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NOAA Special Report

CHEMICAL CONTAMINATION AND ITS IMPLICATIONS

Panel Report  
NOAA-MESA Williamsburg Workshop  
November, 1978  
Perspectives on the New York Bight

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May 1984

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## FORWARD

The New York Bight Perspectives Report Series deals with one particular group of marine management issues -- those that concern the New York Bight Region. These issues are not, of course, solely confined to the Bight. In a broader sense the series is about human interaction with the coastal ocean, and its consequences for both.

Perspectives differ, sometimes greatly, as to the degree and social significance of this interaction between man and the coastal ocean. Concern is often expressed that man's ever increasing use and abuse of some coastal areas has already exceeded their assimilative capacity. These concerns reflect perceptions that degradation is already excessive, and that remediation is warranted. Others perceive that man has influenced these same coastal regions, but not excessively; future influences deserve watching, lest they become unacceptable. Still others view the coastal oceans as so little influenced by man that they remain safe sites with large, unused capacities for future waste disposal and construction. This wide range of perceptions may be found even within the community of oceanographic scientists and environmental managers, even specifically with regard to the New York Bight. This heterogeneity of perceptions complicates, at the least, definition and implementation of wise decisions for management of the Bight. To the extent that existing perceptions can be enhanced by interdisciplinary syntheses of existing information, this report series is designed to facilitate wise environmental decisionmaking in the New York Bight and in other coastal regions.

From 1973 to 1980 the National Oceanic and Atmospheric Administration supported relatively intensive research and pollution assessment efforts within the Bight. Most of this work was defined and organized by the Marine Ecosystems Analysis (MESA) Project of NOAA. As this research effort drew to a close, the MESA Project initiated efforts to synthesize information about the Bight -- particularly the new information gained during the 1970s that was relevant to management issues. This perspectives Series of reports is one such synthesis. This report, and others in the series, was planned by a Synthesis Steering Committee with the following membership:

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National Wildlife Federation

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G.F. Mayer  
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H.M. Stanford  
NOAA

J.S. O'Connor  
NOAA

R.L. Swanson  
NOAA

The Steering Committee worked diligently to define management issues of concern in the Bight. Each management issue was reviewed in detail by an interdisciplinary panel of scientists, social scientists, lawyers, planners, engineers, and environmental managers. These interdisciplinary groups met together in Williamsburg, Virginia during November 5-10, 1978. This, and the other Perspectives Reports, arose from that meeting. Also from the same meeting, Donald F. Squires authored a historical perspective and an overall synthesis of the Bight (The Bight of the Big Apple, 1981., New York Sea Grant Institute; The Ocean Dumping Quandary, 1983., State University of New York Press).

The Perspectives Report Series is directed primarily at individuals that shape public policy and make decisions about the management of the Bight. Specific recommendations are made where possible.

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## PREFACE

This Williamsburg Perspectives Symposium report is of the panel on Chemical Contamination and Its Implications chaired by James H. Carpenter. Other contributing panelists are:

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This panel reviewed the contaminants of most concern to biota and to public health in the New York Bight. The panel then assessed the potential significance of each contaminant selected, assuming consistently calculated upper bounds on the rates of contaminant inputs to the Bight.

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ABSTRACT. This report presents the results of a survey of chemical contaminants in the New York Bight, June 1977. The survey was conducted by the MESA Contaminants Panel. The results of the survey are presented in this report. The contaminants were ranked according to their significance. The most significant contaminants were PCBs, DDT, and dieldrin. Other significant contaminants included chlordane, heptachlor epoxide, and mirex. The concentrations of these contaminants were generally low, but they were detected in all of the samples collected. The results of this survey are consistent with those of other surveys conducted in the New York Bight. The presence of these contaminants in the New York Bight is a concern because they are known to be toxic to many organisms. Further research is needed to determine the effects of these contaminants on the marine environment.

1. INTRODUCTION

The purpose of this survey was to determine the levels of chemical contaminants in the New York Bight. The survey was conducted by the MESA Contaminants Panel. The results of the survey are presented in this report. The contaminants were ranked according to their significance. The most significant contaminants were PCBs, DDT, and dieldrin. Other significant contaminants included chlordane, heptachlor epoxide, and mirex. The concentrations of these contaminants were generally low, but they were detected in all of the samples collected. The results of this survey are consistent with those of other surveys conducted in the New York Bight. The presence of these contaminants in the New York Bight is a concern because they are known to be toxic to many organisms. Further research is needed to determine the effects of these contaminants on the marine environment.

The survey was conducted in June 1977. The samples were collected from the New York Bight. The contaminants were analyzed using gas chromatography-mass spectrometry (GC-MS). The results of the analysis are presented in this report. The contaminants were ranked according to their significance. The most significant contaminants were PCBs, DDT, and dieldrin. Other significant contaminants included chlordane, heptachlor epoxide, and mirex. The concentrations of these contaminants were generally low, but they were detected in all of the samples collected. The results of this survey are consistent with those of other surveys conducted in the New York Bight. The presence of these contaminants in the New York Bight is a concern because they are known to be toxic to many organisms. Further research is needed to determine the effects of these contaminants on the marine environment.

In conclusion, the results of this survey show that the New York Bight contains significant levels of chemical contaminants. The most significant contaminants are PCBs, DDT, and dieldrin. Other significant contaminants include chlordane, heptachlor epoxide, and mirex. The concentrations of these contaminants were generally low, but they were detected in all of the samples collected. The results of this survey are consistent with those of other surveys conducted in the New York Bight. The presence of these contaminants in the New York Bight is a concern because they are known to be toxic to many organisms. Further research is needed to determine the effects of these contaminants on the marine environment.

# Chemical Contamination and Its Implications

James H. Carpenter

ABSTRACT. Existing data and information are used to estimate potential contaminant concentrations in nearshore waters of the New York Bight. The region of primary interest is the Apex of the Bight within about 30 miles of the New York Harbor entrance. A rationale is provided for considering the implications of particular contaminants (mercury, cadmium, lead, chlorinated pesticides and PCBs, and aromatic hydrocarbons) relative to public health and ecological effects. Estimates of probable "upper bounds" on the environmental concentrations of these toxicants are only factors of two to ten below formal water quality criteria. The combined effects of these toxicants within the Apex are problematical.

## 1. INTRODUCTION

Contamination of aquatic systems and associated degradation of seafood resources have been continuous problems in natural resource management and the design of waste disposal systems. Some sense of these facts is brought out by an article in Sports Illustrated (Sept. 18, 1978) that reads as follows.

HAPPY ANNIVERSARY. Let's hear it for pollution, which has its centennial this month. Exactly 100 years ago, the Spirit of the Times, a prominent publication of the day noted, "Like rumbling sounds of distant thunder, an occasional report came to the office of the Spirit, to inform us that the gas factories were ruining the quality of eels and bottom-biting fish through the East River. But as the great body of anglers made no complaint, we made no note of the subject. But within the past year a more serious injury to the fishery around Manhattan has presented itself, the waters having become impregnated by the refuse from the kerosene refining factories to such an offensive degree, as to have not only deteriorated all bottom-feeding fishes, but the striped bass as well have become so permeated by the offensive refuse as to be unfit for the table. This is a great damage, for there are many who made bass fishing near New York their only recreation."

To be precise, the Spirit of the Times story appeared on Sept. 28, 1878, but we thought fishermen might like to know ahead of time so they can get ready to celebrate the big day.

The recent closing of commercial fishing for all fin fishes except shad and large sturgeon in the Hudson estuary by the State of New York shows that the contaminants of concern (polychlorinated biphenyls, in this case) change with the times but the basic problems are always with us.

What follows is an attempt to assess the magnitude of the impacts of the contaminants that seem to present the greatest need for attention in the "Apex" region of the New York Bight. The effects of the contaminants may arise either from direct damage to seafood, making it unfit for human consumption (public-health considerations) or from reducing the production of aquatic organisms by affecting the life cycles of the complex assemblage of plants and animals in the Apex (ecological considerations).

The water-quality criteria for these contaminants are reviewed to provide a framework for the assessments. Current and future contaminant input estimates are used with observed values of the Apex flushing rates to approximate the concentration levels of contaminants that may occur in the Apex. If the guideline values of the water-quality criteria are being approached, new management evaluations and decisions may be necessary.

## 2. CONTAMINANTS FOR CONSIDERATION

In 1978 a panel of experts was convened under the auspices of the New York Bight Marine Ecosystems Analysis (MESA) Project of the National Oceanic and Atmospheric Administration (NOAA) to assess the contaminants that are, or are likely to be, the most serious problem in the New York Bight. The MESA Panel's review appears to be the most definitive statement yet of current information, and its evaluations are accepted in this discussion of the impact of potential contaminants on the Bight. The Panel considered primarily the potentials for long-term chronic impacts and did not fully address the possibility of acute problems resulting from toxic chemical spills (O'Connor and Stanford, 1979).

The MESA Panel was charged with identifying the most serious contaminants of the Bight, given three criteria or categories of seriousness.

- A. Major perceived threats that require continued study: Existing or projected future hazard to human health by any mechanism (ingestion, bodily contact, genetic modification of water-borne pathogens to resist antibiotics, etc.).
- B. Potentially significant threats for which data must be collected and evaluated: Existing or projected future hazard to marine species harvested by man (commercially, or for recreation).
- C. Substances not requiring priority attention on the basis of existing information, but which are or may become a hazard to other marine biota.

The Panel reviewed the priority pollutant lists prepared by various national and international groups and found that the U.S. EPA Toxic Substance List (U.S. EPA et al., 1976) was comprehensive. The materials

on this list were evaluated by the Panel with regard to the following attributes of each substance.

1. Toxicity
  - a. Human
  - b. Aquatic biota
2. Partitioning behavior (n/octanol/water), i.e., tendency for uptake in fatty tissue
3. Discharge estimates
4. Transport (dispersion-dilution)
5. Availability to biota
6. Transformation to nontoxic or to toxic forms
  - a. Physicochemical
  - b. Biological
7. Estimate of concentration factors for biota
  - a. From items 2 to 6
  - b. From field studies

Based on current information, the Panel's evaluation of the contaminants that might present serious problems produced the categories shown in A., B., and C. above. In considering the implications of current and possible future contaminant levels in the Apex area of the New York Bight, we focus on the materials in category A. To provide a framework for assessing the possible implications for these materials, the public-health aspects of toxic materials and the ecological effects are reviewed. Plutonium contamination possibilities are not reviewed because current information (Simpson et al., 1982) on the inputs indicates that current levels can be ascribed primarily to past weapons testing. Observed plutonium concentrations are well below the guideline number for concern.

### 3. GENERAL ASPECTS OF HUMAN HEALTH EFFECTS

The descriptions of the human health effects of selected toxic materials were prepared on the following assumptions. The toxic materials of concern will result in exposures of humans through ingestion of contaminated food organisms. Because of a number of factors, including large dilutions of discharges to ocean waters, there is essentially no chance of acute toxic effects. The descriptions therefore emphasize the effects observed due to chronic low-level ingestion exposures and provide the basic background data available and necessary for comparisons of dietary exposures, potential New York Bight dietary contributions, and regulatory or recommended total intake limits. The reviews given here are intended only as a first step in establishing the probable risks associated with the selected toxic materials; the fuller reviews cited may be consulted for a thorough grasp of the hazards and risks involved prior to regulatory or legislative actions.

#### 4. RATIONALE FOR ASSESSING IMPLICATIONS OF CONTAMINANT LEVELS

The processes involved in the contamination of the Apex region of the New York Bight (Figure 1) are shown diagrammatically in Figure 2. Four kinds of input processes need to be considered: riverine inputs through the Hudson estuary, direct inputs along the shoreline from outfalls serving coastal communities, atmospheric fallout and rainout, and discharges from barge disposal. With continuous discharge, the contaminant materials become distributed throughout the Apex and are either transported out of the region through physical flushing or retained in the bottom sediments and ultimately permanently deposited. Thus, the Apex inventory can be seen to be a balance between the rates of contaminant inputs and the losses to the open ocean and the sediments.

The quasi-state inventory determines the level of uptake of the contaminants by the biota with possible human health effects and also the ecological responses to the contaminant inputs. The current inventory levels and those that might develop if the inputs were changed may be viewed against the U.S. EPA water quality criteria to protect saltwater aquatic life. The most direct procedure for estimating whether or not the water quality criteria are exceeded is the analysis of samples from the area. However, for contaminants that have not been adequately sampled, a simple model comparing inputs with flushing rates provides some insight into whether or not the water quality criteria may be exceeded. Evaluation of the contamination of seafood via the mechanisms involved in the complex food webs in the region makes direct observation obligatory for confidence in such assessments.

The effects of contaminants from the several sources to the Apex (Figure 1) are moderated by the flushing of the Apex with open ocean waters. The Apex physical oceanography is characterized by normally moderate currents that show large variation in direction and strength over 3- to 10-day storm periods. The variations produce complex patterns of motion, but the resulting exchange of Apex waters for shelf waters derived from the mixture of river discharges with open ocean water is variable by a factor of 2 or 3. Using the observed volume of freshwater in the Apex and the time-averaged river flow, Ketchum *et al.* (1951) reported flushing times or time to replace the volume under consideration of 6 to 10.6 days, and the recent work of Han and Niedrauer (1981) showed flushing times of 3.9 to 12 days. The September-October period was found to be the time of slowest flushing; the value of 10 days taken here as being representative of those conditions provides an element of prudence, since usually the flushing rate can be expected to be faster than 10 days.

The volume of the Apex (Figure 1) is  $48 \text{ km}^3$ , so that  $4.8 \text{ km}^3$  of "new" water is transported into the area each day and an equal volume of Apex water is removed. Considering that a cubic meter of water weighs 1 metric ton, this flushing or renewal rate corresponds to  $4.8 \times 10^9$  tons of "new" water per day.

For contaminants retained in the water (either dissolved or associated with suspended particulate matter), the level of contamination is calculated from the ratio of the input rates to the flushing rate. For example, for a material that is being introduced into the Apex by the sum of the several

transport processes (Figure 2) at the rate of 4.8 tons/day, the Apex contamination may be estimated as

$$X = \frac{4.8 \text{ tons/day}}{4.8 \times 10^9 \text{ tons H}_2\text{O/day}} \approx 1 \text{ ppb.} \quad (1)$$

Since the guideline water quality criteria for the more toxic materials are 1 ppb (1 g/liter, neglecting the density) or less, an input of the order of 5 tons/day or less may seriously contaminate the Apex. Note that these maximal estimates of contamination levels do not represent increases over background concentrations for naturally occurring materials, such as heavy metals, but are maximal estimates of the actual concentrations for synthetic organics that may not be present in the ocean water source for the Apex region at significant concentrations. The flushing rate estimates are based on the inventory of fresh water in the Apex as compared with the rate of fresh water input from the Hudson-Raritan estuary, so that the possible recycling of contaminants by transport out of the Apex and then back into the Apex is taken inherently into account by the fresh water tracer technique because the fresh water inventory reflects these processes to the same extent.

## 5. IMPLICATIONS OF INDIVIDUAL CONSTITUENTS

### 5.1 Mercury

#### 5.1.1 Inputs

The total loadings of mercury to the Hudson-Raritan estuary have been estimated by Mueller *et al.* (1982) at about 0.08 tons/day. An additional 0.02 tons/day of mercury in sewage sludge are dumped directly in the Apex (Mueller *et al.*, 1976, adjusted for the 52% increase in annual volumes of sewage sludge dumped since the Mueller *et al.* estimate to 1980-81). Dredged materials from New York Harbor, dumped in the Apex, translocate unknown fractions of the mercury input to the Harbor. Somewhat independent estimates of the mercury in such dredged materials range from 0.13 tons/day (Mueller *et al.*, 1976) to 0.08 tons/day (Dayal *et al.*, 1981). Given the uncertainties in all of these estimates, they are reasonably consistent. The large disparity between dredged material inputs (Mueller *et al.*, 1976 cf Dayal *et al.*, 1981) is due to the use of different bases for estimating average mercury concentrations in dredged materials; the latter estimate has a broader basis and uses more recent data. Some of the mercury in the wastewater and stormwater runoff that must pass through the Hudson-Raritan Estuary to reach the Apex may instead become associated with particles and be deposited in the estuarine sediments. This mercury would contribute to the dredge spoil input to the Apex, but an unknown amount of mercury may be retained in the estuaries.

#### 5.1.2 Potential environmental levels.

The possible significance of mercury in the Apex can be conservatively viewed by assuming that the total estuarine input plus that in sewage sludge

is an upper limit estimate for the increase in mercury concentration in the waters of the Apex. Direct measurements of aqueous mercury concentrations have been below the detection limits of the methods used (E.G. and G., 1978b - <.02 ppb, Energy Resources Co., 1979 - <.1 ppb, Hydrosience, 1974 - < 2 ppb) except for four values between .03 and .05 ppb obtained at the acid waste dump grounds (E.G. and G., 1978a). The upper limit of increase above natural in the concentrations of mercury in water may be estimated simply as the ratio of the input to the flushing rate.

$$\frac{\text{Input}}{\text{Flushing Rate}} = \frac{0.1 \text{ tons/day}}{4.8 \times 10^9 \text{ tons H}_2\text{O/day}} = 0.02 \text{ ppb.}$$

Part of the Apex input of mercury probably passes rapidly to the sediments and would not contribute to contamination of the water column. The extent of this partitioning is unknown, and a cautious view may be taken by assuming that the total input contaminates the water as a theoretical or upper bound estimate. The purpose in making such an estimate is that, if the calculated upper bound value is substantially smaller than the guideline water quality criterion, the inference may be made that current input rates are not producing substantial ecological effects on organisms in the water column. Such "upper bound" estimates may be considered as "conservative" from a risk assessment point of view.

### 5.1.3 Ecological effects

The simple computation given above is not intended as a substitute for definitive field data; it is only a first-order assessment of maximal contamination level. Also, recognition should be given to the possibility that mercury or the other contaminants considered in this review may be having an effect on the benthic organisms that is not considered in the above computation, which applies to the bulk seawater phase. No substitute seems to exist for direct field and laboratory observations of the effects of sediment contamination on the benthic communities. Additive or synergistic effects of simultaneous contamination with several toxic materials may also be suspected, but no adequate basis exists for their assessment. Such effects must be considered as the assessment of Apex contamination continues.

The U.S. EPA (1980) marine water quality criterion for the protection of marine life from mercury is 0.025 ppb. This is based on the facts that (1) natural seawater mercury levels are in this general range, (2) levels 50 times higher have been shown to constitute a distinct threat to selected species of marine organisms, and (3) edible marine fish, like their freshwater counterparts, can bioaccumulate mercury by a factor in excess of 10,000 to 1 (i.e., if the criterion were set any higher the FDA guideline for edible fish, which is now 1 ppm, might be exceeded) (U.S. EPA, 1980). Sublethal effects have also been observed. For example, mercuric chloride "markedly retarded" to "severely disturbed" development of sea urchin eggs, Paracentrotus lividus, at concentrations ranging from 5 to 10 ppb (Soyer, 1963).

Because of their much lower solubility in water, mercurous salts ( $\text{Hg}^+$ ) are much less toxic than the mercuric forms ( $\text{Hg}^{++}$ ) (Stecher, 1968). Oceanic mercury is generally present as an anionic chloride complex which has

a less pronounced tendency to bind to particulate matter and settle out of solution than have mercury compounds found in freshwater (Wallace et al., 1971). However, the discovery that certain microorganisms can convert inorganic and organic forms of mercury to the highly toxic methyl or dimethyl mercury has made any form of mercury potentially hazardous to the environment (Jensen and Jernelov, 1969). Under favorable conditions inorganic mercury in sediments can be readily converted to methyl mercury (Bartlett and Craig, 1981) but the degree of methylation in water is less evident (EPA, 1980).

Bacteria may play an important role in the cycling of mercury in the estuarine environment, particularly at certain times of the year. For example, Colwell and Nelson (1975) present evidence that bacteria may contribute substantially to the mobilization and transformation of mercury from existing deposits in Chesapeake Bay, reaching a peak in the spring.

#### 5.1.4 Human health effects

Ingestion of mercury in seafood has resulted in at least two serious epidemics of mercury poisoning (D'Itri, 1972). These and other cases of elevated mercury burdens in humans consuming fresh or salt water organisms, have been caused by the presence of methylmercuric ion ( $\text{CH}_3\text{Hg}^+$ ) in the organisms. About 95% of this ion is absorbed in humans, while inorganic forms of mercury absorb to 7% and elemental mercury to 0.01% (Nordberg, 1976, p. 21).

Methyl mercury exposures may result in neurological effects weeks to months after an acute exposure (Nordberg, 1976). Effects of short-term exposures are partially reversible, unlike the irreversible effects observed in the Minimata epidemic from chronic exposures. Central nervous system (CNS) effects of increasing severity through blindness to death are produced by chronic exposures at relatively low methyl mercury intakes. These effects can occur in the fetus through placental transfer from the mother in utero (Nordberg, 1976, pp. 58-59).

The World Health Organization (WHO) has established a provisional tolerable weekly intake for adults of 0.3 mg total mercury, with a maximum of 0.2 mg mercury as a methyl mercury compound (WHO, 1972, 1976).

On the basis of the early data available, the FDA set an action limit of  $0.5\mu\text{g Hg/g}$  (wet weight) for fish, shellfish, etc., but this was related to  $1.0\mu\text{g Hg/g}$  as of June 1978 (U.S. FDA, 1978). An overall estimate of U.S. dietary intake by the FDA indicated average weekly levels of 5% of the WHO recommendation (U.S. FDA, 1977, p 10). Mercury in New York Bight organisms has not appeared to be a public health problem (O'Connor et al., 1983).

#### 5.1.5 Ecological and public health implications

As noted above, the U.S. EPA (1980) marine water quality criterion for mercury relative to marine life is 0.025 ppb. The upper-limit estimate of mercury concentration above natural values in the Apex, 0.02 ppb, has reached the EPA guideline criterion.



Earlier work on mercury in ocean water had indicated levels of 0.1 ppb as natural values. However, recent reports by Matsunga et al., (1979) and Fitzgerald and Lyons (1975) indicate that 0.01 to 0.005 ppb are more accurate values.

Consequently, if a large fraction of the estimated upper limit of input reaches the Apex, anthropogenic mercury contamination would represent a significant increase over natural environmental levels in the Apex. Further, a similar fraction of the upper-limit input, combined with background sea water mercury concentrations, would approach or even exceed the EPA criteria for protection of marine life.

The public health aspects of mercury contamination of seafood may be viewed in the framework of the FDA action limit of 1.0 mg/kg or ppm (wet wt.) mercury in fish and shellfish. Recent reports of heavy-metal levels in tissues of New York Bight fishes, molluscs, and crustaceans (Greig and Wenzloff, 1977; Greig et al., 1978; and Wenzloff et al., 1979) show that the mercury levels were below the FDA-established guideline for these seafood products.

As described above, the estimated present inputs of mercury to the Apex may be producing elevated concentrations in the waters and sediments of the area. Some insight into whether or not contamination of marine organisms with mercury is taking place may be gained from the data shown in Figures 3 to 6. The figures were prepared by the MESA Project Office from data on file at the Southeast and Northeast Fisheries Centers of the U.S. National Marine Fisheries Service (NMFS). The mercury concentration data for the spiny dogfish, Squalus acanthias (Figure 3) do not suggest identifiable effects of the mercury inputs to the Apex for this organism. Since the spiny dogfish is a far-ranging migrant (Tempelmann, 1954), the lack of measurable contamination may simply mean that the sample fishes were transients.

The mercury concentration data for the surf clam, Spisula solidissima (Figure 4) do not show a measurable increase in the mercury content of the Apex population compared with clams collected in other areas of the shelf. Wenzloff et al. (1979) also found that the mercury content of this species was below their analytical detection limit (0.05 ppm) and surf clams from the Apex region did not have measurable increases in mercury content.

In contrast, the mercury content of winter flounder, Pseudopleuronectes americanus (Figure 5) appears to be elevated in specimens collected from the Apex as compared with flounders from the shelf, both north and south of the Apex, although the mean mercury concentration for the Apex samples of 0.13 ppm is substantially below the FDA action limit of 1.0 ppm. Studies of the migration of winter flounder (reviewed by Klein-McPhee, 1978) have demonstrated that, aside from local onshore-offshore migration, the winter flounder is a relatively stationary fish. This species might be considered as a useful organism to monitor the Apex as the mercury input is increased or decreased in the future.

The mercury concentration data for the North American lobster, Homarus americanus (Figure 6) show regular differences between body meat, claw meat, and tail meat. The samples from the Apex region were apparently composites of flesh from all parts of the lobster muscles. The total tissue

concentrations of mercury in lobster, during winter or averaged over seasons, may not evidence higher concentrations in the Apex. However, the total tissue concentrations were 275% greater in the Apex than in lobsters from the outer shelf during summer (Sick, in preparation).

Present evidence concerning input rates and the levels of mercury in marine organisms from the Apex suggests that this contaminant is not causing public health damage at present. If large fractions of the mercury reaching the Hudson-Raritan Estuary are transported to the Apex, mercury could be causing or contributing to toxic effects upon biota in the water column. Although not assessed here, toxicant effects on benthic organisms would appear to be even more probable (O'Connor and Stanford, 1979).

## 5.2 Cadmium

### 5.2.1 Inputs

The major source of cadmium to the Apex and the Bight is the 2 metric tons/day introduced via ocean dumping (Mueller et al., 1976), with small secondary sources of 0.2 metric tons/day contributed by wastewater and gaged and ungaged stormwater runoff, and from the atmosphere (Mueller et al., 1976, 1982). To what extent the cadmium introduced with the dredge spoils is transferred to the water or is available to benthic organisms is uncertain. However, cadmium and manganese were found to be continuously released to the overlying water under oxidizing conditions in laboratory simulations of the post-depositional release of metals from dredged materials (Chen et al., 1976; Blom et al., 1976). More direct evidence also indicates that a large fraction of the cadmium in dumped dredged materials is dispersed beyond the dredged materials' dumpsite (Dayal et al., 1981).

### 5.2.2 Potential environmental levels

The possible significance of current inputs of cadmium to the Apex can be viewed conservatively by assuming that the total input is available to produce an upper limit estimate for the increase in the cadmium concentration in the waters of the Apex. The ratio of the input to the flushing rate provides an estimate of the increase in cadmium concentration above the natural background levels.

$$\frac{\text{Input}}{\text{Flushing rate}} = \frac{2.2 \text{ metric tons/day}}{4.8 \times 10^9 \text{ tons H}_2\text{O/day}} = 0.5 \text{ ppb.}$$

This estimate of the possible cadmium concentration increase in the Apex region may be compared with the cadmium measurements reported by Segar and Cantillo (1976). The background or natural cadmium concentration for the seawater source for the Apex appears to be 0.1 ppb, based on their results for bottom-water samples from stations east of the Apex area. This cadmium value is similar to that observed by Windom and Smith (1972) in continental shelf waters of the southeastern United States. The background value of 0.1 ppb plus the contaminant value of 0.5 ppb sum to 0.6 ppb. Segar and Cantillo found an average cadmium concentration of 0.42 ppb in April 1974, and 0.82

ppb for May through November 1974 in Apex waters. The close correspondence between the estimated input and the water-column observations suggests that much of the cadmium input is mobilized into the water column, in agreement with the laboratory results of Blom et al. (1976) and Chen et al. (1976) and the field observations of Dayal et al., (1981).

### 5.2.3 Ecological effects

The EPA criterion value of 4.5 ppb for marine life is based on bioassays and bioconcentration factors (EPA, 1980a). The American Fisheries Society has suggested that the EPA marine criterion is both too high and too arbitrary and has recommended an alternative criterion of 3 ppb (Thurston et al., 1978).

On the basis of short-term bioassays, the most sensitive marine invertebrates include the amphipod, Allorchestes compressa (120-hr LC<sub>50</sub> [static] = 0.2 to 0.4 ppm, EPA, 1977); the polychaete worm, Capitella capitata (96-hr LC<sub>50</sub> [static] = 0.22 ppm, Reish et al., 1976); the grass shrimp, Palaemonetes vulgaris (96-hr LC<sub>50</sub> [static] = 0.42 ppm, Eisler, 1971); the hermit crab, Pogarus longicarpus (96-hr LC<sub>50</sub> [static] = 0.32 ppm, Eisler, 1971); and the sand shrimp, Crangon septemspinosa (96-hr LC<sub>50</sub> [static] = 0.32 ppm). Two life cycle tests on the shrimp, Mysidopsis bahia, resulted in even lower chronic toxicities, at .006 and .008 ppm Cd (Nimmo et al., 1977 and U.S.EPA 1980 in U.S.EPA, 1980a).

Bioassays on marine fishes indicate that the species tested are comparable with or less sensitive than the non-salmonid freshwater species tested. The most sensitive saltwater fish evaluated to date is the mummichog, Fundulus heteroclitus (96-hr LC<sub>50</sub> [static] = 1.3 ppm at low dissolved oxygen and 2.7 ppm at high dissolved oxygen, Voyer et al., 1976).

To date, long-term toxicity data for cadmium using sensitive marine species have not been reported in the literature.

Many marine invertebrates can accumulate cadmium without apparent signs of toxicity (George et al., 1977). However, cadmium can cause abnormal metabolism in lobsters (Thurberg et al., 1977), loss of metabolic flexibility in winter flounder (Gould, 1977) and other disorders in several species (U.S. EPA, 1980a).

Cadmium reaching aquatic systems can be transported from water to aquatic organisms or to sediments. It can be transferred from aquatic prey to either aquatic or terrestrial predators. Presumably, it can also be transferred from sediments to aquatic organisms.

Sorption to sediments (EPA, 1980a) is assumed to be the principal path by which cadmium compounds are removed from the water. Cadmium is strongly sorbed to clays, muds, humic and organic materials, and the hydrous oxides of iron and manganese (Watson, 1973). It can also be precipitated as the hydroxide, carbonate, or sulfide (Baes, 1973).

It is not expected that cadmium will be methylated in the environment (Wood, 1974).

Recent evidence indicates that it is generally the "free" or ionic forms of cadmium that are toxic to aquatic life (Engel, Sunda, and Fowler, 1981). A decrease in toxicity may occur with increased salinity because of complexation (Engel et al., 1981), and as a result of binding by metal-complexing proteins such as metallothioneins, which are apparently found in mammals, teleosts, and bivalves (Overnall et al., 1977; Fowler et al., 1981).

#### 5.2.4 Human health effects

Ingestion of cadmium results in a slow accumulation in the kidneys of humans at normal exposures (Friberg, 1974, p. 55; Nordberg, 1976, p. 24). About 3 to 10% of the ingested cadmium is absorbed, and the concentration in kidney cortex is about 50  $\mu\text{g/g}$  wet weight for 50-year-olds in the U.S. (Friberg, 1974, p. 55 et seq.; Nordberg, 1976, p. 20 et seq.).

The first organ to show increased cadmium content due to exposure is the liver. Induction of metallothionein synthesis occurs in both liver and kidney, and the cadmium metallothionein transfers to and is stored in the kidney. The concentration eventually increases to a point where kidney damage occurs and both cadmium and proteins are excreted in urine.

Serious kidney damage with excretion of cadmium is believed to occur when the concentration reaches 200  $\mu\text{g/g}$  wet weight in the kidney cortex (Nordberg, 1976, pp. 64-66). Chronic exposure in Japan through ingestion of rice contaminated by cadmium in irrigation water produced a complex of unusual symptoms related to irregular calcium metabolism (osteomalacia) (Itai-Itai disease). This occurred in the older segment of the population and may have involved nutritional deficiency interactions (Friberg et al., 1974, pp. 137, 155, 192 et seq.).

The WHO has established a provisional tolerable weekly intake of cadmium at 400 to 500  $\mu\text{g/wk}$  for adults (FAO-WHO, 1972), based on estimates of lifetime intakes of 250 to 440  $\mu\text{g Cd/day}$  required to reach the critical value of 200  $\mu\text{g Cd/g}$  in the kidney cortex (Nordberg, 1976, pp. 66, 67; Kjellstrom and Nordberg, 1978). The FDA has calculated the average U.S. daily intake as 30 to 35  $\mu\text{g Cd/day}$  or 210 to 240  $\mu\text{g/wk}$ , about one-half of the WHO value (US FDA, 1977, p. 10). However, caution is advised, as more representative sampling could result in a higher estimate. There are no long-term trends in the U.S. data for food through 1974, and general levels of  $< 0.5 \mu\text{g Cd/g}$  wet weight provide some margin of safety (U.S. FDA, 1978).

#### 5.2.5 Ecological and public health implications

As noted in section 3, the EPA criterion value for cadmium in seawater is 4.5 ppb (EPA, 1980a) but the American Fisheries Society (Thurston et al., 1978) has suggested a lower value of 3 ppb. While the observations of Segar and Cantillo and the cadmium concentration estimate based on the current probable input rates indicate clearly identifiable cadmium contamination of the Apex, the values are less than the guidelines. Present knowledge suggests that ecological damage due to cadmium exposure of Apex biota is not large and that damage, if it is occurring, would be difficult to observe,

with the possible exception of the dredge-spoil dump site, where elevated concentrations of cadmium in the sediment pore waters may be anticipated. However, it should be noted that Dayal, et al. (1981) found the cadmium concentration in pore waters of the dredged material deposit to be < 2 ppb.

The United States has not yet established guidelines for maximal allowable concentrations of cadmium in the edible portions of fin fish and shellfish. The National Health and Medical Research Council of Australia has set 2.0 ppm wet weight as the limit for cadmium in seafoods (Mackay et al., 1975). As noted in section 4, a value of 0.5 ppm appears to be conservative and provides a margin of safety. Limited sampling of the Apex region has shown that hard clams, oysters, scallops, lobsters, rock crabs, and ocean quahogs often exceed the 0.5 ppm value (Wenzloff et al., 1979; R. A. Greig, unpublished data). The Apex region does not appear to be a commercial source for oysters. However, this region is the source of seawater for the Hudson estuary and Raritan Bay system. Oysters and other shellfish strongly accumulate cadmium. Cadmium in the Apex may contribute to shellfish concentrations in the estuary. This possibility should be examined through the sampling of oysters and other organisms in these estuaries with consideration of all sources.

Data from the National Marine Fisheries Series (NMFS) files for organisms sampled in the Apex region and in areas to the north and south are summarized in Figures 7 to 9. The samples of spiny dogfish, Squalus acanthias, do not show elevated levels of cadmium for the animals collected in the Apex region (Figure 7), which may reflect the short residence time of this migratory species in the area.

The NMFS data for the surf clam, Spisula solidissima, plotted in Figure 8, do not indicate increased cadmium concentrations in this organism in samples from the Apex. Wenzloff et al. (1979) measured cadmium concentrations in surf clams and ocean quahogs, Artica islandica, and found that the cadmium concentration in quahogs was threefold greater than in surf clams. Their data show about a 30% lower cadmium concentration for quahogs collected off the Maryland-Virginia shoreline than in those collected off New York. The cadmium contamination from the Apex region does not appear to be strongly reflected in the body tissues of the surf clam or quahog.

The data for the winter flounder, Pseudopleuronectes americanus, shown in Figure 9, do not display elevated cadmium concentrations in the muscle tissue of specimens collected in the Apex as compared with those collected outside it. Hazen and Kneip (1980) have shown that both laboratory and field exposures resulted in elevated cadmium concentrations in the liver and kidney of gold fish. Fish and crabs collected in the field also showed elevated cadmium concentrations in organs, so that the analysis of flounder muscle to evaluate potential impacts on human diets does not provide the data needed to assess the effects of cadmium exposure on the flounder's liver and kidney metabolism. Gould (1977) found changes in the enzyme activity in flounder kidney tissue following exposure to 5 to 10 ppb cadmium.

In the framework of existing guidelines, cadmium contamination in the Apex region does not appear to be producing severe ecological or public health effects. Local effects on bottom-dwelling organisms at the dredge-material disposal site are not ruled out by existing data and several

shellfish species used as seafood are contaminated at levels that approach or exceed the  $\mu 0.5$  g Cd/g wet weight - a concentration that appears to provide a margin of safety.

### 5.3 Lead

#### 5.3.1 Inputs

The potential inputs of lead to the Apex region total 12.7 metric tons/day, with the predominant sources being dredge spoil, wastewater, and runoff (Mueller et al., 1976). Estimates of loadings to the Hudson-Raritan Estuary made more recently indicate that total potential lead inputs to the Apex have declined to about 9.7 metric tons/day (Mueller et al., 1982). Atmospheric fallout and rainout are also direct inputs of lead to the Apex region that may be highly available as water contaminants. The atmospheric-input estimate of 1.2 metric tons/day to the Bight cannot be simply scaled to the Apex region, which constitutes roughly 5% of the Bight area, in view of the source proximity. Based upon the above figures, the estimate of 12.7 metric tons/day is used to estimate potential environmental levels.

#### 5.3.2 Potential environmental levels

These potential inputs of lead may be viewed cautiously by assuming that the total input is available as an environmental contaminant. Recent studies of the behavior of lead in coastal systems (Benninger et al., 1975; Benninger, 1978) suggest that lead is quite strongly associated with suspended particulates and tends to be transferred to the sediments, with little cycling back into the overlying water. These observations are compatible with laboratory results (Chen et al., 1976; Blom et al., 1976) that showed very little lead was released to the overlying seawater from sediments under oxidizing conditions. Therefore, the assumption that the total lead input might be available as a contaminant is not realistic, but does provide an upper-limit estimate of the potential contamination.

$$\frac{\text{Input}}{\text{Flushing rate}} = \frac{12.7 \text{ metric tons/day}}{4.8 \times 10^9 \text{ tons H}_2\text{O/day}} = 2.6 \text{ ppb}$$

Direct observations of existing lead contamination levels in the Apex region are not extensive. Alexander et al. (1978) reported the results of analyses for samples collected at 5 stations along a transect from Sandy Hook, New Jersey, to Rockaway Point, New York, and found the average particulate lead concentration was < 0.5 ppb and the average soluble lead concentration was < 3 ppb. Piotrowicz et al. (1972) collected 5 samples in the Apex area in May 1971 and found particulate lead concentrations of 0.3 to 0.7 ppb in near surface waters and 1 to 3 ppb in the surface microlayer.

#### 5.3.3 Ecological effects

While the EPA does not specify a saltwater criterion for lead, the report concludes that "The available data for total recoverable lead indicate that

acute and chronic toxicity to saltwater aquatic life occur at concentrations as low as 668 and 25  $\mu\text{g}/\text{l}$  [ppb] respectively, and would occur at lower concentrations among species that are more sensitive than those tested" (U.S. EPA, 1980b, p. B-12). The American Fisheries Society (Thurston *et al.*, 1978) has suggested that the criterion established for freshwater (4 ppb) should be used for marine water. The rationale for such a proposal is the similarity in sensitivity between freshwater and marine species that were tested, but it was noted that there are few supporting data for this assumption. Presumably, the toxicity of lead in marine waters might be reduced in a manner analogous to that observed in hard freshwater, since 99% of the inorganic lead in seawater occurs in the form of carbonate and chloride complexes (Millero, 1977).

Present knowledge is inadequate for assessing the possible ecological effects of lead contamination of marine water. However, lead concentrations of the order of 1 ppb should be scrutinized for effects on the biota.

#### 5.3.4 Human health effects

The ingestion of lead in the diet is a major exposure route for humans. About 5 to 15% of the ingested lead is absorbed by adults (Nordberg, 1976, p. 21), but a larger fraction, possibly as much as 50%, by children (Alexander, 1973 as given in Nordberg, 1976, p. 21). However, the intake of lead in children has not been studied extensively (U.S. EPA, 1980b). Exposures are reflected by lead concentrations in blood and soft tissues, and lead accumulates in bone, with long bones showing continuing increases in lead concentration to about 50 to 60 years of age (Nordberg, 1976, p. 25).

Exposures to lead result in a complex of gastrointestinal, hematological, and nervous system effects. Anorexia, lethargy and anemia can progress in children to coma, convulsions, and death (Nordberg, 1976, p. 47 *et seq.*). Severe central nervous system (CNS) disorders are associated with blood lead levels of 60  $\mu\text{g}/100\text{ g}$  of blood (60  $\mu\text{g}\%$  or 0.6  $\mu\text{g}/\text{g}$ ) (Mahaffey, 1977). Behavioral problems and learning deficiencies in children may be associated with lower blood lead concentrations; however, these factors are subjects of ongoing research. Adults are not as susceptible to CNS effects, but gastrointestinal and peripheral neurological symptoms occur at 60 to 80  $\mu\text{g}/100\text{g}$  (Nordberg, 1976, p. 77).

The provisional tolerable weekly intake of lead has been set at 3 mg/wk by the WHO (FAO-WHO, 1972). Estimates by the U.S. FDA (1977, p. 9) indicate the diet of a teenage male provides an upper estimate of 200 to 250  $\mu\text{g}/\text{day}$  in adult foods, or <1.7 mg/wk. Estimates of acceptable exposures have been reviewed by Mahaffey (1977a). She has recommended control of children's exposures at maxima from all sources to 100  $\mu\text{g}$  Pb/day for the 0 to 6 mo. age group, 150  $\mu\text{g}/\text{day}$  for 6 mo. to 2 yr, 200  $\mu\text{g}/\text{day}$  for 2 to 3 yr, and 300  $\mu\text{g}/\text{day}$  for children over 3 yr. These limits are not easily attained, as the diet alone may account for exposures equal to those recommended for the youngest children.

Because of continuing efforts by the FDA and the food industry, both infant foods and canned adult foods have shown decreasing lead contents recently. The development of the catalytic muffler, which

requires no lead gasoline, and subsequent EPA regulation of lead concentrations in ambient air might result in negligible lead emission by autos by the mid-1980's. This should result in a very significant decrease in the input of lead to the New York Bight.

### 5.3.5 Ecological and public health implications

Since the data for lead toxicity to marine organisms are so meager water quality criteria cannot be estimated at present. No straightforward way exists of assessing the ecological significance of the current input rate of lead to the Apex area. However, in view of the estimates that ecological effects may occur at the 4 ppb or so level and that the input rate could produce contamination of 2 to 3 ppb, the possibility of ecological effects needs future attention.

With regard to public health effects, guidelines or action limits for lead in fish and shellfish have not been established in the United States. The National Health and Medical Research Council of Australia has set 2 ppm wet weight as the limit for lead in seafood (Mackay et al., 1975). Shellfish are particularly effective concentrators of metals, and, as summarized by Wenzloff et al. (1979), lead concentration factors of 4,000 to 6,000 have been observed for 6 species. The concentration factor is the ratio of the lead concentration in the organism to that in the ambient water. If the seafood concentration were to be limited to the Australian guideline of 2  $\mu$ g/kg or ppm, the corresponding water quality criterion would be 0.4 ppb with shellfish as the consideration. The computed potential lead contamination of 2.6 ppb and the observations of Piotrowicz et al. (1972) of particulate lead values of 0.3 to 0.7 ppb suggest that lead contamination may be near levels that require action to protect the public health. In fact, Wenzloff et al. (1978) found lead concentrations of 1.0 to 2.2 ppm in ocean quahogs, Artica islandica, collected in the Apex region.

As shown in Figures 10 and 12, the observed lead concentration data from the NMFS collections for the spiny dogfish, Squalus acanthias, and the winter flounder, Pseudopleuronectes americanus, do not indicate substantially different values for lead content in samples from the Apex region as compared with samples from outside this area. As indicated earlier, the migratory behavior of these species compromises their use as indicators of toxicant concentrations in particular areas. The NMFS data for lead in the surf clam, Spisula solidissima, are near the limit of the analytical method used for the lead analyses and do not show a clear pattern (Figure 11).

While lead contamination of the Apex region is observable, available information is not adequate to assess whether the current input rates are causing deleterious effects on the Apex biota. Shellfish lead concentrations need further documentation, and the significance of this lead input to the human diet should be evaluated.



## 5.4 Polychlorinated Biphenyls

### 5.4.1 Inputs

Comprehensive estimates of the potential inputs of chlorinated pesticides do not seem to be available at this time.

The potential load of polychlorinated biphenyls, PCBs, to the Apex region is discussed by Mueller et al. (1976 and 1982). The largest potential direct source is the estimated 9 kg/day from the Hudson River, largely derived from past discharges at the General Electric Co. plants at Hudson Falls and Fort Edwards. The second largest potential source had been thought to be the roughly 5 kg/day adventitious discharge from the sewage treatment plants. However, O'Connor et al. (1983) reported that the dredged materials and sewage sludge from the Hudson estuary are now a contaminant source at 15 kg/day. The dredged material source may include material from the Hudson River and the sewage treatment plants, but for the sake of an upper-limit estimate the sum of all three sources may be considered; this is 32 kg/day, or 0.03 metric tons/day. The true value is certainly smaller, and 0.02 metric tons/day or so may be reasonable.

### 5.4.2 Potential environmental levels

These estimates of potential PCB inputs may be used with the flushing rate of the Apex region to provide an appraisal of the contamination level.

$$\frac{\text{Input}}{\text{Flushing rate}} = \frac{0.03 \text{ metric tons/day}}{4.8 \times 10^9 \text{ tons H}_2\text{O/day}} = 0.006 \text{ ppb.}$$

The PCBs have a strong affinity for particulate matter. Harvey and Steinhauer (1976) observed that suspended matter in equilibrium with seawater contained 200,000 times more PCB than an equal mass of seawater. PCB concentrations have been measured in several sediment samples collected from the Apex region (Boehm, 1983). The highest concentrations (1,000 ppb) were found in the sewage sludge disposal area. On the basis of the sediment-water distribution coefficient of 200,000, sediment pore waters may have concentrations of 0.005 ppb, to which burrowing organisms would be exposed. West and Hatcher (1980) also measured the PCB concentration in a single water sample from the sludge dumping area following a 48-hr moratorium on sludge dumping and found 0.02 ppb dissolved and 0.006 ppb particulate concentrations. Quite recently, Boehm (1983) reported analytical results for PCB content of particulates collected on a cruise in August 1980. Much lower values (0.0003 ppb) for PCB were found at 12 stations in the Apex region than those reported by West and Hatcher. The difference might be ascribed to either (1) differences in filtration and analytical methods, (2) collection of samples during a different season of the year, or (3) a decrease in the input rate during the time between West and Hatcher's sample collection and that of Boehm. The obvious difficulty in deciding between these possibilities should highlight the need for comprehensive study of the PCB inventory and dynamics in the Hudson estuary and the Apex region that quantitatively takes into account the seasonal and annual variations in the dynamics. This would not be a modest undertaking.

### 5.4.3 Ecological effects

The EPA marine water quality criterion for PCBs is 0.024 ppb (U.S. EPA, 1980d), based primarily on an accumulation factor of 100,000 in the oyster, Crassostrea virginica, and a desire to protect the human consumer. Additional considerations were the toxicity of Aroclor 1254 to sheepshead minnow fry, Cyprinodon variegatus, at 0.1 ppb, and evidence that a food tissue level of 0.5 ppm (0.1 times the FDA level for human consumption) is necessary to protect carnivorous animals.

The acute and chronic toxic effects of PCBs have been determined on a number of marine organisms. For example, PCBs interfere with photosynthetic mechanisms in certain species of phytoplankton at concentrations < 0.1 ppb (Fisher, 1975; Luard, 1973; Keil et al., 1971; Fisher and Wurster, 1973; Mosser et al., 1972a, b; O'Connors et al., 1978; Fisher et al., 1973; Mosser et al., 1974; Moore and Harriss, 1972; Fisher et al., 1974). The toxic effects observed were growth inhibition and a reduction in the rate of carbon fixation. Exposure of the marine diatom Cylindrotheca closterium to Aroclor 1242 yielded results which suggest that the primary effect of PCBs on phytoplankton is to inhibit cell growth and division rather than photosynthesis per se (Keil et al., 1971).

Studies of PCB effects on temperature-stressed marine diatoms illustrate the general effect of PCBs in enhancing environmental stress on sensitive species (Fisher and Wurster, 1973). This may have an important bearing on a species' ability to compete successfully for resources. This is further evidenced by a number of studies of PCB effects on mixed phytoplankton populations. These studies demonstrate that low levels of PCBs can cause changes in community structure in the environment (Mosser et al., 1972a; Fisher et al., 1974). Indeed, PCBs may be more active under mixed than individual culture conditions (Moore and Harriss, 1972). The experiments by O'Connors et al. (1978) show that "Polychlorinated biphenyls at concentrations of 1 to 10 micrograms per liter [ppb] reduced phytoplankton biomass and size in natural estuarine phytoplankton communities grown within dialysis bags in situ in an estuarine marsh. In polychlorinated biphenyls-contaminated waters, these changes could increase the number of trophic levels and divert the flow of biomass from harvestable fish to jellyfish and other gelatinous predators." (p. 178).

PCBs are acutely toxic to estuarine invertebrates in the range of several parts per billion (Hansen, 1976; Duke et al., 1970; Hansen et al., 1974). Even lower levels were toxic in laboratory bioassays lasting longer than two weeks. For example, a 15-day exposure (flowing water) of commercially valuable penaeid (pink) shrimp, Penaeus duorarum, to 0.94 ppb Aroclor 1254 caused 51% mortality as compared with 12% for controls. Comparable results were obtained for the grass shrimp, Palaemonetes pugio (Nimmo et al., 1974).

PCBs appear to increase the sensitivity of exposed shrimp to salinity stress (Nimmo and Bahner, 1974) and their susceptibility to viral disease (Couch and Nimmo, 1974; Friend and Trainer, 1970). Aroclor 1254 decreases the growth rate of oysters, C. virginica, at concentrations as low as 4 ppb (Lowe et al., 1972).

PCBs can have marked effects on species' composition and biological diversity at levels much lower than those required to demonstrate effects with single species. For example, 0.1 ppb Aroclor 1254 reduced the numbers of amphipods, bryozoans, crabs, and molluscs, and increased the number of tunicates that developed from planktonic larvae in seawater flowing through laboratory aquaria (Hansen, 1974).

Among estuarine and marine fish, the longnose killifish, Fundulus similis, is the most sensitive species tested; it is killed by prolonged exposure to 1 ppb Aroclor 1254 (Nimmo et al., 1975).

Fertilized eggs from adult sheepshead minnows, Cyprinodon variegatus, exposed to 0.1 ppb Aroclor 1254 for 4 weeks before being placed in PCB-free water, yielded fry whose survival was diminished (Hansen et al., 1974; Schimmel et al., 1974).

PCBs are known to increase the activity of liver enzymes that degrade steroids, including sex hormones, and to affect reproductive processes (Risebrough et al., 1968; Wasserman et al., 1970). It is not clear whether these effects are due to PCBs or to the chlorinated dibenzofurans ("PCDFs") - often present as contaminants - which are highly toxic (Schultz, 1968; Varrett, 1970).

PCBs have a long life in the environment. In particular, the more highly chlorinated compounds are resistant to metabolic breakdown and persist in water (Oloffs et al., 1972, 1973), sediments (Duke et al., 1970; Nimmo et al., 1975), and in biological tissues (Hansen et al., 1971; Dahlgren et al., 1972; Bush et al., 1974; Bailey and Bunyon, 1972; Burse et al., 1974; Kuratsune et al., 1976), often for periods of years.

PCBs are mobile in the environment and may be transported in solution, by motion of suspended sediments, as vapors, on airborne particulates, and in the tissues of mobile animals (Young et al., 1976; Tucker et al., 1975; Bidleman and Olney, 1974; Harvey and Steinhauer, 1974; Sodergren, 1972; Nisbet and Sarofim, 1972; Veith and Lee, 1971).

Within bodies of water, PCB transport appears to be controlled by the presence and transport of sediments (Munson et al., 1976; Young et al., 1976) and is usually found at much higher concentrations in sediments than in water in contact with them (Young et al., 1976; Crump-Weisner et al., 1974; Dennis, 1976).

Bioaccumulation factors reported in oysters, C. virginica, ranged up to 165,000 (Parrish, 1974). And there is evidence of food chain transfer of PCBs from sediment to large animals (Duke, 1974; Young et al., 1976; Young and McDermott-Ehrlich, 1976). The bioaccumulation factor for the adult fathead minnow, Pimephales promelas, is about 120,000 and 270,000 for Aroclor 1248 and Aroclor 1260, respectively.

Studies of PCB behavior in natural ecosystems suggest that bioaccumulation and biomagnification in wild animals may be much greater than that measured under controlled conditions in the laboratory. For example, Veith and Lee (1971) showed the importance of sediments in enhancing bioaccumulation in fish from the Hudson River, New York. PCB levels in wild

fish were 100 times greater than those in test fish kept in river water isolated from sediments. (See also Young et al., 1975, 1976; McDermott et al., 1975; Coulter and Risebrough, 1973; Risebrough et al., 1968; Delong et al., 1973). Tissue residue levels in nature indicate that predators may bioaccumulate PCBs to factors of millions of times higher than environmental levels.

The National Academy of Sciences (1973) has recommended, for the protection of fish-eating birds and mammals, that PCB concentrations in a homogenate of 25 or more whole fish of any species that is consumed by fish-eating birds and mammals (within the same size range as the fish consumed by any bird or mammal), be no greater than 0.5 mg/kg (or ppm) of the wet weight.

#### 5.4.4 Human health effects

The continual exposure of humans to low concentrations of water-insoluble chlorinated compounds (such as PCBs) results in accumulations in fat. The compounds are readily absorbed from the digestive tract and the less chlorinated biphenyl compounds are metabolized, while the more chlorinated (penta to decachloro) compounds are stored in adipose tissue (NIOSH, 1977, pp. 28-9).

Few published data exist for acute human toxicity to PCBs, but for Aroclor 1221 one value of 500 mg/kg has been reported (NIOSH, 1977a). Halogenated hydrocarbon materials generally cause effects on the liver and kidneys. These may include enlargement, jaundice, development of nodules, enzyme induction, and, in animal experiments, malignant tumors. Central nervous systems effects are critical in humans from many pesticides (Vettorazzi, 1975). Reproductive effects have been observed with PCBs in many animals and in humans. The PCBs cross the placenta and can cause slowed fetal growth, fetotoxicity, and resorption. Toxic effects from exposure to mother's milk have been observed in infants (NIOSH, 1977, p. 33 et seq.).

Contamination of rice oil resulted in PCB exposure to a large population in Japan. The effects described above were observed in a number of individuals. In this case it has been suggested that polychlorinated dibenzofuran (PCDF) contaminants may have been responsible. The PCDFs are present in PCBs to some extent in all samples analyzed (NIOSH, 1977, p.24). Other compounds present in commercial products include chlorinated benzenes, naphthalenes, and terphenyls. Thus, effects on humans due to environmental or occupational exposures cannot be attributed to a single compound or even a given class of compounds.

#### 5.4.5 Ecological and public health implications

As noted in section 3, the EPA marine-water quality criterion for PCB is 0.001 ppb. While the estimated upper limit of contamination is in excess of this guideline, the average measured concentrations approach the guideline. The probability that ecological damage is occurring in the Apex region cannot be assessed unequivocally with the present limited information, but clearly these possibilities need continued attention.

## 5.5 Aromatic Hydrocarbons

### 5.5.1 Inputs

While a number of aromatic hydrocarbon compounds have been implicated as major contributors to the toxicity of petroleum and refined products, most of the experimental or test observations of the responses of aquatic organisms to these contaminants have been expressed in terms of exposure to various concentrations of "total oil" rather than of individual compounds. Since petroleum products from different sources have different chemical compositions, different degrees of toxicity can be expected and, ideally, inputs to aquatic environments would be expressed in terms of the amounts of the individual toxic compounds. However, in the absence of the desired information, the total input of petroleum products will be considered as a first approximation of the potential environmental contamination of the Apex region.

The total load of oil and grease to the New York Bight region is ~ 350 metric tons/day (Mueller et al., 1982). A recent report of the National Academy of Sciences (1975) suggested that the petroleum hydrocarbon content of oil and grease wastes is in the range from 50 to 70% and averages 60%. Therefore, the potential input of petroleum hydrocarbons to the Apex region would approximate 210 metric tons/day. An independent estimate of 92 metric tons/day of petroleum hydrocarbon loadings to the Estuary was made by Connell, (1982).

### 5.5.2 Potential environmental levels

The potential petroleum hydrocarbon contamination of the Apex region waters can be approximated in terms of the ratio of the input rate to the flushing rate of the Apex region.

$$\frac{\text{Input}}{\text{Flushing rate}} = \frac{210 \text{ metric tons/day}}{4.8 \times 10^9 \text{ tons H}_2\text{O day}} = 44 \text{ ppb}$$

This computed value may be considered an upper limit. The independent estimate of 92 metric tons of petroleum hydrocarbons per year to the region (Connell, 1982) yields a maximum estimate of 19 ppb in the Apex.

No reports exist of systematic surveys of petroleum hydrocarbons in waters of the Apex region. Searl et al. (1977) reported observations for stations in the Ambrose Channel area as averaging 35 ppb in the water during March 1975 and 18 ppb in June 1975. These observations are not incompatible with the range of 19-44 ppb computed from the estimated inputs, since some of the petroleum compounds may be expected to be transferred to the atmosphere and microbiological metabolism would further reduce the inventory. MacLeod et al. (1981) reported that individual polyaromatic hydrocarbon (PAH) compounds were below their detection limits in 10 samples.

### 5.5.3 Ecological effects

Because the terms "petroleum hydrocarbons" and "oil and grease" are not definitive chemical categories but include thousands of organic compounds with varying physical, chemical, and toxicological properties, the EPA (1980c), has not set specific numerical criteria for these compounds. Instead, the EPA has specified two qualitative and one semiquantitative criteria for aquatic life. The semiquantitative criterion of 1/100 of the lowest continuous-flow 96-hr LC<sub>50</sub> to several important marine species, each having a demonstrated high susceptibility to oils and petrochemicals, is based on the persistence, toxicity, and bioaccumulation of many petroleum products. There are no criteria for levels in the sediments, where the real or potential problem is most likely to occur.

It is virtually impossible to discuss the ecological effects of aromatic hydrocarbons without including the general term "petroleum hydrocarbons," since much of the experimental work has been done on whole oils and most of the PAHs that enter the environment do so as components of oils. Crude oil is generally less toxic to marine organisms than refined products. The acute toxicity of a petroleum compound is directly correlated with its overall content of soluble aromatic derivatives (Neff et al., 1976). The monoaromatics are least toxic, and toxicity increases with increasing molecular size until the 4- and 5-ring compounds are reached. Aromatic hydrocarbons of 4 or more rings are less acutely toxic, possibly because of their large molecular size and low aqueous solubility. The toxicity of di- and tri-aromatic compounds appears to increase with alkylation.

PAHs enter the environment as components of oil, but in the New York Bight a significant input from combustion sources via aeolian and remobilized aeolian sources is apparent (Farrington and Tripp, 1977). The most toxic hydrocarbon evaluated to date is 1-methylphenanthrene (Anderson, 1977). Naphthalene and its alkyl derivatives are toxic to marine organisms in the 0.1 to 1.0 ppm range.

Considerable laboratory experimental work has been conducted to determine the effects on marine organisms of oils and some individual components such as naphthalenes. In addition to acute toxicity, workers have documented inhibition or delay in phytoplankton cell division (Mironov, 1970), suppression and stimulation of photosynthesis of phytoplankton (Gordon and Prouse, 1973), developmental abnormalities in Black Sea turbot (Mironov, 1970), and disruption of feeding and phototactic behavior in plaice larvae (Wilson, 1970). Reduction in chemotactic reception by snails was reported by Jacobson and Boylan (1973), and Bellan et al. (1972) found decreased fecundity in the polychaete Capitella capitata. Many of the experiments involved unrealistically high concentrations (>1 ppm) of hydrocarbons in the water column, but some of the effects have been demonstrated at levels of 10 to 100 ppb, which is more in line with the concentrations found in contaminated harbors.

There is a considerable amount of literature documenting the effects of oil spills on marine life. In very high concentrations, total mortality of invertebrate populations may occur, as was seen at West Falmouth (Michael et al., 1975). The aromatic fraction persisted for at least six years in the sediments; this caused structural changes in the benthic communities.

Concentrations of 20 to 50 ppm total hydrocarbons can induce structural changes in the benthic communities (Michael et al., 1975). Oils of any kind can cause drowning of seabirds resulting from loss of buoyancy or death by exposure due to loss of insulation capacity of the feathers. Recent work shows that ingestion of small amounts of oil by herring gull chicks can interfere with osmoregulation which result in death by dehydration and reduced ability to survive stress (Miller et al., 1978). Significant mortalities are produced by microliter quantities of crude and fuel oil applied directly to the surface of incubating eggs (Albers, 1977; Szaro and Albers, 1977). Even though many aromatic compounds oxidize fairly readily, they have been known to persist in marine sediments for as long as seven years after an oil spill, particularly where anaerobic conditions exist (Michael et al., 1975; Keizer et al., 1978).

The U.S. EPA (1980c) has concluded that insufficient data are available to establish an ambient water criterion for PAHs. It has, however, proposed an interim criterion value of 0.2 ppb for total PAHs based on the WHO (1970) recommendations concerning the levels of combination of three carcinogenic PAHs in drinking water.

#### 5.5.4 Human health effects

Of the thousands of compounds that make up crude oil, the aromatic hydrocarbons are the most toxic and most persistent in the marine environment. Petroleum hydrocarbons can be separated into three major compounds. Paraffin hydrocarbons (alkanes) are straight-chain structures with as many as 60 carbon atoms per molecule. Cycloalkanes (naphthenes) consist of saturated five or six-member carbon rings and polycyclic combinations of these. Aromatics include unsaturated ring structures such as benzene, alkyl-substituted benzenes, naphthalenes, and polynuclear hydrocarbons with multiple alkyl substitutions. This group includes the compounds commonly referred to as PAHs, polynuclear aromatics (PNA) or polynuclear aromatic hydrocarbons (PNAH), which are of particular interest because of carcinogenic effects in mammals. Aromatics make up 30 to 50% of crude oils.

The EPA list of toxic substances that should receive priority attention (U.S. EPA, 1975) includes polynuclear aromatics that are, or are suspected of being, carcinogenic: benzantracenes, benzene, benzofluoranthene, benzopyrenes, chrysenes, dibenzanthracenes, ethyl benzene, idenopyrenes, naphthalene, and toluene. Benzene is suspected of producing leukemia after chronic exposures (Laskin and Goldstein, 1977). Several PAH compounds are known human carcinogens, with benzo(a)pyrene (BaP) frequently the most active of the group found in the environment (NAS, 1972). The polycyclic compounds are metabolized by enzymes in the tissues to active species (proximal carcinogens) which react with the target cells to initiate the production of cancer.

Additional carcinogens are present in the mixture of petrochemicals entering the Apex region of the Bight but only limited data on the inputs and resulting concentration inventory have been reported (Mueller et al., 1982; MacLeod, et al., 1981).

Aromatic hydrocarbons include compounds with a wide range of molecular weights, vapor pressures, and chemical properties. All compounds in the aromatic group contain one or more unsaturated six-membered carbon rings.

Exposures to aromatic compounds cause activation of liver enzyme systems and may cause CNS, irritant, and other effects (NIOSH, 1973, 1975).

The PAHs are contained in aerosols emitted from fuel-burning sources, including autos, power plants, and incinerators. This is perhaps significant in terms of dietary exposures, as food generally contains much less than 10 g BaP/kg, while smoked fish and meats have been found to contain from 10 to 30  $\mu$ g BaP/kg. Evidence of increased gastrointestinal cancer exists in Japanese and Scandinavian populations having large dietary components of smoked products (IARC, 1973, p. 50).

#### 5.5.5 Ecological and public health implications

As previously noted, quality criteria for oil and grease in marine waters and sediments have not been established by the EPA because petroleum from different sources have different compositions. However, the lack of straightforward guidelines should not direct attention away from the damaging effects of chronic exposure of marine organisms to petroleum hydrocarbons.

The EPA report (1976) tabulated the results of exposure of organisms petroleum and summarized them as follows. "The most susceptible category of organisms, the marine larvae, appear to be intolerant of petroleum pollutants, particularly the water soluble compounds at concentrations as low as 100 ppb. The long-term sublethal effects of oil pollution refer to interferences with cellular and physiological processes such as feeding and reproduction that do not lead to immediate death of the organism. Disruption of such behavior apparently can result from petroleum product concentrations as low as 10 to 100 ppb."

The computed potential contamination level of 44 ppb together with measured values of the same order, suggest that current inputs can cause damage to marine organisms inhabiting the Apex region, primarily through mechanisms that cause reductions in feeding and reproductive success. The actual level of contamination in Apex waters needs to be documented by seasonal sampling throughout the Apex region.

The public health aspects of petroleum contamination arise either from the tainting of edible species, which makes the organisms less desirable as food, or from the incorporation of carcinogenic PAHs in their tissues. The magnitude of these effects needs to be assessed by sampling the organisms in the Apex region and analyzing for the tainting and carcinogenic compounds.

In view of the observed changes in the kinds of organisms that inhabit the benthic environment in cases of petroleum contamination, there is little question that the Apex region benthic communities have been extensively stressed. Concentrations of 20 to 50 ppm total hydrocarbons have been observed to produce changes in the composition of benthic communities (Michael *et al.*, 1975). The observed hydrocarbon concentrations of 300 to 1800 ppm (Farrington and Tripp, 1977) must be contributing to effects on the benthic populations that have been reported to correlate with waste disposal activities.



Table 1. Ranking of significance of chemical contaminants of the New York Bight, June 1977, by MESA Contaminants Panel. Contaminants are listed alphabetically within categories.

Category A	Category B	Category C
Cadmium	Arsenic	Chromium
Chlorinated pesticides <sup>1</sup>	Benzidenes	Haloalkyl ethers
Mercury	Chlorobenzenes	Nitrobenzenes
Polynuclear aromatic hydrocarbons (PNAHs)	Chlorophenols	Nitrophenols
Polychlorinated biphenyls (PCBs)	Diphenylhydrazine	Phenols
Plutonium	Halogenated diphenyl ethers	Pthalates
Lead	Isopherone	Selenium
	Low-molecular-weight, halogenated hydrocarbons (LMHHs) <sup>3</sup>	Silver
	Petroleum hydrocarbons (PHCs) other than PNAHs, other than alkanes and cycloalkanes	
	Thallium	

<sup>1</sup>Aldrin/Dieldrin, Chlordane (technical mixture and metabolites), DDT and metabolites, Endosulfan and metabolites, Endrin and metabolites, Heptachlor and metabolites, hexachlorocyclohexane (all isomers), and Toxaphene.

<sup>2</sup>Aromatic compounds with unsaturated ring structures: Benzene, alkyl-substitute benzenes, and polynuclear hydrocarbons with multiple alkyl substitutions.

<sup>3</sup>Carbon tetrachloride, chloroform, chlorinated ethanes (included 1,2-dichloroethane, 1,1,1-trichloroethane, and hexachloroethane), 1,1-tetrachloroethylene, trichloroethylene, and vinyl chloride.

## 6. SUMMARY

Current estimates of the input rates of contaminative materials to the Apex region of the New York Bight have been used with flushing rates to compute "upper bound" values for the possible resulting contamination levels. These computed concentrations, as well as the results of sampling the Apex region, may be viewed against water quality criteria to assess the probability of public health or ecological effects.

Present evidence concerning input rates and contamination of marine organisms from the Apex region suggest that Apex-wide ecological or public health damage due to anthropogenic sources of mercury, cadmium, lead, or polychlorinated biphenyls is not occurring at present. However, the computed contamination levels for these various materials are only factors of 2 to 10 below the water quality criteria, and synergistic (combined) deleterious effects are problematical. Also, localized impacts around input areas, where above-average contamination occurs, can be expected. The input rates for aromatic hydrocarbons are so high that substantial questions of damage to indigenous marine organisms, particularly the benthic populations, need attention.

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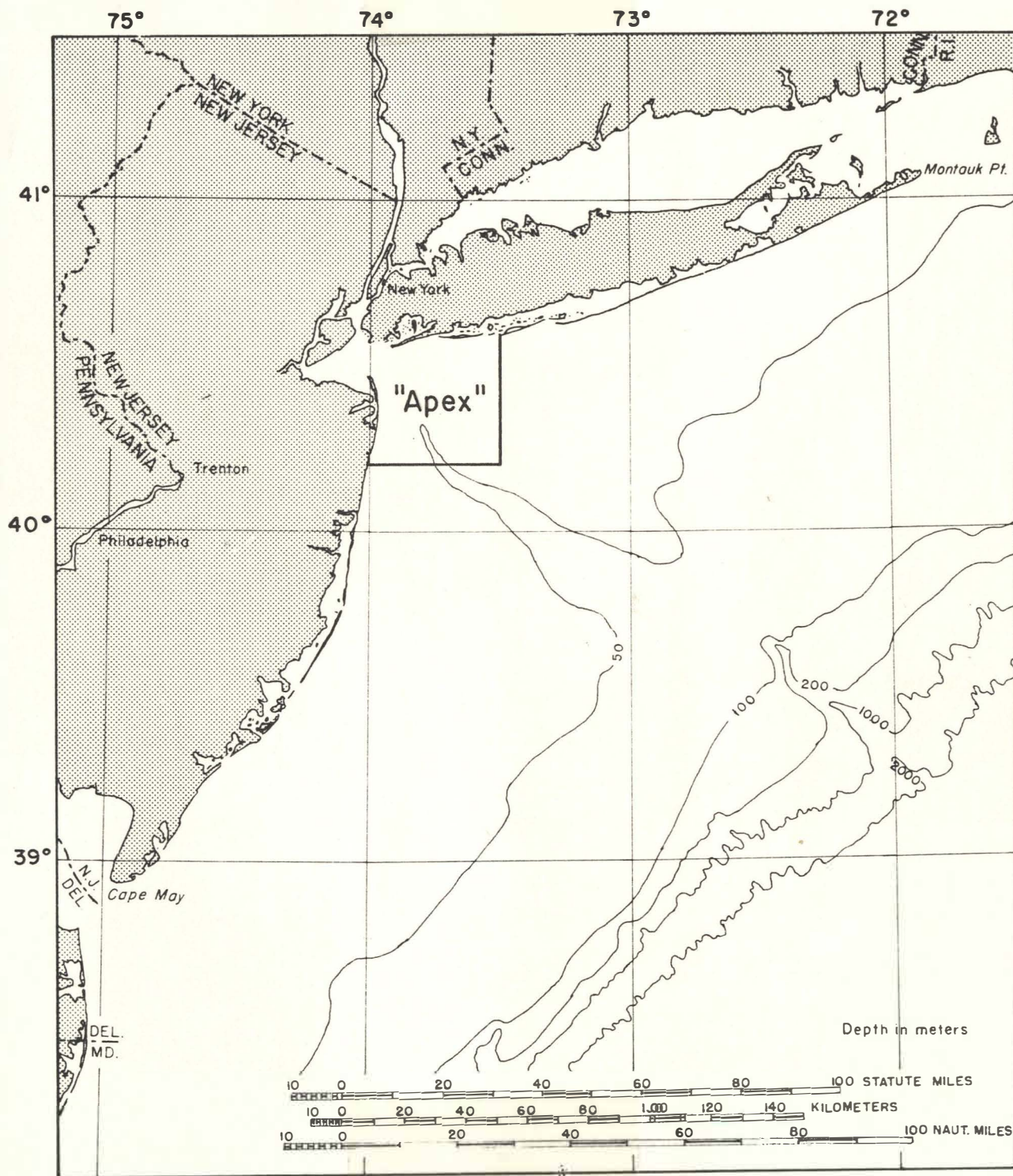


Figure 1. Definition of the "Apex" area.

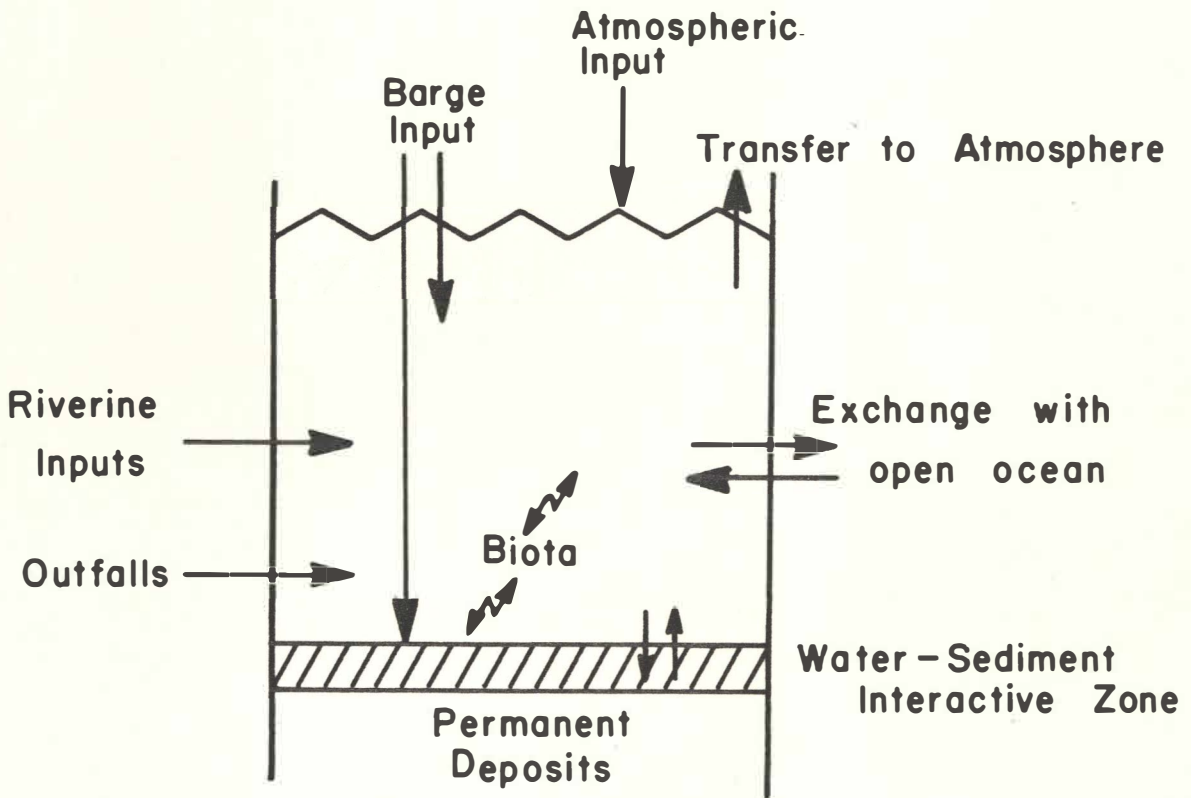


Figure 2. Processes affecting contamination in the Apex region of the New York Bight.

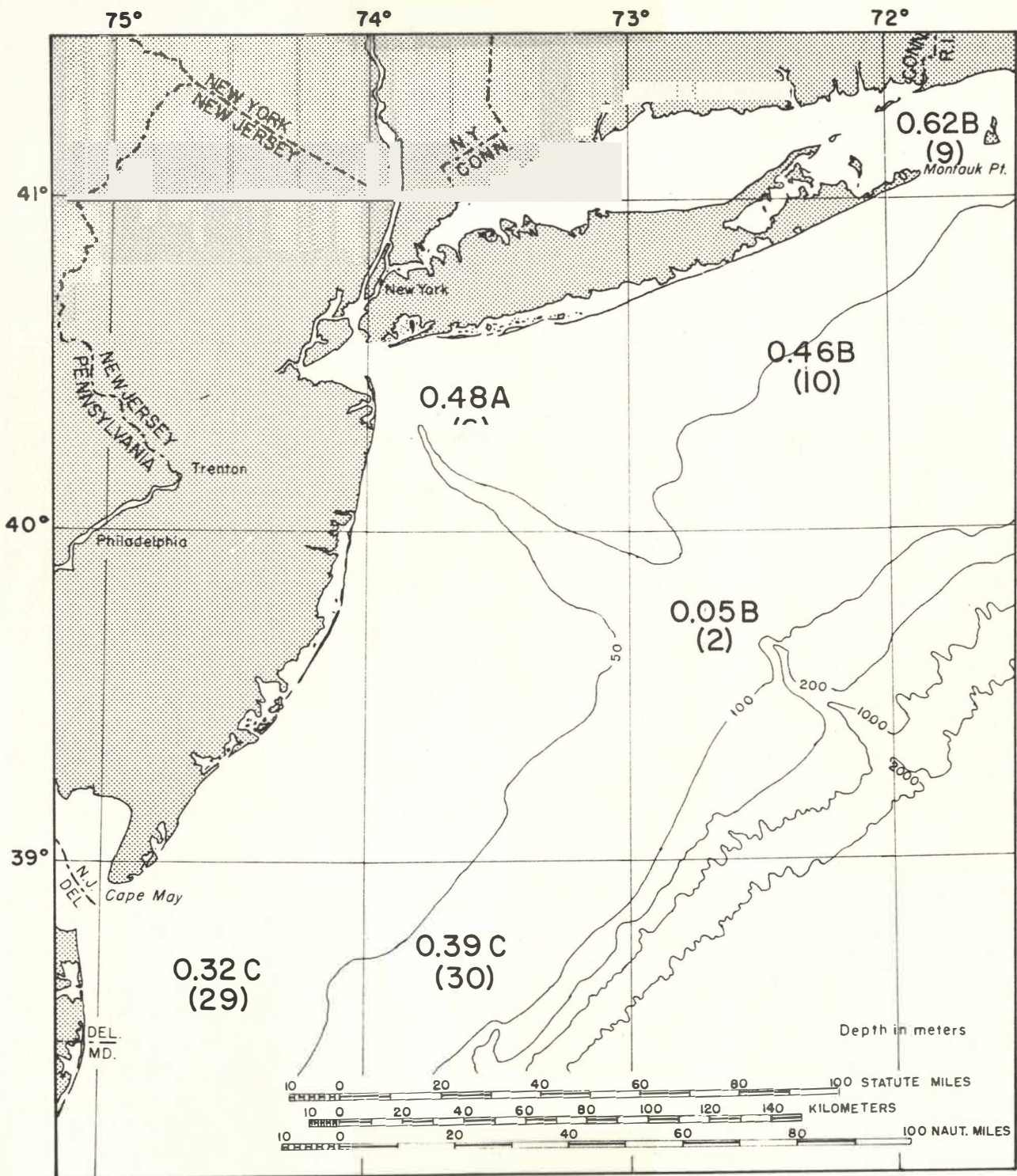


Figure 3. Mercury content of muscle from spiny dogfish (*Squalus acanthias*) as ppm wet weight. Letter A indicates SEFC data; B, NEFC data; and C, combined NEFC /SEFC data. Number of organisms studied is shown in parentheses.



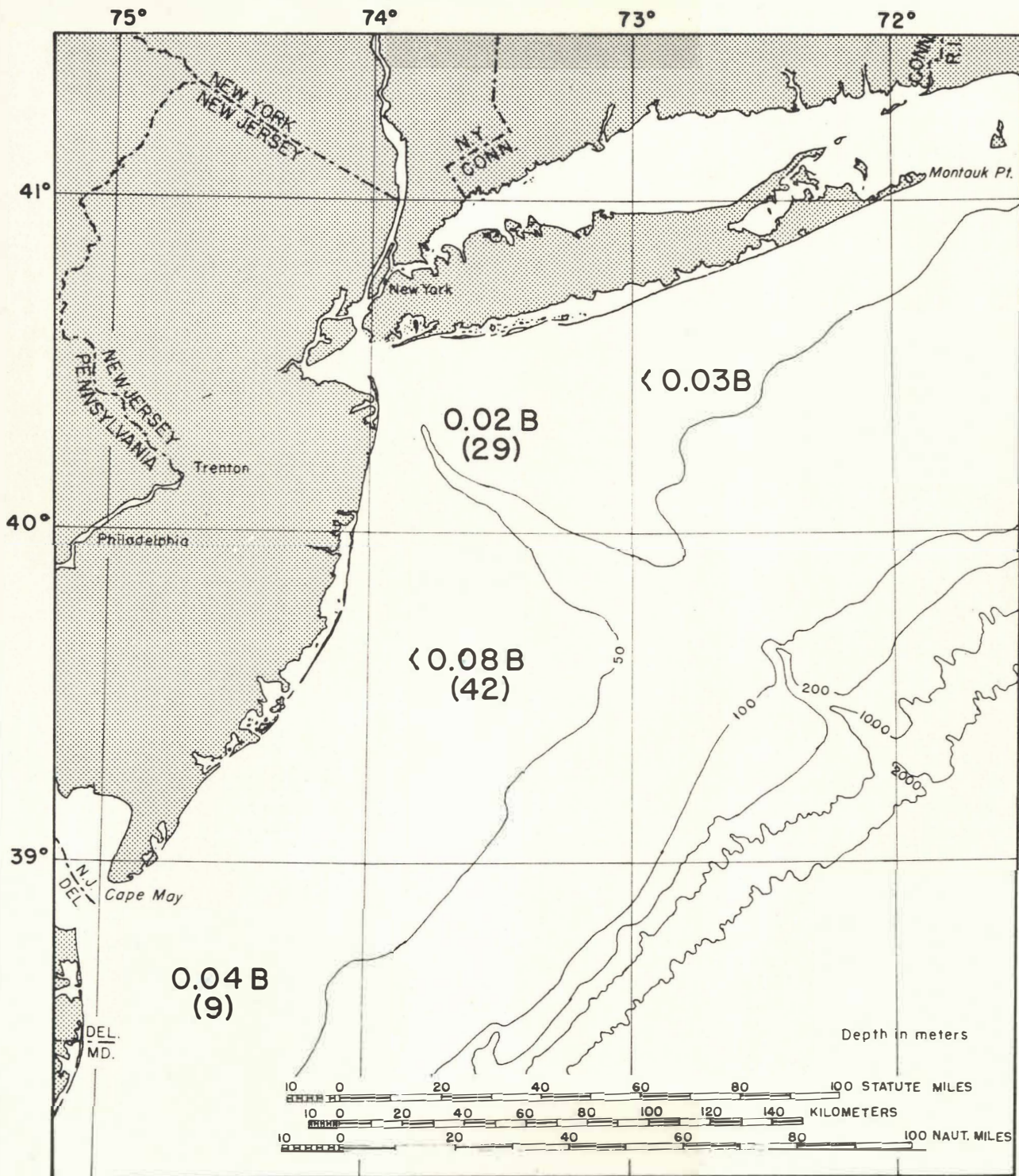


Figure 4. Mercury content of whole bodies of surf clams (*Spisula solidissima*) as ppm wet weight. Letter A indicates SEFC data; B, NEFC data; and C, combined NEFC/SEFC data. Number of organisms studied is shown in parentheses.

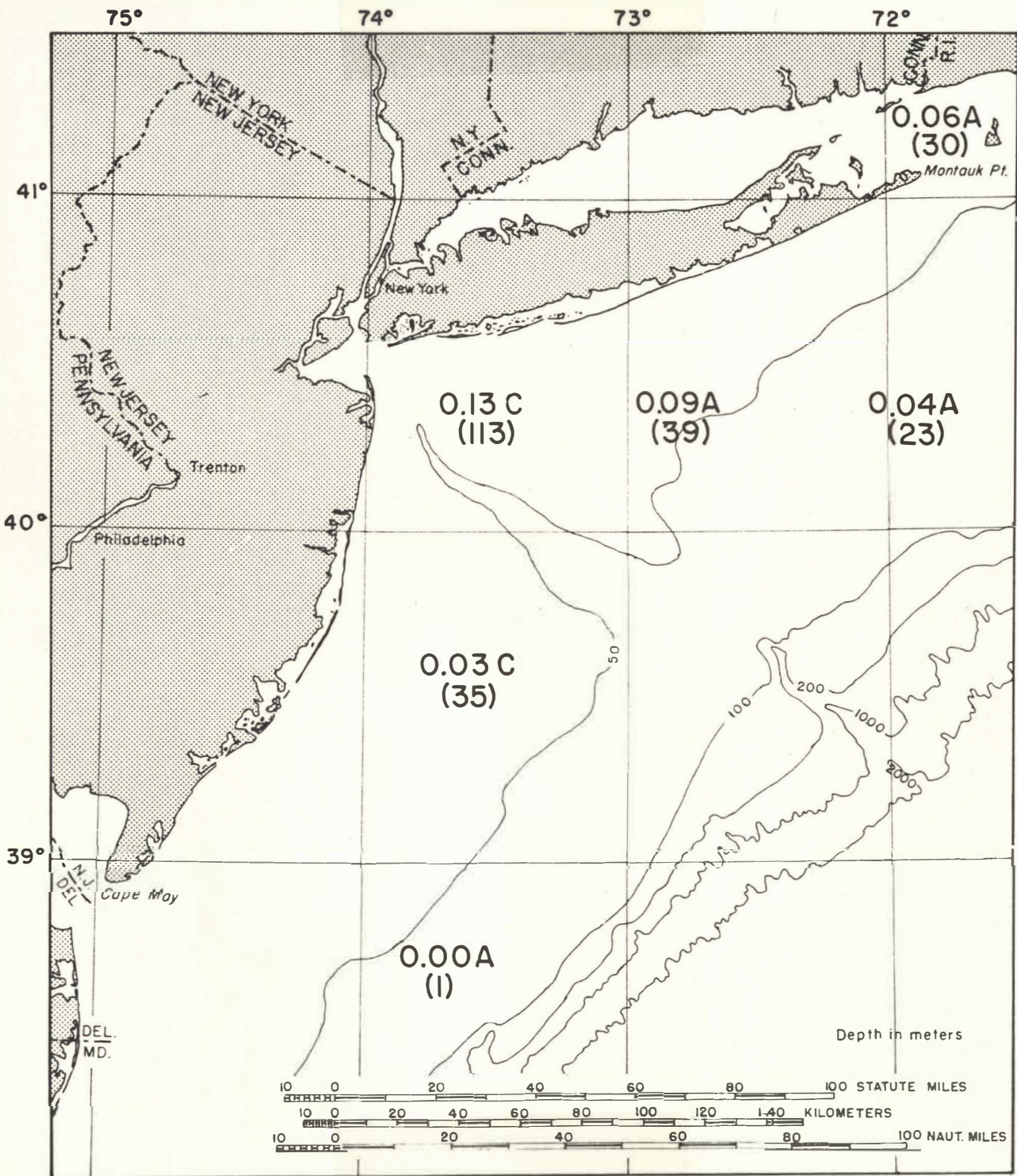


Figure 5. Mercury content of muscle from winter flounder (*Pseudopleuronectes americanus*) as ppm wet weight. Letter **A** indicates SEFC data; **B**, NEFC data; and **C**, combined NEFC/SEFC data. Number of organisms studied is shown in parentheses.

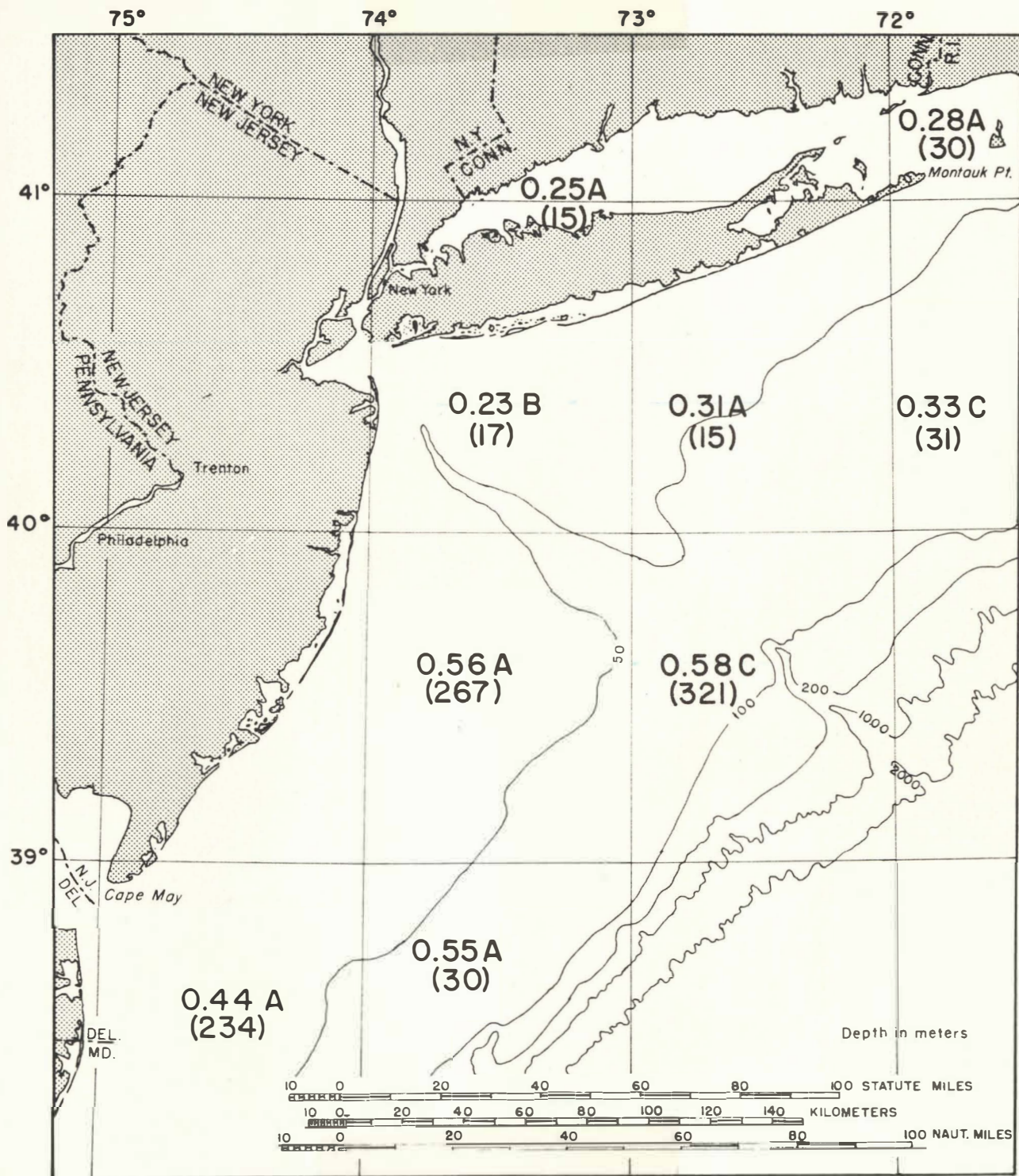


Figure 6. Mercury content of body, claw and tail of North American lobster (*Homarus americanus*) as ppm wet weight. Letter **A** indicates SEFC data; **B**, NEFC data; and **C**, combined NEFC/SEFC data. Number of organisms studied is shown in parentheses.

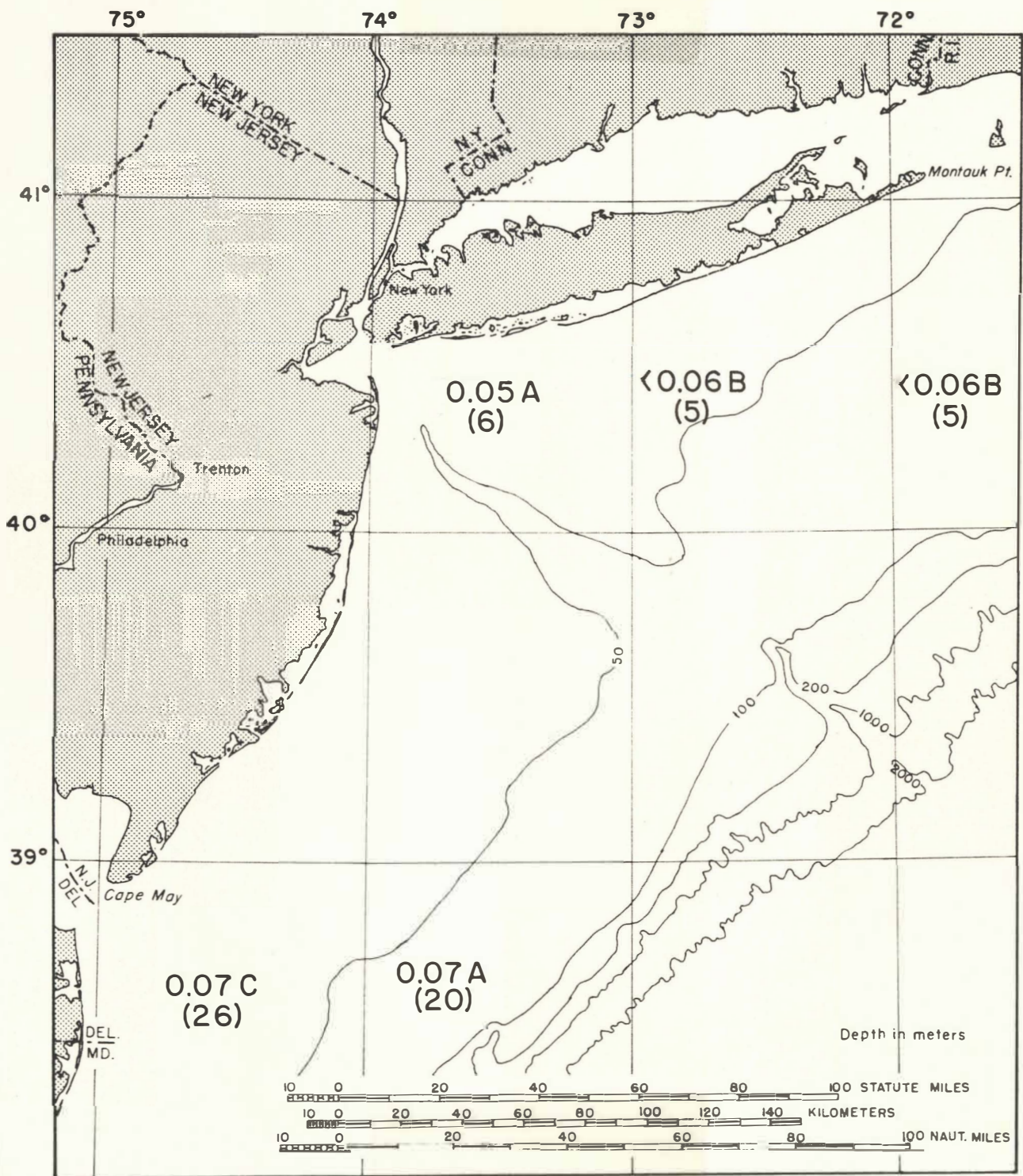


Figure 7. Cadmium content of muscle from spiny dogfish (*Squalus acanthias*) as ppm wet weight. Letter A indicates SEFC data; B, NEFC data; and C, combined NEFC/SEFC data. Number of organisms studied is shown in parentheses.

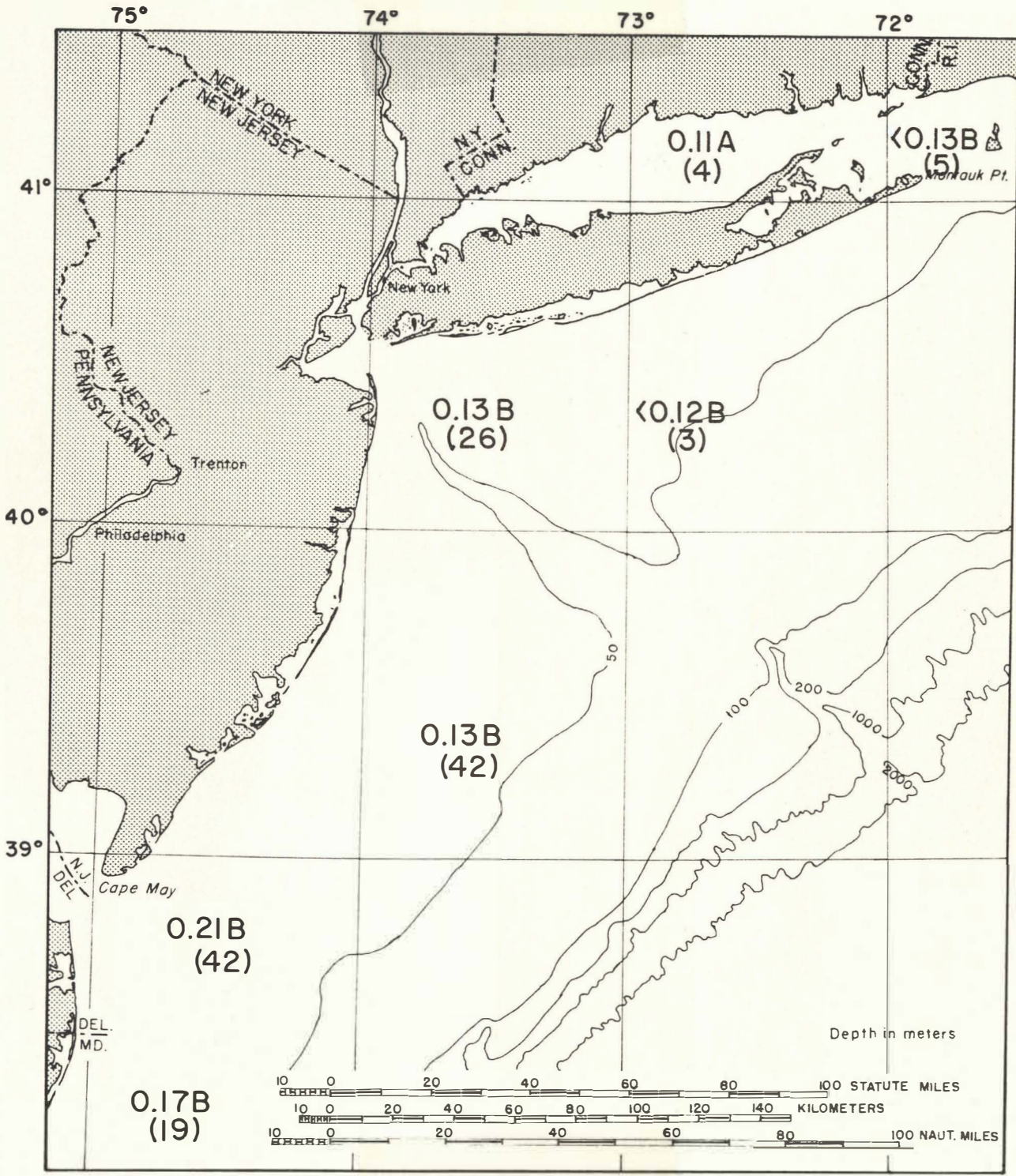


Figure 8. Cadmium content of whole bodies of surf clams (*Spisula solidissima*) as ppm wet weight. Letter A indicates SEFC data; B, NEFC data; and C, combined NEFC/SEFC data. Number of organisms studied is shown in parentheses.

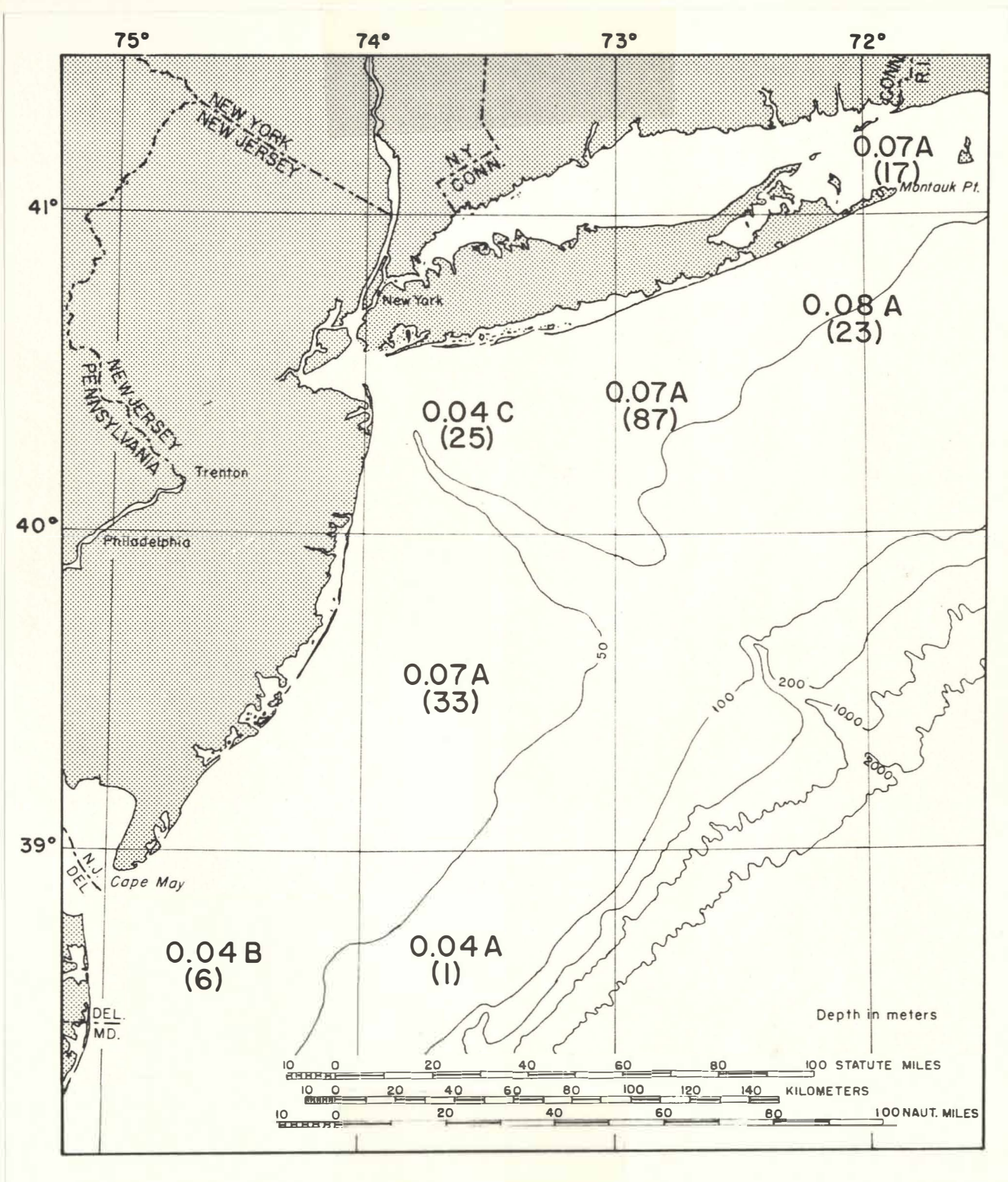


Figure 9. Cadmium content of muscle from winter flounder (*Pseudopleuronectes americanus*) as ppm wet weight. Letter A indicates SEFC data; B, NEFC data; and C, combined NEFC/SEFC data. Number of organisms studied is shown in parentheses.

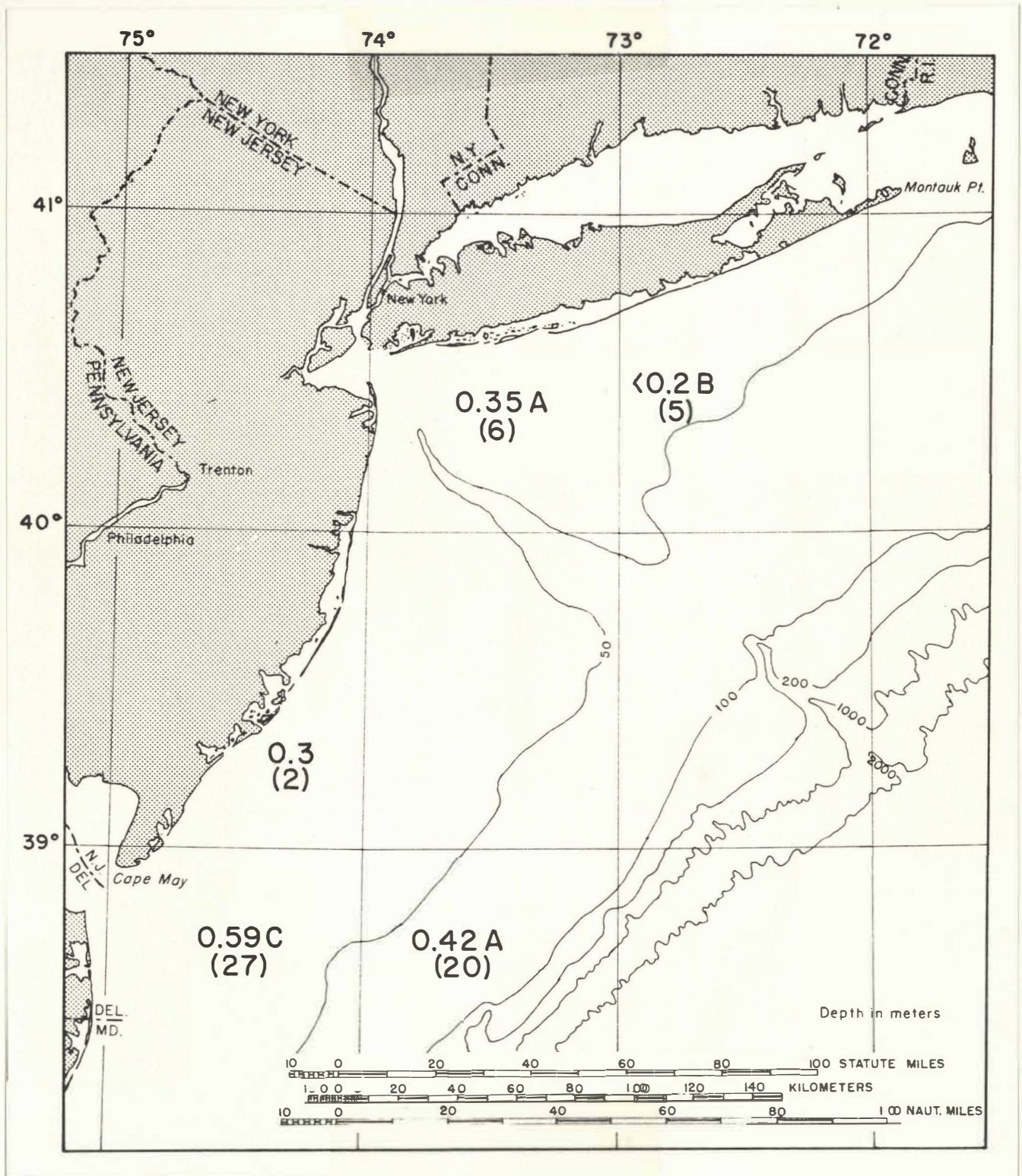


Figure 10. Lead content of muscle from spiny dogfish (*Squalus acanthias*) as ppm wet weight. Letter A indicates SEFC data; B, NEFC data; and C, combined NEFC/SEFC data. Number of organisms studied is shown in parentheses.

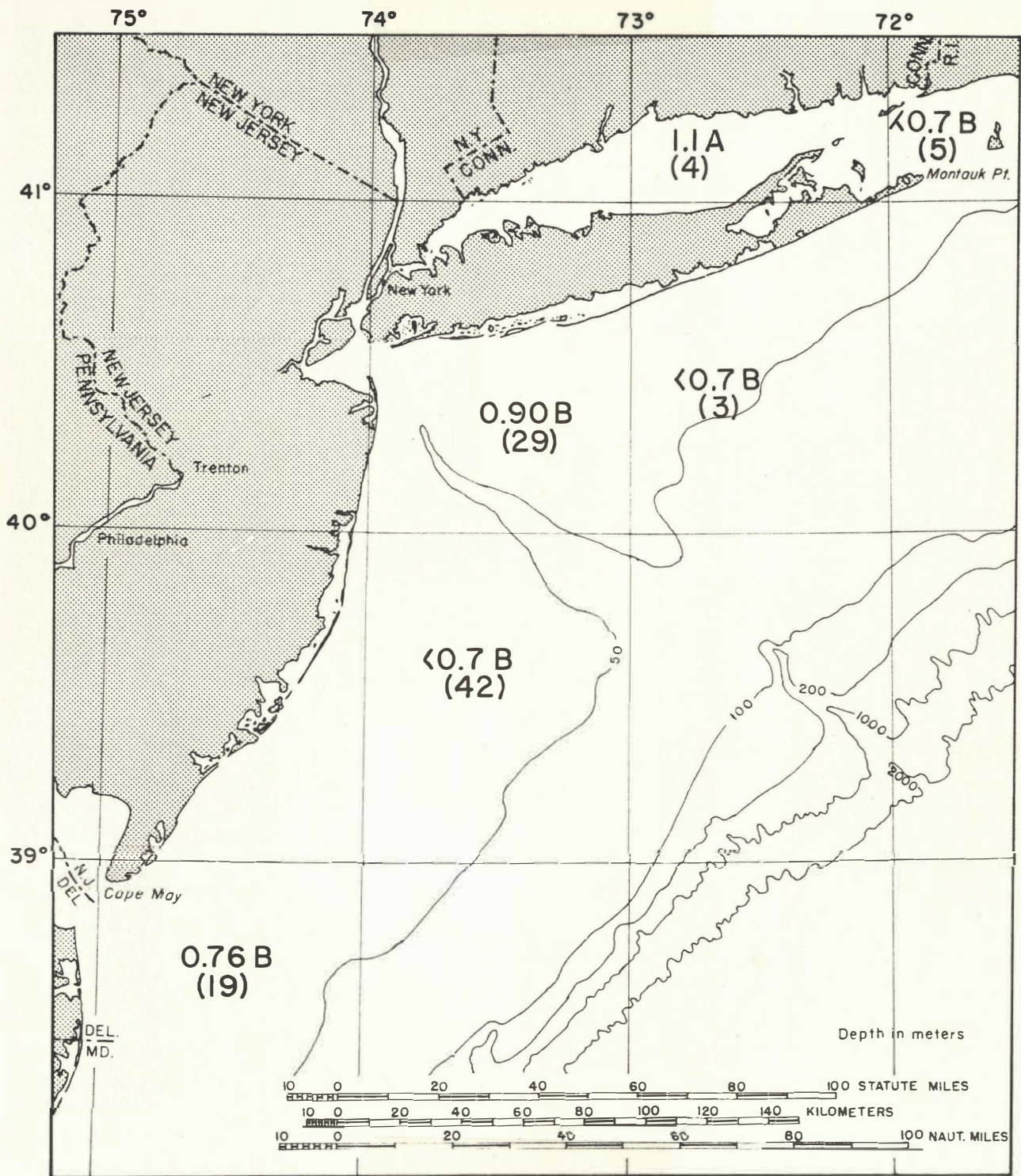


Figure 11. Lead content of whole bodies of surf clams (*Spisula solidissima*) as ppm wet weight. Letter A indicates SEFC data; B, NEFC data; and C, combined NEFC/SEFC data. Number of organisms studied is shown in parentheses.



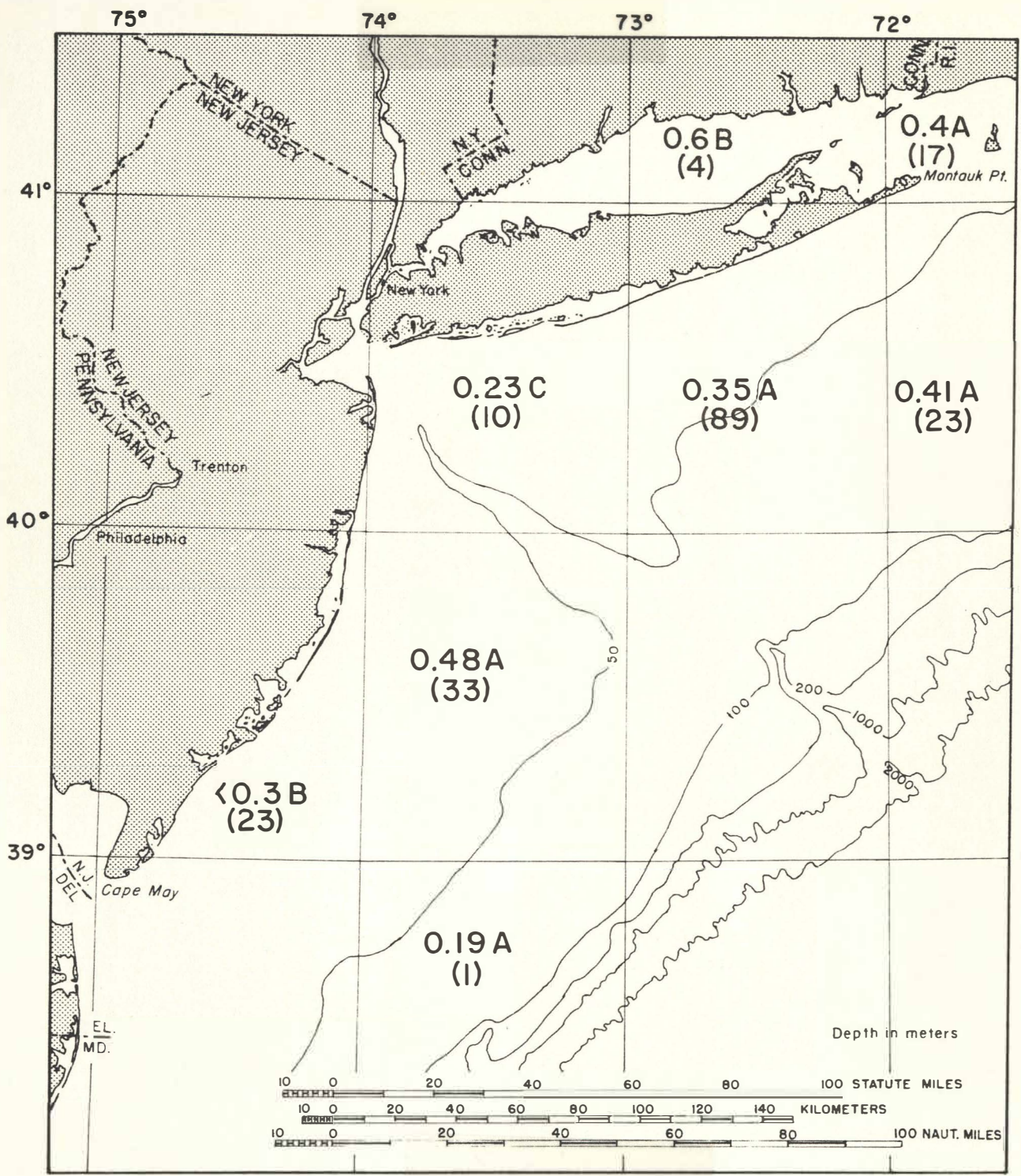


Figure 12. Lead content of muscle from winter flounder (*Pseudopleuronectes americanus*) as ppm wet weight. Letter A indicates SEFC data; B, NEFC data; and C, combined NEFC/SEFC data. Number of organisms studied is shown in parentheses.