977b	Demersal

TABLE B-7
OTHER DATA ON EFFECTS OF CADMIUM ON MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

American lobster, Homarus americanus	Cadmium chloride	21 days	BCF=25	-	Eisler et al., 1972	Benthic
American lobster, Homarus americanus	Cadmium chloride	30 days	Increase in ATPase activity	6	Tucker, 1979	Benthic
Hermit crab, Pagurus longicarpus	Cadmium chloride	7 days	25% mortality	270	Eisler & Hennekey, 1977	Benthic
Hermit crab, Pagurus longicarpus	Cadmium chloride	60 days	LC56	70	Pesch & Stewart, 1980	Benthic
Rock crab, Cancer irroratus	Cadmium chloride	4 days	Enzyme activity	1000	Gould et al., 1976	Demersal
Rock crab (larvae), Cancer irroratus	Cadmium chloride	28 days	Delayed development	50	Johns & Miller, 1982	Demersal
Blue crab, Callinectes sapidus	Cadmium chloride	7 days	LC50 (10 g/kg salinity)	50	Rosenberg & Costlow, 1976	Demersal
Blue crab, Callinectes sapidus	Cadmium nitrate	7 days	LC50 (30 g/kg salinity)	150	Rosenberg & Costlow, 1976	Demersal
Blue crab (juvenile), Callinectes sapidus	Cadmium nitrate	4 days	LC50 (1 g/kg salinity)	320	Frank & Robertson, 1979	Demersal
Mud crab (larva), Eurypanopeus depressus	Cadmium chloride	8 days	LC50	10	Mirkes, et al., 1978	Benthic
Mud crab (larva), Eurypanopeus depressus	Cadmium chloride	44 days	Delay in metamorphysis	10	Mirkes, et al., 1978	Benthic
Mud crab, Rhithropanopeus harrisii	Cadmium nitrate	11 days	LC80 (10 g/kg salinity)	50	Rosenberg & Costlow, 1976	Benthic
Mud crab, Rhithropanopeus harrisii	Cadmium nitrate	11 days	LC75 (20 g/kg salinity)	50	Rosenberg & Costlow, 1976	Benthic
Mud crab, Rhithropanopeus harrisii	Cadmium nitrate	11 days	LC40 (30 g/kg salinity)	50	Rosenberg & Costlow, 1976	Benthic
Fiddler crab, Uca puilator	-	10 days	LC50	2900	O'Hara, 1973a	Benthic
Fiddler crab, Uca puilator	Cadmium chloride	-	Effect on respiration	1	Vernburg, et al., 1974	Benthic
Starfish, Asterias forbesi	Cadmium chloride	7 days	25% mortality	270	Eisler & Hennekey, 1977	Benthic

TABLE B-7
OTHER DATA ON EFFECTS OF CADMIUM ON MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Herring (larvae), Clupea harengus	Cadmium chloride	-	100% embryonic survival	5000	Westernhagen et al., 1979	Nekton
Pacific Herring (embryo), Clupea harengus pallasi	Cadmium chloride	< 1 day	17% reduction in volume	10000	Alderdice et al., 1979a	Nekton
Pacific Herring (embryo), Clupea harengus pallasi	Cadmium chloride	4 days	Decrease in capsule strength	1000	Alderdice et al., 1979b	Nekton
Pacific Herring (embryo), Clupea harengus pallasi	Cadmium chloride	2 days	Reduced osmolality of perivitetine fluid	1000	Alderdice et al., 1979c	Nekton
Mummichog (adult), Fundulus heteroclitus	Cadmium chloride	2 days	LC50 (20 g/kg salinity)	60000	Middaugh & Dean, 1977	Demersal
Mummichog (adult), Fundulus heteroclitus	Cadmium chloride	2 days	LC50 (30 g/kg salinity)	43000	Middaugh & Dean, 1977	Demersal
Mummichog, Fundulus heteroclitus	Cadmium chloride	21 days	BCF=48	-	Eisler, et al., 1972	Demersal
Mummichog (larva), Fundulus heteroclitus	Cadmium chloride	2 days	LC50 (20 g/kg salinity)	32000	Middaugh & Dean, 1977	Demersal
Mummichog (larva), Fundulus heteroclitus	Cadmium chloride	2 days	LC50 (30 g/kg salinity)	7800	Middaugh & Dean, 1977	Demersal
Atlantic silverside, Menidia menidia	Cadmium chloride	2 days	LC50 (20 g/kg salinity)	13000	Middaugh & Dean, 1977	Demersal
Atlantic silverside, Menidia menidia	Cadmium chloride	2 days	LC50 (30 g/kg salinity)	12000	Middaugh & Dean, 1977	Demersal
Atlantic silverside, Menidia menidia	Cadmium chloride	19 days	LC50 (12 g/kg salinity)	160	Voyer et al., 1979	Demersal
Atlantic silverside, Menidia menidia	Cadmium chloride	19 days	LC50 (20 g/kg salinity)	540	Voyer et al., 1979	Demersal
Atlantic silverside, Menidia menidia	Cadmium chloride	19 days	LC50 (30 g/kg salinity)	970	Voyer et al., 1979	Demersal
Atlantic silverside (larvae), Menidia menidia	Cadmium chloride	2 days	LC50 (20 g/kg salinity)	2200	Middaugh & Dean, 1977	Demersal
Atlantic silverside (larvae), Menidia menidia	Cadmium chloride	2 days	LC50 (30 g/kg salinity)	1600	Middaugh & Dean, 1977	Demersal
Stripped bass (juvenile), Morone saxatilis	Cadmium chloride	90 days	Significant decrease in enzyme activity	5	Dawson et al., 1977	Nekton

TABLE B-7
OTHER DATA ON EFFECTS OF CADMIUM ON MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Stripped bass (juvenile), Morone saxatilis	Cadmium chloride	30 days	Significant decrease in oxygen consumption	0.5-5.0	Dawson et al., 1977	Nekton
Spot (larva), Leiostomus xanthurus	Cadmium chloride	9 days	Incipient LC50	200	Middaugh et al., 1975	Demersal
Cunner (adult), Tautogolabrus adspersus	Cadmium chloride	60 days	37.5% mortallity	100	MacInnes et al., 1977	Demersal
Cunner (adult), Tautogolabrus adspersus	Cadmium chloride	30 days	Depresses gill tissue oxygen consumption	50	MacInnes et al., 1977	Demersal
Cunner (adult), Tautogolabrus adspersus	Cadmium chloride	4 days	Decreased enzyme activity	3000	Gould & Karolus, 1974	Demersal
Winter flounder, Pseudopleuronectes americanus	Cadmium chloride	8 days	50% viable hatch	300	Voyer et al., 1977	Benthic
Winter flounder, Pseudopleuronectes americanus	Cadmium chloride	60 days	Increased gill tissue oxygen respiration	5	Calabrese et al., 1975	Benthic
Winter flounder, Pseudopleuronectes americanus	Cadmium chloride	17 days	Reduction of viable hatch	586	Voyer et al., 1982	Benthic
Diatom, Skeletonema costatum	-	-	Decreased growth	10-25	Berland et al., 1977	Nekton/Plankton
Crab, Pontoporeia affinis	-	265 days	Reduced F1 life span	6.5	Sundelin, 1983	Demersal
Mysid shrimp, Mysidopsis spp.	-	23-27 days	Molt inhibition	10	Gentile et al., 1982	Demersal
Mysid shrimp, Mysidopsis spp.	-	23-27 days	No effect	5.1	Gentile et al., 1982	Demersal

Table taken from USEPA, 1985a, and Eisler, 1985.

TABLE B-8
LEAD ACUTE TOXICITY DATA FOR MARINE ORGANISMS

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

SPECIES	METHOD	CHEMICAL	LC50 or EC50 (ug/l)	SPECIES MEAN ACUTE VALUE (ug/l)	REFERENCE	HABITAT GROUP
Amphipod, Ampelisca abdita	R, U	Lead nitrate	547	547	Scott et al. Manuscript	Benthic
Atlantic silverside, Menidia menidia	S, U	Lead nitrate	10000	>10,000	Berry, 1981	Demersal
Copepod, Acarti clausi	S, U	Lead nitrate	668	668	Gentile, 1982	Nekton/Plankton
Dungeness crab, Cancer magister	-	Lead	575	575	Reish & Gerlinger, 1984	Demersal
Inland silverside, Menidia beryllina	FT, M	Lead nitrate	3140	>3,140	Cardin, 1981	Demersal
Mummichog, Fundulus heteroclitus	S, U	Lead nitrate	315	315	Dorfman, 1977	Demersal
Mysid, Mysidopsis bahia	FT, M	Lead nitrate	3130	3130	Lussier, et al. Manuscript	Demersal
Plaice, Pleuronectes platessa	-	Diethyl Pb	75000	-	Maddock and Taylor, 1980	Demersal
Sheepshead minnow, Cyperinodon variegatus	FT, M	Lead nitrate	3140	-	Cardin, 1981	Demersal
Shrimp, Crangon crangon	-	Trimethyl Pb	8800	-	Maddock and Taylor, 1980	Demersal
Alga, Phaeodactylum tricornutum	-	Trimethyl Pb	800	-	Maddock and Taylor, 1980	Nekton/Plankton
Alga, Phaeodactylum tricornutum	-	Pb+2	>5000	-	Maddock and Taylor, 1980	Nekton/Plankton
Alga, Phaeodactylum tricornutum	-	Triethyl Pb	100	-	Maddock and Taylor, 1980	Nekton/Plankton
Alga, Phaeodactylum tricornutum	-	Tetraethyl Pb	100	-	Maddock and Taylor, 1980	Nekton/Plankton
Alga, Phaeodactylum tricornutum	-	Tetramethyl Pb	1300	-	Maddock and Taylor, 1980	Nekton/Plankton

TABLE B-8
LEAD ACUTE TOXICITY DATA FOR MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Amphipod, Amplisca abdita	-	Lead	547	-	EPA, 1985	Benthic
Dungeness crab, Cancer magister	S, U	Lead nitrate	575	-	Martin et al., 1981	Demersal
Mummichog, Fundulus heteroclitus	-	Lead	315	-	EPA, 1985	Demersal
Plaice, Pleuronectes platessa	-	Tetraethyl Pb	230	-	Maddock and Taylor, 1980	Demersal
Plaice, Pleuronectes platessa	-	Tetramethyl Pb	50	-	Maddock and Taylor, 1980	Demersal
Plaice, Pleuronectes platessa	-	Triethyl Pb	1700	-	Maddock and Taylor, 1980	Demersal
Plaice, Pleuronectes platessa	-	PB+2	180000	-	Maddock and Taylor, 1980	Demersal
Plaice, Pleuronectes platessa	-	Dimethyl Pb	300000	-	Maddock and Taylor, 1980	Demersal
Plaice, Pleuronectes platessa	-	Trimethyl Pb	24600	-	Maddock and Taylor, 1980	Demersal
Shrimp, Crangon crangon	-	Tetramethyl Pb	110	-	Maddock and Taylor, 1980	Demersal
Shrimp, Crangon crangon	-	Tetraethyl Pb	20	-	Maddock and Taylor, 1980	Demersal
Shrimp, Crangon crangon	-	Triethyl Pb	5800	-	Maddock and Taylor, 1980	Demersal
Shrimp, Crangon crangon	-	Pb+2	375000	-	Maddock and Taylor, 1980	Demersal
Blue mussel, Mytilus edulis	-	Pb+2	>500000	-	Maddock and Taylor, 1980	Benthic
Blue mussel, Mytilus edulis	-	Tetraethyl Pb	100	-	Maddock and Taylor, 1980	Benthic
Blue mussel, Mytilus edulis	40 days	Lead chloride	30000	-	Talbot et al., 1976	Benthic
Blue mussel, Mytilus edulis	-	Tetramethyl Pb	270	-	Maddock and Taylor, 1980	Benthic

TABLE B-8
LEAD ACUTE TOXICITY DATA FOR MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Blue mussel, Mytilus edulis	150 days	Lead nitrate	500	-	Schultz-Baldes, 1972	Benthic
Blue mussel, Mytilus edulis	-	Triethyl Pb	1100	-	Maddock and Taylor, 1980	Benthic
Blue mussel, Mytilus edulis	-	Trimethyl Pb	500	-	Maddock and Taylor, 1980	Benthic
Blue mussel, Mytilus edulis	S, U	Lead nitrate	476	-	Martin et al., 1981	Benthic
Blue mussel (larva), Mytilus edulis	-	Pb^{+2}	476	-	EPA, 1985	Benthic
Eastern oyster, Crassostrea viginica	S, U	Lead nitrate	2450	-	Calabrese et al., 1973	Benthic
Pacific oyster, Crassostrea gigas	S, U	Lead nitrate	758	-	Martin et al., 1981	Benthic
Polychaete worm, Ophryotrocha diadema	4 days	Lead acetate	14100	-	Reish et al., 1976	Benthic
Polychaete worm, Ophryotrocha diadema	2 days	Lead acetate	100000	-	Parker, 1984	Benthic
Polychaete worm, Capitella capitata	4 days	Lead acetate	1200	-	Reish et al., 1976	Benthic
Quahog clam (larva), Mercenaria mercenaria	S, U	Lead nitrate	780	-	Calabrese & Nelson, 1974	Benthic
Sandworm, Neanthes arenaceodentata	-	Lead	7700	-	Reish & Gerlinger, 1984	Benthic
Sandworm, Neanthes arenaceodentata	-	Lead	10700	-	Reish & Gerlinger, 1984	Benthic
Soft-shell clam, Mya arenaria	S, U	Lead nitrate	27000	-	Eisler, 1977	Benthic
Soft-shell clam, Mya arenaria	7 days	Lead nitrate	8800	-	Eisler, 1977	Benthic

Table taken from USEPA, 1980, and Eisler, 1988.

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

TABLE B-9
OTHER DATA ON EFFECTS OF LEAD ON MARINE ORGANISMS

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

SPECIES	CHEMICAL	DURATION	EFFECT	RESULT (ug/l)	REFERENCE	HABITAT GROUP
Alga, Dunaliella salina	Lead	-	65% growth reduction	900	Pace et al., 1977	Nekton/Plankton
Alga, Dundaliella tertiolecta	Tetramethyl lead	4 days	EC50	1650	Marchetti, 1978	Nekton/Plankton
Alga, Dundaliella tertiolecta	Tetraethyl lead	4 days	EC50	150	Marchetti, 1978	Nekton/Plankton
Alga, Chorella stigmatophora	Lead	21 days	50% growth inhibition	700	Christensen, et al., 1979	Nekton/Plankton
Alga, Champia parvula	Lead	-	Reduced tetrasporophyte growth	23.3	Steele & Thursby, 1983	Nekton/Plankton
Diatom, Phaeodactylum tricornutum	Lead	3 days	No growth inhibition	1000	Hannan & Patoulliet, 1972	Nekton/Plankton
Diatom, Asterionella japonica	Lead	-	EC50	207	Fisher & Jones, 1981	Nekton/Plankton
Diatom, Ditylum brightwelli	Lead	-	EC50	40	Centerford & Centerford, 1980	Nekton/Plankton
Diatom, Phaeodactylium tricornutum	Lead	1 day	Completely inhibited photosynthis	10000	Woolery & Lewin, 1976	Nekton/Plankton
Diatom, Skeletonema costatum	Lead	12 days	EC50 (growth rate)	3.7	Rivkin, 1979	Nekton/Plankton
Diatom, Phaeodactylum tricornutum	Lead	2-3 days	Reduced photosynthesis and respiration	100	Woolery & Lewin, 1976	Nekton/Plankton
Natural phytoplankton populations	Lead	4 days	Reduced biomass	21	Hollibaugh, et al., 1980	Nekton/Plankton
Natural phytoplankton populations	Lead	5 days	Reduced chlrophyll a	207	Hollibaugh, et al., 1980	Nekton/Plankton
Phytoplankton, Platymonas subcordiformes	Lead	3 days	Retarded population growth	2500	Hessler, 1974	Nekton/Plankton
Eastern oyster Crassostera virginica	Lead	1 yr	BCF = 326	-	Kopfler & Mayer, 1973	Benthic

TABLE B-9
OTHER DATA ON EFFECTS OF LEAD ON MARINE ORGANISMS
(continued)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

Polychaete worm, Ophryotrocha diadema	Lead	21 days	Suppressed reproduction	1000	Relsh & Carr, 1978	Benthic
Polychaete worm, Ctenodrilus serratus	Lead	21 days	Suppressed reproduction	1000	Relsh & Carr, 1978	Benthic
American lobster Homarus americanus	Lead	30 days	Reduced enzyme activity	50	Gould & Greig, 1983	Benthic
Mud crab, Rhithropanopeus harrisii	Lead	-	Delayed larval development	50	Benijts-Claus & Benijts, 1975	Benthic
Mummichog (embyro), Fundulus heterociltus	Lead	-	Depressed axis formation	100	Weis & Weis, 1977	Planktonic
Mummichog (embyro), Fundulus heterociltus	Lead	-	Retarded hatching	10000	Weis & Weis, 1982	Planktonic

Table taken from USEPA, 1980.

TABLE B-10
COPPER MATC ESTIMATES FOR ORGANISMS AT NEW BEDFORD HARBOR

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

TAXON	SLOPE	INTERCEPT	MATC	TOTAL VARIANCE
Marine Fish	1.02	0.75	2.517	1.319
Crustacea	0.8	0.43	1.816	2.708
Mollusca	0.98	-0.6	1.223	0.420
Polychaeta	1.0	-0.88	1.480	0.210
Alga			1.081	0.069

Notes:

(1) The basic regression equation that defines the extrapolation is Y = Intercept + (X * Slope), where X is the acute toxicological estimate and Y the extrapolated MATC value.

(2) No extrapolation was done for the alga, rather chronic data were used to estimate the benchmark value for the taxon.

(3) All units expressed as Log (base 10) ug/l.

TABLE B-11
CADMIUM MATC ESTIMATES FOR ORGANISMS AT NEW BEDFORD HARBOR

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

TAXON	SLOPE	INTERCEPT	MATC	TOTAL VARIANCE
Marine Fish	1.02	0.75	1.505	0.698
Crustacea	0.8	0.43	1.022	1.824
Mollusca	0.98	-0.6	2.757	0.424
Polychaeta	1.0	-0.88	3.106	0.212
Alga			1.997	0.115

Notes:

(1) The basic regression equation that defines the extrapolation is Y = Intercept + (X * Slope), where X is the acute toxicological estimate and Y the extrapolated MATC value.

(2) No extrapolation was done for the alga, rather chronic data were used to estimate the benchmark value for the taxon.

(3) All units expressed as Log (base 10) ug/l.

TABLE B-12
LEAD MATC ESTIMATES FOR ORGANISMS AT NEW BEDFORD HARBOR

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

TAXON	SLOPE	INTERCEPT	MATC	TOTAL VARIANCE
Marine Fish	1.02	0.75	2.176	1.028
Crustacea	0.8	0.43	1.548	2.317
Mollusca	0.98	-0.6	2.433	0.421
Polychaeta	1.0	-0.88	3.149	0.210
Alga			2.370	0.909

Notes:

(1) The basic regression equation that defines the extrapolation is Y = Intercept + (X * Slope), where X is the acute toxicological estimate and Y the extrapolated MATC value.

(2) No extrapolation was done for the alga, rather chronic data were used to estimate the benchmark value for the taxon.

(3) All units expressed as Log (base 10) ug/l.

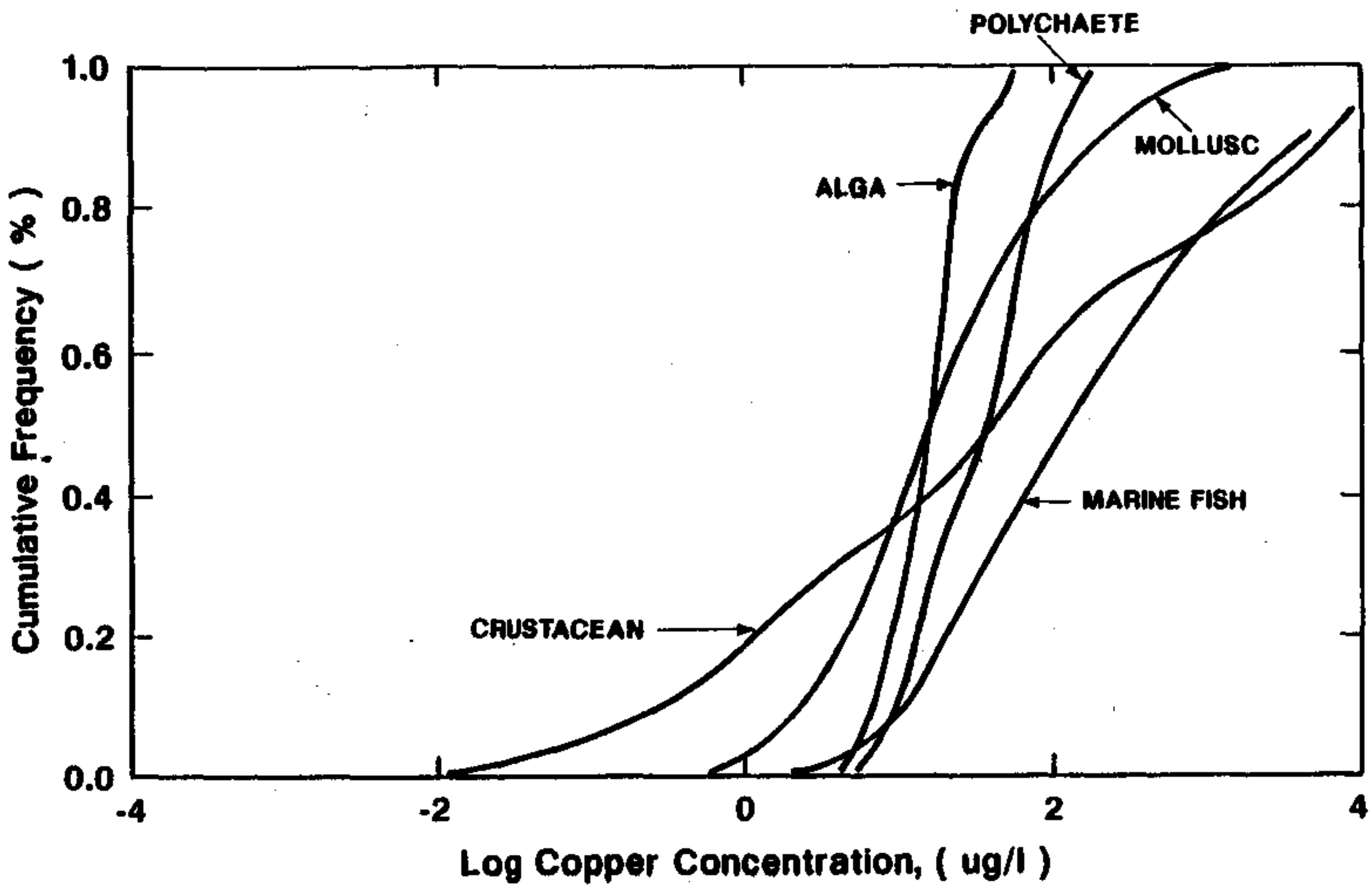

FIGURE B-1
MATC CURVES FOR COPPER
NEW BEDFORD, MASSACHUSETTS

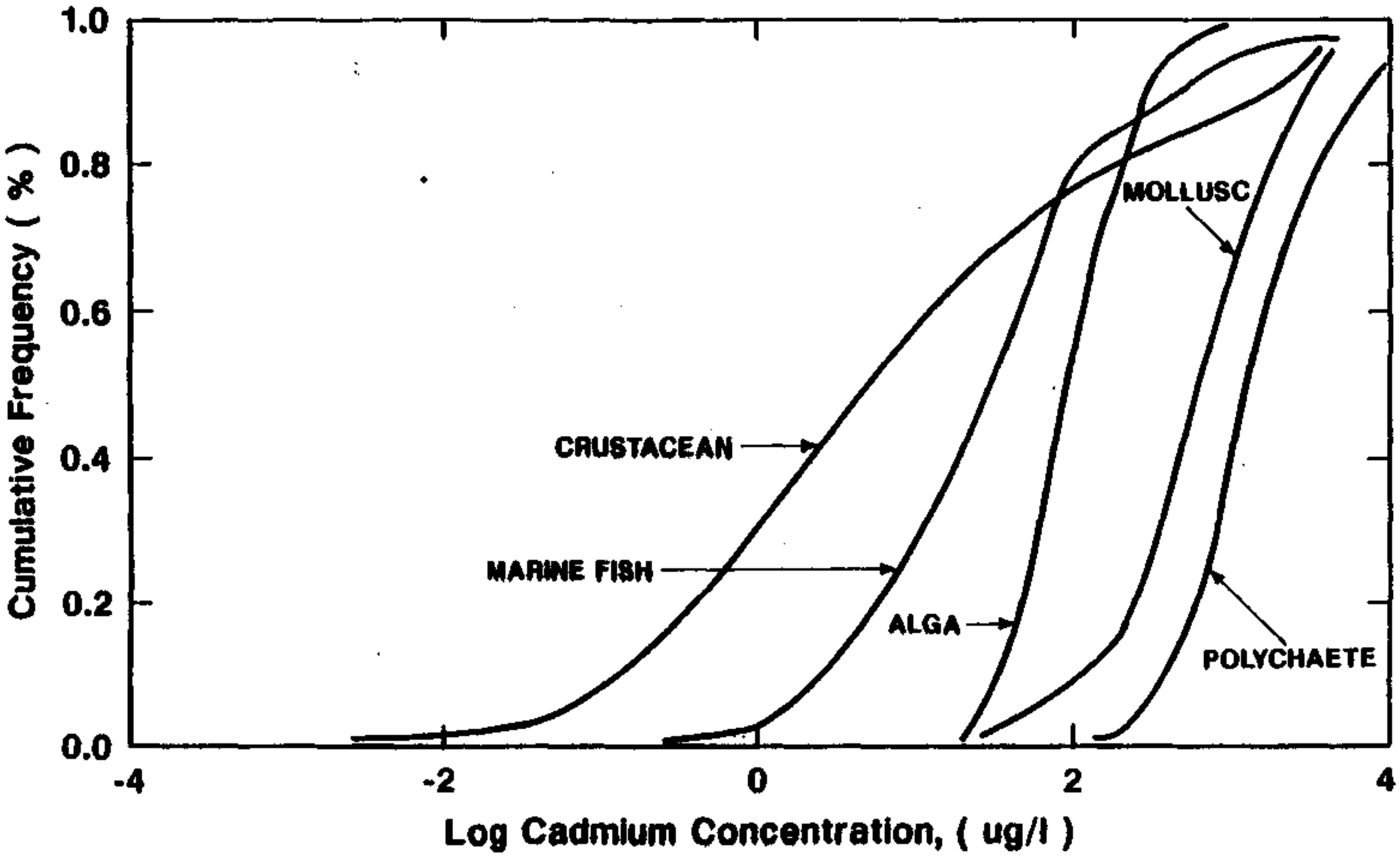

FIGURE B-2
MATC CURVES FOR CADMIUM
NEW BEDFORD, MASSACHUSETTS

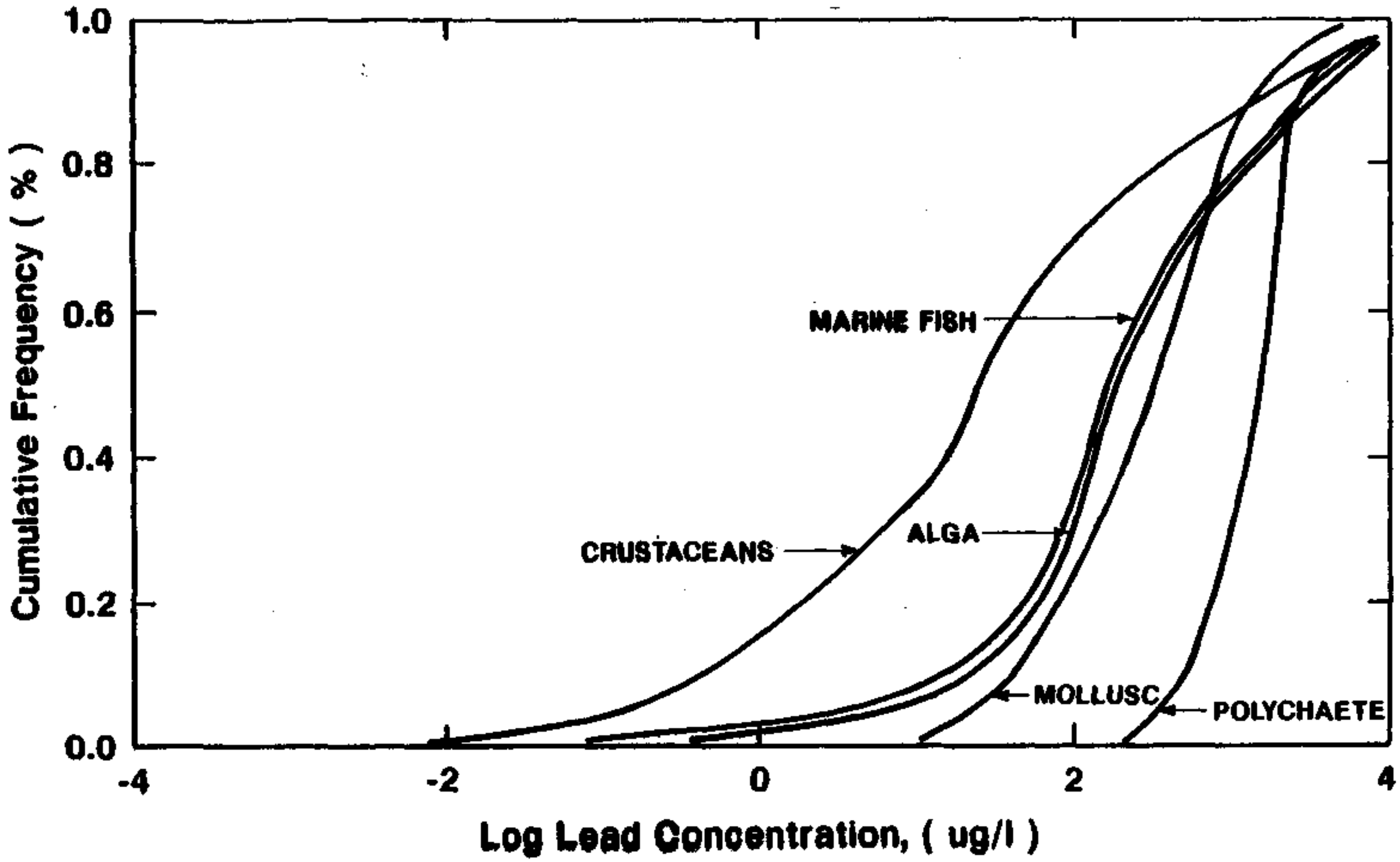

FIGURE B-3
MATC CURVES FOR LEAD
NEW BEDFORD, MASSACHUSETTS

APPENDIX C

MATCs, EECs, and CHRONIC EFFECTS PROBABILITIES

FOR

COPPER, CADMIUM, AND LEAD

TABLE C-1
CUMULATIVE PROBABILITY THAT THE EXPECTED EXPOSURE CONCENTRATION WILL EXCEED THE COPPER MATC FOR THE PARTICULAR TAXON.

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

HARBOR ZONE	MARINE FISH	CRUSTACEA	MOLLUSCA	POLYCHAETA	ALGA
1, Water Column	0.03	0.19	0.09	0.01	0.00
2, Water Column	0.04	0.22	0.15	0.02	0.03
3, Water Column	0.04	0.22	0.15	0.02	0.03
4, Water Column	0.03	0.19	0.10	0.01	0.02
5, Water Column	0.01	0.12	0.03	0.00	0.00
1, Pore Water	0.02	0.11	0.05	0.02	0.04
2, Pore Water	0.01	0.08	0.04	0.02	0.03
3, Pore Water	0.02	0.11	0.05	0.02	0.04
4, Pore Water	0.01	0.08	0.02	0.00	0.01
5, Pore Water	0.00	0.04	0.00	0.00	0.00

Notes:

Probabilities calculated as the area under a normally-distributed curve defined by a particular Z score, where Z = (Mean EEC - BM) / (Var EEC + Var BM)^2. Equation presented by Suter et al., 1986.

EEC = Expected Environmental Concentration

BM = Bench Mark, which in this application are the MATCs developed by extrapolation, in the case of Marine Fish, Crustaceans, Mollusks, and Polychaetes. For Alga, the bench mark was based on available chronic toxicity data.

TABLE C-2
CUMULATIVE PROBABILITY THAT THE EXPECTED EXPOSURE CONCENTRATION WILL EXCEED THE CADMIUM MATC FOR THE PARTICULAR TAXON.

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

HARBOR ZONE	MARINE FISH	CRUSTACEA	MOLLUSCA	POLYCHAETA	ALGA
1, Water Column	0.01	0.11	0.00	0.00	0.00
2, Water Column	0.01	0.14	0.00	0.00	0.00
3, Water Column	0.00	0.10	0.00	0.00	0.00
4, Water Column	0.00	0.08	0.00	0.00	0.00
5, Water Column	0.00	0.05	0.00	0.00	0.00
1, Pore Water	0.01	0.12	0.00	0.00	0.00
2, Pore Water	0.03	0.13	0.00	0.00	0.00
3, Pore Water	0.01	0.09	0.00	0.00	0.00
4, Pore Water	0.00	0.06	0.00	0.00	0.00
5, Pore Water	0.00	0.03	0.00	0.00	0.00

Notes:

Probabilities calculated as the area under a normally-distributed curve defined by a particular Z score, where Z = (Mean EEC - BM) / (Var EEC + Var BM)^2. Equation presented by Suter et al., 1986.

EEC = Expected Environmental Concentration

BM = Bench Mark, which in this application are the MATCs developed by extrapolation, in the case of Marine Fish, Crustaceans, Mollusks, and Polychaetes. For Alga, the bench mark was based on available chronic toxicity data.

TABLE C-3
CUMULATIVE PROBABILITY THAT THE EXPECTED EXPOSURE CONCENTRATION WILL EXCEED THE LEAD MATC FOR THE PARTICULAR TAXON.

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

HARBOR ZONE	MARINE FISH	CRUSTACEA	MOLLUSCA	POLYCHAETA	ALGA
1, Water Column	0.03	0.18	0.00	0.00	0.01
2, Water Column	0.02	0.17	0.00	0.00	0.01
3, Water Column	0.02	0.13	0.00	0.00	0.01
4, Water Column	0.01	0.08	0.00	0.00	0.00
5, Water Column	0.01	0.06	0.00	0.00	0.00
1, Pore Water	0.04	0.18	0.01	0.00	0.03
2, Pore Water	0.03	0.13	0.01	0.00	0.02
3, Pore Water	0.02	0.14	0.00	0.00	0.01
4, Pore Water	0.00	0.06	0.00	0.00	0.00
5, Pore Water	0.01	0.10	0.00	0.00	0.01

Notes:

Probabilities calculated as the area under a normally-distributed curve defined by a particular Z score, where Z = (Mean EEC - BM) / (Var EEC + Var BM)^2. Equation presented by Suter et al., 1986.

EEC = Expected Environmental Concentration

BM = Bench Mark, which in this application are the MATCs developed by extrapolation, in the case of Marine Fish, Crustaceans, Mollusks, and Polychaetes. For Alga, the bench mark was based on available chronic toxicity data.

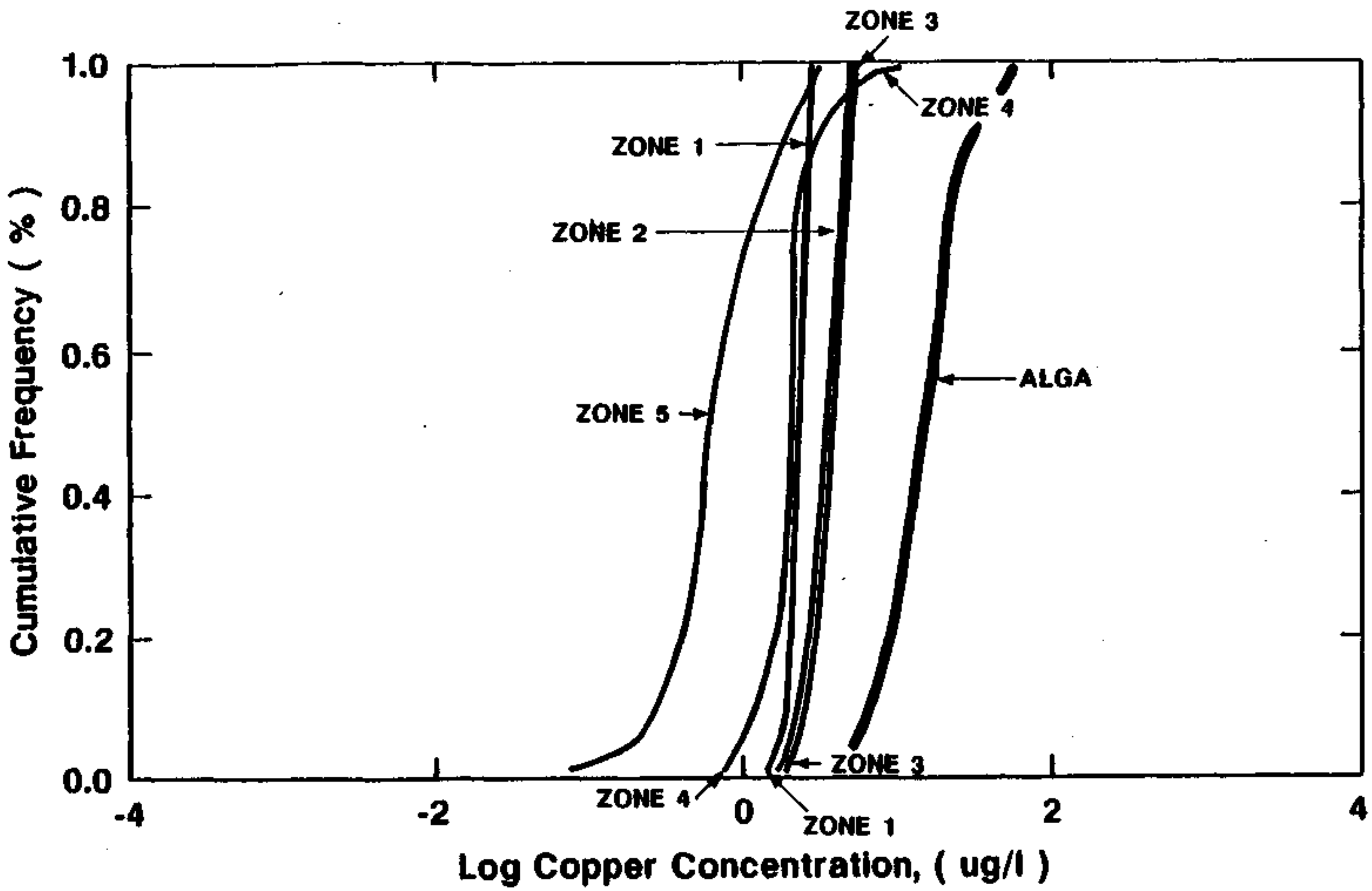

FIGURE C-1
MATC FOR ALGA AND
EECs FOR ALL ZONES, COPPER, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

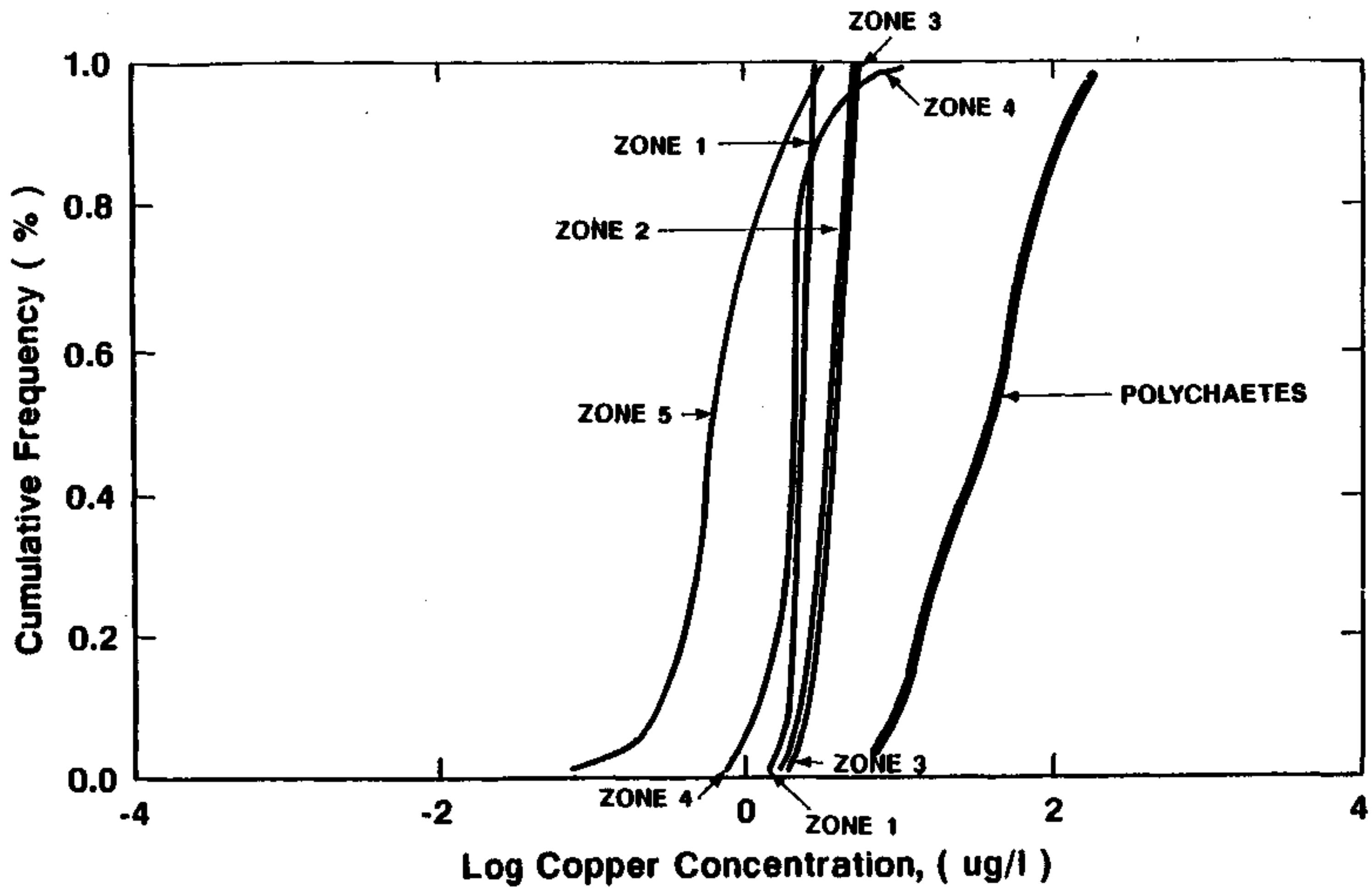

FIGURE C-2
MATC FOR POLYCHAETES AND
EECs FOR ALL ZONES, COPPER, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

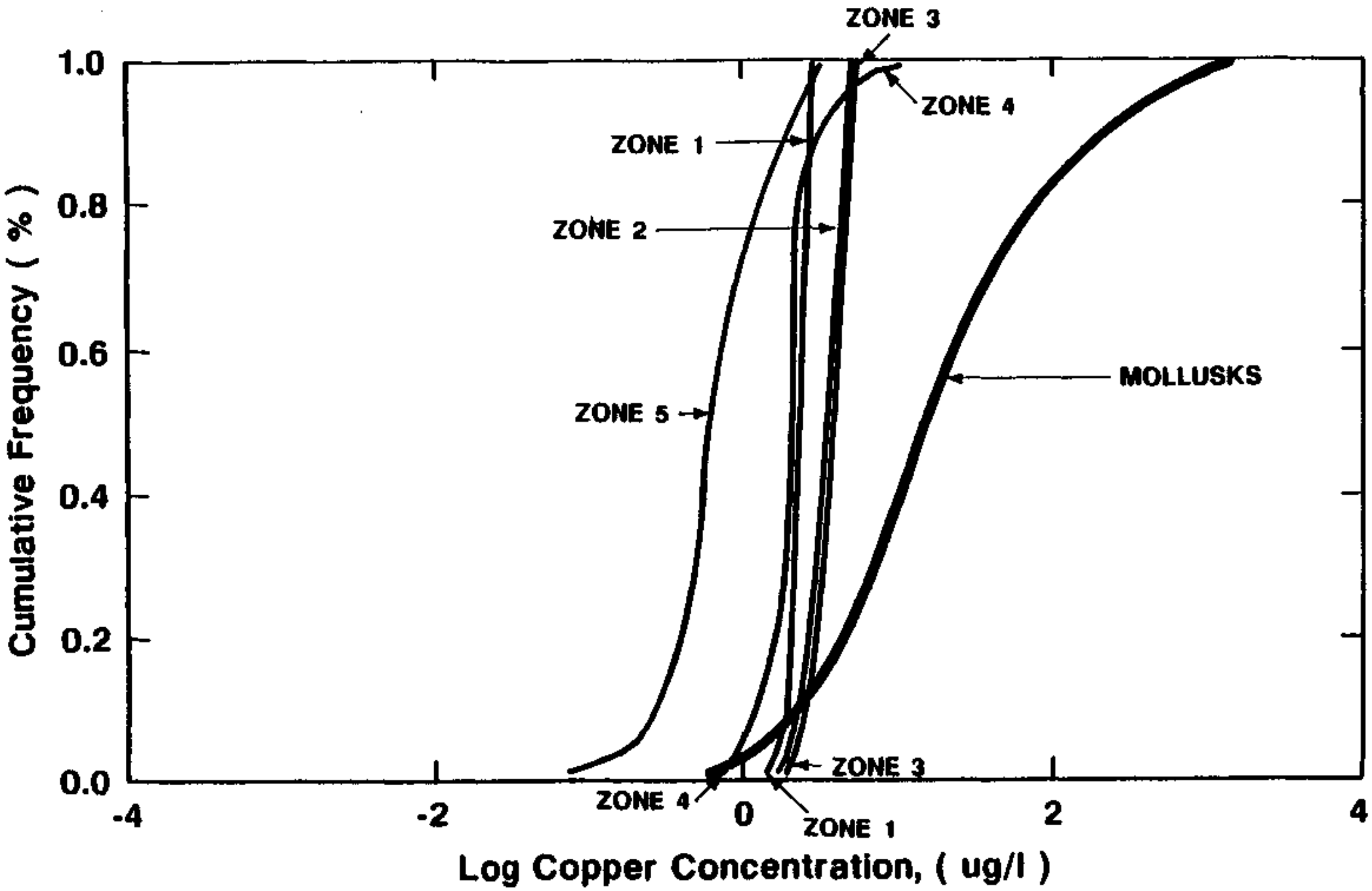

FIGURE C-3
MATC FOR MOLLUSKS AND
EECs FOR ALL ZONES, COPPER, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

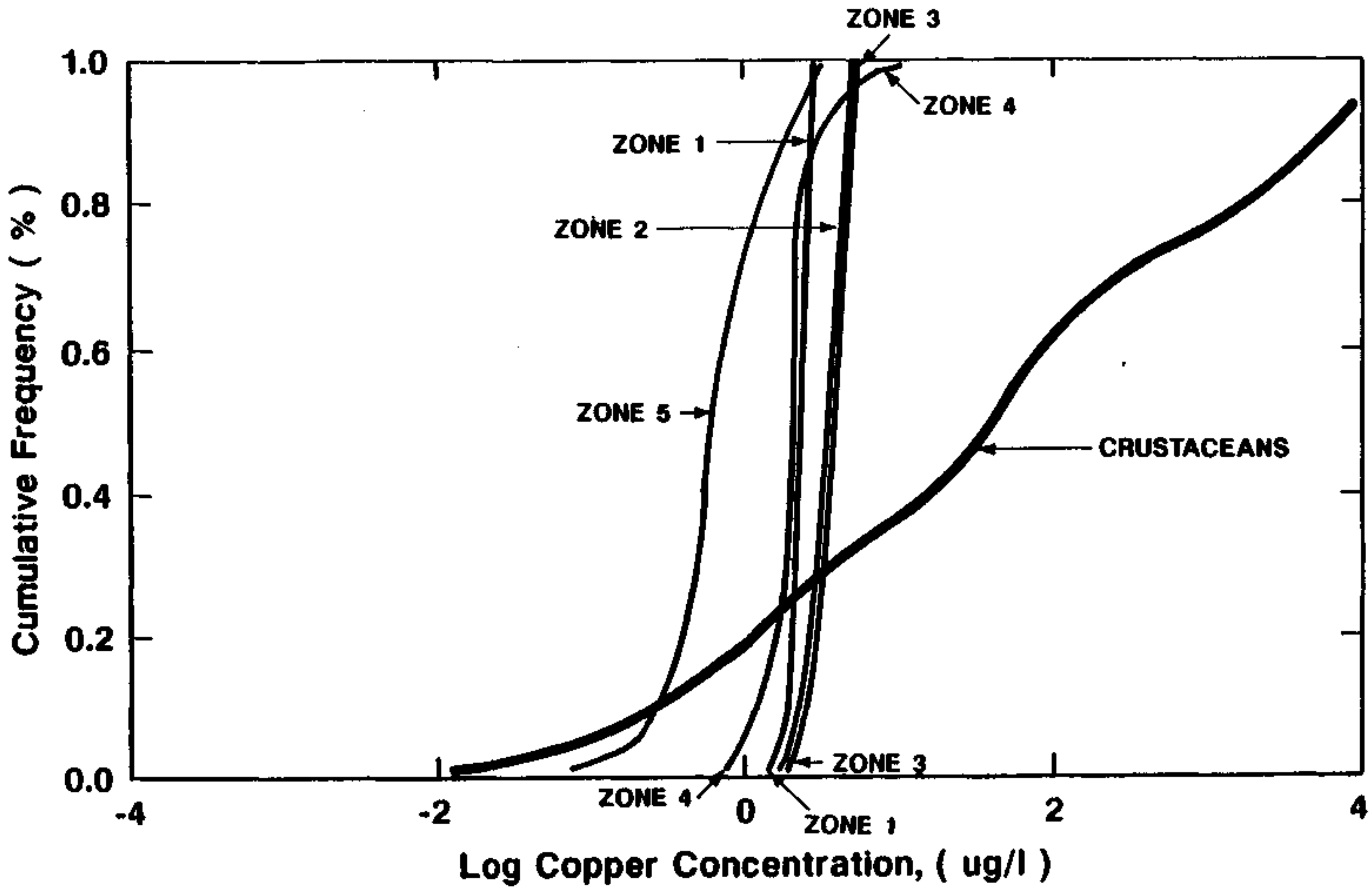

FIGURE C-4
MATC FOR CRUSTACEANS AND
EECs FOR ALL ZONES, COPPER, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

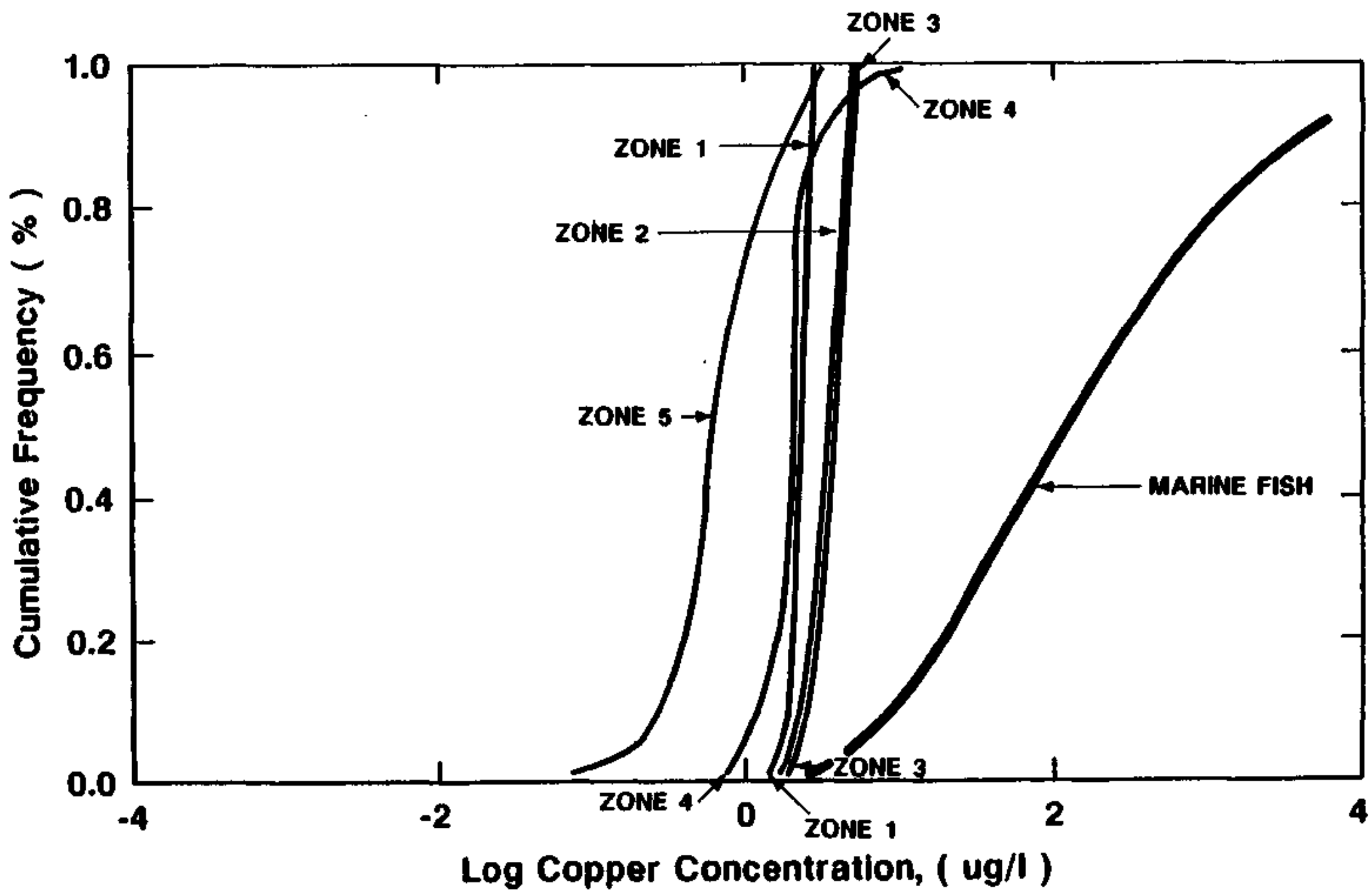

FIGURE C-5
MATC FOR MARINE FISH AND
EECs FOR ALL ZONES, COPPER, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

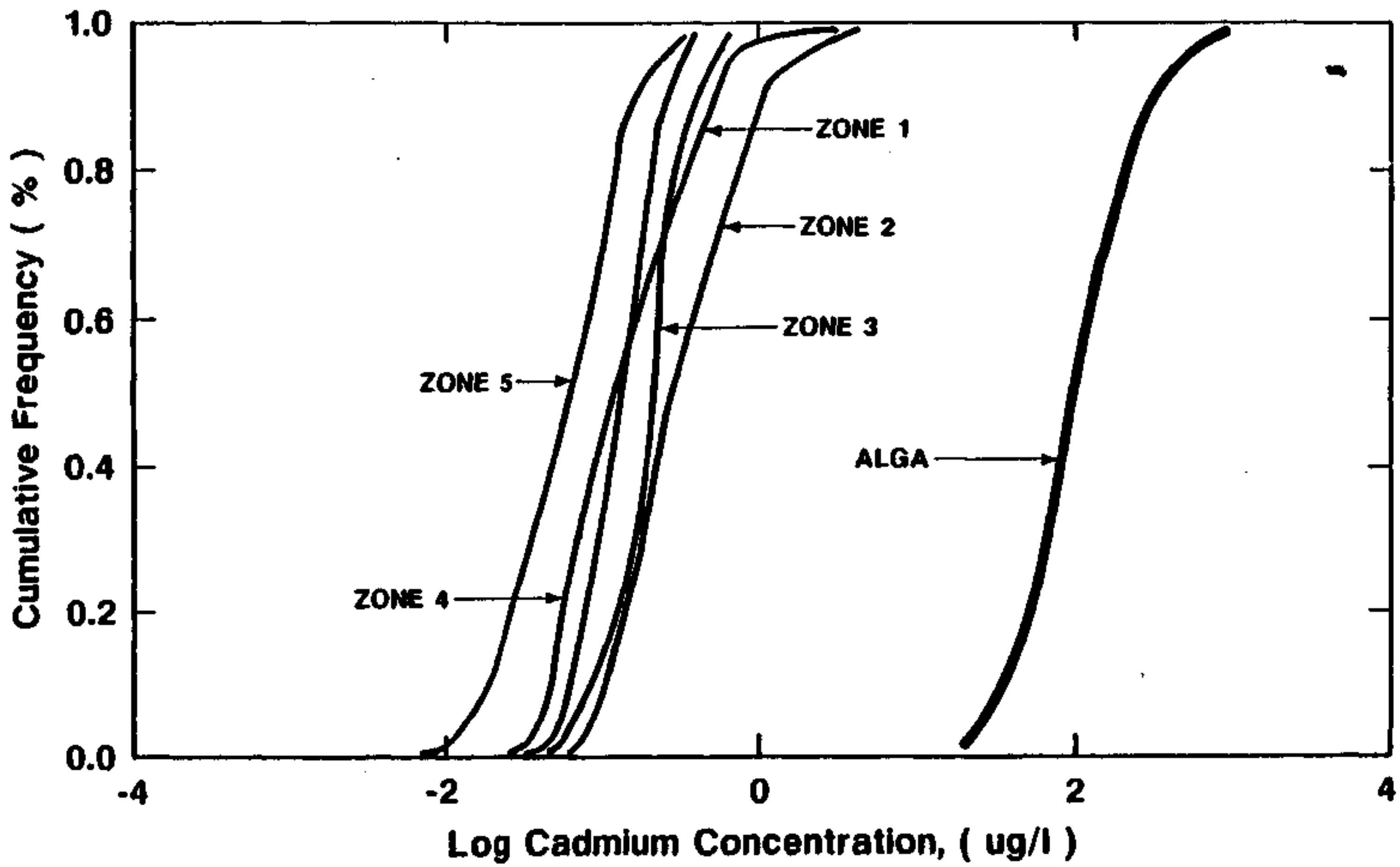

FIGURE C-6
MATC FOR ALGA AND
EECs FOR ALL ZONES, CADMIUM, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

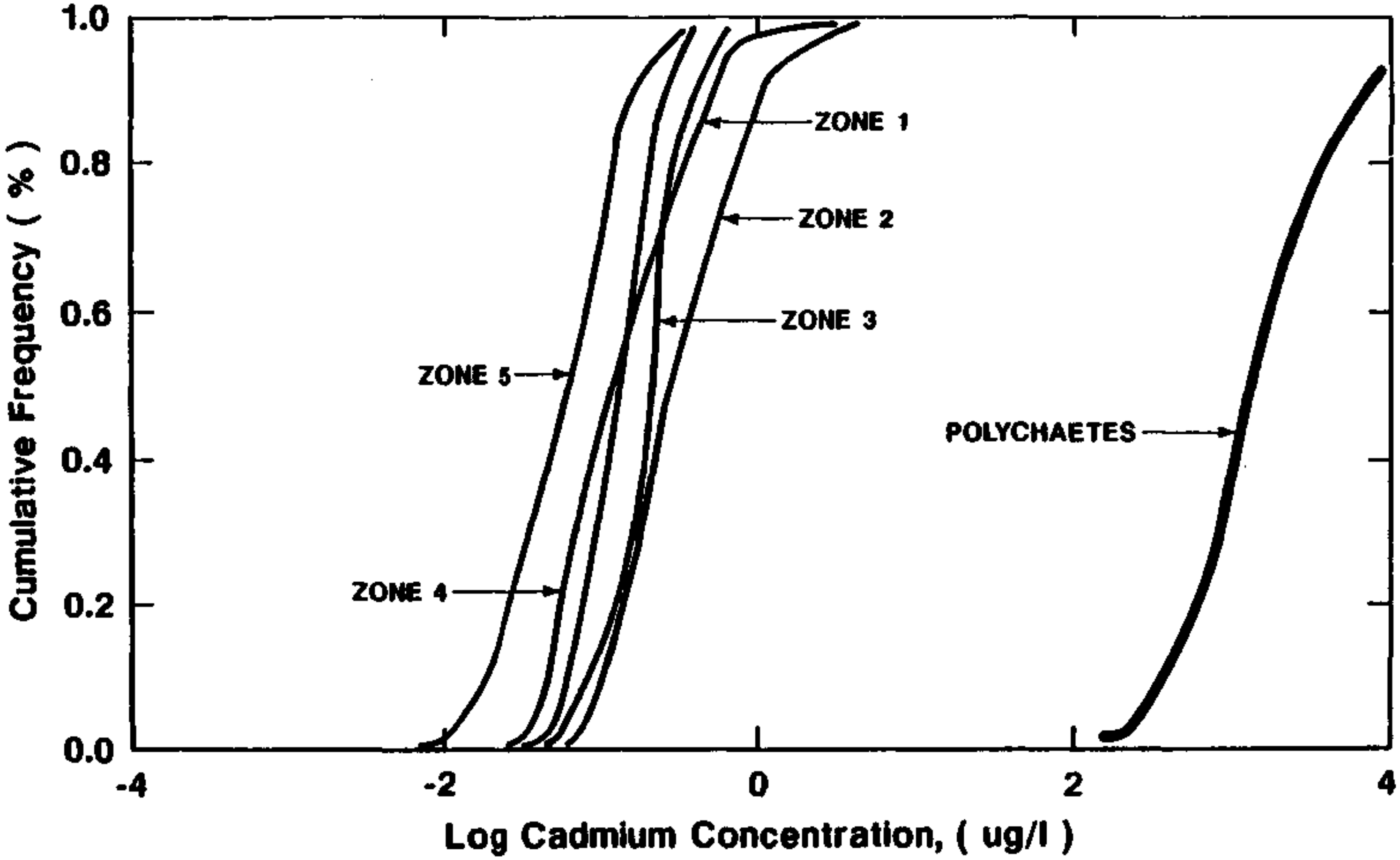

FIGURE C-7
MATC FOR POLYCHAETES AND
EECs FOR ALL ZONES, CADMIUM, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

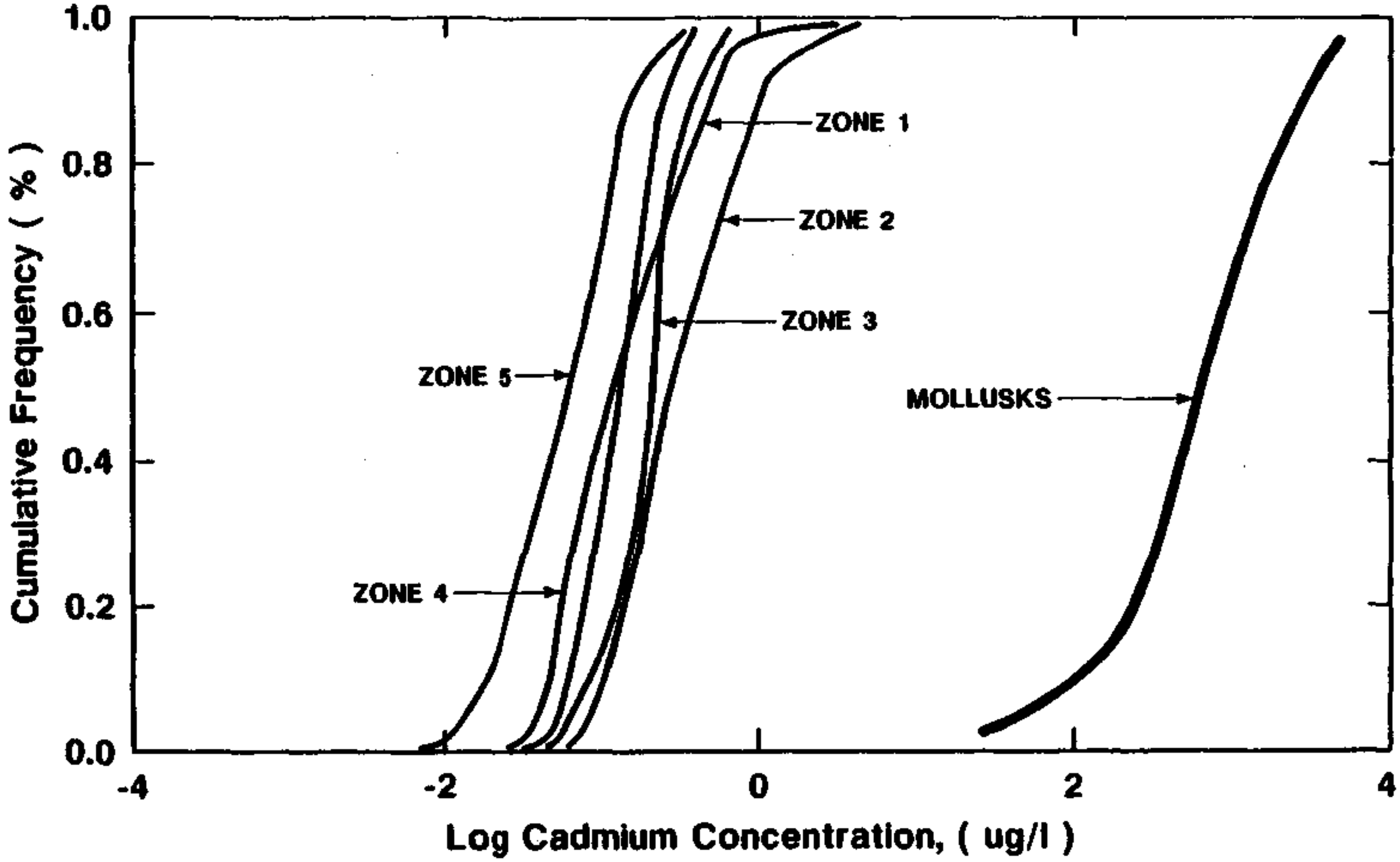

FIGURE C-8
MATC FOR MOLLUSKS AND
EECs FOR ALL ZONES, CADMIUM, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

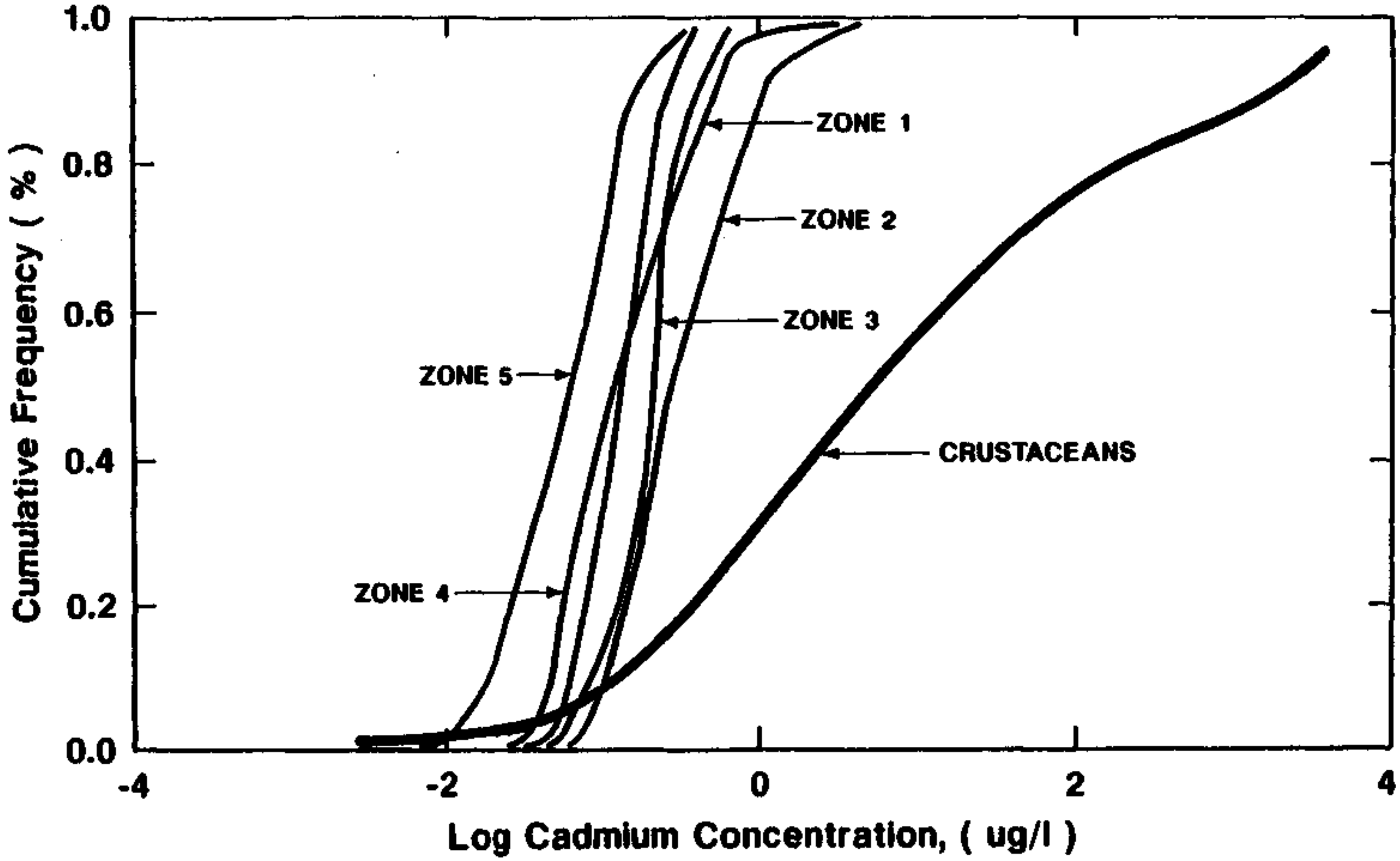

FIGURE C-9
MATC FOR CRUSTACEANS AND
EECs FOR ALL ZONES, CADMIUM, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

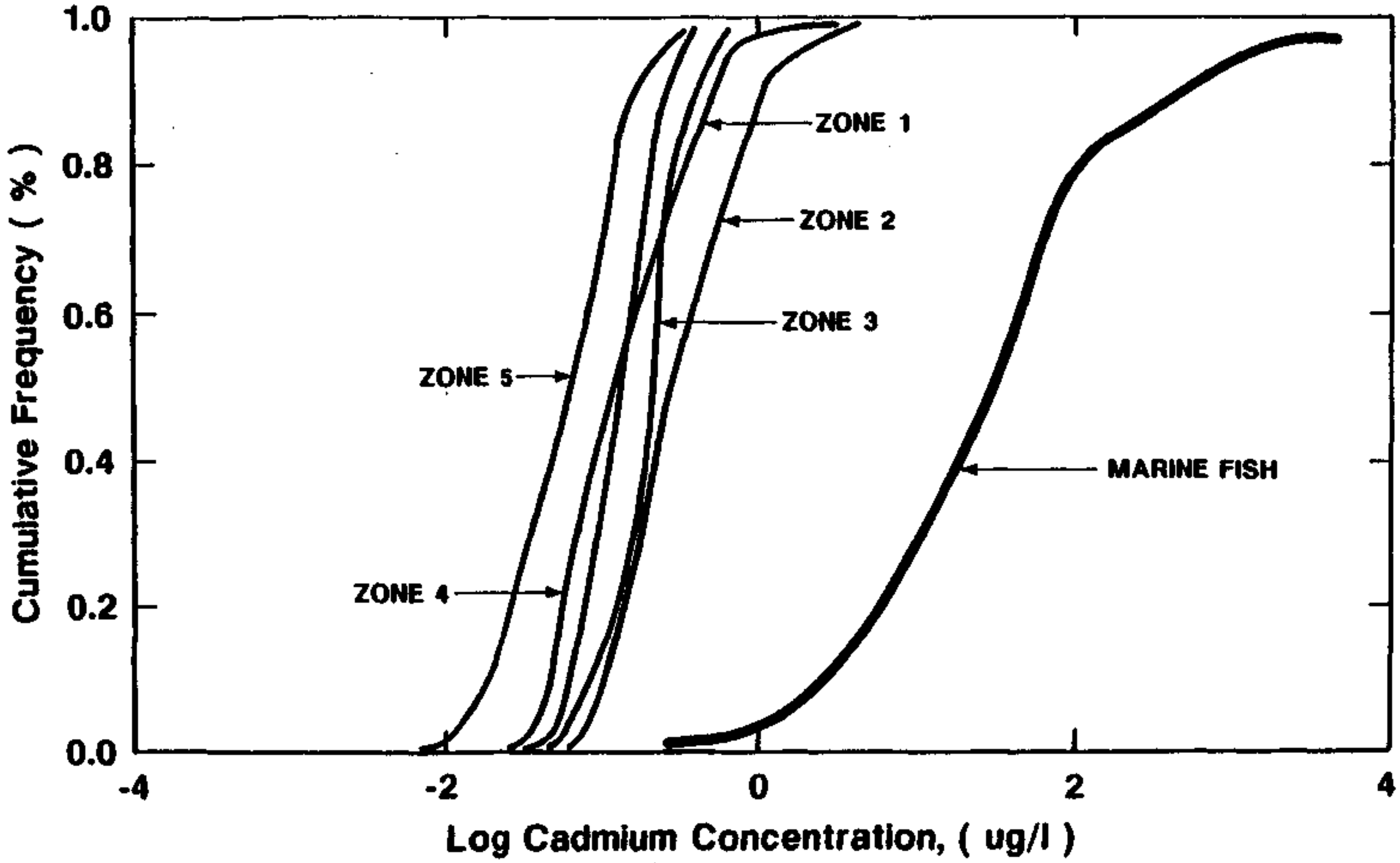

FIGURE C-10
MATC FOR MARINE FISH AND
EECs FOR ALL ZONES, CADMIUM, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

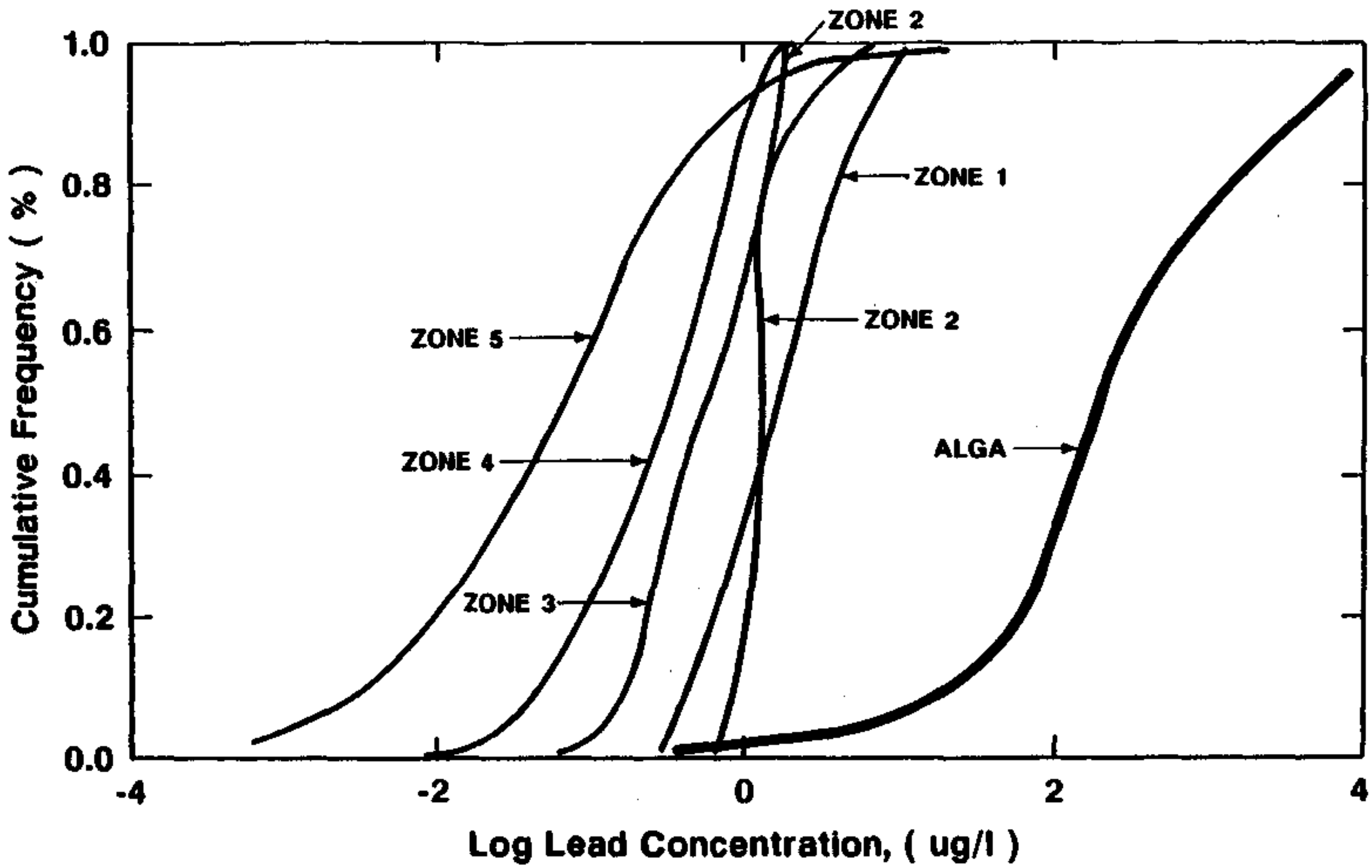

FIGURE C-11
MATC FOR ALGA AND
EECs FOR ALL ZONES, LEAD, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

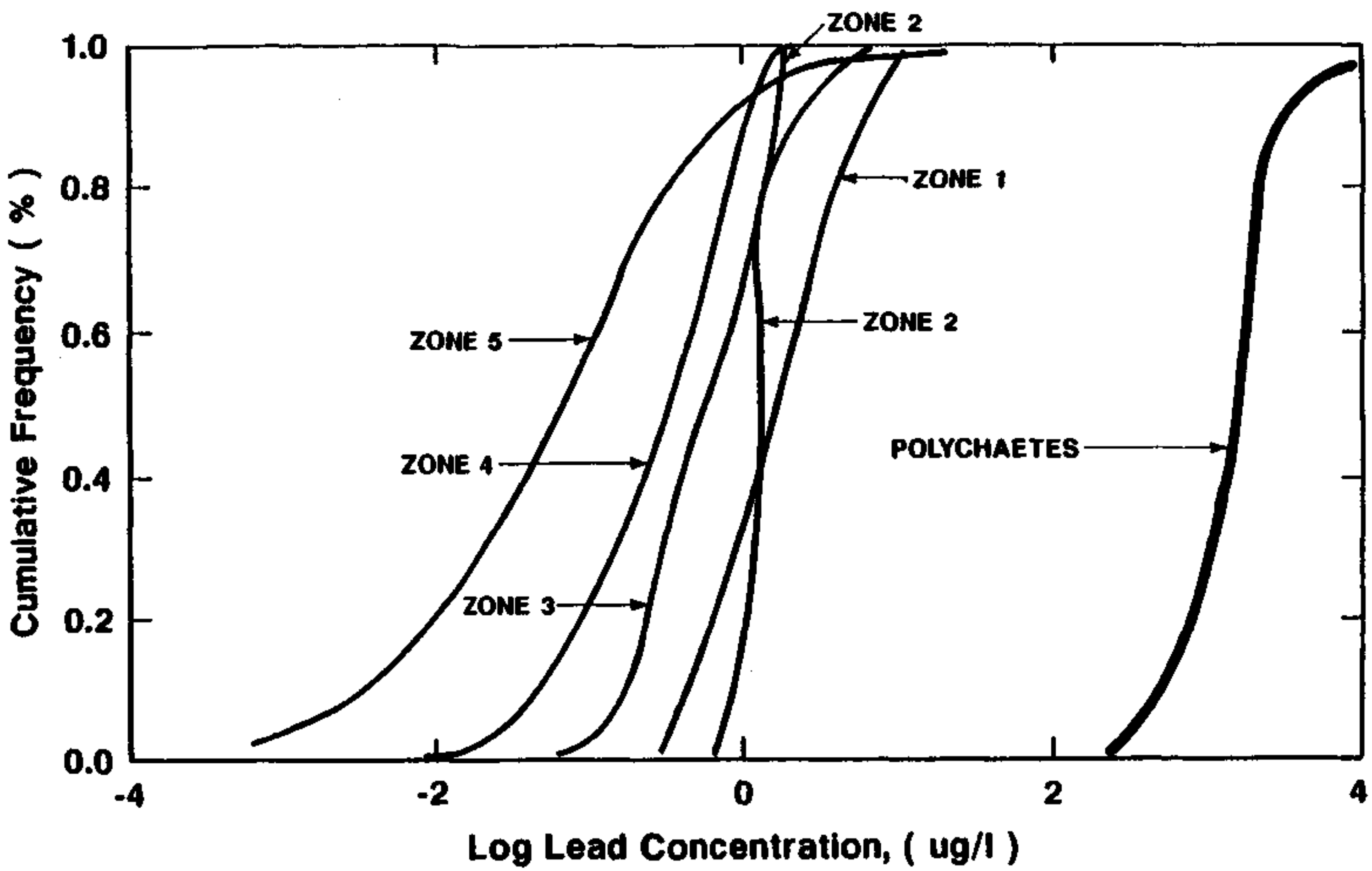

FIGURE C-12
MATC FOR POLYCHAETES AND
EECs FOR ALL ZONES, LEAD, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

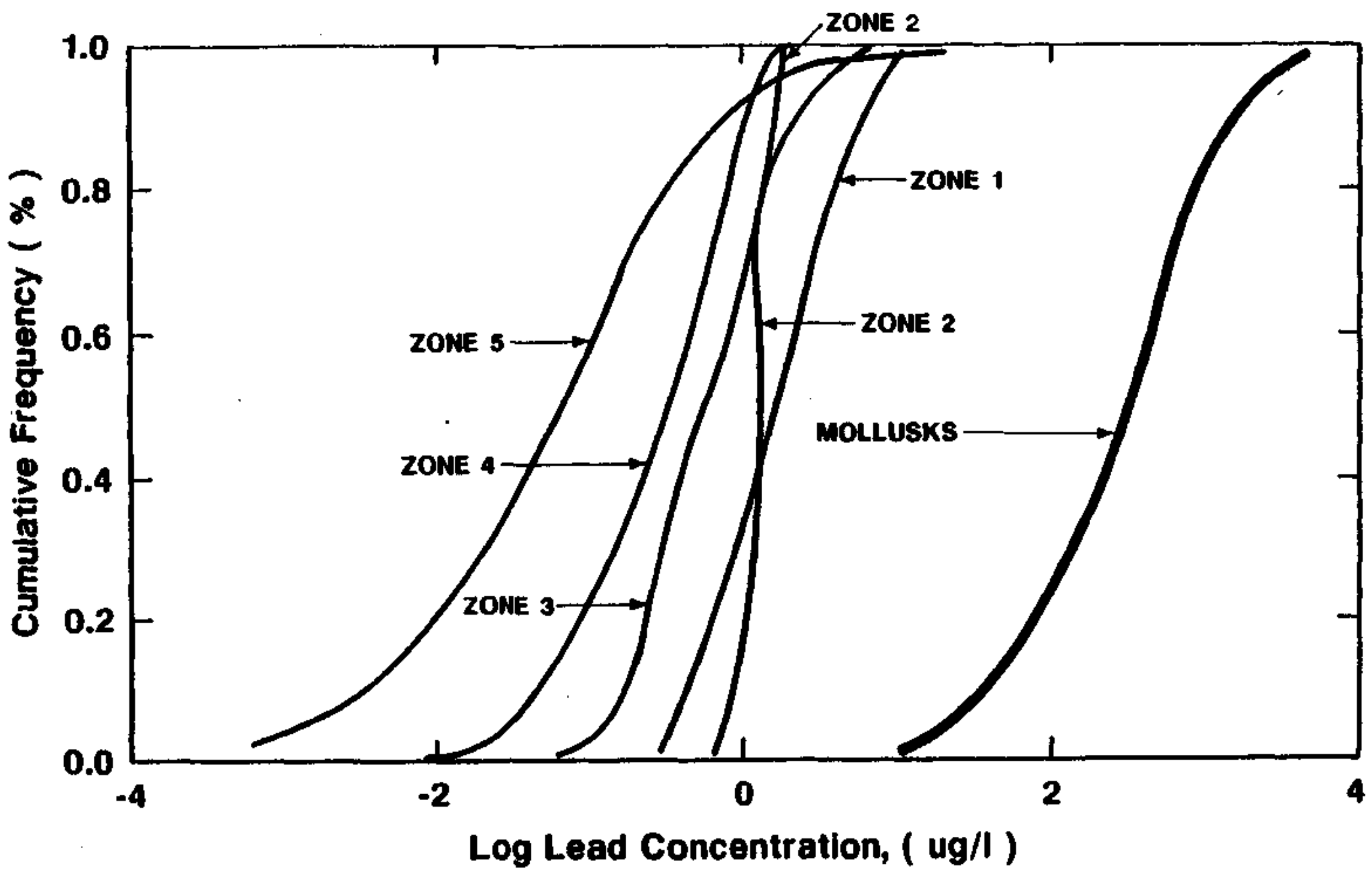

FIGURE C-13
MATC FOR MOLLUSKS AND
EECs FOR ALL ZONES, LEAD, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

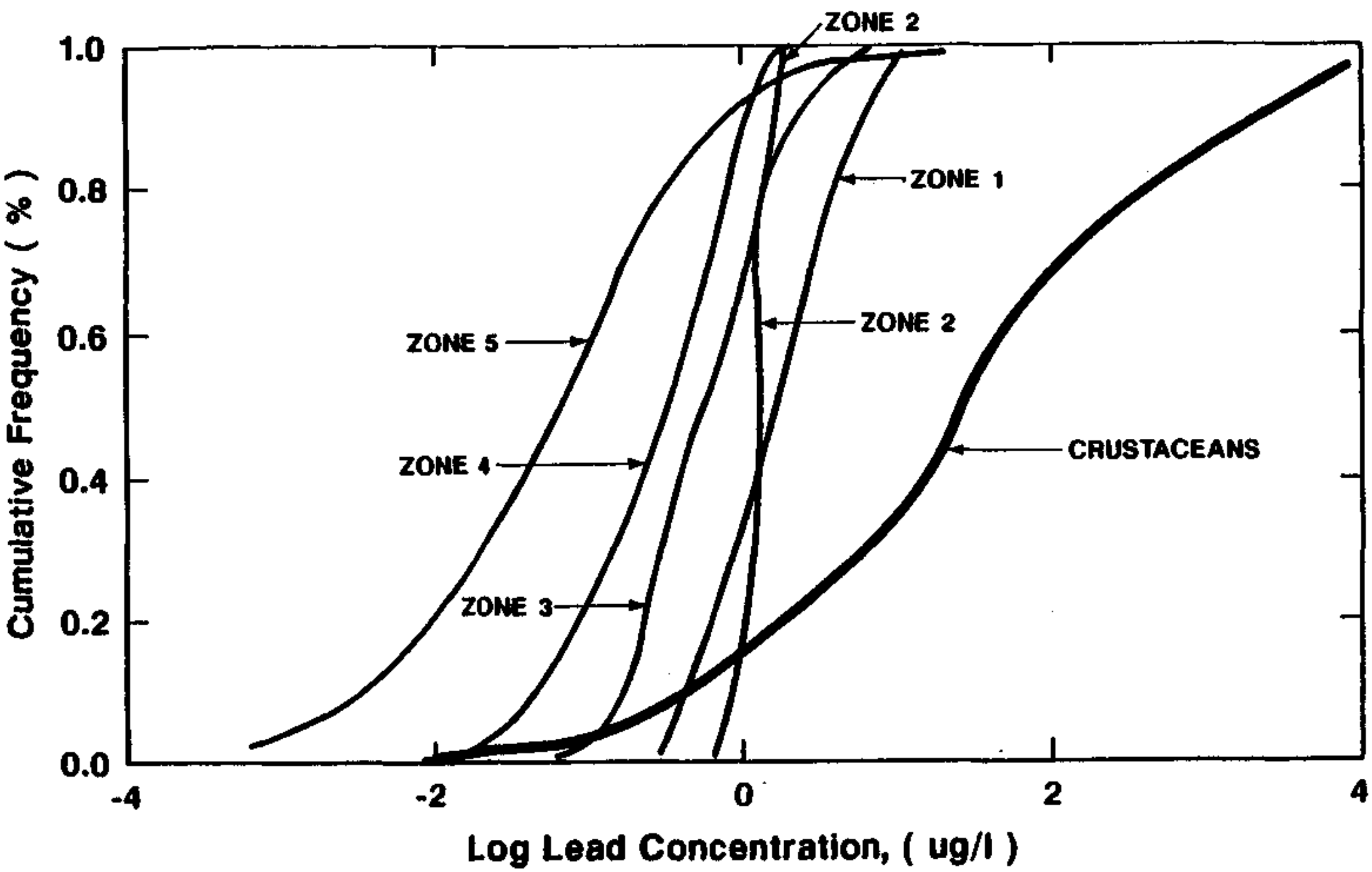

FIGURE C-14
MATC FOR CRUSTACEANS AND
ECCs FOR ALL ZONES, LEAD, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

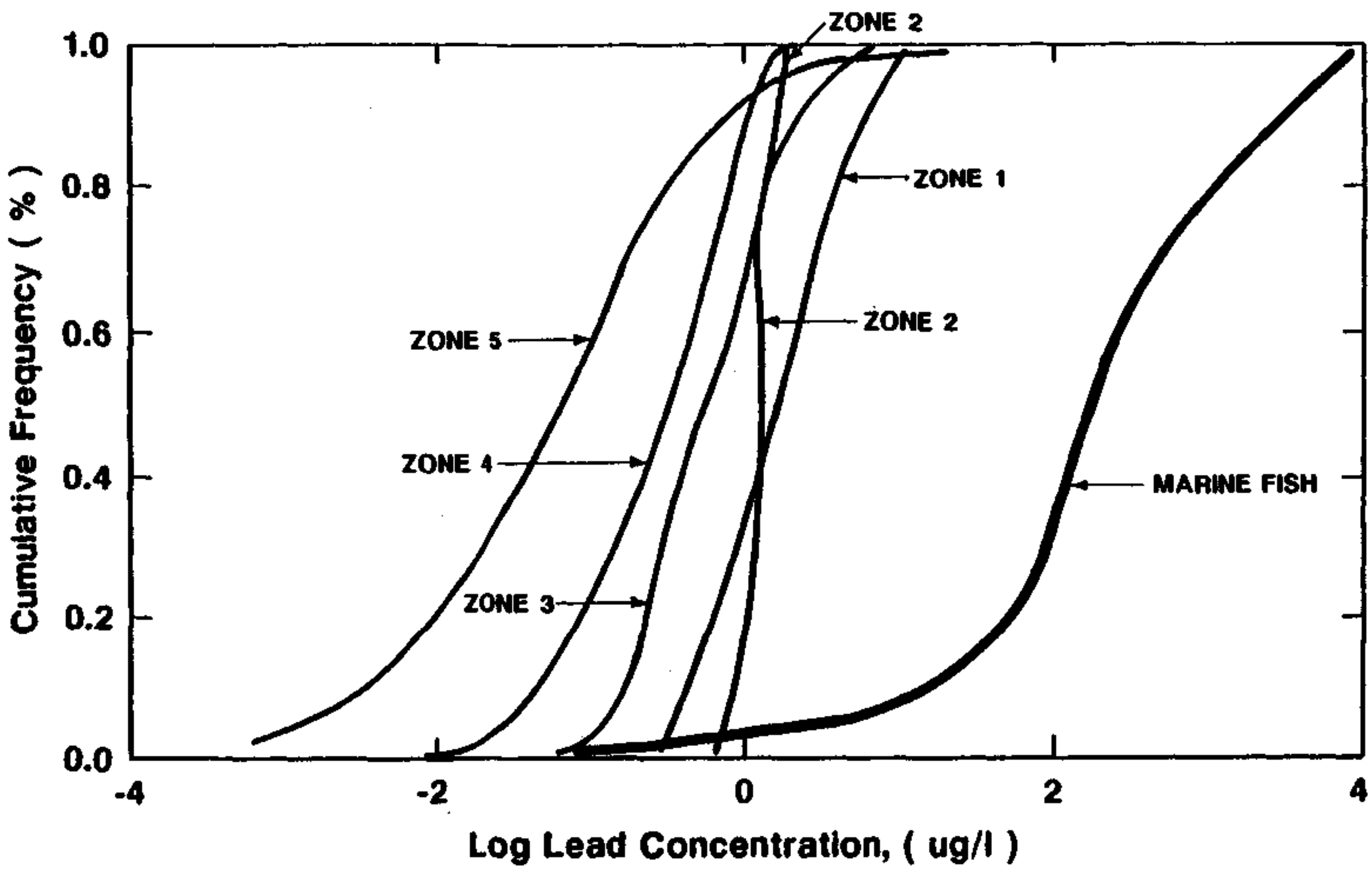

FIGURE C-15
MATC FOR MARINE FISH AND
EECs FOR ALL ZONES, LEAD, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

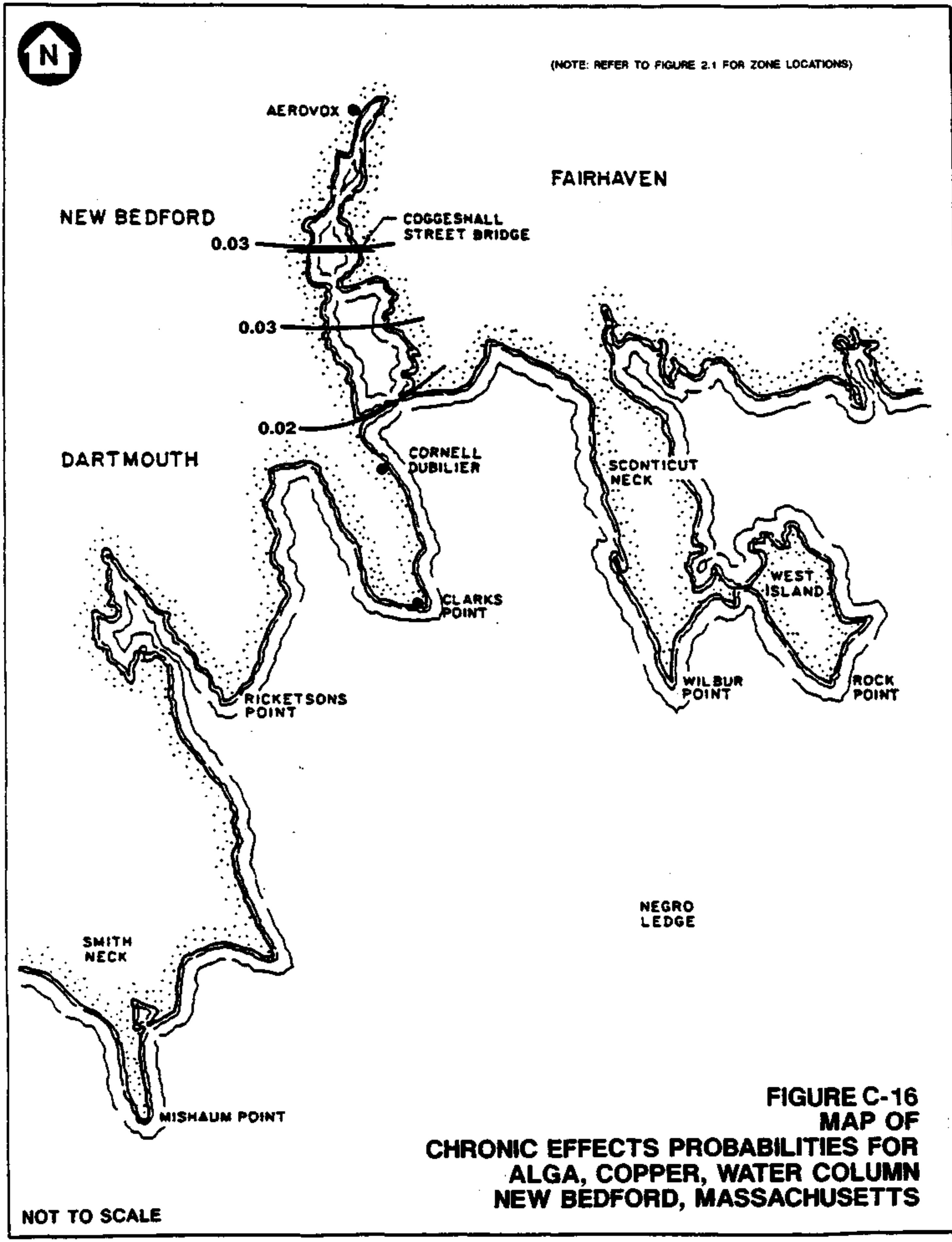

FIGURE C-16
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR ALGA, COPPER, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

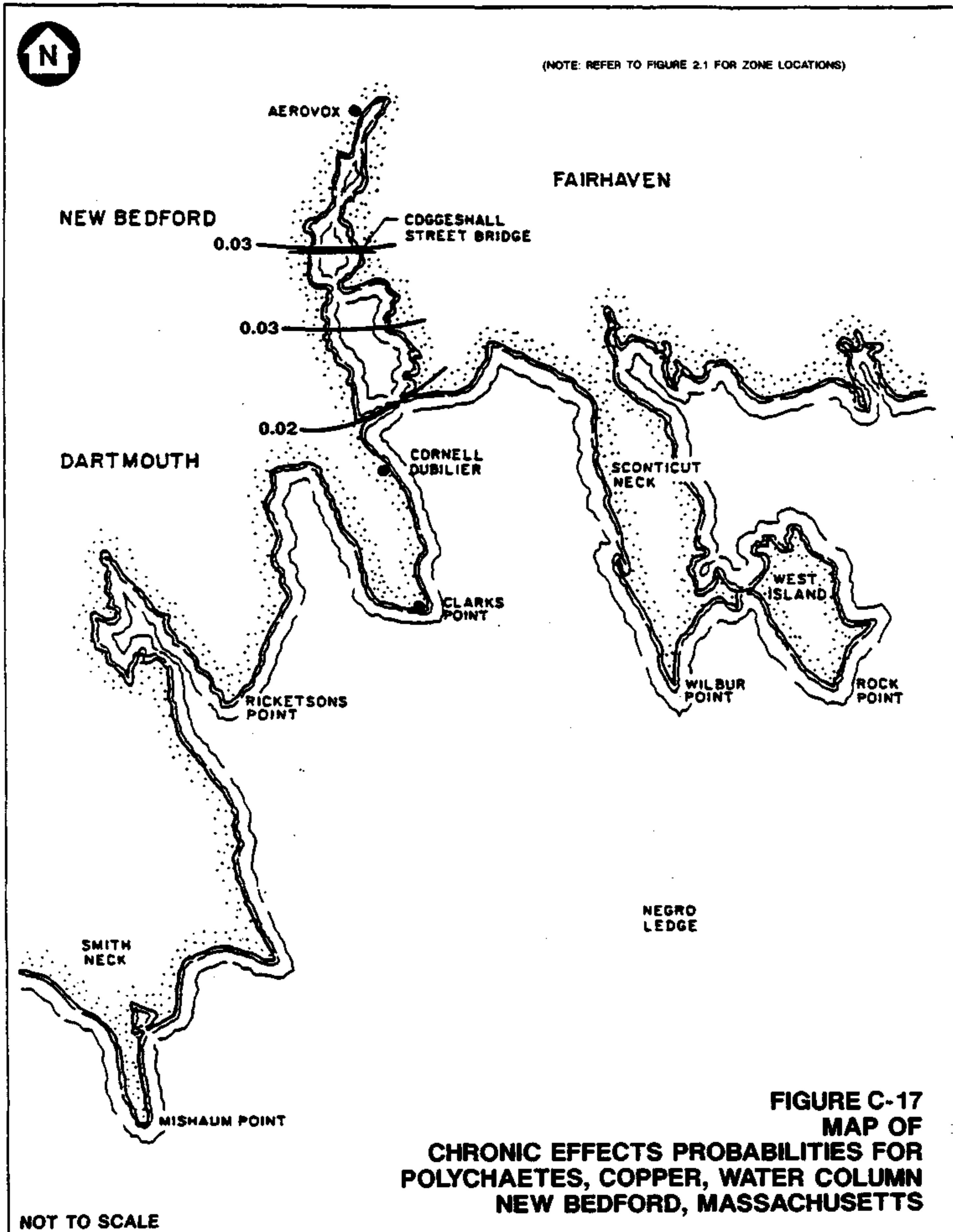

FIGURE C-17
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR POLYCHAETES, COPPER, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

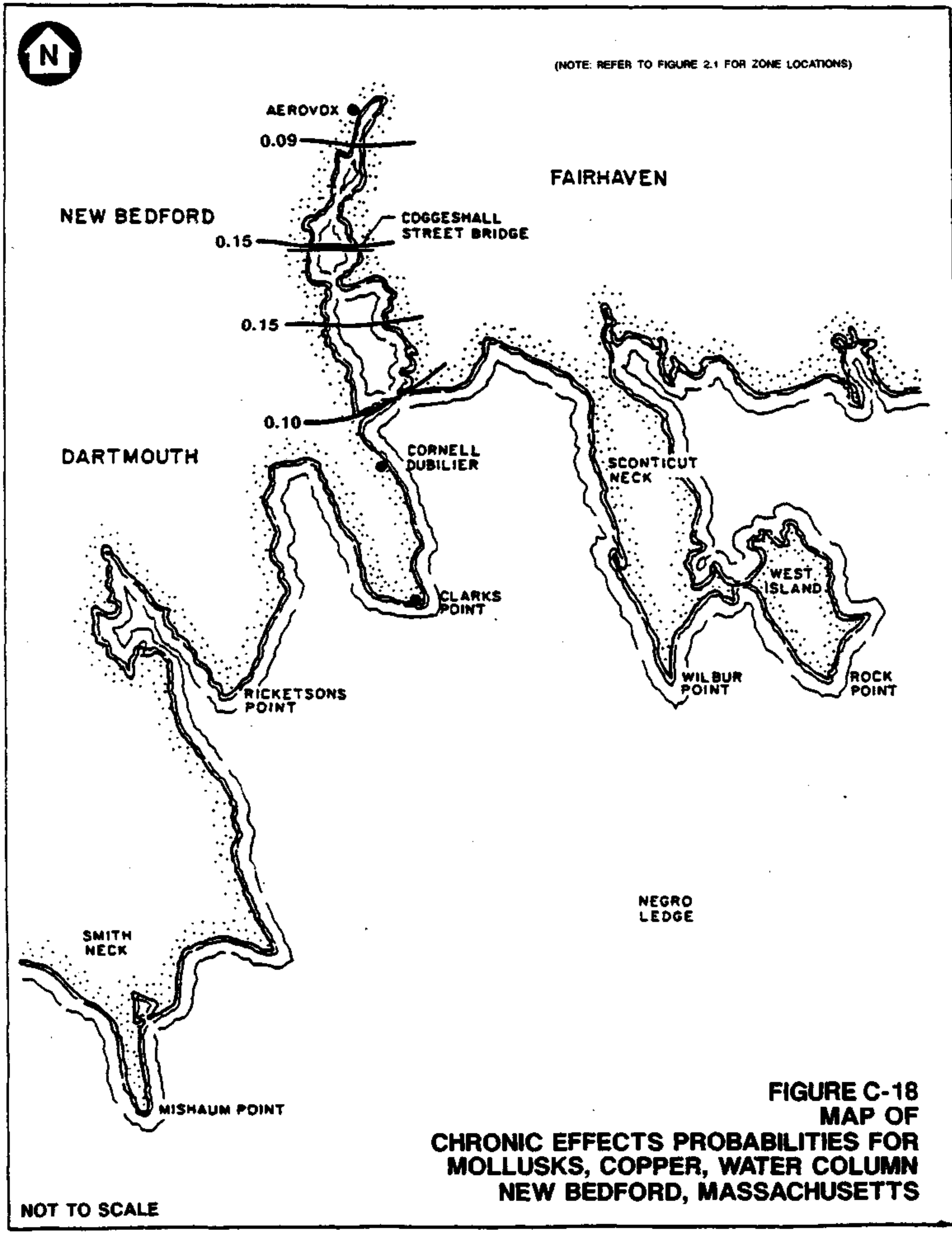

FIGURE C-18
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR MOLLUSKS, COPPER, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

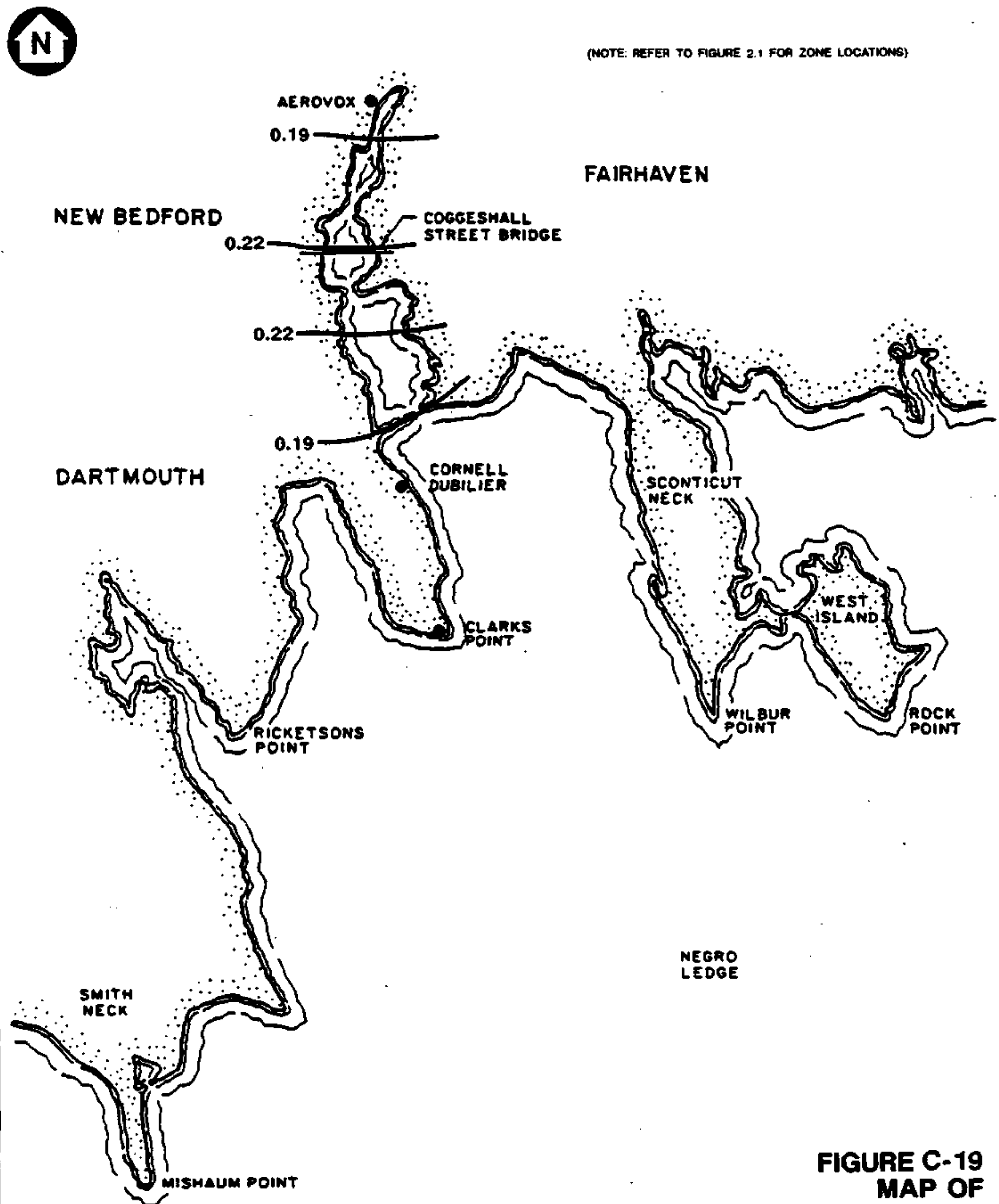

FIGURE C-19
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR CRUSTACEANS, COPPER, WATER COLUMN NEW BEDFORD, MASSACHUSETTS

NOT TO SCALE

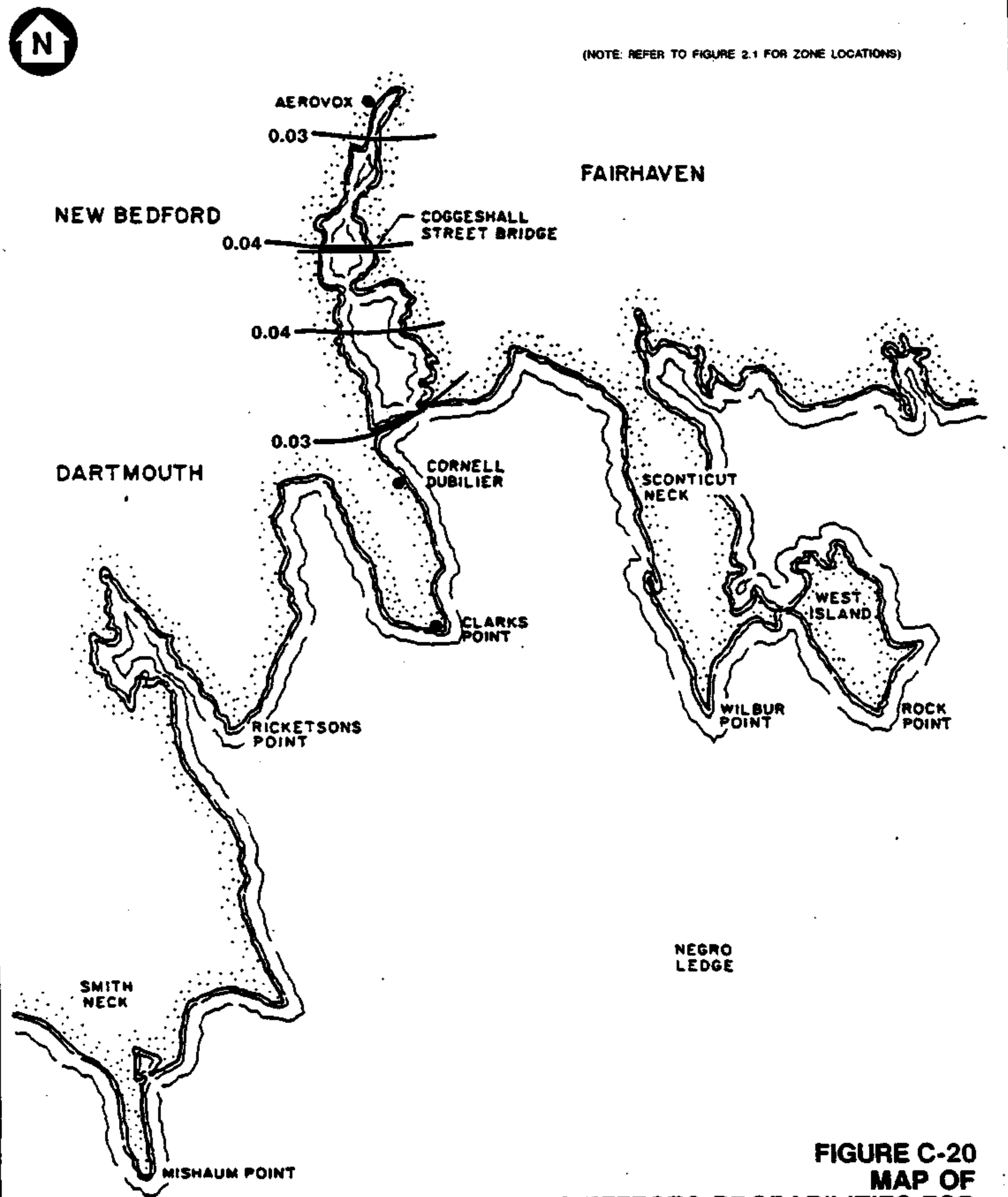

FIGURE C-20
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
MARINE FISH, COPPER, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

NOT TO SCALE

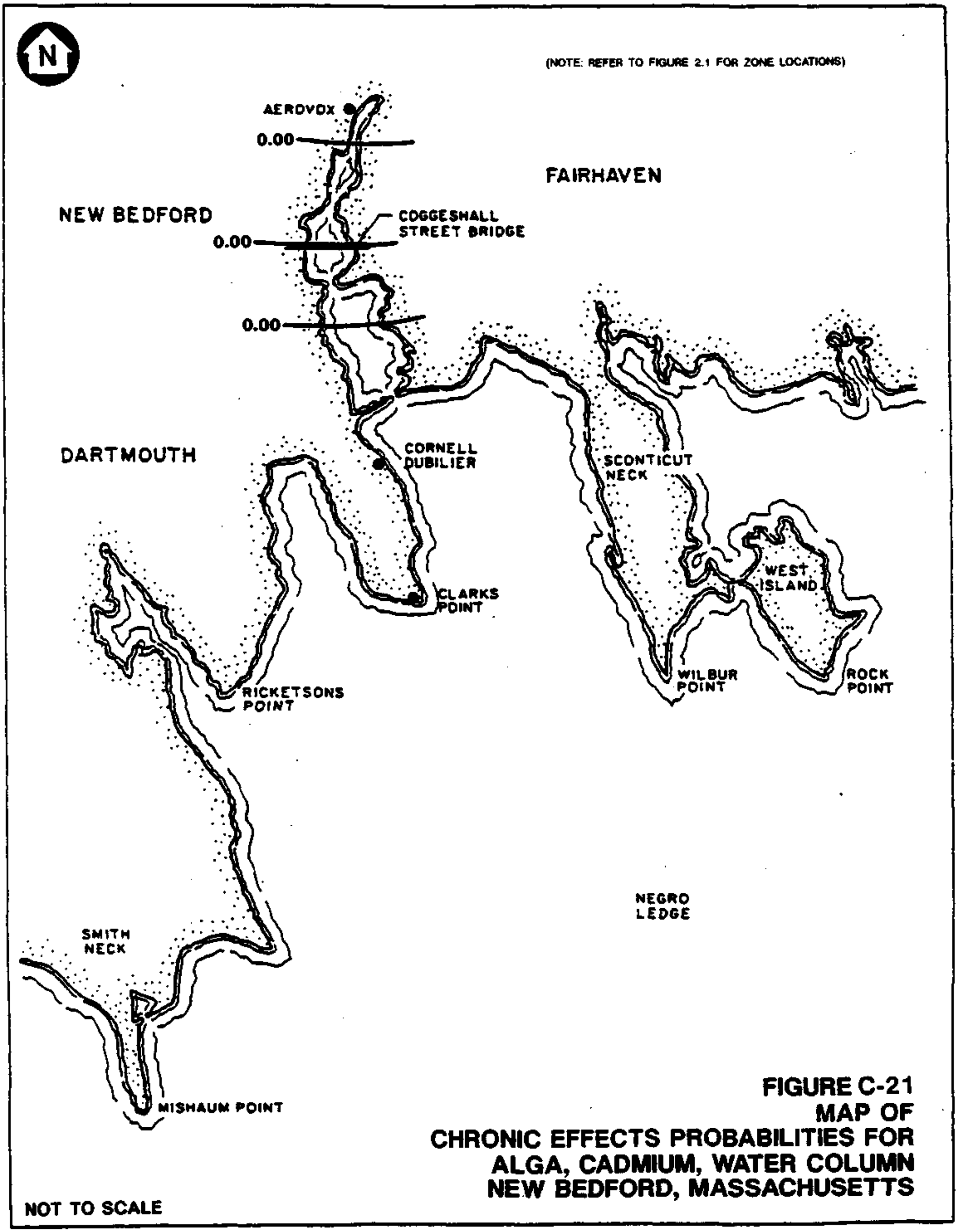

FIGURE C-21
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
ALGA, CADMIUM, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

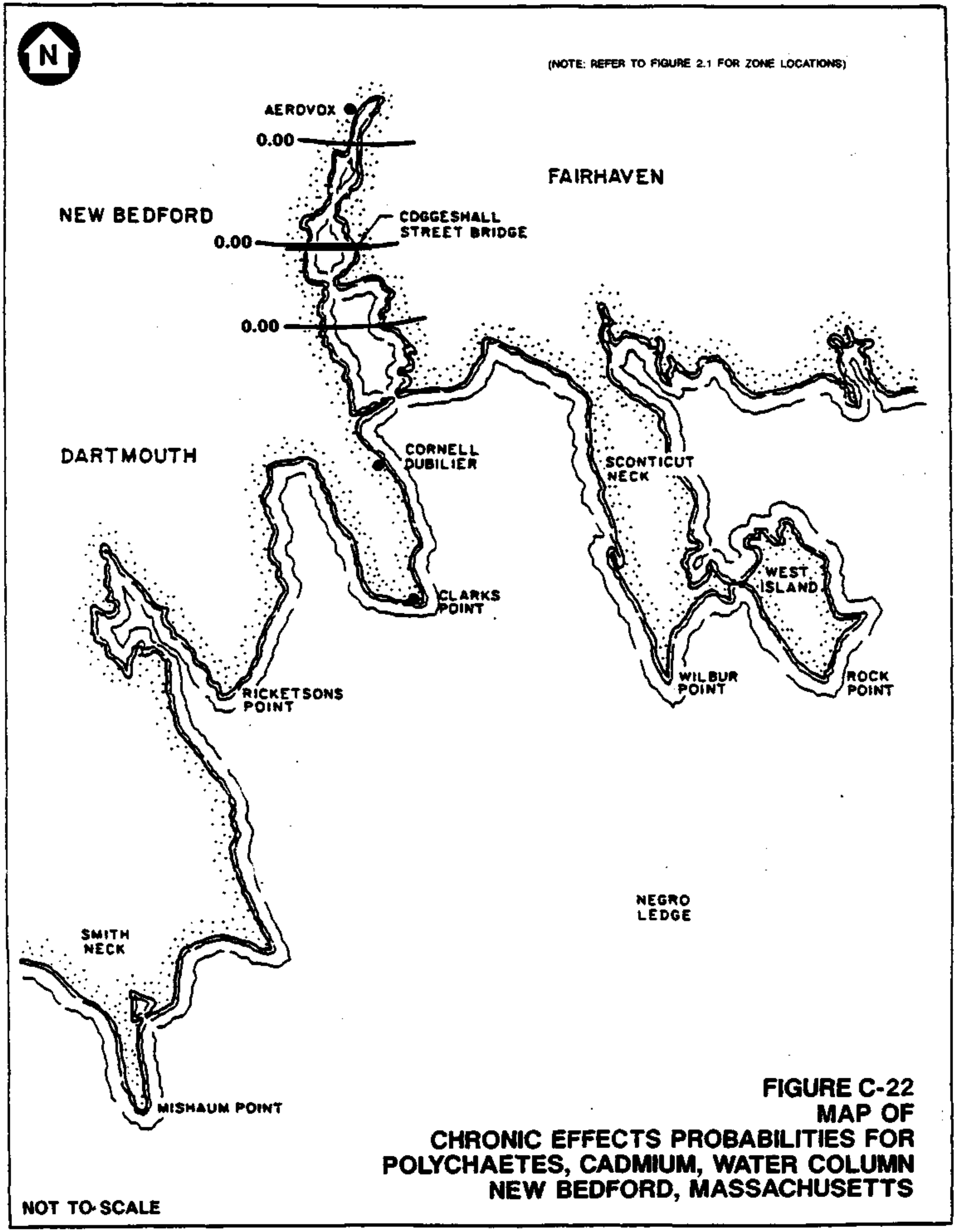

FIGURE C-22
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR POLYCHAETES, CADMIUM, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

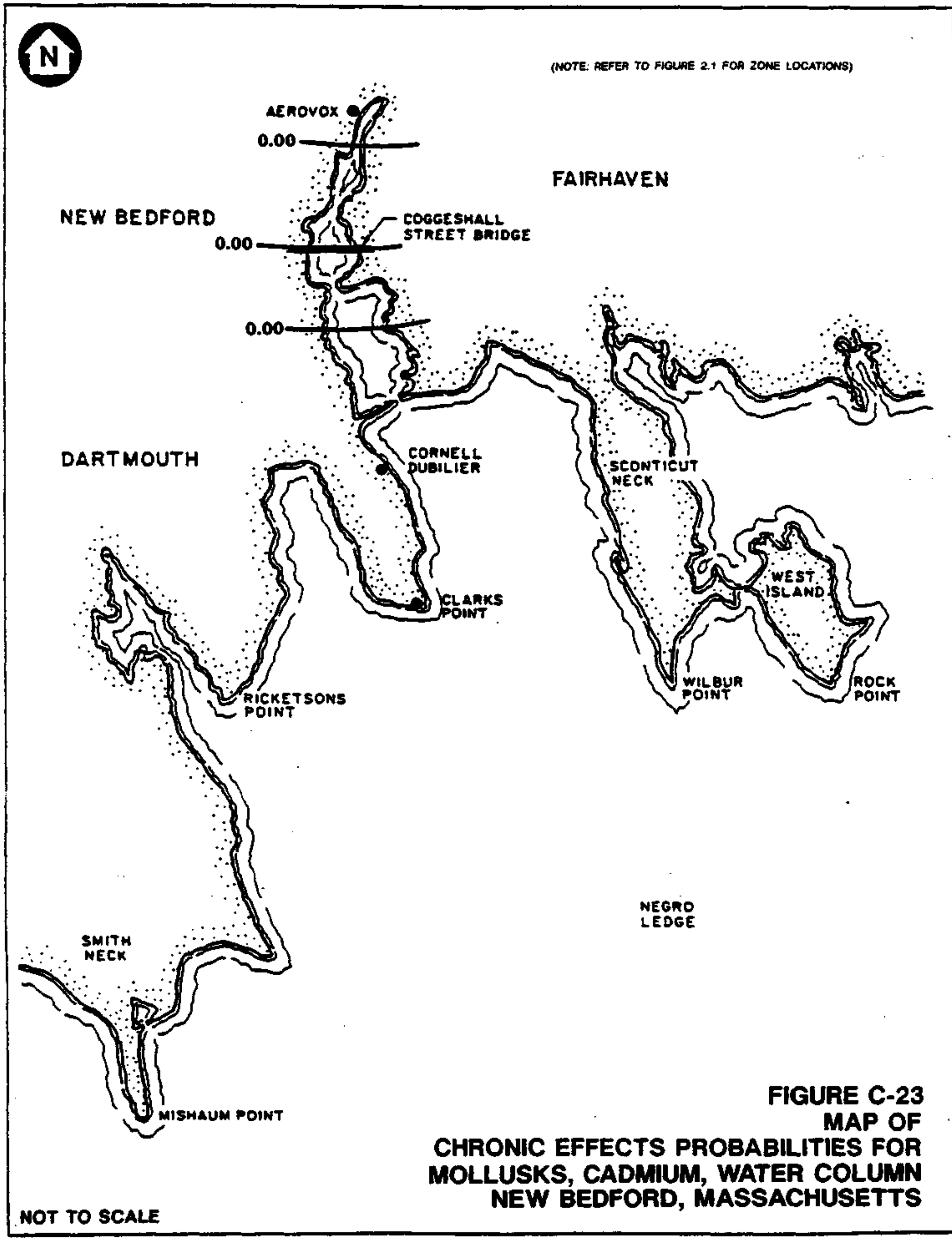

FIGURE C-23
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
MOLLUSKS, CADMIUM, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

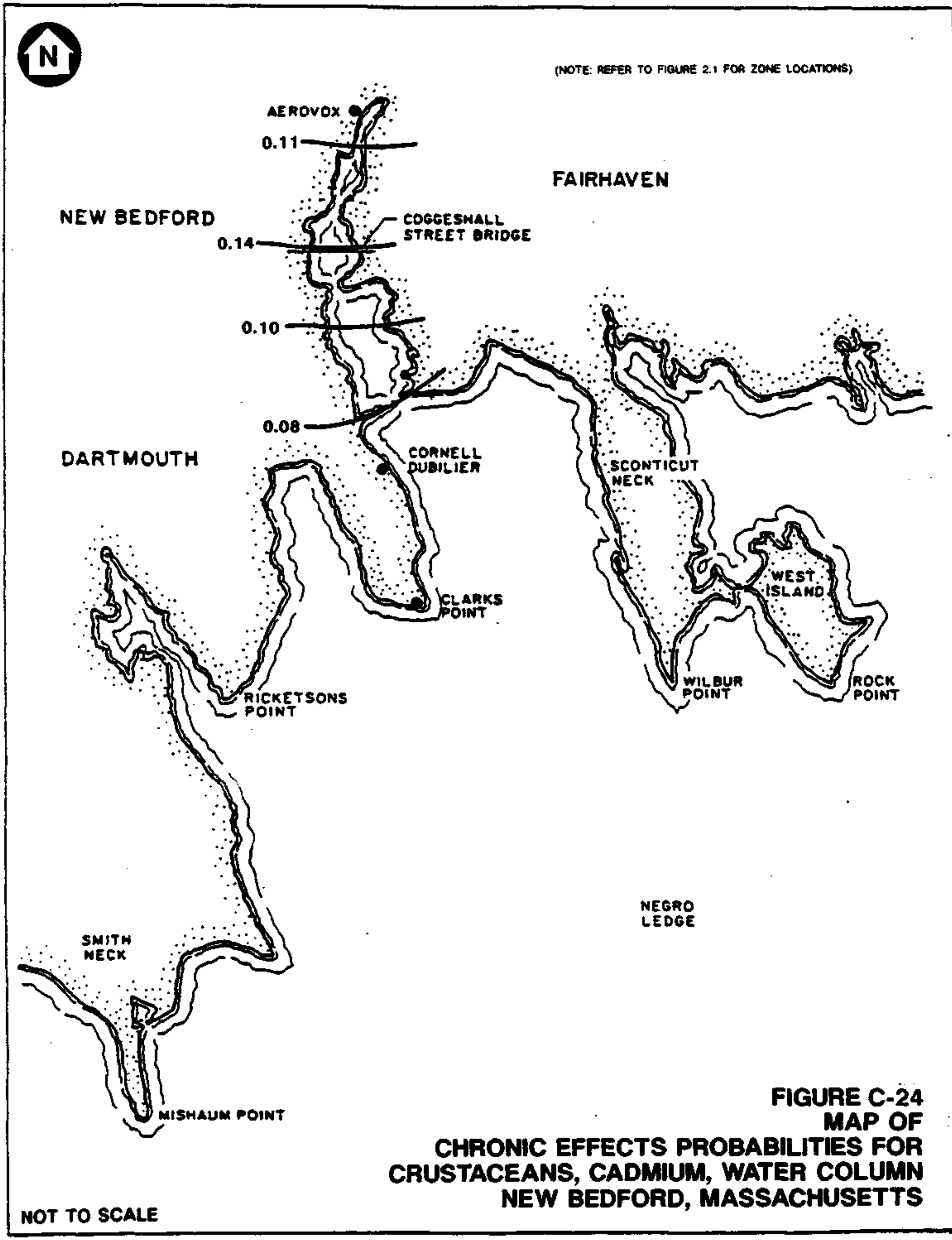

FIGURE C-24
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
CRUSTACEANS, CADMIUM, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

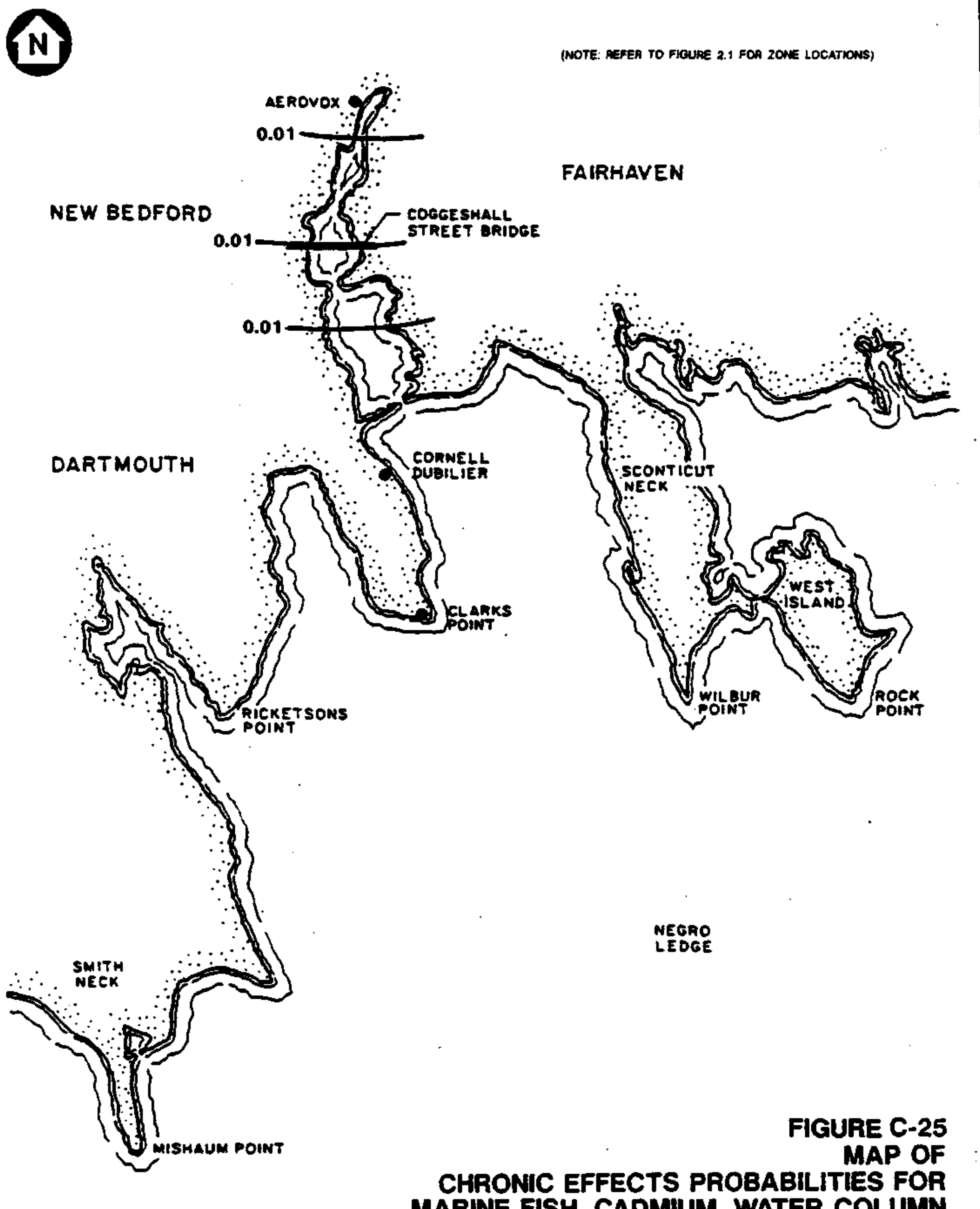

FIGURE C-25
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR MARINE FISH, CADMIUM, WATER COLUMN NEW BEDFORD, MASSACHUSETTS

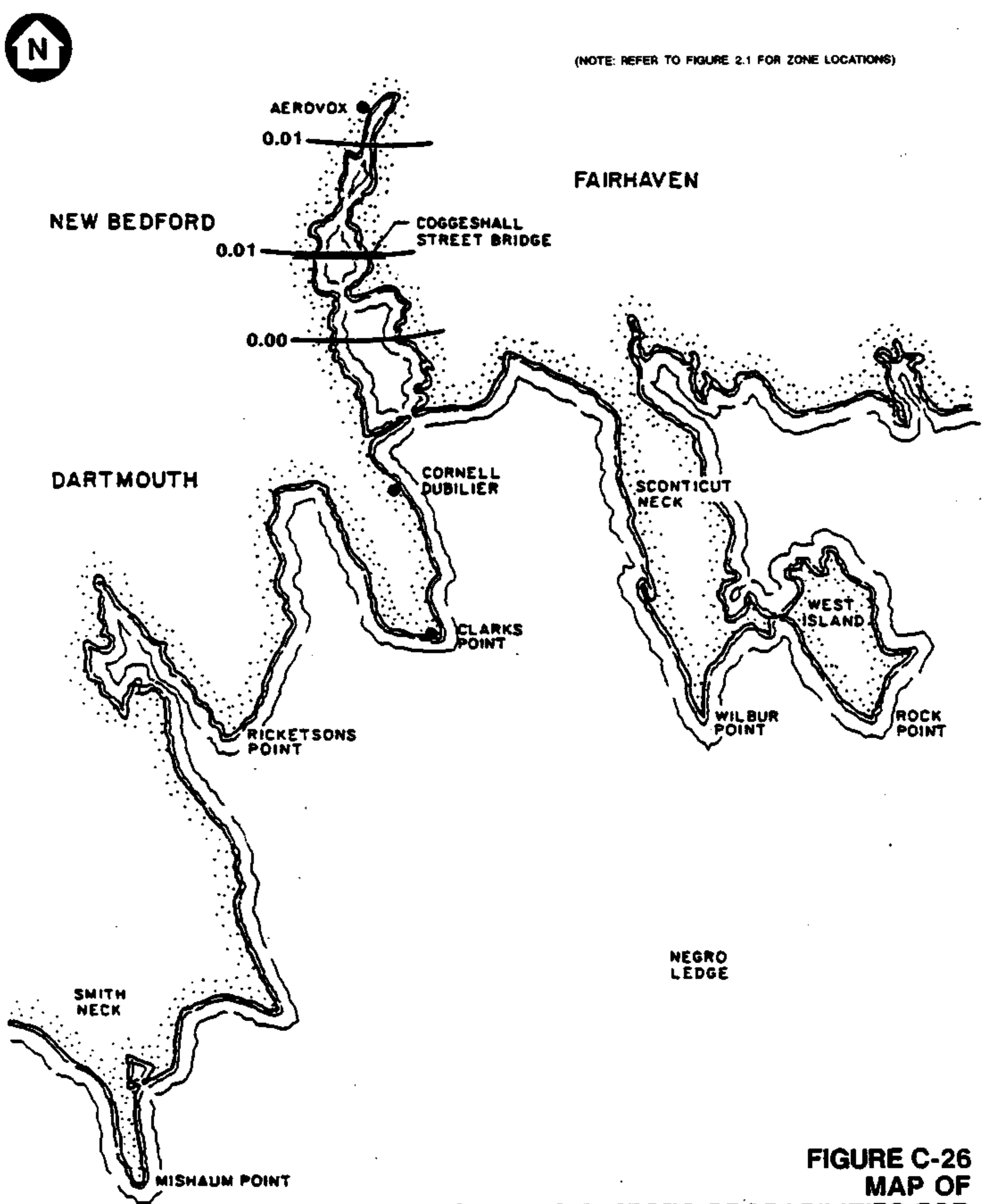

FIGURE C-26
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
ALGA, LEAD, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

NOT TO SCALE

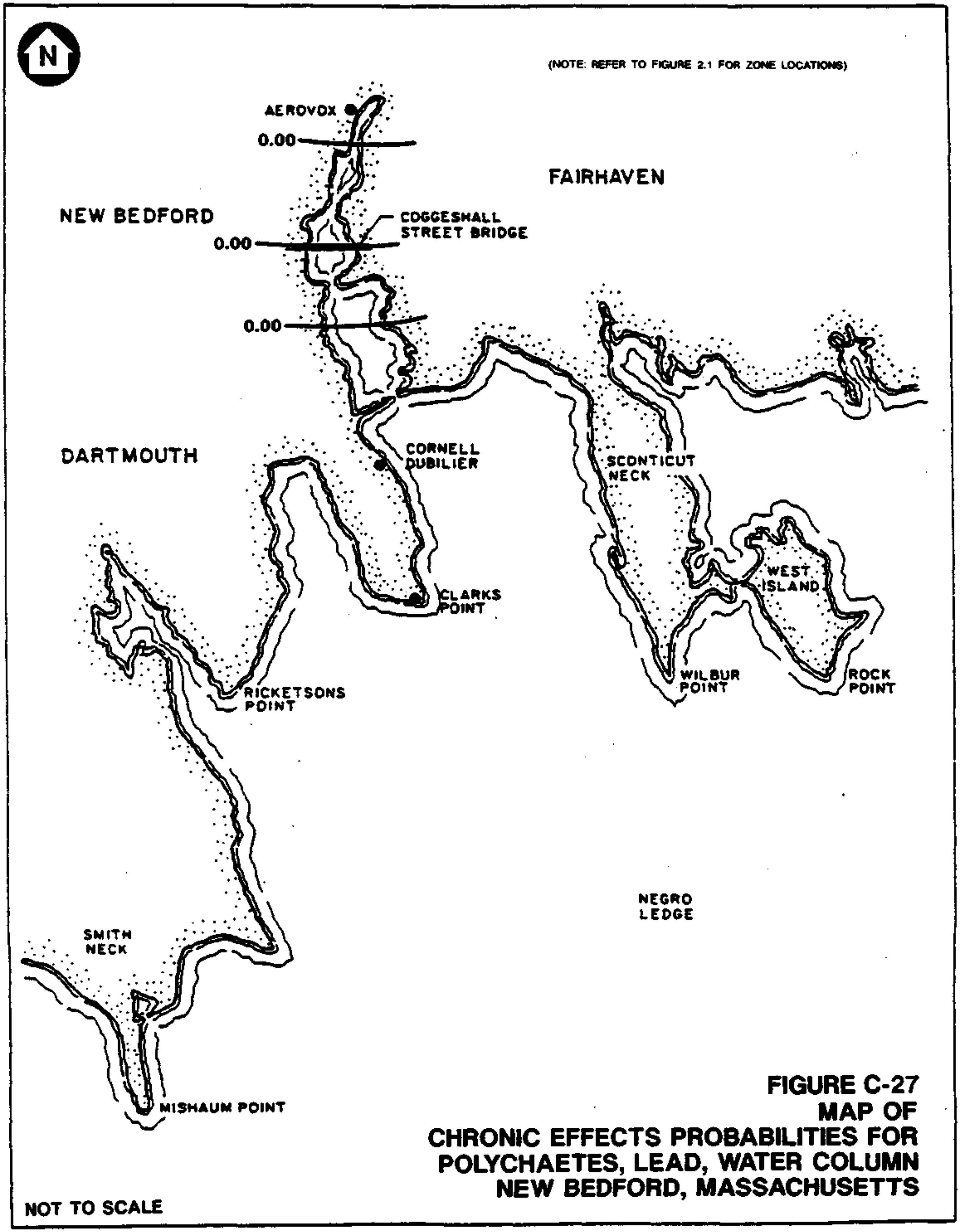

FIGURE C-27
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR POLYCHAETES, LEAD, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

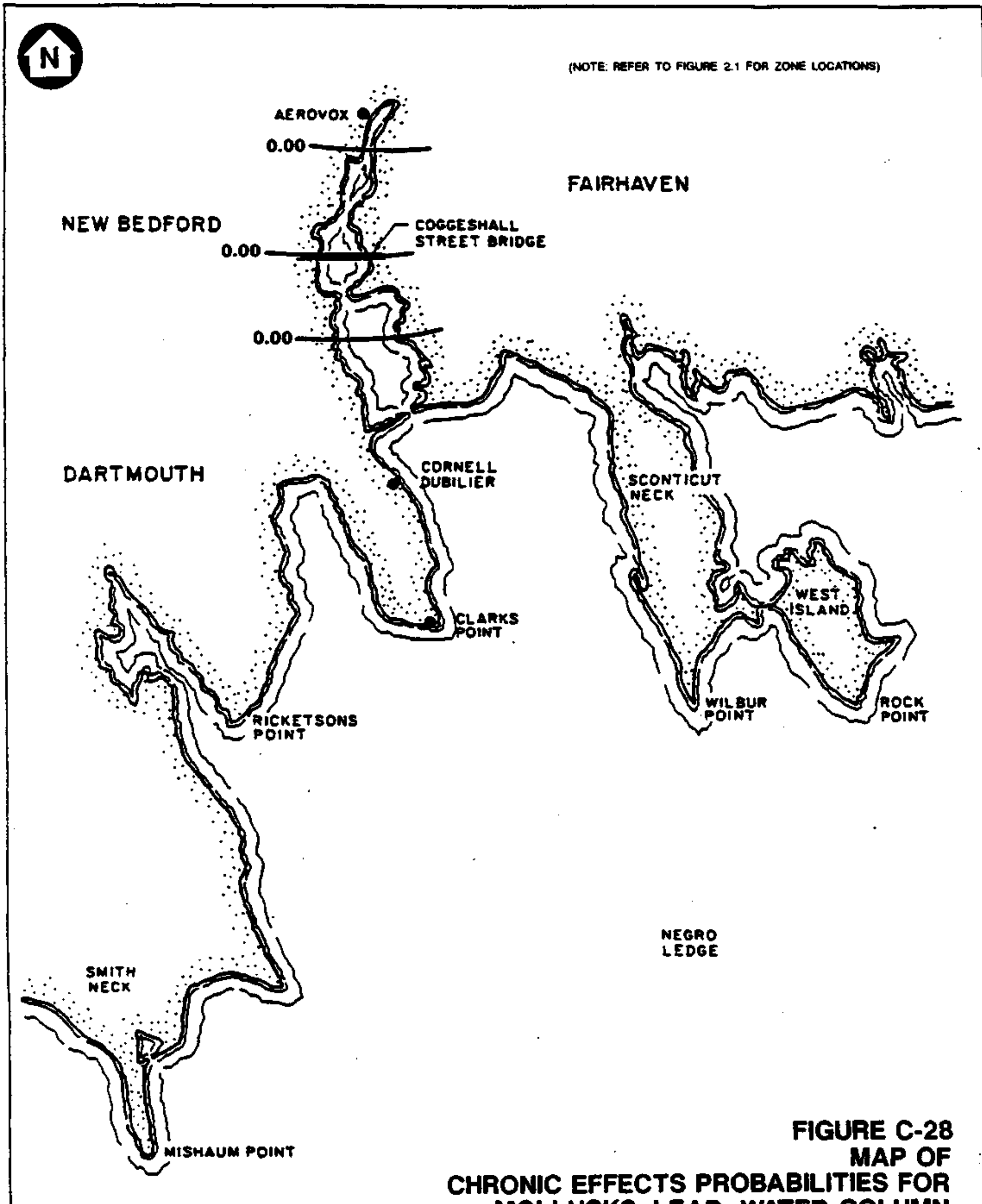

FIGURE C-28
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
MOLLUSKS, LEAD, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

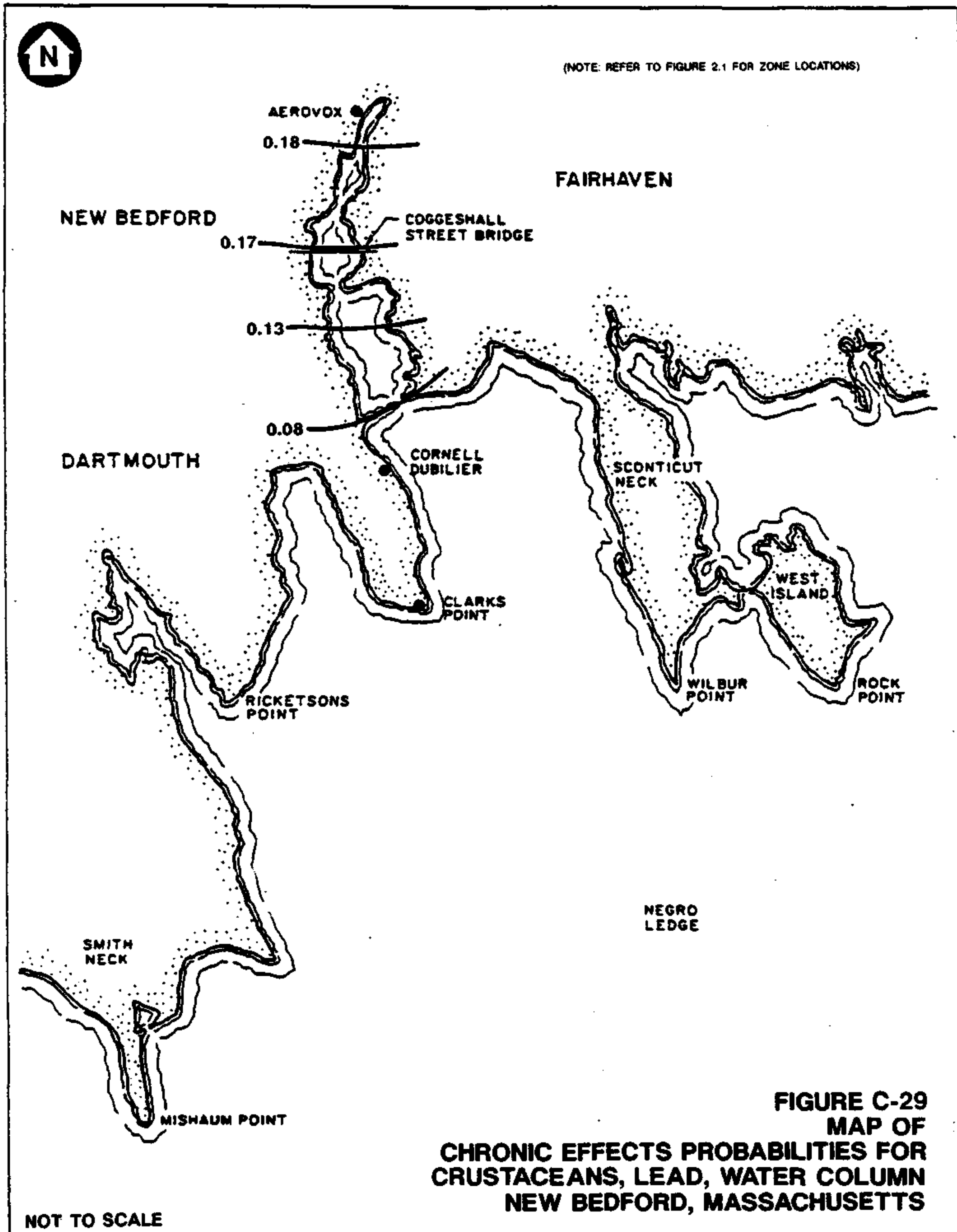

FIGURE C-29
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR CRUSTACEANS, LEAD, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

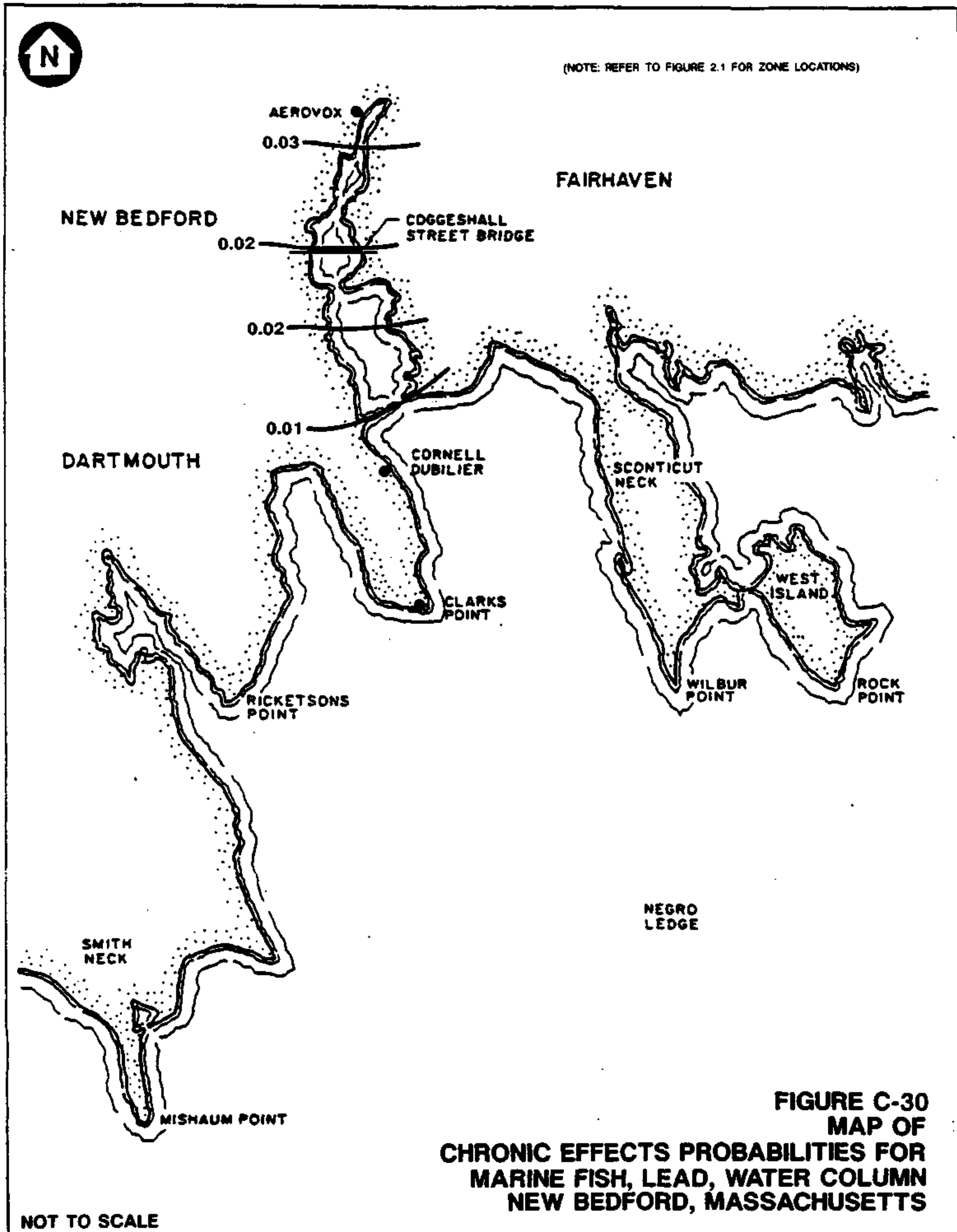

FIGURE C-30
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
MARINE FISH, LEAD, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

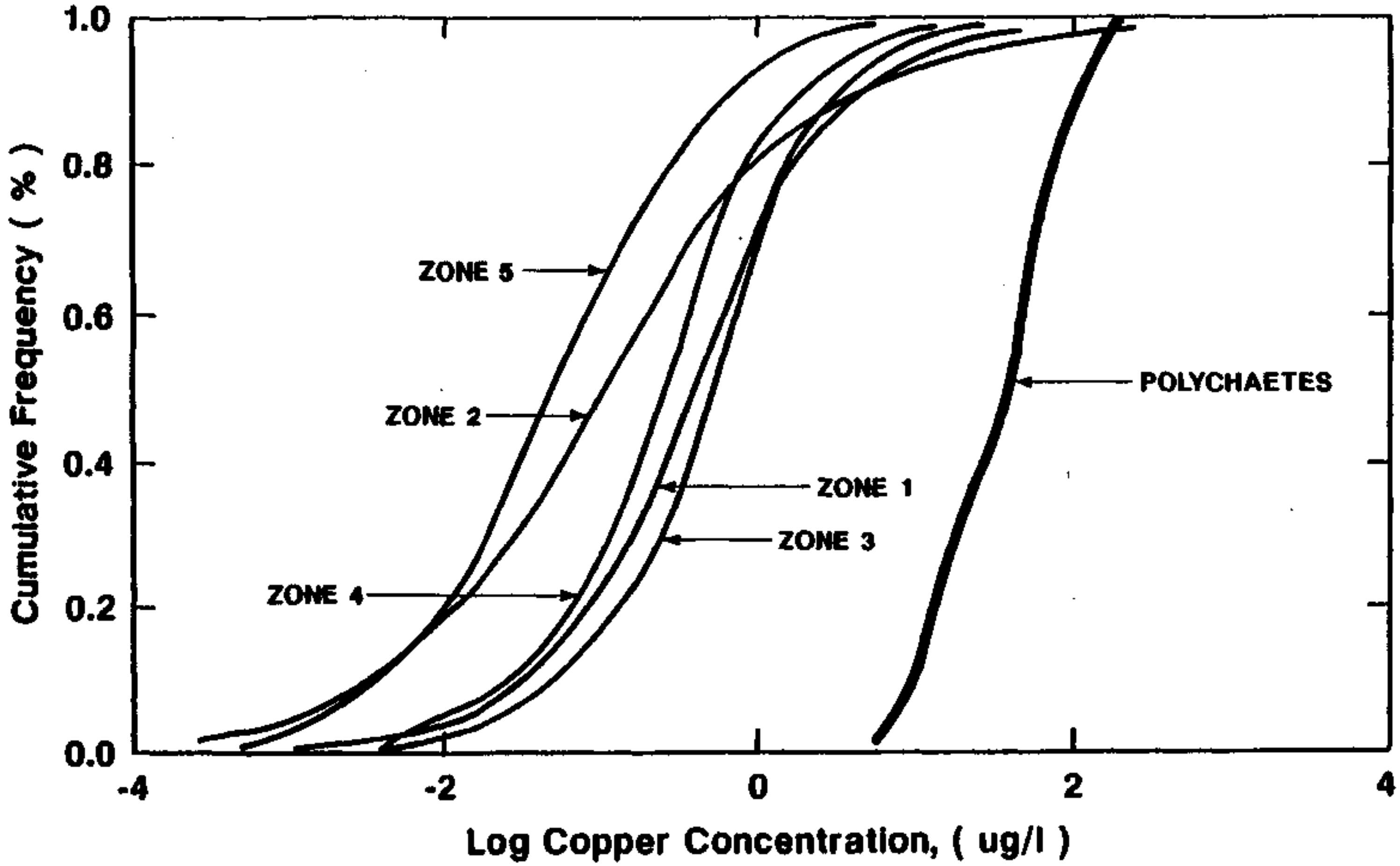

FIGURE C-31
MATC FOR POLYCHAETES AND
EECs FOR ALL ZONES, COPPER, PORE WATER
NEW BEDFORD, MASSACHUSETTS

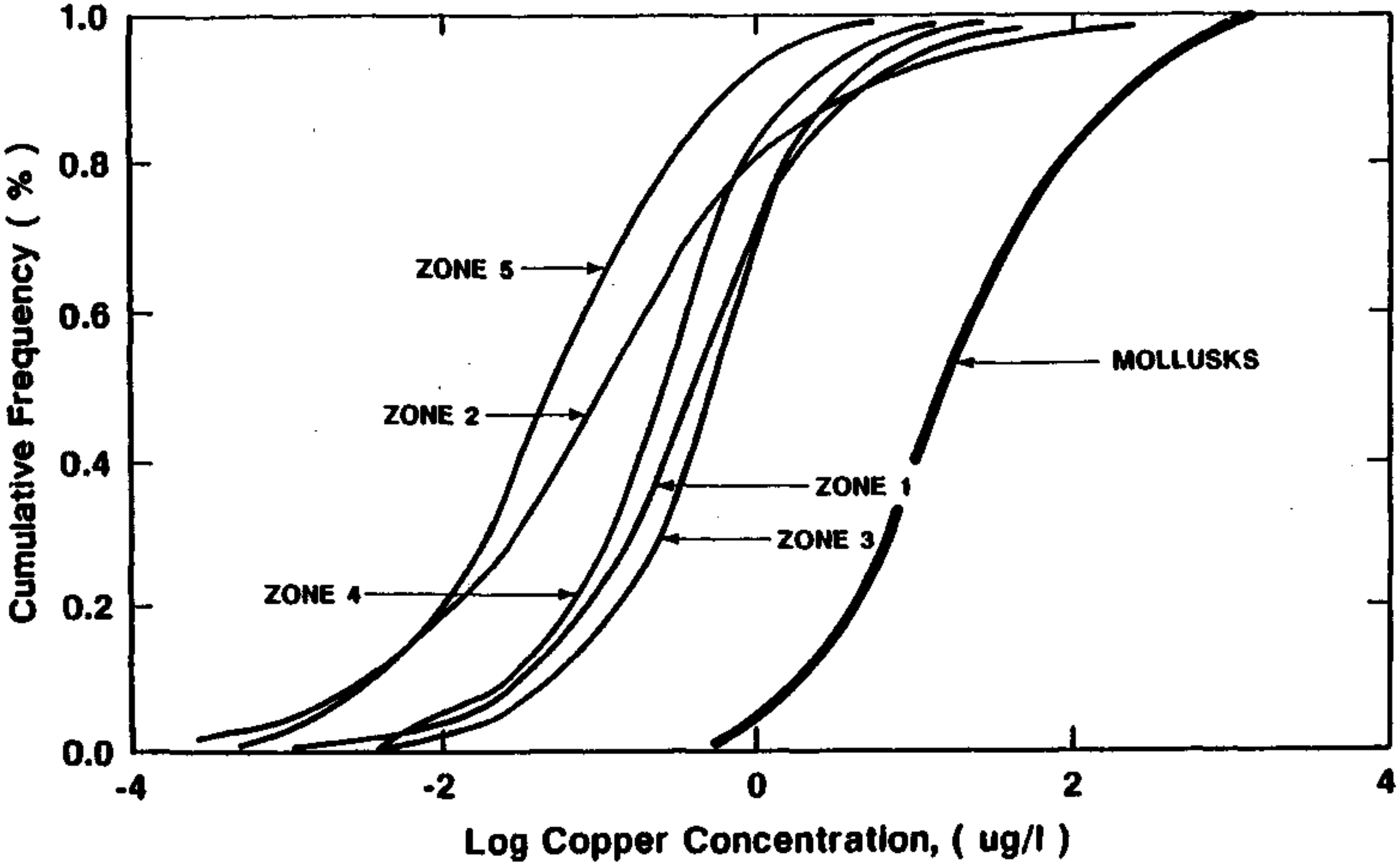

FIGURE C-32
MATC FOR MOLLUSKS AND
EECs FOR ALL ZONES, COPPER, PORE WATER
NEW BEDFORD, MASSACHUSETTS

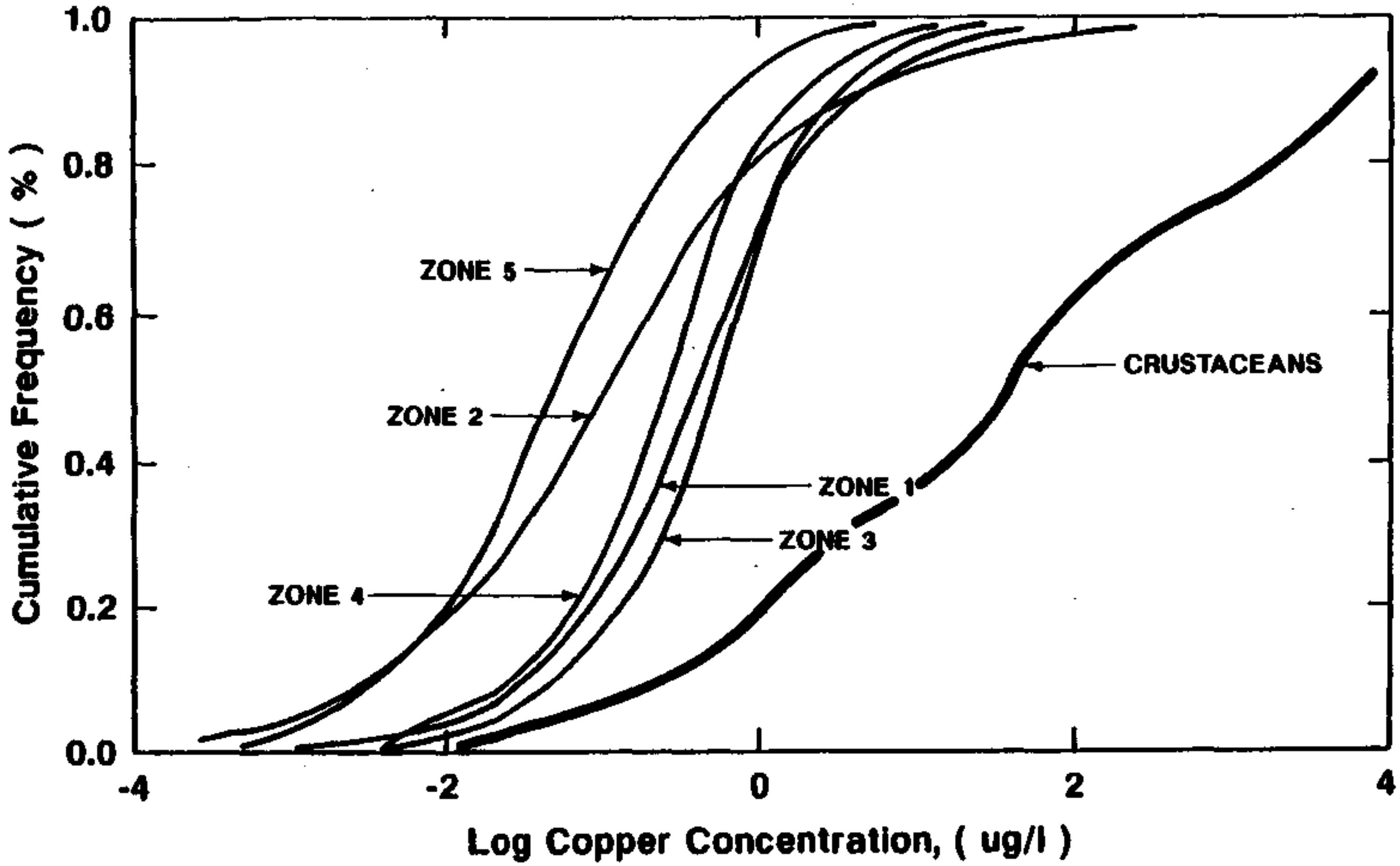

FIGURE C-33
MATC FOR CRUSTACEANS AND
EECs FOR ALL ZONES, COPPER, PORE WATER
NEW BEDFORD, MASSACHUSETTS

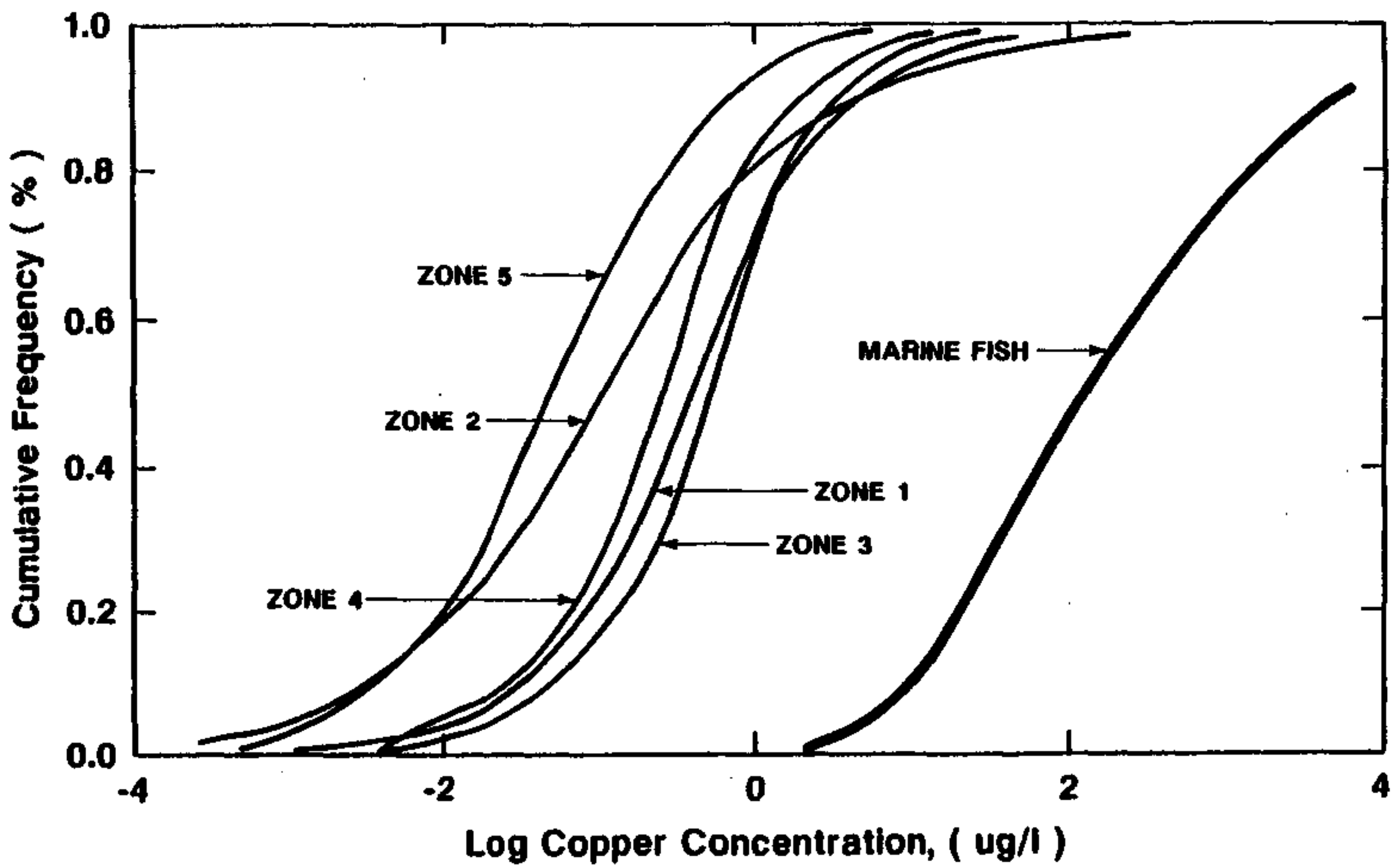

FIGURE C-34
MATC FOR MARINE FISH AND
EECs FOR ALL ZONES, COPPER, PORE WATER
NEW BEDFORD, MASSACHUSETTS

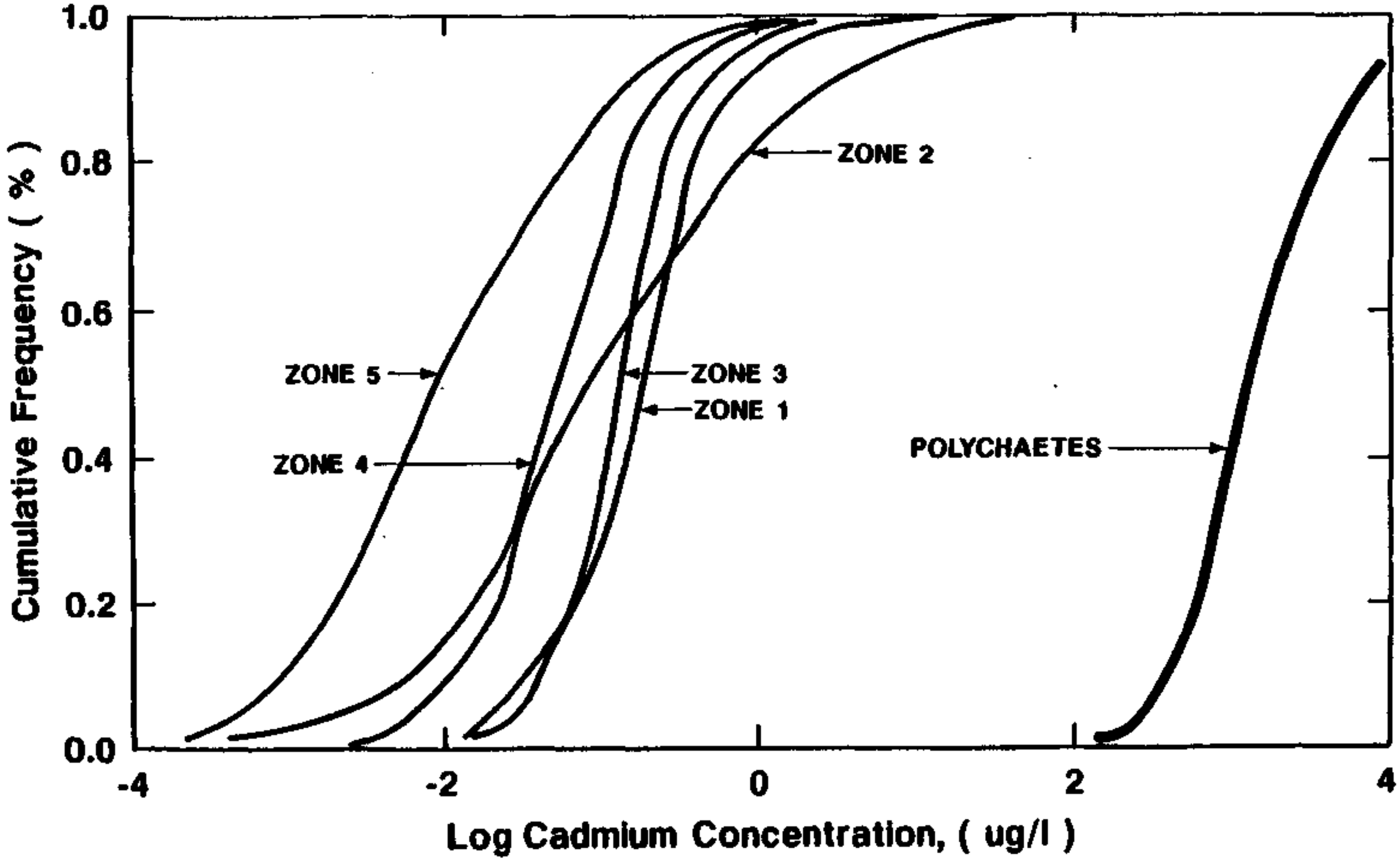

FIGURE C-35
MATC FOR POLYCHAETES AND
EECs FOR ALL ZONES, CADMIUM, PORE WATER
NEW BEDFORD, MASSACHUSETTS

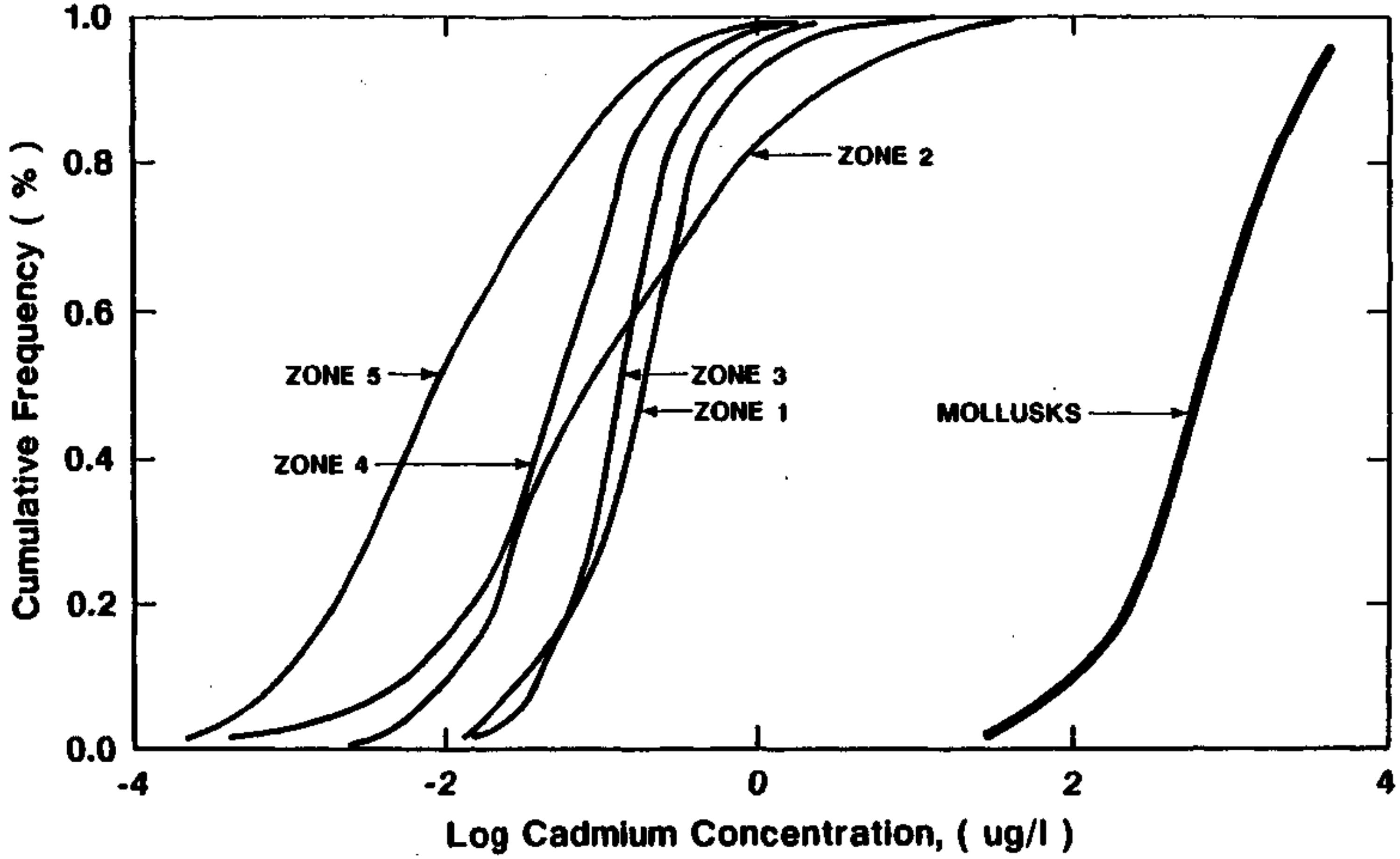

FIGURE C-36
MATC FOR MOLLUSKS AND
EECs FOR ALL ZONES, CADMIUM, PORE WATER
NEW BEDFORD, MASSACHUSETTS

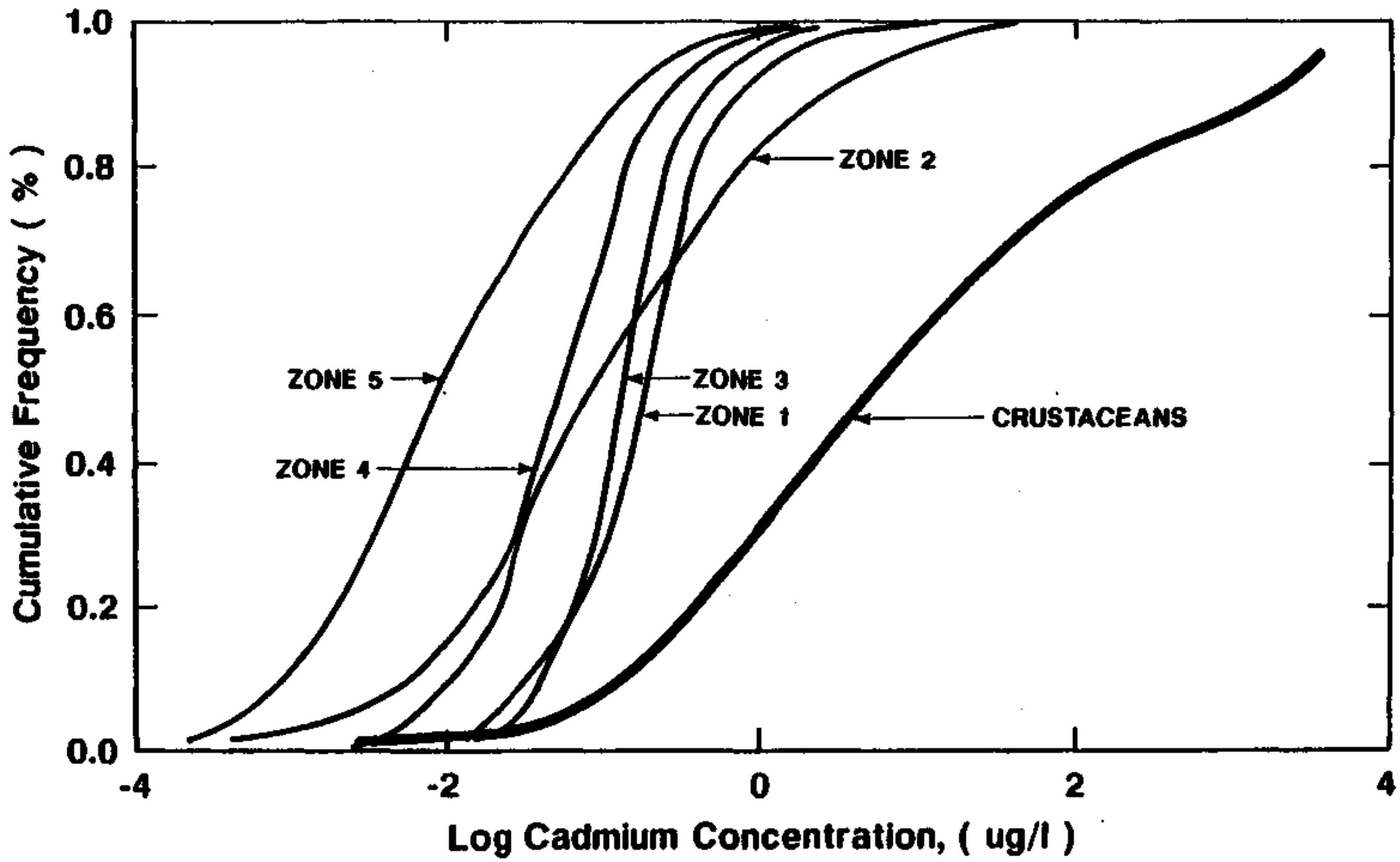

FIGURE C-37
MATC FOR CRUSTACEANS AND
EECs FOR ALL ZONES, CADMIUM, PORE WATER
NEW BEDFORD, MASSACHUSETTS

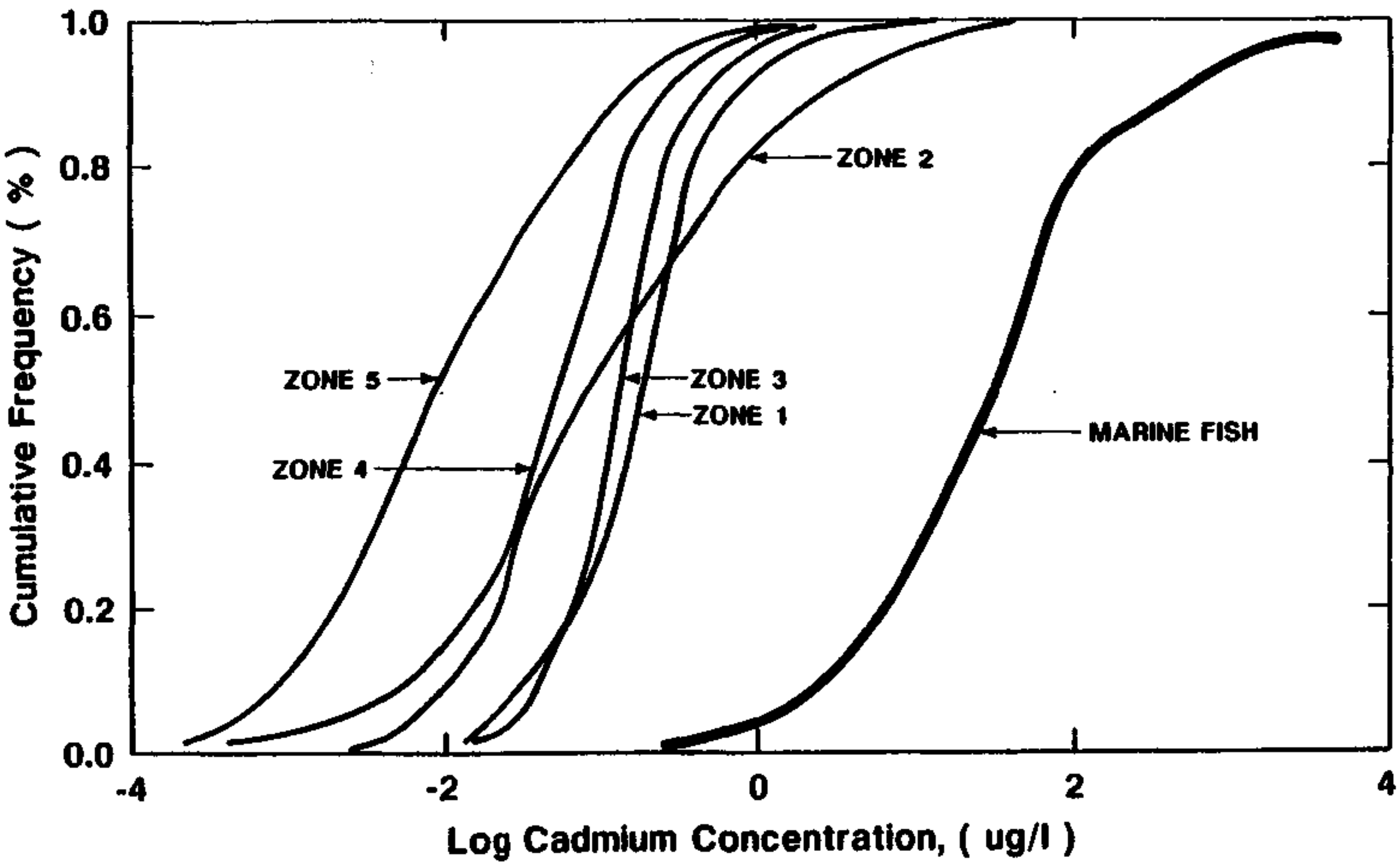

FIGURE C-38
MATC FOR MARINE FISH AND
EECs FOR ALL ZONES, CADMIUM, PORE WATER
NEW BEDFORD, MASSACHUSETTS

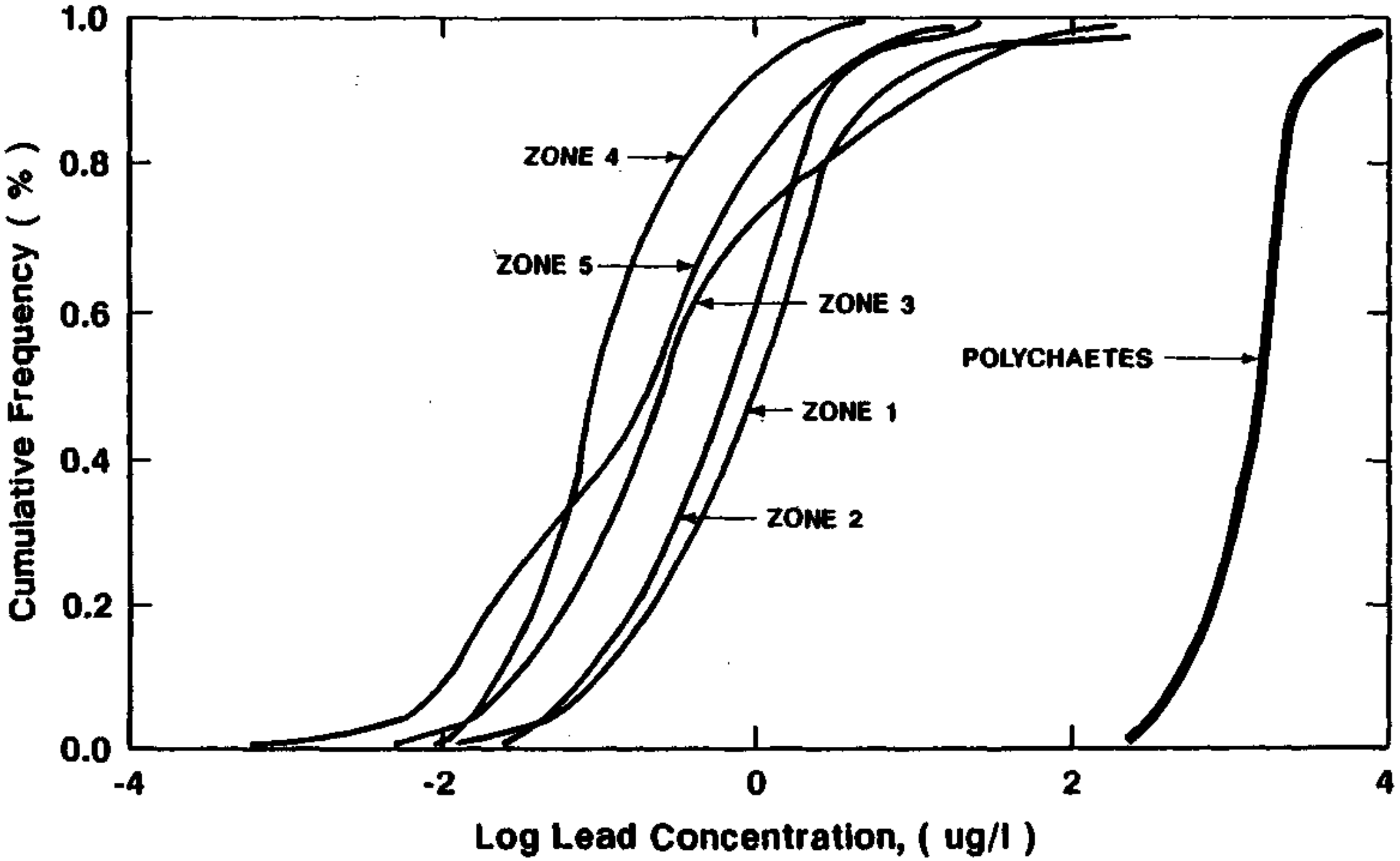

FIGURE C-39
MATC FOR POLYCHAETES AND
EECs FOR ALL ZONES, LEAD, PORE WATER
NEW BEDFORD, MASSACHUSETTS

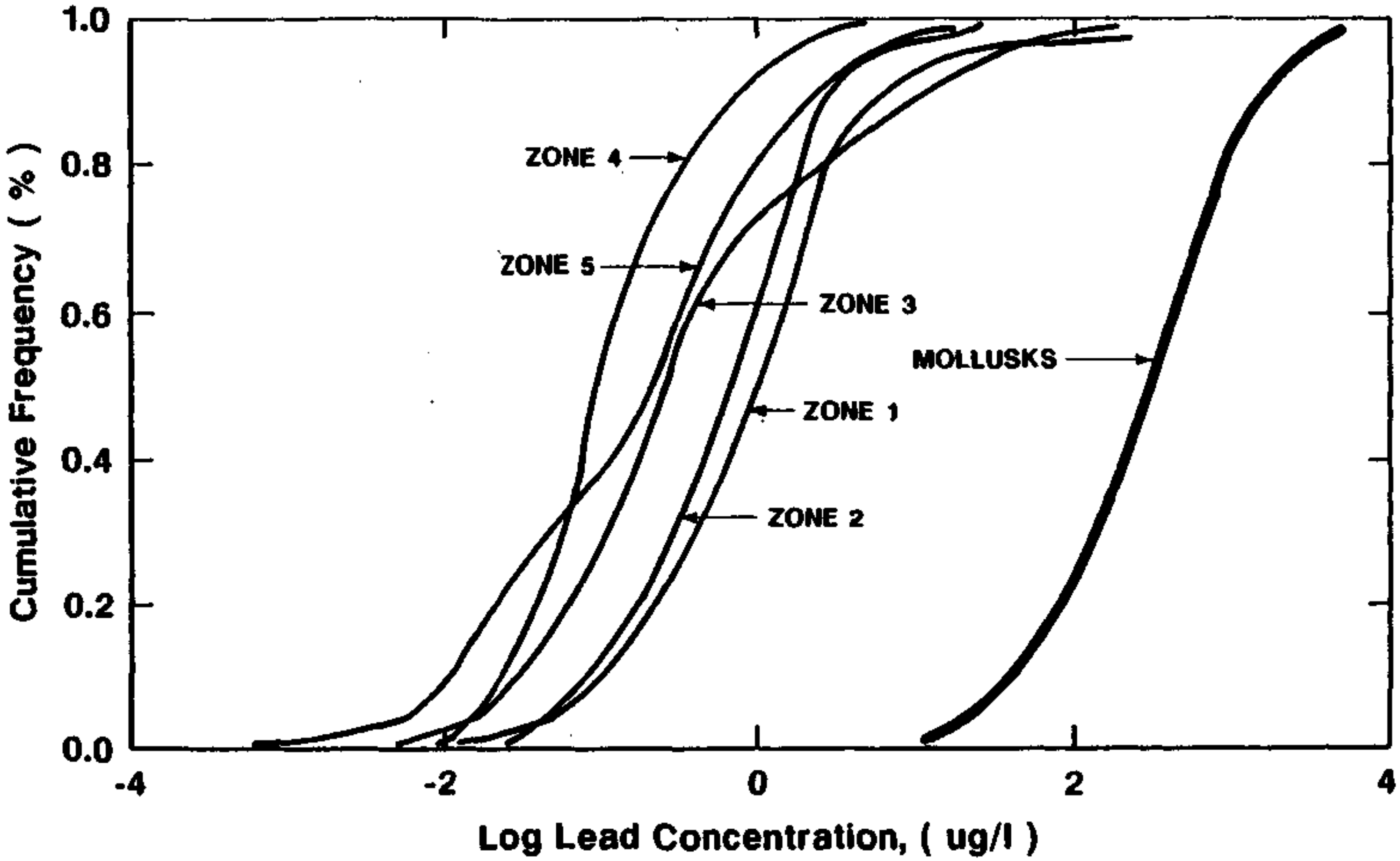

FIGURE C-40
MATC FOR MOLLUSKS AND
EECs FOR ALL ZONES, LEAD, PORE WATER
NEW BEDFORD, MASSACHUSETTS

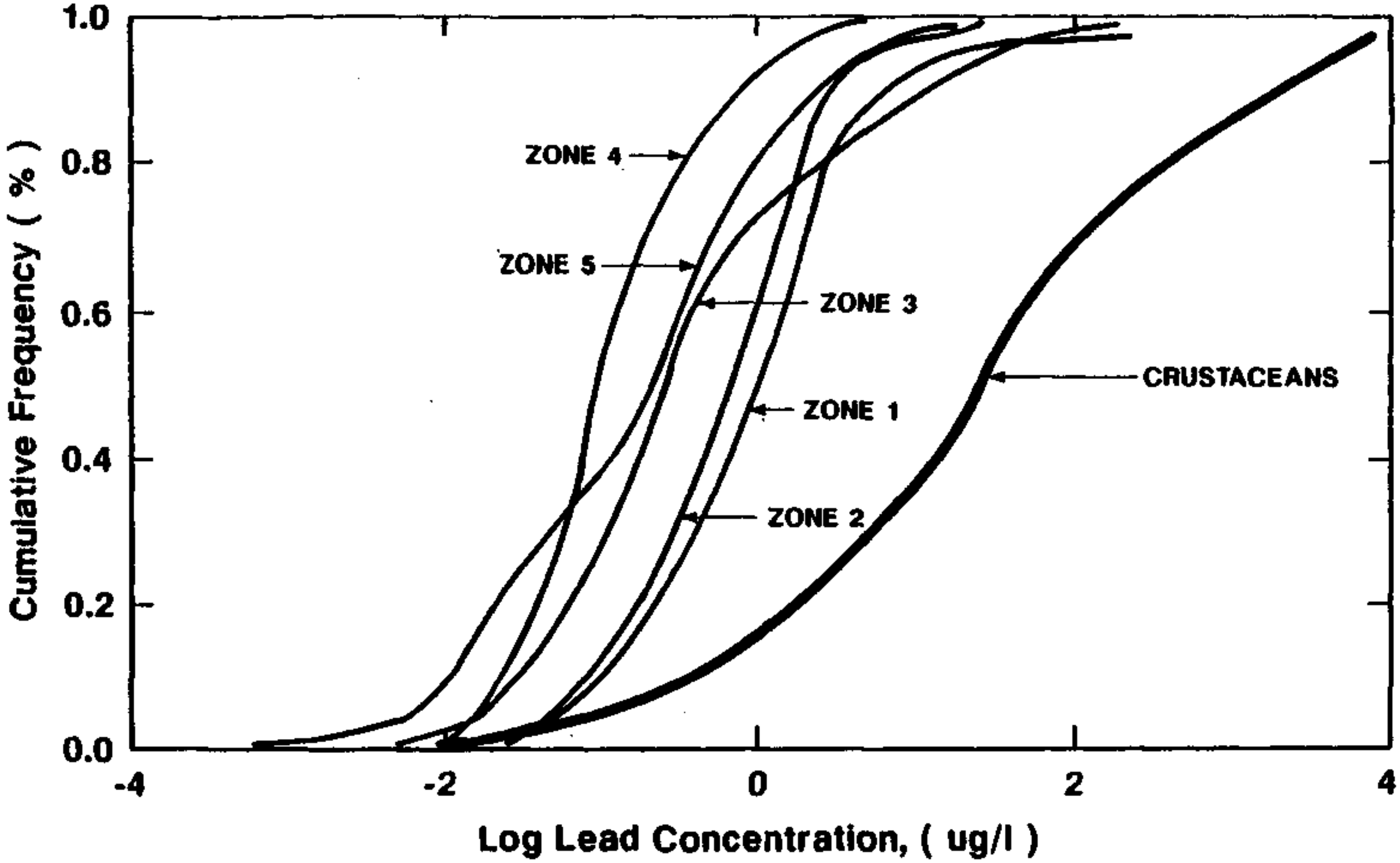

FIGURE C-41
MATC FOR CRUSTACEANS AND
EECs FOR ALL ZONES, LEAD, PORE WATER
NEW BEDFORD, MASSACHUSETTS

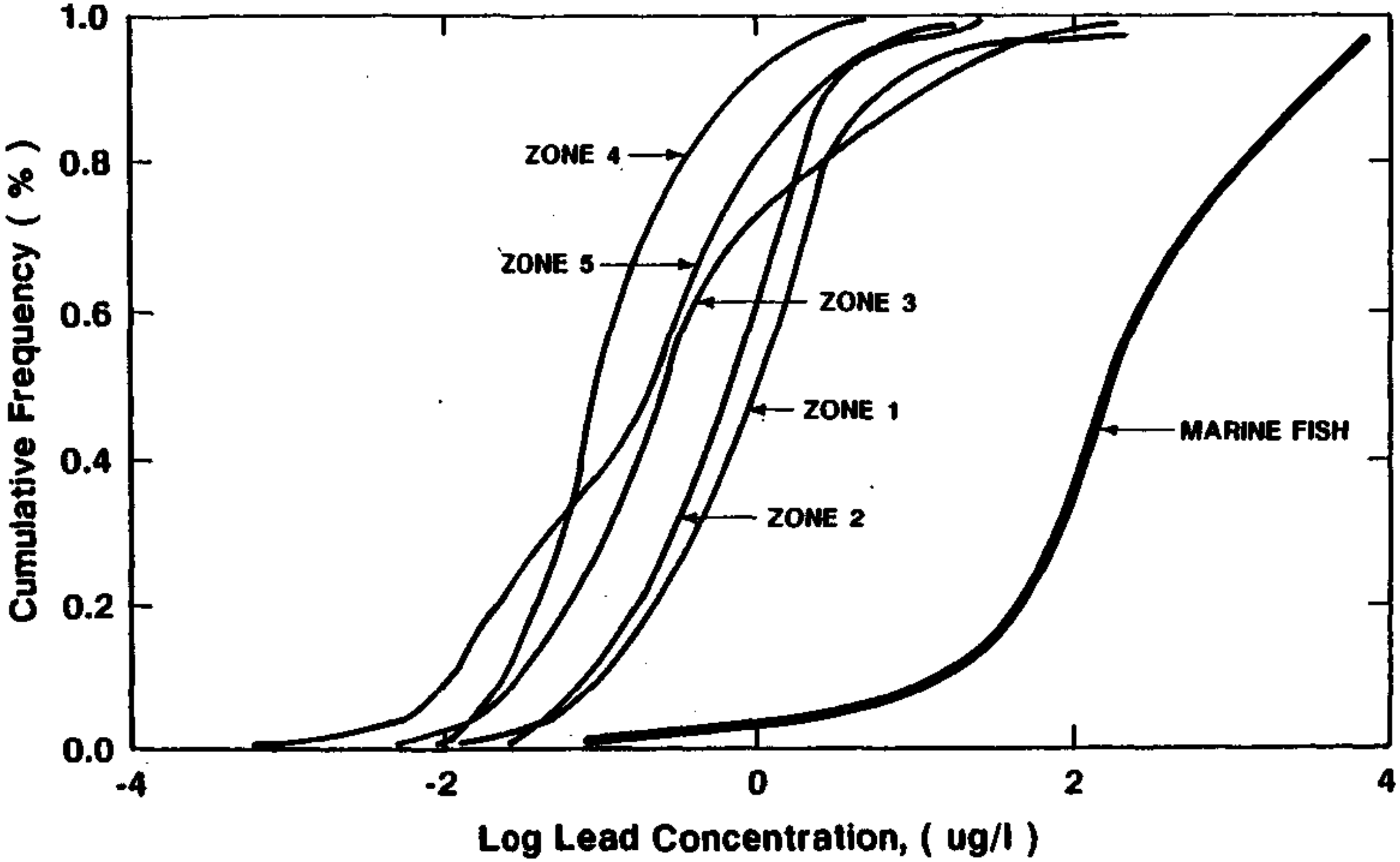

FIGURE C-42
MATC FOR MARINE FISH AND
EECs FOR ALL ZONES, LEAD, PORE WATER
NEW BEDFORD, MASSACHUSETTS

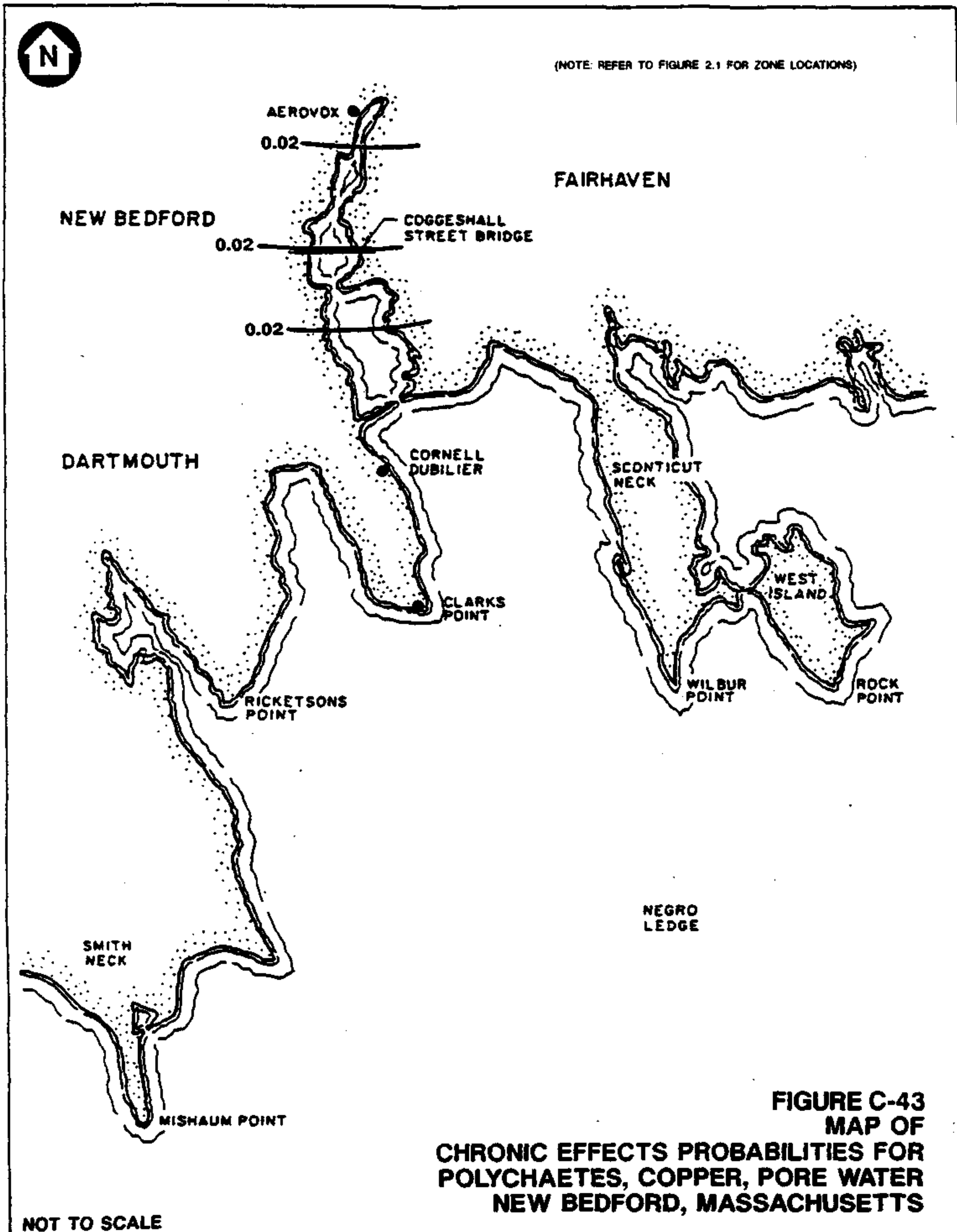

FIGURE C-43
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR POLYCHAETES, COPPER, PORE WATER NEW BEDFORD, MASSACHUSETTS

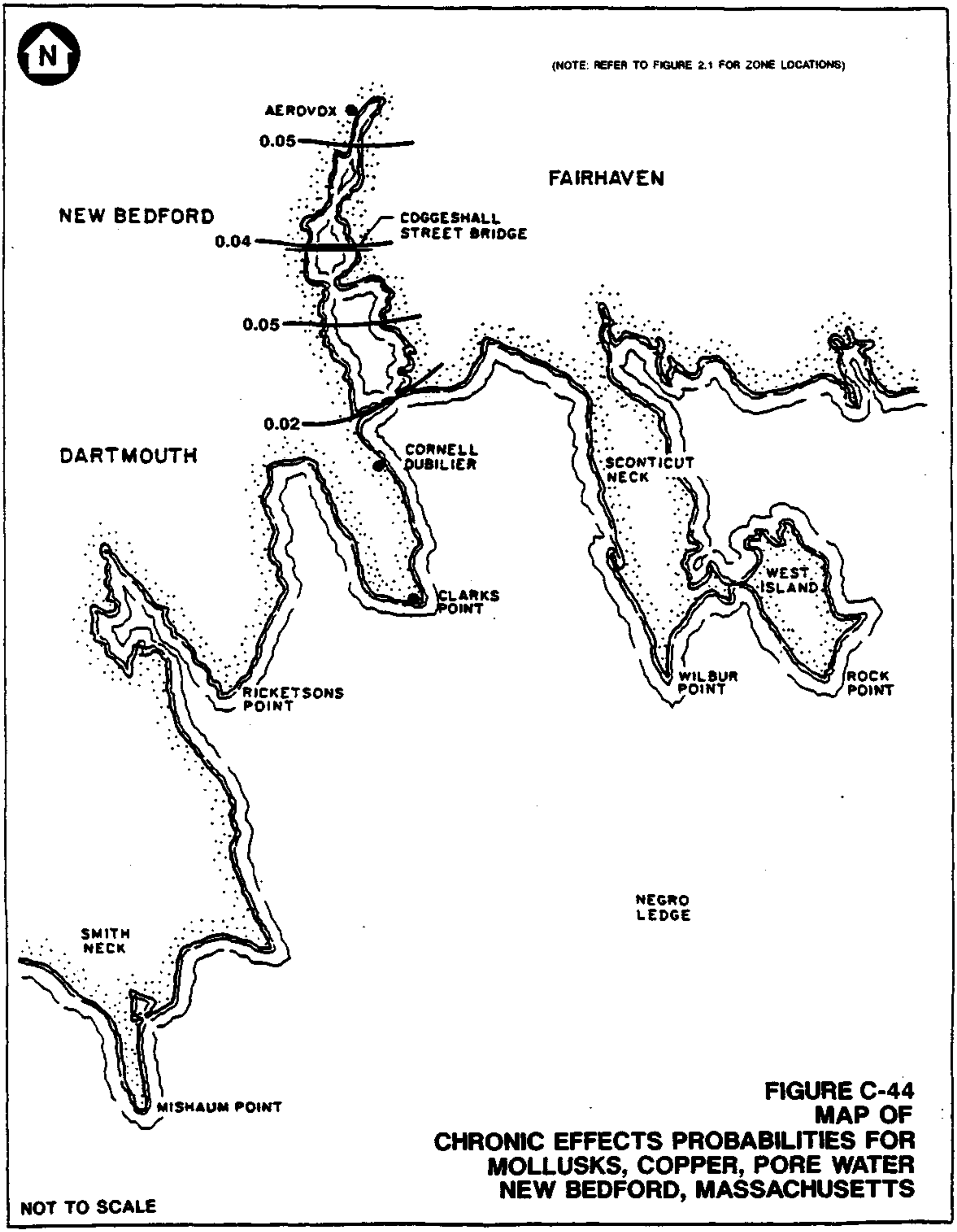

FIGURE C-44
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR MOLLUSKS, COPPER, PORE WATER
NEW BEDFORD, MASSACHUSETTS

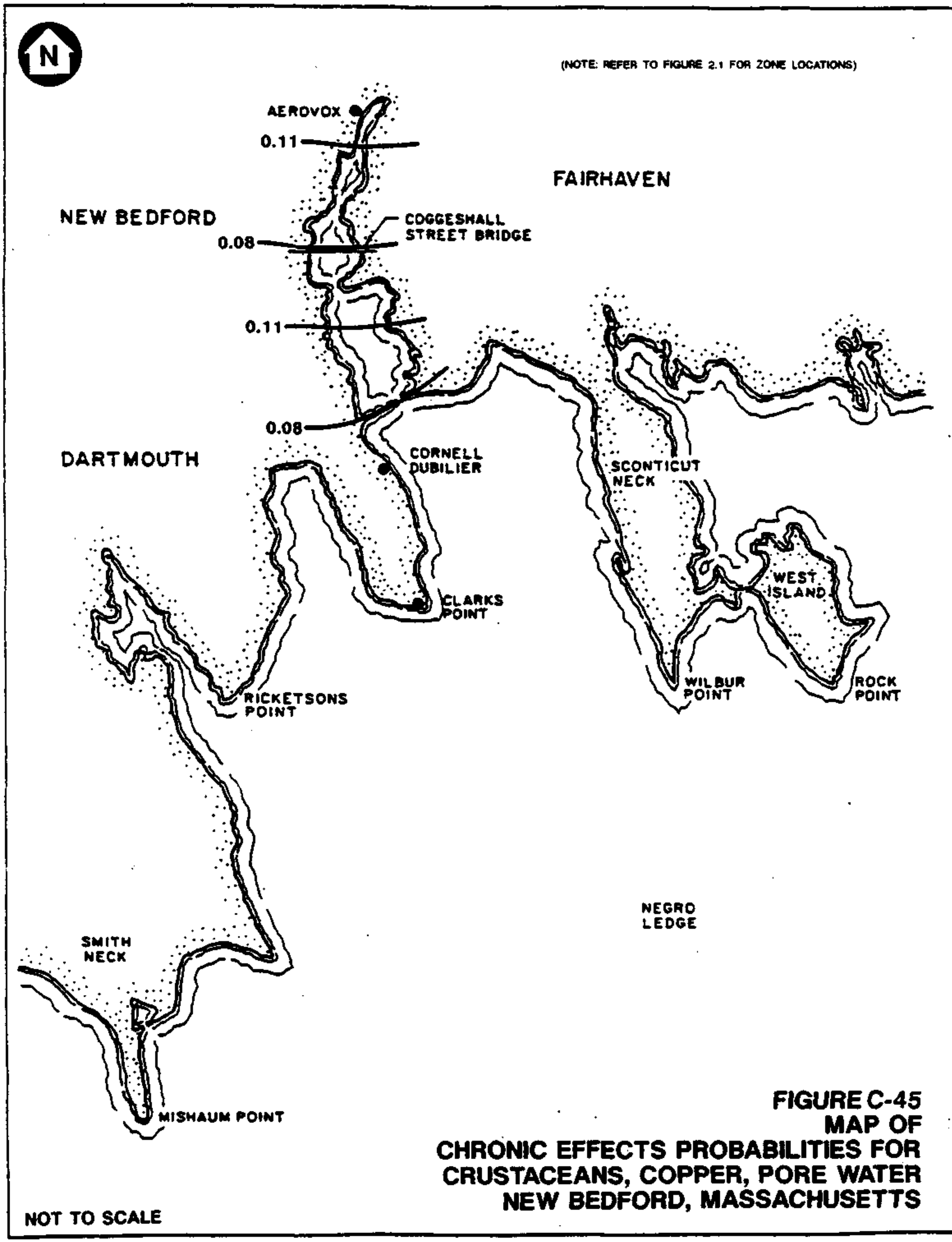

FIGURE C-45
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
CRUSTACEANS, COPPER, PORE WATER
NEW BEDFORD, MASSACHUSETTS

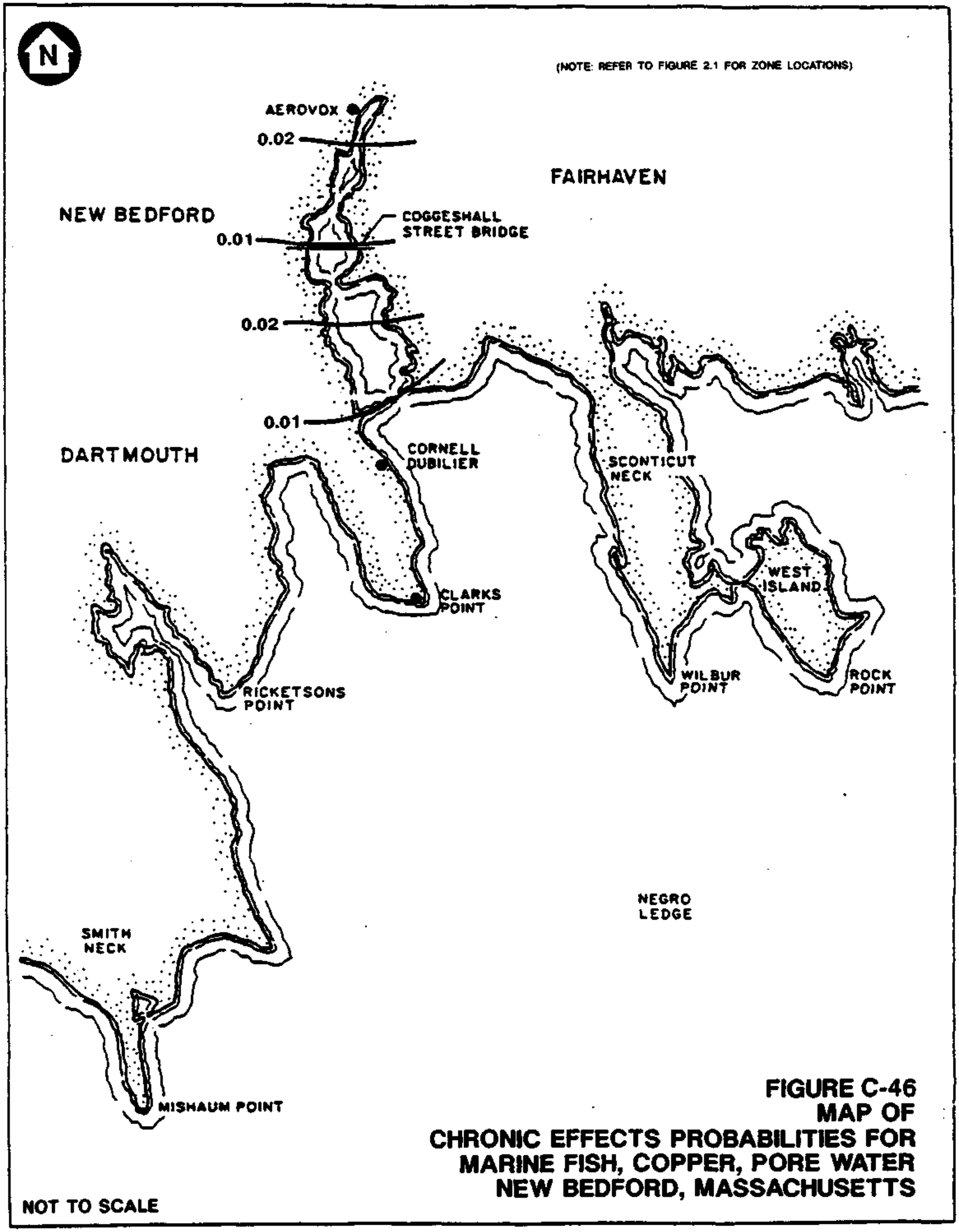

FIGURE C-46
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
MARINE FISH, COPPER, PORE WATER
NEW BEDFORD, MASSACHUSETTS

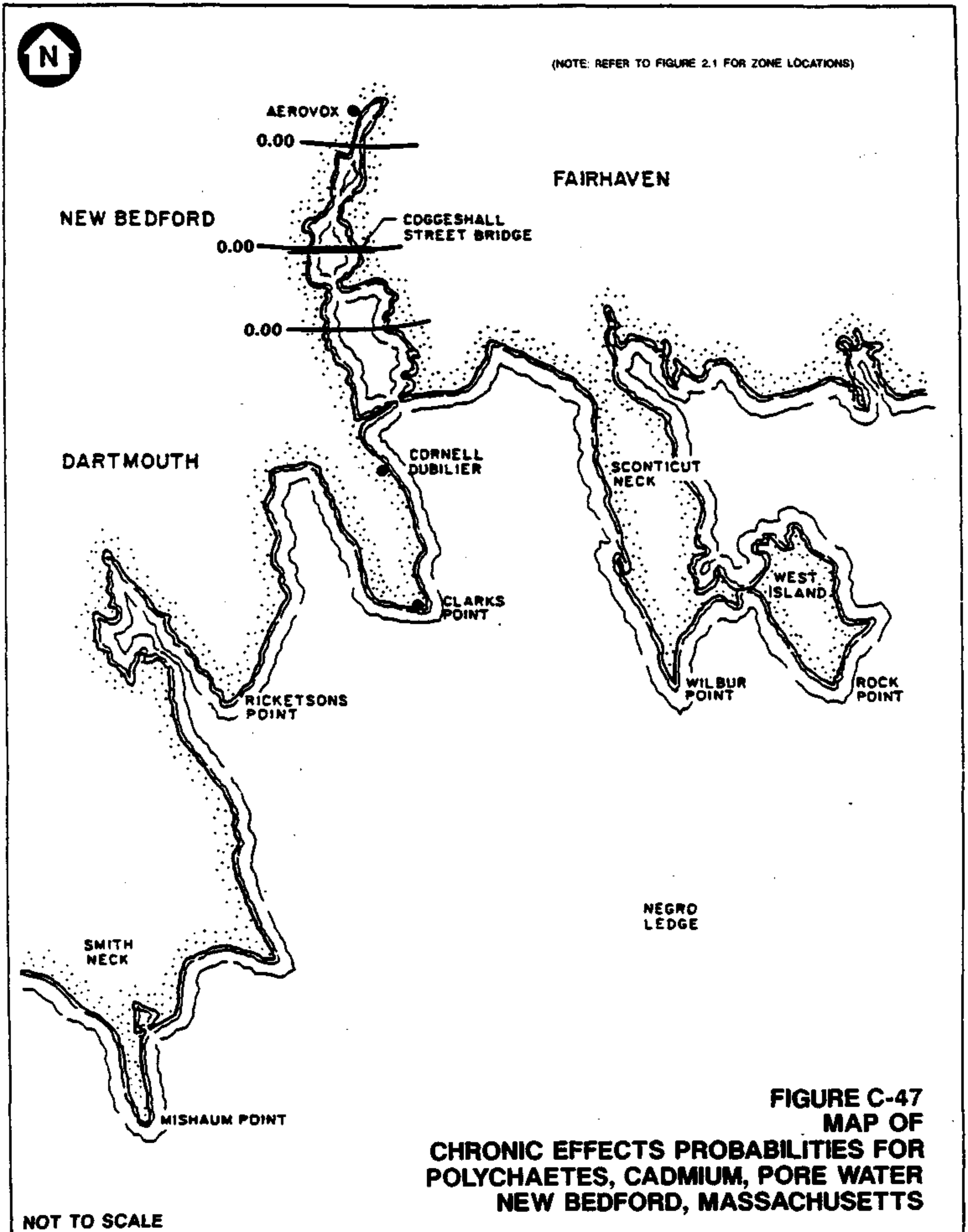

FIGURE C-47
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR POLYCHAETES, CADMIUM, PORE WATER NEW BEDFORD, MASSACHUSETTS

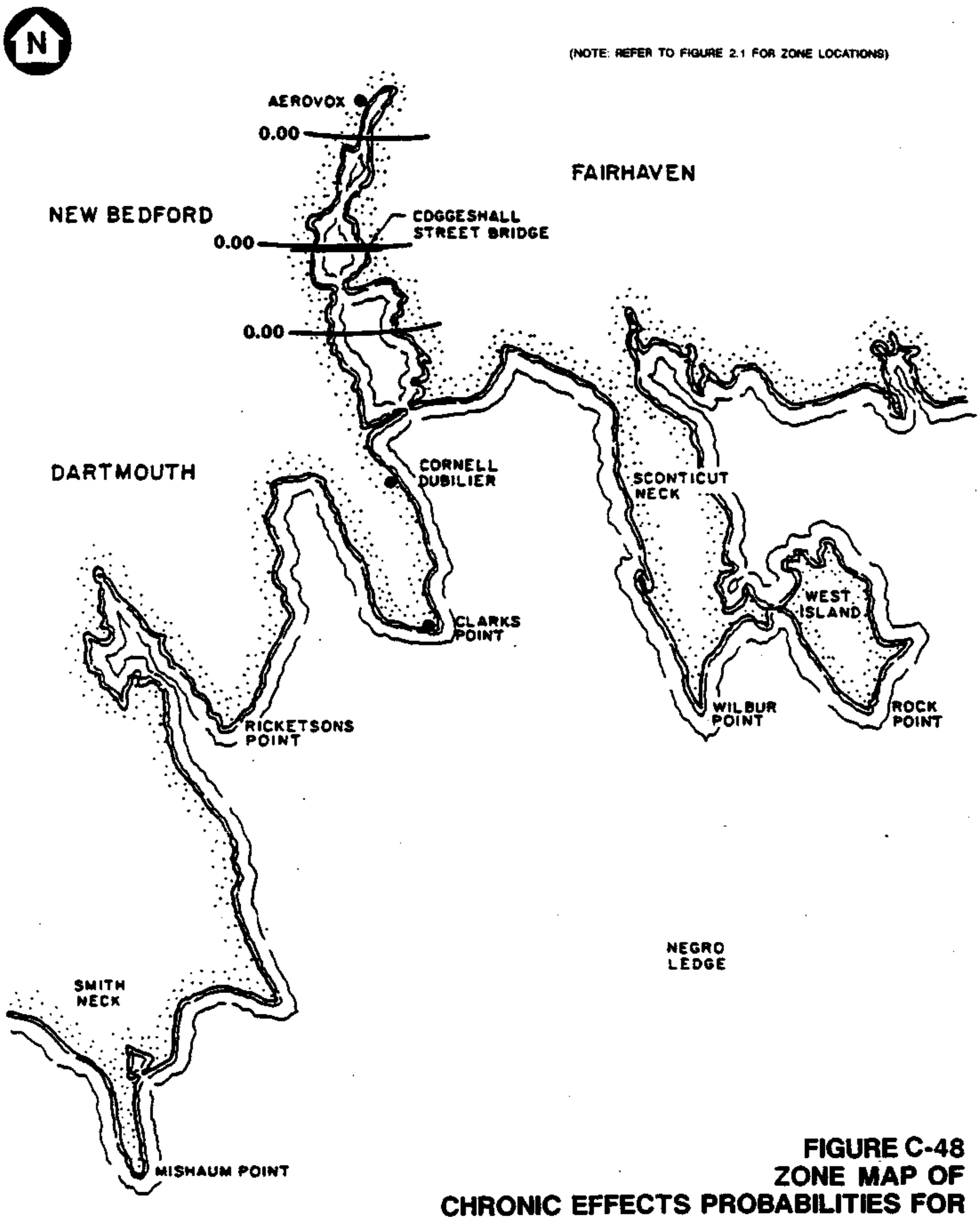

FIGURE C-48
ZONE MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
MOLLUSKS, CADMIUM, PORE WATER
NEW BEDFORD, MASSACHUSETTS

NOT TO SCALE

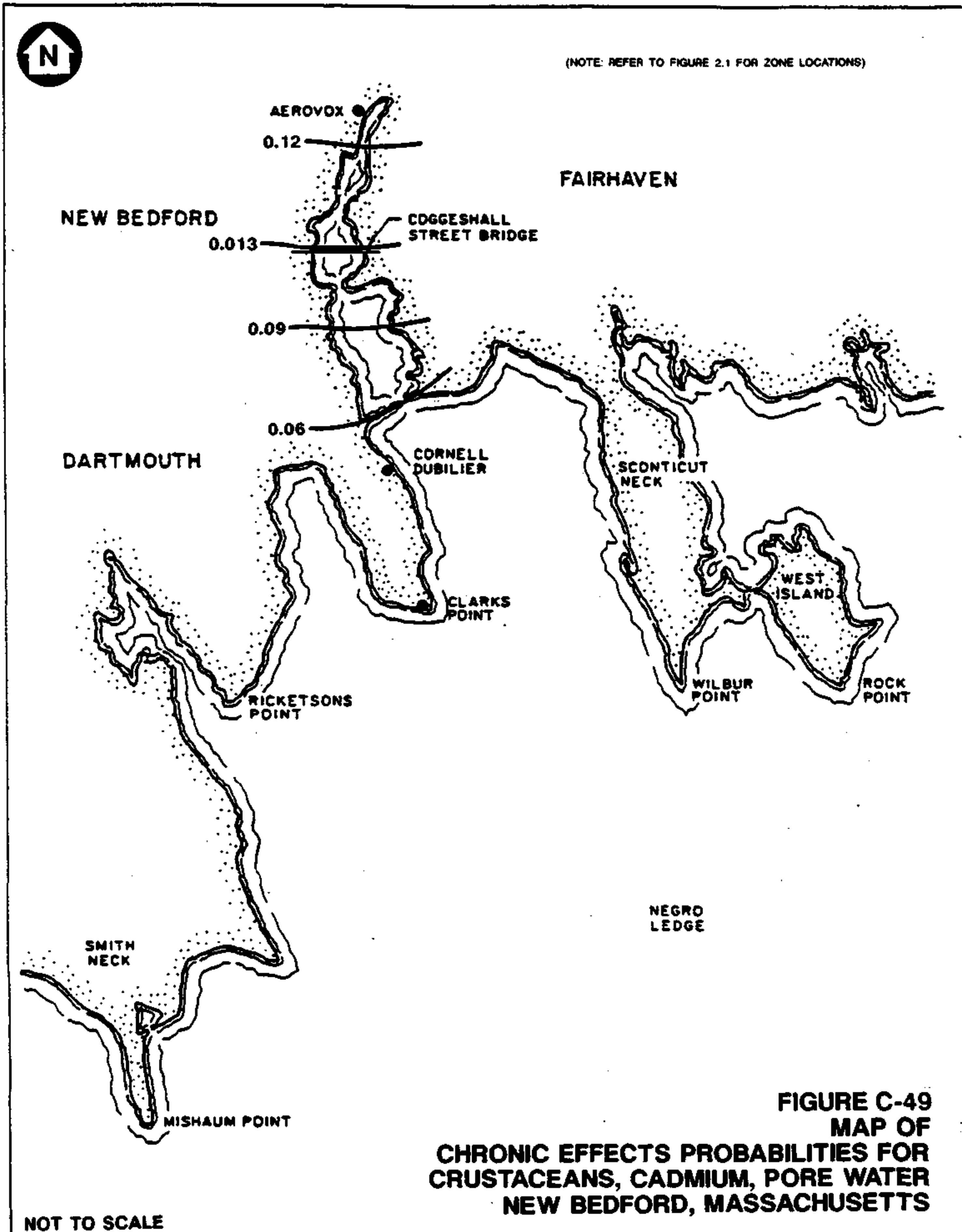

FIGURE C-49
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR CRUSTACEANS, CADMIUM, PORE WATER
NEW BEDFORD, MASSACHUSETTS

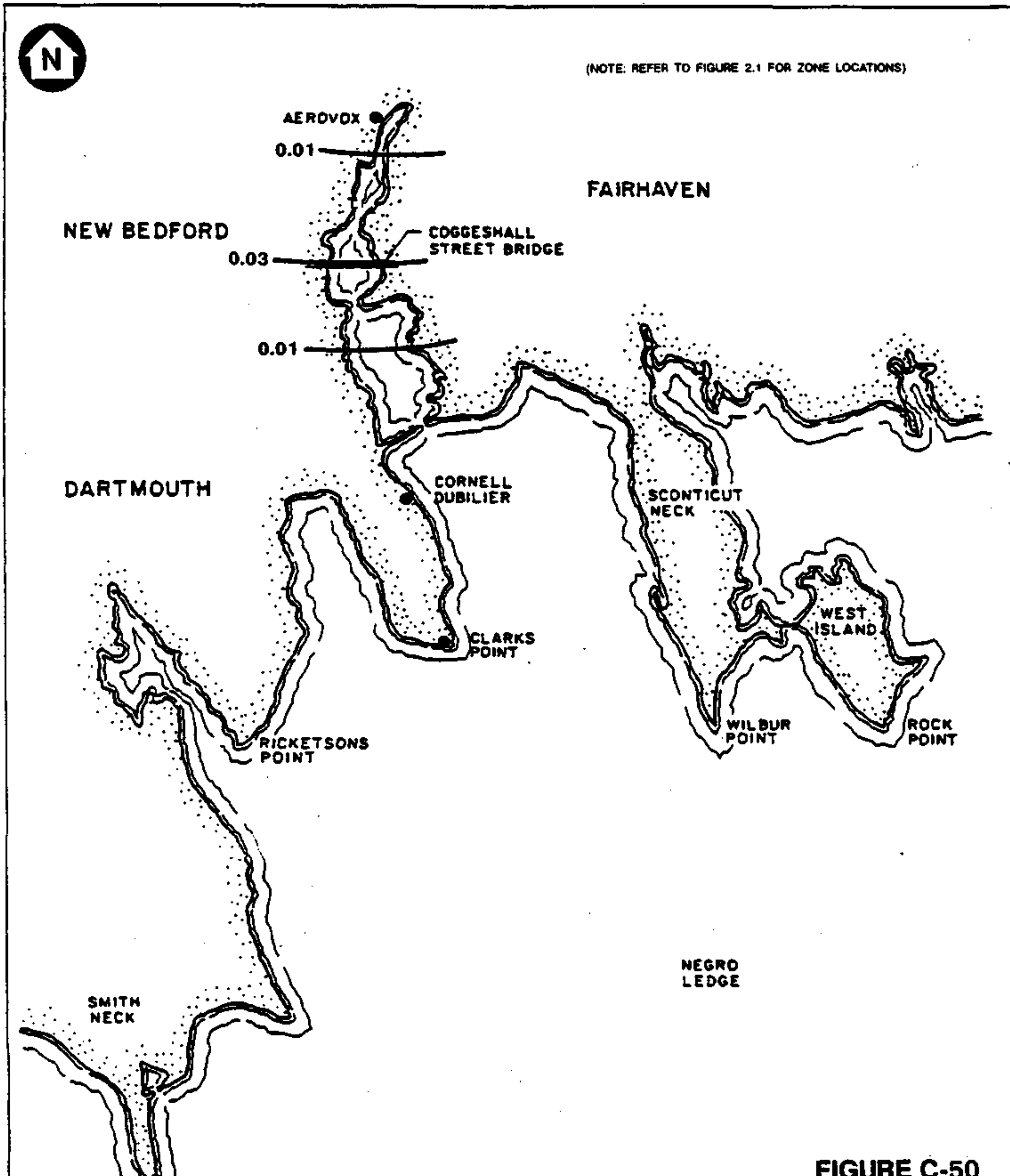

FIGURE C-50
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
MARINE FISH, CADMIUM, PORE WATER
NEW BEDFORD, MASSACHUSETTS

NOT TO SCALE

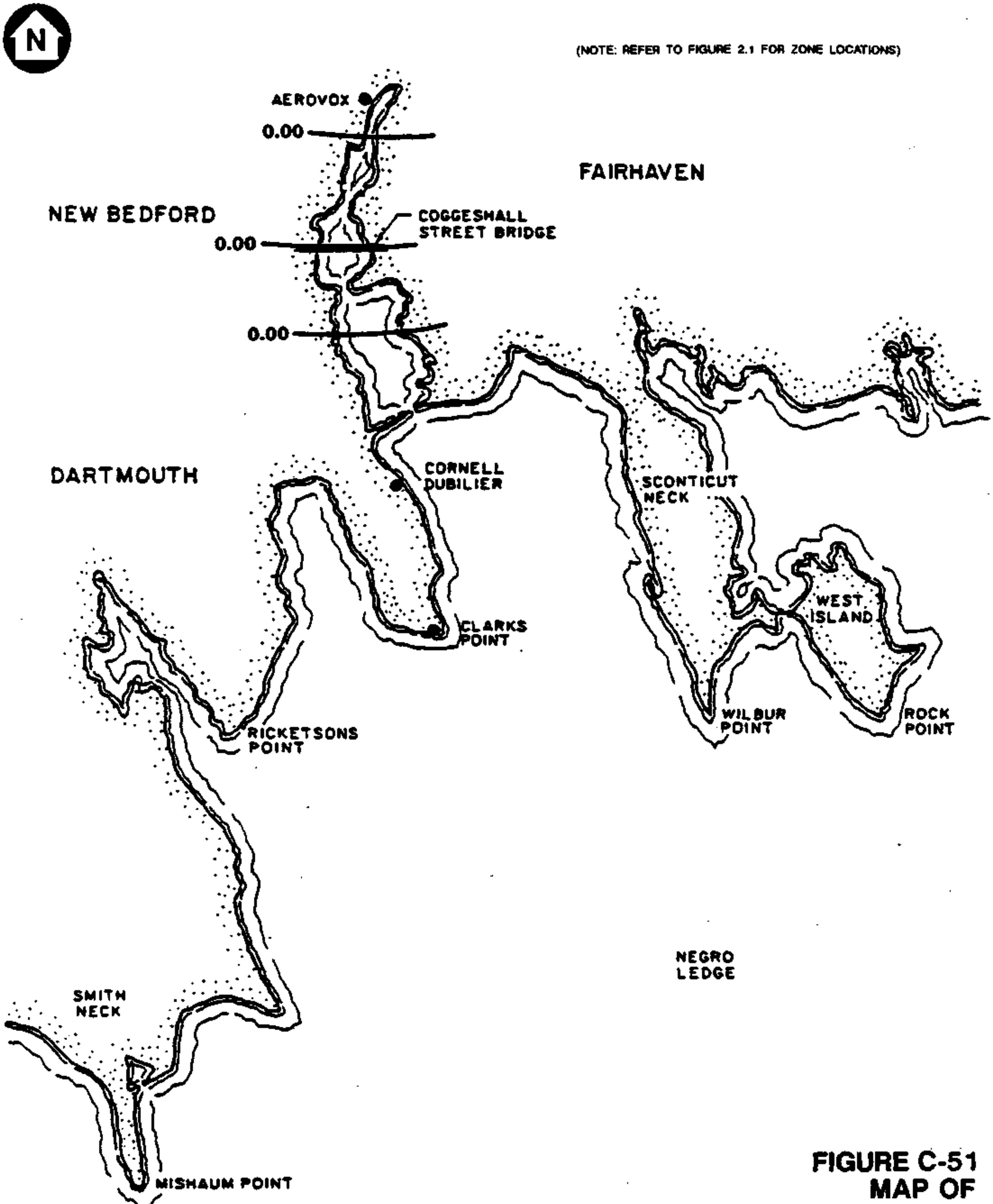

FIGURE C-51
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
POLYCHAETES, LEAD, PORE WATER
NEW BEDFORD, MASSACHUSETTS

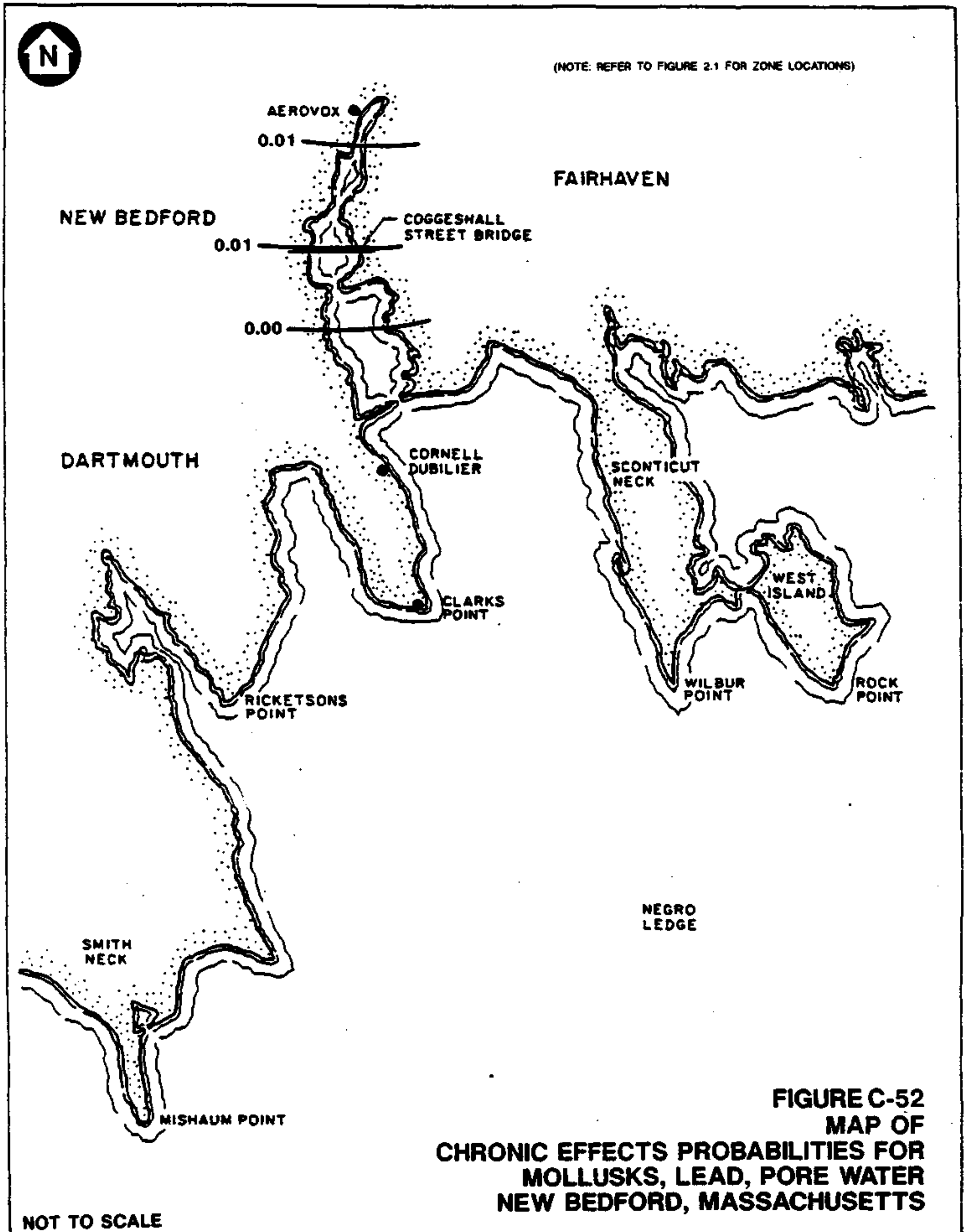

FIGURE C-52
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
MOLLUSKS, LEAD, PORE WATER
NEW BEDFORD, MASSACHUSETTS

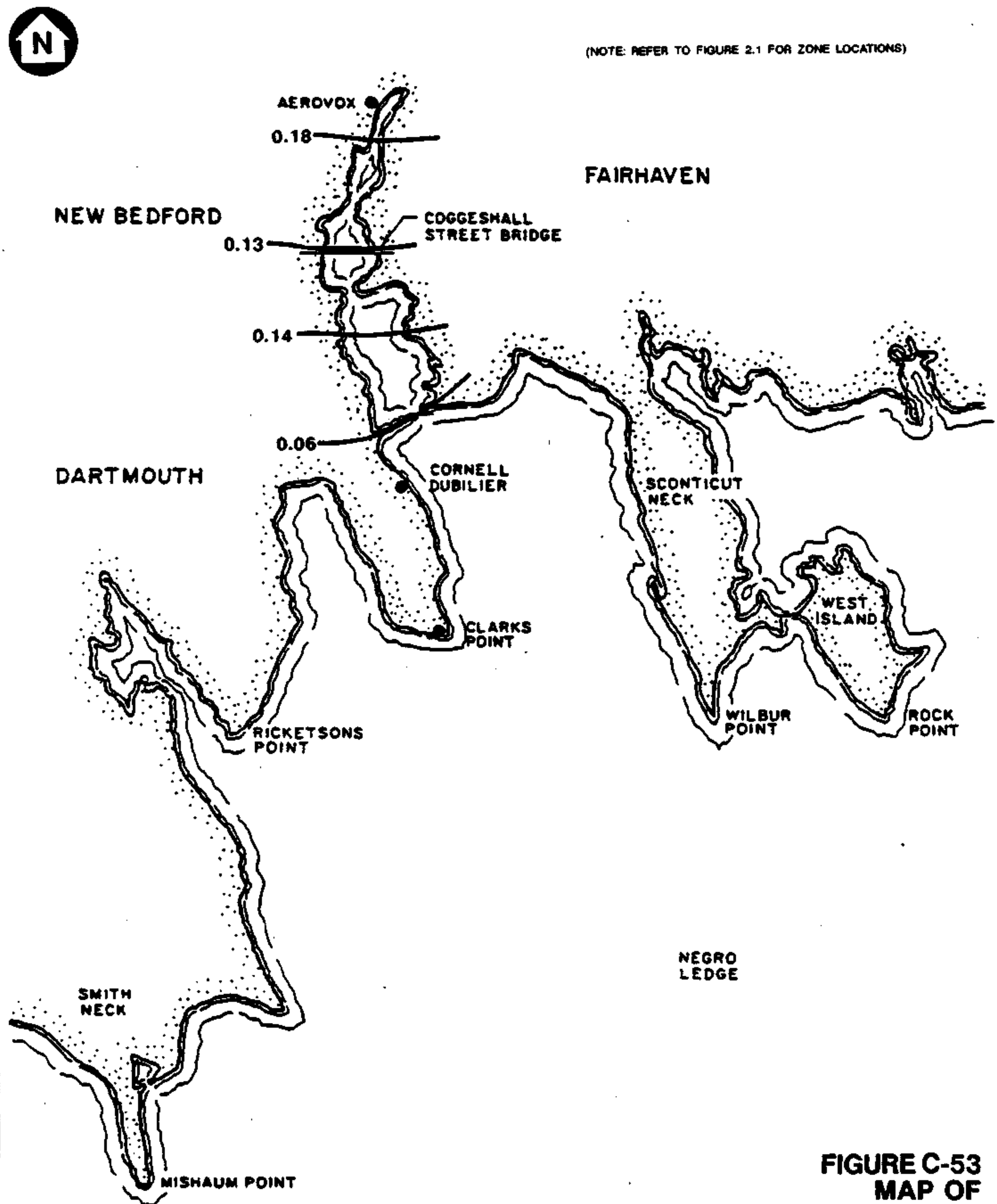

FIGURE C-53
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR CRUSTACEANS, LEAD, PORE WATER
NEW BEDFORD, MASSACHUSETTS

NOT TO SCALE

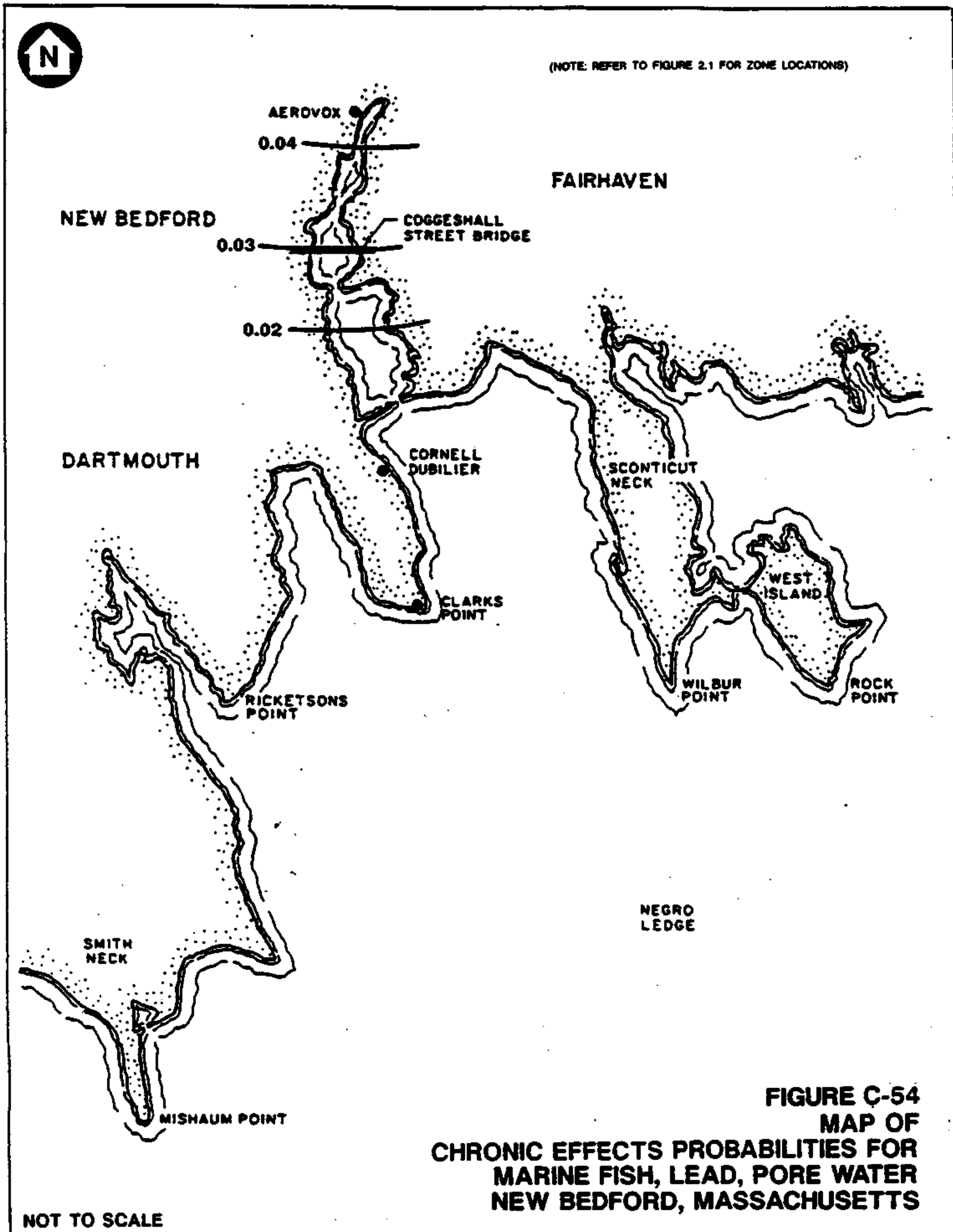

FIGURE C-54
MAP OF
CHRONIC EFFECTS PROBABILITIES FOR
MARINE FISH, LEAD, PORE WATER
NEW BEDFORD, MASSACHUSETTS