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A Survey of Potential Problems Related to Toxic Organic Chemical Contamination of Aquatic Environments

Carol B. Rideout
Virginia Institute of Marine Science

Michael E. Bender
Virginia Institute of Marine Science

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by

Carol B. Rideout and Michael E. Bender

Virginia Institute of Marine Science
School of Marine Science
College of William and Mary
Gloucester Point, Virginia

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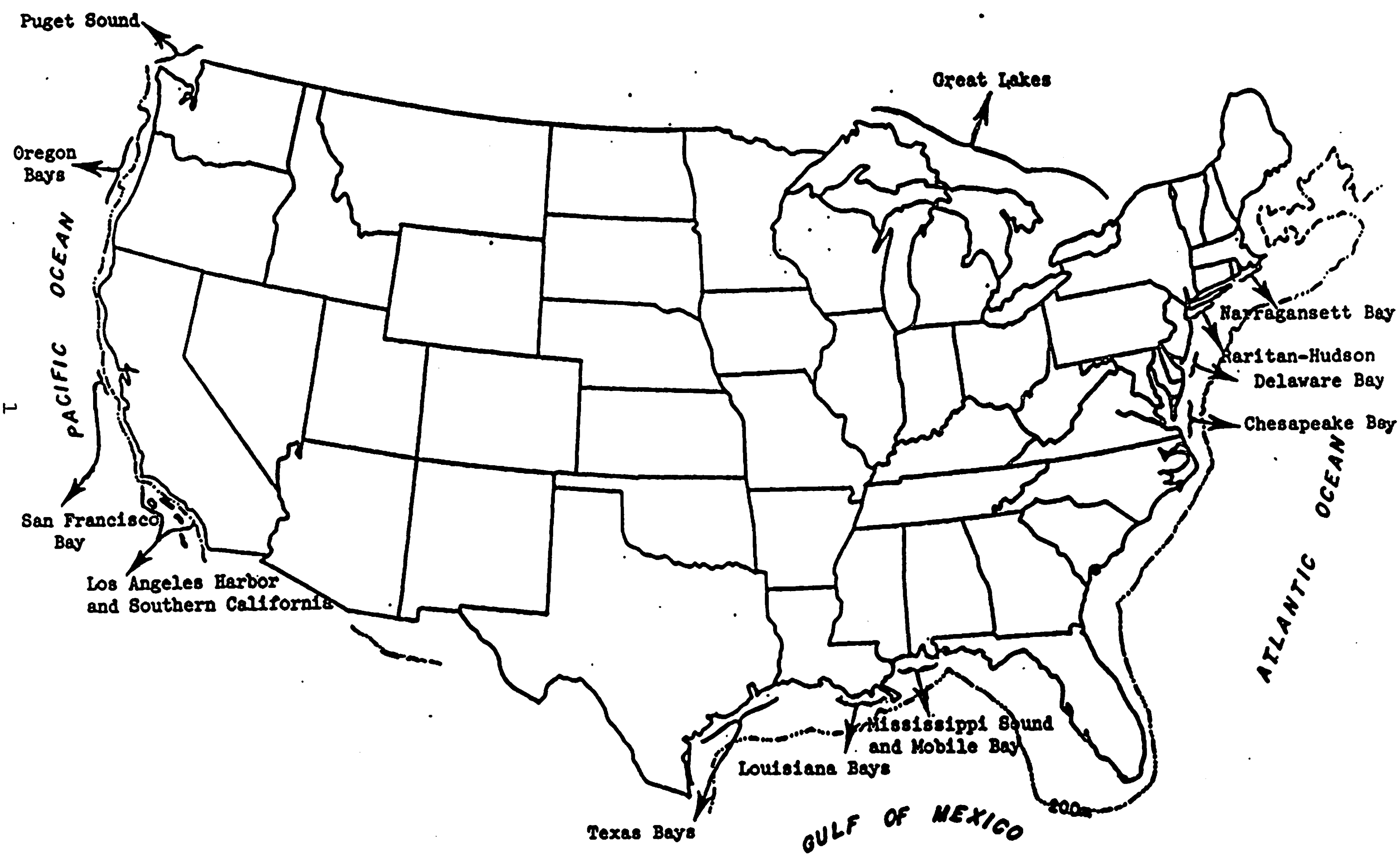
Figure 1. Map of continental U.S.A. showing study areas.



SUMMARY AND RECOMMENDATIONS

Toxic organic chemicals have affected aquatic resources by (1) restricting harvest; (2) causing biological damage to harvestable stocks; and (3) damaging other biological resources, eg. benthic animals and birds.

Geographically the areas of the country most affected are the Great Lakes region and localized coastal areas, along the east and west coasts. However, the actual extent of problems related to toxic organics is not known mainly because only a limited number of cause and effects studies have been conducted.

In the authors' opinion there is a generic need for the following:

(1) more refined monitoring of organic contaminants in aquatic animals and sediments, i.e. inclusion of enclosed areas in sampling, inclusion of a wider variety of organics in analysis, improved and more uniform analytical procedures;

(2) research to better establish cause and effect relationships between individual contaminants and mixtures, i.e. more thorough investigation of the possibility of synergistic effects of organic contaminants and metals in inducing cancerous growth in aquatic organisms and further investigation of the role of metabolites of organic contaminants in inducing abnormalities;

(3) research and monitoring to better establish the extent, i.e. geographic areas, which are affected by toxic organics.

The authors further recommend consideration of introducing or improving control methods in problem areas. Control might include measures for reducing pollutant input at nonpoint sources before pollutants enter receiving waters and better controls on those inputs which enter municipal wastewater treatment facilities. Such measures might include

(1) detention basins
Latimer, et al. (1986) recommended the use of detention basins as an effective and economical method of reducing the impact of petroleum hydrocarbon and PAH pollution from storm runoff on receiving water. They collected influent and effluent samples from a small detention basin serving a large shopping center parking lot. They found that treatment of hydrocarbons by the detention pond was 70% better and treatment of PAHs was 14% better "than treatment efficiencies of a combined sanitary/storm municipal treatment plant studied in Providence, RI [by] (Hoffman, et al. 1984)" (Latimer, et al., 1986).

(2) legislation re slash burning

In the authors' opinion legislative bodies, especially those governing places where there is logging activity, should consider putting restrictions on slash burning or should consider legislation offering logging companies incentives to use alternate practices, as pyrogenic PAHs resulting from slash burning are a threat to health of the bodies of water they enter with runoff and a potential threat to consumers of living natural resources there.

(3) legislation re disposal of used motor oil
Federal laws, state regulations, or city and county ordinances might prohibit disposal of used motor oil except at designated collection sites from which the oil might be taken to recycling facilities as those suggested by Quinn (1979) described under Narragansett Bay in this Summary and Recommendations section.

For specific areas of the coast we suggest the following:

Puget Sound

The effects of organic contaminants on biota have been extensively studied in Puget Sound. In fact the best research in the nation regarding the potential impacts of organic pollutants has been performed on this area. However, more needs to be done particularly in elucidating cause and effect relationships. In the opinion of the authors continued and expanded support should be given to the investigations being conducted by the Northwest and Alaska Fisheries Center of the National Marine Fisheries Service. In addition colaborative efforts between the Fisheries Center and other groups, eg. the studies proposed by Malins and Huggett, should be given serious consideration for funding at the national level.

Oregon Bays

We recommend continued monitoring for PAH levels in shellfish and the addition of regular monitoring of sediments at sites in embayments and near marinas in Oregon Bays. The practice of slash burning large forest areas after logging, the presence of creosoted pilings, fish processing plants, marinas, and automobile exhaust components in road runoff (although admittedly there is not as much automobile traffic in Oregon as in more heavily populated areas of the west coast) all contribute PAHs to Oregon bays; therefore levels should be watched.

We believe that analysis of sediments and biota should include PCB and pesticides analysis as well. There is a possibility of pesticides from farm lands east of the coastal forests reaching the bay waters and contaminating biota there.

San Francisco Bay

Major studies of organic contaminants and problems of San Francisco Bay have concentrated on large central areas in the Bay, in a sort of regional approach. Results of these studies paint a "cleaner" picture of the Bay when compared to results of voluminous studies of Puget Sound areas and of major sewage outfalls in Southern California. The comparison lacks credibility until there are studies of small bays and inlets and specific sites near potential sources of organic pollutants. We recommend such studies be conducted.

Also we recommend further studies to sort out the factors and combinations of factors contributing to the decline of the fisheries in the Bay.

Citizens for a Better Environment (1983) pointed out that most effluent standards for the Publicly Owned Treatment Works (POTWs) were based on values recommended in California's Ocean plan. Since the Bay is an enclosed estuary, they fear these standards may be inappropriate and not sufficiently protective. They said standards for toxic organics "are grossly deficient or totally non-existent" as is adequate monitoring of toxic organic pollutants.

They recommended that the POTWs in the Bay area develop uniform prohibited discharge standards and that the San Francisco Bay Region of the Regional Water Quality Control Board require effluent monitoring of industrial dishargers and sewage treatment plants that would include analysis for detection of relevant toxic organic pollutants. One specific recommendation, with which we concur, was to replace broad analysis for "oil & grease" with analysis for petroleum hydrocarbons, PAHs and total aromatic hydrocarbons.

Southern California

We believe there is need for more comprehensive and more current measurements of PAHs in biota of southern California. We would also like to see sediment toxicity studies to determine lethal and sublethal effects of sediments from various locations in southern California, especially Los Angeles Harbor and Palos Verdes shelf, on species indigenous to the area.

Texas Bays

Pesticide contamination problems in the Arroyo Colorado are severe as indicated by residue levels in biota and fishes. Impacts on the biota are unknown and additional studies are needed to determine the potential impacts of this contamination.

Davis (Personal communication, July 12, 1985) reported that EPA will fund studies of effects of contaminants on biota, indicating highest acceptable contaminant levels. He commented that very little of this type research had been done to date.

The Texas Natural Resources Information System (TNRIS) is a highly sophisticated system with a large collection of potentially valuable data available for use in making decisions concerning environmental protection. TNRIS provided the authors with data tapes containing State Monitoring Network data and Coastal Data System data which resulted from monitoring efforts in estuarine waters of Texas 1968-1984. The data included physical, chemical and biological parameters. No PAH levels were included in the chemical data. A careful examination of all the data revealed, however, that in only very few instances, ~10, were there biological data, eg. benthic diversity, from the same location and general time period as chemical data, i.e. sedient concentrations of PCB and/or a pesticide.

We would like to see future monitoring plans coordinated so that a number of stations sampled for chemical data also be sampled for biological data.

The lack of available PAH data needs to be corrected. Jack Davis, benthic ecologist for Texas Department of Water Resources, Water Quality Management Division (Davis, personal communication, July 12, 1985) has sampled Corpus Christi inner harbor, Arroyo-Colorado, and other Texas locations for EPA priority polllutant analysis which, of course, includes PAHs. Apparently they have been sampling only sites suspected of having elevated pesticide levels.

Louisiana and Texas

With regard to persistant hydrocarbon contamination from oil spills and chronic releases, Boesch, et al. (1985) recommended the following:

> "The long-term effects of oil spills are potentially the most serious of the effects of offshore oil and gas development activities, but also the hardest to define and control. There is a long history of research on the fate and effects of oil in the marine environment, yet viewpoints in the scientific community on some issues diverge widely (Royal Commission on Environmental Pollution, 1981; Clark, 1982; National Research Council, 1984). The research approaches outlined in Table 1.5 address the most important unresolved issues concerning effects of persistent contamination by petroleum hydrocarbons and degradation products in environments conducive to such retention--fine-grained sediments and cold environments. The study approaches recommended are far more complex and multifaceted than those addressing subsequent issues. They also involve more generic experimental research; the other issues are more heavily dependent on field observations."
>
> "The recommended approaches addressing the long-term effects of hydrocarbon contaminants fall into three groups: a) the sedimentologic and geochemical dynamics of hydrocarbon contaminants and their degradation products (what compounds persist, for how long, and where are they transported?) b) bioavailability (are the persistent compounds taken up by the biota, are they bioaccumulated by air-breathing animals, what insight can be provided by measuring

body burdens in stranded animals?); and c) chronic and sublethal effects."

"Research on the long-term fate of medium and high molecular weight aromatics, heterocyclics and their degradation products depends on understanding the conditions which allow them to persist. Most of th existing information is based on field observations or small-scale laboratory experiments. Controlled experiments are required to separate multiple factors, and experimental approaches should be generally scaled up for increased realism. Thus, mesocosm and field experimental approaches are particularly recommended. These will allow better extrapolation to natural conditions through application of site-specific sediment transport models."

"Research on bioavailability should concentrate on the uptake and retention of contaminants from bottom sediments and on the potential for long-term build up of contaminants in the tissues of birds, mammals and turtles, in which the primary uptake route is probably direct ingestion."

"Determination of chronic and sublethal effects on the biota is always difficult. During th past decade, there has been a proliferation of stress indicator techniques for evaluating responses of organisms to pollutants (McIntyre and Pearce, 1980), yet the relationship of the response to survival of the individual, much less the population, is frequently unknown. Particularly sensitive are biochemical responses that relate to energy metabolism and membrane function (such as lysosomal stability), biochemical responses that relate to detoxification (such as induction of mixed-function oxidases), and physiological responses (such as scope for growth) that measure the energy available for growth and reproduction."

"Induction of mixed-function oxidase activity in marine organisms is a response to petroleum hydrocarbons which detoxifies and removes hydrocarbons but also produces more toxic primary and secondary metabolites. Recent evidence has suggested a direct relationship between detoxification processes and a) loss of reproductive effort (Spies et al., 1983), b) developmental and energetic abnormalities (Capuzzo et al., 1984), and c) histopathologic changes (Malins et al., 1983). Other parameters that may prove to be useful in predicting significant effects on populations include changes in blood glucose, energy metabolism or hormonal levels, particularly when these measurements can be made in conjunction with estimates of mixed-function oxidase activity, metabolite formation and organismal effects, such as fecundity and development rate. No single index can provide the predictive capability to evaluate population change; hence future efforts should emphasize the relationship of multiple response indices."

Table 1. Recommended study approaches for the resolution of potential long-term effects resulting from the persistence of medium and high molecular weight aromatic hydrocarbons, heterocyclics, and their partial degradation products.

STUDY APPROACHES	FEASIBILITY[1]	REGIONAL FOCUS	DURATION (YEARS)	SEQUENCING
1. Sedimentologic and geochemical dynamics of hydrocarbons and products				
a. Determine the persistence of parent compounds and their degradation products under various conditions (temperature, light, oxygen, microbial populations).	High	Generic experimental research	5	Immediate
b. Conduct mesocosm and field experiments and make long-term field observations on sedimentation, retention, degradation and erosional transport of petroleum hydrocarbons in sediments.	High-Potential	Generic experimental research	5	Laboratory studies concurrent with and dependent on results of 1, a
		Field research should emphasize areas with extensive nearshore, fine-grained sedimentary or cold environments, particularly Gulf of Mexico and Alaska.	10	Field studies dependent on results from 1, a and experimental studies; long-term for life of development or activity being assessed; low-level effort
c. Develop process models which relate the fate of contaminants to sediment type, erosion and deposition and likelihood of contamination.	Potential	Generic research with emphasis on environments likely to be developed and contaminated	3	Initial descriptive models to direct the above research and quantitative models which employ results of 1, a and 1, b

Table 1. continued

STUDY APPROACHES	FEASIBILITY[1]	REGIONAL FOCUS	DURATION (YEARS)	SEQUENCING
2. Bioavailability of petroleum hydrocarbons and their degradation products to various trophic groups of benthos and valuable species				
a. Conduct experiments and make field observations to determine bioavailability of compounds from contaminated sediments.	High-Potential	Generic laboratory research Field research should emphasize areas with extensive nearshore, fine-grained sedimentary or cold environments, particularly Gulf of Mexico and Alaska; contingent on results of 1, b.	5 10	Laboratory studies concurrent with and dependent on 1, a Field studies follow at a low level from progress in 1, b
b. Determine the bioaccumulation potential via ingestion by birds, mammals and turtles.	High-Potential	Research should emphasize: Alaska and California (mammals, shearwaters, storm petrels, alcids); Gulf of Mexico and Atlantic (turtles, mammals).	5	Immediate
c. Relate bioavailability of major compounds to their persistence in sediments.	Potential	Generic research	1 2	Initial models following results of 1, a Subsequent refinement following results of 1, b and 2, a-b

Table 1. continued

STUDY APPROACHES	FEASIBILITY[1]	REGIONAL FOCUS	DURATION (YEARS)	SEQUENCING
3. Sublethal effects of petroleum hydrocarbons and their degradation products				
a. Determine biochemical and physiological responses to exposure in potentially susceptible biota.	Potential	Generic laboratory research Field research as above in 1, b and 2, a-b and in chronically contaminated environments	5	Following progress of 1, a-b and 2, a-b
b. Determine the quantitative importance of detoxification reactions that produce more toxic intermediate metabolites.	Potential	Generic laboratory research	5	Following progress of 2, a-b
c. Assess the relationship between tissue concentrations and pathological condition of selected species.	High	Generic laboratory research Field research (as in 2, a-b)	5 10	Concurrent with 2, a-b Long-term observations dependent on results of 2, a-b
d. Monitor the population dynamics of selected susceptible colonies of marine birds, turtles and mammals.	Potential	As in 2, b	10+	Long-term, low-level effort coupled with 2, b and 3, a.

Table 1. continued

STUDY APPROACHES	FEASIBILITY[1]	REGIONAL FOCUS	DURATION (YEARS)	SEQUENCING
e. Determine relationship between biochemical and physiological indicators of stress and population-level responses.	Limited	Generic research	5	Following progress of 3, a, c and d
f. Evaluate the consequences of long-term effects on populations to living resources and ecosystem support of these resources.	Limited	Generic research	1	Initial models based on results from 1, a-b; 2, a-b; 3, a and c
			2	Refinement of models based on all results

[1]Feasibility of approach judged as "High" (can be satisfactorily accomplished within a 10-year time frame using available methods); "Potential" (requires development of methods or innovative approaches); and "Limited" (probably infeasible within a 10-year time frame).

(Boesch, et al., 1985)

Mississippi Sound and Mobile Bay

We notice a shortage of investigations of fish abnormalities and a lack of species diversity and richness studies in Mississippi Sound and in Mobile Bay. We believe data from such studies are needed. We recommend that prevalence of fish abnormality and benthic community data be tested for possible relation to organic pollution level in tissues of the fish and benthos and to organic concentration in sediments of their habitat.

In Mississippi Sound levels of organic contaminants in economically important mollusks and finfish should be regularly monitored.

We found no reports of sediment toxicity studies of Mobile Bay. We recommend that a comprehensive study similar to the Lytles' four year study of Mississippi Sound be conducted for Mobile Bay. The study should include measurement of organic concentrations in sediments, sediment toxicity, and location of biological communities possibly in danger of distress by anthropogenic pollution. We suggest that concentrations of individual aromatic hydrocarbon compounds be detected and reported to allow comparison with data from studies of other estuaries. We also recommend that the sediment toxicity studies include investigations of sublethal effects (disturbances of respiration, reproduction, etc.) of sediments on local biota.

Couch (1985) suggested that perodical studies of the health of Gulf Coast biota should be made since "the Gulf Coastal Plain will probably be the fastest growing region in the USA in the next decade..."

Chesapeake Bay

The Chesapeake Bay and particularly certain of its sub-estuaries have been adversly affected by both pesticides and PAHs.

Additional studies are needed in several areas with regard to PAH contamination in the Bay. They include

(1) studies to define the levels of sediment contamination necessary to cause both acute and chronic effects on fishes and shellfish;

(2) surveys of the Bay and tributary streams to determine whether increases in incidences of abnormalities in fishes are related to PAH contamination; and

(3) laboratory investigation to determine potential synergism between metals and PAHs.

In the area of pesticides studies are needed to

(1) determine the acute and chronic toxicity of tributyltin to a variety of estuarine organisms;

(2) measure the degradation rates which govern the fate of TBT in the natural environment;

(3) measure bioaccumulation rate constants for TBT, specifically in oysters and clams; and

(4) determine the distribution of organotin compounds in estuarine water and sediments.

Continued monitoring of Kepone levels in the James River is necessary to

(1) protect public health; and

(2) provide a long term record, as an example of the fate of a non-degradable pesticide in an estuarine system.

Delaware Bay

The Delaware River Basin Commission (1984) emphasized that the available data concernig toxic compounds in the water, sediments, and biota are limited. We believe that more samples need to be taken at more locations, particularly including sites close to shore and in small embayments. We notice an apparent lack of recent data for organic contaminant concentrations in tissues of mollusks from Delaware Bay that shoud be remedied.

We also recommend that parameters monitored include PAH, PCB, pesticides, and dioxin concentrations. It seems especially important to routinely analyze samples for PAHs, considering the carcinogenic nature of some of them, and the potential for finding them in the Delaware River Estuary and Bay which are heavily used for navigation and petroleum refining.

New York Harbor and Raritan Bay

The authors agree with recommendations of Pearce (1979a) and Wolfe (1982).

Pearce (1979a) said that "steps must be taken to reduce sewage discharge and terrestrial runoffs along the length of the major streams and their tributaries carrying wastes to the estuary." He additionally recommended resurrection of marshlands, as their brackish water vegetation can effectively filter organic matter (as well as inorganic suspended matter, other contaminants, and nutrients).

Wolfe, et al. (1982) recommended that community characteristics, especially for benthos, be monitored at selected stations to measure long term effectiveness of whatever management practice is selected and instituted.

The authors also support continued monitoring of sediments and biota in the area, but particularly encourage vigilance in monitoring effluents from industries to prevent future extensive damage to natural resources such as the long term degradation that resulted from PCBs in the effluents of two capacitor manufacturing plants in the upper Hudson.

Narragansett Bay

Quinn (1979) recognized the serious nature of chronic input of anthropogenic hydrocarbons to the water, sediments and biota of the Narragansett Bay estuary from sewage effluents and recommended correcting the problem by taking steps to prevent oil from entering the sewage system. One suggestion was institution of a national waste-oil recycling program. As an example of an advantage of this recycling, he reported the University of Rhode Island saved $7,300/month burning a mixture of fuel oil and used crankcase oil in its heating plant. Another suggestion in Quinn (1979) for preventing entrance of oil into the sewage system was adoption of "new and tougher regulations for the effluent discharge of fossil fuel compounds."

Quinn (1979) also recommended establishment of permissable levels of anthropogenic hydrocarbons in shellfish and fish taken for human consumption.

Great Lakes

Contamination levels of both PCBs and DDT in fishes from the Great Lakes have generally declined since the early 1970s. However, closures due to PCB contamination are still in effect for many species (see Fig. 13 in the section on the lakes). With respect to health problems related to consuming these fishes, Sonzogni and Swain (1984) stated:

> "Individuals who habitually consume large quantities of fish from the Great Lakes, especially from Lake Michigan, will have substantially higher intakes of PCBs than the general population. Based on conservative extrapolations from animal studies, such individuals face an increased, although relatively small, risk of developing cancer. More information, such as may be provided by ongoing epidemiological studies of subpopulations exposed to high PCB levels, is needed to better define the associated risks. Further, more information is needed on the possible health effects of PCBs transferred from mother to infant, as well as on possible synergistic effects of PCBs."

> "Despite concern over potential health effects of contaminants found in the Great Lakes, the fact that levels of contaminants such as PCBs and DDT are decreasing is cause for guarded optimism. It would appear that control measures for these compounds are beginning to be effective. As environmental levels of contaminants decrease, the health risk associated with these contaminants should also decrease. Optimism aside, it is clear

that our knowledge of the chronic human health effects of Great Lakes contaminants is minimal, and the need for careful and innovative health effects research will continue into the foreseeable future."

With regard to persistent toxic substances the Water Quality Board of the International Joint Commission in 1985 recommended the following:

PERSISTENT TOXIC SUBSTANCES

Atmospheric Contaminants

The Water Quality Board recommends that:

1. The Great Lakes jurisdictions explicitly recognize the effects of air emissions on water quality in general, and on Great Lakes water quality in particular, in the formulation of air quality statutes, regulations, and standards.

2. The Parties standardize sampling and analytical methods for the United States and Canadian atmospheric deposition monitoring networks, place greater emphasis on gathering organic pollutant deposition data, and maintain the resulting data in a computerized data base.

3. The Parties continue to improve mathematical models which could be used to describe the transport, deposition, and fate of atmospheric pollutants in the Great Lakes Basin.

Groundwater Contamination

The Water Quality Board recommends that:

4. The Parties undertake programs to establish the impact of contaminated groundwater on the surface waters of the Great Lakes Basin ecosystem.

In-Place Pollutants

The Water Quality Board recommends that:

5. The Parties, through appropriate research and demonstration efforts, catalogue and evaluate the effectiveness of alternative approaches for the isolation, removal, treatment, and/or disposal of in-place pollutants.

Investigation of Sublethal Effects

The Water Quality Board recommends that:

6. The Parties support the continuing development and application of suitable diagnostic techniques for the assessment of potential sublethal effects of Great Lakes contaminants on the health of fish and other aquatic organisms in the Great Lakes ecosystem.

Toxicity of Alkyl Lead Compounds

The Water Quality Board recommends that:

7. The jurisdictions continue to monitor lead concentrations in the edible portion of fish, at locations where inputs are known to exist, in order to establish the potential for human exposure.

8. The Parties establish consumption guidelines for lead in fish, in order to protect human health.

SURVEILLANCE

9. The jurisdictions maintain or enhance, where appropriate, funding of surveillance programs which are designed to determine compliance with Agreement objectives, elucidate trends in water quality, evaluate effectiveness of remedial programs, and identify emerging or previously unrecognized problems.

10. The jurisdictions place greater emphasis on the use of atmospheric and biological data in the assessment of water quality and the elucidation of trends.

11. The jurisdictions utilize to the fullest extent, water intake data and other water quality indicators such as beach closures and fish consumption warnings, for tracking long-term water quality trends.

12. The jurisdictions report incidences of fish tumors as part of Great Lakes surveillance programs, and expand research to definitively identify the causative factors.

13. The jurisdictions make every effort to achieve data compatibility among jurisdictional monitoring and surveillance programs.

14. Because dieldrin levels in herring gull eggs and certain fish in the Great Lakes Basin have not responded significantly to remedial measures, the jurisdictions should direct research efforts to identify all sources of dieldrin, determine the relative contribution of these sources to the current levels found in the Great Lakes biota, and understand the environmental dynamics of this substance.

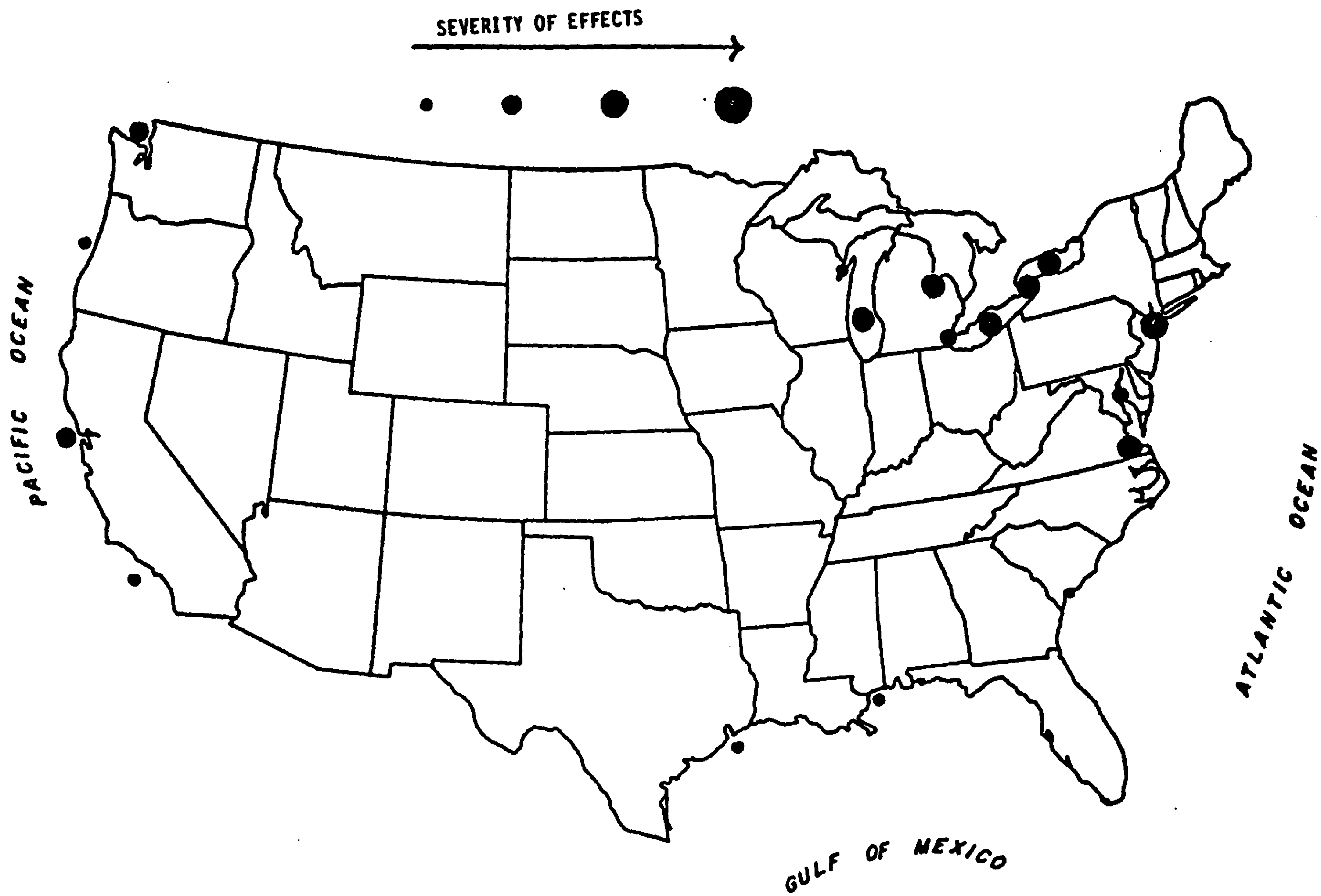


Figure 2. Relative severity of effects.

PUGET SOUND

Puget Sound, a huge arm of the sea which breaks into the northwest corner of the continental U.S., contains many embayments. Some of these are surrounded by heavily populated metropolitan areas such as Seattle and Tacoma in which heavy industry, international shipping traffic, and municipal wastewater, all contribute to its pollution. Also along the shores of this estuary the cities of Bremerton, Olympia, Everett, and others contribute domestic wastes. Other open areas of the sound and many northern areas are relatively pristine. Puget Sound supports anadromous salmonid populations and other important commercial and recreational fishery resources. Figures 1 and 2 show Puget Sound indicating areas that will be discussed in this summary. Investigators considered three of the labeled sites, Presidents Point, Case Inlet, and Port Madison clean enough to use as reference sites.

In recent years scientists have studied Puget Sound intensively for contaminant levels and their impact. EPA, NOAA, USACOE, and METRO, have sponsored research and for the most part reputable government scientists and private consulting firms have conducted this research. They have used three main methods of studying effects of contaminants: (1) fish histopathology, (2) biossays with contaminated sediments and (3) benthic community changes (Ed Long, personal communication, June 28, 1985). The combined reports of these recent studies cover the entire area very well. These studies taught us much about the Sound and about techniques of sampling, analysis, and effects studies in general.

Many investigators reported measurements of PAHs or aromatic hydrocarbons in general from Puget Sound sediments (Anderson, 1985; Anderson and Crecelius, 1985; Barrick, et al., 1985a; Battelle, PNL, 1985 Draft; Crecelius, et al., 1985; Crecelius, Bloom, and Gurtisen, 1984; Dexter, 1981; Galvin, et al., 1984; Hileman and Matta, 1983; Konasewich, et al., 1982; Krahn, et al., 1986; Long, 1983; Long, 1982; Long, 1981; Macleod, et al., 1982; Malins and Roubal, 1985 in Press; Malins, et al., 1985 a., b. & c.; Malins, et al., 1984 a., b., c.; Malins, et al., 1982; Malins, et al., 1980; McCain, et al., 1982; Pavlou and Weston, 1984; Riley, et al., 1980; Romberg, et al., 1984; Tetra Tech, Inc., 1985; USACOE, 1985; Varanasi, et al., 1985). Only a few give values for PAHs in the water column as well.

PAHs in Water and Sediments of Everett Harbor, Elliott Bay and Duwamish Estuary

Anderson (1985) reported total aromatic hydrocarbons, mostly PAHs, in sediment samples from eleven stations representing eight potential Everett disposal sites averaged ~10,015 ppb (dry wt.). This included one completely

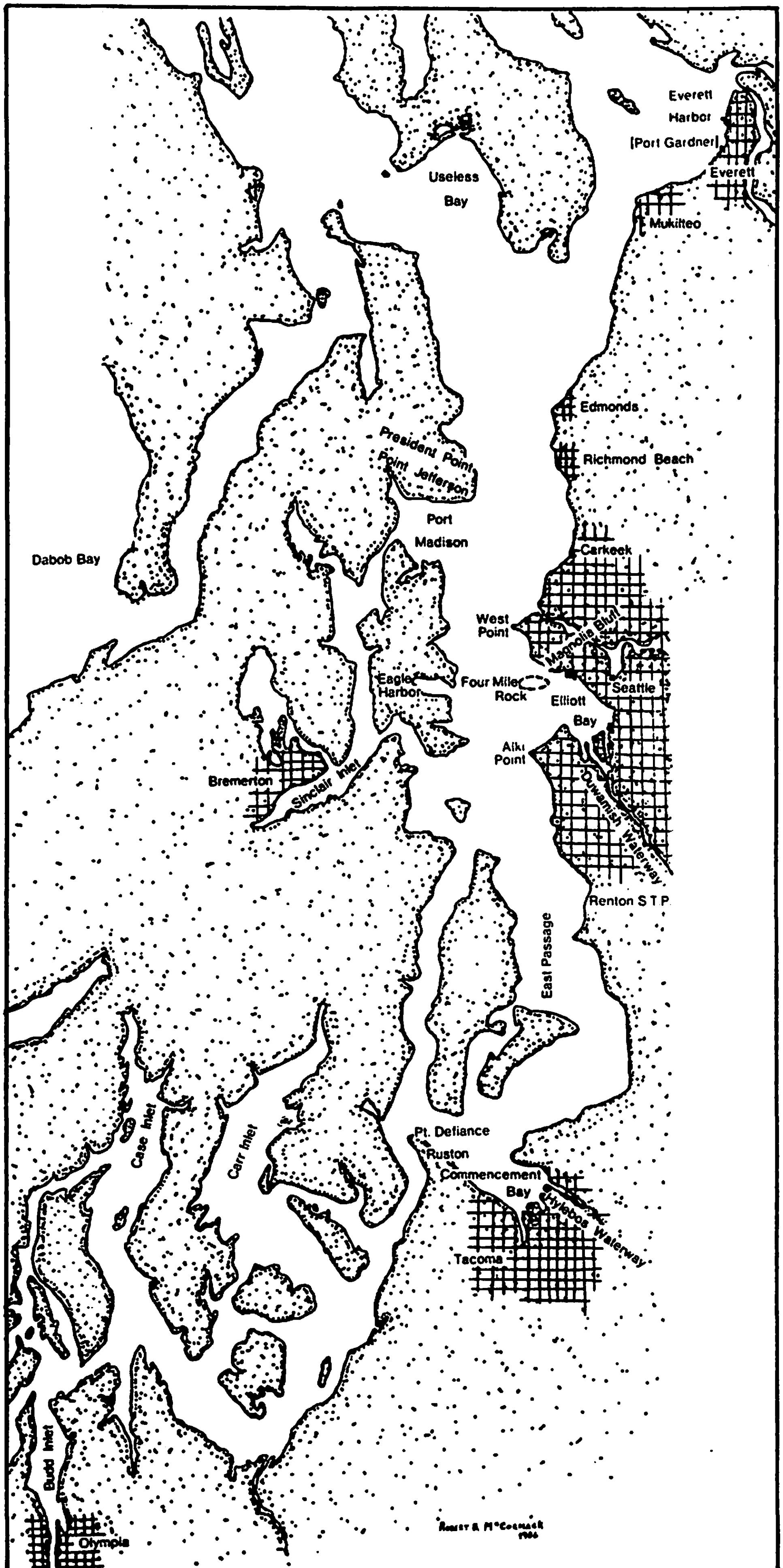


Figure 1. Puget Sound

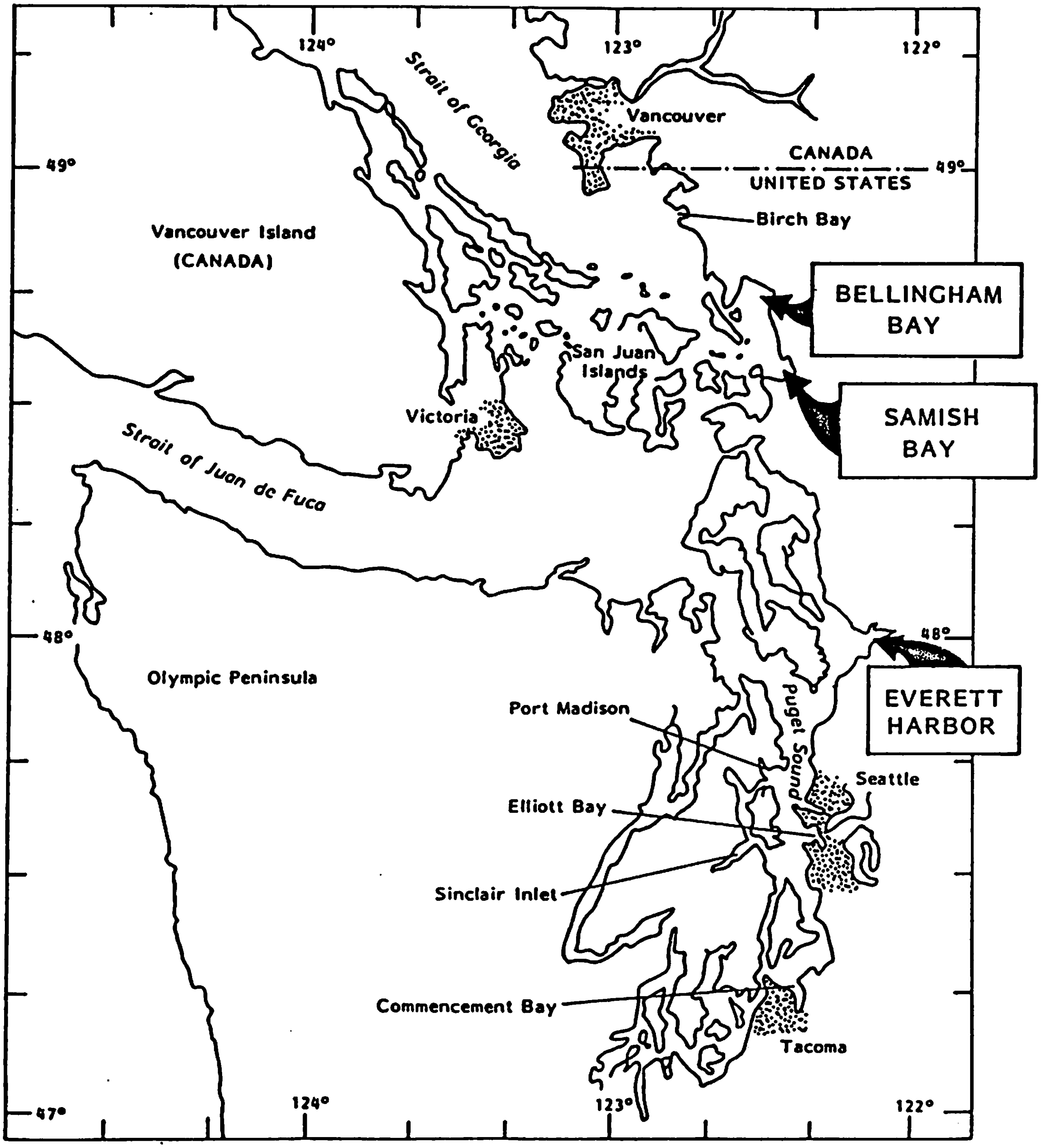


Figure 2. Puget Sound and Straits of Georgia
and Juan de Fuca (Chapman, et al. 1984b).

clean station. In the same study the average total aromatic hydrocarbons in sediments from six Everett East Waterway sediments was 4,000 - 5,000 ppb (dry wt.). Battelle scientists compared PAH concentrations found in sediments analyzed in this study with concentrations from their two earlier studies in 1985 which averaged ~1,336 ppb (dry wt.). They explained the higher concentrations by the difference in sampling depths. Average sample depth in this recent study was 106.6 cm. as opposed to 132 cm. for the core samples from their two studies earlier in 1985. They reported the total PAH concentration as an inverse function of the average sampled depth.

Anderson and Crecelius (1985) in Battelle's Pacific Northwest Research Laboratory final report to USACOE, Seattle District for the U.S.Navy considered levels of PAH in East Waterway sediments of Everett Harbor elevated. They presented results of analysis in the East Waterway showing a range from 151,000 - 250,000 ppb (dry wt.) in the most contaminated areas near loading docks.

In Volume I of Battelle, PNL, 1985 Draft final report of their eight bays study - four urban, four baseline - total aromatic hydrocarbon concentrations in sediments varied. Most baseline bays sediments did not contain detectable aromatic hydrocarbons; three contained 60 to 334 ppb. (One or more samples from each of three of the urban bays) contained concentrations of more than 10,000 ppb. One Elliott Bay station had 25,000 ppb aromatic hydrocarbons in sediment samples.

Averages of PAH (2 ring - 5 ring) concentration values from two East Waterway's core samples analyzed and reported by Crecelius, et al. (1984) are as follows for various core depths:

0- 5cm.	=	24,155 ppb
10-15cm.	=	23,481 ppb
20-25cm.	=	23,583 ppb
30-35cm.	=	16,763 ppb
40-45cm.	=	16,170 ppb

The highest PAH concentration appeared at drastically different depths in each of these two cores. For their sediment core #29 the highest concentration of PAH was 49,908 ppb found at 55-60cm.; for core #30 the highest was 258,790 ppb for total PAHs at 115-120cm.

In the METRO study reported by Galvin, et al. (1984) PAH levels in shallow Elliott Bay/Duwamish sediments ranged 0-13,600 ppb (dry wt.), with a mean of 1,300 ppb. In Figure 29 of their summary report several "hot spots" for concentration of high molecular weight (combustion) PAHs in surface sediments of Puget Sound stand out. Elliott Bay waterfront and Duwamish West Waterway each contain sites with over 24,000 ppb (dry wt.).

Both Malins, et al. (1982) and Long (1982) in describing results of the Marine Ecosystems Analysis (MESA) Puget Sound project report a mean concentration of 13,000 ppb (dry wt.) of total aromatic hydrocarbons in Elliott Bay sediments; the highest value measured there was 63,000 ppb.

In their earlier report concerning the MESA Puget Sound Project, Malins, et al. (1980) separate values for PAHs in sediments into concentrations of 1- and 2-ring aromatic hydrocarbons and concentrations of 3- to 5-ring aromatic hydrocarbons. They also show separate values for each of the three carcinogenic PAHs, phenanthrene, benz[a]anthracene, and benzo[a]pyrene. (All sediment samples contained) all three of these PAHs. Samples were taken from twelve locations in central Puget Sound. They found that concentrations of PAHs varied a great deal within subareas, but average PAH concentrations from the subareas of Elliott Bay (Duwamish Waterway, Seattle Waterfront, and West Point), Commencement Bay (Hylebos Waterway, Commencement Waterways and SW Commencement Bay), and Pt. Herron station of Sinclair Inlet were much higher than those of other subareas.

MacLeod, et al. (1982) compared results of Duwamish sediment analyses by 13 laboratories. They showed values for 18 selected aromatic hydrocarbons. The mean of the means from the various laboratories was highest for the PAH fluoranthene [1,700-1,800 ppb (dry wt.)] in subtidal sediment.

Among compounds they found in sediments of the Duwamish Waterway and two other Puget Sound estuaries, McCain et al. (1982) gave concentrations for ten individual PAHs. Of these the concentrations are highest for fluoranthene, pyrene, and benz[a]anthracene in the three estuaries. The highest concentrations by far were from the Duwamish Waterway sediments: floranthene, 1,200 ppb (dry wt.); pyrene and benz[a]anthracene, 1,000 ppb (dry wt.) each.

In the Romberg, et al. (1984) study, partly because the water column was so clean that detection and quantification of pollutants was difficult, they collected sediment samples and "the most complete set of sediment chemistry data that exists for central Puget Sound" resulted. They show "chemical hot spots" via contour maps which define regions with similar sediment concentrations. The three main hotspots they identify are the old North Trunk sewer outfall, north of Seattle at West Point; the Four Mile Rock dredge spoils disposal site; and inner Elliott Bay and the lower Duwamish. Their contour maps of both high molecular weight, or "combustion" PAHs and of low molecular weight PAHs show distinctively high values at these places: nearshore at the old North Trunk sewer outfall, the combined sewer outfall (CSO) off Magnolia Bluff, the entire waterfront in Elliott Bay and the lower Duwamish, and two sites off Elliott Bay (one near Four Mile rock disposal site).

A study was conducted under the auspices of the United States Army Corps of Engineers, USACOE, to provide the U.S. Navy with data from sediment analyses to assist in making plans for the design and construction of its proposed homeport at the East Waterway of Everett Harbor that would minimize possible adverse impacts of disturbing contaminated sediments there. Battelle's Pacific Northwest Laboratory sampled sediments in July, 1984 for the Corps of Engineers and made suggestions for further study about dealing with the contaminated dredged material that would result from the proposed construction (UASCOE, 1985).

Prior to their bioaccumulation studies, Varanasi, et al. (1985) analyzed and reported concentrations of aromatic hydrocarbons found in Duwamish River Delta sediments. They combined values for aromatic hydrocarbons with the same number of benzoid rings and reported the following:

2-ring AHs -	- 600 ppb ± 100 (wet wt.);	3,000 (approx. dry wt.)
3-ring AHs -	- 2,000 ppb ± 300 (wet wt.);	10,000 (approx. dry wt.)
4-ring AHs -	- 6,900 ppb ± 1100 (wet wt.);	34,500 (approx. dry wt.)
5-ring AHs -	- 5,300 ppb ± 600 (wet wt.);	26,500 (approx. dry wt.)
6-ring AHs -	- 1,400 ppb ± 400 (wet wt.);	7,000 (approx. dry wt.)

As in other studies (MacLeod, et al., 1982; McCain, et al., 1982) concentrations of 4-ring aromatic hydrocarbons - totals of fluoranthene, pyrene, benz(a)anthracene, and chrysene concentrations - are higher in sediments than other AHs.

PAHs in Water and Sediments of Commencement Bay

Barrick, et al. (1985a) detected PAHs in Commencement Bay sediments, with the highest concentration found in the Hylebos Waterway finger of the Bay.

According to Crecelius, et al. (1985) concentrations of aromatic hydrocarbons in age-dated sediment cores indicate the period of maximum contamination of Commencement Bay sediments was as early as 1900. They calculated concentrations of aromatic hydrocarbons to be approximately 10 times higher in bay sediments than in sediments from nonindustrial sites outside the central basin of Puget Sound.

Malins, et al. (1985c) and Malins, et al. (1984 a & b) reported detection of over 900 individual organic compounds in sediments from Commencement Bay. They investigated four urban embayments in Puget Sound and four nonurban ones. Sums of mean concentrations of 25 PAHs (ΣAHs) in sediments for the four "urban embayments . . . were as much as 46 times the mean concentration (0.28 ppm) of ΣAHs in sediments from the nonurban embayments . . ." (Malins, et al. 1984a)

Malins, et al. (1982) reported that a chromatogram from a single sediment sample from Hylebos Waterway, Commencement Bay revealed more than 500 aromatic hydrocarbons. When they summed (ΣAHs) mean concentration of their 27 target aromatic hydrocarbons from sediment analyses, they found the areas with highest ΣAHs were in Elliott Bay, [13,000 ppb (dry wt.)] and in Commencement Bay [9,700 ppb (dry wt.)].

In the Romberg, et al. (1984) study, values for low molecular weight PAHs in sediments from three southern stations towards Tacoma were OOstinctively high.

Riley, et al. (1981) collected samples of water, suspended matter and sediment cores from Hylebos and Blair Waterways, located adjacent to

Commencement Bay. Most core sections contained PAHs in a range of molecular weights, the highest PAH concentrations appearing in one replicate sample from a station located in Commencement Bay between the two waterways. Data in this report suggest that organic compounds have impacted these waterways in the past 25 years and that neither dredging nor biological degradation affect much decrease in the persistence of these compounds in the sediments.

Riley, et al. (1980) examined water and suspended matter from nine sampling stations located in Elliott Bay, Sinclair Inlet, Budd Inlet and Port Madison, their reference area. Of these nine the Hylebos and the Blair Waterways in Commencement Bay appeared to be the most contaminated with organic compounds. Aromatic hydrocarbons from naphthalene (2-ring) to pyrene (4-ring) were present in suspended matter at all stations; higher molecular weight aromatic compounds seemed to be site specific. They identified five or more of nine PAHs on EPA's priority pollutants list in suspended matter from each sampling site.

PAHs in Water and Sediments of Other Puget Sound Locations

Sediments from two sampling sites at Mukilteo, Washington, which is on Puget Sound not far from Everett, contained substantially higher concentrations of aromatic hydrocarbons than sediments from a reference site, President Point in Puget Sound. Aromatic hydrocarbons comprised 7,800 and 33,000 ppb of the sediments from the two sites - one near a sewer outfall and the other adjacent to the fuel tanks (Malins, et al. 1985a).

Figure 8 of Malins, et al. (1985c) showed mean concentration of chemicals in sediments from various areas of Puget Sound. Values of ΣAHs from Mukilteo are highest, followed by the next highest: Commencement Bay's Hylebos Waterway, then the Duwamish Waterway in Elliott Bay, then Everett.

Krahn (1986) found concentrations of aromatic hydrocarbons in sediments from eleven sites in Puget Sound. They found the highest concentrations [310,000 ppb (dry wt.)] in Eagle Harbor. Sediment samples from the Duwamish, Clinton, inner and outer Everett Harbor, Richmond Beach, West Point, Carkeek, President Point, Edmonds, and Useless Bay were also analyzed.

Malins and Roubal (1985) reported the presence of free radicals derived from aromatic hydrocarbons in sediments from Eagle Harbor. Malins, et al. (1985b) attributed high concentrations of aromatic hydrocarbons in sediments from the three Eagle Harbor sites they studied to heavy creosote pollution. Differences among concentrations from the three sites and "high standard deviations in the mean concentrations of individual compounds of these sites, demonstrate . . . 'patchiness', of creosote in the harbor."

When Romberg, et al. (1984) compared concentrations of toxicants from deep ocean water at Pillar Point in the Strait of Juan de Fuca with concentrations from Point Jefferson in Puget Sound, they detected very few organic compounds, including some PAHs, in particulates at either location, and saw even fewer in dissolved water samples. As most would expect, they saw more toxicants in the Puget Sound samples. Ambient water concentrations of PAHs (averaging about 55 pptr) were low when compared to lowest reported chronic or acute toxic concentrations.

Riley, et al. (1983) found an average of 346 ppb (dry wt.) of PAHs (2-ring naphthalene - 5-ring perylene) in sediments sampled from southern and northern Case Inlet in southern Puget Sound.

PCBs and Pesticides in Water and Sediments

Using their equilibrium partitioning approach on contaminant concentrations in Puget Sound, Pavlou and Weston (1984) found among synthetic organics only PCBs and DDT consistently exceeded their proposed sediment criteria in urban embayments.

Cline, et al. (1979) did not consider pollution a problem in Puget Sound at that time, but they did consider monitoring certain variables important "to identify and quantify early signs of deterioration" A monitoring program would require a good baseline survey. "Monitoring for concentrations of pollutants is especially attractive since this approach depends only on chemical analytical methodology, which is often more reliable and reproducible than biological methods."

Long (1981) reported that in the Marine EcoSystems Analysis (MESA) Project the range of PCBs detected in 42 Puget Sound sediment samples was .5 - 1,200 ppb (dry wt.).

Malins, et al. (1984a) usually found concentrations of PCBs much higher in major urban embayments than in nonurban ones, but concentrations varied within urban embayments. Concentrations of the pesticides lindane, heptachlor, aldrin, α-chlordane, and trans-nonachlor were usually <2 ppb and DDT and its derivatives <10 ppb in urban embayments they studied.

Malins, et al. (1980) summed the concentrations of isomers of PCBs they detected in sediment samples from various locations in Puget Sound. Their results in order of average PCB concentrations by location follow:

Subarea	ppb (dry wt.) PCBs in Sediments
Hylebos Waterway	500
Duwamish Waterway	300
Seattle Waterfront	300
Outer Elliott Bay	200
Commencement Waterways	100
Sinclair Inlet	100
West Point	60
Southwest Commencement Bay	40
Budd Inlet	10
Port Madison	6
Case Inlet	2
Brown's Point	2

PCBs and Pesticides in Water and Sediments of Everett Harbor, Elliott Bay, and Duwamish Estuary

Anderson (1985) sampled sediments from eleven stations at potential disposal sites in the East Waterway of Everett Harbor. They also collected samples to form six East Waterway native sediment composites. Analysis revealed below detection level PCBs (Arochlor 1254) in three of the six native sediment composite samples and in two of the eleven samples from proposed disposal sites. He reported an average of 11 ppb (dry wt.) in the three native sediment composite samples containing detectable levels of PCBs and an average of 103 ppb (dry wt.) PCBs in the nine proposed disposal site samples containing detectable levels. The one potential disposal site composite that contained the highest value for PCBs only contained 307 ppb. These levels are well below the 610 ppb found at Fourmile Rock disposal site.

Anderson and Crecelius (1985) reported that concentrations of PCBs (Arochlor 1254) were scattered in East Waterway, Everett Harbor sediments. Concentrations were highest in the upper region of the waterway around loading docks. They found the highest concentration of PCBs [717 ppb (dry wt.)] in the top section of the core samples from one of the stations. In some other samples Arochlor 1254 was not detectable.

In Volume I of Battelle, PNL, 1985 Draft final report of their eight bays study, both Table 18. on page 87 and Figure 29. on page 77 show relative rank of PCB 1254 loadings in the eight bays. This implicated Sinclair Inlet as the most adversely impacted with PCBs; next was Elliott Bay - Fourmile Rock, followed by the other two urban bays: Everett Harbor - Port Gardner, then Bellingham. The highest concentration found in a Sinclair Inlet sediment station was about 1200 ppb (Battelle, PNL, 1985, Vol. 2). They detected no PCBs in any of the four baseline bays.

Crecelius, et al. (1984) analyzed in triplicate sediment samples from two East Waterway, Everett core samples. They detected less than 10 ppb (dry wt.) PCBs (Arochlor 1254) in bottom sections of each core. The highest concentrations of PCBs they found was 1610 ppb from the center section (95 - 100 cm depth) of one of their core samples. Average concentration of PCBs they discovered in upper and center core sections was 795 ppb (dry wt.). These and other chemical data from the two Everett core samples "indicate that 20 to 30 cm of contaminated sediments are accumulating per year."

Malins, et al. (1985c) found the mean concentration of PCBs in sediments of Seattle's Duwamish Waterway was >100 times greater than in their reference area sediments from Port Madison and Case Inlet. Malins, et al., 1984a. sampled sediments in four major urban embayments - Port Gardner, Elliott Bay, Commencement Bay, and Sinclair Inlet. They reported highest mean concentrations of PCBs (380 ppb) in Elliott Bay sediment samples.

Data available to Galvin, et al. (1984) showed PCB levels in Elliott Bay sediments to be higher than in many other areas of the country. Their own reported data included a mean value of 670 ppb (dry wt.) of PCBs in shallow Elliott Bay/Duwamish sediments and a mean of 120 ppb (dry wt.) in the deep central basin. They also reported pesticide data in ppb (dry wt.). Their mean value for DDTs in shallow Elliott Bay/Duwamish sediments was 20, and in deep central basin, <2. Mean value of other misc. pesticides in shallow Elliott Bay/Duwamish sediments was 11 and in deep central basin, 7.

Analysis of Four Mile Rock sediment as reported by USACOE (1985) showed the sum of PCB 1016, 1232, 1242, 1248, 1254, and 1260 as 610 ppb and the sum of DDD, DDE, and DDT as 7 ppb.

Malins, et al. (1982) reported 540 ±83 ppb (dry wt.) PCBs in Duwamish River Waterway sediments, but only 4.2 ±2.9 ppb (dry wt.) in Case Inlet/Port Madison sediments. Analysis of water by Pavlou and Dexter (1979) disclosed very low (near detection) levels in the Duwamish River and in Elliott Bay.

In describing spacial distribution of PCBs in Puget Sound, Pavlou and Dexter (1979) noted that generally values correlated well with areas of increased industrial and municipal activity. For example, they observed that water, suspended particulate matter and sediments in the Duwamish River Estuary contained the highest PCB concentrations in Puget Sound. They observed no significant horizontal or vertical gradients of PCBs in most regions; but in the Elliott Bay - Duwamish River system they did see distinct gradients which correlated well with the distribution of the brackish layer of the river.

PCB concentrations have increased by a multiple of 13 since 1900. This represents the highest numeric increase among classes of organic compounds which were detected in Puget Sound prior to 1900 (Romberg, et al., 1984). Average of mean values in ppb (dry wt.) of PCBs (Arochlors 1242, 1248, 1254, & 1260) measured in sediments from selected areas by Romberg, et al. (1984) follow:

East Elliott Bay	474
NE Elliott Bay/Denny Way CSO	321
Four Mile Rock (>100m deep)	186
Central Elliott Bay (>100m deep)	185
South Elliott Bay/Duwamish	163
Urban Central Basin, off Elliott Bay (>100m deep)	96
Urban Central Basin, off West Point (>100m deep)	41
North Central Basin (>100m deep) .	41
Old North Trunk Sewer Outfall . . .	37

PCBs and Pesticides in Water and Sediments of Commencement Bay

Barrick, et al. (1985a) found that the sediments in Hylebos Waterway contained the highest levels of chlorinated organic compounds in their project area. The distribution of PCBs there was patchy "with elevated levels occurring throughout subtidal sediments of the waterway." Pesticides such as aldrin, lindane, and DDT that had been previously identified at elevated levels in certain areas of Commencement Bay were apparently absent or low concentrations were found in samples analyzed this study. "The only pesticide detected and confirmed by GC/MS identification was DDT. . . @ 50 ppb (dry wt.) in the City Waterway sediments of Commencement Bay" (Barrick, et al., 1985b).

Average dry wt. concentrations of sums of chlorinated pesticides (lindane, aldrin, heptachlor, -chlordane, nonachlor, and the DDT family of compounds) were highest in Commencement Bay sediment samples of subareas examined by Malins, et al. (1980). Concentrations from Hylebos Waterway were 80 ppb; from Commencement Waterways, 30 ppb; and from Southwest Commencement Bay, 40 ppb.

Of eight areas selected by Malins, et al. (1985c) concentrations of PCBs in sediments were highest in Commencement Bay. Long (1982) reported that results of the Marine EcoSystems Analysis (MESA) of Puget Sound water and sediments at that stage indicated particularly high PCBs at the mouth of Hylbos Waterway (Commencement Bay). Riley, et al. (1981) also reported highest concentrations of chlorinated biphenyls in sediments collected near the mouth of the Hylebos Waterway. Malins, et al. (1982) report a mean concentration of 270 ppb (dry wt.) for PCBs in Commencement Bay sediments.

PCBs and Pesticides in Water and Sediments of Other Puget Sound Areas

Malins, et al. (1985b) reported relatively low dry wt. concentrations of PCBs and other chlorinated organic compounds in sediments from three Eagle Harbor sites they sampled (44 ±14, <6 ±3, <5 ppb). In southern Puget Sound, Riley, et al. (1983) did not find PCBs at measureable concentrations at most locations, including Case Inlet stations. Battelle (1985 Draft) implicated Sinclair Inlet as the bay most heavily loaded with PCBs of the eight bays in their study.

PAHs, PCBs, and Pesticides in Biota

Interest in levels of organic contaminants in sediments and water is unjustified if these organics do not accumulate or bioconcentrate in tissues of aquatic biota and perhaps affect them in some way that might eventually affect human health, or economic well-being. You will find evidence of these relationships discussed in a large number of the studies listed in the attached annotated Puget Sound bibliography.

Presence of organic contaminants in tissues of Puget Sound organisms is well documented. PAH and PCB concentration data gathered from sampling and tissue analysis in a number of Puget Sound studies is given in Tables 1 and 2.

Table 1. PAH, PCB and pesticide concentrations in Puget Sound invertebrates in ppb (dry wt.).

Organism	PAH	PCB	Pesticide	Source
Clam				
Duwamish Waterway	5,300 (ΣAH)	312	.4**	2.
	1,600 "	1,300	undetectable**	
Hylebos Waterway	5,100 "	874	10**	
	1,300 "	340	130**	
Case Inlet/Port Madison	160 "	24	undetectable**	
		160	undetectable**	
Clam				
Duwamish Waterway	4,900 (3-5 ring)	300	.4**	4.
Hylebos Waterway	1,000 "	300	130**	
Port Madison	200 "	20	<.3**	
Shrimp				
Duwamish Waterway	720 (ΣAH)	2,100	2**	2.
Hylebos Waterway	1,300 "	3,800	80**	
Case Inlet/Port Madison	150 "	300	1**	
	1,500 "	134	undetectable**	
Crab hepatopancreas				
Duwamish Waterway	2,000 "	32,000	30**	
Hylebos Waterway	1,900 "	28,000	1800**	
Case Inlet/Port Madison	15 "	480	2**	
Worm				
Duwamish Waterway	7,400 "	1,800	2**	
Hylebos Waterway	17,000 "	1,220	140**	
	1,200 "	380	370**	
Case Inlet/Port Madison	200 "	190	undetectable**	
Shrimp				
Duwamish Waterway	500 (3-5 ring)	2,100	2**	4.
Hylebos Waterway	1,200 "	3,000	80**	
Case Inlet/Port Madison	1,400; 100 "	100; 300	<.3; 1	
Crab hepatopancreas				
Duwamish Waterway	500 (3-5 ring)	33,000	30**	4.
Hylebos Waterway	900 "	28,000	1,800**	
Case Inlet	undetectable	400	2**	
Sinclair Inlet	200 (3-5 ring)	4,200	2**	
Worm				
Duwamish Waterway	6,500 "	1,800	2**	
Hylebos Waterway	8,400 "	800	260**	
Port Madison	200 "	200	<1**	
Crab hepatopancres				
Duwamish Waterway		32,000		3.

Table 1. Continued.

Organism	PAH	PCB	Pesticide	Source
Zooplankton and benthic shrimp				
Duwamish River	480 (3-5 ring)	2,050		5.
Hylebos Waterway	1,210 "	3,054		
Sinclair Inlet	90 "	680		
Case Inlet	1,380 "			
Case Inlet/Port Madison		219		
Crab, edible tissue				
Elliott Bay	600*	375*	25* (DDT)	1.
Zooplankton, lipid				
Elliott Bay		6,790		6.
Zooplankton				
Elliott Bay		6,800 ± 3,100		7.
Sinclair Inlet		16,000 ± 7,000		

1. Romberg et al. 1984.
2. Malins et al. 1982.
3. Long 1982.
4. Malins et al. 1980.
5. Dexter et al. 1981. (based on data from Malins et al. 1980.).
6. Cline et al. 1979.
7. Pavlou and Dexter. 1979.

* Wet weight converted to dry weight. Dry wt. = .20 (wet wt.).

** hexachlorobenzene

Table 2. PAH, PCB and pesticide concentrations in Puget Sound finfish in ppb (dry wt.).

Organism	PAH	PCB	Pesticide	Source
English sole, liver				
Mukilteo	undetectable	3,400		2.
President Pt.	68 (napthalene)	1,000		
English sole, liver				
Eagle Harbor	97 (10, 2-3 ring)	1,100		3.
President Pt.	68 (napthalene)	1,000		
English sole, muscle				
Hylebos Waterway	undetectable	3,400	170 ± 130**	6.
Duwamish Waterway	undetectable	4,800 ± 3,400	8 ± 9**	
English sole, liver				
Hylebos Waterway	undetectable	39,000	1,100 ± 520**	6.
Duwamish Waterway	undetectable	47,000 ± 25,000	35 ± 17**	
Case Inlet/Port Madison	undetectable	2,300 ± 700	10 ± 0**	
English sole, liver				
Hylebos Waterway	675 (ΣAH)	14,000	1,735**	8.
Duwamish Waterway	undetectable	12,500	10**	
English and Rock sole composite livers				
Hylebos Waterway	400 (3-5 ring)	11,000	990**	11.
Duwamish Waterway	100 "	17,000	5**	
Case Inlet	50 "	1,100	7.5**	
English sole, liver				
Duwamish Waterway		35,000		9.
English sole, bile				
Duwamish Waterway	1,400 ± 2,200 (BaP)			4.
Eagle Harbor	2,100 ± 1,500 "			
Everett inner harbor	520 ± 410 "			
Everett outer harbor	270 ± 220 "			
President Pt.	100 ± 90 "			
Useless Bay	67 ± 45 "			
English sole, liver				
Duwamish Waterway		17,500		10.
English sole, liver				
Duwamish River	110 (3-5 ring)	35,290		12.
Hylebos Waterway	430 "	15,140		
Case Inlet	50 ± 490	1,600 ± 3830		
Sinclair Inlet	not quantifiable	8,550		
English sole, bile				
Duwamish	986 (some metabolites)			7.

Table 2. Continued.

Organism	PAH	PCB	Pesticide	Source
Sole, edible tissue				
Elliott Bay	undetectable	58*	1.4* (DDT)	5.
Cod, edible tissue				
Elliott Bay	undetectable	36*	1.2* (DDT)	5.
Salmon, edible tissue				
Elliott BAy	94*	270*	2.6* (DDT)	5.
English sole, muscle				
Hylebos Waterway		1,660*	undetectable	1.
Commencement Bay @ Tacoma		535*	undetectable	

1. Barrick et al. 1985b.
2. Malins et al. 1985a.
3. Malins et al. 1985b.
4. Krahn et al. 1985.
5. Romberg et al. 1984.
6. Malins et al. 1984a.
7. Burrows et al. 1983.
8. Malins et al. 1982.
9. Long 1982.
10. McCain et al. 1982.
11. Malins et al. 1980.
12. Dexter et al. 1981. (based on data from Malins et al. 1980.).

* Wet weight converted to dry weight. Dry wt. = .20 (wet wt.)

** hexachlorobenzene

Investigators have conducted bioaccumulation studies using sediments from various Puget Sound areas with a variety of organisms. Anderson and Crecelius (1985) placed the top section of sediments from their East Waterway, Everett Harbor stations in aquaria along with clams, Macoma inquinata; in some aquaria they placed mussels, Mytilus edulis, on racks above the sediment. Their control aquaria contained Sequim Bay sediments. Nine of the ten mussels in the tank containing the highest PCB concentration (717 ppb, dry wt.) died. Anderson (1985) in discussing results of this same bioaccumulation study reported that PAH levels were not magnified above sediment level by the clams, but one mussel did appear to show bioaccumulation.

Anderson (1985) also described a second phase of bioaccumulation testing for the U.S. Navy's consideration in planning a homeport facility in Everett Harbor. This experiment lasted 21 days, eight days longer than the earlier one, and results, all reported as dry wt., did indicate biomagnification of PAHs by clams in control sediments (from 200 to 16,130 ppb) and in four of six Everett Harbor composite sediment samples (from 7,000 to 254,800; 2,020 to 5,010; 5,890 to 36,090; and from 5,860 to 19,540 ppb). PCB levels were low in control sediments and in most experimental composite samples as well, but biomagnification of PCBs by clams was evident, though in most sediments just barely. The mussels, however, clearly exhibited bioaccumulation of PCBs. In all six Everett Harbor composite sediments mussels accumulated much higher PCB levels than surrounding sediments:

PCBs in sediments =	< 1 ppb	PCBs in mussels =	1410 ppb
	< 1 ppb		1420 ppb
	< 1 ppb		470 ppb
	12 ppb		70 ppb
	12 ppb		1800 ppb
	9 ppb		1490 ppb

Results also showed biomagnification of PAHs by mussels in control sediments and in four of six experimental composites: from 7,000 ppb in sediment to 20,740 ppb in mussels; from 3,570 to 16,910; from 2,020 to 20,830, and from 5,860 to 87,950 ppb in mussels.

Malins, et al. (1982) demonstrated bioaccumulation of aromatic hydrocarbons and PCBs in clams. They moved clams from a relatively non-polluted area to cages buried in the Hylebos Waterway. The concentration of these contaminants not only increased over clams from the non-polluted area [concentration of ΣAHs in caged clams, 15,000 ppb (dry wt.); concentration of ΣAHs in clams from the same non-polluted area, 970 ppb (dry wt.)] but concentrations "in the caged clams were similar to mean concentrations in the sediments" (Malins, et al., 1984c)

Results of the Varanasi, et al. (1985) study show accumulation of PAHs by invertebrates exposed to Duwamish River delta sediments:

ΣPAHs in	Sediment	Clams	Amphipods: Eohaustorius washingtonianus	Amphipods: Rhepoxynius abronius
2-ring	3000 ± 100	trace	trace;	not detected
3-ring	10000 ± 1650	215 ± 70	480 ± 25	400 ± 30
4-ring	34500 ± 5500	2700 ± 285	18000 ± 950	7000 ± 1200
5-ring	26500 ± 3000	1400 ± 60	4600 ± 600	950 ± 750
6-ring	7000 ± 2000	215 ± 35	not detected;	not detected

Concentrations are shown in ppb (dry wt.), converted from Varanasi, et al. (1985) using dry wt. = .20(wet wt.).

William L. Reichert (Personal interview, June 26, 1985) described the experimental set up for this bioaccumulation study (See Figure 3) and discussed some of his thoughts concerning results of this and similar studies. He noted that various PAHs are bioavailable to different degrees and that species, even the two amphipod species they used, handle benzo(a)pyrene in different ways.

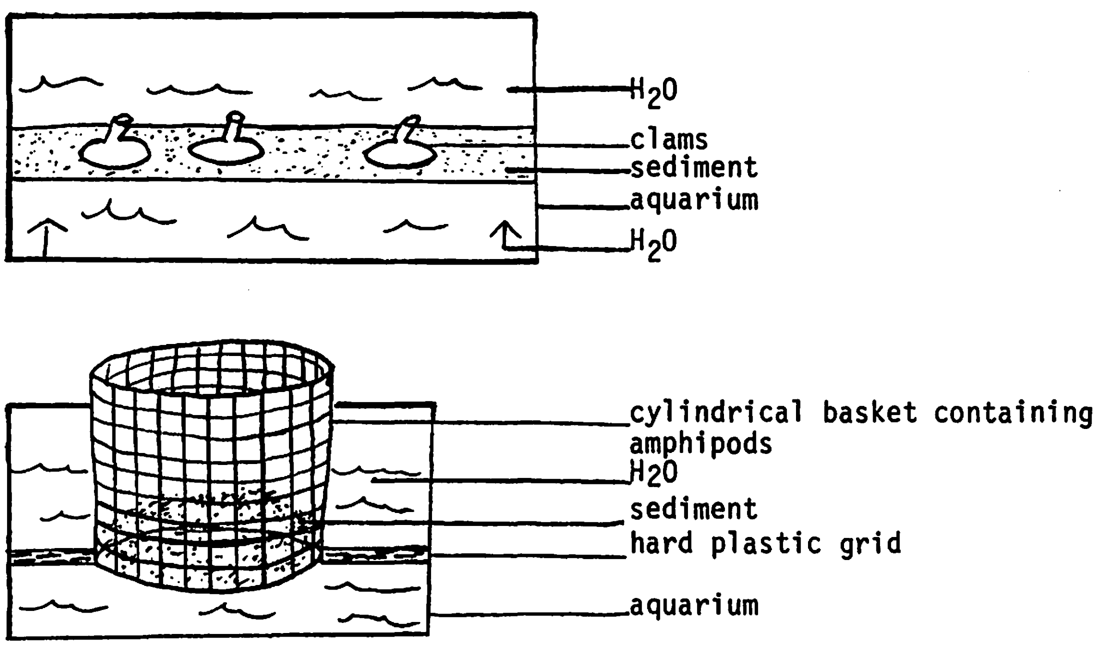


Figure 3. Bioaccumulation Experiment Set Up

Investigators have studied the transport of aromatic hydrocarbons from sediments to finfish in Puget Sound and have either not detected the chemicals in the fish livers from areas containing polluted sediments, or have found them only in trace amounts (Malins, et al., 1985c; Malins, et al., 1984c; Malins, et al., 1982; McCain, et al., 1982). They hypothetically attributed this to the well known fact that aromatic hydrocarbons readily convert to metabolites in fish livers.

Malins and Roubal (1985) reported the occurrence of free radicals in bottom-dwelling fish from creosote-polluted Eagle Harbor. Their study supports the possibility that free radicals from aromatic hydrocarbons may be present in liver and bile of English Sole from organically polluted environments; "however, proof of their presence has thus far not been obtained."

Roubal and Malins (1985) studied English Sole from the highly polluted Duwamish River and found free radicals of xenobiotics in liver microsomes of some.

Malins, et al. (1984c) commented that "metabolism of the parent AHs or nitrogen heterocycles by fish limits transfer of these compounds further through the food chain; however, the potential for transfer of metabolites is largely unknown." Then Malins, et al. (1985a) reported their study of adult English sole captured near Mukilteo and from President Point. They found a higher concentration of aromatic hydrocarbons in the stomach contents of fish from Mukilteo than in those from President Point, and they found concentrations of BaP-like and naphthalene-like metabolites in bile from Mukilteo fish to be significantly higher than concentrations in President Point fish. The authors offered their results as documentation of "a dietary route of uptake by English sole of environmental chemicals, including known carcinogens".

Malins et al. (1982) noted that chromatograms of aromatic hydrocarbons from biota were less complex than those from sediments in their environment. They identified fewer than 12 aromatic hydrocarbons in biota and as many as 500 aromatic hydrocarbons in sediments. Some marine organisms may take in aromatic hydrocarbons and quickly metabolize them to compounds which have more toxic effects.

McCain, et al. (1982) studied English sole and starry flounder in the Duwamish Waterway and in other estuaries in Puget Sound and found that concentrations of PCBs in liver tissues clearly corresponded to concentrations in sediments where the fish were caught; but they did not find that aromatic hydrocarbon levels in the fish corresponded with sediment levels.

Effects of Organic Contaminants on Populations

Several studies have examined the possible effect of organic contaminants on species population and diversity, especially for benthos in

Puget Sound (Barrick, et al., 1985 a & b; Battelle, 1985; Chapman, et al., 1984a; Comiskey, et al., 1984; Long, 1982; Malins, et al., 1982; Malins, et al., 1980; Pavlou, et al., 1982; Stober and Chew, 1984; Swartz, et al., 1982; Wingert, et al., 1976; Word, et al., 1984).

Barrick, et al. (1985 a & b) collected 119,095 benthic macroinvertebrates of 407 species from 56 stations in nine study areas of Commencement Bay. The polychaete worm, Tharyx multifilis, and the clam, Axinopsida serricata accounted for 59% of all benthic organisms collected. The next most abundant taxonomic groups were nematode worms and crustaceans, mainly amphipod crustaceans. In two reference type areas, Carr Inlet and the Ruston - Pt. Defiance Shoreline, numbers of species tended to be higher and total abundances lower than in the other seven study areas. Within the waterways of Commencement Bay there were lower numbers of species and higher number of individuals, with enhanced abundances (dominance) of a few species - a situation indicative of an environmentally stressed area. Two stations in this study were nearly devoid of benthic invertebrates. Their depressed abundances indicated severe stress. One of these stations was the closest one to a smelting plant located along the shore near Ruston since the 1800s. The slag generated from this plant contains high concentrations of arsenic and other toxic metals. The other nearly abiotic station was located at the mouth of one of the waterways of Commencement Bay, St. Paul Waterway, at the site of the Champion International pulp mill. Again the authors chose a metal, in this case copper, as a likely contributing factor to the absence of fauna. In summarizing results of their studies in Commencement Bay and nearshore tideflats, Barrick (1985b) noted that within the waterways organic content and grain size of sediments have a great deal of influence on benthic community structure.

Swartz, et al. (1982) sampled from several sites in Commencement Bay and its associated waterways and at Browns Point, on the edge of the mouth of the Bay, to study amphipod distribution there. They found spatial differences in amphipod species composition, particularly in the family Phoxocephalidae. Members of this family are small, burrowing infaunal organisms. Phoxocephalid amphipods were completely absent from all sampling sites in Commencement Bay waterways and at Browns Point, but were ubiquitous out in the Bay. Total amphipod density and species richness were lower in the waterways than in the central Bay.

Battelle, Pacific Northwest Laboratory's (1985) eight bays study found a total of 2,941 benthic infaunal organisms of 120 species at 48 stations. Species richness, abundance, and encounter index were strongly correlated with grain size of the sediments, so they could only compare infauna from urban and rural bays within sediment type. Poor community structure was more associated with depth and clay content than with contaminant concentration. They found high organic carbon levels in all sediment types where infauna were dominated by "organic enrichment opportunists." Three of the four baseline bays were characterized by higher mean species richness than the other baseline bay and the four urban bays. Of the urban bays Everett Harbor (Port Gardner) had the lowest species richness value and highest value for number of individuals. Case Inlet, one of the baseline bays in this study, and Elliott Bay at Fourmile Rock, an urban bay, had low species richness values as well as low values for number of individuals. In each bay polychaete species were the dominant group as a percentage of the mean number of species per station. Battelle ranked the eight bays in order as follows along a baseline-urban continuum based on species richness, abundance and encounter index results:

1. Samish Bay
2. Sequim Bay
3. Dabob Bay
4. Sinclair Inlet
5. Everett Harbor (Port Gardner)
6. Elliott Bay (Fourmile Rock)
7. Bellingham Bay

They ranked Case Inlet eighth (last) because of its sparse infauna, even though they considered it one of the baseline bays for this study.

Chapman, et al. (1984a) chose benthic samples collected by NMFS from 12 stations in three urban bays and two rural bays, and EPA samples from seven stations in the Commencement Bay waterways, and METRO samples from four stations in Elliott Bay for taxonomic analysis based on similarities of depth and sediment texture at the 23 stations. Results from calculation of the Index of Benthic Degradation [simplified from J. S. O'Connor's index (O'Connor, unpublished report) to reflect use of samples only from similar depth and sediment texture.]* indicated that the urban samples were significantly degraded. "Benthic communities in urban embayments were characterized by a high proportion of polychaetes and mollusks, a low proportion of arthropods and echinoderms and the virtual absence of sensitive amphipod families. The reverse situation existed in the rural embayments." Multi-variate statistical analyses of the benthic community data indicated that the Case Inlet reference area was more similar to the urban embayments than to Samish Bay, the other reference area.

* "$I = \text{antilog}\,(f_c) x (F_c) x \Sigma[(n_i) \log(6 + {}^{0.05}/_{P_{irc}})]$
where f_c is the fraction of the total representative taxa that show significant population differences between control and study areas, F_c is the fraction of Phyla that contain taxa that show significant population differences between areas, n_i are the number of taxa that show significant population differences between areas at a selected confidence level and P_{irc} is the probability that differences are not significant (one minus the confidence level). Representative taxa are those present in sufficient abundance frequency to allow reasonable analysis of spatial differences in their distributions." (Chapman, et al., 1984a)

The primary emphasis in the Comiskey, et al. (1984) study for METRO focused on using benthic community structure as an evaluation of biological effects to determine whether or not a toxicant related problem existed. They selected two study sites, each near a known source of toxicant input - one near the West Point Treatment Plant outfall where primary treated effluent is discharged into Puget Sound, and the other located along the Elliott Bay waterfront near the Denny Way combined sewer overflow (CSO), the largest CSO along Elliott Bay. They chose sampling stations at various distances from the input source. In the final stage of these benthic studies they identified taxa, pollutants, and other variables during both the wet and the dry season of year at 26 stations. These investigators as others previously mentioned noted that sediment grain size and depth were major natural factors which appeared to be controlling community structure. They also noted that these two factors were themselves strongly related, i.e. finer sediments are in deeper areas. Community structure data analysis indicated that the Denny Way CSO was unquestionably detrimental to organisms near the outfall, but this CSO did not appear to substantially contribute to burdens in the rest of Elliott Bay. The values for community parameters at all stations in Elliott Bay were lower than values at comparable depths in the central basin. One taxonomic group which included the polychaete worm, Capitella capitata, an indicator of organic enrichment, showed a preference for stations nearest the Denny Way CSO, while taxa which included a sensitive amphipod family seemed to avoid the same stations. The West Point outfall did not appear to strongly influence benthic communities in its vicinity. They could explain most trends observed near West Point as responses to environmental conditions unrelated to pollutant levels.

Malins, et al. (1982) reported the average highest species richness values for sediment-dwelling invertebrates in the MESA Puget Sound Project at Port Madison and West Point, reference areas. They found the lowest values in the urban areas Hylebos Waterway, and inner portions of the urban associated areas, as Budd Inlet. They procured data concerning species composition from sampling 37 stations quarterly in 1979 and twice in 1980. Malins, et al. (1980) reported consistantly low species richness values for infaunal invertebrates from the Duwamish Waterway.

Pavlou, et al. (1982) investigated impacts of open-water disposal into Elliott Bay of dredged material contaminated with elevated PCB concentrations. In summarizing their biological results they stated that particular taxa in close proximity to and within the dredge material disposal site may have higher abundances. They also found that stations close to the disposal site were more similar to each other than to more distant stations. Their reported preliminary data indicated the dredged material and its associated PCBs was stable and no major long-term impact was apparent.

In 1982 it became necessary to expand the Renton Sewage Treatment Plant to meet the needs of increased population in metropolitan Seattle. Secondary effluent had been discharged into the Duwamish River, but that policy had to be changed to comply with water quality standards required for the Duwamish River. The municipality of metropolitan Seattle (Metro) conducted the "Renton Sewage Treatment Plant Project: Seahurst Baseline Study" (Stober and Chew, 1984) to collect ecological data useful in considering Seahurst Bay, located due west of the Renton Sewage Treatment

Plant, as a new outfall site for the effluent from the plant. Before completion of the planned three year study, the METRO council decided to change the new outfall site to Duwamish Head in outer Elliott Bay. During the study, however, Metro obtained useful information about central Puget Sound. In the Word, et al.'s (1984) subtidal sediment study they observed greater abundance and fewer species of benthic infauna in areas with greater organic loading to sediments. In such areas they predicted two pelecypod species were likely to increase in abundance, while abundance of echinoderms and certain arthropods would decrease. They saw seasonal trends. The maximum number of species occurred in summer months; the greatest abundance occurred in late fall and early winter. They also noted a temporal trend. They compared numbers of species and numbers of individual organisms in this current study to recorded results from a 1969-70 study conducted by F. L. Nichols. "More species and more individuals are present in the system now than were there fifteen years ago." Results of this study agreed with others (Barrick, et al., 1985b.; Battelle, 1985; Chapman, et al., 1984a.; Comisky, et al., 1984) in considering substrate type a primary factor accounting for differences in community structure. Analysis from the Seahurst Study gave more importance to habitat type than to geographic location or season in explaining differences among samples in species composition and abundance.

In the MESA Puget Sound Project (Long, 1982 and Malins, et al., 1982) unexpected results showed highest fish abundance, species richness and species diversity in Elliott and Commencement Bays, the most contaminated bays studied. Malins, et al. (1982) noted that these two are estuarine bays being compared to inlets and open bays.

Wingert, et al. (1976) investigated the ecology of demersal fishes near METRO-operated sewage outfalls and in the Duwamish waterway. They used one site, Point Pully, which lacked an outfall and was considered nonstressed, Point Pully, was used as a "control" site. They observed that generally the standing crop and species richness were highest at Point Pully. Starry flounder, English sole, Pacific staghorn sculpin, longfin smelt and snake prickleback were among species predominant in the Duwamish waterway.

Effects of Organic Contaminants: Abnormalities in Fish and Invertebrates

A number of studies have included a look at the prevalence of incidences of fish or invertebrate abnormalities in various Puget Sound areas to see if there is a a significant correlation between this prevalence and the level of organic or general pollution in the area (Barrick, et al., 1985 a & b; Battelle, 1985 Draft Final Report; Dexter, et al., 1981; Gronlund, et al., 1983; Krahn, et al., 1986; Long, 1982, 1981; Malins, et al., 1985 a, b, c, 1984, 1984 a & b, 1983, 1982, 1980; McCain, et al., 1982; Wellings, et al., 1976; Wingert, et al., 1976). According to Dexter, et al. (1981), "Past research has documented the existence of histopathological abnormalities in biota of subregions of Puget Sound receiving considerable toxicant loads."

Battelle, PNL's (1985 Draft) final report, discusses prevalence of fish and shellfish disease in relation to physical and chemical properties of associated sediments. They found two types of lesions only in shellfish from two urban bays. In Bellingham Bay and Elliott Bay at Fourmile Rock they found two shrimp species with degenerative disorders in their antennal gland and Dungeness crab with degeneration/membrane lysis in the hepatopancreas. They did not find these abnormalities in any of the other six bays studied.

In Battelle's same study they collected 138 English sole and 83 Dover sole from Everett Harbor/Port Gardner, Elliott Bay/Fourmile Rock, Sinclair Inlet, Case Inlet, Eliza Island (west of Bellingham Bay), and in Elliott Bay at Duwamish Head and West Point. Prevalences of liver, kidney, and gill lesions in English sole from Sinclair Inlet were highest, but were low in Case Inlet (compared to other urban bays including the Duwamish waterway). They did not detect these lesions in sole from Eliza Island.

According to Long (1982), "The frequency of liver lesions in fish, and gill and antennal gland disorders in crustaceans, appeared to be greatest in areas with high contaminant concentrations (Elliott Bay, Commencement Bay)". He further reported the positive correlation ($p \leq 0.01$) of frequency of "preneoplastic" lesions in English sole with high aromatic hydrocarbon concentrations in sediments.

Long (1981) discussed results of research supported by the MESA Project in which 2,951 fish and 618 invertebrates were examined for gross and microscopic abnormalities. The most common abnormalities were in livers of fish and in hepatopancreas of crustaceans. Percent incidences of abnormalities were higher in industrialized waterways and waterfronts than in outer bays or reference areas.

Gronlund, et al. (1983) reported that of 66 English sole collected from Everett Harbor for the Port Gardner Bottomfish Survey, 70% had at least one type of liver lesion. One of the most serious types of liver lesions found were neoplasms (tumors, 12%). English sole from Port Madison, a reference area, were virtually free of this liver disease.

Dr. Donald Malins, Director, Environmental Conservation Division, Northwest and Alaska Fisheries Center, NMFS, NOAA, addressed the Puget Sound Water Quality Conference held Sept. 30 and Oct. 1, 1983 in Seattle explaining much that was known about organic and other pollutants, their sources and their effects on marine life in urban embayments of Puget Sound. He spoke of the conversion of aromatic hydrocarbons to potentially toxic (some carcinogenic) substances by environmental processes and through metabolism inside animals (including fish). He pointed out that "tumors, or liver diseases in general, exist in fairly high frequencies in the polluted industrial areas such as Commencement Bay, Elliott Bay, and the Everett Harbor". He also said, "You don't find high frequencies of the lesions in the so-called 'clean' areas, Port Madison. ., and Case Inlet. .".(Malins, 1984).

Malins also commented about shellfish noting that sometimes they find quite a high frequency of disease in crabs in polluted areas, but they also find diseased crabs in areas with no obvious pollution. So, there seems to be more of a mystery surrounding the possiblity of implicating organic pollutants in crustacean abnormality than in fish abnormality.

Barrick et al. (1985 a & b) reported that lowest English sole lesion prevalences were at reference sites; highest were in the waterways of Commencement Bay; intermediate prevalences were along the Ruston-Pt. Defiance Shoreline. They found no hepatic neoplasms in English sole from the Ruston-Pt. Defiance Shoreline or from Carr Inlet, but they did find them in every other study area.

Malins, et al. (1984a) examined bottom dwelling fish from four urban and four nonurban embayments for abnormalities and noted that most lesions were in the liver, kidney, and gills. The highest prevalences of hepatic neoplasms were in English sole from the Duwamish Waterway and from the eastern, Port Gardner subarea of Everett Harbor. Prevalences for Rock sole were not as high as for English sole, but highest prevalence of hepatic neoplasms in Rock sole were found in subareas in Everett Harbor and in the Hylebos Waterway.

Malins, et al. (1985a) found high prevalences of liver lesions of unknown cause, including 7.5% hepatic neoplasms, in English sole from waters near Mukilteo where potential sources of pollution include a major fuel storage depot, a ferry terminal, an abandoned boat ramp, and a municipal sewage outfall. Sediments contained high concentrations of aromatic hydrocarbons there and fish bile contained high concentrations of metabolites of aromatic hydrocarbons. They found no hepatic neoplasms in English sole from President Point where very low background levels of organic contaminants exist in sediments, though they did find some other types of hepatic lesions there.

Findings from Krahn, et al. (1986) provided further evidence of the putative relationships between aromatic hydrocarbons and serious idiopathic liver diseases in English sole. They measured concentrations of metabolites of aromatic hydrocarbons in bile of sole from eleven Puget Sound sites (two had no known anthropogenic source of contaminants). Using the Spearman's rank correlation procedure, they found a significant positive correlation (p ε≤σ 0.0125) between prevalence of total hepatic lesions and the concentrations of metabolites of aromatic compounds in fish bile.

Dr. Sin Lam Chan (Personal Interview, June 26, 1985) spoke of good, strong, but not absolute evidence of a relation between PAHs and fish abnormalities. He used as an example Eagle Harbor where creosote is apparently the only source of PAH pollution. There about 30% of the fish sampled had liver tumors; yet no tumors were found in fish from many reference areas sampled. The mean concentration of metabolites of aromatic compounds measured in bile of Eagle Harbor fish was 2,100±1,500 ppb and only 100±89 ppb for President Point fish (Malins, et al., 1985b).

Malins et al. (1985c) reviewed research results concerning the nature and prevalence of liver neoplasms in demersal marine fishes. They observed that in almost all cases hepatic neoplasms were reported in fish from urban waters and not in fish from nonurban Puget Sound waters. They cited high prevalences of hepatic neoplasms found in English sole from the following areas:

Everett Harbor, 12% (n = 66)
Duwamish Waterway, 8.2% (n = 537)
Muktilteo, 7.5% (n = 66)

Malins, et al. (1985c) also reviewed studies showing statistical analyses comparing chemicals in Puget Sound sediments with prevalence of hepatic lesions in English sole. Prevalence was positively correlated with sediment concentrations of aromatic hydrocarbons and metals (Malins et al., 1984a.). Malins, et al. (1984b) found positive correlations between the frequencies of liver neoplasms and other liver lesions in English sole and concentrations of aromatic hydrocarbons in sediment. They did not find such correlations with chlorinated hydrocarbons.

In another study of liver lesions of unknown cause in English sole, Malins, et al. (1983) found that most liver samples contained multiple lesion types. 51 of 54 sole with such lesions were from the highly polluted Duwamish River and Commencement Bay sites. These idiopathic liver lesions were in all fish examined from Commencement Bay, but were not in fish from the non-industrialized embayment of Meadow Point.

Malins, et al. (1982) captured and examined crabs and shrimp for grossly visible and microscopic abnormalities. They found histopathological lesions most frequently in the hepatopancreas, the antennal glands, the gills and the midgut. They considered these lesions to be of unknown cause. Only the crab gill lesions were externally visible abnormalities. Crustaceans with some of these lesions were common in urban areas, but they captured too few animals to statistically analyse the significance of this observation.

In the same investigation English sole examined most commonly either had lesions associated with infectious agents or had lesions caused by unknown factors. They either found the lesions of unknown cause only in fish from urban embayments, or they found the idiopathic lesions most prevalent in these areas. Three main types of lesions of unknown cause seen in the English sole were liver neoplasms, "preneoplastic" lesions, and necrotic liver lesions. The first two types were most prevalent in the 535 Fish from Seattle's Duwamish Waterway and in the 573 fish in Tacoma's Waterways. Specific necrotic liver lesions were most prevalent in the Duwamish Waterway (18% of the 535 fish) and along Seattle's waterfront (20% of 161 fish). They used two statistical methods to evaluate possible relationships between prevalence of English sole with these types of lesions and chemical composition of sediments in their environment. Results of cluster group analysis showed an apparent association between prevalence of these lesions and sediment concentrations of metals and aromatic hydrocarbons. Results of the Spearman rank correlation supported this association. Prevalence of two types of idiopathic lesions, liver neoplasms and specific necrotic lesions was positively correlated ($p \leq 0.05$ significance level) with metal and aromatic hydrocarbon concentrations in sediments. Prevalence of English sole with "preneoplastic" lesions was positively correlated ($p \leq 0.01$ significance level) only with aromatic hydrocarbon level.

In the earlier study, Malins, et al. (1980), concluded that results supported the hypothesis that "the hepatic lesions found in this study were caused or enhanced by exposure of fish to one or more toxic chemicals found in their environment".

McCain, et al. (1982) reported prevalence of English sole with "preneoplastic" and/or neoplastic lesions were 20.5% (113 of 551 fish) for the Duwamish Waterway and 20.4% (10 of 49 fish) for Lake Washington Ship Canal. The prevalence of English sole with liver neoplasms was 12.9% (71 of 551 fish) for the Duwamish Waterway and 8.4% (4 of 49 fish) for Lake Washington Ship Canal. Prevalence of starry flounder with liver neoplasms in the Duwamish Waterway was considerably lower than that of English sole.

Wellings, et al. (1976) studied fin erosion disease of starry flounder and English sole in the Duwamish River. They suggested that fin erosion incidence is related to an interaction between genetic constitution, a variety of chemical pollutants and other environmental variables. Wingert, et al. (1976) also communicated ideas arising from an investigation of fin erosion disease of demersal fishes. Observation of the progression of the disease suggested that it is not caused by a microorganism. Observation of most frequent occurrance of the disease on fins closely associated with the sediment suggested that the causal agent is in the sediment. They also observed that large numbers of starry flounder from the Duwamish River suffering from fin eriosion had livers with an abnormal color. Histological examination revealed that the abnormal color was a gross indicator of liver pathogenesis.

Sediment (and Water Column) Toxicity: Commencement Bay

In a number of recent studies (Barrick, et al., 1985a. & b.; Chapman, et al., 1985; Chapman and Fink, 1984; Chapman, et al., 1984a.; Chapman, et al., 1983, 1982; Long, 1984, 1983; Swartz, et al., 1982) investigators have studied toxicity of Commencement Bay sediments.

In Barrick, et al. (1985 a & b) one test to evaluate the relative toxicity of Commencement Bay sediments was an amphipod mortality bioassay. Using sediments from 52 stations, they measured direct lethal response and compared toxicity of these sediments to toxicity of Carr Inlet sediments. Mortalities in sediments from 18 test stations from locations in every waterway, and in Ruston, and Tacoma were significantly different ($p \leq 0.05$) from Carr Inlet samples. Mortality ranged from 20 to 100 percent. Analyses indicated no significant difference ($p \leq 0.05$) in mean mortality values among sediment controls; mortality ranged from 4 to 10 percent. In dilution bioassays using Commencement Bay test sediments from six stations (one of these stations really represented combined sediments from two stations) they diluted test sediments with varying amounts of clean sediment. Results indicated that a 50 to 75 percent dilution with clean sediment was sufficient to eliminate toxic response to sediments from five of the six stations. Combined sediments from two Ruston (just south of Pt. Defiance) stations near the ASARCO smelter were still highly toxic when they made up only 10 percent of the sediment mixture (90% dilution). The smelter would more likely contribute metalic rather than organic contaminants to sediments.

Chapman, et al. (1985) investigated and reported acute toxicity of sediments to amphipod crustaceans, oligochaete worms and stickleback fish. They tested twenty members of each species separately with each of the test and control sediments and reported less than 15% mortalities for each of the species. This was not significantly different than mortalities for controls. They concluded that acute lethal tests are relatively insensitive compared with sublethal tests.

Chapman and Fink (1984) reported larval mortality of the polychaete worm, <u>Capitella capitata</u>, significantly different ($p \leq 0.05$) from controls (50% mortality after 50 days) only in tests using sediment elutriate from one of their Commencement Bay and Waterways stations. This station was located about midway of the Hylebos Waterway. When using whole sediments from that same station, there was 70% mortality after 35 days. There were similar results with whole sediments from several other stations in the waterways (75% mortality from a station lower in Hylebos Waterway, 65% from one Blair Waterway station, 100% mortality from the Sitcum Waterway station, 65% from a City Waterway station).

Results of lethality bioassays with an oligochaete, an amphipod, and fish reported by Chapman, et al. (1982) indicated that no acute lethality occurred using bottom water samples and slurries of Commencement Bay and other Puget Sound sediments. From those results and other observations it appeared that "direct, rapid lethality is not a major factor for the majority of fauna exposed to and living in or near chemically contaminated Puget Sound sites".

Swartz, et al. (1982) measured the toxicity of 175 Commencement Bay sediment samples by survival of amphipods, <u>Rhepoxynius abronius</u>, after a 10 day exposure to test sediments. It is interesting to note that sediment samples from Yaquina Bay, Oregon were used in this benthic bioassay as reference sediments. Compared to these Yaquina Bay samples, survival in sediments from the offshore, deeper part of Commencement Bay was high, and survival in sediments from some parts of the waterways, especially in Hylebos Waterway, was low. Mean survival in five replicate sediment samples from each of three station locations was significantly less ($P \leq 0.05$) than in Yaquina Bay "controls". These stations were in City and Hylebos Waterways and at Browns Point (near the eastern shore of the mouth of Commencement Bay). The lowest mean survival in any of the waterways or in offshore or near shore Commencement Bay sediments occurred in Hylebos Waterway station sediments.

Results of several studies have demonstrated sublethal effects of sediments from various locations in Commencement Bay. They include effects on respiration, reproduction, and development, and other abnormalities in several invertebrates and in fishes.

Barrick, et al. (1985 a & b) conducted oyster larvae abnormality bioassays. "Exposure to sediments from 15 of their 52 sites caused significant oyster larvae abnormality ($p \leq 0.05$)." These 15 sites were in Hylebos, St. Paul, and City Waterways, and along the Ruston shore (south of Pt. Defiance). Results of oyster larvae sediment dilution bioassays indicated that greater than 75 percent dilutions were necsessary to reduce abnormalities to control levels. In this study reported abnormalities were developmental ones.

Chapman, et al. (1985) made respiration rate measurements with marine oligochaete worms in sediment elutriate from various Commencement Bay stations. They also measured genotoxicity in vitro by growing rainbow trout gonad cells on glass slides, exposing them to sediment extracts during mitotic division, then examining fixed, stained anaphase cells for chromosomal defects and for cell division inhibition. In five Commencement Bay subareas - outer Bay, inner Bay, Hylebos upper waterway, Blair upper waterway and City waterway - there were significant respiration rate responses or genotoxic responses in over 60% of the sampling sites. Sediments from 67% of the stations in Hylebos upper waterway apparently induced altered respiration rates; 100% of them apparently induced genotoxic responses. Sediments from 75% of City Waterway stations illicited respiration rate responses; 100% illicited genotoxic responses. In tests using oyster larvae, surf smelt eggs and larvae, and polychaete trochophore larvae, reproductive impairment resulted in all species from association with Hylebos upper waterway sediments from all (100%) of the sampling sites. In sediment from one lower Hylebos Waterway station, surf smelt hatching success was <2%.

Chapman and Fink (1984) observed effects of sediment elutriate and whole sediments on growth and on reproduction of polychaete worms. In general growth of the worms in whole sediments was about half that in sediment elutriates. In elutriate tests only one test sediment (from a station midway in Blair Waterway) had a growth rate prior to sexual maturation that was significantly different from controls. The growth rate was almost twice as high in this sediment elutriate. In whole sediment toxicity tests, worm growth was significantly different in sediments from three Hylebos waterway stations and from one City Waterway station.

Chapman, et al. (1983) collected sediment samples from selected stations in Elliott Bay and the Duwamish River, Commencement Bay and associated waterways, Sinclair Inlet, and Port Madison and tested for reproduction impairment effects. Results indicated that Commencement Bay Waterways and the lower Duwamish River were the most toxic areas. Outer Commencement Bay stations showed low toxicity. Oyster larvae (Crassostrea gigas) bioassay data revealed high mean percentage of abnormal larvae resulting from association of the larvae with replicate sediment samples from certain waterway stations and very low percents abnormality in larvae tested with reference and control sediments:

Station Location	Percent Abnormality
Hylebos	91
Hylebos	42
Blair	32
City	94
Port Madison (ref)	4
Sediment Control	2
Seawater Control	1

Chapman et al. (1983) conducted cell reproduction studies by exposing rainbow trout gonad (RTG-2) and bluegill fry (BF-2) cells to sediment extracts during logarithmic growth. Sediment extracts from all Commencement Bay waterways were toxic at some concentration to rainbow trout gonad cells; the two most toxic sediments were from Blair Waterway. Sediments from only three stations inhibited bluegill fry cell proliferation. They were located in Blair, Sitcum, and City Waterways.

Results from Chapman, et al. (1982) showed significantly (P ε≤σ 0.05) depressed or elevated respiration responses of oligochaete worms, Monopylephorus cuticulatus, to Commencement Bay Waterway water and sediment slurry samples. They detected significant respiratory anomalies compared to controls in 20 of 37 sediments from Commencement Bay. They reported elevated respiration rates from worms in sediments of all stations in Hylebos, Blair and Sitcum Waterways in which effects were shown. The three Commencement Bay areas with the highest percent of stations showing detrimental effects on respiratory rate were City Waterway (75%), upper Hylebos Waterway (67%) and offshore (67%) according to Long (1983).

Results also showed significant genotoxicity responses of in vitro rainbow trout gonad cells to sediments and water extracts (Chapman, et al., 1982). Sediment extracts tested from seven stations in the waterways and two along the Ruston-Tacoma shoreline were among stations considered to have the greatest inhibitory activity on cell proliferation of all the 97 Puget Sound stations sampled and tested in this study. Mutagenicity occurred from association with sediment extracts from all of the stations in the upper Hylebos Waterway and from all stations in City Waterway (Long, 1983).

Sediment (and Water Column) Toxicity: Other Areas

In their eight bays study, Battelle (1985 Draft Final Report) found that amphipod survival results of their bioassays did not clearly distinguish urban bays from baseline bays. Correlation analysis revealed important relationships of sediment grain size, percent water, and organic compound load to amphipod survival. They found low amphipod survival (<50% after 10 day exposure to sediments) in bioassays of sediments from five Everett Harbor stations, three Dabob Bay stations and six Case Inlet stations.

Chapman, et al. (1985) conducted acute lethality tests in which they exposed amphipod crustaceans, oligochaetes, and stickleback fish to resuspended sediments in seawater for ten days. They noted an acute lethality response significantly different than controls only from exposure of amphipods to sediments from one Elliott Bay station. A 40% amphipod mortality resulted. Bioassays of other organisms in other sediments resulted in less than 15% mortalities.

When Chapman and Fink (1984) investigated effects of Puget Sound sediments on the polychaete worm, Capitella capitata, there was significantly higher mortality than controls in sediment elutriates from two stations in Elliott Bay and one station in the Duwamish Waterway.

Chapman, et al. (1984) conducted acute lethal bioassays with the sensitive amphipod, Rheopoxynius abronius, and partial life-cycle bioassays with oyster larvae of Crassostrea gigas using sediments from two industrialized Puget Sound embayments, Bellingham Bay and Everett Harbor and using sediments from Samish Bay as a reference. In sediments from one station in Bellingham Bay and one station in Everett Harbor there were significant acute lethal effects to amphipods. Reference station sediments showed no significant effects in either the amphipod or the oyster larvae tests. Three stations in Everett Harbor and four stations in Bellingham Bay had sediments in which oyster larvae mean relative survival was quite low, ranging from 8-26%.

Dinnel, et al. (1984) report and Stober and Chew (1984) report on the Central Puget Sound Basin, which extends north as far as Alki Point and south to (but not including) Commencement Bay. They reported mortality of oyster embryos resulting from 48 - hour exposure to water from various Central Basin sites. Mortality was highest in surface waters and within the East passage. These results essentially agreed with 1962-1976 Washington Department of Fisheries data. They also reported results of ten day exposures of amphipods to sediments from bioassays conducted twice each year for two years. Reduced amphipod survival occurred in sediments from the northern East Passage area and was correlated with grain size and toxicant concentration.

McCain, et al. (1982) report results of laboratory experiments conducted to determine the effect of exposure to bottom sediments on English sole. Sole were kept in aquaria for about three months. The mortality rate of sole exposed to Duwamish Waterway sediments was not statistically different from sole on reference area sediments. When extracts of sediments from the Duwamish Waterway were intraperitoneally injected into juvenile English sole, significantly ($p \leq 0.05$) more fish died than fish injected with extracts of reference area sediments.

Battelle (1985 Draft Final Report) found that their analyses of oyster larval bioassay results indicated that urban bays were not clearly distinct from baseline bays. Samish Bay, Case Inlet, and Sequim Bay sediments demonstrated the lowest percent abnormalities; Bellingham Bay, the highest percent abnormality; Dabob Bay, classified as a baseline bay, the second highest percent abnormalities; and Everett Harbor - Port Gardner demonstrated an intermediate response.

Chapman, et al. (1985) reported respiration rate measurements of marine oligochaetes during exposure to sediment elutriate, genotoxicity measurements made exposing in vitro rainbow trout gonad cells to sediment extracts during mitotic division, and results of reproductive impairment tests using oyster larvae, surf smelt eggs, and rainbow trout gonad and bluegill fry cells in vitro. They observed toxic respiration responses of oligochaete worms in sediment elutriates from more than 50 percent of the sampling sites within Elliott Bay, off the Denny Way CSO and within outer Sinclair Inlet.

Genotoxic responses resulted from exposure to sediment elutriates from more than 50 percent of the sampling sites within outer Elliott Bay, off the Denny Way CSO in Elliott Bay, the lower and upper channels of the Duwamish River, inner Sinclair Inlet, and within Birch Bay, which is considered a control area. During reproductive impairment tests, surf smelt eggs exhibited extremely low hatching success (2% or less) in certain Elliott Bay and upper Duwamish sediments and larvae hatched prematurely when eggs were exposed to certain Duwamish River sediments.

Chapman and Fink (1984) grew polychaete worms, Capitella capitata, in whole sediments for 35 days and measured survivors to test effect of sediments on growth. Mean lengths of worms grown in sediments from two Duwamish Waterway stations and from one Sinclair Inlet station were significantly less than controls, indicating slower growth.

Chapman, et al. (1984) conducted sediment toxicity tests on the industrial embayments Bellingham Bay and Everett Harbor and on Samish Bay, their chosen reference area. They examined sublethal effects on developing oyster larvae, on respiration in oligochaete worms, on cell reproduction of rainbow trout gonad and bluegill fry cells, and on condition of chromosomes in rainbow trout cells during the anaphase stage of mitosis. They observed more than 20% larval abnormality in oyster embryos exposed to sediments from three Bellingham Bay and three Everett Harbor stations. (They observed lowest survival rates in these same sediments.) In sediment elutriates from three of ten Bellingham Bay stations and four of ten Everett Harbor stations, oligochaete worms had significantly different respiration rates than controls, but reference station elutriates did not.

Chapman, et al. (1984) reported significantly reduced cell reproduction in rainbow trout gonad cells exposed to sediment extracts from one reference station, one Bellingham Bay station, and six Everett Harbor stations. Sediment extracts from one reference station, five Bellingham Bay stations, and two Everett Harbor stations caused significant chromosonal damage in the fish cells during anaphase.

It is interesting to note that the same seven stations from Chapman, et al. (1984) whose sediments significantly altered respiration rates of oligochaete worms, exhibited one or two other significant sublethal effects as well. These results suggest that effects of contaminants are site specific and do not spread out into general geographical areas.

In Chapman, et al. (1983) results of oyster larvae bioassays indicated Duwamish River sediments, and sediments from one Elliott Bay station near the Denny Way CSO were very toxic. High mean percentages of abnormal larvae resulted from 48 hour association with sediment samples. Mean values for percent abnormalities from each of the four Duwamish River and Waterways sediment samples were 78%, 50%, 86%, and 31%. About 31% abnormal larvae resulted from exposure to Elliott Bay samples from near the Denny Way CSO.

In another test for sublethal effects of sediments, Chapman, et al. (1983) reported detrimental effects to rainbow trout gonad cell reproduction in sediment extracts at concentrations below 10,000 ppb. in samples from the Magnolia Bluff station, two Elliott Bay stations (not the Denny Way CSO station), two Duwamish River stations, the East Duwamish Waterway, and one Sinclair Inlet station.

When Chapman, et al. (1982) exposed oligochaete worms to sediment slurry from 13 of 37 Elliott Bay and Duwamish stations, 6 of 12 Sinclair Inlet stations, and 1 of 6 Port Madison stations, these worms showed significantly altered respiratory responses compared to controls. Significant respiratory anomalies were not detected in any of the Birch Bay (control site) sediments. They also found that sediments from the following Puget Sound stations had a high inhibitory (genotoxic) effect on proliferation of rainbow trout gonad cells in vitro: West Point, Alki Point, mouth of the West Duwamish Waterway, mouth of the East Duwamish Waterway, (3) Duwamish River stations, the East Passage station, (3) Port Madison (reference area) stations, and (3) Birch Bay (control area) stations.

Comisky, et al. (1984) conducted bioassay tests using sediments from two study sites, one near the West Point Treatment Plant outfall from which primary treated effluent is discharged, and the other near the Denny Way combined sewer overflow (CSO) located along the Elliott Bay waterfront, in an attempt to determine relative toxicity. They tested amphipod survival, oligochaete respiration and chromosome changes on a number of samples. Results of all tests were inconclusive. Results of the first phase of acute lethal amphipod bioassays did not agree with results from their follow-up bioassays in their second phase. Oligochaete respiration tests at Denny Way sites indicated a biological stress response at 40 percent of the stations and at 56 percent of the West Point stations, but there was no correlation between these responses and levels of priority pollutants at those sites. The lack of consistency made interpretation of results difficult and created doubts particularly about using amphipod tests in screening for priority pollutants.

Sediment Toxicity: Summary

Long (1984) summarized sediment bioassay data from several studies of Puget Sound to show relative toxicity of sediments from various areas. He included data from studies from the late 1970's through some time in 1984. He only omitted data from a few major studies such as Battelle's detailed "eight bays" study. He reviewed results of acute toxicity studies and identified the following areas where more than 50% of the sediment samples elicited a toxic response:

outer Bellingham Bay
inner Everett Harbor
Quilcene Bay
outer Dabob Bay
Elliott Bay, off the Denny Way CSO
inner Sinclair Inlet
inner nearshore of Commencement Bay
upper Hylebos
lower Hylebos
City Waterway
outer Case Inlet

His summary of studies also identified these areas in which 50% or more sediment samples elicited sublethal toxic responses:

Birch Bay (intertidal)
Bellingham Bay
inner Everett Harbor
Port Gardner
Elliott Bay, West Point area
Elliott Bay in general
inner Elliott Bay
Elliott Bay off Denny Way CSO
West Duwamish Waterway
upper Duwamish
mid Duwamish
lower Duwamish
Sinclair Inlet in general
Sinclair Inlet shipyards
outer Sinclair Inlet
inner Sinclair Inlet
inner or nearshore Commencement Bay
upper Hylebos Waterway
upper Blair Waterway
lower Blair Waterway
Sitcum Waterway
City Waterway

Results of reproductive impairment tests from Chapman, et al. (1983) and results from the earlier broad scale toxicity survey in Chapman, et al. (1982) indicated that Commencement Bay Waterways and the lower Duwamish River were the most toxic areas tested.

When Long (1983) summarized data from sediment toxicity studies reported in Ott, et al. (1982), Swartz, et al. (1982), and Chapman et al. (1982), he found these Puget Sound subareas with the highest percent of sites with significantly different toxic response values than controls:

1. Elliott Bay, off the Denny Way CSO
2. lower Duwamish River
3. upper Hylebos Waterway
4. City Waterway
5. outer Sinclair Inlet (near Bremerton)

Correlation of Contaminant Levels with Observed Effects

Chapman, et al. (1984a) offered a ranking of the five Puget Sound areas they studied. They used their "Triad" Index in which sediment chemistry, benthic community and sediment bioassay data were compared for this ranking:

1. Commencement Bay
2. Elliott Bay
3. Sinclair Inlet
4. Case Inlet
5. Samish Bay

The three urban bays tested are significantly degraded. Commencement Bay Waterways were more degraded than Elliott Bay; though Elliott Bay had higher sediment contaminant levels. Case Inlet was more similar to the urban bays than to Samish Bay. Samish Bay is a suitable reference.

In Battelle's (1985 Draft) Final Report, Volume I, they name three types of investigations which proved useful in their study of eight Puget Sound bays - analyses of physical and chemical characteristics, analysis of infaunal communities, and amphipod bioassay. When they ranked the bays, the urban ones were significantly more impacted than the baseline ones. They found the worst problems in Elliott Bay - Port Gardner and Sinclair Inlet, and noted that Case Inlet responded more like an urban than a baseline bay, just as Chapman, et al. (1984a) had noted. They also concluded that metals may not be as important for toxicity to infauna or amphipods as organics (PCBs and PAHs) in Puget Sound sediments.

Barrick, et al. (1985 a & b) found that sediment toxicity and number of significant benthic effects were highest in the most chemically contaminated of the areas (Commencement Bay and Waterways, Ruston - Tacoma shoreline, Carr Inlet) they studied. However, "no one contaminant or contaminant group correlated with effects observed in all areas" (Barrick, et al. 1985a).

The research findings in Everett Harbor reported by Gronlund, et al. (1983) concur with studies from other portions of Puget Sound. Serious liver diseases in certain species of bottomfish are characteristic of urban areas with high toxic chemical sediment loads.

Human Health Risk

The May, 1985 update for the "Commencement Bay Nearshore/Tidalflats Superfund Investigation" described the revised advisory issued in April, 1985 from the Tacoma - Pierce County Health Department and Department of Social and Health Services about eating fish in the area. The superfund study identified PCBs as the chemical groups causing the highest potential health risk. the advisory recommended the following:

1. Do not consume fish or crab from any of the Commencement Bay Water ways.
2. If you do consume fish from the waterways limit consumption to fish muscle tissue.
3. Do not consume fish livers from anywhere in Commencement Bay.
4. It's okay to consume fish or crabs caught in Carr Inlet or along the Ruston/Point Defiance shoreline.

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OREGON BAYS

We have found virtually no data regarding measurement of PCBs or other pesticides in Oregon Bays, nor general water quality or population data. We have some data, however, from important research revealing PAH levels in mollusks, water, and sediments and possible effects of these chemical burdens on mussels. We have also seen in the literature a discussion of human health hazard from organic contaminants in Oregon Bays.

Dr. Michael Mix of Oregon State University in Corvallis, Oregon has reported results of sampling mollusks from various Oregon Bays (See Figure 1) for a number of years and analyzing them for PAHs. He has compared these results with concentrations of PAHs found in mollusk tissue samples from many remote sites and has observed that PAHs are ubiquitous at approximately the same levels (Mix, 1982; Mix, 1984; Mix, Personal Communication, June 24, 1985).

Mix's early investigations into PAH levels (Mix, et al., 1977) show a clear contrast between mollusks from industrialized and those from non-industrialized sites. "Certain populations of indigenous bivalve mollusks from the industrialized Oregon Bays . . .contain shellfish with significant levels of BaP in their tissues. Shellfish from the nonindustrialized bays . . . contain non-detectable levels of BaP." At the time Mix believed that BaP concentration served as an indicator of total PAH concentration in mollusks (Mix, 1979). Mix concluded, though, after more thorough investigations (Mix, 1982) that "Benzo(a)pyrene was not a significant variable for predicting total PNAH at any site. . . .the use of BaP for making decisions about the quantities and presence or absence of other PNAH should be abandoned or modified." In a June 24, 1985 personal communication he made additional comments about research procedure. He recommended that percent recovery be determined before proceeding with chemical analysis and that animals be allowed to purge themselves of sediment before their tissue is analyzed.

In another early study (Mix and Schaffer, 1979a) Mix and Schaffer sampled mussels, Mytilus edulis, bimonthly for two years from thirteen sites in Yaquina Bay. Inexplicably the average concentration of B(a)P from one of these sites along the Oldtown bayfront was significantly higher, (130ppb, dry wt.), than the mean of the average concentrations from the other twelve sites, (20 ppb, dry wt.). Mussels in eleven of the sites were exposed to creosoted pilings or a creosoted floating dock, potential sources of PAHs. The particular combination of those sources plus presence of a fish processing plant, and nearby marinas may explain the higher BaP concentration at the one site. In this study as in others BaP concentration in mollusk tissues depended upon geographical location.

In more recent studies Mix's results confirm the influence of industrialization of sites on PAH tissue levels. "The values in Table 13 indicate that shellfish in relatively pristine areas of the three bays have a baseline PNAH load of approximately 50 µg/kg.. Increased concentrations

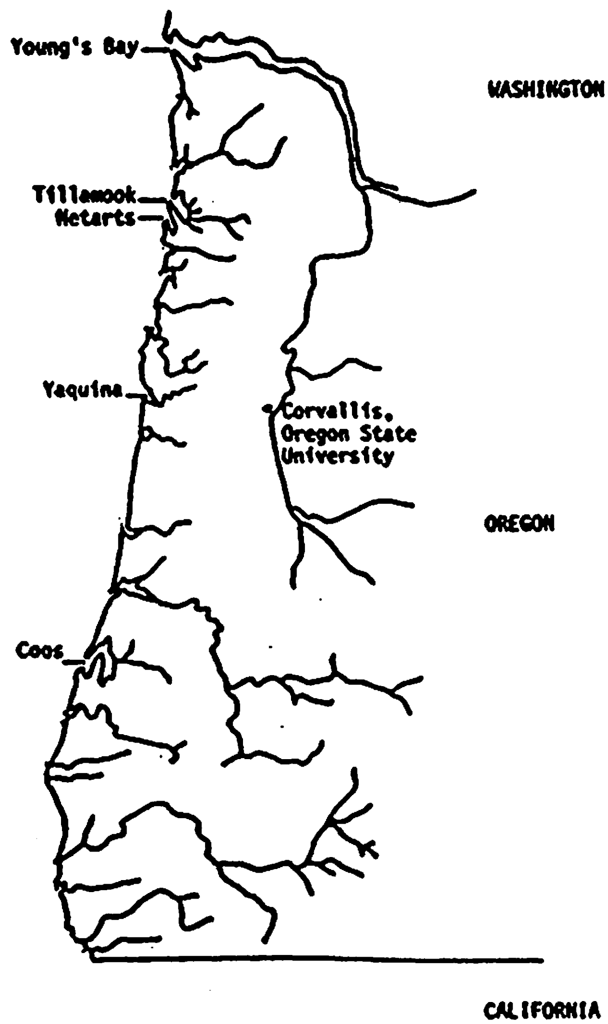

Figure 1. Location of Oregon Bays and Estuaries (Mix, et al., 1977).

occurred in direct relation to the degree of industrialization and human onshore activity." (Mix, 1982).

In measuring PAH levels in bay mussels, Mytilus edulis, from a remote site in Yaquina Bay and from a more industrialized bayfront site during 1979 - 1980, Mix and Schaffer, (1983a), found concentrations from the remote site significantly lower than those from the bayfront (1369.5 ppb, dry wt. compared to 4931 ppb, dry wt.). They also observed that "smaller, more water soluble, compounds were concentrated to one or two orders of magnitude above the larger, less soluble PNAH."

Measurements of PAH levels in several different mollusk tissues from various Oregon Bays in recent studies concer with earlier results. In a 1978 - 1979 study, (Mix and Schaffer, 1983b), concentrations of PAHs in clams, Mya arenaria, from an industrialized bayfront averaged 2775.5 ppb (dry wt.), significantly higher than the average, 381.5 ppb (dry wt.), for clams from more remote areas of Coos Bay. In relatively pristine Oregon Bays such as Tillamook and part of Coos, Mix again found that mollusks contained baseline concentrations of PAH, approximately 50 ppb (wet wt.), (Mix, 1984), or 250 ppb (dry wt.). In this study also PAH concentrations in mollusks in more highly industrialized areas were significantly higher. The highest concentrations Mix found were in mollusks from a heavily industrialized site in Yaquina Bay where the PAH concentrations ranged from 3375 - 6625 ppb (dry wt.).

For the period January - May of 1979 scientists (Mix, et al., 1982) compared BaP concentrations in somatic tissue to concentrations in gonadal tissues of bay mussels from the site in Yaquina Bay in which PAH concentrations were highest in mussels in an earlier study (Mix and Schaffer, 1979) to see if changes in levels in these tissues could explain the winter - spring fluctuations in BaP oncentrations encountered earlier (Mix and Schaffer, 1979). They concluded that storage of the organic contaminant occurred primarily in somatic tissues compared to gonadal tissue, even during spring spawning. In another study (Mix, 1984) Mixconcluded that mollusks accumulate more PAHs in winter than in other seasons even though winter is not a time of lipid buildup.

Dr. Mix investigated the possibility of detrimental effects of organic contaminants in the form of abnormalities on Oregon Bay mollusks in several studies. He found that the prevalence of cellular proliferative disorders in mussels from two sites in Yaquina Bay where mussels always contained high levels of BaP was significantly higher than prevalence for this type abnormality in two sites where mussels had low or no detectable levels of BaP (Mix, 1979).

Later (Mix, 1982) Mix also reported results of histological examination of mollusks for cellular proliferative disorders resembling neoplastic conditions in vertebrates. Characteristic cells were not found in clams from Coos Bay or mussels from Tillamook Bay. The disorder was rarely found in a mussel population from a relatively clean site across Yaquina Bay from the site where high PNAH concentrations were encountered in mussels. These abnormalities, however, were quite prevalent in Yaquina Bay mussels with the highest PNAH measurements.

Although no causative relationship of high PAH concentrations in mollusk tissue to occurrance and prevalence of cellular disorders has yet been established, (Mix, 1982; Mix, 1983; Mix, Personal Communication, June 24, 1985), the belief exists (Malins, Personal Communication, June 26, 1985), that a cause - effect relation will some day be shown. In Mix's study of haematic neoplasms in subpopulations of bay mussels, Mytilus edulis L., in Yaquina Bay, 1976 - 1981, he found significant differences in the occurrance of the disorder were related to geographical location. The mussel subpopulations differed in their proximity to sources of anthropogenic contaminants (Mix, 1983).

Mix included a discussion of the risk to human health of organic contamination levels of Oregon Bays in one of his reports (Mix, 1979). He reported that blue mussels, Mytilus edulis, from Yaquina Bay are heavily contaminated and accessible to Oregon shellfishermen, but are not heavily exploited by them. High concentrations of BaP were found in Mya arenaria from certain industrialized areas of Coos Bay and these soft shell clams are accessible to the public, but they are not usually harvested from those particular sites. These two species occupy different habitats. M. edulis are found on pilings and rocks in low to high salinity; M. arenaria are found in soft or sandy mud in low salinity. Mix found other clam species and a cockle species that are heavily exploited to be only lightly contaminated. In 1979 Mix saw "no clear risk to users of the shellfish resources of Oregon at the present time." He did, however, have this word of caution: "Since humans possess the enzymes necessary to convert unaltered PNAH to highly mutagenic or carcinogenic derivatives. . .(Mollusks retain polynuclear aromatic hydrocarbons.). . .sufficient quantities could pose a public health problem."

OREGON BAYS

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SAN FRANCISCO BAY

San Francisco Bay shown in figure 1 is located on the western edge of California with about 1/3 of the state north of it. Large metropolitan areas such as the cities of San Francisco, Berkeley, and Oakland discharge over 500 million gallons of treated sewage into the Bay each day (Jung, et al. 1984). There are also some fairly rural/suburban areas along the shores of the Bay. Heavy traffic is obvious all around the bay, which is transversed east to west by four large bridges, and north to south at it's opening to the Pacific Ocean by the famous Golden Gate Bridge. Sailboats and other recreational boats as well as ferries on their regular routes are seen daily in the Bay. There have been recent declines in productivity and condition of fish important to the commercial fisheries of San Francisco Bay.

Organic Contaminants in Sediments

Robert Spies (Personal Communication, October 7, 1985) in describing his recent studies of sediments in San Francisco Bay concentrated on the difference between levels of contaminants found in the urbanized, central section bounded on the east by Berkeley, Oakland, San Leandro, Alameda, etc. and levels of contaminants in San Pablo Bay. Since San Pablo Bay is a less urbanized area, he treated it as a reference site. Spies, et al. (1985a) found about twice the level of total identified PAHs in mid-Bay sediments at Berkeley and Oakland as in San Pablo Bay sediments. Following is a summary of concentrations, shown in ppb (dry wt.), of organic contaminants in sediments reported in Spies, et al. (1985a):

Location	Total PAH	Total PCB	Total DDT
San Pablo Bay	2600	30	84
Berkeley	4600	8	32
Oakland	5660	24	76
Alameda	470	5	12

Spies, et al. (1985b) also reported concentrations of organic contaminants in sediments from San Francisco Bay. They took two sediment samples 100 meters apart in San Pablo Bay, two in Richmond, and two in Berkeley, and sampled one site each in Oakland and Alameda. It was apparent from their sediment data that "samples taken only 100 meters apart can have disparate contaminant levels." Their analysis identified in all their sediment samples a compound, benzothiozole-2, which had not to their knowledge previously been identified as a contaminant in marine samples. It is an anti-oxidant and an impurity in compounds used in making rubber. It is widely used in making automobile tires. The presence of benzothiozole-2 and of combustion PAHs suggested that San Francisco Bay's contamination may be from motor vehicle wastes entering the Bay as street runoff.

Fong, et al. (1982) described the procedure developed and used by the San Francisco District Corps of Engineers for determining impacts of dredging and disposal and the results they obtained after a year of testing sites in San Francisco Bay in 1979. They studied sediments from Oakland and Richmond harbors, the Main Ship Channel (outside the Golden Gate), Mare

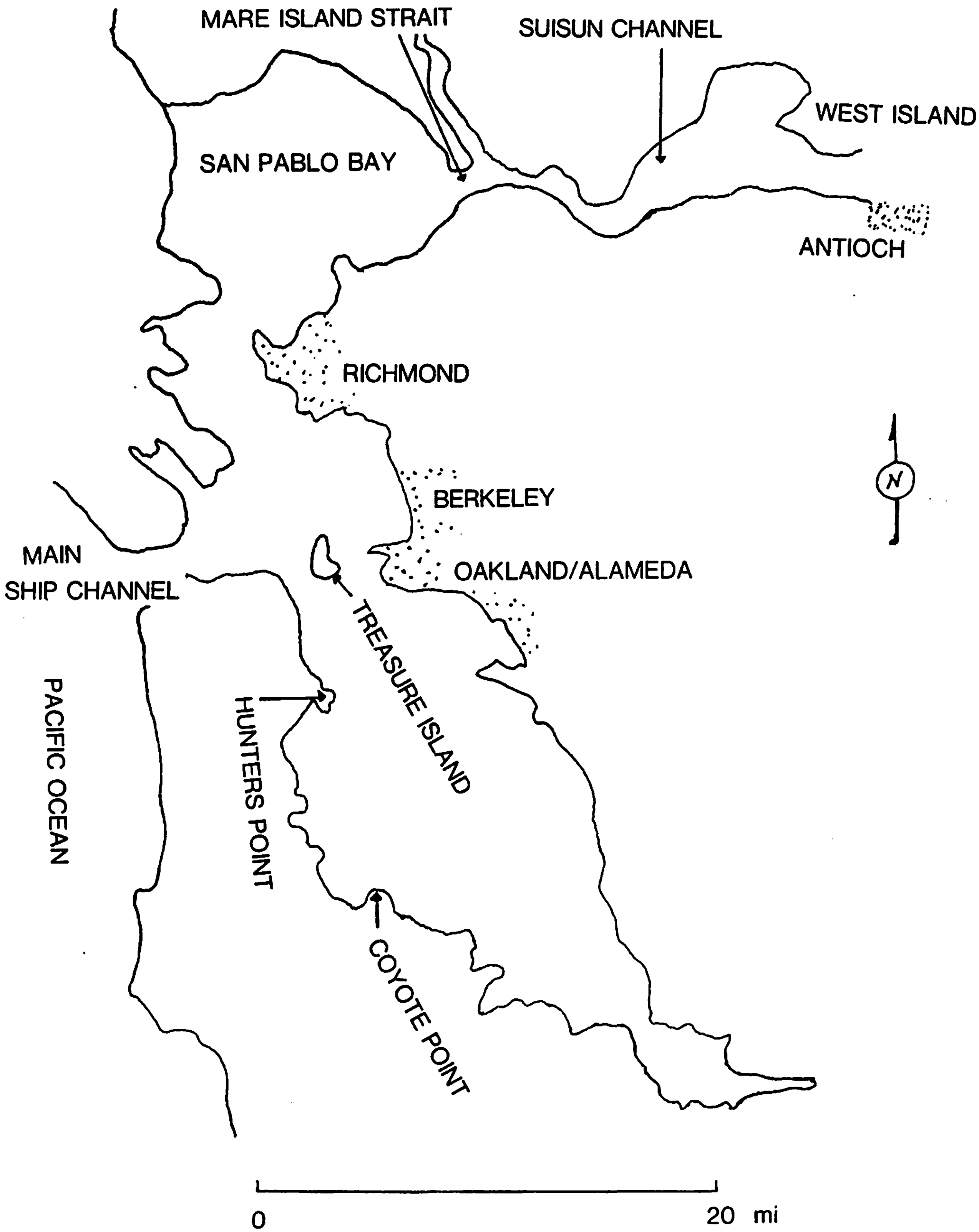

Figure 1. San Francisco Bay

Island Strait, and Suisun Channel first for physical characteristics. They found that Suisun and the Main Ship Channels contained sediments which are predominantly sand, so they considered further testing unnecessary. They analyzed elutriates from Oakland Inner and Outer Harbor and Oakland Army Terminal sediments and from Richmond Harbor and Mare Island Strait sediments. They compared concentrations of contaminants in these sediment elutriates with concentration in water samples from two disposal sites in San Francisco Bay. They found concentrations of petroleum hydrocarbons and PCBs in most elutriates significantly ($p=.95$) lower than concentrations in disposal site water and only considered further testing (bioassay) necessary for Oakland Army Terminal sediments.

Organic Contaminants in Mollusks and Crustaceans

We found data about concentrations of organic contaminants in mollusks of San Francisco Bay in Butler (1973), Risebrough (1978), and in California State Mussel Watch reports from 1977 sampling through 1983.

Butler (1973) monitored pesticide levels in mollusks in California from 1966 through 1972. The average maximum DDT residue he found in San Franciso Bay samples was 3,840 ppb (dry wt.). Maximum DDT residue was highest (11,400 ppb, dry wt.) in mollusks at the West Island sampling site. Butler found an average maximum dieldrin residue of 120 ppb (dry wt.) in mollusks from San Francisco Bay sampling sites. He found the highest maximum dieldrin residue (215 ppb, dry wt.) in mollusks at Coyote Point. Butler detected PCBs in mollusks from most San Francisco Bay sites but could not quantify them.

Risebrough (1978) gave concentrations (acquired via personal communication with L. H. DiSalvo) of PAHs in hermit crabs from the Point San Pablo area: 500 ppb (dry wt.) for entire body; 21,800 ppb (dry wt.) for eggs. The hepatopancreas of a crab, *Cancer antennarius*, from the same area contained 3,100 ppb (dry wt.) PAH while no PAH was detectable in a control crab from a coastal area.

Risebrough (1978) reported concentrations of DDT and its derivatives and PCB concentrations in mussels, *Mytilus edulis*, collected in April, 1976 from 29 sites in San Francisco Bay:

120.6 ppb (dry wt.) average total DDT + derivatives/site
560.7 ppb (dry wt.) average PCB conc./site

Risebrough (1980) as part of the California State Mussel Watch program reported concentrations of some individual PAHs in *Mytilus edulis* from sites in San Francisco Bay. Concentrations in ppb (dry wt.) of each PAH in mussels from two of these sites follow:

	Hunters Point cement pilings	Treasure Island wooden pilings
Fluorene	< 3	9
Dihydroanthracene	< 6	7
Phenanthrene	6	1000
1-Methyl phenanthrene	5	73
Fluoranthene	11	4200
Pyrene	140	1900
Chrysene	200	3800
Benzo (a) pyrene	< 6	2000

Mussels from other sites contained lower levels of the various PAHs, some at a level too low to accurately quantify.

Risebrough (1980) gave a level of 790 ppb (dry wt.) PCBs and 56 ppb DDE for San Francisco Bay mussels analyzed in winter of 1978, but gave no values for these contaminants in San Francisco Bay mussels prior to that year.

In State Water Resources Control Board's (1982) summary of California Mussel Watch 1980-81, they reported monitoring mussels along the open coast and within San Francisco Bay in their general "San Francisco Bay Region." They found that levels of toxicants increased at sites nearest the Bay's entrance. They found PCB concentrations in Bay mussels about ten times higher than concentrations in mussels from bays north and south of San Francisco. They reported that concentrations of the pesticide chlordane were ten times higher than those measured in Northern California and were similar to concentrations in mussels found near urban centers in Southern California. They found that a site near the mouth of the Bay produced mussels with substantially higher concentrations of dieldrin in 1980 than in their earlier measurements in 1979.

Martin, et al. (1982) reported data for organic compounds in mussels, Mytilus californianus, from 1980 California State Mussel Watch sites in San Francisco Bay. They transplanted mussels from Bodega Head (an area considered "clean") to stations in the Bay, then collected them after about five or six months and analyzed them. Average concentrations in ppb (dry wt.) among nine stations for each of three compounds follow:

total chlordane ...	66.8
dieldrin	31.6
PCB (1254)	1136.7

Stations where mussels contained the highest PCB concentrations were at Hunters Point (1800 ppb, dry wt.) and Treasure Island (1500 ppb, dry wt.). PCB levels in mussels from San Francisco Bay stations were 10 to 70 times greater than those from the reference station at Tomales Bay. Chlordane levels were highest in mussels from the south San Francisco Bay stations. "Dieldrin showed little spatial variation within San Francisco Bay" (Martin, et al., 1982).

Ladd, et al. (1984) reported results of two years (1981-1983) of monitoring by the California State Mussel Watch project. This study repeated the 1980-1981 study in order to detect possible changes in baseline levels of pollutants and spatial and temporal patterns of distribution. Results from 1981-1983 showed about the same trends as previous years.

Spatial patterns for the three pesticides pp' DDE, chlordane, and dieldrin in San Francisco Bay were all different. Concentrations of pp' DDE were greater in mussels from the north Bay than in the south. Concentrations of total chlordane were highest in mussels from the south Bay, intermediate in mussels from the north, and lowest in mussels tranplanted to the central Bay. Dieldrin concentrations increased from the north to the south of the Bay.

Ladd, et al. (1984) noted temporal patterns with respect to some pesticide concentrations in mussels, but not for all pesticides. He noted that no significant changes had yet occurrred in pp' DDE concentrations in Bay mussels since the beginning of the mussel study about three years earlier. He did note "consistently lower chlordane values between 1980 and 1983." Concentrations of dieldrin in transplanted mussels, though, increased slightly in this mussel watch study at all but one of the San Francisco Bay stations.

Ladd, et al. (1984) reported concentrations of PCBs (Arochlor 1254) in San Francisco Bay mussels. Mussels collected during the first two months of 1982 contained a station average of 191 ppb (dry wt.). Those collected in December, 1982 contained an average concentration per station of 296 ppb (dry wt.).

Organic Contaminants in Fish

In 1979 the State Water Resources Control Board decided to commit funds for investigation of the effects of water pollution on the striped bass (Morone saxatilis) fishery in San Francisco Bay-Delta. Jung, et al. (1981) reported progress of this project, called the Cooperative Striped Bass Study. Striped bass from Coos River Oregon were larger and had lower concentrations of pesticides in their ovaries than San Francisco Bay-Delta bass. Average concentrations of some pesticides in livers of bass collected from four Antioch, California locations follow:

PCB-1260	5262.5 ppb, dry wt.
Total DDT and metabolites	4665.0 ppb, dry wt.
Total chlordane	1511.0 ppb, dry wt.
Dieldrin	269.0 ppb, dry wt.

In the third progress report of the Cooperative Striped Bass study Whipple, et al. (1983) reported that PCB concentrations were significantly higher in gonads of San Francisco Delta fish (San Joaquin and Sacramento Rivers) than in gonads of Coos River, Oregon fish. PCB concentrations in muscle tissue of Delta fish were also higher than in Coos River fish. Some data about pollutant levels in striped bass tissue, reported by Whipple, et al. (1983) are shown here in ppb (dry wt.):

SFB-D,	Monocyclic Aromatic Hydrocarbons in liver:	50-50000
1978-80	" " " in gonad:	50-50000

SFB-D, 1978-80	DDT and metabolites in liver: 350-7000 " " " in gonad: 1650-22500
SFB-D, 1978-80	PCBs in liver: 1250-65000 " " gonad: 4050-65000
Antioch, 1978	PCB-1260 in ovaries: 10000 DDT and metabolites in ovaries: 6460 Total chlordane in ovaries: 2310 Dieldrin in ovaries: 480
Antioch, 1980	PCB-1260 in ovaries: 850 DDT and metabolites in ovaries: 3065 Total chlordane in ovaries: 1100 Dieldrin in ovaries: 155

SFB-D = San Francisco Bay-Delta

In the Summary Report of the Cooperative Striped Bass Study (COSBS), Jung, et al. (1984) noted some interesting comparisons and trends in contaminant levels in striped bass. Bass collected from Coos River, Oregon in 1980-1981 "had significantly lower concentrations of contaminants than those collected from the Bay-Delta, even though the Oregon fish were 2 to 3 times older and larger than the Delta bass collected." Fish from the Hudson River had higher concentrations of PCBs, chlordane, and dieldrin in gonads than fish from the San Francisco Bay-Delta; but San Francisco Bay-Delta fish had higher levels of DDT and metabolites than Hudson River fish.

Both DDT and PCB residues in bass declined steadily in 1972, but showed a slight increase in recent COSBS samples. In spite of the general decline, [See Figures 2 and 3 from Jung, et al.'s (1984) Figure's 2.2 and 2.4], the COSBS found DDT residues at levels that could have detrimental effects on egg and larval development. They found PCB residues in bass eggs at levels strongly correlated with poor hatching success in Atlantic salmon eggs and in some cases levels in bass fillets that were above protective limits. "The concentrations of monocyclic aromatic hydrocarbons (MAH) in striped bass ovaries were sometimes as high as those observed in the laboratory to have adverse effects on the condition of eggs." (Jung, et al. 1984).

Spies, et al. (1985b) found that chlorinated hydrocarbon levels were generally higher in livers than in eggs of starry flounder they sampled from San Francisco Bay. They also found that the Richmond-Berkeley area flounder had the highest mean concentration of chlorinated hydrocarbons in their livers and San Pablo Bay flounder had the lowest mean concentration. Folowing is a summary of some of their DDT and PCB concentration data for eggs and livers of starry flounder (Values are means of concentrations from all spawning female fish analyzed from each area, shown here in ppb, dry wt.):

	DDT in eggs	DDT in liver	PCBs in eggs	PCBs in liver
San Pablo Bay	315	1815	461	3105
Berkeley	426	4722	588	9270
Richmond	388	7638	568	11675
Alameda Naval Station	633	3440	653	6295
Oakland, 7th. St. Pier	410	2140	390	2888

Annual mean concentration of DDT + metabolites
in Delta fish, 1964-81

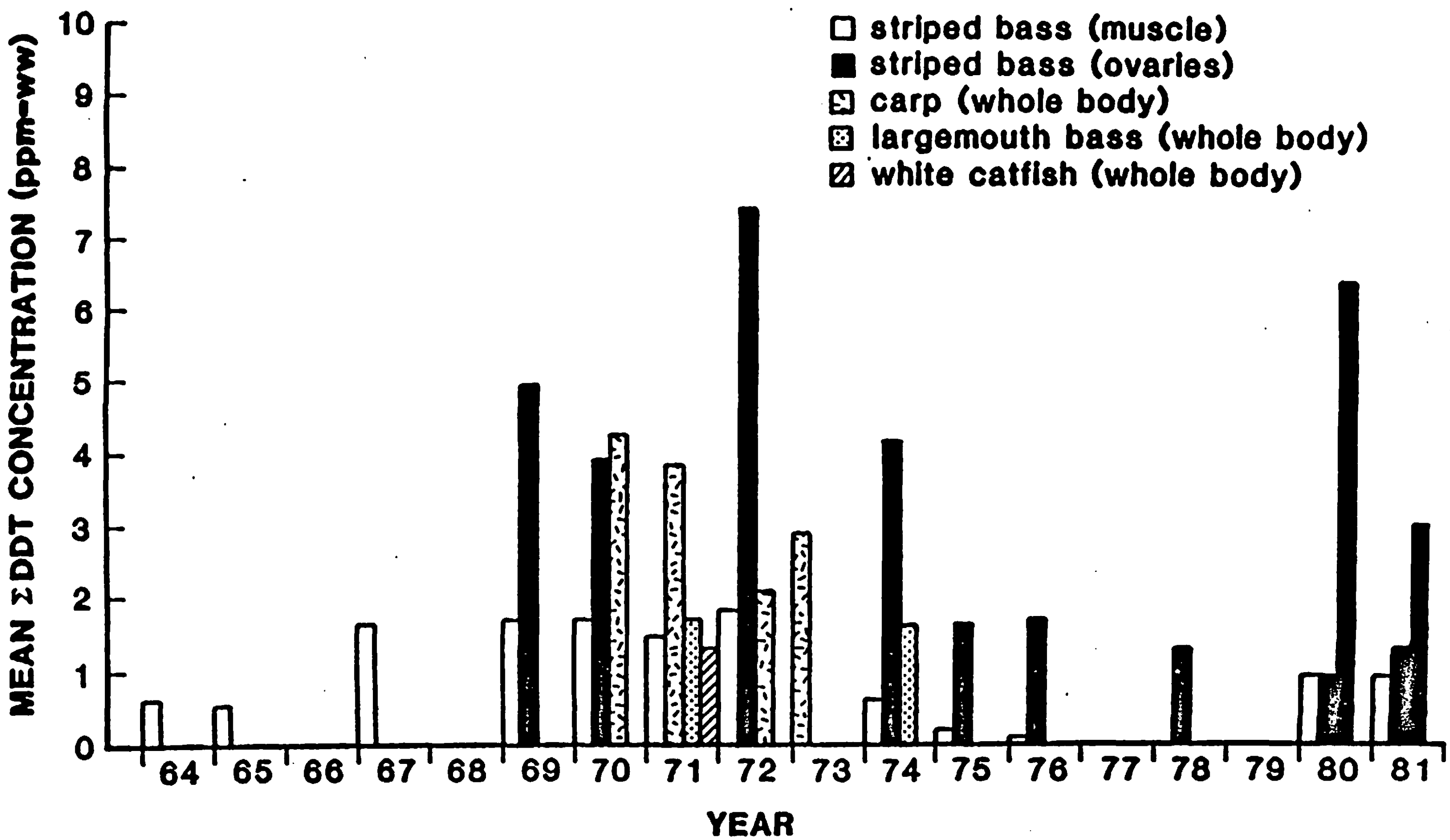

Figure 2. (Jung, et al., 1984)

Annual mean concentrations of PCB residues in Delta fish, 1969-81

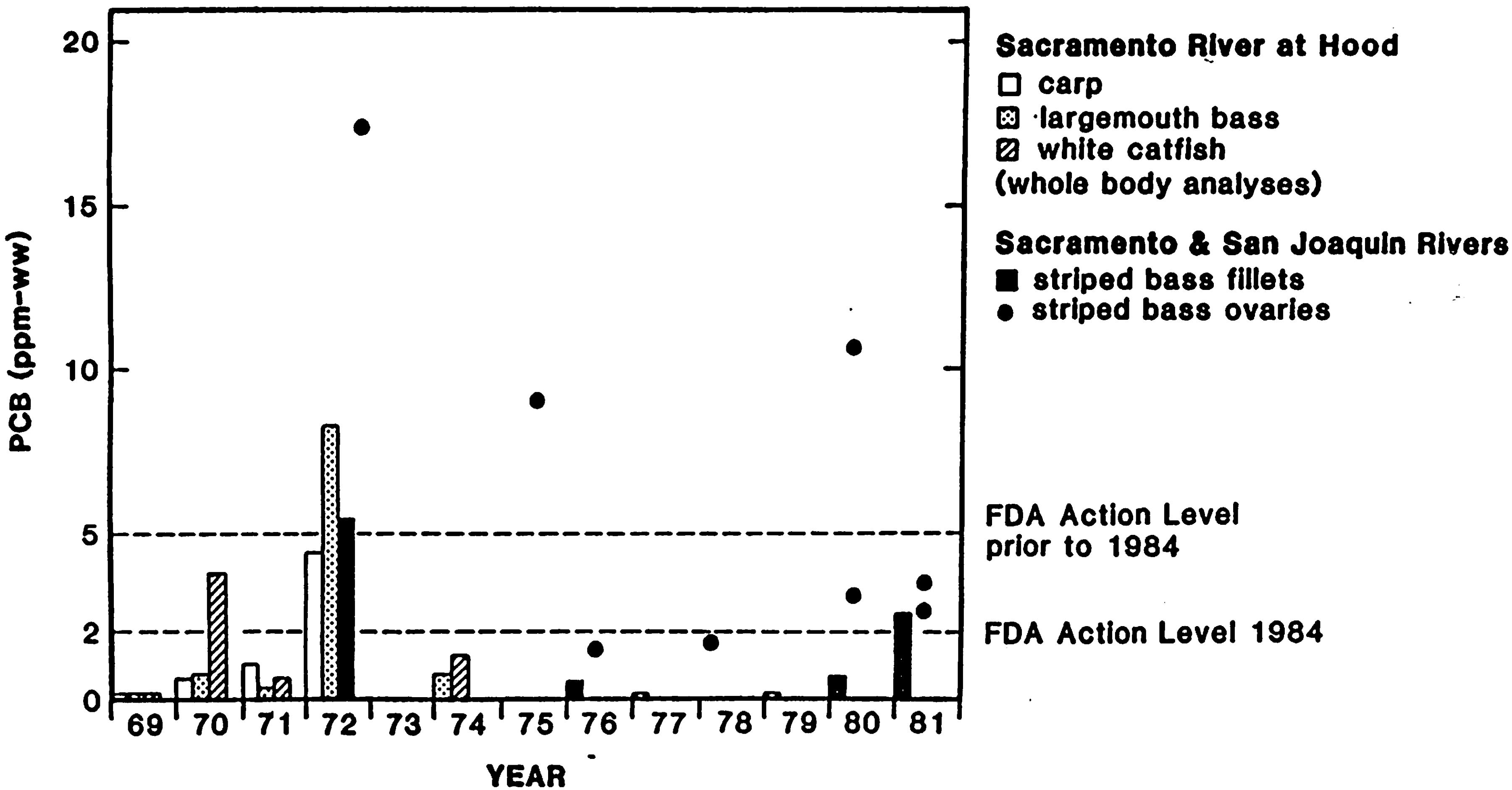

Figure 3. (Jung, et al., 1984)

Spies, et al. (1985a) measured and compared concentrations of toxic organic contaminants in starry flounder for several sites within San Francisco Bay. They found that the urbanized central bay sites had significantly higher concentrations of chlorinated and polynuclear aromatic hydrocarbons in fish livers than the less urbanized San Pablo Bay site. Spies, et al. (1985a) combined liver extracts from all fishes at each station and used EPA method 610 (using HPLC equipped with a fluorescence detector) to produce one number per station for total PAH value. Following is a summary of the average total PAH, total PCB, and total DDT they found in starry flounder liver extracts:

Locations	PAH	PCB	DDT
San Pablo Bay	700	1700	1000
Berkeley	13000	8000	2050
Oakland	7000	11500	2000
Alameda	70000	11000	2000

[Concentrations converted from ppb, wet wt. to ppb dry wt. via dry wt = .20 (wet wt.); therefore ppb, dry wt. = 5(ppb, wet wt.)]

Effects of Organic Contaminants: Invertebrates

Armstrong, et al. (1971) explained the development of the water quality criterion that a waste discharge must receive a 25 to 1 dilution of its relative toxicity in San Francisco Bay. During the the San Francisco Bay-Delta Water Quality Control Program conducted in the 1960s it became evident that toxicity criteria were needed to plan for control of toxic discharges and that control must be related to changes in aquatic biota. Previous studies had shown good DO and BOD in the Bay, yet there were more fish kills and fish populations were decreasing. They proceeded to calculate benthic animal species diversity indices to measure the relative health of the Bay, then to correlate the indices with values for chlorosity and sand content of sediments from non-polluted areas. Data indicated that damage would be observed in benthic fauna where the relative toxicity is not reduced by a factor of 25.

Nichols (1979) reported on benthic community structure in San Francisco Bay determined by his sampling in 1973 and from historical data. Numbers of benthic macrofaunal species were greatest in the central region of the Bay near San Francisco, but he found highest total biomass of benthic macrofauna in the southern region of the Bay. Explanation for his findings included differences in salinity and sediment texture of these regions as well as human influences such as introduction of species and diversion of fresh water flow. It might be informative to determine more current benthic community structure data and analyze these data along with sediment structure, salinity and sediment contaminant concentration data. Such a study might offer clues regarding possible effects of organic contaminants and/or other factors on community structure.

Starry Flounder Abnormalities

Spies, et al. (1985a) examined 104 starry flounder from San Francisco Bay for various histopathological abnormalities. They found no fish with liver abnormalities in Alameda samples, only 7% in San Pablo Bay samples, 11% in Oakland samples, and 14% in Berkeley samples. We noticed that the areas with least fish liver abnormality (Alameda) and second least (San Pablo Bay) correspond exactly to the areas containing least (Alameda) and second least (San Pablo Bay) PAH in sediments from Spies, et al. (1985a) sediment data. Spies, et al. (1985a) noted that Berkeley was the site with the greatest extent of fish liver damage and was also the site containing the highest sediment concentrations of the toxic, carcinogenic PAH, benzo(a)pyrene. They found no abnormal eye tissue in any of the 104 fish and found only four fish with neoplastic or preneoplastic lesions. Two were from the Berkeley area and one each were from San Pablo Bay and Oakland. None were from Alameda. Spies, et al. (1985a) considered the relationship between levels of contaminants and abnormalities still not well established.

Robert Spies (Personal Communication, October 7, 1985) relayed that he and others do see effects on fish and on fish reproduction in San Francisco Bay, but that the $64,000 question everyone is trying to answer is what the consequences are on the population. Spies discussed the strong evidence that PCBs are having an effect. PCB content of eggs of fish that are captured in the field and spawned in the laboratory correlate with reduced reproductive success, particularly embryological success. Spies said that they had been worried that the negative correlation between percent reproductive success and total PCBs in eggs might somehow be a result of lipid content, so they normalized their values to micrograms PCBs/gram lipid and got an even more significant relationship. Instead of $p = .04$, they had a "p" value of about .02 or .01.

Spies (Personal Communication, October 7, 1985) also discussed the more indirect, complex relationship of mixed function oxidase (MFO) activity in fish to sublethal effects in these fish. They looked at the MFO activity of spawning female fish and found a highly significant ($p = .001$) negative correlation with percent fertilization and with embryological success ($p = .045$). According to Spies, et al. (1984) exposure of fish to aromatic hydrocarbons induces their MFO enzyme system. PCBs are inducers of MFO activity, so they may be the cause along with PAHs for increased MFO activity and consequently adverse effects on fish reproduction, etc. (Spies, Personal Communication, October 7, 1985). MFO enzymes are responsible for initial transformations of organic contaminants, most notably PAHs, into mutagens and carcinogens (Varanasi and Malins, 1977)

Spies, et al. (1985b) noted flounder populations from the urbanized Richmond-Berkeley area in the central region of the Bay had the highest liver concentrations of total chlorinated hydrocarbons, highest hepatic MFO activity and lowest fertilization success. San Pablo Bay flounder populations exhibited opposite traits. Their livers had the lowest concentrations of total chlorinated hydrocarbons, lowest hepatic MFO activity and their reproductive success was highest.

Striped Bass Abnormalities

Jung, 1981, in a progress report of the Cooperative Striped Bass Study referred to unpublished data from F. Fisher, California Department of Fish and Game showing an 11.8% incidence of open lesions in striped bass collected in 1979 from the Delta at the confluence of the Sacramento and San Joaquin Rivers and an incidence of 8.4% in bass collected there in 1980. Toxic pollutants was only one of several hypotheses suggested as causes of the lesions.

Results in the third progress report of the Cooperative Striped Bass Study (Whipple, et al. 1983) include sublethal effects observed in San Francisco Bay-Delta striped bass. Fish with higher levels of monocyclic aromatic hydrocarbons (MAH) including benzene in their liver had more lesions and host reactions to parasites in general, had redder or hemorrhaged livers, had significantly poorer egg condition and lower fecundity. Fish with higher levels of MAH in their ovaries had poorer ovary and egg condition. Also fish with higher levels of MAH in their gonads had significantly more monocytes in their peripheral blood. There did not appear to be any correlation between gonad PCB concentrations and egg, body, or liver condition of the fish.

Whipple, et al. (1983) hypothesized that "chronic exposure to pollutants significantly decreases condition, growth and/or reproduction, when fish are stressed by extremes in natural environmental factors, such as increased salinity or temperature, or decreased flow." They had several hypotheses about interactions and mechanisms which may cause reduction of striped bass populations in the San Francisco Bay-Delta area.

Jung, et al. (1984) and Whipple (1984) reported that San Francisco Bay-Delta fish were in poorer health than Coos River, Oregon fish. Jung, et al. (1984) also observed egg condition was significantly poorer in San Francisco Bay-Delta fish than in fish from any other area they had sampled. They estimated that reduction in fecundity per striped bass spawner in San Francisco Bay-Delta due to the combined effects of pollutants and parasitism was at least 50 percent in 1978. They thought that "sublethal exposure to pollutants combined with parasitic infections may increase the mortality of older fish and reduce their fecundity ... during periods of extreme stress." They think long term presence of pollutants in striped bass may be of ecological concern since they may affect the reproductive success of the fish by damage to the reproductive condition of parent fish, to the egg condition and to the development and survival of fertilized eggs and larvae.

Whipple (1984) noted that some pollutants found in relatively high levels in adult striped bass "showed strong correlations with poor health and condition, parasite burdens and impaired reproduction." She reported that DDT, though not the metabolites DDD and DDE, in liver and gonads were associated with abnormal egg development and death of eggs. "Delayed egg maturation rates ... were associated with PCBs in ovaries" (Whipple 1984). She listed pollutants most implicated in deleterious effects on fish in order. The top four pollutants on her list were organic compounds: ethylbenzene; 1,2-dimethylcyclohexane; benzene; and toluene. Results of laboratory experiments concerning effects of benzene disclosed that benzene induced egg resorption in prespawning females similar to resorption in field fish; benzene accelerated and increased inflammatory response in the fish to roundworm larvae; benzene was associated with destruction of blood cells followed by increased production of immature red and white blood cells.

Impacts

In personal interviews on October 7, 1985 with both Jeanette Whipple at Tiberon Fisheries Laboratory, the Southwest Fisheries Center for the National Marine Fisheries Service and with Robert Spies at the Lawrence Livermore National Laboratory in Livermore, California it was clear that there has been a definite impact on fisheries in San Francisco Bay. They cannot yet determine if contamination is the cause of the reduced commercial fish densities or if it is one contributing factor.

Whipple's (Personal Communication, October 7, 1985) data analyses of striped bass results show that many pollutants do relate to certain problems in the fish. She thinks that a combination of DDT and other hydrocarbons cause problems. They are positiviely correlated. DDT accounts for most of the variation in step-wise regressions; but in no case did egg resorption occur where only DDT was present. Since laboratory experiments have shown MAHs can induce effects in fish similar to effects observed in nature, Whipple is pretty sure MAHs are involved, and perhaps PAHs as well, in egg resorption.

Neither Whipple nor Spies seemed to consider over-fishing a major factor in the decline of fish populatins in the Bay. They both discussed possible effects of changes in net flow in the Bay and effects of net flow combined with effects from pollutants. Spies also questioned nutrient runoff or possibly insufficient primary productivity as contributing factors.

Two recent drought years may have been a big factor in increasing adult mortality (Whipple, Personal Communication, October 7, 1985), though not enough measurements are being taken at present to document whether things are really worse in dry years or wet years (Spies, Personal Communication, October 7, 1985). Spies commented that a certain amount of rain can bring very high concentrations of contaminants into the Bay in runoff from the land. Continued rain could provide enough dilution, though, to flush out the Bay.

Both Spies and Whipple commented about concern that exists about possible detrimental effects of an aquaduct to shunt water from northern to southern California.

Whipple (Personal Communication, October 7, 1985) mentioned somewhat of a public health problem in that PCBs are found over FDA limits in body tissues of older Bay fish. She also verbalized her concern for the decline of striped bass in San Francisco Bay. It is almost to the point of a reduced gene pool which could lead to extinction because genetic variability may be so reduced.

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SOUTHERN CALIFORNIA

In this summary of information about organic pollution and its effects in southern California estuarine waters, we concentrate attention on the Los Angeles and Long Beach Harbor area and the nearby Palos Verdes shelf. According to the 1986 World Almanac the Los Angeles/Long Beach - Anaheim - Riverside metropolitan area's human population (11,497,568) was second in the U.S.A. in 1980 only to the New York City metropolitan area. Figure 1 shows some key locations along the coast of southern California and includes an insert showing Los Angeles and Long Beach Harbors and associated San Pedro Bay.

Literature reporting PCB and pesticide levels in tissues of fish, mussels, and a few other indigenous fauna and in sediments was ample. Data about PAH levels were available, but not as abundant. Several investigators discussed possible links between organic contaminants and abnormalities found in southern California fishes. Quite a few reports of studies included discussions of benthic community structure at sites in the area from different periods in time. Investigators usually looked for signs of geographic trends possibly related to location of sewage effluent outfalls or petroleum seeps, not attempting to check for correlation between benthic community structure and concentration of specific organic contaminants in effluents, sediments and in the biota themselves.

Organic Contaminants in Sediments

There have been many reports of concentrations of organic contaminants in southern California sediments. Most investigators measured levels of PCB, DDT, and other pesticides. Some discussed temporal and geographical trends. In one of the few early studies of PAHs, Orr and Grady (1967) reported about the distribution of perylene in basin sediments. They found a mean concentration of 167 ppb (dry wt.) of it in their four sediment stations in Santa Barbara Basin.

Tables 1a & 1b summarize PCB and pesticide concentration data from studies of southern California sediments from 1971 through 1984. You may note particularly high levels in sediments off the Palos Verdes Peninsula. This area receives Los Angeles County Sanitation District outfall. Soule and Oguri (1980) in their report about the condition of the harbors in Los Angeles and Long Beach said "The Whites Point outfall of the Los Angeles County Sanitation Districts was the major source of DDT in the local area until it was controlled..." They also mentioned temporal trends in pesticides. "The levels of DDT increased by an order of magnitude in 1978 over the 1973-74 levels, but most harbor stations were free of DDT" They noted that since 1973-74 total PCBs have decreased fourfold and in some areas have been eliminated. Schafer (1978) gave combined annual emission rates of total DDT and total PCB of southern California's five largest municipal wastewater discharges from 1971 through 1977 indicating a steady decreasing trend. Values are shown in kg/year:

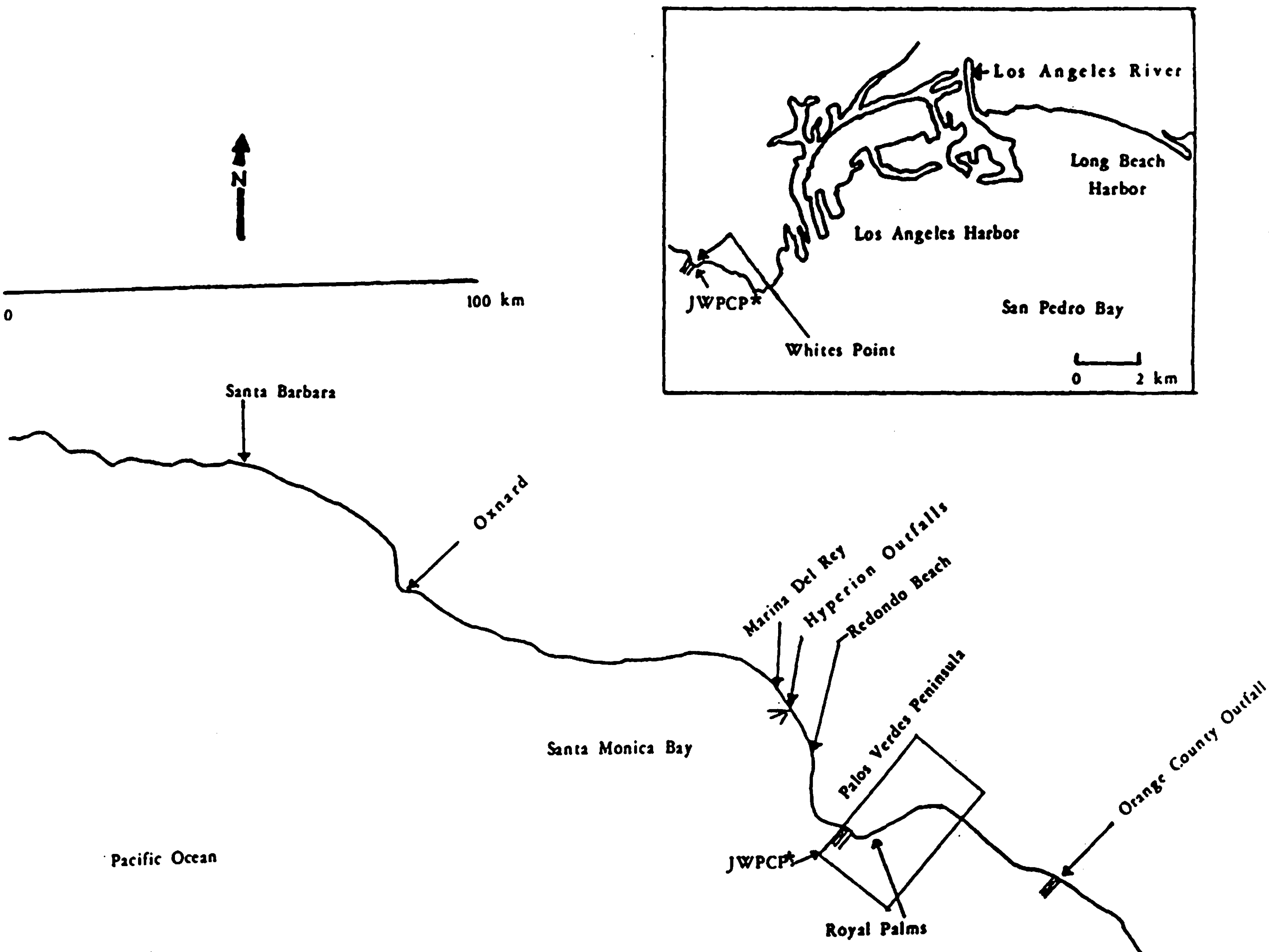

Figure 1. Southern California
*Los Angeles County Sanitation Districts Outfalls

Table 1. Concentrations of organic contaminants in southern California sediments.
A. Concentrations of DDT, Chlordane, etc. in sediments.

Reference	Sampling Year	Location	Mean Conc. (ppb, dry wt.)
SCCWRP (1973)	1971	Santa Monica Bay around the Hyperion Outfall System	ΣDDT: 513 (100-2,400)[1]
SCCWRP (1973)	1971	Orange Co. Outfall area north of Newport Beach	ΣDDT: 48
SCCWRP (1973)	1972	Santa Monica Bay around the Hyperion Outfall System	ΣDDT: 130
Smith (1973)	1972	Los Angeles Harbor	DDE + DDD: 3,655
Young, et al. (1977)	1972	off Palos Verdes Peninsula	ΣDDT: 65,837
SCCWRP (1974)	1972	off Palos Verdes Peninsula	ΣDDT: 61,327
Chen + Lu (1974)	1973	Los Angeles and Long Beach Harbors	ΣDDT: 491
Chen + Lu (1974)	1973	San Pedro Bay	ΣDDT: 252
Emerson (1974)	?	Inner Los Angeles Harbor, at S. Calif. gas pipeline crossings	ΣDDT: 864
Young, et al. (1977)	1975	off Palos Verdes Peninsula	ΣDDT: 40,976
Young + Heeson (1978)	1976	Los Angeles Co. Sanit. Dist. Outfall Zones off Palos Verdes	ΣDDT: 94,000
Bascom, et al. (1979)	1977	Santa Monica Bay	ΣDDT: 227*
Bascom, et al. (1979)	1977	Palos Verdes	ΣDDT: 175,000*
Bascom, et al. (1979)	1977	S. San Pedro Bay	ΣDDT: 3*
Bascom, et al. (1979)	1977	Point Loma, near San Diego	ΣDDT: 1*
Bascom, et al. (1979) In: Word + Mearns (1979)	?	Southern California Bight	ΣDDT: 20

Soule + Oguri (1985)	1984	Entrance Channel, Marina del Rey	Chlordane: 721
Soule + Oguri (1985)	1984	Oxford Flood Control Basin, Marina del Rey	Chlordane: 468
Bascom (1985)	83-84	Santa Monica Bay	ΣDDT: 120
Bascom (1985)	83-84	Santa Monica Bay	DDT metabolites: 800
Bascom (1985)	83-84	Palos Verdes	ΣDDT: 19,100
Bascom (1985)	83-84	Palos Verdes	DDT metabolites: 78,800

B. Concentrations of ΣPCB in sediments.

SCCWRP (1973)	1971	Santa Monica Bay, around the Hyperion Outfall System	924 (10-12000)[1]
SCCWRP (1973)	1971	Orange Co. Outfall area north of Newport Beach	29
SCCWRP (1973)	1972	Santa Monica Bay, around the Hyperion Outfall System	64
Chen + Lu (1974)	1973	Los Angeles and Long Beach Harbors	876
Chen + Lu (1974)	1973	San Pedro Bay	176
Emerson (1974)	?	Inner Los Angeles Harbor, at S. Calif. gas pipeline crossings	2,488
Young, et al. (1977)	1975	off Palos Verdes Peninsula	3,400
Young + Heeson (1978)	1976	Los Angeles Co. Sanit. Dist. Outfall Zones, off Palos Verdes	10,900
Bascom, et al. (1979)	1977	Santa Monica Bay	288*
Bascom, et al. (1979)	1977	Palos Verdes	10,900*

Bascom, et al. (1979)	1977	S. San Pedro Bay	254*
Bascom, et al. (1979)	1977	Point Loma, near San Diego	37*
Bascom, et al. (1979) In: Word + Mearns (1979)	?	Southern California Bight	10
Bascom (1985)	83-84	Santa Monica Bay	18
Bascom (1985)	83-84	Santa Monica Bay	PCB metabolites: 6,700
Bascom (1985)	83-84	Palos Verdes	670
Bascom (1985)	83-84	Palos Verdes	PCB metabolites: 36,400

1 Ranges are shown in parentheses under mean value when the range of concentrations is wide.

* maximum concentration

	1971	1972	1973	1974	1975	1976	1977
ΣDDT	21700	6600	4120	2120	1989	1673	1219
ΣPCB	8730	9830	4620	9390	6011	4310	2183

Data from Chan and Lu (1974) in Table 1b illustrate the relatively high concentrations of PCBs in Los Angeles-Long Beach Harbor surface sediments in 1973 compared to San Pedro Bay sediments.

Young, et al. (1976) considered DDT and its metabolites the most important of the synthetic organic contaminants of the marine ecosystem off southern California. They blamed municipal wastewaters passing through the Joint Water Pollution Control Plant (JWPCP) outfalls of the Los Angeles County Sanitation Districts for over 95 percent of the DDT introduced into southern California Bight waters in 1972. Total output of DDT from southern California's five largest municipal wastewater dischargers decreased by 20-30 percent per year 1971-1981, though "PCBs have not dropped much since 1979" (Bascom, 1983).

Young, et al. (1977) pointed out the persistance of chlorinated hydrocarbons in bottom sediments in southern California. "Despite major decreases in DDT and PCB emissions over a three year period by the dominant discharger to a coastal site, only minor ... decreases were observed in concentrations ... in bottom sediments" (Young, et al., 1977).

Organic Contaminants in Biota

According to Smokler, et al. (1979) total annual emissions of DDT from the Los Angeles County Sanitation District's underwater outfalls decreased through source control by a factor of 30 between 1971 and 1977; yet DDT residue levels in Dover sole did not decrease. Their study did, however, reveal a decrease in DDT residues in black perch flesh. They reported that in Spring, 1971, 50 percent of Dover Sole specimens collected from the Joint Water Pollution Control Plant of the Los Angeles County Sanitation Districts exceeded the U.S. Food and Drug Administration (FDA) guideline of 5 ppm, wet wt. DDT (25000 ppb, dry wt.). In Fall, 1971, 80 percent of the Dover Sole specimens exceeded this guideline. In Spring, 1975, 83 percent exceeded the guideline for DDT residue.

Young (1978) noted a gradual decline in levels of DDT and PCB in mussels from the Palos Verdes environment, specifically Royal Palms mussels, following initiation of the source and use control of these contaminants in the early 1970's.

The California State Mussel Watch Monitoring Program sponsored by the State Water Resources Control Board and performed by the California Department of Fish and Game began monitoring accumulation of metals and organic toxicants in marine mussels in 1977. Mussels proved to be excellent indicators of marine water quality. They are ubiquitous along California's coast. They concentrate toxicants in their tissues and, of course, they are stationary (Risebrough, et al., 1980; Ladd, et al. 1984; State Water Resources Control Board, Surveillance and Monitoring Section, 1982). According to the summary by the State Water Resources Control Board, Surveillance and Monitoring Section (1982), in considering results of Mussel Watch along the entire coast, "The Southern California Bight exhibits the greatest number of sites that have consistently displayed problematical levels of toxicants." In this same summary they reported finding chlordane concentrations about ten times higher in Marina del Rey mussels than in mussels from adjacent sampling stations. Marina del Rey mussels had chlordane concentrations 100 times those of Northern California mussels. They also reported strikingly higher DDT and PCB concentrations in Los Angeles-Long Beach Harbor mussels than in mussels from reference stations. DDT concentrations in these harbor area mussels were 300 times above oceanside reference station mussels. PCB levels were 10 to 100 times greater than reference station mussels.

Ladd, et al. (1984) discussed patterns of DDE and PCB distribution in mussels along California's coast. They noted that organic pollutants seemed to remain concentrated near point sources. They found that both pp' DDE and PCB concentrations were elevated in the Southern California Bight, specifically near major wastewater outfalls such as those around Royal Palms.

Mearns (1981) showed that concentrations of DDT in muscle tissue of popular seafood organisms from the Palos Verdes shelf during 1975-76 sampling period appeared to increase with increase in trophic level and with proximity to the benthic environment. He also reported that of seafood organisms analyzed, abalone, scallop, prawn, crab, sanddab, scorpionfish and boccacio, only the sanddab's DDT concentration exceeded U.S. Food and Drug Administration standard. Mearns and Sherwood (1977), however, reported that DDT concentrations in muscle tissue of many Palos Verdes fish specimens exceeded the FDA standard for safe human consumption.

Dunn and Young (1976) reported concentrations of benzo(a)pyrene in whole soft mussel tissue collected from southern California sites in 1974. Mytilus californianus from a rock substrate off the Palos Verdes peninsula contained 1.5 ppb, dry wt. benzo(a)pyrene; M. californianus from a rock substrate at the Long Beach breakwater contained 1.0 ppb, dry wt.; M. caliornianus from pilings in the Santa Barbara Harbor contained 0.3 ppb, dry wt.

Spies, et al. (1982) collected the common flatfishes, Citharicthys sordidus and C. stigmaeus from four areas during 1979-1980: (1) central Monterey Bay, a relatively pristine area in northern California, (2) a petroleum seep in the Santa Barbara Channel, (3) the Los Angeles sewer system discharge area in Santa Monica Bay and (4) the Los Angeles County Sanitation Districts outfall on the Palos Verdes shelf. Then they measured hepatic mixed function oxidases in these flatfish. They also conducted experiments in which they fed flatfishes diced squid without contaminants, diced squid augmented with seep-oil, and diced squid augmented with the PCB 1254. Their data showed that pollutant induced fishes could exhibit quantitative and qualitative differences in mixed function oxidase activity. Induction by petroleum hydrocarbon, PCB, or other xenobiotics from sewage discharge produced the same qualitative mixed function oxidase response. "The hepatic MFO system in the California flatfishes Citharicthys sordidus and C. stigmaeus is a sensitive and useful way to monitor biologically meaningful levels of xenobiotics in the coastal environment" (Spies, et al., 1982).

Following in Tables 2a and 2b and Tables 3a and 3b is a summary of data concerning concentrations of organic contaminants in fish and invertebrates of southern California.

Table 2a. ΣDDT concentrations in southern California fish in ppb (dry wt.).

Fish*	Location	Sampling Year	Mean Concentration**	Reference
Dover sole, muscle	Palos Verdes	71-72	65,000	SCCWRP (1973)
Dover sole, muscle	Santa Monica	71-72	4,500-36,000[1]	SCCWRP (1975)
Dover sole, muscle	Palos Verdes	71-72	16,000-225,000[1]	SCCWRP (1975)
Dover sole, muscle	Orange County	71-72	3,500-22,000[1]	SCCWRP (1975)
Dover sole, muscle	Palos Verdes	71-72	125,000	Young, et al. (1976)
Dover sole, muscle	L.A. Co. Sanitation Dists. outfall	1972	130,000	Young, et al. (1977)
Dover sole, muscle	Palos Verdes	1973	2,600-13,000[1]	SCCWRP (1974)
Dover sole, muscle	L.A. Co. Sanitation Dists. outfall	1973	80,000	Young, et al. (1977)
Dover sole, muscle	L.A. Co. Sanitation Dists. outfall	1974	75,000	Young, et al. (1977)
Dover sole, muscle	Santa Monica	74-75	3,000-8,500[1]	SCCWRP (1975)
Dover sole, muscle	Palos Verdes	74-75	48,500-125,000[1]	SCCWRP (1975)
Dover sole, muscle	Orange County	74-75	3,500-130,000[1]	SCCWRP (1975)
Dover sole, muscle	Palos Verdes	1975	4,000-50,000[1]	SCCWRP (1975)
Dover sole, muscle	Orange County	1975	1,000-2,000[1]	SCCWRP (1975)
Dover sole, liver	Palos Verdes	1975	600,000-950,000[1]	SCCWRP (1975)
Dover sole, liver	Orange County	1975	8,000-12,000[1]	SCCWRP (1975)
Dover sole, gonads	Palos Verdes	1975	28,000-150,000[1]	SCCWRP (1975)
Dover sole, gonads	Orange County	1975	1,000-4,000[1]	SCCWRP (1975)
Dover sole, muscle	L.A. Co. Sanitation Dists. outfall	1975	125,000	Young, et al. (1977)
Dover sole, muscle	L.A. Co. Sanitation Dists. outfall	1976	55,500	Young + Heesen (1978)
Dover sole, liver	L.A. Co. Sanitation Dists. outfall	1976	1,225,000	Young + Heesen (1978)
Dover sole, muscle	Santa Monica Bay	?	3,000-8,500[1]	Mearns + Sherwood (1977)
Dover sole, muscle	Palos Verdes	?	48,500-125,000[1]	Mearns + Sherwood (1977)
Dover sole, muscle	S. San Pedro Bay	?	3,500-130,000[1]	Mearns + Sherwood (1977)
Dover sole, liver	Dana Point (Control Site)	1976	300	Young + Gossett (1980)
Dover sole,	L.A. Co. Sanitation Dists. outfall	1977	1,650,000	Young + Gossett (1980)

liver				
Scorpionfish, muscle	Dana Point (Control Site)	75-77	485	Young + Gossett (1980)
Scorpionfish, muscle	L.A. Co. Sanitation Dists. outfall	75-77	18,000	Young + Gossett (1980)
Pacific sanddab, liver	Hyperion	1978	130,000	Young + Gossett (1980)
White croaker, muscle	Palos Verdes	81-82	38,150	Bascom (1983)
White croaker, muscle	Whites Point	81-82	38,000	Bascom (1983)
White croaker, muscle	Santa Monica Bay	81-82	2,850	Bascom (1983)
White croaker, muscle	Redondo Area Piers	81-82	1,100	Bascom (1983)
White croaker, muscle	Malibu/Santa Monica Pier	81-82	265	Bascom (1983)
White croaker, muscle	Marina del Rey + Venice Pier	81-82	3,650	Bascom (1983)
White croaker, muscle	Los Angeles + Long Beach	81-82	5,400	Bascom (1983)
Scorpionfish, muscle	Palos Verdes	81-82	1,340	Bascom (1983)
Spiny dogfish, muscle	Palos Verdes	81-82	327,500	Bascom (1983)
Dover sole, muscle	Palos Verdes	81-82	35,850	Bascom (1983)
Scorpionfish, muscle	Whites Point	81-82	3,800	Bascom (1983)
Scorpionfish, muscle	Santa Monica Bay	81-82	2,700	Bascom (1983)
Spiny dogfish, muscle	Palos Verdes	?	220,000	Bascom (1982)
Scorpionfish, muscle	Palos Verdes	?	1,100	Bascom (1982)
Dover sole, muscle	Palos Verdes	?	29,950	Bascom (1982)
White croaker, muscle	Palos Verdes	?	37,600	Bascom (1982)
Yellowchin sculpin, liver	Santa Monica	83-84	65,000	Bascom (1985)
Yellowchin sculpin, liver	Palos Verdes	83-84	370,000	Bascom (1985)
Longspine combfish, liver	Santa Monica	83-84	60,000	Bascom (1985)
Longspine combfish, liver	Palos Verdes	83-84	430,000	Bascom (1985)
Pacific sanddab, liver	Santa Monica	83-84	210,000	Bascom (1985)
Pacific sanddab, liver	Palos Verdes	83-84	3,050,000	Bascom (1985)
Yellowchin sculpin, liver	Palos Verdes	83-84	345,000	Bascom (1985)
Scorpionfish, liver	Santa Monica	83-84	70,000	Bascom (1985)

Scorpionfish, liver	Palos Verdes	83-84	140,000	Bascom (1985)
Longspine combfish, liver	Santa Monica	83-84	not detected (DDT metabolites)	Bascom (1985)
Longspine combfish, liver	Palos Verdes	83-84	315,000 (DDT metabolites)	Bascom (1985)
Pacific sanddab, liver	Santa Monica	83-84	85,000 (DDT metabolites)	Bascom (1985)
Pacific sanddab, liver	Palos Verdes	83-84	1,315,000 (DDT metabolites)	Bascom (1985)

* Fish species: Dover sole (Microstomus pacificus)
Scorpionfish (Scorpaena guttata)
Pacific sanddab (Citharichthys sordidus)
White croaker (Genyonemus lineatus)
Spiny dogfish (Squalus acanthias)
Yellowchin sculpin (Icelinus quadriseriatus)
Longspine combfish (Zaniolepis latipinnis)

1. Range given.

** Wet wt. converted to dry wt. via Dry wt. = .20 (Wet wt.); therefore, ppb, Dry wt. = 5(ppb, Wet wt.)

Table 2b. ΣPCB concentrations in southern California fish in ppb (dry wt.).

Fish*	Location	Sampling Year	Mean Concentration**	Reference
Dover sole, muscle	Palos Verdes	71-72	9,800	SCCWRP (1973)
Dover sole, muscle	Santa Monica	71-72	2,000-14,000[1]	SCCWRP (1975)
Dover sole, muscle	Palos Verdes	71-72	3,500-33,000[1]	SCCWRP (1975)
Dover sole, muscle	Orange County	71-72	500-6,000[1]	SCCWRP (1975)
Dover sole, muscle	L.A. Co. Sanitation Dists. outfall	1972	11,500	Young, et al. (1977)
Dover sole, muscle	L.A. Co. Sanitation Dists. outfall	1973	10,000	Young, et al. (1977)
Dover sole, muscle	L.A. Co. Sanitation Dists. outfall	1974	7,500	Young, et al. (1977)
Dover sole, muscle	Santa Monica	74-75	5,000-11,500[1]	SCCWRP (1975)
Dover sole, muscle	Palos Verdes	74-75	300-11,000[1]	SCCWRP (1975)
Dover sole, muscle	Orange County	74-75	1,500-13,000[1]	SCCWRP (1975)
Dover sole, muscle	Palos Verdes	1975	500-5,500[1]	SCCWRP (1975)
Dover sole, muscle	Orange County	1975	3,000-7,000[1]	SCCWRP (1975)
Dover sole, liver	Palos Verdes	1975	55,000-90,000[1]	SCCWRP (1975)
Dover sole, liver	Orange County	1975	24,000-65,000[1]	SCCWRP (1975)
Dover sole, gonads	Palos Verdes	1975	4,000-26,000[1]	SCCWRP (1975)
Dover sole, gonads	Orange County	1975	3,500-23,000[1]	SCCWRP (1975)
Dover sole, muscle	L.A. Co. Sanitation Dists. outfall	1975	9,500	Young, et al. (1977)
Dover sole, with fin erosion, muscle	Palos Verdes	75-76	9,100	SCCWRP (1976)
Dover sole, with fin erosion, liver	Palos Verdes	75-76	93,500	SCCWRP (1976)
Dover sole, no fin erosion, muscle	Palos Verdes	75-76	795	SCCWRP (1976)
Dover sole, no fin erosion, liver	Palos Verdes	75-76	18,450	SCCWRP (1976)
Dover sole, muscle	L.A. Co. Sanitation Dists. outfall	1976	3,700	Young + Heesen (1978)
Dover sole, liver	L.A. Co. Sanitation Dists. outfall	1976	88,000	Young + Heesen (1978)
Dover sole, muscle	Santa Monica Bay	?	5,000-11,500[1]	Mearns + Sherwood (1977)
Dover sole, muscle	Palos Verdes	?	300-12,500[1]	Mearns + Sherwood (1977)

Dover sole, muscle	S. San Pedro Bay	?	1,500-14,000[1]	Mearns + Sherwood (1977)
Scorpionfish, muscle	Dana Point (Control Site)	75-77	215	Young + Gossett (1980)
Scorpionfish, muscle	L.A. Co. Sanitation Dists. outfall	75-77	2,850	Young + Gossett (1980)
White croaker, muscle	Palos Verdes	81-82	1,915	Bascom (1983)
White croaker, muscle	Whites Point	81-82	1,900	Bascom (1983)
White croaker, muscle	Santa Monica Bay	81-82	1,000	Bascom (1983)
White croaker, muscle	Redondo Area Piers	81-82	275	Bascom (1983)
White croaker, muscle	Malibu/Santa Monica Pier	81-82	75	Bascom (1983)
White croaker, muscle	Marina del Rey + Venice Pier	81-82	548	Bascom (!983)
White croaker, muscle	Los Angeles + Long Beach	81-82	885	Bascom (1983)
Scorpionfish, muscle	Palos Verdes	81-82	220	Bascom (1983)
Spiny dogfish, muscle	Palos Verdes	81-82	25,600	Bascom (1983)
Dover sole, muscle	Palos Verdes	81-82	1,395	Bascom (1983)
Scorpionfish, muscle	Whites Point	81-82	330	Bascom (1983)
Scorpionfish, muscle	Santa Monica Bay	81-82	650	Bascom (1983)
Spiny dogfish, muscle	Palos Verdes	?	15,500	Bascom (1982)
Scorpionfish, muscle	Palos Verdes	?	240	Bascom (1982)
Dover sole, muscle	Palos Verdes	?	1,275	Bascom (1982)
White croaker, muscle	Palos Verdes	?	1,770	Bascom (1982)
Yellowchin sculpin, liver	Santa Monica	83-84	6,000	Bascom (1985)
Yellowchin sculpin, liver	Palos Verdes	83-84	11,500	Bascom (1985)
Longspine combfish, liver	Santa Monica	83-84	6,500	Bascom (1985)
Longspine combfish, liver	Palos Verdes	83-84	29,500	Bascom (1985)
Pacific sanddab, liver	Santa Monica	83-84	24,000	Bascom (1985)
Pacific sanddab, liver	Palos Verdes	83-84	70,000	Bascom (1985)

* Fish species: Dover sole (*Microstomus pacificus*)
Scorpionfish (*Scorpaena guttata*)
Pacific sanddab (*Citharichthys sordidus*)

White croaker (Genyonemus lineatus)
Spiny dogfish (Squalus acanthias)
Yellowchin sculpin (Icelinus quadriseriatus)
Longspine combfish (Zaniolepis latipinnis)

1. Ranges given.

** Wet wt. converted to dry wt. via Dry wt. = .20 (Wet wt.); therefore, ppb, Dry wt. = 5(ppb, Wet wt.).

Table 3a. DDT and other pesticide concentrations in whole soft tissue (unless otherwise indicated) of southern California invertebrates in ppb (dry wt.).

Organism*	Location	Sampling Year	Mean Conc.**	Reference
M. californianus (mussel)	Santa Monica	1971	1,900[1]	SCCWRP (1973)
M. californianus (mussel)	Palos Verdes	1971	20,500[1]	SCCWRP (1973)
Cancer anthonyi (crab)	Palos Verdes	71-72	3,750[1]	SCCWRP (1974)
Cancer anthonyi (crab muscle)	Palos Verdes	71-72	5,100[1]	Young, et al. (1976)
M. edulis (mussel)	L.A. + Long Beach Harbors	1974	2,095[1]	Young + Heesen (1974)
M. californianus (mussel)	Palos Verdes Peninsula	1974	4,875[1]	SCCWRP (1975)
Cancer anthonyi (crab muscle)	L.A. Co. Sanit. Dist. Outfalls	75-77	7,500[1]	Young + Gossett (1980)
Cancer anthonyi (crab muscle)	Dana Point (Control Site)	75-77	20[1]	Young + Gossett (1980)
M. californianus (mussel)	L.A. Co. Sanit. Dist. Outfalls	75-77	7,000[1]	Young + Gossett (1980)
intertidal mussels	Royal Palms	1977	2,000[1]	Young (1978)
Mytilus sp. (mussels)	L.A. Harbor	1978	230[2]	Risebrough, et al. (1980)
M. californianus (mussel)	Royal Palms	1978	250[2]	Risebrough, et al. (1980)
Mysids and decapod shrimp	Palos Verdes	?	1,950[1]	Bascom (1982)
Sicyonia (prawn muscle)	Palos Verdes	?	1,415[1]	Bascom (1982)
M. californianus (mussel)	Royal Palms	1980	4[3]	Martin, et al. (1982)
M. edulis (mussel)	L.A. + Long Beach Harbors	1980	13[3]	Martin, et al. (1982)
M. californianus (mussel)	L.A. Harbor transplant station	1980	35[3]	Martin, et al. (1982)
M. californianus (mussel)	Royal Palms	1980	46[4]	Martin, et al. (1982)
M. edulis (mussel)	L.A. + Long Beach Harbors	1980	67[4]	Martin, et al. (1982)
M. californianus (mussel)	Marina del Rey transplant station	1980	780[4]	Martin, et al. (1982)
M. californianus (mussel)	L.A. + Long Beach Harbors	1980	111[4]	Martin, et al. (1982)
M. californianus (mussel)	Palos Verdes Peninsula	1980	1,305[1]	Martin, et al. (1982)
M. californianus (mussel)	L.A. + Long Beach	1980	1,465[1]	Martin, et al. (1982)
M. edulis (mussel)	L.A. + Long Beach	1980	2,282[1]	Martin, et al. (1982)

Mysids + decapods	Palos Verdes	81-82	1,615[1] Bascom (1983)
Ridgeback prawn	Palos Verdes	81-82	1,635[1] Bascom (1983)
M. californianus transplanted mussels	L.A. + Long Beach Harbors	81-82	1,341[1] Ladd, et al. (1984)
M. edulis (mussel)	L.A. + Long Beach Harbors	82-83	1,296[1] Ladd, et al. (1984)
M. californianus (mussel)	Royal Palms	82-83	11[3] Ladd, et al. (1984)
Red pointer crab hepatopancreas	Santa Monica	83-84	25,000[1] Bascom (1985)
Red pointer crab hepatopancreas	Palos Verdes	83-84	220,000[1] Bascom (1985)
Ridgeback prawn hepatpancreas	Santa Monica	83-84	60,000[1] Bascom (1985)
Ridgeback prawn hepatopancreas	Palos Verdes	83-84	245,000[1] Bascom (1985)
Red pointer crab hepatopancreas	Santa Monica	83-84	245,000[5] Bascom (1985)
Red pointer crab hepatopancreas	Palos Verdes	83-84	700,000[5] Bascom (1985)
Ridgeback prawn muscle	Santa Monica	83-84	155[1] Bascom (1985)
Ridgeback prawn muscle	Palos Verdes	83-84	4,250[1] Bascom (1985)

1. ΣDDT
2. DDE
3. Dieldrin
4. Chlordane
5. DDT metabolites

* Ridgeback prawn (*Sicyonia ingentis*)
Red pointer crab (*Mursia gaudichaudii*)

** Wet wt. converted to dry wt. via Dry wt.= .20(Wet wt.); therefore, Dry wt. = 5(ppb, wet wt.)

Table 3b. PCB concentrations in whole soft tissue (unless otherwise indicated) of southern California invertebrates in ppb (dry wt.).

Organism*	Location	Sampling Year	Mean Conc.**	Reference
Cancer anthonyi (crab)	Palos Verdes	71-72	3,375[2]	SCCWRP (1974)
M. californianus (mussel)	Santa Monica	1971	750[2]	Young + Heesen (1978)
M. californianus (mussel)	Palos Verdes	1971	2,250[2]	Young + Heesen (1978)
M. californianus (mussel)	Palos Verdes	1974	600[2]	Young + Heesen (1978)
M. edulis (mussel)	L.A. + Long Beach Harbors	1974	1,135[2]	Young + Heesen (1974)
M. californianus (mussel)	Palos Verdes	1974	610[2]	SCCWRP (1975)
Cancer anthonyi (crab muscle)	L.A. Co. Sanit. Dist. Outfalls	75-77	1,850[1]	Young + Gossett (1980)
Cancer anthonyi (crab muscle)	Dana Point (Control Site)	75-77	160[1]	Young + Gossett (1980)
M. californianus (mussel)	L.A. Co. Sanit. Dist. Outfalls	75-77	850[1]	Young + Gossett (1980)
intertidal mussels	Royal Palms	1977	300[1]	Young (1978)
Mytilus sp. (mussels)	L.A. Harbor	1978	270[1]	Risebrough, et al. (1980)
M. californianus (mussel)	Royal Palms	1978	250[1]	Risebrough, et al. (1980)
Mysids and decapod shrimp	Palos Verdes	?	155[1]	Bascom (1982)
Sicyonia (prawn muscle)	Palos Verdes	?	255[1]	Bascom (1982)
M. californianus (mussel)	Royal Palms	1980	390[2]	Martin, et al. (1982)
M. edulis (mussel)	L.A. + Long Beach	1980	490[2]	Martin, et al. (1982)
M. californianus (mussel)	Marina del Rey	1980	1,800[2]	Martin, et al. (1982)
Mysids + decapods	Palos Verdes	81-82	160[1]	Bascom (1983)
Ridgeback prawn	Palos Verdes	81-82	305[1]	Bascom (1983)
Mussels	L.A. + Long Beach Harbors	1982	829[2]	Ladd, et al. (1984)
Mussels	Whites Point	1982	110[2]	Ladd, et al. (1984)
Mussels	Royal Palms	1982	150[2]	Ladd, et al. (1984)
Mussels	Marina del Rey, North Docks	1982	1,000[2]	Ladd, et al. (1984)
M. edulis (mussel)	L.A. + Long Beach Harbors	82-83	869[2]	Ladd, et al. (1984)
M. californianus	Royal Palms	82-83	290[2]	Ladd, et al. (1984)

(mussel)				
Red pointer crab hepatopancreas	Santa Monica	83-84	2,500[1]	Bascom (1985)
Red pointer crab hepatopancreas	Palos Verdes	83-84	23,500[1]	Bascom (1985)
Ridgeback prawn hepatopancreas	Santa Monica	83-84	15,000[1]	Bascom (1985)
Ridgeback prawn hepatopancreas	Palos Verdes	83-84	9,000[1]	Bascom (1985)

1. ΣPCB
2. PCB 1254

* Ridgeback prawn (<u>Sicyonia ingentis</u>)
Red pointer crab (<u>Mursia gaudichaudii</u>)

** Wet wt. converted to dry wt. via Dry wt. = .20(Wet wt.); therefore, ppb Dry Wt. = 5(ppb, Wet wt.)

Effects: Populations

Reish (1980) investigated the effect of the discharge of domestic wastes on benthic communities in southern California and made predictions of these effects based on the quantity of primary effluent discharged per day. He noted biological enhancement, i.e. an increase in biomass, number of species and specimens, and diversity, where less than 18 million liters of primary effluent per day were discharged. The biomass, number of specimens, and area affected increased, but the number of species, diversity, and richness decreased where the amount of discharge exceeded 40-180 million liters per day. The discharge he reported from all the major sewage outfalls in southern California in 1977 exceeded the amount likely to degrade marine benthic communities:

Source of Effluent	Discharge Location	Primary Effluent Discharged
San Diego	Point Loma	440 million liters per day
Los Angeles County	Whites Point	1270 million liters per day
Orange County	N. of Newport Bay	606 million liters per day
Los Angeles City	Santa Monica Canyon	830 million liters per day

Reish (1971) studied changes in communities of fauna inhabiting sites in Los Angeles Harbor following the termination of oil refinery discharge in September, 1970 in compliance with an order by the California Regional Water Quality Control Board. The first change noted was improved water quality and appearance of dissolved oxygen. Reish (1971) reported "Successful settlement and growth of a greater diversity of organisms occurred more rapidly on the boat floats than the benthos because of the lack of accumulative wastes characteristic of the bottom." The water quality improvement appeared sufficient for Reish to predict that *Mytilus edulis* would become the dominant organism on the boat docks in 1971.

Bascom (1982) evaluated the benthic infaunal situation in areas surrounding sewage outfalls in southern California and at 28 control stations where there were no indications of human disturbance. He used three parameters, number of species, number of individuals per square meter, and biomass per square meter, to determine Infaunal Index and to establish three descriptions for areas: "normal," "changed," and "degraded." The size of the changed area around outfalls varied. The changed area for Los Angeles City was $48 km^2$; for Los Angeles County it was $85 km^2$. Around the San Diego outfall $4 km^2$ were changed; around Orange County outfalls, $10 km^2$. Conditions in the Ventura area fell within the normal range. The only two degraded areas in southern California according to Bascom were $3 km^2$ at Los Angeles City and $9 km^2$ at Los Angeles County.

Bascom's summarizing comments were hopeful. "In areas where serious damage to the environment has been observed in the past, there has been notable recovery of the marine life" (Bascom, 1982). He cited reappearance of pelicans and species of algae including the giant kelp. He certainly recognized room for improvement in environmental conditions, but felt "there are no very serious problems either."

Concurring with Bascom (1982) was Mearns (1981) who commented, "Conditions attributable to sewage discharges were considerably better in 1980 than in 1970." Mearns also cited return of kelp beds, dramatic increases in abundance and diversity of seaweeds and invertebrats during the previous four years at Palos Verdes, apparently in response to decreased emission from the discharge there. However, Mearns (1981) noted that "repeated and standardized benthic surveys" indicate changed communities at all outfall sites in southern California except at Oxnard (between Santa Monica Bay and Santa Barbara). He called the changes most pronounced at Palos Verdes and at the Hyperion outfall in Santa Monica Bay where communities are dominated by deposit-feeding polychaetes and mollusks.

Mearns and Young (1983) summarized knowledge gained from several studies of impacts of municipal wastewater on the southern California coastal environment. They reported the main benthic response to wastewater discharges at all major sites is "a shift from communities dominated by suspension feeding infauna to those dominated by an abnormally high biomass of deposit-feeding invertebrates" They noted most obvious effects of municipal effluent on the communities along the Palos Verdes shelf where there was a major decrease in benthic species diversity, dominance by deposit feeding organisms, and fish catches that were significantly lower than control sites in abundance, number of species, and biomass.

Bascom, et al. (1979) reported results of sampling benthic invertebrates by trawling at 60m in control areas and areas impacted by point-source waste discharges in southern California. He referred to sections of SCCWRP (1977) by Word, Mearns and Allen and by Word and Mearns to describe a depression in echinoderm abundance in the Palos Verdes area, approximately two orders of magnitude lower than expected, based on abundances at other 60m stations. The area of severely depressed echinoderm abundance extended down to ≥610 meters and horizontally out about 5km from the outfalls.

Bascom, et al. (1979) also reported the absence of infaunal microcrustacea in a large area around a discharge site on the Palos Verdes shelf. They hypothesized that DDT levels in sediments there might be the cause of the disappearance. Bascom, et al. (1979) also presented data from the Southern California Coastal Water Research Report (1978) which showed patterns of changes in abundance of benthic invertebrates and fishes around the outfall system on the Palos Verdes shelf. Many types of invertebrates were depressed there and a few were enhanced. Also listed were five depressed fish species and three enhanced fish species.

Soule and Oguri (1980) described changes in benthic communities of San Pedro Bay during the past 150 years. Harbor improvements made drastic irreversible changes there. Then pollution from sewage and industry advanced degradation of the harbor. In the late 1960s, however, federal and state environmental laws strengthened cleanup efforts and macrofauna began to live in areas previously devoid of all but bacterial life.

Soule and Oguri reported results of benthic sampling in 1973-1974 and in 1978. These results indicated downward trends in mean numbers of species throughout the harbor, except at stations at the river mouth. At outer Los Angeles Harbor stations mean numbers of species and individuals first rose 1971-1972, following initiation of water quality standards enforcement in 1970. Then the "population means dropped steeply in 1975 and continued to deline through 1978" (Soule and Oguri, 1980). At outer Long Beach Harbor and main channel stations the mean numbers of species peaked in 1973 and

1975 and remained only slightly above the 1972 level in 1978. At inner Los Angeles Harbor and main channel stations the mean species numbers leveled off at considerably higher numbers in 1978 than in 1972, reflecting, according to Soule and Oguri (1980), "more accurately the effects of increasing environmental quality." At stations in the Los Angeles River mouth the mean number of species was high in 1973, then dropped to less than half in 1975 and returned to the 1973 level in 1978. The following table from Soule and Oguri (1980) shows that the mean number of benthic species remained almost the same while the mean number of individuals dropped about threefold for all stations from the 1973-74 period to 1978:

# Species			# Individuals	
Period	Range	Mean	Range	Mean
73-74	1-60	28	32-80,000±	22,264
1978	4-68	28.7	240-38,000±	8,102

In 1978 the same three benthic species, (1) Cossura candida (a polychaete species), and (2) Mediomastus californiensis and (3) Tharyx sp. (annelid species), that were dominant in numbers were also most widely distributed at almost all the 41 harbor stations sampled. In the 1973-74 period these three were also the dominant species found, but their dominance rank differed. After considering transitory impacts of rainy seasons, and other natural variations, "Harbors Environmental Projects Associates" concluded that decreases in the outer harbor were largely due to changes in waste treatment (Soule and Oguri, 1980).

Davis and Spies (1980) took a census of infaunal benthic invertebrates at a natural petroleum seep near Santa Barbara, California and at a control area nearby without fresh petroleum in sediments. They found higher densities of individuals in ~60% of the populations at the petroleum seep, but no dramatic difference in diversity. They also noted "The most abundant populations at the seep site were dominated by deposit feeders . . . ., especially oligochaetes which are extremely rare at the comparison site."

Spies (1984) sought and found some evidence that alterations in the distribution of communities of bottom-feeding fishes occurred as a result of changes in the distriution and abundance of their benthic prey around major sewage outfalls in the Southern California Bight. Spies called this the "trophic coupling" hypothesis. He discussed factors which make measuring of trends in finfish very difficult. These included the high level of mobility of finfish and natural year-to-year variability in fish abundance. In summary Spies declard that the "definitive study on fish distribution around sewage outfalls has yet to be made." He thought the available data suggested lower fish diversity and higher biomass near outfalls, but that trend was not consistantly apparent.

Smith (1973) analyzed benthic population data and abiotic data from the vicinity of waste discharges in outer Los Angeles Harbor. He found the "distributions of species groups, biomass, and species diversity all showed a relationship to the distance from shore and the [cannery] effluent sources." Basically he reported lowest species diversity and biomass values at nearshore stations. There was a similar geographical trend in the level of dissolved oxygen. DO levels decreased toward the sources of discharge. Smith reported the polychaete Capitella capitata was dominant in the species group which occurred mostly at stations closer to the effluent sources.

This polychaete is characteristically found in bottom areas where the sediment is considered "polluted."

Southern California Coastal Water Research Project (SCCWRP) reports (1973, 1974, 1975, 1976, 1977) contained data about benthic community characteristics and the relation of proximity of the communities to sewage treatment outfalls. SCCWRP (1973) compared conditions around the "old" Orange County outfall before and after termination of discharge in April, 1971. After termination of discharge the total benthic invertebrate density decreased (from 16,000 to 10,000 organisms per square meter), the total number of species per sample increased (from 65 to ~84), and species diversity, as measured by the Shannon-Weaver function, increased. Also there were major changes in species composition. The polychaete, Capitella capitata, was one of the three most abundant species before termination of the discharge, but was not among the sixteen most abundant species afterwards. SCCWRP (1973) also compard characteristics of benthic macrofaunal communities around the "new" Orange County outfall before and after initiation of discharge there in April, 1971. After initiation of the discharge at the new outfall the total population density more than doubled and the number of species per sample increased from 90 to 100. Total community diversity decreased and the number of polychaete species per sample remained nearly constant. Also there were changes in relative abundance for most species. For example, before the discharge Capitita ambisata accounted for only 4.3% of the total individuals, but it was the most abundant species in the postdischarge period, accounting for >25% of the total number of individuals.

SCCWRP (1974) tried to evaluate effects of physical and chemical factors on commuities from data collected in a summer, 1973 survey. Characteristics of "outfall" and "nonoutfall" areas did differ. The characteristic abiotic factors they found in outfall areas were higher nitrogen, more particulate matter, finer sediments, and higher H_2S, DDT, and Hg than in nonoutfall areas. They concluded that "changes in depth and various outfall-related factors are responsible for the similarities and dissimilarities in species compositions at various sites and are causing changes in species abundance." DDT and organic nitrogen were the only organic contaminants they measured. ". . .some of the physical and chemical factors that affect a species distribution may not have been measured and included in this analysis. . ."(SCCWRP, 1974).

SCCWRP (1975) analyzed data collected during annual monitoring in the area of major municipal outfalls on the Palos Verdes shelf and data collected during the Los Angeles County Sanitation Districts' benthic survey there in summer, 1973. Using different data analysis techniques from those reported in SCCWRP (1974), they reported similar results. The presence of two very active outfalls was reflected in the distribution of total species. The number of species per site was greatly reduced at stations near the outfalls. Other data such as distribution of numbers of individuals was complex and seemed impossible to evaluate objectively.

SCCWRP (1975) also contained qualitative results from a preliminary survey of benthic fauna surrounding the 1- and 5-mile Orange County outfalls conducted in February and March, 1975. ". . . benthic fauna may be characterized as normal, grading near the 5-mile outfall into patterns reminiscent of somewhat enriched regions around other outfall sites in southern California" (SCCWRP, 1975).

SCCWRP (1976) presented results from comparing statistics about benthos at five wastewater outfall areas along the southern California coast with control areas. The five outfall areas were Oxnard, Santa Monica Bay, Palos Verdes coastal shelf, Santa Ana River area of south San Pedro Bay, and Point Loma at San Diego. The trend was of "increasing biomass (but not necessarily abundance) and decreasing number of species, diversity, and richness associated with increasing amounts of discharge" (SCCWRP, 1976). They found that effects at discharge sites could be related quantitativly to the amount of wastewater discharged. They also noted a geographical trend of general increase in biomass and decrease in diversity as you approach the Los Angeles coastal region from north to south.

Abnormalities

In the section of Bascom (1985) called "Tumors in Fish Collected on the Palos Verdes Shelf" Jeffrey Cross discussed results of collection and examination of fish on the Palos Verdes shelf from 1971 through 1983. Over 90 percent of the tumorous fish were young Dover sole, Microstomus pacificus, less than two years old. Trend analysis of data revealed no significant changes in percent of Dover sole with epidermal tumors during the time period, 1971-1983; however, data analysis did indicate percent of tumorous fish increased with proximity to outfalls. Cross found evidence suggesting that these tumors are caused by parasitic amoebae whose incidence is apparently enhanced by Palos Verdes shelf outfalls.

Marjorie Sherwood and Alan Mearns also discussed skin and mouth tumors of Dover sole in their chapter, "Tumors in Southern California Demersal Fishes" in SCCWRP (1974). They found no obvious geographical trends in frequency of tumorous Dover sole trawl-caught in southern California coastal waters, nor did the prevalence of tumor disease seem to them to be related to proximity to any particular wastewater discharge. Several factors suggested to them that skin tumors in Dover sole were genetic or developmental diseases.

Bascom (1982) discussed two types of fish diseases prevalent in southern California coastal waters - skin tumors and fin erosion. Since the occurence of skin tumors, which are commonly found in young Dover sole, is widely distributed over time and geographical space, Bascom suggested the tumors are not related to municipal wastewater discharge. Occurrence of fin erosion suggested the opposite. Thirty nine percent of Dover sole and twenty eight percent of calico rockfish samples taken with trawls on the Palos Verdes shelf in 1977 had fin erosion. A short distance north or south of Palos Verdes the frequency of fin erosion dropped to ten percent. Less than one percent of Dover sole from control stations had the disease. Bascom hypothesized that outfalls may attract the Dover sole and induce fin erosion in them.

Bascom (1979) had reported that most species of fish with fin erosion collected by otter trawl from Santa Monica Bay to Dana Point, 1972-1976, "had highest frequencies of the disease in the vicinity of the wastewater discharge on the Palos Verdes shelf."

Mearns and Young (1983) declared fin erosion the only disease conclusively linked to wastewater discharges. Although the exact cause of the disease is still unknown, the fish apparently must have contact with bottom sediments containing high concentrations of PCBs such as those at Palos Verdes discharge areas, and must accumulate the PCBs in their tissue for the disease to develop. They concluded that prevalence of fin erosion in Palos Verdes Dover sole would decline but would still affect about nine percent of these fish in 1988. They were cognizant of the uncertainties about causes of fin rot. Parhaps total PCB is not the only cause. Mearns (1981) said that fin erosion "has been induced in the laboratory by exposing healthy fish to sediments from Palos Verdes".

Mearns and Sherwood (1977) reported results of a study of disease distribution patterns in coastal waters off southern California. About five percent of the 290,000 fishes collected and examined in their 1969-76 study were affected with such external disease symptoms as fin and tail erosion, tumors, abnormal coloration, and attached macroparasites. Fin erosion in Dover sole was the only one of these diseases that appeared "to be directly associated with discharge of municipal waste waters in southern California." The disease appeared to result from exposure to contaminated sediments on the Palos Verdes shelf. Mearns and Sherwood (1977) concurred with Bascom (1982), SCCWRP (1974), and others in noting that the tumor diseases did not appear related to municipal waste water discharge sites. They further did not consider tumor diseases related to "other discrete sources of pollutants." They found prevalence of lip papillomas in white croaker decreased from prevalence in earlier years. Lip papillomas were primarily in fish from off the coasts of Orange and Los Angeles Counties.

Sherwood and Mearns (1977) collected demersal fishes from southern California coastal waters from 1972 to 1976 and found the Dover sole the species most often affected by fin erosion and found the Palos Verdes shelf the site of the largest percentage of fish with fin erosion. Their Table 2 contained the following data concerning percentages of total trawl caught fish with fin erosion for each of five most frequently affected species:

	Palos Verdes Shelf	Santa Monica Bay	S. San Pedro Bay	Dana Point
Dover sole	39%	3.5%	2.0%	0.67%
Rex sole	21%	2.1%	0.0%	0.0%
Slender sole	21%	5.5%	.06%	0.0%
Green-striped rockfish	18%	0.0%	0.0%	0.0%
Vermilion rockfish	11%	0.0%	0.0%	0.0%

SCCWRP (1974) reported results of 1972-73 trawl surveys in which prevalence of fin erosion in Dover sole from the Palos Verdes shelf was 42.6% and from Santa Monica only 2.5%.

In a chapter of SCCWRP (1977) entitled "Histology of Liver Tissue from Dover Sole" Kenneth Pierce, Bruce McCain and Marjorie Sherwood reported on a preliminary study of differences between histological characteristics of livers of Dover sole collected from the Palos Verdes shelf and livers of

Dover sole from a control area, Dana Point. Liver structures of the Dana Point Dover sole were normal. Palos Verdes Dover sole without fin erosion had abnormal liver structure: larger and more irregular fatty vacuoles in cells, disarray in tissue structures, larger and more numerous melanin-macrophage centers, and more interstitial fibrotic tissue. In livers of two of the three Palos Verdes specimens with severe fin erosion, the changes were more extreme.

SCCWRP (1974) contained information from histopathological examination of fish. They reported that fin lesions in Dover sole, rex sole, and slender sole were non-infectious and non-inflammatory. Since the epidermis over damaged areas in most cases appeared intact, the lesions possibly developed from within the subcutaneous tissue of the fin tip rather than from trauma through the outer skin.

Deirdre McDermott and Marjorie Sherwood in the chapter entitled "DDT and PCB in Diseased Dover Sole" in SCCWRP (1975) considered possible relationships between those contaminants and fin erosion in Dover sole. They found a strong, though not quite statistically significant relationship between high levels of total PCB and fin erosion in fish off Palos Verdes and Orange County. They also found a strong tendency for diseased fish from off Palos Verdes to have higher levels of DDT than unaffected specimens. Two of their ideas were interesting speculations. DDT and PCB in combination with other constituents associated with the Palos Verdes region could be involved in the development of fin erosion; or possibly diseased fish just have enhanced uptake of PCB and/or DDT, so the higher tissue levels of these contaminants may be a result, rather than a cause of the disease.

SCCWRP (1973) reported frequency of tail erosion and lip papillomas in white croaker from four trawls in 1970 at the old Orange County outfall (discharge terminated in April, 1971):

tail erosion	lip papillomas
Feb......26%	Feb......4.4%
May......11%	May......0.6%
Aug......14%	Aug......0.9%
Nov......74%	Nov......0.0%

Sediment Toxicity

Emerson (1974) used two benthic polychaete species, Ophryotrocha nr. labronica and Capitella capitata, as bioassay organisms to determine the toxicity of resuspended sediment from Los Angeles Harbor. In all short term (96 hour) exposures Capitella capitata trochophore and metatroch mortality was less than 50%. In short term tests with Ophryotrocha nr. labronica no mortality occurred; however, sublethal effects did occur at the higher concentrations of elutriate. Long term (28 day) exposure to increasing concentrations of sediment correspondingly decreased reproductivity in Ophryotrocha. On the other hand, low resuspended sediment concentration (100:1) produced a stimulatory effect on reproduction. As a consequencse of these results Emerson presented the idea that harbor dredge spoils might have value as a sort of fertilizer for improved productivity in specified areas of the marine environment.

SOUTHERN CALIFORNIA

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The coastal bays of Louisiana have been the site of intensive oil drilling and production for decades. And most studies related to toxic organic compounds conducted in these bays or offshore areas have been related to either oil spills or chronic oil discharges. However, the major environmental problems along the coast are attributed to physical alterations of the coast line caused by erosion (Boesch, 1982). Man's alterations particularly the channelization of the Mississippi have resulted in shifting i.e. with time, sediment supplies, altering natural depositional patterns. Recent rises in sea level and subsidence have also contributed to the problem (Morgan, 1974). Flood control measures have diverted normal freshwater in flows to many bays altering salinity regimes and biological populations.

Figure 1, shows the chronology of the naturally shifting deltas of the Mississippi.

Wells and Kemp (1982) reviewed the recent literature and stated, "The works of Morgan and Larimore (1957), Gagliano and van Beek (1970), and Adams et al. (1978) as well as others, establish that the shoreline of Louisiana, taken as a whole, is currently retreating. These authors also point out, however, that this retreat shows a high degree of spatial variability. For example, in the case of the modern Mississippi delta front, retreat is virtually nonexistent and in the case of the Atchafalaya Delta complex, it is significantly reversed. We can then draw a picture of a modern shoreline that is undergoing erosion and transgression, but that is dynamically stable at the Mississippi River delta front and is locally progradational near the Atchafalaya River mouth."

"The 200-km section of shoreline extending west from Marsh Island to the Texas border is distinct in plan view from the rest of the Louisiana coast (Figure 2). The complex indentations and barrier/lagoon systems that characterize the shorelines flanking the modern Mississippi River course are not found west of Vermilion Bay. The smooth and relatively straight form of the western half of the coast reflects a depositional history different from that of the rest of Louisiana's coastal plain. Early workers hypothesized that this section evolved during the Holocene as a marginal deltaic sequence of prograding mudflats that were intermittently partially reworked into sand/shell ridges called "cheniers" (Russell and Howe 1935; Price 1955). More recently, Gould and McFarlan (1959) reconstructed the development of the "chenier plain" and adjacent shelf from cores using radiocarbon dating techniques. Their interpretation indicates that, as sea level rose from -5 m to its present level, a transgressive sequence of marine sediments was deposited over the dissected Pleistocene Prairie Formation, first filling estuaries, then later spreading across shallow bay and marsh environments."

Organic Contaminants in Water and Sediments

Bender, et al. (1979) reported on attempts to measure discharges from petroleum platforms in Timbalier Bay and offshore sites (Figure 3). Figure 4 shows the levels of organic carbon (mean & ranges) at the offshore platforms and the control site, from June of 1972 through February of 1974, while Figure 5 displays the levels of organic carbon, BOD & COD along a transect from the bay to the offshore platforms. They concluded that:

"Although the concentrations of TOC, BOD, and hydrocarbons do not appear to differ between platform and ambient sites, the ternary diagrams in Figures 6 and 7 suggest that relative amounts of types of hydrocarbons in the air/sea interface and in sediments may differ between platform and ambient sites. The arrows in Figure 6 indicate how to read a ternary diagram. "Platform" represents all data of all samples collected within a range of 0 to 2,500 m from 54A, spanning an azimuth of 360 degrees, which includes platform 66D. "Ambient" represents all data within a range of 2,501 to 12,500 m from 54A, spanning an azimuth of 360 degrees. The platform data tend to cluster and, with two-notable exceptions, have the highest levels of benzene eluates (aromatics). Since aromatics usually disappear most rapidly with weathering of oil, this analysis suggests that the platform may be the source of fresh petroleum."

Figure 7 indicates that aromatic hydrocarbons do not tend to accumulate in sediments offshore, perhaps because they rapidly disappear during weathering of oil. The two samples within 750 m of 54A had the highest aromatic content. There were no samples taken within 750 m of platform 66D. All platform samples were taken within 2,500 m of 54A and ambient samples within the range of 2,501 to 12,500 m.

Data in Figure 7 indicate the petroleum hydrocarbons present in sediments are more weathered (have lower aromatic content) than those present in air/sea interface and beach samples. This diagram presents all the hydrocarbon data for all of the cruises.

"In summary, we do not feel that the OEI data quantitatively demonstrate that platforms are significant sources of TOC, BOD, or hydrocarbons; on the other hand, display of the data in ternary diagrams suggests that platforms are sources of unweathered oil. The OEI data are inadequately replicated to support rigorous statistical treatment of platform versus ambient comparison even with the redefinition of control sites" (Bender et al., 1979).

Data on hydrocarbons in bottom sediments from the same area but covering a larger experimental area the "Central Gulf Platform Study" are shown in Figure 8. Sharp & Appan (1982) summarized the data at the four primary platforms:

"In the C.G.S., four 'primary' platforms (of 20 platforms and four control sites investigated) were sampled at 16 stations (100, 500, 1000 and 2000 m along north, east, south and west transects) in one season, and at eight of these same stations (500 and 2000 m along these azimuths) in two successive seasons. For these 200 samples, the maximum and mean values were 371 and 37 $\mu g\ g^{-1}$ respectively, and 13 samples exceeded 100 $\mu g\ g^{-1}$. Figure 8 shows the three-season average of concentrations of heavy molecular mass hydrocarbons (heavy hydrocarbons) at the four primary platforms. Except for primary platform no. 1, neither these averages nor single-season values show consistently high values nor radial gradients."

Effects of Organic Contamination on Benthos

Sharp and Appan (1982) examined the Central Gulf Study with respect to the responses of benthic animals to hydrocarbon concentrations in sediments. Figure 9 plots diversity of benthos vs distance from platforms: Hydrocarbon data are shown in Figure 8, for these same stations. They concluded that: "Comparison of the H' values with heavy hydrocarbon concentrations show (a) no significant change nor gradient in H' corresponding to heavy hydrocarbon gradients, (b) no significant differences among the four widely separated platforms despite significant differences in heavy hydrocarbon concentrations, and (c) values of H' in the range 1.7-3.3 with a mean value of 2.60 for 140 samples taken over three seasons. Additional measures of potential effects were investigated, eg. community structure and species dominance. They summarized their investigations as follows:

"These and numerous other investigations of C.G.S. data fail to reveal significant correlations of concentration of petroleum hydrocarbons in the sediments with measures of macroinfaunal structure, abundance or diversity. Nor do they reveal evidence of general degradation, change, or unnatural distribution in this community that can be attributed to either point source or distributed source contamination from petroleum activities. The effects of the Mississippi River, with its large hydrocarbon load and natural variables such as depth and sediment type, are those variables correlating with macroinfaunal communities with greatest statistical confidence. The most evident correlations are with the faunistic zones that are defined by ecosystem properties and processes."

Comparisons of faunal similarity at study sites in Timbalier Bay are shown in Figures 10 and 11. Unfortunately no hydrocarbon data are available to help clarify the differences between study sites. However, Bender, et al. (1979) concluded:

"The index of similarity does not seem to indicate any marked seasonal differences in the benthic populations. There is a slight indication of greater similarity between stations in summer than in winter. This can be noted particularly in the comparisons of ambient station AB1 with the other stations."

"The benthic biological results in Timbalier Bay were subjected to cluster analysis using all polychaete, crustacean, and mollusc data and using all polychaete data. The combined data included collections made in winter, spring, and summer 1973, and the polychaete data included collections made during the same seasons plus two samplings taken in fall 1973. With the combined data for the three animal groups, three distinct assemblages are noted, which coincided, in part, with the seasons. The four summer stations fell out as a group. The four winter stations were grouped with two spring stations. The two remaining spring stations were together in one group. No benthic biological differences existed between the platform stations and the ambient stations when this method of analysis was used."

"The dendrogram drawn from the polychaete data showed a separation of the stations according to the season. The four winter collections were separated as a group with two subdivisions. The remaining 10 collections were grouped according to season but were added one-by-one in a step-wise cluster."

Catallo (1984) studied a cresote spill site in Bayou Bonfouca a tributary of Lake Pontchartrain. Figure 12 shows his sampling stations and Table 1 the concentrations of PAHs detected in water and sediments.

He examined the effects of the high levels of PAH contamination on the microbial and meiofaunal populations and the related physical-chemical characteristics of the sediments. He concluded: "ATP analyses of bacteria, fungi, protozoans, and direct counts of meiofauna indicated highly significant reductions in biomass of all communities exposed to creosote relative to the control. Microbial plate counts generally supported ATP data and demonstrated altered community structure in polluted areas. The effect of creosote upon benthic microbial assemblages was also manifested in the physical-chemical properties of the sediments: Redox potential (Eh) was progressively less reducing in contaminated areas and terminal accumulations of detritus were observed."

Abnormalities in Fish and Shellfish

Neoplasms of fish and shellfish have only infrequently been reported in aquatic animals along the Gulf of Mexico. Mix (in press) reviewed the information on this topic and reported: "Despite significant efforts to discover such conditions in aquatic species inhabiting what are (or were) considered to be the most polluted areas (e.g. Galveston Bay, Louisiana oil fields, Mobile Bay), only a few neoplasms have been found in such surveys."

"In a little known study of massive proportions, eastern oysters from Louisiana, near oil fields, were the subject of intense investigations from 1947 through 1960. While those studies were not directly related to neoplasms in oysters, and they occurred many years ago, they are considered to be highly relevant to the present review and so will be described in this section."

"The origin of the project (reported fully in Mackin & Hopkins, 1961) can be traced to 1946 and 1947 when leaseholders of private oyster-growing grounds in several Louisiana parishes filed claims, some of which later became lawsuits, asking compensation for claimed financial losses. Those claims were based on abnormally high mortality of oysters on their leases, allegedly caused by operations of certain oil and sulfur companies. Several of the oil companies requested scientists at the Texas A&M Research Foundation to conduct scientific studies to investigate the extent and cause of the oyster mortality in Louisiana and also in Texas. The projects became widely known as Projects 9 and 23. Project 9 began on February 1, 1947, and ended May 31, 1950, at which time Project 23 began; it lasted until 1960."

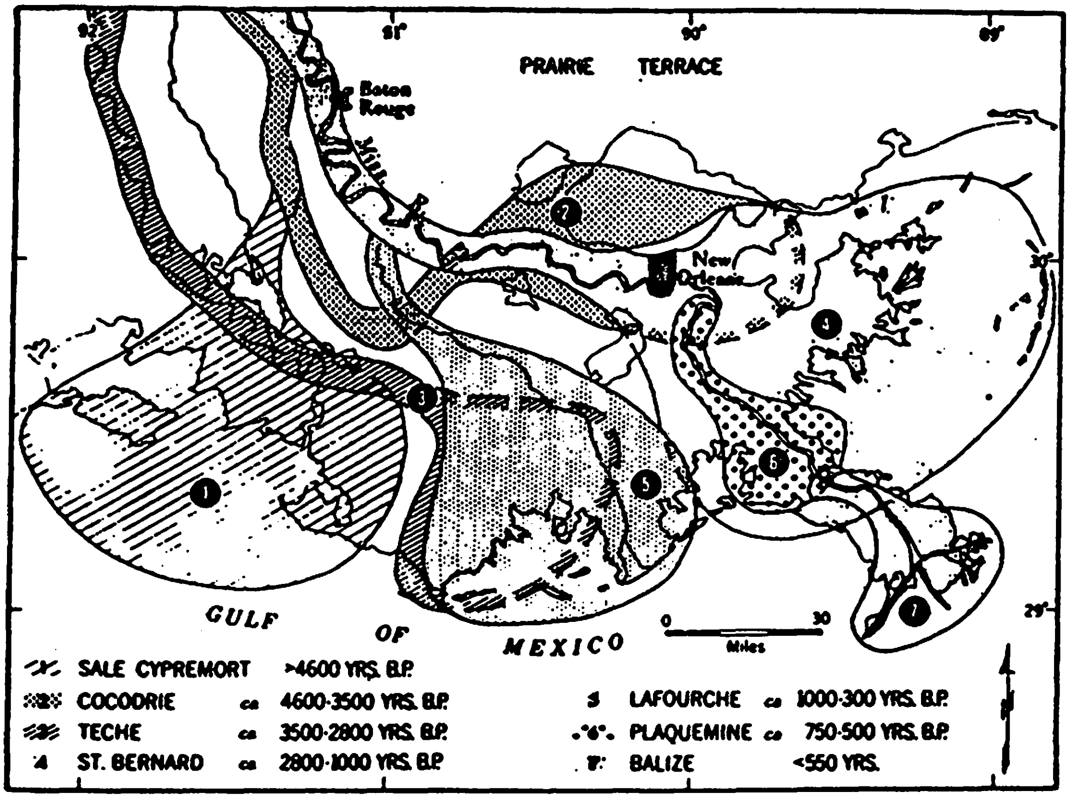
"Eventually, these two projects were to produce more than 200 scientific reports. Thousands of oysters were collected and examined microscopically from many oyster-growing areas. Some of the areas were obviously polluted with oil since the oysters were described as having an oily taste and even the crude analytical methods of the period were capable of detecting hydrocarbons in water samples (Mackin & Hopkins, 1961). No correlation was found between rates of oyster mortality and proximity to oil fields, and no oyster mortalities occurred as a result of oil spills. Ultimately, after the causes of mortality could not be traced to oil, a disease organism was found which killed thousands of oysters every summer when the water temperatures and salinities increased to a critical level (Mackin, 1961)."

"Although thousands of oysters were examined, no neoplasms were reported by the Texas A & M investigators. An important questions about those results is, would those scientists have detected neoplasms if they existed? Reports of neoplasms were not common at that time and no internal neoplasms had been described; thus, there was no information base to utilize in diagnosing such conditions. However, Mackin and his co-workers were the preeminent invertebrate pathologists of that period. Given some of their discoveries of unicellar parasites, it seems unlikely that they would have overlooked something as deviant as neoplasms if they had been present."

"In another research project involving Project 23 personnel, oysters exposed to oil from a wild well were studied for several months after exposure. In 1957, a well went out of control, caught fire, and spilled oil for a period of about two weeks. It was believed that the oil loss was probably the largest ever sustained in the oyster producing area of Louisiana to that time (Mackin & Sparks, 1961). Oysters were exposed to oil for at least two months and oil slicks were evident in the areas studied for at least six months; it can be assumed that extremely heavy oil pollution impacted the area for several months. Oysters were studied for six months after the spill and, again, a large number were examined histologically. Both native oysters and oysters planted in trays at different sample sites were studied and, again, no neoplasms were reported, although a variety of tissue conditions were described. Since one of the authors of those reports (Sparks) was later to co-author one of the first descriptions of a hemic neoplasm in oysters (Farley & Sparks, 1970), it seems unlikely that any neoplasms would have been overlooked."

Mix (in press) concluded his review of Gulf investigations including sites in Louisiana, Texas and Florida with the following: "Thus, while thousands of oysters have been examined from different locations in the Gulf of Mexico, some known to be contaminated with different pollutants, relatively few neoplasms have been identified. There was little evidence from the various studies that supports a pollution-neoplasm relationship."

Figure 1



CHRONOLOGY OF DELTAS THAT COMPOSE THE HOLOCENE MISSISSIPPI RIVER DELTAIC PLAIN. Modified from Kolb and Van Lopik (1958).

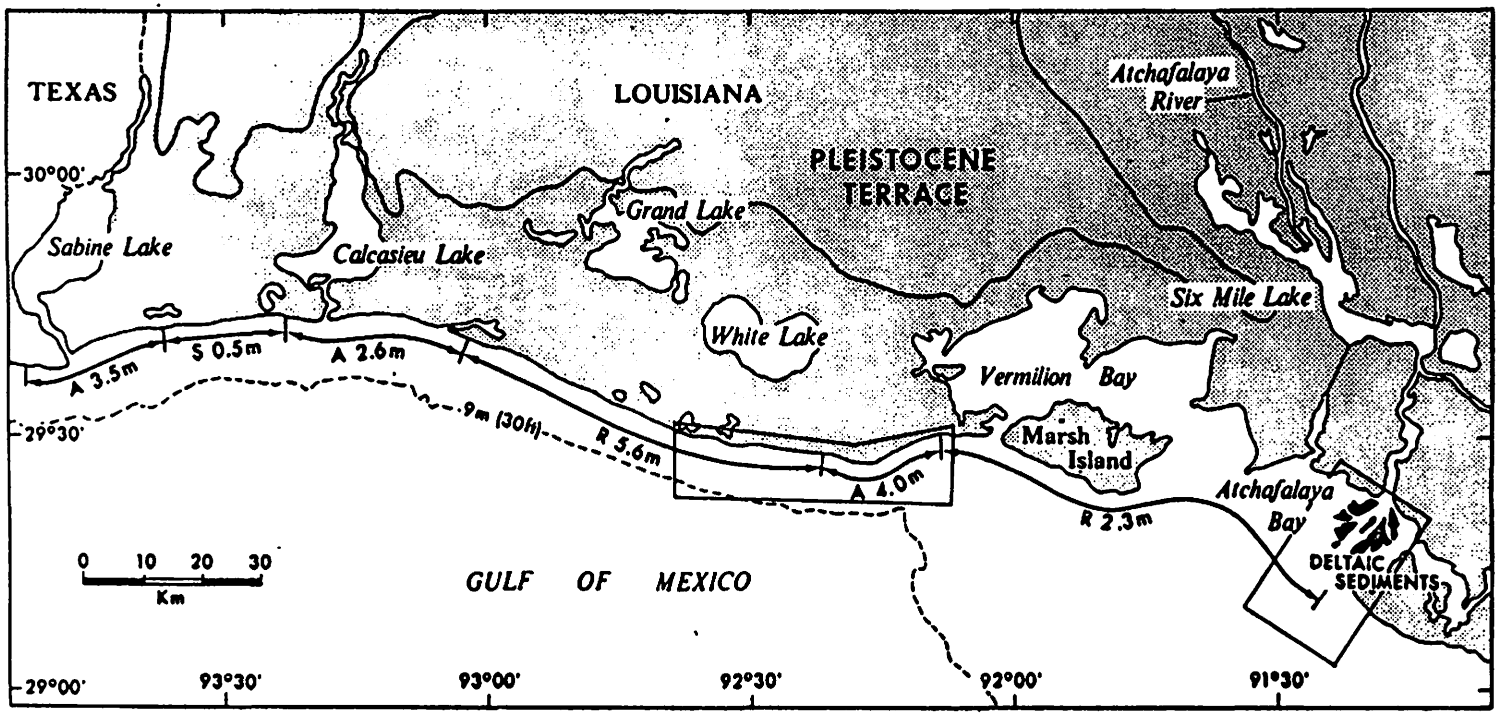
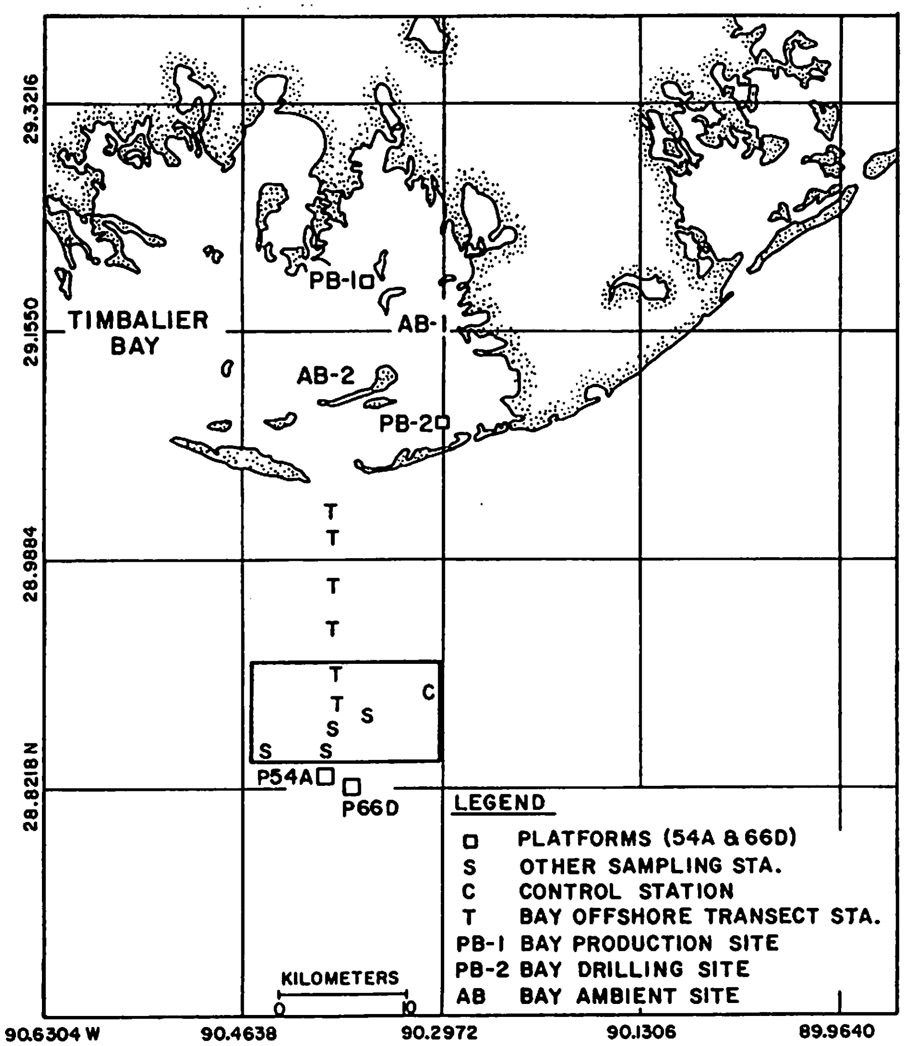


Figure 2 Coast of central and western Louisiana showing average annual rates of shoreline change from 1812 to 1954 (from Morgan and Larimore 1957). A = accretion, S = stable, R = retreat. Enclosed region of Atchafalaya Bay shows source of fine-grained sediments; enclosed segment of chenier plain is present-day downdrift recipient of these sediments.

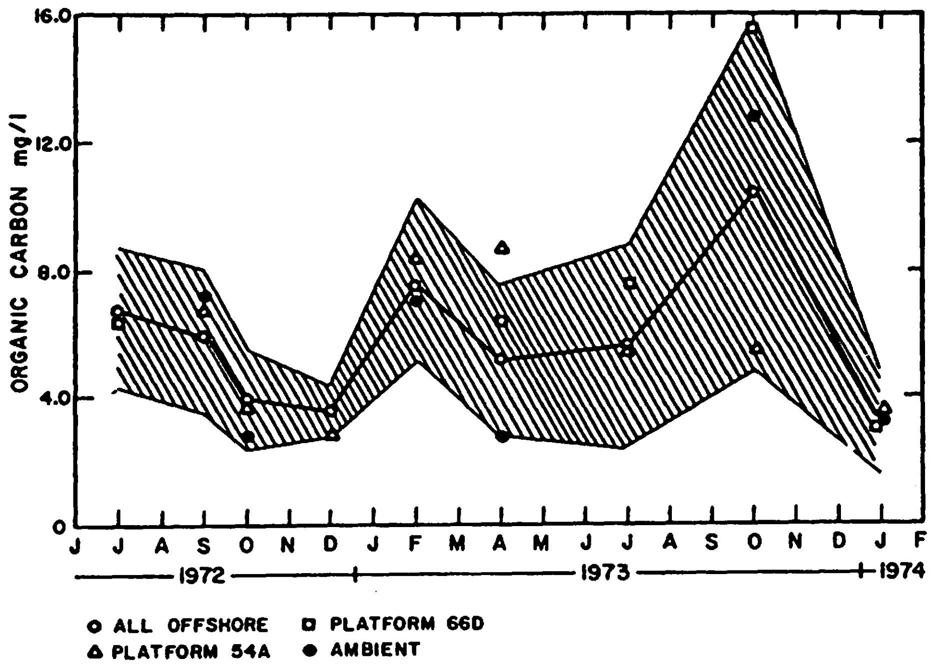
Figure 3



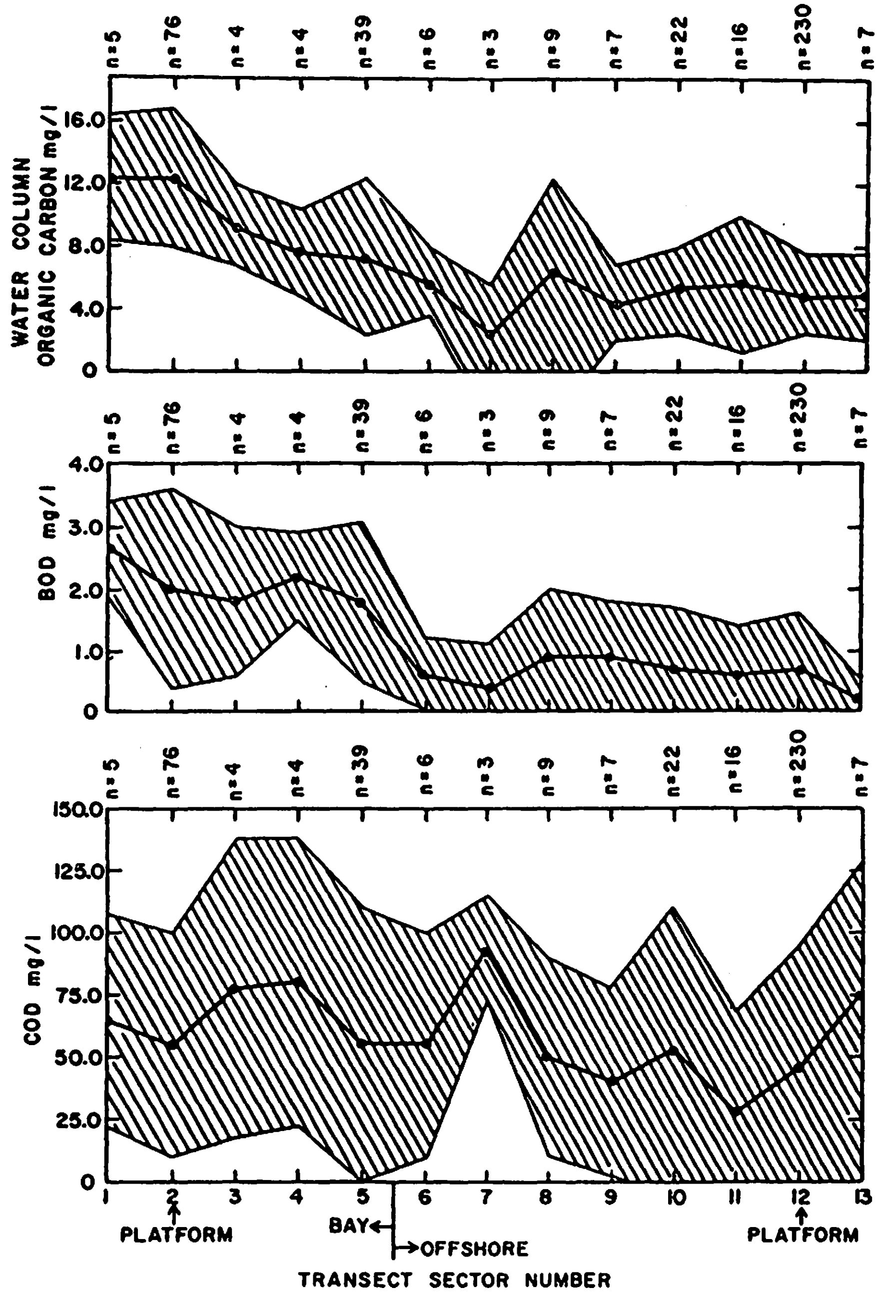
LOCATIONS OF BAY AND OFFSHORE SAMPLING STATIONS used in the re-examination of the Offshore Ecology Investigation. (Bender et al., 1979)

Figure 4

AMB	n=	6	2		1	2		4	9
66D	n= 16					2	15	1	27
54A	n=	6	2	2	6	2	14	2	23
ALL	n=22	42	28	9	16	19	53	12	99



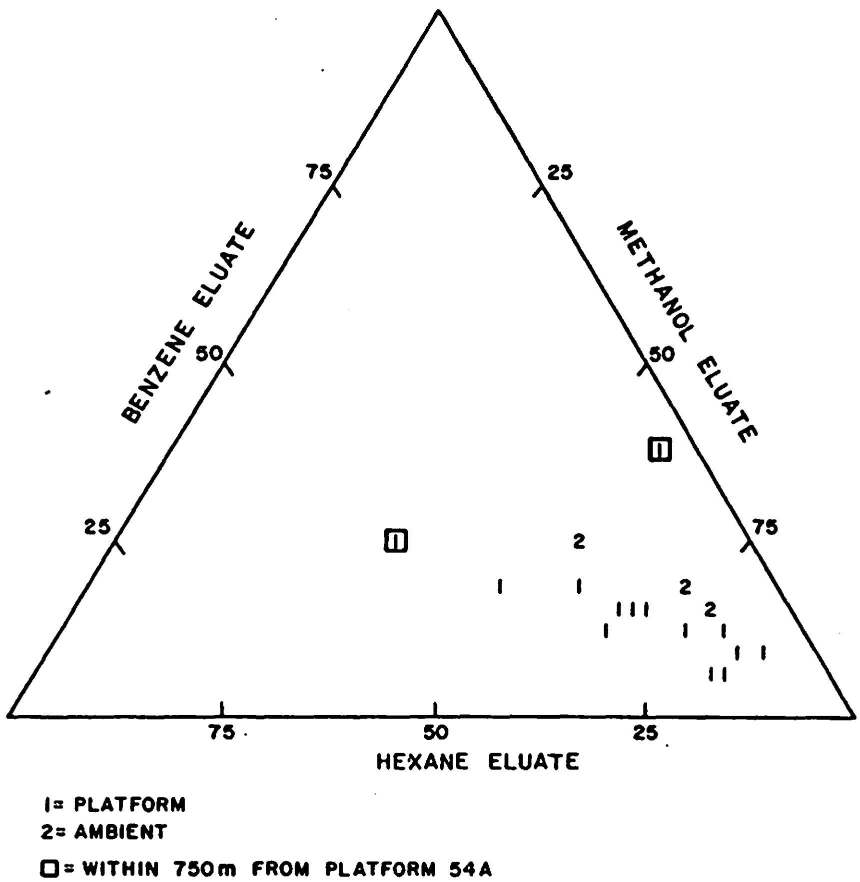
SEASONAL VARIABILITY OF ORGANIC CARBON IN THE WATER COLUMN OFFSHORE. (Bender et al., 1979)



VARIABILITY OF ORGANIC CARBON, BOD, AND COD IN THE WATER COLUMN FROM INNER BAY TO OFFSHORE. (Bender et al., 1979)

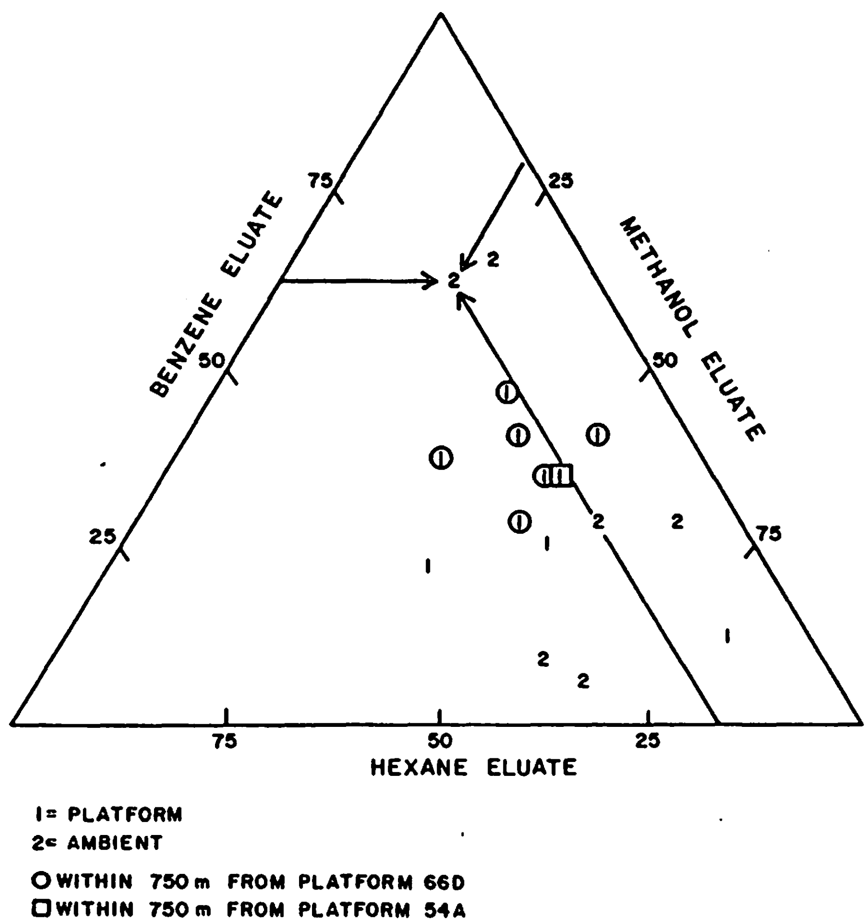
Figure 5

Figure 6

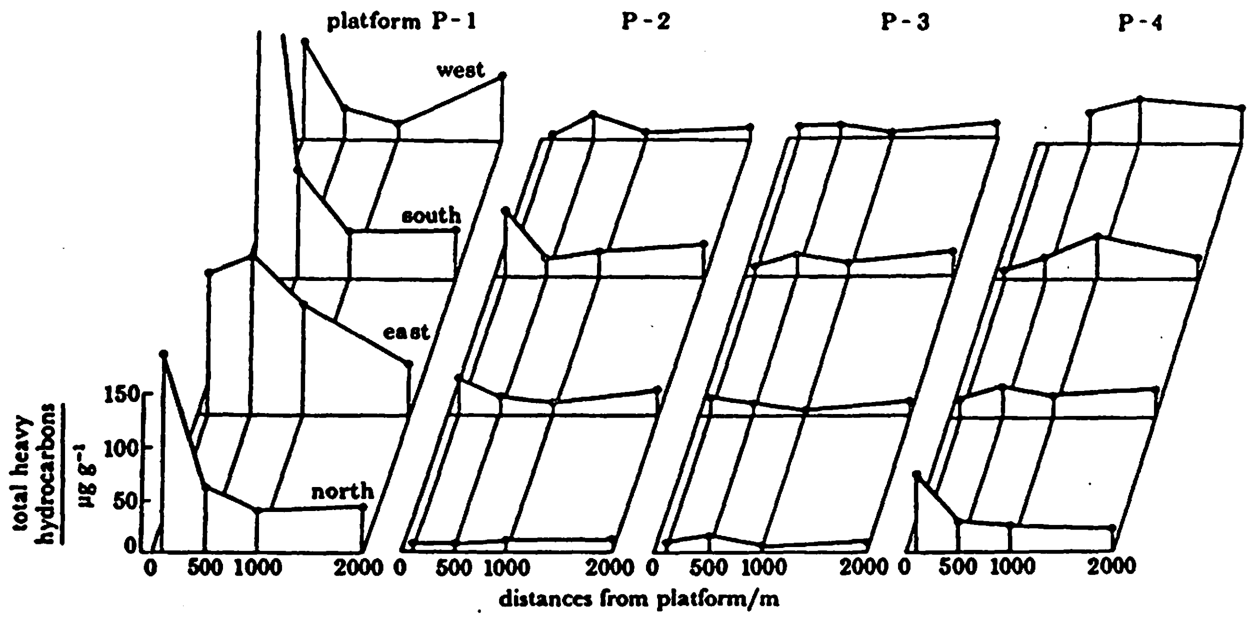


TERNARY DIAGRAM OF HEXANE, BENZENE, AND METHANOL ELUATES OF SEDIMENT SAMPLES AT PLATFORM AND AMBIENT STATIONS OFFSHORE. (Bender et al., 1979)

Figure 7



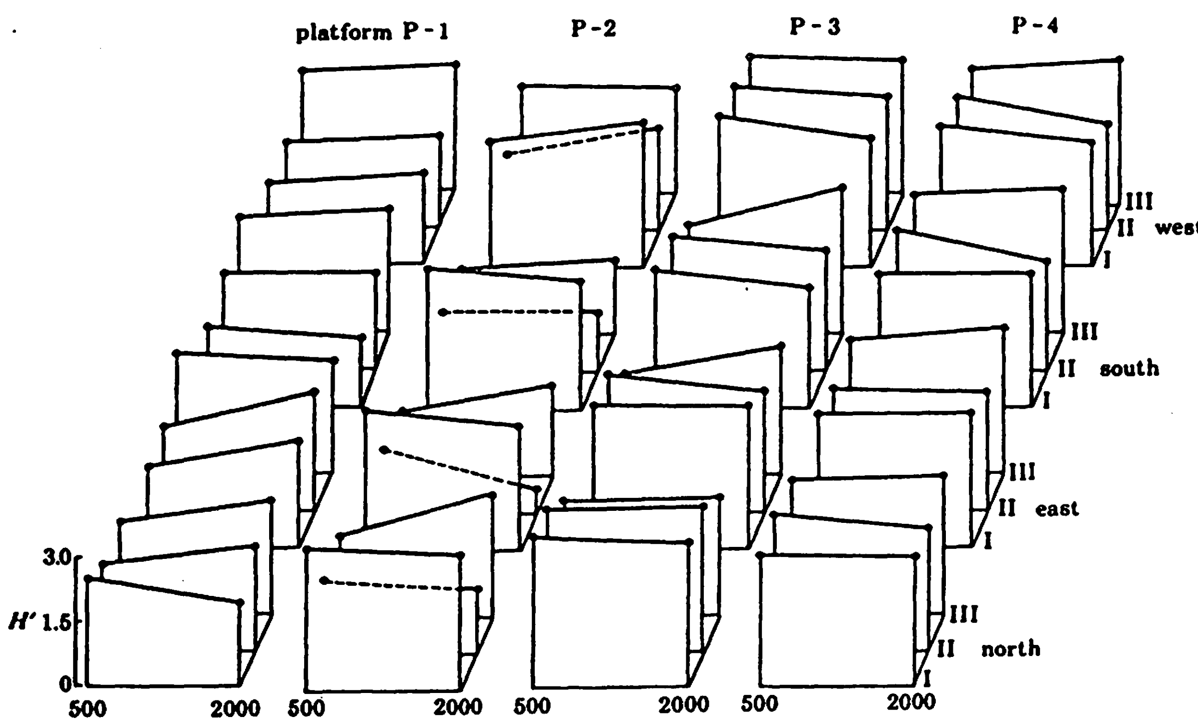
TERNARY DIAGRAM OF HEXANE, BENZENE, AND METHANOL ELUATES OF SURFACE WATER SAMPLES AT PLATFORM AND AMBIENT STATIONS OFFSHORE. (Bender et al., 1979)



Concentrations of heavy hydrocarbons in the bottom sediments at the four primary C.G.S. platforms.

(Sharp and Appan, 1982)

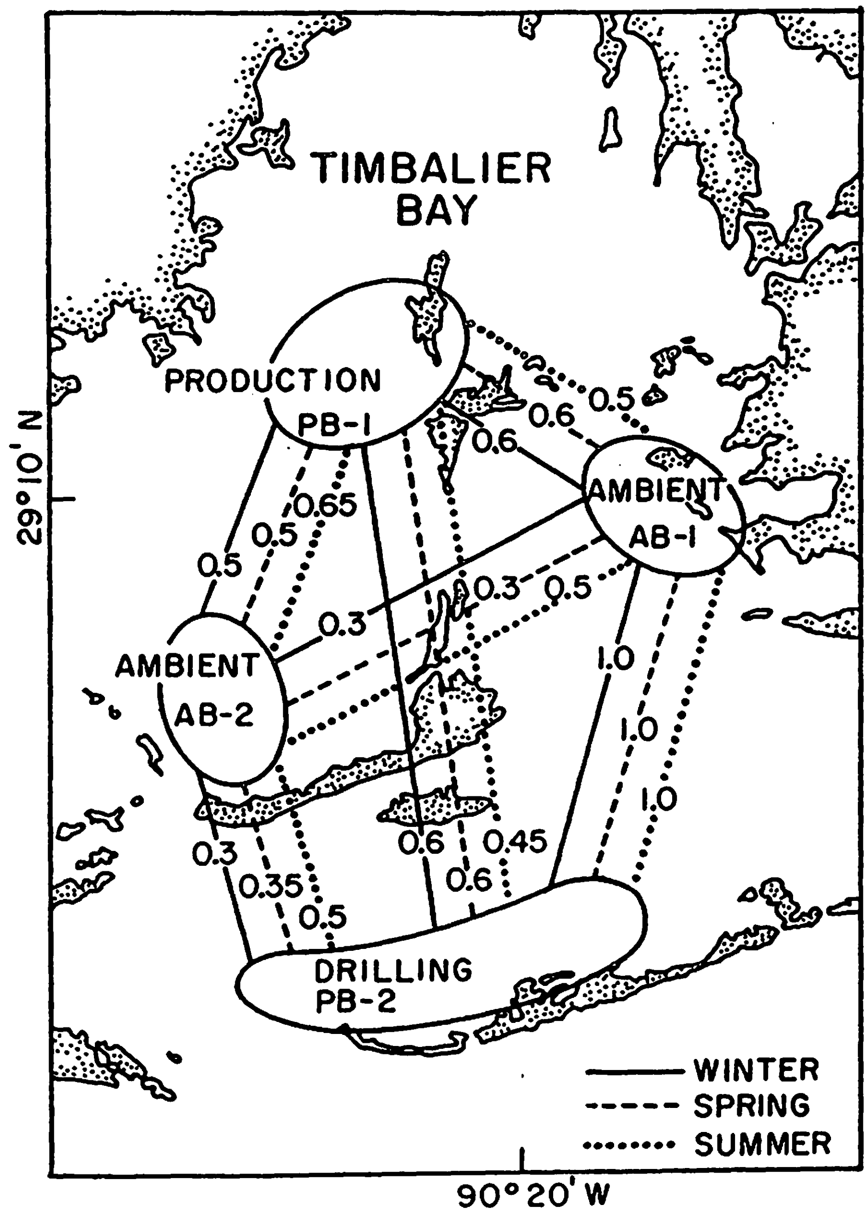
Figure 8



Radial gradients of Shannon-Weaver species diversity index, H', at four primary C.G.S. platforms for three seasons: I, May 1978; II, August 1978; III, January 1979.

(Sharp and Appan, 1982)

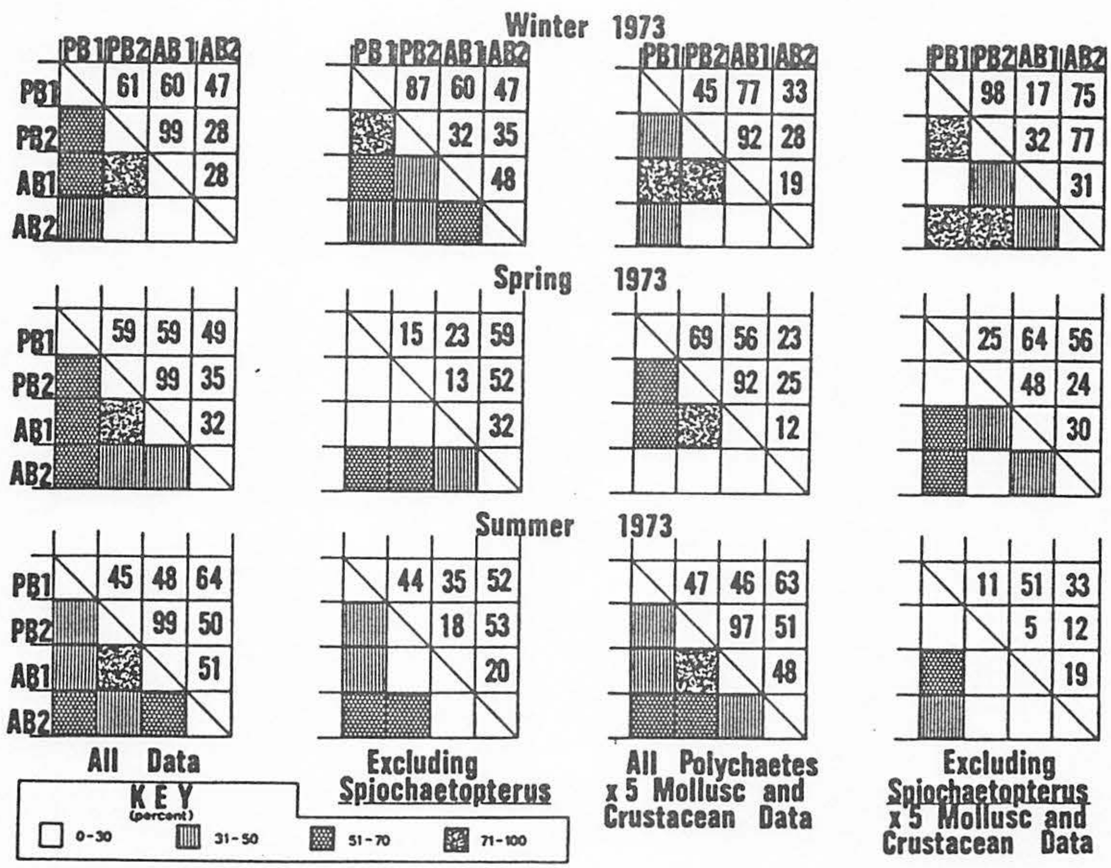
Figure 9



DIAGRAMMATIC REPRESENTATION OF THE MORISITA-ONO INDEX OF FAUNAL SIMILARITY IN TIMBALIER BAY (from Bender et al.. 1979; Copyright, Offshore Technology Conference, 1979; reproduced by permission).

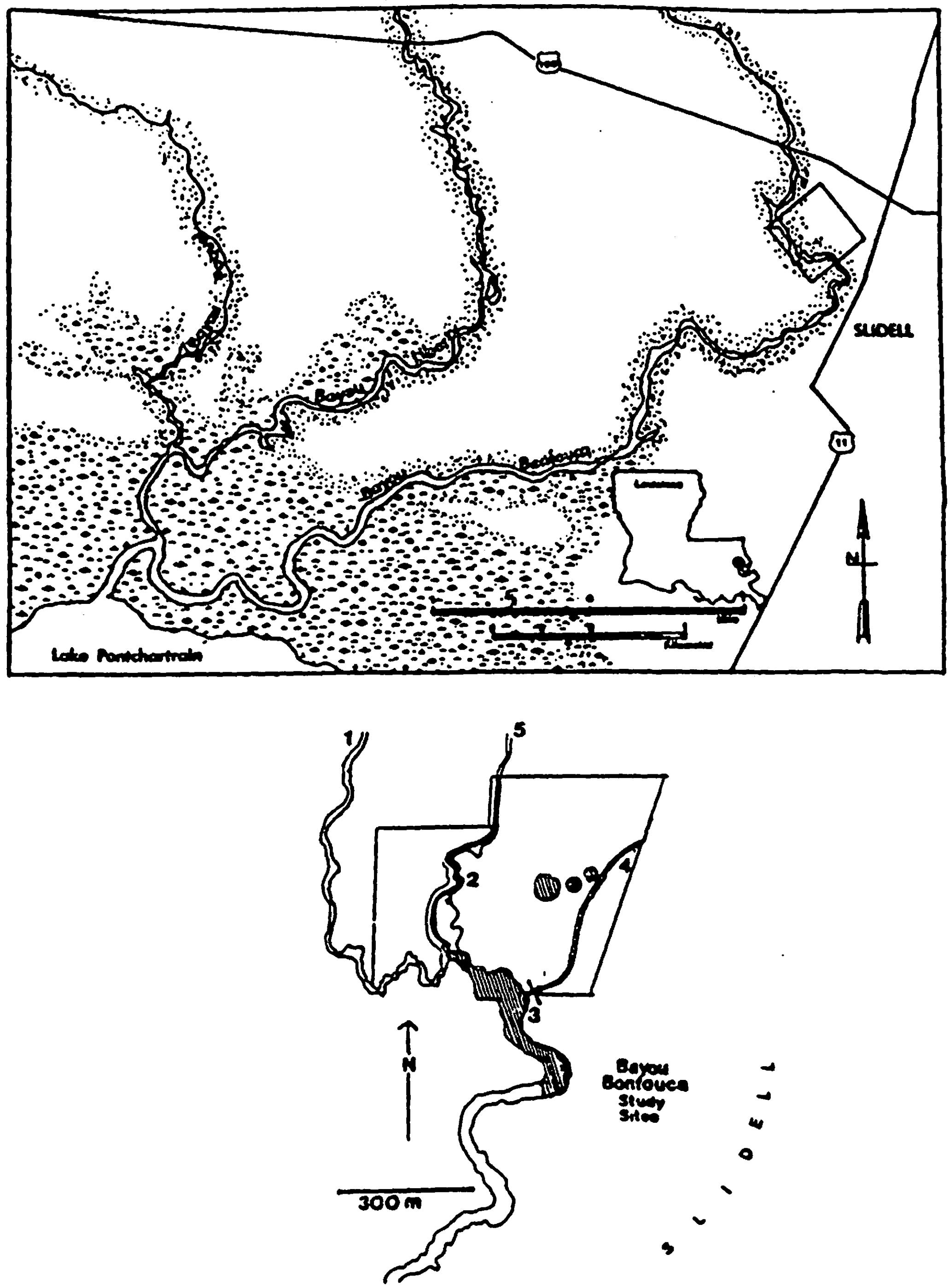
Figure 10

Figure 11



TRELLIS REPRESENTATION OF THE MORISITA-ONO INDEX OF FAUNAL SIMILARITY (× 100) IN TIMBALIER BAY. (Bender et al., 1979)

Fig. 12   Location of Experimental Sites



(Catallo, 1984)

TABLE 1 CONCENTRATIONS OF POLYCYCLIC AROMATIC HYDROCARBONS IN SEDIMENT AND WATER FROM BAYOU BONFOUCA, SLIDELL, LOUISIANA

Site	Compound	Surface Water*	Sediments*
1	Three unknowns eluting in the range of naphthalene and flourene		
2	Naphthalene	8.3	1380
	Flourene	7.6	358
	Phenanthrene	28.7	435
	Anthracene	14.7	107
	Flouranthene	27.1	608
	Pyrene	18.3	178
	Benzo(k)flouranthene/ benzo(j)flouranthene	4.9	75
3	Naphthalene	0.7	7720
	Flourene	0.6	2180
	Phenanthrene	2.3	17670
	Anthracene	0.4	1110
	Flouranthene	1.2	3360
	Pyrene	2.1	800
	Benzo(k)flouranthene/ benzo(j)flouranthene	nd +	940
	Benzo(a)pyrene	0.3	40
4	Naphthalene	14.1	7117
	Flourene	12.3	nr±
	Phenanthrene	155.0	29310
	Anthracene	39.7	1650
	Flouranthene	110.0	6580
	Pyrene	85.0	1660
	Benzo(k)flouranthene/ benzo(j)flouranthene	5.5	2280
	Benzo(a)pyrene	6.6	610

* Parts per million (ppm)
+ "Not detected
± "Not resolved"

(Catallo, 1984)

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MISSISSIPPI SOUND AND MOBILE BAY

Mississippi Sound, shown in Figure 1, is a shallow embayment which forms the southern border of Mississippi. Rivers, small bays, marshes, and bayous are along the coast. A chain of offshore islands form the southern limit of the Sound. Mobile Bay is at the eastern boundary and Lake Borgne at the western boundary of the Sound.

Mississippi Sound "is one of the most valuable fisheries nursery grounds of the world" (Lytle and Lytle, 1985). Sections of its shore presently support a tourist industry. Some local residents are becoming concerned, however, that with growth of industrial interests, other than traditional fishing and tourism, pollution of the Sound may occur. The area seems destined for expansion in industrial, residential, and petroleum resource exploitation.

Mobile Bay, shown in Figure 2, cuts deeply north into the short southern coastal border of Alabama. It supports an important commercial seafood industry (especially oysters). Changes in the Mobile Bay area currently threaten to produce environmental problems. According to Marion and Settine (1983), industrial and technological development is proceeding rapidly and major chemical and petrochemical industries have been built on the edges of Mobile Bay in recent years. They expect barge and tanker traffic will increase and that the population in the area will expand.

Julia and Thomas Lytle of the Gulf Coast Research Lab in Ocean Springs, Mississippi have recently completed a four year study of Mississippi Sound (Lytle and Lytle, 1985). Research reported by the Lytles and their associates at the Gulf Coast Research Lab comprise the major body of organic contaminant information concerning Mississippi Sound. Ken Marion and Robert Settine's (of University of Alabama in Birmingham) studies of organic pollutants in bivalves supplied most of the Mobile Bay information.

PAHs in Sediments

Lytle and Lytle (1985) analyzed surface sediment samples from sites in Mississippi Sound and found an astonishing contrast in levels of total hydrocarbon in bay and river sediments to levels in the Open Sound. They found 100 to 1000 times the Open Sound level of total hydrocarbons in the Pascagoula River System, a region of diverse and concentrated industrialization. They observed large variations in hydrocarbon levels between stations in close proximity in the Pascagoula River and Biloxi Bay. The samples they collected in Bayou Casotte, considered part of the Pascagoula River System, contained the highest total concentration of hydrocarbons, but surprisingly not the highest aromatic hydrocarbon load, even though oil tanker traffic serves a refinery in Bayou Casotte. They described the total Pascagoula River and Biloxi Bay ecological system as "heavily polluted but only in very well-defined localized areas."

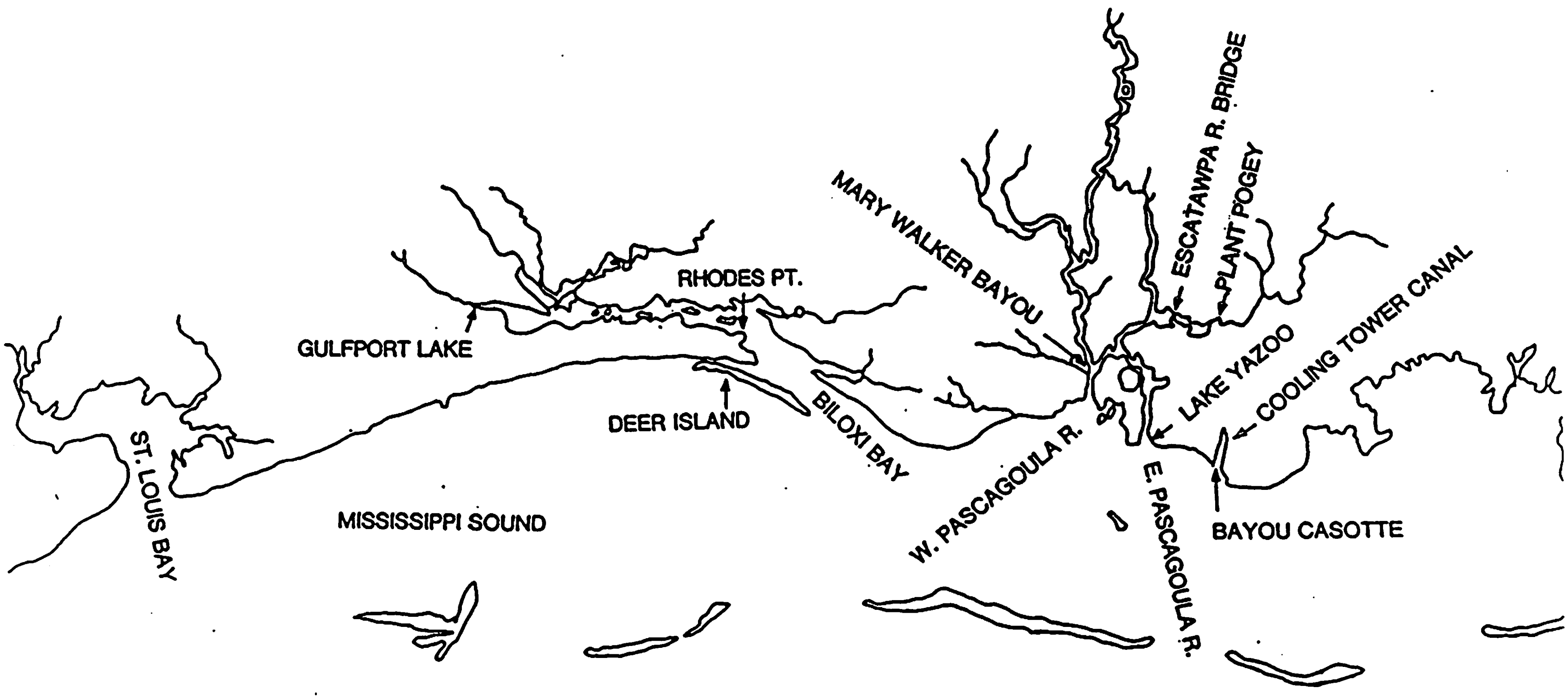

Figure 1. Mississippi Sound (Lytle and Lytle, 1985).

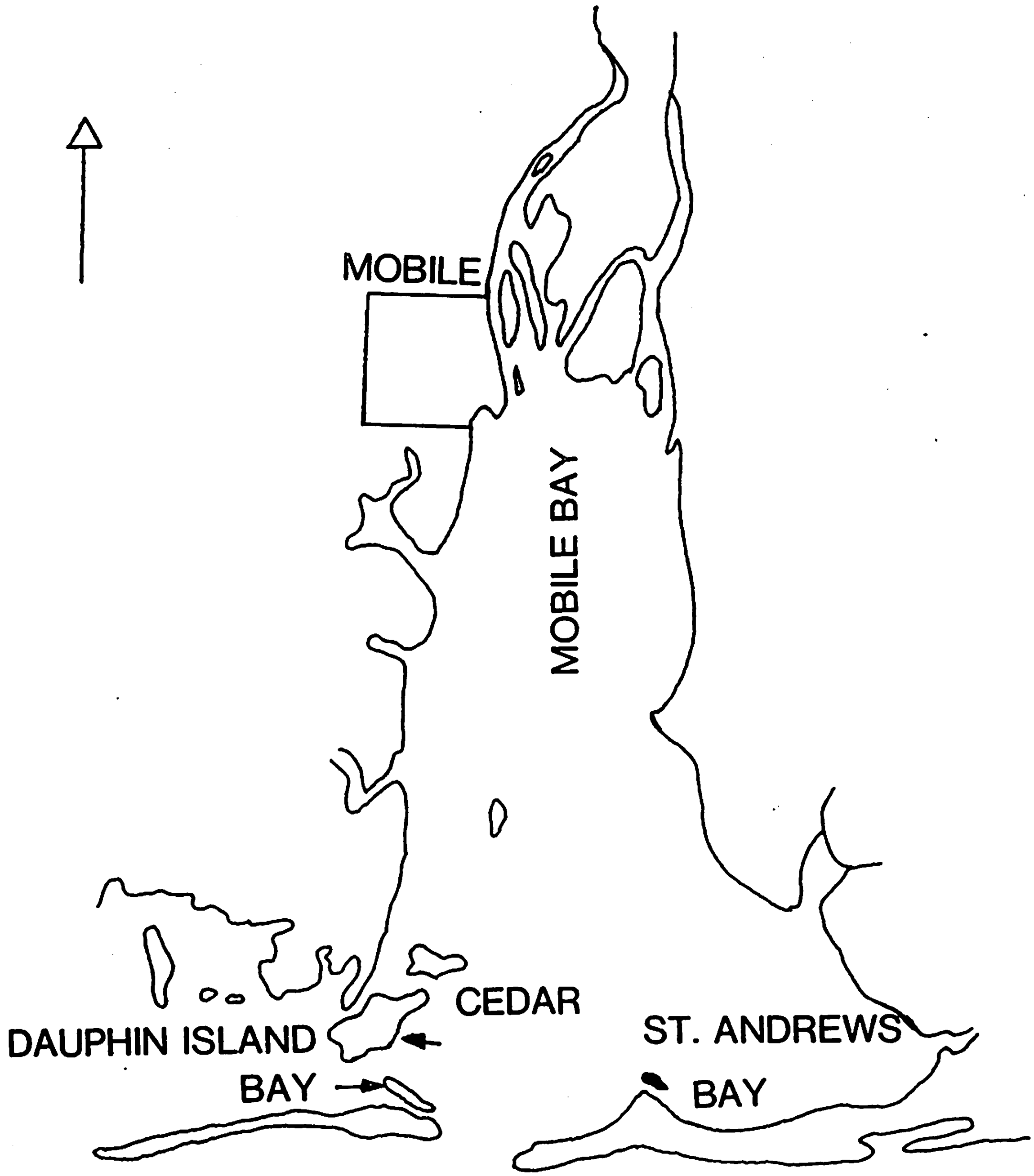

Figure 2. Mobile Bay (Marion and Settine, 1983).

Following are listed general areas the Lytles studied in order from those they considered most seriously polluted down to the least impacted:

1. Pascagoula River System:
   (mean conc. aromatic hydrocarbons in surface sediments = 211,500 ppb, dry wt.; range = 0 - 1,930,000 ppb, dry wt.)
2. Biloxi Bay System:
   (mean conc. aromatic hydrocarbons in surface sediments = 186,000 ppb, dry wt.; range = 410 - 2,610,000 ppb, dry wt.)
3. St. Louis Bay System:
   (mean conc. aromatic hydrocarbons in surface sediments = 7,370 ppb, dry wt.; range = 3,520 - 9,770 ppb, dry wt.)
4. Mississippi Sound System (or Open Sound):
   (mean conc. aromatic hydrocarbons in surface sediments = 3,110 ppb, dry wt.; range = 70 - 11,000 ppb, dry wt.)

Lytle and Lytle (1985) in summarizing facts that emerged from their pollutant distribution data also stated the following:

> "The highest pollution levels were found in only very localized regions, not broad areas."
>
> "The greatest fluctuations in pollutant values and greatest pollution problems were noted in hydrocarbons."
>
> "Fairly uniform distributions of organics occur in the Open Sound System in sharp contrast with the river and bay systems."

PCBs and Pesticides in Sediments

Walker (1976) collected sediment samples from 37 sites in the Biloxi Bay - Mississippi Sound estuary system and analyzed for pesticides using a procedure capable of detecting concentrations as low as 1 ppb. He detected no insecticide residues in any of these sediment samples. He felt that "the waters and sediments of the Mississippi Coastal Zone are free from contamination by these noxious chemicals." He explained that drainage areas to the north of this area are forested and receive only minimal pesticide applications.

Walker (1976) also studied persistence of insecticides in Mississippi Sound water and sediment. He found that microbial degradation was a substantial factor in dissipation of several insecticides from sediments, but of little consequence in water. He believed that if insecticides enter the Mississippi Sound estuary, we can expect them to stay unchanged for a long time, especially if they are associated with the water column.

PAHs, PCBs and Pesticides in Biota

Butler (1973) gave 1965-1972 data on pesticide levels in eastern oysters from Mississippi Sound stations. Gulf Breeze Laboratory analysts detected DDT in 61% of the samples. They found the highest maximum residues in oysters from Biloxi Bay (~ 675 ppb, dry wt.) and in oysters from Bay St. Louis (~620 ppb, dry wt.). These maximum DDT residues in Mississippi oysters were lower than those in twelve of the other fourteen states reported in Butler (1973). In 1971 70% more Mississippi Sound oyster samples had low (< 50 ppb, dry wt.) DDT residues than in previous years; then the trend reversed during the first six months of 1972 when 44% of the residues were > 50 ppb, dry wt. Analysts detected dieldrin in 47% of the oyster samples analyzed 1965-1972. They found dieldrin residues in oyster samples from four of Butler's eight Mississippi stations. The maximum dieldrin residue in Biloxi Bay oysters was 95 ppb, dry wt. and 100 ppb, dry wt. in Bay St. Louis oysters.

Butler (1973) also summarized 1968-1969 data on pesticide levels in eastern oysters from Mobile Bay stations. Gulf Breeze Laboratory detected DDT in 100% of these samples and detected dieldrin in 18% of the samples. The highest maximum DDT residue of samples at all Alabama stations was ~3080 ppb, dry wt.; the highest maximum dieldrin residue was ~ 105 ppb, dry wt. When you compare maximum levels of DDT from sites in the fifteen states where Butler sampled oysters, you find Alabama's maximum level in oysters is fifth highest.

Settine, et al. (1983) reported on the first two years, 1981 and 1982, of their study of oysters, Crassostrea virginica, in Mobile Bay. They identified and measured a large number of organic contaminants in the oysters, providing environmental baseline data which should be useful in planning for industrial development of Mobile Bay. They used Cedar Key, Florida as a control sampling site. Results of these initial studies indicated concentrations of individual PAHs and pesticides were below levels thought to be a health hazard. They found a definite trend of higher levels of pollutants in oyster tissues in spring than in fall. They also found a greater variety of organic pollutants in some sites in Mobile Bay and they found them in higher concentrations than in other sites. Even the worst scenario results of oyster tissue concentration of PAHs and DDTs, namely from spring, 1981 sampling in Cedar Point and in spring, 1982 in Dauphin Island Bay, were not alarmingly high levels:

	Mobile Bay	
	Cedar Point Spring, 1981	Dauphin Island Bay Spring, 1982
PAHs	209 ppb (wet wt.) 1,045 approx. ppb (dry wt)*	421 ppb (wet wt.) 2,105 approx. ppb (dry wt.)*
Pesticides	69 ppb (wet wt.) 347 approx. ppb (dry wt.)*	158 ppb (wet wt.) 790 approx. ppb (dry wt.)*

*approx. dry wt. = .20 (wet wt.)

Marion and Settine (1983) reported on their studies continued from that reported in Settine, et al. (1983). In 1983, they sampled oysters from five of the six Mobile Bay sites they had sampled during the two previous years, plus one additiional site. They used the sites at Cedar Key, Florida again as control sites and added ten clam (Rangia cuneata) sampling sites, mostly further north in Mobile Bay. Many of the compounds they found in bivalves are known to be detrimental to human health, but again in 1983, their results showed concentrations of individual compounds too low to indicate a health hazard. They continued to notice a trend, although not as strong a trend, toward higher levels of organics in tissues sampled in the spring over fall samples. They selectively searched for PCB's in oyster and clam samples, but could not confirm such low levels, near the detection limit of their instrument. They detected DDT and its metabolites at a few sites, but levels were lower than those of 1982. They found greater, though not significantly greater levels of total organic pollutants in oysters from two sites in Mobile Bay: St. Andrews Bay and Dauphin Island Bay. Comparably high levels of naphthalene and phenanthrene indicated presence of diesel fuel and boat traffic in the more enclosed Dauphin Island Bay. Following are total PAH and pesticide levels they found in oysters in the two most contaminated sites in their 1983 Mobile Bay oyster study:

	Dauphin Island Bay Spring, 1983	St. Andrews Bay Spring, 1983
PAHs	323 ppb (wet wt.)	546 (wet wt.)
	1,625 approx. ppb (dry wt.)*	2,730 approx. ppb (dry wt.)*
pesticides	4 ppb (wet wt.)	7 ppb (wet wt.)
	20 approx. ppb (dry wt.)*	35 approx. ppb (dry wt.)*

*approx. dry wt. = .20 (wet wt.)

Marion and Settine sampled and analyzed clams from ten stations in Mobile Bay in 1983 and found 25.5 ppb (wet wt.) or 127.5 ppb (approx. dry wt.) for the average sum of DDT and its derivatives. Clams collected from their sites at Mobile, Alabama, near the McDuffie Island coal-loading facility had the highest concentration of total PAHs (309 ppb, wet wt. or approx. 1545 ppb dry wt.) of their ten clam sampling sites. Clams from Polecat Bay, north of the causeway had the lowest total PAH concentration (58 ppb wet wt./approx. 290 ppb dry wt.) of clams in Mobile Bay. The average concentration of total PAHs found in clams from their ten sites was 211.6 ppb, wet wt. (approx. 1058 ppb, dry wt.).

In 1984 Marion and Settine continued their investigation of Mobile Bay, sampling Rangia in the north and Crassostrea in the south. They sampled closer to industrial sites and added experimental oyster transplanting. Their results were much the same as those of earlier testing. They found pollutants were in low ppb category and were dispersed evenly throughout the Bay. The only exception were results of sampling close to Mobile, Alabama, where levels of pollutants were higher. (Ken Marion, Personal Communication, February 17, 1986)

Marion and Settine (1984) reported concentrations of a large variety of organic contaminants including PAHs and insecticides in oyster and clam tissue from Mobile Bay. They detected metabolites of DDT as in previous years' studies, but found no DDT in 1984. They did not find the obvious seasonal fluctuations in tissue levels of clams in this study that they had noted in oysters the previous three years. They did note a geographical trend that clams north in Mobile Bay and close to the city of Mobile contained a wider variety and higher levels of pollutants than clams from other sites. One of these northern sites, McDuffie Island site, is downstream from the harbor area and just below a coal loading facility. The clams collected there had the highest tissue levels of pollutants. Marion and Settine (1984) did not consider any of their sampling sites "hotspots" of contamination; rather they thought that currents and hydrologic forces in the Bay probably distribute contaminants from areas, such as the harbor in Mobile, to widespread areas of Mobile Bay.

Effects of Organic Contaminants: Sediment Toxicity

In order to evaluate the biological impact of disturbing Mississippi Sound sediments, Lytle and Lytle (1985) collected surface sediment samples and conducted toxicological bioassays. They exposed mysid shrimp and sheepshead minnows to a liquid phase, consisting of a 0.45 micron filtrate from sediment mixed with site water at a 1:4 proportion by volume and exposed these organisms to a particulate phase, consisting of a 1:4 by volume mixture of unfiltered sediment and water. They also tested mysid shrimp and an amphipod in a solid phase, consisting of the sediment settling from the particulate phase with a fresh portion of site water. They diluted site water with seawater for further testing in cases where they observed significant mortalities compared to their seawater controls. They recorded mortalities at 24 hour intervals for 96 hours for bioassays in liquid and particulate phases and only at the end of 96 hours for the solid phase bioassay.

Only 12 of the 34 sedimentary sites the Lytles tested produced significant mortalities. Eight of these were Pascagoula River System sites and four were Biloxi Bay Area sites. They noted that "high levels of hydrocarbons in sediments, particularly aromatic hydrocarbons, are invariably accompanied by significant mortalities in the toxicity bioassays."

Lytle and Lytle (1985) examined six experimental parameters to establish "Environmental Stress Index" values for Mississippi Sound locations:

1. sediment toxicity - harmfulness of resuspended sediments.
2. suspension stability - quantity of sediment that may be suspended and length of time it would remain suspended after disturbance.
3. disturbance probability - likeliness that disturbances may occur by boat traffic, dredging, mainstream flow, etc.
4. biota susceptability - vulnerability of indigenous species

inhabiting the community; extent to which a pollution incident might affect them.

5. pollutant level - concentration of pollutants in sediments.
6. sediment leachability - likeliness that pollutants would be released from sediments into the water column after a disturbance.

Using results of their sediment bioassays and suspension stability experiments and their knowledge of disturbance probability and biota susceptibility, the Lytles assigned a rating of 1-5, with "5" indicating highest potential risk from sediment pollution, to each of these four parameters for each site they studied. Then they multiplied the four values for each site to find an "index product." In this way they identified three trouble areas. The areas with the hightest ratings were the Pogey Plant in the Escatawpa River where there are chemical industries and fish processors (Lytle and Lytle, 1980), Lake Yazoo at the mouth of East Pascagoula River where there are industrial discharges (Lytle and Lytle, 1980), and Gulfport Lake in the Industrial Seaway of Bernard Bayou of western Biloxi Bay, a heavy industrial area with a major sewage outfall and a fisheries trawl station (Lytle and Lytle, 1983).

Some sites they considered in intermediate level danger were Escatawpa River Bridge, Mary Walker Bayou in the west Pascagoula River where boat traffic is heavy (Lytle and Lytle, 1983), Cooling Tower Canal, Rhodes Point which is a multi-use industrial site (Lytle and Lytle, 1983), and Deer Island, a fine-grained depository from Biloxi Bay (Lytle and Lytle, 1983). Of the three large river-bay systems along the coast, the Pascagoula River System, or eastern Sound appeared to have the most polluted sites which may pose a significant threat to the environment. The Lytles found the St. Louis Bay System to be the most pristine along the Gulf Coast, with only background levels of hydrocarbons (Lytle and Lytle, Personal Communication, July 15, 1985). The St. Louis Bay System also had the lowest calculated values for Environmental Stress Index product (Lytle and Lytle, 1985). In all phases of their study the Lytles found that "regions of danger do not cover extensive tracts of waterways, but are limited to relatively small regions...".

Effects of Organic Contaminants: Fish, Crustacean and Mollusk Abnormalities

Lytle and Lytle (1985) referring to John Couch's (U.S. EPA, unpublished, Biological Laboratories, Gulf Breeze, Florida 33561) data, stated that "Aromatics may be responsible for the relatively high incidence (compared with other Gulf of Mexico sites) of presumed neoplastic cells found in the American oyster (Crassostrea virginica) and other mollusks in the Pascagoula River region of the Sound." Creosote plants and coal-fired power plants in the region may be sources of aromatics, and careless disposal of used motor oils may contribute petroleum hydrocarbons to the urban storm runoff (Tanacredi, 1977; Dunn and Fee, 1979; Hunter, et al., 1979).

Couch (1985) studied frequencies of diseases, including neoplasms in oysters and fish from three estuaries with varying degrees of human activity: Mobile Bay, Pascagoula Harbor in Mississippi Sound, and two bays in NW Florida. He found a higher frequency of diseased fish and shellfish in heavily industrialized sites in Pascagoula Harbor than in the other estuaries. Twenty of 4486 oysters from Pascagoula Harbor (.44%) had suspect neoplasms; .13% from Mobile Bay had them; and only .04% from Pensacola Bay in Florida had them.

Tumorous growths, even those caused by protozoans, rarely manifest themselves in Mississippi Sound fishes (Overstreet and Howse, 1977); however, Edwards and Overstreet (1976) found 1 to 10 fibrous tumors in subcutaneous tissue of each of five Mississippi Sound striped mullet they studied. First they examined 1,300 mullet netted off Deer Island and from Davis Bayou in Mississippi Sound and found nine fish with at least one tumor. The tumors they studied did not seem to have been caused by parasites, but had properties which designated them as true neoplasms. Edwards and Overstreet (1976) noted a lack of reports of this type tumor in striped mullet prior to their study. They suggested a posssible relationship between the mullet neoplasms and increasing pollution in Mississippi Sound since both are developments in recent years.

Overstreet and Edwards (1976) reported their unusual discovery of neoplasms in an adult southern flounder collected in September, 1971 from Davis Bayou and in a sea catfish trawled in October, 1974 in the ship canal off the Mississippi coast. Tumors in the two fish resembled each other histologically. The Overstreet and Edwards report was the first ever describing tumors from the southern flounder, Paralichthys lethostigma and the sea catfish, Arius felis, and one of few reports of tumors from any fishes in the entire Gulf of Mexico.

Overstreet and Van Devender (1978) monitored postlarval brown, white and pink shrimps from four Mississippi Sound localities semimonthly for five years, January, 1972 - May, 1977. They observed an abnormality, a protrusion of muscle overgrowth through the ventral portion of the sixth abdominal segment in 33 of the brown and the white shrimp predominantly from one polluted habitat. They found none of these abnormal growths in the 4,573 pink shrimp they examined. They suggested "that an unknown pollutant or combination of pollutants related to sewage or use of boats interferred with the shrimp's normal growth process to cause the abnormality in the shrimp we observed." Although only a small portion of shrimp examined were afflicted, Overstreet and Van Devender found almost all of the afflicted postlarvae were from a site adjacent to a small boat harbor receiving Ocean Springs' treated sewage. That site was presumably most heavily polluted of their four sites. Some of their results also infer that a seasonal concentration of a pollutant may have induced the abnormality. They also suspected a pollutant as a probable cause of the abnormality because they did not see evidence of injury or disease associated with the abnormality and they had valid reason to doubt any direct genetic influence.

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TEXAS BAYS

The Texas coastal system is comprised of a complex series of estuaries and bays (Figure 1). Generally as one moves up the coast from Laguna Madre the bays and estuaries receive more freshwater inflow and in general are more developed. Evapotranspiration and diversions for irrigation are very important factors in altering the "normal" estuarine characteristics of these estuaries (Lind, 1980).

The Texas Department of Water Resources (TDWR) conducts water quality studies on the fresh and coastal waters of the state. These data are computerized and were made available to us from the Texas Natural Resources Information System. Unfortunately the majority of the data available on organic contaminants were not collected at the same locations where the biological samples were collected. However, between 1979 and 1981 the TDWR conducted surveys in four areas to measure priority pollutants in water, sediments and aquatic organisms. In addition to the state data network several special studies related to energy development have been conducted in the area.

Organic Contaminants in Water

The Corpus Christi Inner Harbor which flows into Corpus Christi Bay was surveyed for priority pollutants in September of 1981. Sampling stations for water and sediments are shown in Figure 2. Table 1 presents the priority pollutants detected in water samples; no PAHs or pesticides were found in these samples.

In March of 1981 the Arroyo Colorado estuary (Figure 3) was sampled in a similar manner, and these results are presented in Table 2. In contrast to the Corpus Christi Inner Harbor and Bay a considerable number of organic compounds including halogenated aliphatics and pesticides were found at levels which were of concern as defined in Table 3.

Figure 4 shows the stations sampled in the September 1980 studies of the Neches and Sabine Rivers, while Table 4 presents the results for the water samples. The only organics which were detected at levels of concern were phthalate esters.

Organic Contaminants in Sediments

Table 5 presents the results of the sediment analysis for the Corpus Christi Inner Harbor. Relatively high levels of several PAHs and phthates were found at station 3 near the entrance to Corpus Christi Bay.

Results of the sediment surveys for priority pollutants in the Arroyo Colorado are show in Table 6. DDE levels were elevated at 3 stations as were phthalate esters.

Table 7 presents the results of the sediment samples from the Sabine and Neches Rivers. PAHs were elevated in the Neches relative to the Sabine.

Organic Contaminants in Biota

Residue data for various aquatic animals from Corpus Christi Inner Harbor are shown in Table 8. Phthalate ester residues in trout, flounder and blue crabs were very high at one station indicating a point source. Jensen (1979) reported elevated levels of metals in fish and shellfish from the same station.

Table 9 shows the priority pollutants detected in fish from the Arroyo Colorado in 1981. Residues of chlordane were high in several of the species sampled.

Davis 1984c summarized the history of DDT and toxaphene contamination of the river:

> "Persistently elevated pesticide concentrations have been of particular interest in the Arroyo, as the intensively cultivated watershed is one of the most heavily pesticided areas in the world (Donald White, U.S. Fish and Wildlife Service, personal communication). The USFWS observed that fish collected from the Rio Grande near Mission from 1966-1980 as part of the National Pesticide Monitoring Program had consistently contained DDT, even though USEPA cancelled its usage in 1972. This observation stimulated a succession of studies by several agencies on various aquatic systems in the lower Rio Grande Valley, including the Arroyo Colorado. These studies, which are summarized in (Table 9), have generally shown elevated levels of DDT and toxaphene in fish tissue samples from the Arroyo. There is some evidence that fish-eating birds are being adversely affected, and that a potential human health hazard exists through the consumption of contaminated fish. Although the effects of elevated DDT and toxaphene levels on the aquatic biota of the Arroyo Colorado system have not been adequately documented, there are indications of a degree of adverse impact. Decline of the speckled trout fishery in the lower Laguna Madre during the 1960's was attributed in part to DDT and other organochlorine pesticides contributed by the Arroyo Colorado (Table 10) and several fish kills in the area may have resulted from pesticides."

Butler (1973) summarized data utilizing the eastern oyster to monitor chlorinated pesticides along the Texas coast. DDT, DDE, TDE, dieldrin, endrin and toxaphene were monitored at 13 stations between 1965 and 1972. The Arroyo Colorado, Nueces Bay and Lavaca Bay were the most highly contaminated areas over the period.

Biological Effects

Neches River - Water quality problems in this estuary are attributed to low dissolved oxygen levels during periods of stagnation, a condition worsened by inputs of DO consuming organics from swamps and domestic wastes (Davis, 1984b). Several investigators have characterized the macrobenthic communities of the river and related changes in diversity to (1) industrial pollution (Harrel et al. 1976); (2) depressions in dissolved oxygen (Adsit and Hagen 1978) and (3) turbidity (Darville and Harrel 1980).

Arroyo Colorado - Twidwell (1978) studied the tidal portion of the river and concluded that depressions in benthic diversity were due to depressed DO levels resulting from stratfication and organic loading. As mentioned previously high residues of DDT and toxaphene are found in fishes from the river, however, the effects of these residues on the animals are unknown (Davis 1984). Residues in fish eating birds are within the ranges known to produce adverse effects in certain species (White et al. 1983).

Corpus Christi Inner Harbor - Bowman and Jensen (1985) reported the results of a water quality survey conducted in Corpus Christi Inner Harbor in 1982. They concluded that water quality had improved compared to an earlier study conducted in 1973. Although benthic populations were still limited in the inner harbor they attributed the paucity of the fauna to dredging operations and low oxygen levels due to limited flushing.

Armstrong, et al. (1979) studied the effects of discharges from an oil separator platform in Trinity Bay, from July of 1974 to December of 1975. They found a definite correlation between sediment naphthalene concentrations and the number of species of benthic animals. The benthic fauna increased in both numbers of species and individuals as distance from the discharge increased. They suggested that low 2 ppm of naphthalenes in sediments could be responsible for restricting many species.

Mix (in press) reviewed the literature concerning abnormalities in fish and shellfish from the Gulf of Mexico. With regard to abnormalities in fish he concluded, "The relative scarcity of tumors in fish from the Gulf of Mexico remains somewhat of a mystery. Other surveys, some rather extensive (Sparks, personal communication), have failed to find aquatic animals affected by neoplasms, even though some of the biota inhabiting Texas bays, particularly Galveston Bay, contain a variety of chemical pollutants including phthalates, chlorobenzenes, PCBs, chlorinated pesticides and other compounds (Ray & Giam, 1984). However, compared to the Great Lakes, Puget Sound and other more limited bodies of water, the Gulf of Mexico cannot be considered to have been studied intensively."

Chambers and Sparks (1959) surveyed the fish and shellfish populations of in the Houston Ship Channel and in Galveston Bay. Even through extensive organic contamination was detected no tumorous animals were found. Oysters from Galveston Bay were analyzed for PAHs by Ehrhardt (1972) and found to contain residues of 750 ng/kg (wet weights). "Although he did not look for abnormalities, (Ray, personal communication) examined hundreds of oysters during that time period and found no neoplasms" (Mix, in press).

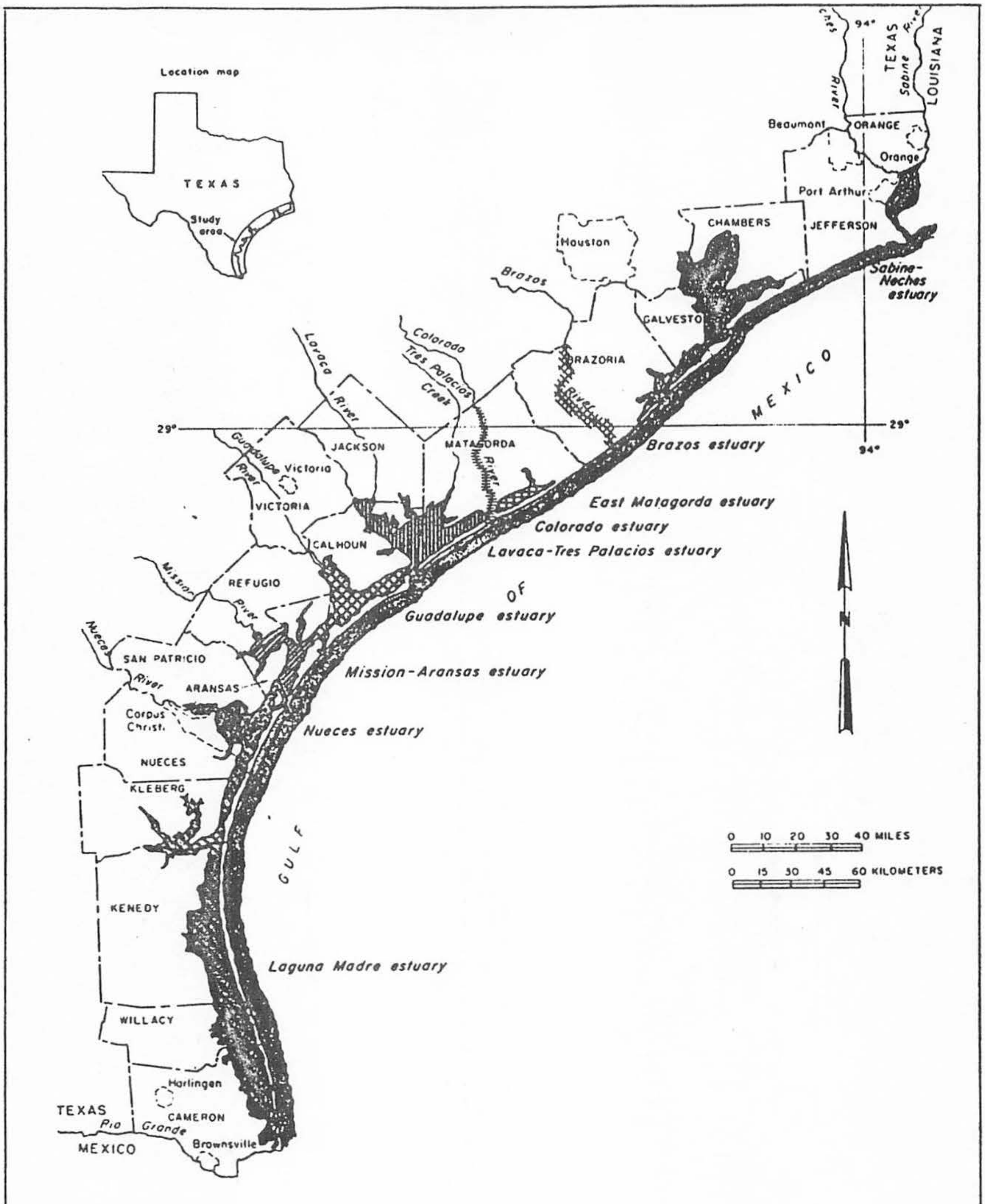

Figure 1

Locations of the Estuaries

(Lind, 1980)

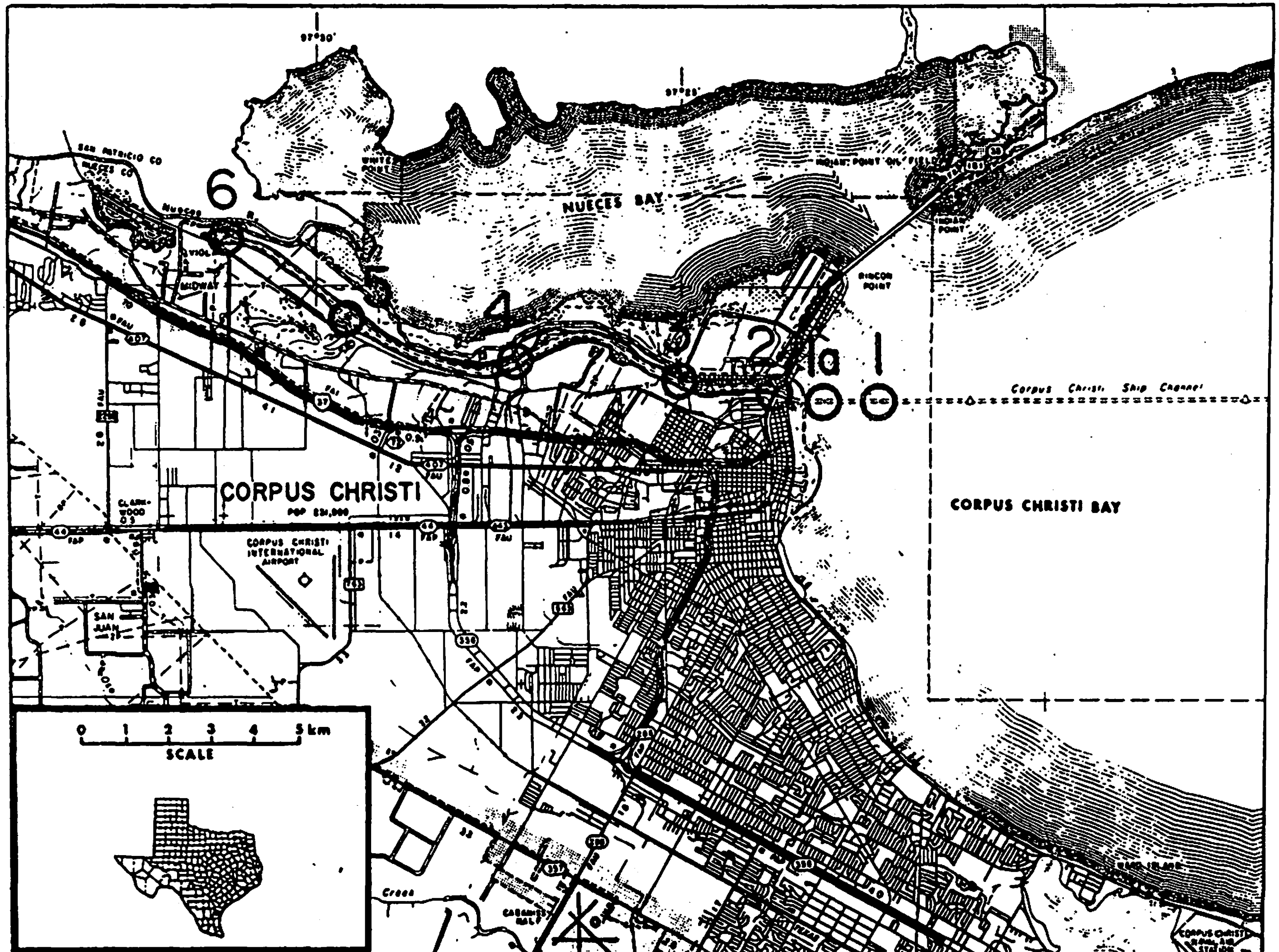

Figure 2 Study area and sampling stations, priority pollutant survey, Corpus Christi Bay and Corpus Christi Inner Harbor, September 1-3, 1981. (Davis, 1984a)

Figure 3

Study area and sampling stations, priority pollutant survey, Arroyo Colorado, March 24-25, 1981.

(Davis, 1984c)

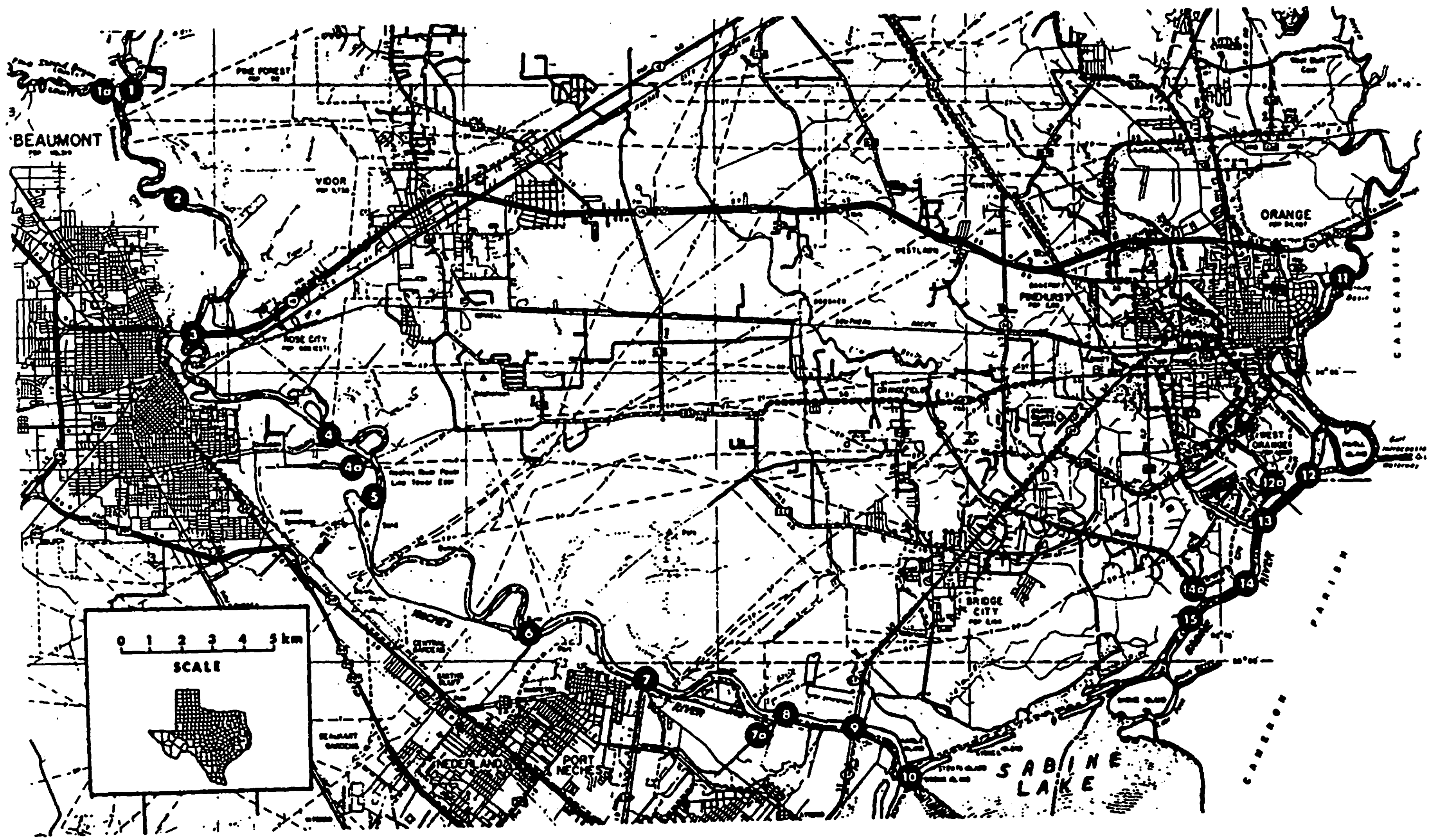

Figure 4 Study area and sampling stations, priority pollutant survey, Sabine River, Neches River, and Sabine Lake, September 22-25, 1980. (Davis, 1984b)

TABLE 1

Priority Pollutants Detected in Water Samples from Corpus Christi Bay and Corpus Christi Inner Harbor, September 1-3, 1981

Parameter[a]	Stations					
	1	2	3	4[b]	5	6
PHENOLS AND CRESOLS						
phenolics recoverable				<10-12		
METALS						
arsenic	2,000*	2,000*				
beryllium	2*	3*	3*	3-4*	5*	3*
cadmium	4	4	5*	4-29*	4	3
chromium	2	2	2	3	2	2
copper				<100-100*	100*	100*
lead	1	2	1	<1-3	2	
mercury	0.6*	0.9*	0.6*	0.9-4.2*	1.2*	
nickel	34*	32*	33*	29-31*	33*	31*
silver	28*	25*	22*	23-27*	28*	29*
thallium				<450-460*	624*	624*
zinc	67*	51	77*	154-172*	96*	84*
GENERAL INORGANICS						
cyanide					12*	

a - all in µg/L

b - range of concentrations measured on triplicate samples

* - Value exceeded one or more of the screening criteria listed in Appendix C

Blanks in the table indicate that the parameter was analyzed but not detected.

(Davis, 1984a)

TABLE 2

Priority Pollutants Detected in Water Samples from the Arroyo Colorado, March 24-25, 1981

Parameter[a]	Station						
	1	2	3	4	5[b]	6[c]	7[c]
PHENOLS AND CRESOLS							
phenolics recoverable	76*		25*	20	35-48*	480*	
HALOGENATED ALIPHATICS							
bromodichloromethane					ND-7.0		
bromoform					ND-10.1*		
chloroform					ND-8.1		
dibromochloromethane					ND-14.3*		
methylene chloride	23.9	31.2		40.7	ND-27.0		
1,1,2,2-tetrachloroethane					ND-28.8*		
tetrachloroethylene	6.6	66.9*	77.2*	65.6	63.4-79.9*		
1,1,2-trichloroethane					ND-20.3*		
trichloroethylene	67.3*	12.4*	15.2*	14.9*	9.2-27.9*	6.3	42.4*
1,3-trans-dichloropropene					ND-13.7*		
MONOCYCLIC AROMATICS							
benzene					ND-14.1		7.2
chlorobenzene					ND-5.7		
toluene					ND-3.6		
METALS							
arsenic	20*	20*	20*	30*	10-30*	20*	20*
mercury					ND-1*		
zinc			30				

TABLE 2 (Continued)

Parameter[a]	Station						
	1	2	3	4	5[b]	6[c]	7[c]
<u>PESTICIDES</u>							
chlordane						0.03*	
p,p' DDD		0.56*	0.10		ND–0.32*		
gamma-BHC (lindane)					0.03–0.20*		
heptachlor epoxide		0.28*	0.23*	0.11*	ND–0.51*		
<u>GENERAL INORGANICS</u>							
cyanide		20*	20*	20*			

a - all values reported in μg/L

b - range of concentrations measured on triplicate samples; ND = not detected

c - tidal stations (all others freshwater)

* - value exceeded one or more of the screening criteria listed in Appendix C

Blanks in the table indicate that the parameter was analyzed but not detected.

(Davis, 1984c)

TABLE 3

Priority Pollutants of Primary Significance with Regard to Future Monitoring Emphasis in the Arroyo Colorado

Parameter	Water				Sediment	Tissue		
	National 85th Percentile Exceeded at Multiple Stations	10^{-5} Lifetime Cancer Risk Value Exceeded at Multiple Stations	Acute Toxicity Levels Exceeded	Chronic Toxicity Levels Exceeded at Multiple Stations	National 85th Percentile Exceeded at Multiple Stations	National 85th Percentile Exceeded at Multiple Stations	USFDA Levels for Edible Tissue Exceeded	Metals Levels Exceeded 1.0 mg/kg
1) phenolics recoverable	X							
2) tetrachloroethylene	X							
3) trichloroethylene	X							
4) arsenic	X	X				X		X
5) cadmium*	X			X				
6) chromium						X		
7) copper						X		X
8) mercury					X			
9) nickel								X
10) zinc						X		X
11) chlordane						X	X	
12) p,p' DDD	X							
13) p,p' DDE					X			
14) DDT + metabolites		X		X				
15) heptachlor epoxide	X							
16) toxaphene**						X	X	
17) cyanide				X				

* - did not exceed criteria in present study; exceedances based on the findings of Black and Veatch, 1981

** - not detected in the present study; exceedances based on the findings of White et al., 1983

(Davis, 1984c)

TABLE 4
Priority Pollutants Detected in Water Samples from the Sabine River, Neches River, and Sabine Lake, September 22-25, 1980

Parameter[b]	Station 1[a]	1a[a]	2[a]	3[a]	4	4a	5	6	7	7a
PHENOLS AND CRESOLS										
pentachlorophenol										
phenol (C_6H_5OH) single compound	<1[c]	<1	<1				ND-<1[d]		<1	
phenolics recoverable						2			.	3
HALOGENATED ALIPHATICS										
bromoform						1.3	<1-1.1			
chloroform				<1			ND-<1	0.7	<1	
dibromochloromethane										
methylene chloride	2	1	2	<1			ND-1		1	
1,1,1-trichloroethane										
POLYCYCLIC AROMATIC HYDROCARBONS										
fluoranthene										
phenanthrene										
pyrene										
MONOCYCLIC AROMATICS										
toluene										
METALS										
antimony	3								2	
arsenic	2*	35*	2*	2*	3*	2*	1-3*	3*	2*	3*
cadmium	20*	3*					7*		25*	
chromium	64*	4*	3*	3*			24*		88*	
copper	39*	9*	7*	7*			13*		45*	
lead	293*						63*		415*	
nickel	67*	8	10	10			22*		93*	
selenium	7						2		8	
silver		1*							1	
thallium	1								3	
zinc	77*	6	6	6			28		101*	20

TABLE 4 (Continued)

Parameter[b]	Station 1[a]	1a[a]	2[a]	3[a]	4	4a	5	6	7	7a
PESTICIDES										
aldrin									< 0.1	
alpha benzene hexachloride	<0.1	<0.1	<0.1	<0.1			ND-<0.1			
delta benzene hexachloride	<0.1	<0.1	<0.1	<0.1			ND-<0.1		< 0.1	
dieldrin			<0.1							
endosulfan alpha			<0.1							
endosulfan beta									< 0.1	
endosulfan sulfate		0.1					ND-<0.1		< 0.1	
endrin	< 0.1		< 0.1		--					
endrin aldehyde										
gamma benzene hexachloride (lindane)	<0.1		<0.1				ND-<0.1		< 0.1	
heptachlor	<0.1	<0.1	<0.1	<0.1						
heptachlor epoxide	<0.1	<0.1	<0.1						< 0.1	
P,P' DDD	<0.1		<0.1							
P,P' DDE										
PHTHALATE ESTERS										
bis(2-ethylhexyl) phthalate	2	2	10*	2			ND-14*		45*	
butyl benzyl phthalate										
diethyl phthalate	< 1	< 1	< 1	< 1			ND-<1		< 1	
dimethyl phthalate										
di-n-butyl phthalate	5	3	2	1			ND-<1		< 1	
di-n-octyl phthalate										
GENERAL INORGANICS										
cyanide					2*		0.005*	0.005*		

TABLE 4 (Continued)

Parameter[b]	Station 8	9	10	11	12	12a	13	14	14a	15
PHENOLS AND CRESOLS										
pentachlorophenol					<1					
phenol (C_6H_5OH) single compound		<1		<1	5		<1			
phenolics recoverable						2			4	
HALOGENATED ALIPHATICS										
bromoform	1	ND-0.5			1	2.1	1	1	2.1	<1
chloroform		ND-<1	<1	<1	<1		<1	<1		
dibromochloromethane					<1		<1			
methylene chloride		1	1	1	2		1	1		1
1,1,1-trichloroethane		ND-0.5		<1						
POLYCYCLIC AROMATIC HYDROCARBONS										
fluoranthene								<1		
phenathrene								<1		
pyrene								<1		
MONOCYCLIC AROMATICS										
toluene							<1			
METALS										
antimony		3					3			2
arsenic	3*	1-3*	2*	2*	2*		1*	3*	3*	2*
cadmium		17*	18*	18*	21*		10*	25*		20*
chromium		55*	53*	56*	60*		16	110*		75*
copper		35*	32*	37*	33*		23*	44*		35*
lead		256*	217*	350*	262*			419*		301*
nickel		61*	68*	69*	60*		28*	90*		52*
selenium		7	4	7	7			4		4
silver			1	1						
thallium		3	2	2	3		1	2		1
zinc		65*	62*	71*	73*		30	114*		86*

TABLE 4 (Continued)

Parameter[b]	Station									
	8	9	10	11	12	12a	13	14	14a	15
PESTICIDES										
aldrin										
alpha benzene hexachloride				<0.1			<0.1			<0.1
delta benzene hexachloride			<0.1	<0.1	<0.1		<0.1	<0.1		<0.1
dieldrin		ND-<0.1								
endosulfan alpha										
endosulfan beta										
endosulfan sulfate		ND-<0.1		0.1						
endrin										
endrin aldehyde		ND-<0.1								
gamma benzene hexachloride (lindane)										
heptachlor										<0.1
heptachlor epoxide		ND-<0.1								
P,P' DDD		<0.1								
P,P' DDE		<0.1								
PHTHALATE ESTERS										
bis(2-ethylhexyl) phthalate		5-8*	2	4	2		1	16*		
butyl benzyl phthalate							<1	2		
diethyl phthalate		<1-70*	<1	<1	<1		<1	<1		<1
dimethyl phthalate					<1		<1	<1		
di-n-butyl phthalate		<1-1	1	1	2		<1	<1		<1
di-n-octyl phthalate								<1		
GENERAL INORGANICS										
cyanide	0.006*	0.005*				5*			5*	

a - freshwater stations (all others tidally influenced)
b - all in µg/L, except cyanide (mg/L)
c - a number appearing as <x indicates a parameter detected at an unquantifiably low concentration
d - range of concentrations measured on multiple samples; ND = not detected
* - value exceeded one or more of the screening criteria listed in Appendix C

Blanks in the table indicate that the parameter was analyzed but not detected.

(Davis, 1984b)

TABLE 5

Priority Pollutants Detected in Sediment Samples from Corpus Christi Bay and Corpus Christi Inner Harbor, September 1-3, 1981

Parameter[a]	Station					
	1	2	3	4	5	6
PHENOLS AND CRESOLS						
phenolics recoverable	2.1	1.2	1.4	2.1	3.4	1.1
HALOGENATED ALIPHATICS						
tetrachloroethylene					44	
POLYCYCLIC AROMATIC HYDROCARBONS						
benzo(A)anthracene 1,2-benanthracene			198	330	99	
fluoranthene		31	534			
pyrene		32	369			
MONOCYCLIC AROMATICS						
1,2-dichlorobenzene	10	16				
METALS						
antimony				1.60	0.39	1.00
arsenic	5.55*	5.62*	4.40	6.99*	6.58*	4.00
cadmium	1.62	0.60	1.90	7.98*	5.92*	4.50*
chromium	18.8	20.8	41.5	49.9	26.4	18.7
copper	15.1	14.9	25.0	49.9	26.7	17.5
lead	15.7	23.8	49.2	145.0*	28.9	29.8
mercury		0.807*	0.472	1.900*	1.060*	0.076
nickel	14.1	12.8	15.0	13.0	12.6	9.00
silver		0.255	0.909	1.300	0.485	0.200
zinc	89.0*	138.0*	285.0*	125.0*	593.0*	5.0
PHTHALATE ESTERS						
bis(2-ethylhexyl)phthalate			91			
diethyl phthalate	140	642	666			
di-n-butyl phthalate	61					
GENERAL INORGANICS						
cyanide						1.1

a - all in μg/kg, except phenolics recoverable, metals, and cyanide (mg/kg)

* - value exceeded one or more of the screening criteria listed in Appendix C

Blanks in the table indicate that the parameter was analyzed but not detected.

(Davis, 1984a)

TABLE 6

Priority Pollutants Detected in Sediment Samples from the Arroyo Colorado, March 24-25, 1981

Parameter[a]	Station 1	2	3	4	5	6	7
PHENOLS AND CRESOLS							
phenolics recoverable	2.4	1.4	2.2	1.7	1.1	1.0	1.3
POLYCYCLIC AROMATIC HYDROCARBONS							
napthalene		30			19		
MONOCYCLIC AROMATICS							
toluene		252	24				
METALS							
antimony	1.60	1.03	0.79		0.77	0.88	0.80
arsenic	2.59	2.74	5.42*	1.81	4.23	3.29	3.61
beryllium	1.20	1.03	1.97	0.40	1.73	1.76	2.00
cadmium		0.17	0.20			0.22	0.50
chromium	13.6	14.1	21.5	5.3	20.9	21.1	19.2
copper	15.0	17.1	61.6*	5.0	24.0	18.9	17.5
lead	41.5	19.5	22.9	1.6	38.9	19.3	11.0
mercury		0.343		4.240*	0.471	3.970*	1.230*
nickel	13.0	12.9	15.8	10.3	14.4	17.6	16.0
silver	0.10	0.51	0.39		0.87	0.33	
zinc	64.6	66.9	86.5*	16.8	83.8*	76.1*	73.7
PESTICIDES							
p,p' DDE		22*	39*		10	23*	

TABLE 6 (Continued)

Parameter[a]	Station						
	1	2	3	4	5	6	7
PHTHALATE ESTERS							
bis(2-ethylhexyl)phthalate					284	168	
diethyl phthalate	24			122	278	150	54
di-n-butyl phthalate	34						

a - all values reported in μg/kg, except metals and phenolics recoverable (mg/kg)

* - value exceeded one or more of the screening criteria listed in Appendix C

Blanks in the table indicate that the parameter was analyzed but not detected.

(Davis, 1984c)

TABLE 7
Priority Pollutants Detected in Sediment Samples from the Sabine River, Neches River, and Sabine Lake, September 22-25, 1980

Parameter[d]	Station									
	1	1a	2	3	4	4a	5	6	7	7a
<u>PHENOLS AND CRESOLS</u>										
phenol (C_6H_5OH) single compound										
phenolics recoverable				167	157	180				
<u>POLYCYCLIC AROMATIC HYDROCARBONS</u>										
acenapthylene										
anthracene							det.-13[b]			
anthracene + phenanthrene[d]					2500					
benzo(a)pyrene						5000	ND-det.[c]			
benzo(k)fluoranthene + benzo(B)fluoranthene[d]					det.		ND-det.			
chrysene							det.-34			
chrysene + benzo(A)anthracene 1,2-benzanthracene[d]					1200	5500				det.
fluoranthene				det.	2200	1000	74-1100		31	det.
fluorene					det.					
phenanthrene										
pyrene				det.	1500	6000	55-1000		52	det.
<u>METALS</u>										
arsenic	15*	18*	16*	19.9*		36*	ND-19*	27.9*	21*	23.7*
beryllium	0.44	0.39	0.36				ND-0.5		0.5	
chromium	19	17	16	66.4*	40.6	763*	21-96*	60.8*	28	136*
copper	1.9	0.75	1.2	18.6	4.8	86.7*	1.7-19	13.8	3.3	109*
lead	2.3	1.3	2.7	47.8	26.3	867*	6.4-38		1.3	40.8
mercury	0.013	0.006	0.01			0.41	0.01-0.11		0.04	0.43
nickel	2.2	2.6	2.3	8.0	7.2	30.4	2.5-7.7	13.8	2.9	19.1
zinc	20	16.2	14.4	125*	76.5*	377*	13.5-103*	83*	18	109
<u>PHTHALATE ESTERS</u>										
bis(2-ethylhexyl)phthalate										
<u>GENERAL INORGANICS</u>										
cyanide						1.35	0.92			0.157

TABLE 7(Continued)

Parameter[a]	Station									
	8	9	10	11	12	12a	13	14	14a	15
PHENOLS AND CRESOLS										
phenol(C_6H_5OH) single compound			0.004		0.036					
phenolics recoverable										
POLYCYCLIC AROMATIC HYDROCARBONS										
acenapthylene					110		17	46		
anthracene					24					
anthracene + phenanthrene[d]									det.	
benzo(A)pyrene									det.	
benzo(K)fluoranthene + benzo(B)fluoranthene[d]									det.	
chrysene								31		
chrysene + benzo(A)anthracene 1,2-benzanthracene[d]									det.	
fluoranthene			77		180	det.	14	17	det.	
fluorene										
phenanthrene					21		11	10		
pyrene					220	det.	21	46	1200	
METALS										
arsenic	13.8*	15-40.2*	15*	19*	28*	22.6*	20*	22*	24.7*	15*
beryllium		0.5	0.45	0.43	0.44		0.43	0.45		0.43
chromium	50.7	21-85.8*	18	20.5	26	45.3	14.5	16	46.2	20
copper	31	4.1-23.4	3.6	1.1	4.85	20.1	3.3	1.8	34.7	4.2
lead		2.6	2.8	1.2	2.05	25.1	3.9	2.2	0.023	1.3
mercury		0.03-0.12	0.028	0.016	0.021		0.033	0.026		0.012
nickel	11.3	3.5-23.4	2.3	2.25	2.2	10.1	0.86	2.7	30	2.3
zinc	84.5*	24-133*	22	16.5	25	90.5*	11	16	74	24
PHTHALATE ESTERS										
bis(2-ethylhexyl)phthalate						det.				
GENERAL INORGANICS										
cyanide						0.0341				

a - all in μg/kg, except metals, phenols, cyanide (mg/kg)
b - det. = detected at an unquantifiably low concentration
c - ND = not detected
d - compounds not separable by the analytical method employed
* - value exceeded one or more of the screening criteria listed in Appendix C

Blanks in the table indicate that the parameter was analyzed but not detected.

(Davis, 1984b)

TABLE 8

Priority Pollutants Detected in Aquatic Organism Tissue Samples from Corpus Christi Bay and Corpus Christi Inner Harbor, September 1-3, 1981

Parameter[a]	Stations											
	1a							4				
	southern flounder (1)b	menhaden (5)	white shrimp (20)	sand trout (5)	speckled trout (2)	hardhead catfish (5)	blue crab (5)	American oyster (25)	speckled trout (2)	anchovy (200)	southern flounder (2)	blue crab (5)
PHENOLS AND CRESOLS												
phenolics recoverable	2.7	1.4						5.9			3.6	2.4
ETHERS												
2-chloroethyl vinyl ether										10		
4-bromophenyl phenyl ether										10		
HALOGENATED ALIPHATICS												
methylene chloride	24		17				16					
POLYCYCLIC AROMATIC HYDROCARBONS												
napthalene	34	26	14	23	13	43		101		25	19	31
MONOCYCLIC AROMATICS												
1,2-dichlorobenzene						15						
1,2,4-trichlorobenzene						29						
METALS												
arsenic	0.116	0.163	0.110	0.188	1.160*	0.096	0.109	0.373*	0.131	0.102	0.110	0.119
beryllium	0.029	0.018										
cadmium	0.108	0.044	0.030	0.020	0.010	0.010	0.097	15.496*	0.009	0.093	0.094	1.368*
chromium	2.233*	0.470*	0.622*	0.196	0.209	0.283	1.413*	2.217*	0.131	1.132*	0.643*	0.789*
copper	1.606*	1.828*	5.525*	2.427*		0.585	12.190*	69.464*	2.518*	3.452*	1.748*	13.482*
lead	0.343	0.568	0.099	0.088	0.119	0.185	7.343*	0.910*	0.122	0.186	0.085	0.190
mercury	0.206	0.156	0.130	0.160	0.222	0.319	0.164	0.454	0.349	0.215	0.307	0.218
nickel	0.235	0.089	0.197	0.117	0.080	0.088	0.339	0.237	0.056	0.130	0.321	0.180
selenium	0.116	0.163	0.110	0.188	0.086	0.096	0.109	0.373	0.131	0.102	0.110	0.119
silver							0.010	0.950*				0.020
zinc	16.059*	14.549*	17.168*	13.408*	16.216*	21.546*	21.411*	5,798.536*	11.232*	53.818*	26.930*	56.527*
PCB'S AND RELATED COMPOUNDS												
total pcb's		trace		886					trace	trace		
PHTHALATE ESTERS												
bis(2-ethylhexyl)phthalate											8,530	9,663
diethyl phthalate	126	130	92					181	3,003	96		
dimethyl phthalate	15								64			
di-n-butyl phthalate							49					118
GENERAL INORGANICS												
cyanide								3.9	1.2			

TABLE 8 (Continued)

Parameter[a]	Station 5				
	blue crab (5)	sand trout (5)	hardhead catfish (5)	pinfish (5)	menhaden (1)
PHENOLS AND CRESOLS					
phenolics recoverable	3.0	3.2	1.7	1.3	10.6
ETHERS					
2-chloroethyl vinyl ether					
4-bromophenyl phenyl ether					
HALOGENATED ALIPHATICS					
methylene chloride					
POLYCYCLIC AROMATIC HYDROCARBONS					
napthalene		14	14	13	
MONOCYCLIC AROMATICS					
1,2-dichlorobenzene					
1,2,4-trichlorobenzene					
METALS					
arsenic	0.112	0.962*	0.101	0.093	0.545*
beryllium		0.038	0.010		
cadmium	2.144*	0.019	0.136	0.098	0.345*
chromium	0.626*		0.136	0.195	0.792*
copper	11.687*	3.173*	0.988	0.790	2.073*
lead	0.239	0.125	0.242	0.380	0.705
mercury	0.202	0.343	0.177	0.219	0.114
nickel	0.184	0.038	0.068	0.088	0.144
selenium	0.112	0.128	0.352	0.093	0.545
silver					
zinc	43.250*	8.369*	28.973*	27.501*	20.297*
PCB'S AND RELATED COMPOUNDS					
total pcb's					
PHTHALATE ESTERS					
bis(2-ethylhexyl)phthalate					
diethyl phthalate	20	11			
dimethyl phthalate					
di-n-butyl phthalate	67				
GENERAL INORGANICS					
cyanide	1.0		1.4		2.2

a - all in μg/kg, except phenolics recoverable, metals, and cyanide (mg/kg)

b - numbers in parentheses are the number of individuals composited for analysis

* - value exceeded one or more of the screening criteria listed in Appendix C

Blanks in the table indicate that the parameter was analyzed but not detected.

(Davis, 1984a)

TABLE 9

Priority Pollutants Detected in Aquatic Organism Tissue Samples from the Arroyo Colorado, March 24-25, 1981

Parameter[a]	Station 1			Station 3				Station 5					Station 7		
	gizzard shad (5)b	channel catfish (1)	sailfin molly (20)	gizzard shad (5)	channel catfish (5)c	channel catfish (1)d	carp (3)	gizzard shad (3)	channel catfish (5)e	channel catfish (1)f	carp (4)	large-mouth bass (1)	croakers (10)	hard-head catfish (5)	shrimp (20)
PHENOLS AND CRESOLS															
phenolics recoverable		1.5	1.3		3.3	5.2	3.8			2.3	3.8	1.7		3.8	2.9
HALOGENATED ALIPHATICS															
1,2-dichlorobenzene										14					
1,3-dichlorobenzene										16					
POLYCYCLIC AROMATIC HYDROCARBONS															
napthalene	30	68							46	27					
METALS															
arsenic	0.108	0.103	0.132	0.118	0.120	0.147	0.102	0.121	0.148	0.098	0.106	0.720*	0.102	1.100*	1.360*
beryllium	0.038		0.029	0.038	0.010	0.009		0.008							
cadmium	0.019	0.018	0.019	0.019	0.109	0.018	0.021	0.016	0.020	0.024	0.039	0.010	0.021	0.018	0.048
chromium	0.672*	0.192	0.544*	0.255	0.283	0.532*	0.126	0.154	0.051		0.145	0.105	0.144	0.109	
copper	1.939*	1.172*	2.848*	1.728*	1.608*	1.677*			0.935	0.435		1.918*	0.759	0.672	8.541*
lead	0.422	0.128	0.330	0.217	0.234	0.119	0.199	0.138	0.071	0.040	0.174	0.147	0.072	0.109	0.019
mercury	0.169	0.245	0.280	0.193	0.262	0.182	0.263	0.125	0.312	0.318	0.204	0.300	0.174	0.253	0.120
nickel	0.298	1.108*	0.311	0.094	0.166	0.284	0.167	0.114	0.112	0.040	0.058	0.189	0.164	0.182	0.076
selenium	0.108	0.103	0.132	0.118	0.120	0.098	0.102	0.121	0.148	0.098	0.106	0.120	0.102	0.096	0.101
silver			0.058												
zinc	12.289*	15.936*	13.024*	7.480*	16.660*	11.730*	19.253*	14.940*	13.618*	5.210*	37.303*	11.636*	11.079*	97.131*	12.730*
PESTICIDES															
chlordane	537*	593*				542*	trace	trace	trace	874*	trace	608*			
p,p' DDE											473				
PHTHALATE ESTERS															
bis(2-ethylhexyl)phthalate		136													
diethyl phthalate		180	513	307					161	127	241				82
dimethyl phthalate			101	18							23				
GENERAL INORGANICS															
cyanide		1.4	1.0			2.0	1.1	1.3	1.2		1.4	1.5		1.0	1.3
PCB'S AND RELATED COMPOUNDS															
total pcb's							trace								

(footnotes on next page)

Table 9 (cont.)

[a] - all values reported in μg/kg, except phenolics recoverable, metals, and cyanide (mg/kg)

[b] - numbers in parentheses are the number of individuals composited for analysis

[c] - composite sample of 5 channel catfish, each approximately 25 cm in total length

[d] - channel catfish approximately 36 cm in total length

[e] - composite sample of 5 channel catfish, each approximately 30 cm in total length

[f] - edible tissue sample (fillets) taken from a channel catfish weighing approximately 6 kg

* - value exceeded one or more of the screening criteria listed in Appendix C

Blanks in the table indicate that the parameter was analyzed but not detected.

(Davis, 1984c)

Table 10

Chronology of DDT and Toxaphene Investigations on the Arroyo Colorado

1969–70	Study by Texas Parks and Wildlife Department (TP&WD) indicated decline of speckled trout fishery in lower Laguna Madre during 1960's, attributed to DDT and other organochlorine pesticides contributed by the Arroyo Colorado.
1970–72	Study by Texas Department of Agriculture revealed DDT levels in the upper Arroyo Colorado higher than any previously observed in Texas.
1972	Use of DDT restricted by U.S. Environmental Protection Agency (USEPA), allowing only limited usage for public health purposes.
1973–76	Fish tissue samples collected by TP&WD in lower Arroyo indicated elevated levels of DDT and toxaphene, but values were less than the 5.0 mg/kg U.S. Food and Drug Administration (USFDA) limits for edible fish tissue.
1976	Whole fish samples collected by U.S. Fish and Wildlife Service (USFWS) in the upper Arroyo contained levels of DDT and toxaphene 3–6 times the USFDA limits. Edible tissue samples were not analyzed.
1977	Fish tissue samples collected by the Texas Department of Water Resources (TDWR) in the lower Arroyo showed DDT and toxaphene levels well within USFDA limits.
1978	Fish tissue samples collected by USFWS in the upper Arroyo showed DDT and toxaphene levels 2–6 times the USFDA limits in whole fish samples, and above the USFDA limits in edible fillets.
February 1979	Copy of 1976 USFWS data forwarded to Texas Department of Health (TD of H).
March 1979	Copy of 1978 USFWS data forwarded to TD of H.
November 1979	Joint fish tissue study conducted by USFWS and TDWR with samples split between three laboratories. Data showed levels of DDT and toxaphene in whole fish in the upper Arroyo 3–6 times higher than USFDA limits. However, the majority of analyses were performed on whole fish and not edible tissue for which USFDA guidelines are intended.

Table 10 (cont.)

July 1980	USFWS in letter to USEPA recommended that the Arroyo Colorado between McAllen and the Port of Harlingen be closed to sport fishing or at least that warning signs be posted. USEPA notified TD of H of USFWS recommendations. TD of H then undertook an independent study to ascertain levels in edible fillets since all prior studies dealt mainly with whole fish.
September 1980	TD of H data indicated levels of DDT and toxaphene exceeding USFDA criteria for edible fish tissue. TD of H then issued a public advisory discouraging consumption of fish from the Arroyo Colorado near the Port of Harlingen and from Llano Grande Lake.
1980	Study published documenting a decline in fish-eating bird populations along the Texas coast as a result of direct mortality and decreased reproductive success due to eggshell thinning induced by DDT and DDE (King et al., 1980).
March 1981	The study described in this report was conducted by TDWR.
June 1981	Lower Rio Grande Valley Development Council (LRGVDC) published 208 study conducted from October 1979 to September 1980 pertaining to levels and sources of organic pesticides and metals in water, sediment, and fish tissue in the lower Rio Grande Valley. Data indicated recent introductions of DDT into the Rio Grande and its subsequent distribution throughout the valley by irrigation canals. Concentrations in the Arroyo Colorado were the highest observed in the Rio Grande Valley. Strong circumstantial evidence indicated that DDT enters the Rio Grande from Mexico. Other possible sources mentioned were runoff from abandoned formulating plant sites and/or storage areas, leachates from abandoned solid waste disposal sites, and illegal, small-scale usage of DDT purchased in Mexico. Recommendations were made that a comprehensive survey of the Arroyo Colorado be conducted in an attempt to locate point and nonpoint inputs of DDT (Black and Veatch, 1981).
April 1982	LRGVDC published draft report of follow-up 208 study conducted in an attempt to identify sources of DDT input to the Arroyo Colorado. Data indicated higher levels of DDT in fish from Llano Grande Lake than from the lower Rio Grande or the North Floodway. DDT residue was relatively uniformly distributed in sediments in the Arroyo between Mission and Llano Grande Lake. Runoff from the Hayes-Sammons site in Mission, a former DDT formulating and storage site, was proposed as the possible major contribution of DDT to the Arroyo. DDT was generally higher in wastewater treatment plants that receive washings

	from fruit and vegetable processing operations than in plants not receiving such wastes. DDT and toxaphene were detected in two of ten samples taken from subsurface agricultural drains. Recommendations were made that further investigations be conducted into the sources of DDT in the Arroyo (Black and Veatch, 1982).
August 1983	Study published documenting elevated DDT and toxaphene residues in fishes and birds in the lower Rio Grande Valley (White et al., 1983), based in part on USFWS studies mentioned above. DDE and toxaphene levels in fish from the Arroyo Colorado were reported to be higher than samples from other areas in the valley, with both ranging up to 31.5 mg/kg. Median DDE residues in fish-eating bird carcasses from near the Arroyo ranged up to 34 mg/kg, and 81 mg/kg in individual specimens. The levels of contamination of fish and birds were within, or above, ranges known to produce adverse effects in certain species. Some evidence of adverse effects on birds was obtained through observations that eggs of some species are high in DDE, eggshells are thin, and reproduction is poorer than in birds at other localities along the Texas coast. The impacts of DDE and toxaphene on the fishes and aquatic invertebrates of the Arroyo Colorado were reported to be unknown, and additional research was recommended to elucidate these impacts. Sources of contamination to the Arroyo Colorado system were theorized to include past and present use of pesticides on surrounding croplands, and runoff from the aforementioned abandoned pesticide plant at Mission.

(Davis, 1984c)

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The Chesapeake Bay (Figure 1) and particularly its more industrialized sub-estuaries are subjected to a variety of toxic chemicals. In some cases, e.g. Kepone, damages to resource uses are well documented (Bender and Huggett, 1984), while in others, e.g. PAHs, potential impacts are indicated from field and laboratory studies (Hargis et al., 1984 and Huggett et al., 1986).

Organic Contaminants in Sediments

Bieri et al. (1982a,b) reported on a comprehensive program to determine the levels of PAHs in sediments from the main stem of the Bay and the Elizabeth and Patapsco rivers.

Figure 2 shows the location of their sediment sampling stations in the Bay. Figures 3 and 4 present data on total aromatics in surface sediments at these stations in the spring and fall of 1979, respectively. Figures 5 and 6 give the concentration sums of PAHs related to the combustion of carbonaceous matter for the same sampling period. An examination of data shown in Figures 5 and 6 indicates that tributary streams are the major source of these compounds to the Bay proper (Station 1 - Lynnhaven; Station 3 - James; Station 5 - York; Station 8 - Rappahannock; Station 13 - Potomac; Station 23 - Patapsco; Station 26 and 27 - Susquehanna).

Figures 7 and 8 present data on levels of total resolved PAHs and pyrogenic PAHs in the Elizabeth River. Note that the levels found here are 2-4 orders of magnitude higher than those found in the Bay proper (Figures 3,4,5 and 6). A more detailed discussion of the sources and effects of these elevated PAH levels is presented latter.

Figures 9 and 10 show data on the levels of total resolved PAHs and pyrogenic PAHs in the Patapsco River (Baltimore Harbor). As was the case in the Elizabeth levels in the Patapsco are several orders of magnitude higher than those found in the Bay proper.

Bender et al. (1986) sampled sediments along the James, York and Rappahannock rivers for PAHs. The results from this survey are shown in Table 1.

Bender and Huggett (1984) reviewed sediment contamination of the James with Kepone relative to the residues found in fish and shellfish. A more detailed discussion of this data is presented in the section on Kepone effects.

Organic Contaminants in Biota Residues and Effects

Polynuclear Aromatic Hydrocarbons

Field observations suggests that fishes in the Elizabeth River, Virginia are severely stressed because of the contamination of the sediments by polynuclear aromatic hydrocarbons (PAHs). Figure 11, from Huggett et al.

(1986) shows the distribution of one PAH, benzo(a)pyrene, in surface sediments along the Southern Branch of the Elizabeth River. Incidences of abnormalities, e.g. skin lesions, cataracts and fin erosion in native fishes collected along the river increase at stations which are heavily contaminated (see Table 2 and Figure 12).

In laboratory exposures of spot to contaminated sediments from the Elizabeth River dermal lesions and fin rot similar to those in fish from the Elizabeth River were observed (Hargis et al., 1984). Weeks and Warinner (1984) found that the phagocytic efficiency of macrophages from spot (Leiostomus xanthurus) and hogchokers (Trinectes maculatus) resident in the Elizabeth River was reduced when compared to fish from control locations.

The bioavailability of PAHs in the river is demonstrated in Figure 13, which shows the residue levels observed in oysters transplanted along the river system (Huggett et al., 1986). During this transplant study we also observed depressed lipid levels in oysters which were exposed at the more contaminated stations (Figure 14).

PAHs are widespread contaminants of freshwater and estuarine systems and have been implicated in causing effects on fishes and shellfish in other areas [Black (1983), the Niagara River; Mix (1984), Oregon Bays; and Malins et al. (1985), Puget Sound].

A recent survey of PAH contamination in Virginia's major river systems (see Figure 15 for station locations) indicates higher residues in shellfish collected from estuaries draining the more industrialized or populated basins (Bender, 1985). Figure 16 shows the residues of total resolved PAHs in oysters (Crassostrea virginica) and brackish water clams (Rangia cuneata) along the James, York, and Rappahannock rivers in the fall of 1984. In the James River residues of total PAHs in oysters declined with increasing distance from the river mouth while residues in clams, although generally higher than oysters, showed no distinct trends with distance. Residues in Rangia collected from the Chickahominy River (an undeveloped tributary of the James) were considerably lower than those in Rangia from the James River stations.

In the York River concentrations of total aromatics increased dramatically at the most upstream oyster rock sampled, and clams collected from just below West Point had the highest residues observed during the survey (Figure 16). A detailed examination of the clam samples from the York, Pamunkey and Mattaponi rivers indicated that compounds derived from resin acids of plants accounted for a significant proportion of the resolved aromatics in these samples. The concentrations of the "resin acid derived compounds" in the York, Pamunkey and Mattaponi rivers are shown in Figure 17.

Concentrations of hydrocarbons in the unresolved envelopes (mixtures of degraded aromatic hydrocarbons) from these samples are shown in Figure 18. Oysters and clams collected from the Rappahannock showed no evidence of unresolved envelopes (UCMs). In both the York and James rivers substantial increases in the UCM were observed in both oysters and clams collected near the turbidity maximum zone. The lack of a UCM in the Rappahannock samples and the relatively low concentration observed in the Chickahominy samples suggest anthropogenic origins for the envelopes.

At present we have no conclusive evidence to indicate that shellfish populations which show higher PAH residues are adversely affected. It should be noted that Rangia populations in the upper York and the lower Mattaponi and Pamunkey rivers are very small compared to those in the James and Rappahannock rivers. In addition, clams from these areas generally appear to be in poor condition, i.e. they have lower dry weight to wet weight ratios than clams from other river systems.

Organotins

The recent proposal by the U.S. Navy to utilize tributyltin (TBT) based paints as antifouling coatings on all Navy vessels has prompted concern over the potential environmental effects of TBT in the Chesapeake Bay (Fed. Reg., 1985). TBT-based paints have been used on pleasure craft in Europe and the United States for about 15 years.

Recent studies in England and France, summarized by Stebbing (1985), have implicated tributyltin in causing decreased spatfall, decreased growth and shell malformations in oysters (Crassostrea gigas). Thain and Waldock (1983) showed that very low concentrations of tributyltin oxide (TBTO), 0.15 μg/ℓ, inhibited the growth of young oysters (C. gigas). Beaumont and Budd (1984) reported about 50% mortality of mussel larvae (Mytilus edulis) after 15 days exposure to TBTO concentrations of 0.1 μg/ℓ. For adult mussels of the same species, 96 hr LC_{50} values of 20-60 μg/ℓ have been reported (U.S. Navy, 1983). Thain (1983) found a 48 hr LC_{50} of 1800 μg/ℓ (TBTO) for adult oysters (C. gigas) and an LC_{50} of only 1.6 μg/ℓ for their larvae. Laughlin, et. al. (1984) showed a significant reduction in the survival of Gammarus oceanicus larvae at 0.3 μg/ℓ of TBTO, although adults were not affected at this level.

Recent studies have found TBT concentrations in water near marinas in the lower Bay region as high as 68 ng/ℓ, as Sn [~.15 μg/ℓ as bis(tributyltin)oxide] (Unger and Huggett, personal communication). Compared to bioassay data, e.g. with mussel larvae, such levels are within the acutely toxic range.

Preliminary data indicate that TBT is toxic to fertilized embryos of the Eastern oyster (Crassostrea virginica) at 0.7 μg/ℓ (LC_{50}) in 48 hours (Roberts, personal communication). Further tests are necessary to determine the acute toxicity to oyster larvae at various developmental stages. In addition, bioassays on other non-target species are needed, e.g. the hard clam (Mercenaria mercenaria), the bay scallop (Aequipecten irradians), and others.

Kepone

Restrictions on the commercial harvest of some species of finfish are still in effect in the James River, 10 years after the discovery that the pesticide Kepone (decachloroctahydro-1, 3, 4-metheno-2H-cyclobuta (cd) pentalin-2-one) had contaminated the estuary. Kepone was produced in the City of Hopewell by two firms between 1967 and 1975. The pesticide entered the tidal James River through a variety of routes including chemical plant discharges, runoff from contaminated land fills and sewage effluents.

Bender and Huggett (1984) reviewed the data available through 1982 on the status of Kepone contamination in the estuary, and this discussion is based primarily on that review.

Bottom sediments of the river are contaminated from the source at Hopewell to near the river mouth. Figure 19 shows the mass of Kepone estimated to be present in the upper 32 cm of bed sediments as a function of time and location. As can be seen from the figure, the rate of burial or rate of dilution, i.e. the slope, is greatest in the turbidity maximum zone, followed by the upper estuary, and is considerably lower in the lower estuary. Since these bed sediments serve as the source of Kepone available to organisms, the rate at which burial or dilution occurs is extremely important in determining exposure levels for aquatic animals.

Figure 20 shows the relationship between the Kepone residues observed in 4 species of animals as a function of the change in mass of Kepone in river sediments over time.

Figure 21 depicts the change in third quarter (July-September) residues for croaker and spot from the lower river as a function of time (1976-1985). Residues for both species declined through 1980 but then increased in 1981. Since 1981 residues have again declined but at a much slower rate.

After the Kepone contamination of the James River was discovered in 1975, numerous studies were conducted to estimate its impact on the biota of the river. The majority of these investigations to establish effects levels were conducted by researchers at the U.S. Environmental Protection Agency laboratory in Gulf Breeze, Florida, and by the staff and students of the Virginia Institute of Marine Science. Space precludes a detailed discussion of each of their findings. For more information, the reader is referred to the original references cited in Bender and Huggett (1984).

Acute, partial chronic and chronic toxicity tests have all been utilized to estimate the effects of Kepone on aquatic life. In some cases the bioassays established no effects levels, i.e. an exposure level at which no significant difference in growth, reproduction, etc. was observed versus a control group. Other studies estimated the maximum acceptable toxicant concentration (MATC) by use of the application factor. The application factor is defined as the ratio of the MATC to the 96 hour LC_{50}.

Figure 22 compares the measured no effect level for the animals indicated to the levels of Kepone found in the river. As can be seen from the figure, exposure levels are well below these no effect levels. Figure 23 shows the MATC's for fishes using an application factor of 0.001, the most conservative estimate of an application factor (Goodman et al., 1982).

In summary, these laboratory bioassays have shown that Kepone can produce acute and chronic effects on marine and freshwater animals. However, the concentrations necessary to cause effects are considerably greater than those found in the river. If these conclusions about effects on the biota are correct, then the major impact of Kepone contamination in the river is through economic losses due to the restrictions on fishing. Quantitative estimates of the magnitude of the economic impacts are not available. Many commercial fishermen along the river participated in legal actions for damages against the manufacturing firms, claims which were settled out of court.

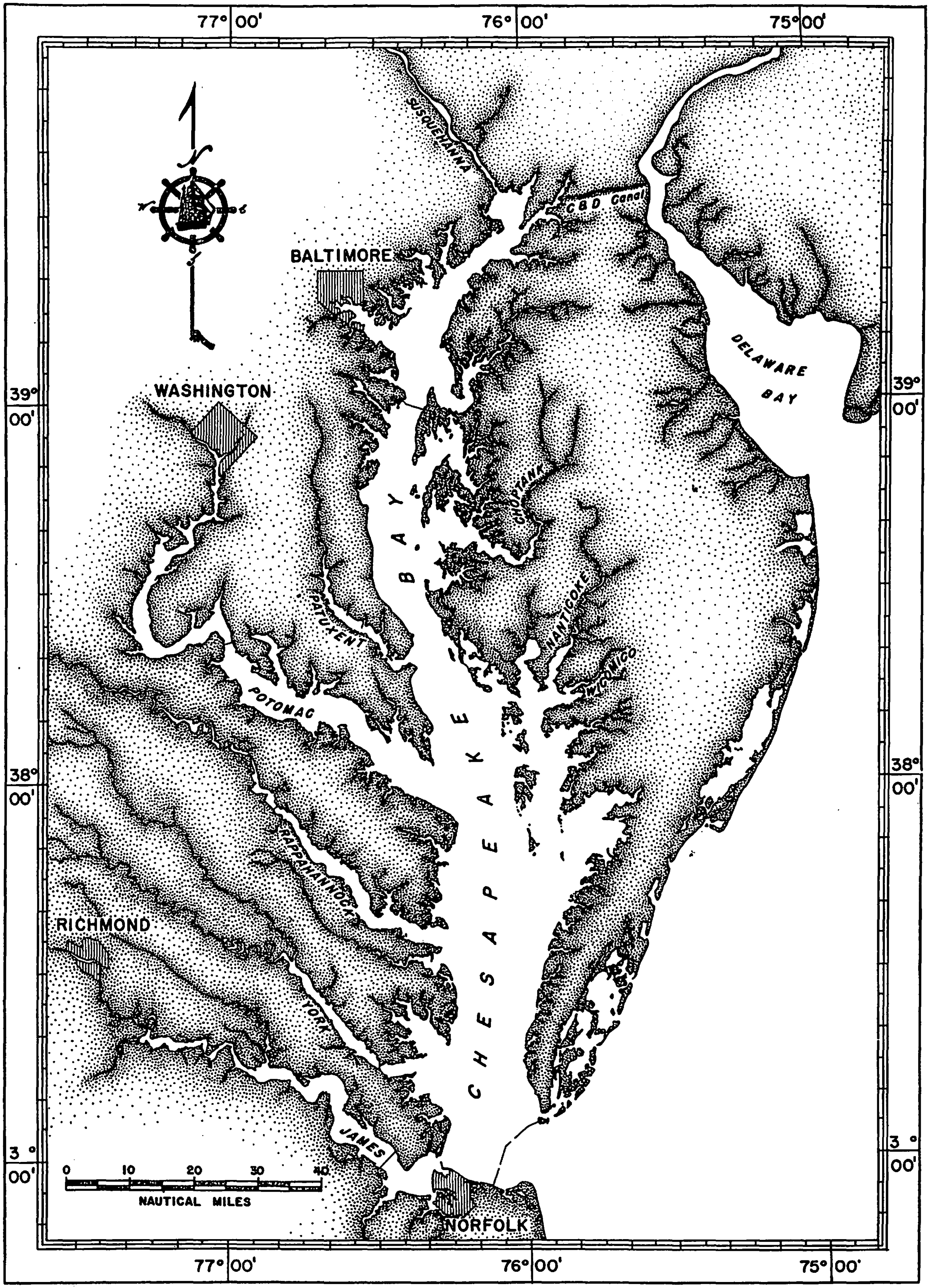
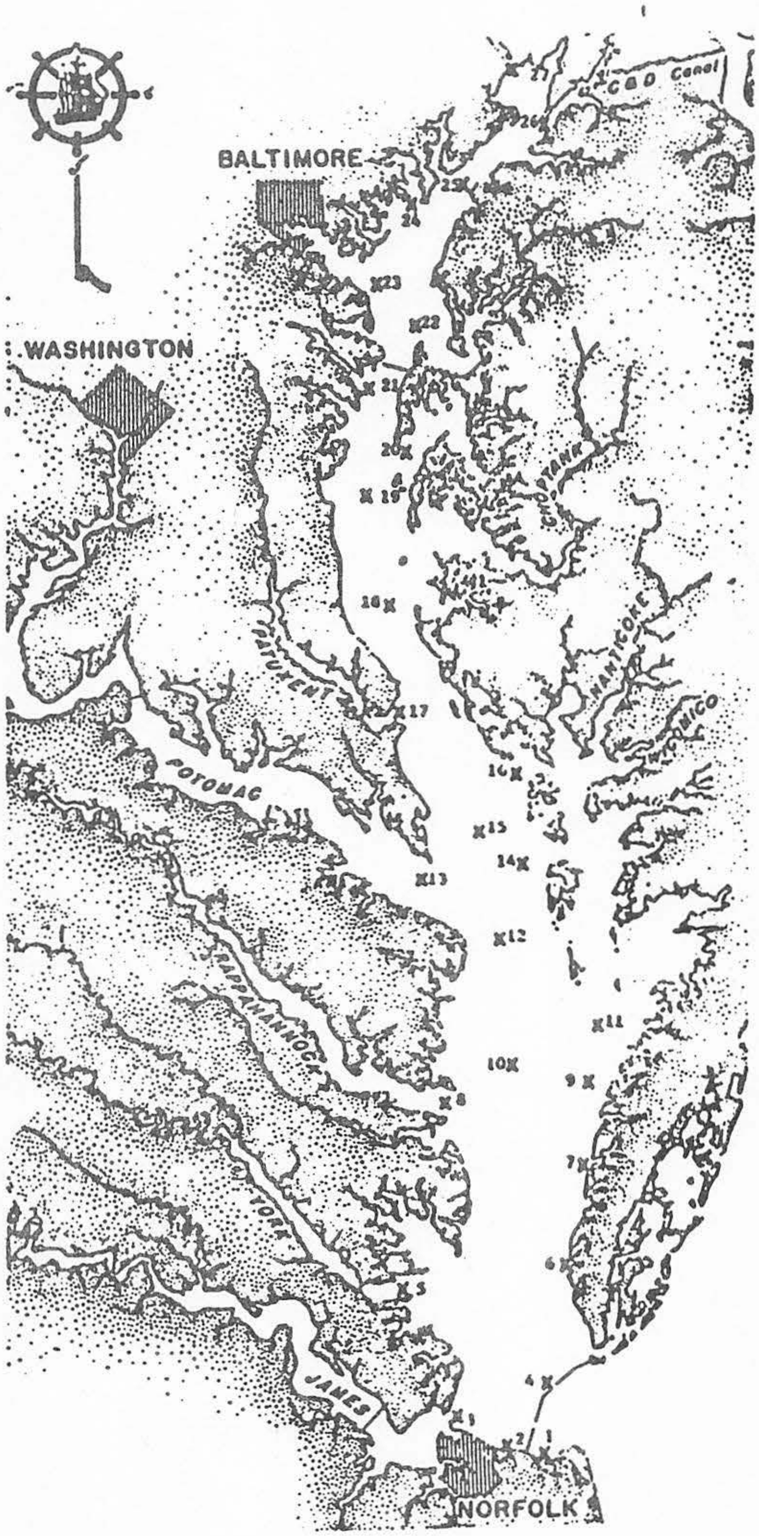


Figure 1. Chesapeake Bay

Figure 2



Chesapeake Bay Sediment Sample Locations

(Bieri, et al., 1982a)

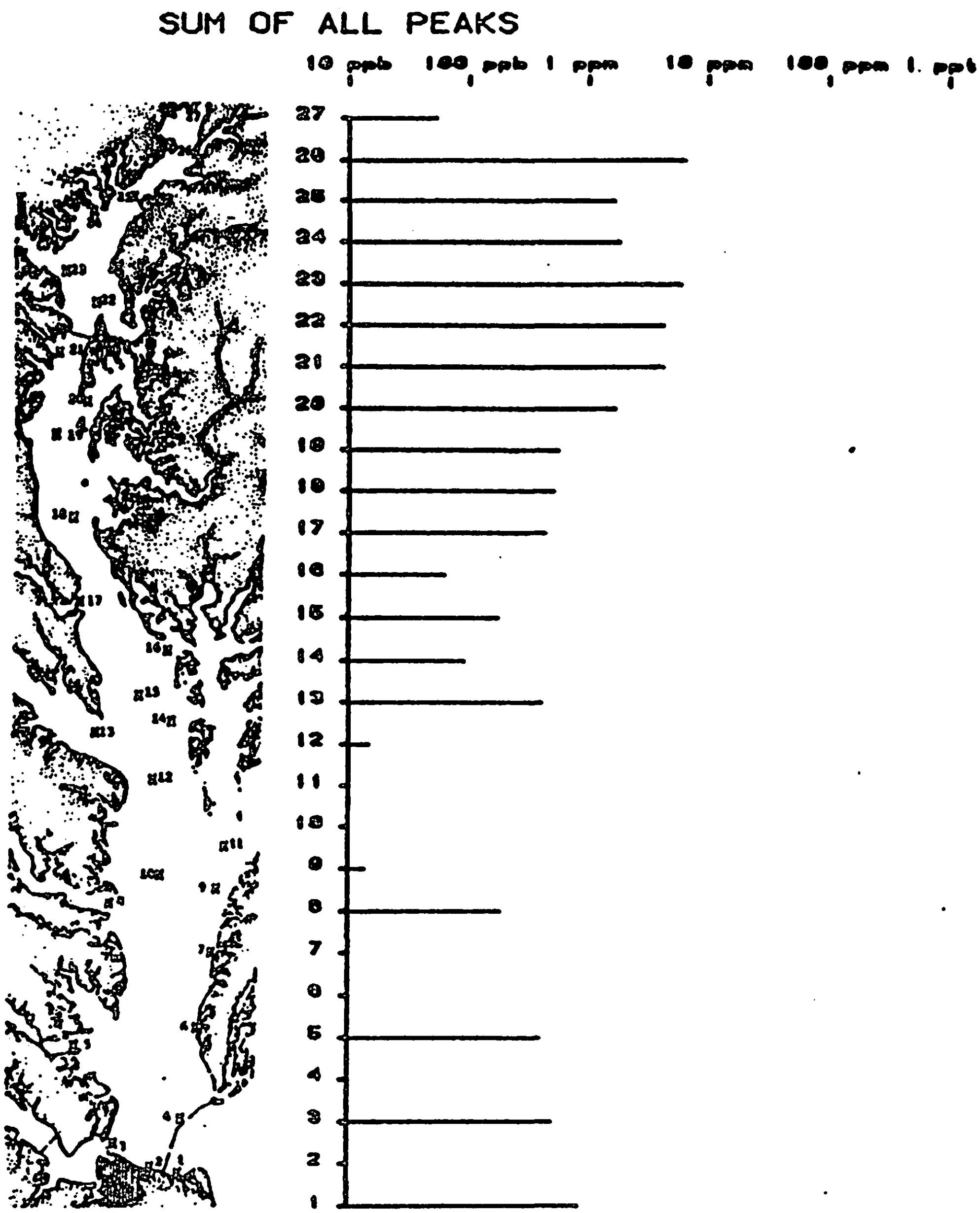


Figure 3 Distribution of organic compounds in surface sediments of the Chesapeake Bay, Spring 1979. The height of individual bars represents the logarithm of the concentration sum of all resolvable peaks. The sample locations are found by locating the numbers to the left of the bars in the map.
(Bieri, et al., 1982a)

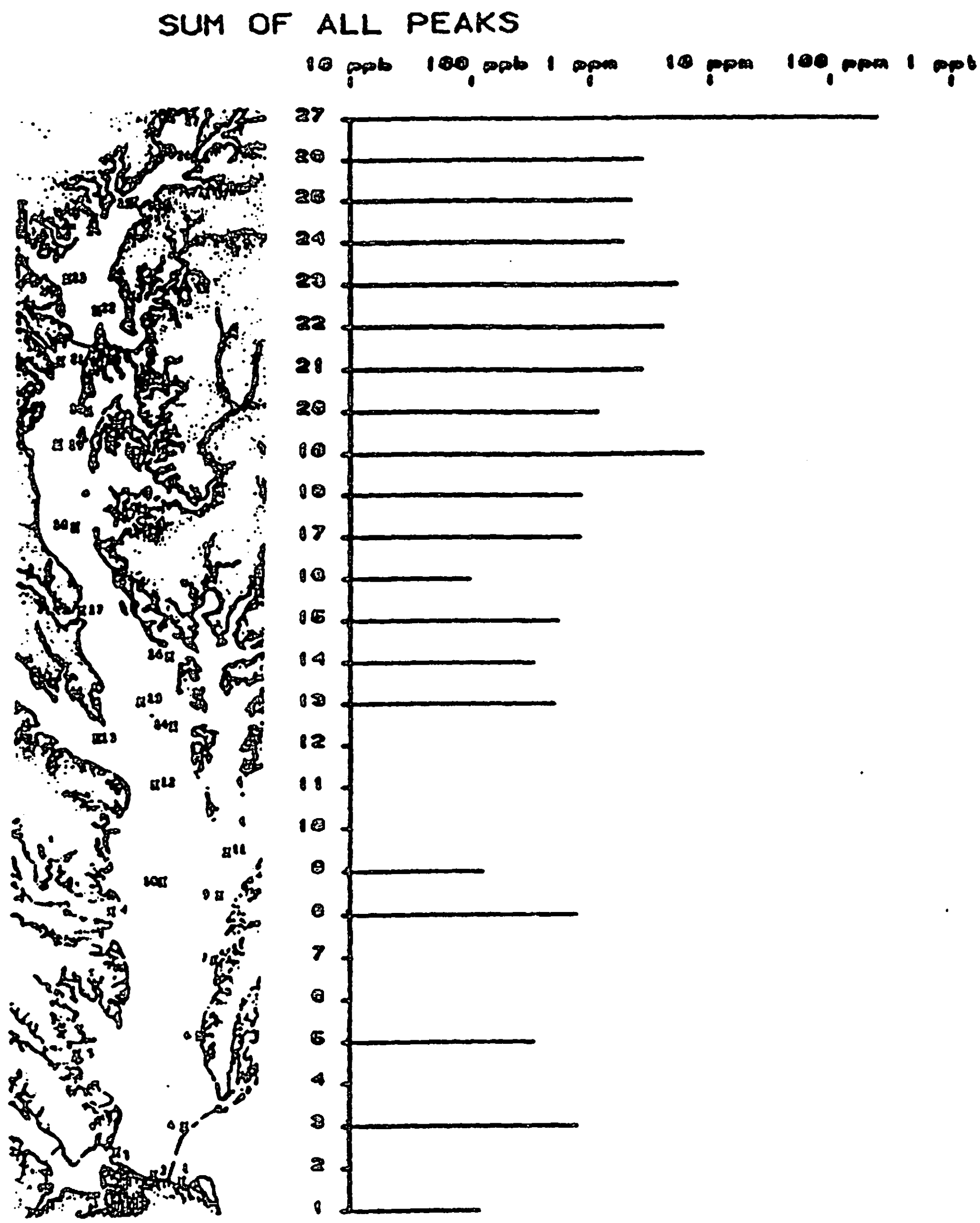


Figure 4 **Distribution of organic compounds in surface sediments of the Chesapeake Bay, Fall 1979. The height of individual bars represents the logarithm of the concentration sum of all resolvable peaks.** (Bieri, et al., 1982a)

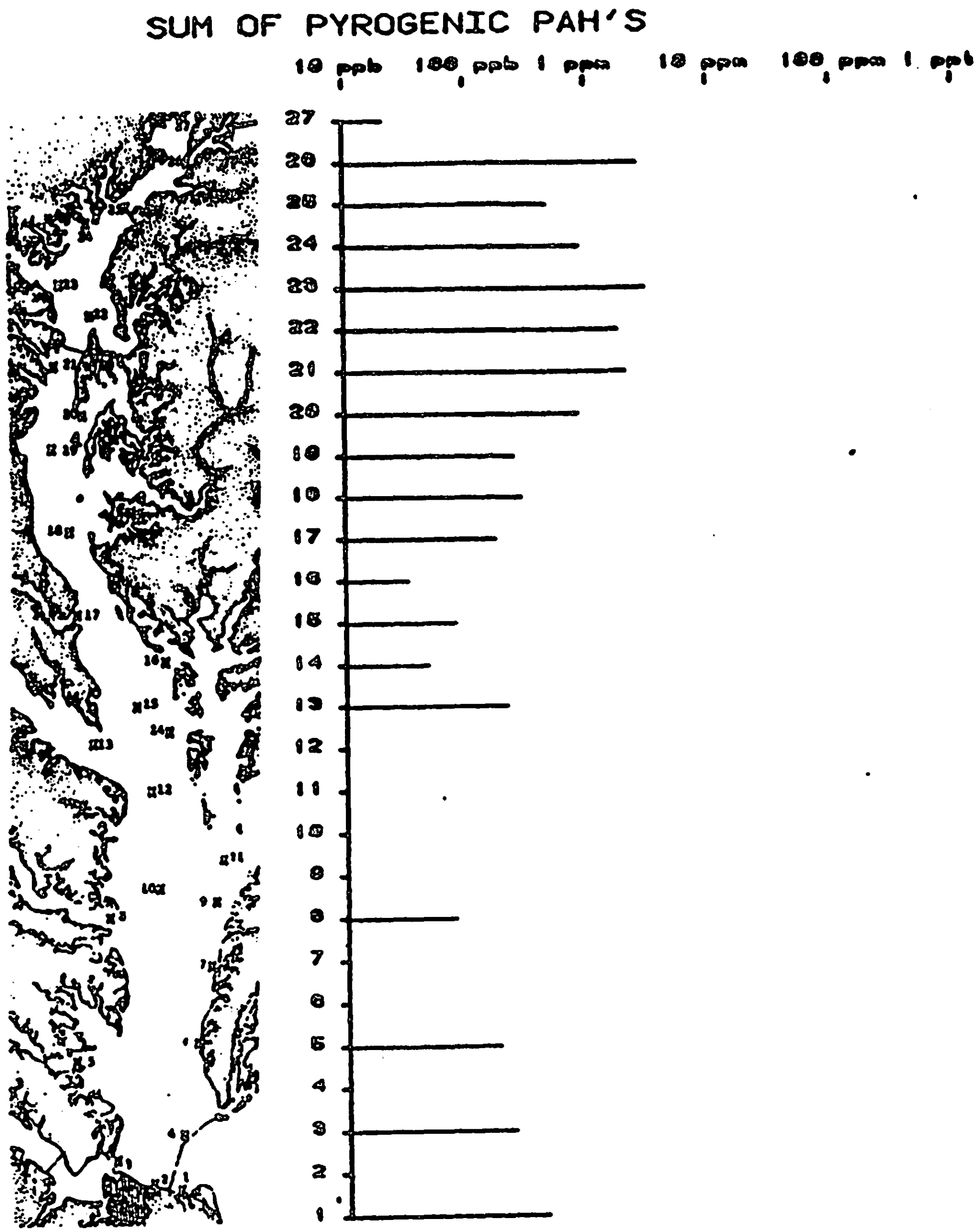


Figure 5 Concentration sums of polynuclear aromatic hydrocarbons in surface sediments, related to the combustion of carbonaceous matter, Spring 1979. (Bieri et al., 1982a)

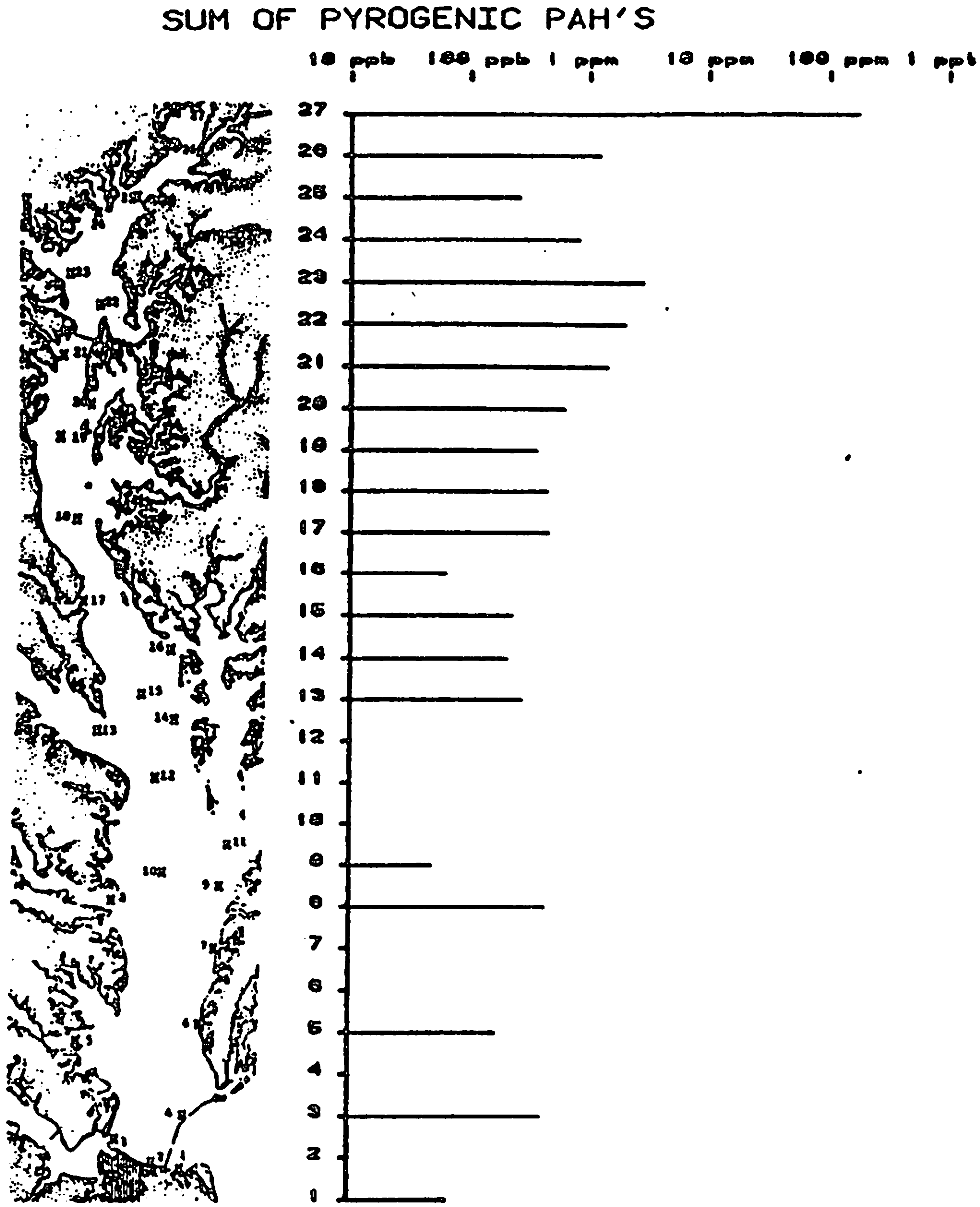
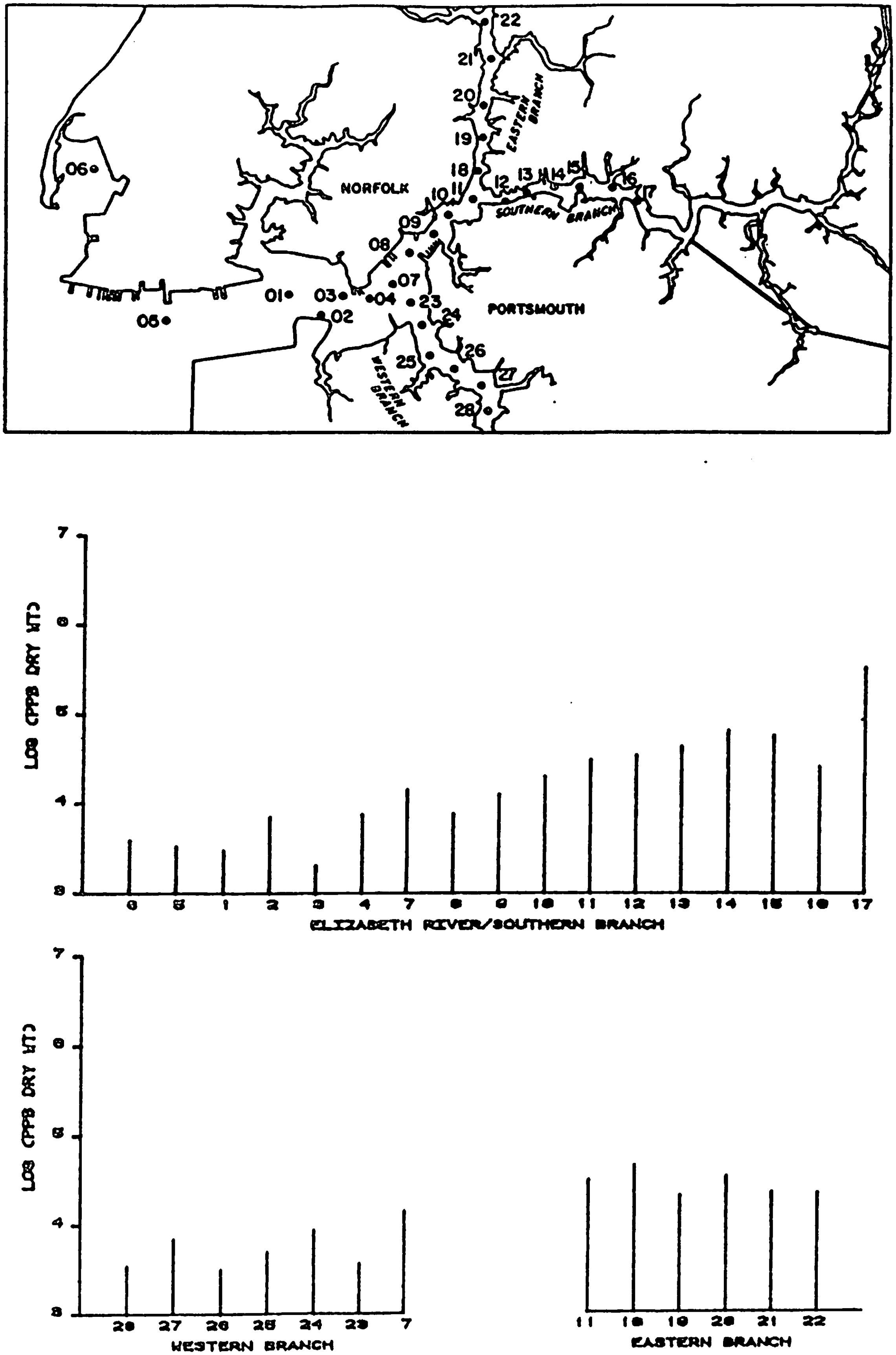


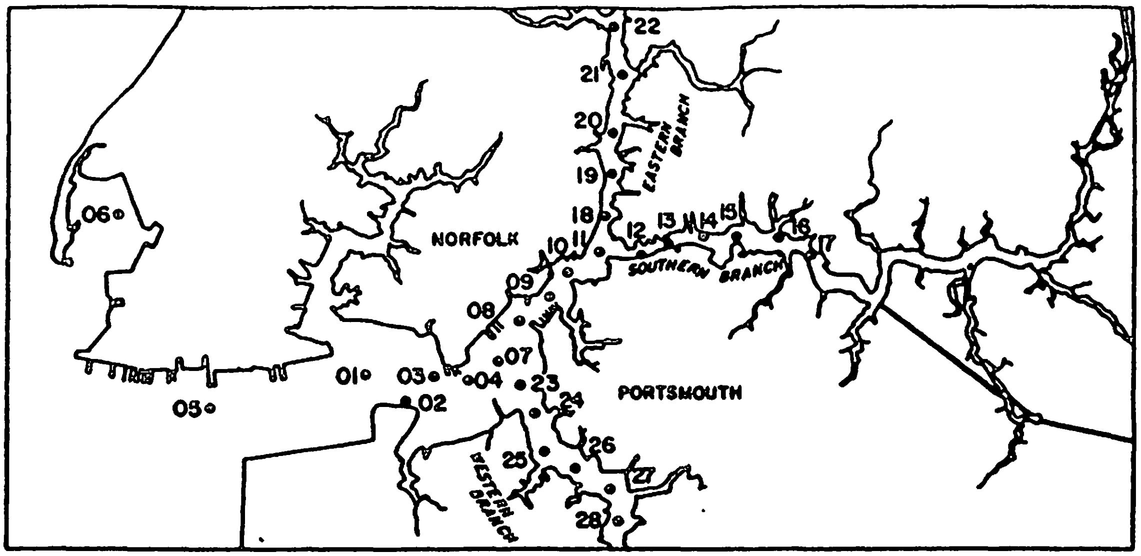
Figure 6 Concentration sums of polynuclear aromatic hydrocarbons in surface sediments, related to the combustion of carbonaceous matter, Fall 1979. (Bieri et al., 1982a)

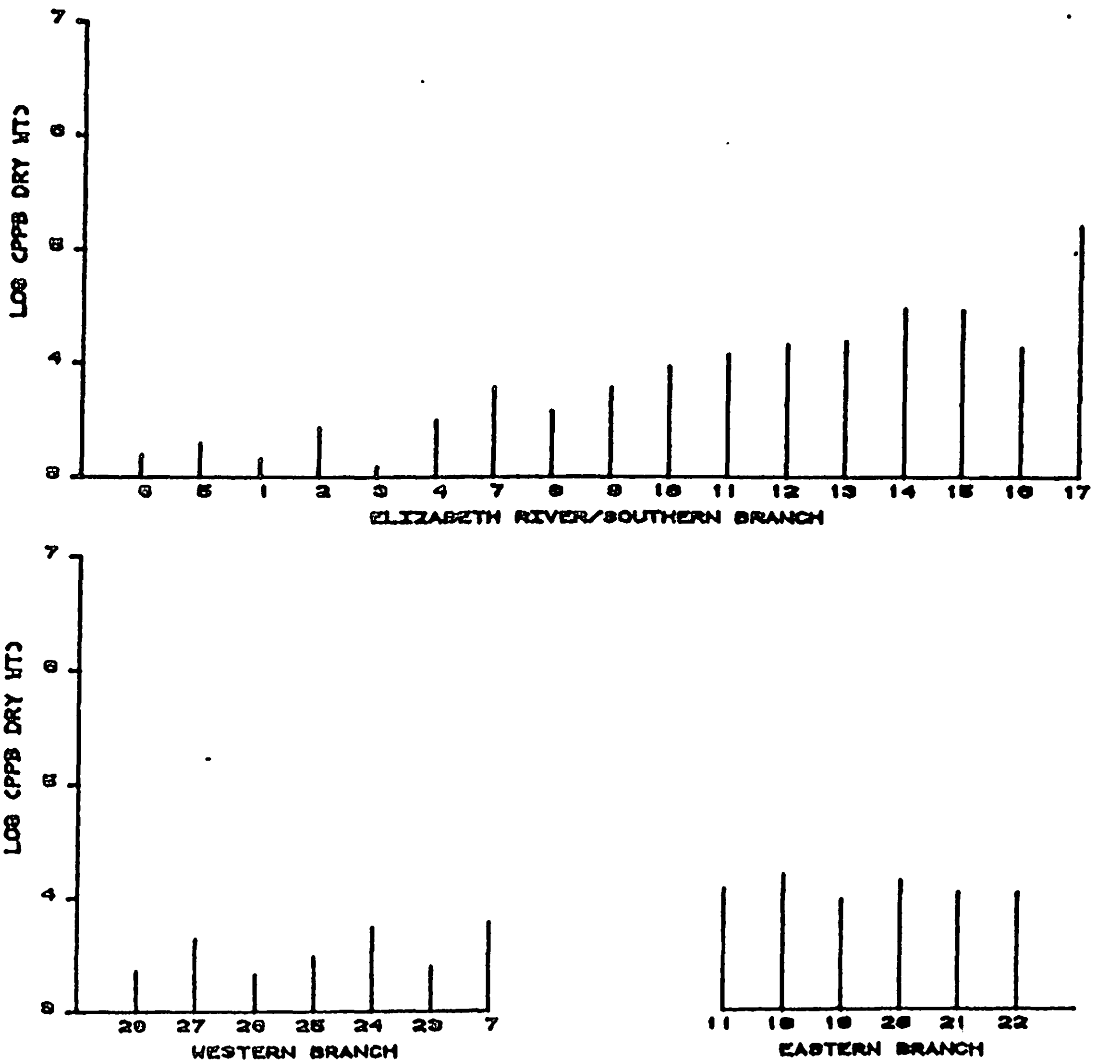
Figure 7



TOTAL RESOLVED PEAKS
(Bieri, et al., 1982b)

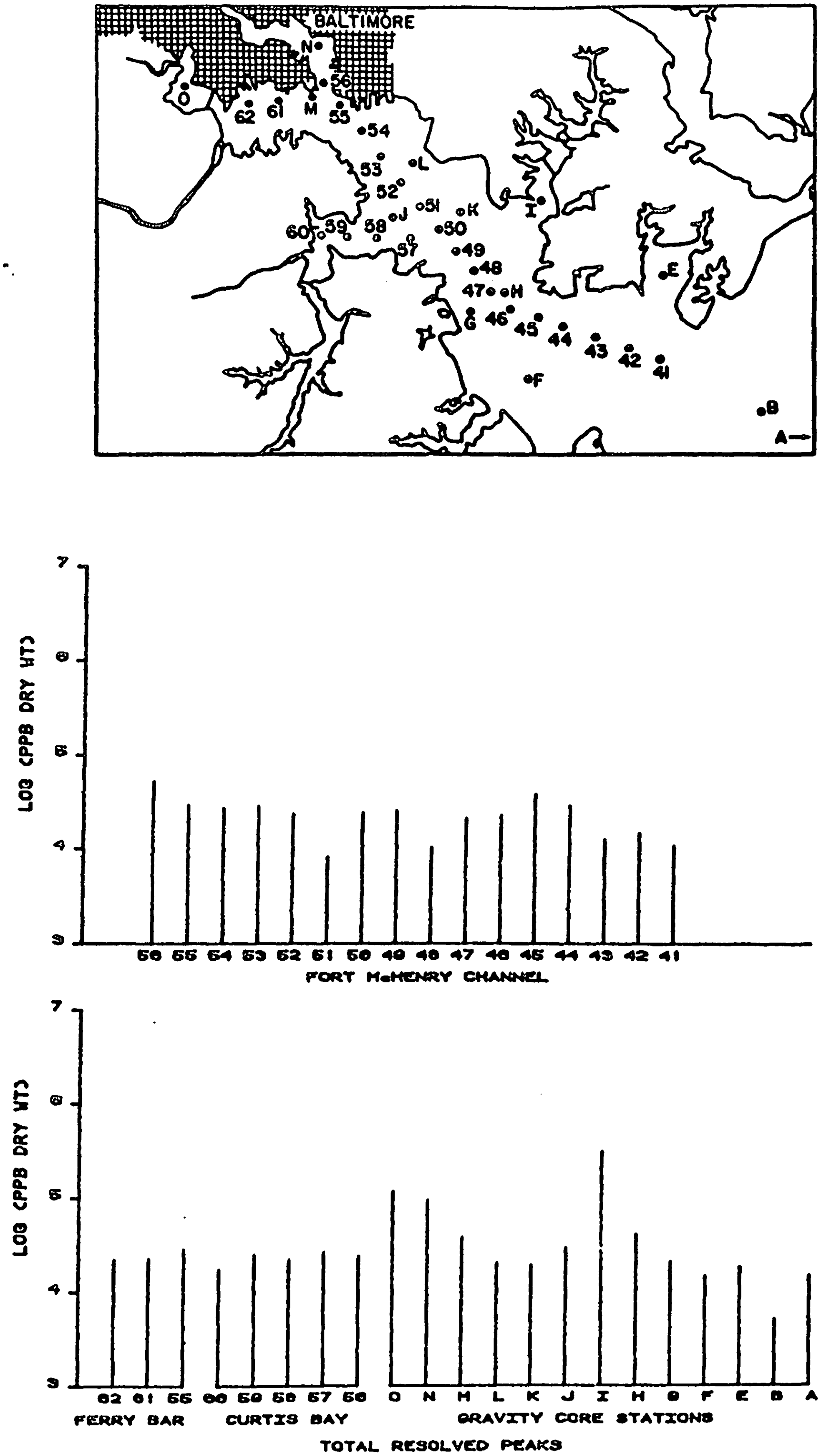
Figure 8





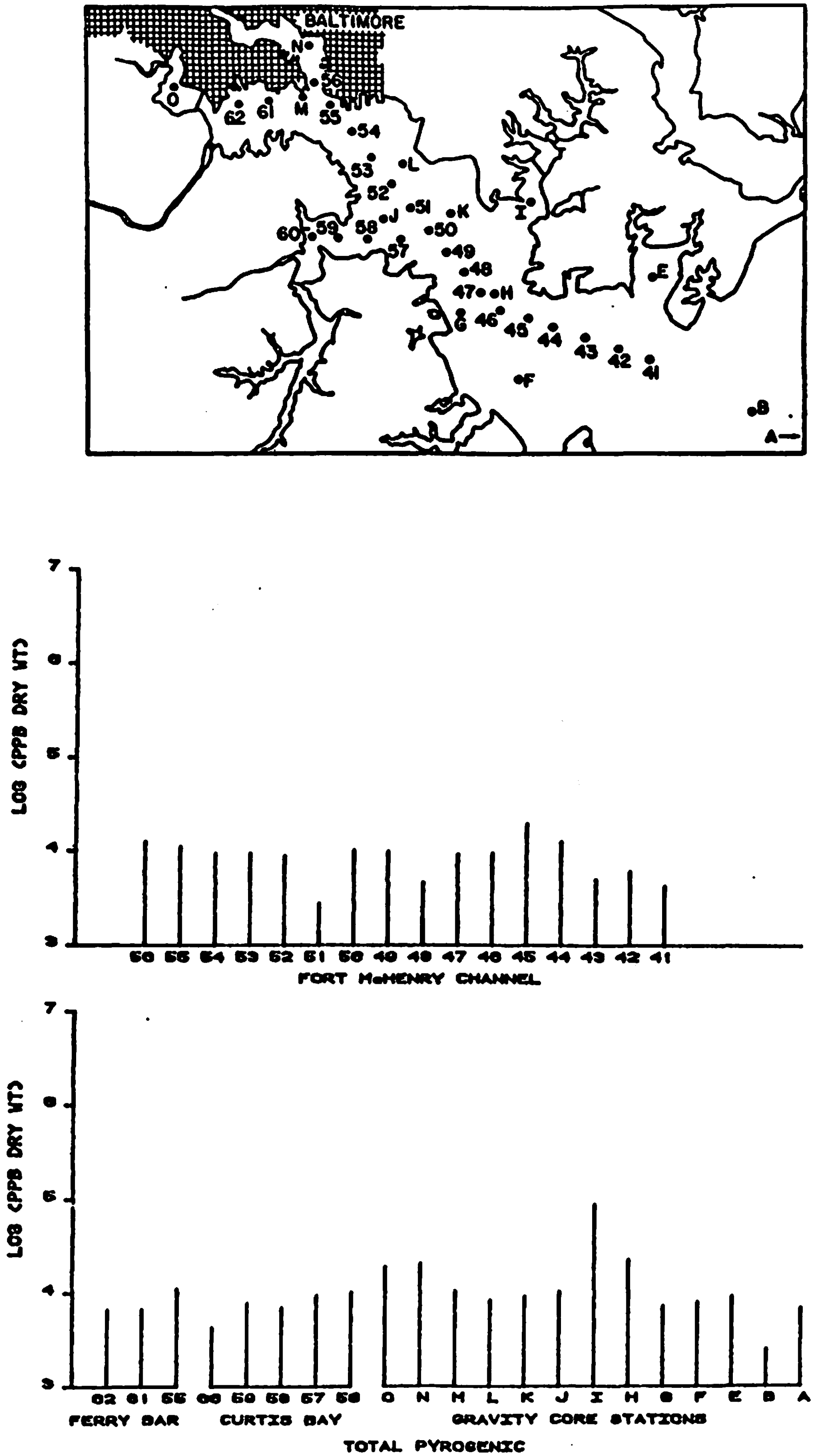
TOTAL PYROGENIC PEAKS
(Bieri et al., 1982b)

Figure 9



(Bieri, et al., 1982b)

Figure 10



(Bieri, et al., 1982b)

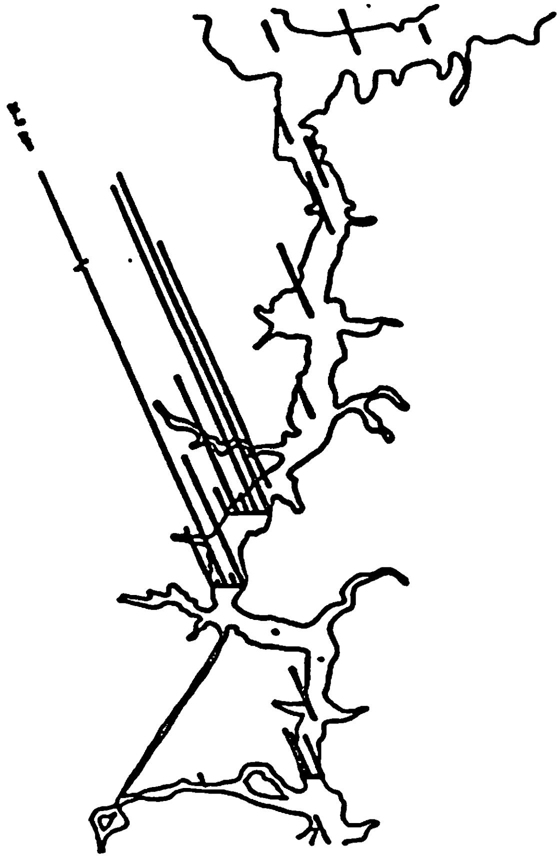


Figure 11 Surface sediment concentrations (mg/kg-dry wt) of benzo(a)pyrene along the Elizabeth River [from Huggett, et al. 1985] 0.5 cm = 1 ppm

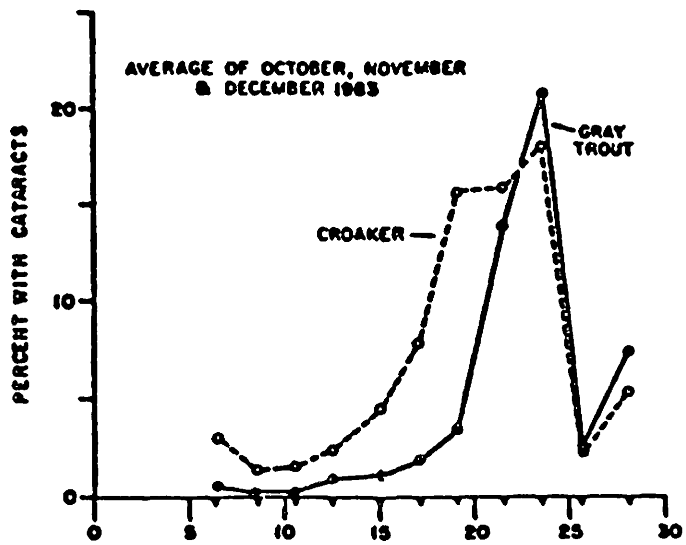


Figure 12 Average occurrence of cataracts in croaker and gray trout from stations along the Elizabeth River [from Huggett, et al. 1985]

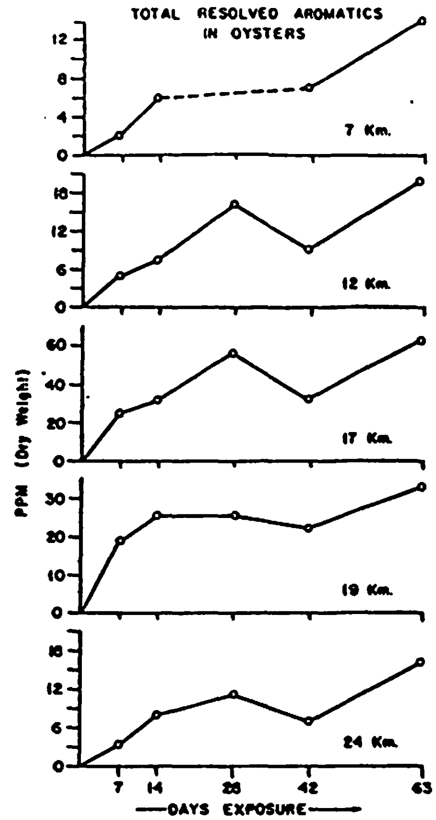


Figure 13 Total resolved aromatic hydrocarbons in oysters along the Elizabeth River (Huggett, et al., 1986 in Press)

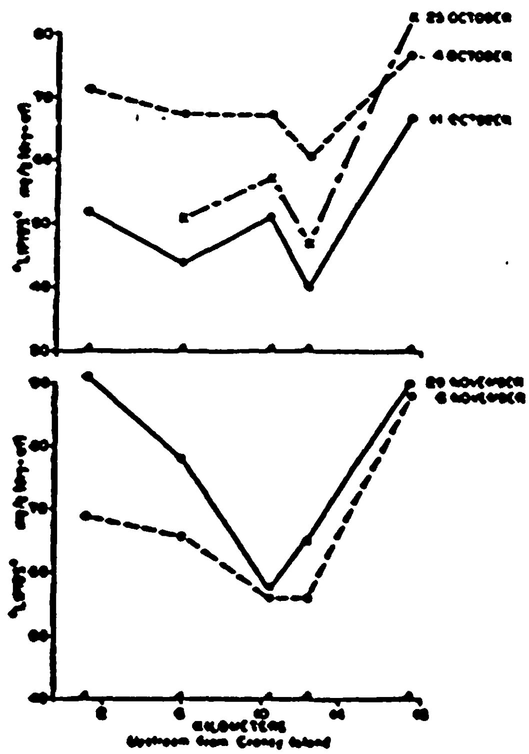


Figure 14 Lipid levels in oysters along the Elizabeth River (Bender and Huggett, 1986)

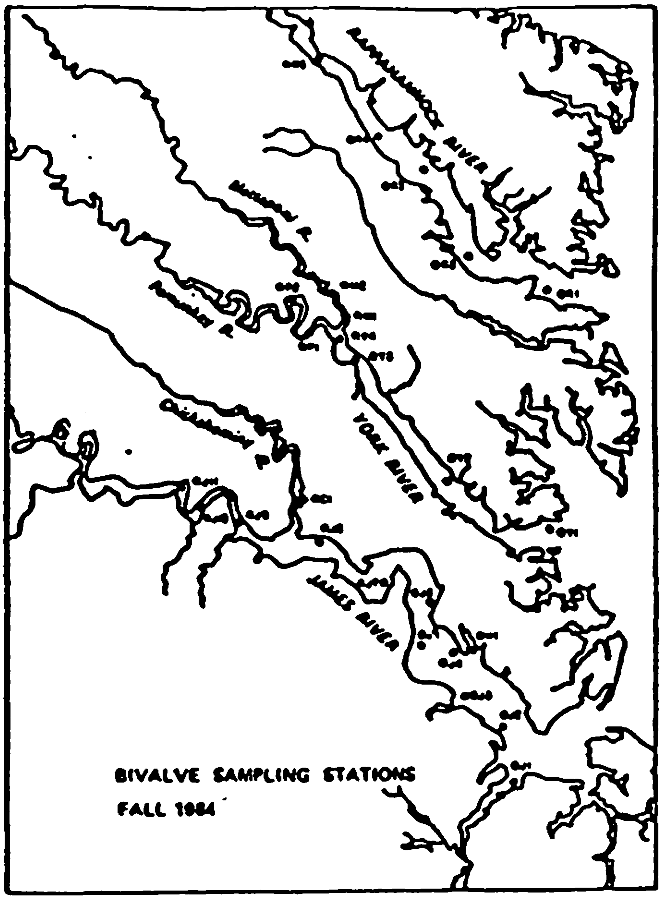


Figure 15 Bivalve sampling stations for PAHs [from Bender 1985]

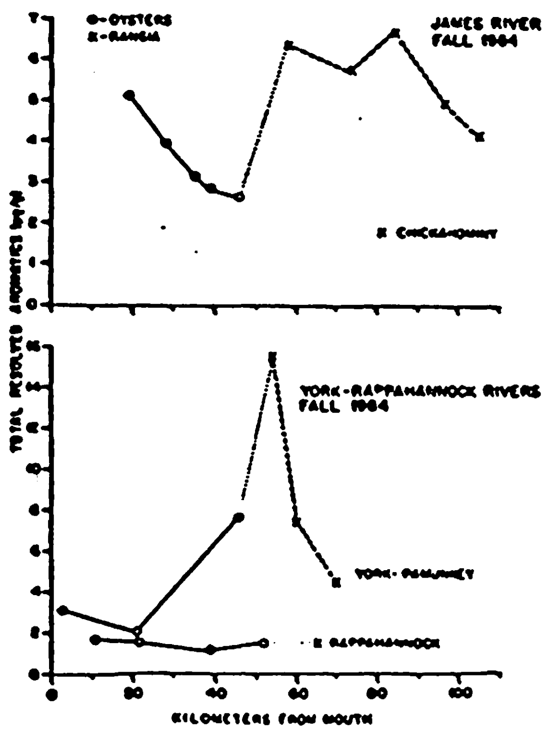


Figure 16 Total resolved aromatics in oysters and Rangia [from Bender 1985]

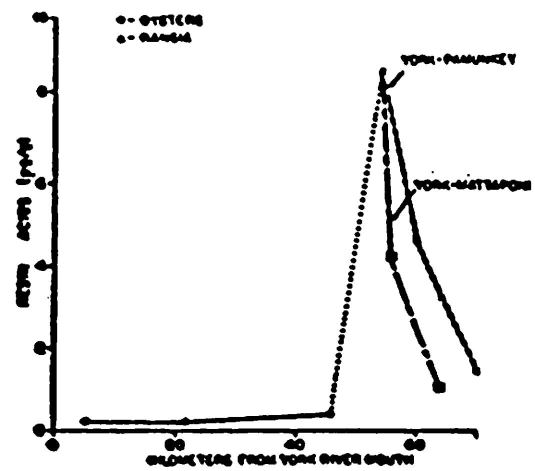


Figure 17 Resin acids in oysters and Rangia [from Bender 1985]

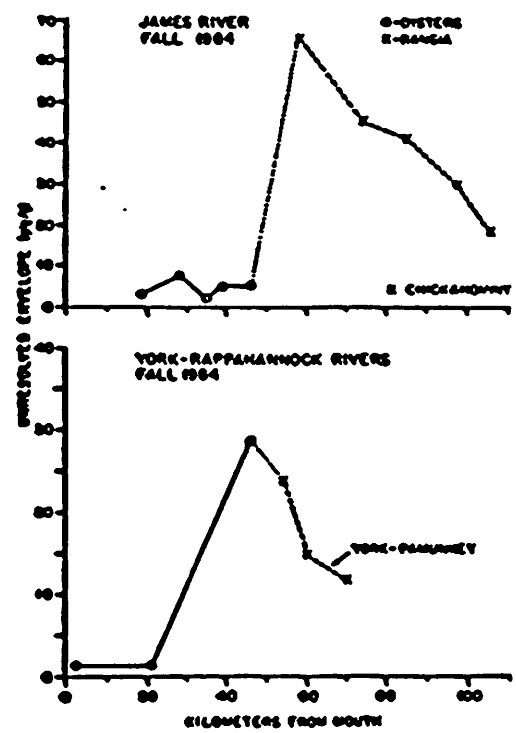


Figure 18 Unresolved complex mixture in oysters and Rangia [from Bender 1985]

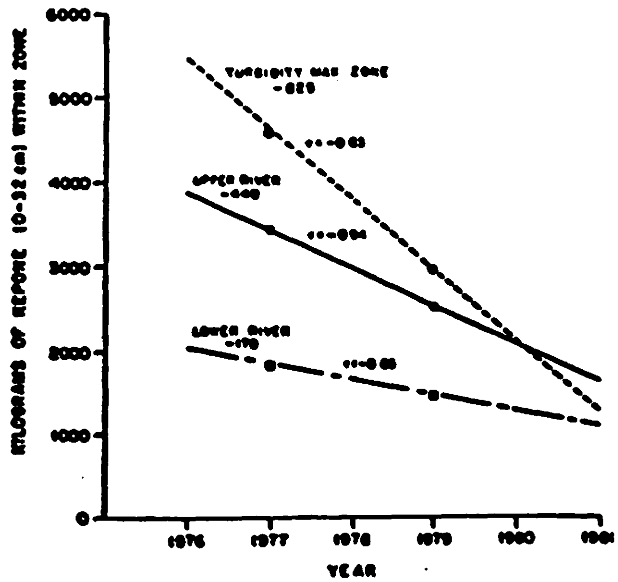


Figure 19 Kepone in James River sediments, calculated regression lines from three zones [from Bender and Huggett 1984]

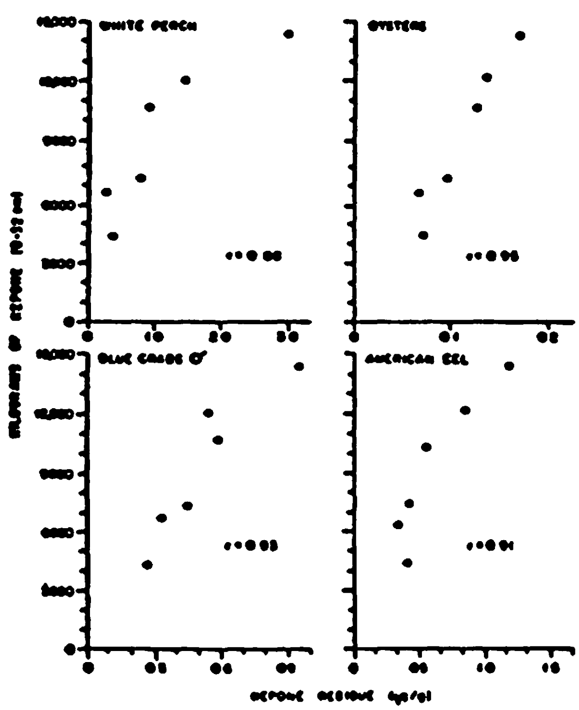


Figure 20 Kepone in James River sediments (0-32 cm) 1976-1981 vs Kepone residues in 4 species of animals [from Bender and Huggett 1984]

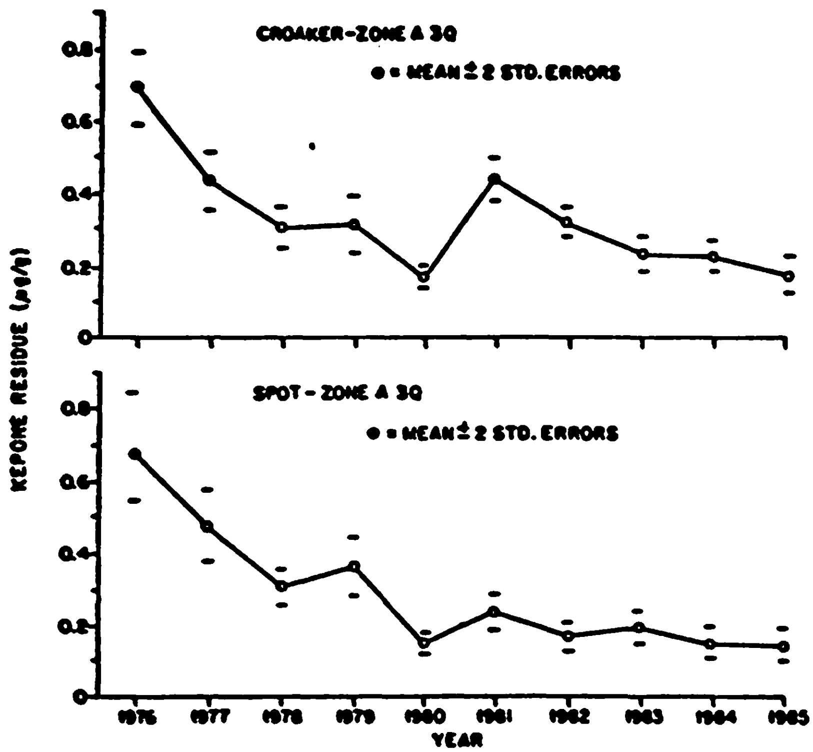


Figure 21 **Kepone residues in croaker and spot 1976-1985** (Bender and Huggett, 1986)

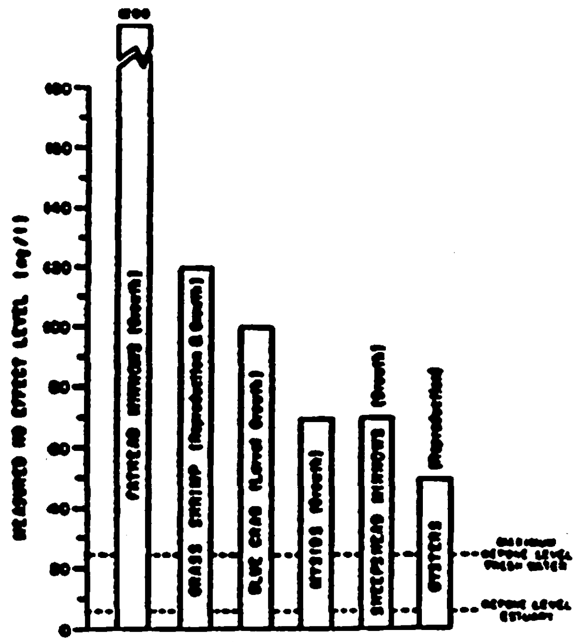


Figure 22   Kepone - measured no effect levels [from Bender & Huggett 1984]

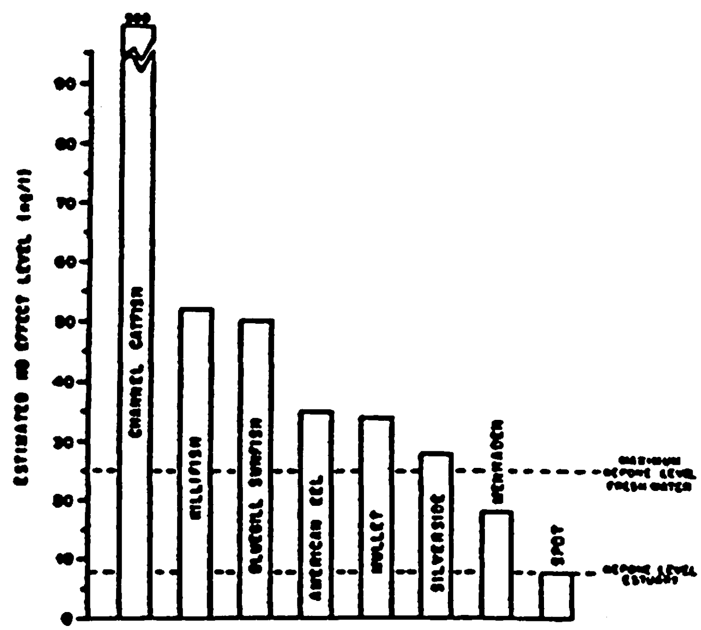


Figure 23   Kepone - estimated no effect levels [from Bender & Huggett 1984]

Table 1

Sediment PAH Concentrations
ppm - dry weight

River	Station	Total Resolved	Natural	Anthropogenic
James	1	1.36	0.06	1.30
James	2	1.04	0.12	0.92
James	3	6.22	0.93	5.29
James	4	7.12	1.45	5.67
James	5	13.90	1.26	12.64
James	6	5.69	0.55	5.14
James	7	7.25	1.05	6.20
James	8	5.02	0.57	4.67
James	9	6.36	0.38	5.98
York	1	0.97	0.07	0.90
York	2	1.03	0.09	0.94
York	3	0.98	0.14	0.84
Pamunkey	1	14.22	12.25	1.97
Pamunkey	2	1.75	0.49	1.26
Mattaponi	1	1.19	0.25	0.94
Mattaponi	2	0.57	0.26	0.31
Rappahannock	1	0.09	0.01	0.08
Rappahannock	2	0.62	0.03	0.59
Rappahannock	3	0.94	0.26	0.68
Rappahannock	4	0.67	0.04	0.63
Rappahannock	5	0.82	0.09	0.73
Rappahannock	6	0.89	0.20	0.69
Rappahannock	7	1.28	0.45	0.83
Rappahannock	8	5.11	2.12	2.99
Rappahannock	9	0.32	0.12	0.20

(Bender, et al., 1986)

Table 2 Percentage of fish showing gross abnormalities from exposure to contaminants in the Elizabeth River. Data are means of three samples taken October, November, and December, 1983*

	Kilometers from the Mouth										
	6.5	8.5	10.5	12.5	15.0	17	19	21.5	23.5	25.5	28
Fin Erosion											
Hogchoker (1)	0.7	0	0	0.4	1.4	5.5	4.3	11.2	1.9	0	0.5
Toadfish (2)	0	0	11.0	5.0	0	11.5	30.1	26.3	25.0	0	0
Cataracts											
Spot (3)	0	0	0.1	0	3.0	0.8	9.6	6.0	0.2	0.3	0
Gray Trout (4)	0.2	0	0	0.8	1.0	1.8	3.5	14.0	21.0	2.5	7.5
Croaker (5)	3.3	1.4	1.5	2.2	4.5	7.9	15.8	15.9	18.1	2.5	5.6

(1) *Trinectes maculatus*
(2) *Opsanus tau*
(3) *Leiostomus xanthurus*
(4) *Cynoscion regalis*
(5) *Micropongonias undulatus*

* from Huggett, et al., 1985

CHESAPEAKE BAY
AND TRIBUTARIES

BEAUMONT, A. B. AND M. D. BUDD. 1984. "HIGH MORTALITY OF THE LARVAE OF THE COMMON MUSSEL AT LOW CONCENTRATIONS OF TRIBUTYL TIN" MARINE POLLUTION BULLETIN 15: 402 - 405. (ACUTE TOXICITY OF TBT TO MUSSEL LARVAE)

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DELAWARE BAY

Delaware Bay is a horn shaped, well mixed estuary with no density currents and very small or no salt water wedges (Gross, personal communication, November, 1985).

The Delaware River flows into the bay. Four large cities located along the tidal Delaware River Estuary are Trenton, New Jersey; Philadelphia, Pennsylvania; Camden, New Jersey; and Wilmington, Delaware. The combination of these cities forms our nation's fourth largest metropolitan area and is the site of the second largest oil-refining petrochemical center in the nation. Delaware River is one of the most heavily used in the nation for recreation. Delaware Bay "serves as a major shipping lane from the Atlantic Ocean to port facilities in the Philadelphia-Camden Area" (Delaware River Basin Commission staff, 1984).

Since no major industries feed directly into Delaware Bay, tributaries are usually monitored and Delaware Bay information is scarce (Otto, personal communication, November, 1985). Otto described priorities for EPA's sampling at some area stations in a national water quality monitoring program. They primarily analyzed shellfish, secondly finfish, and in the absence of either of those, they analyzed sediments. Otto reported that EPA data showed very low levels of contaminants in the fish.

Water Quality

The Delaware River Basin Commission (1984) reported that the middle portion of the Delaware Estuary in the Philadelphia-Camden area had the worst water quality of the entire area they assessed. The parameters they assessed included dissolved oxygen, pH, phenols, fecal coliforms, alkalinity, chlorides, and sodium. The commission declared that 9% of the river miles did not meet designated water uses specified in the water quality standards of the commission and of the four states, Delaware, New Jersey, New York, and Pennsylvania, through which the Delaware River flows. Using data from water evaluated in 1982 and 1983, the Delaware River Basin Commission judged water in the 29 river miles passing through the Philadelphia-Camden area not "fishable" or "swimmable" and not of adequate quality to support maintenance of resident fish, passage of anadromous fish or secondary contact recreation. They considered water quality in the area acceptable for public, industrial, and agricultural supply and for navigation.

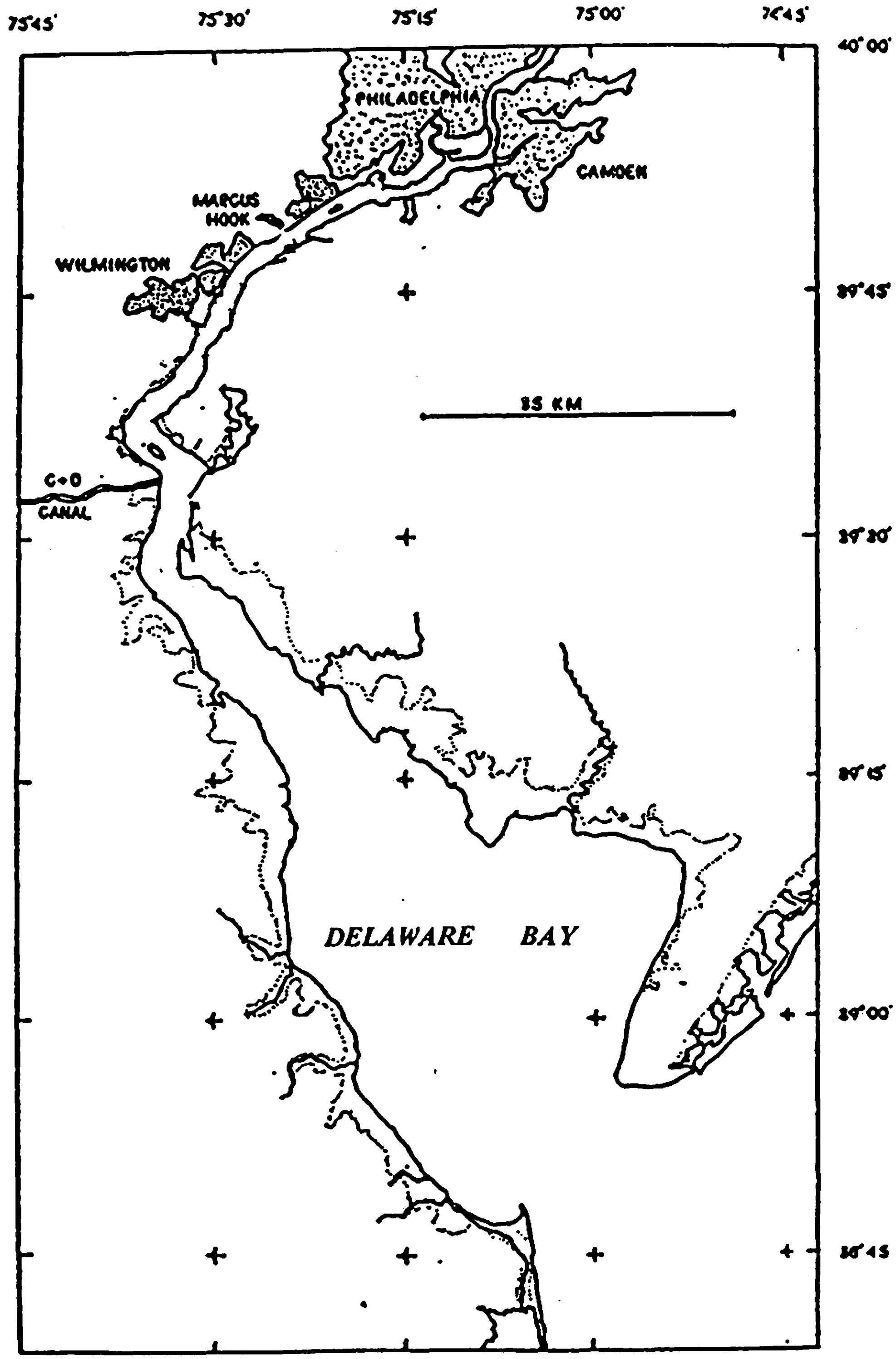

Figure 1. Delaware Bay and lower Delaware River Estuary (Fiest, 1977).

Organic Contaminants in Sediments & Water

Delaware River Basin Commission (1984) noted that "data for non-metallic toxic substances in the water column are generally not available." They did refer to a May, 1981 Delaware Department of Natural Resources and Environmental Control survey in which they found no chlorinated pesticides in the water column in the Delaware Estuary or Bay. They reported finding neither PCB nor chlorinated pesticides in the repeat survey in April, 1982.

The commission's 1984 report mentioned only a few incidences of sediment sample analyses for organic contaminants in the Delaware River Estuary and Bay. The Corps of Engineers tested dredged spoils from the navigation channel during 1979-1983. They found low concentrations of pesticides and organic compounds there. In 1982 the Delaware Department of Natural Resources and Environmental Control took a sediment sample from the estuary near the mouth of Delaware Bay. They found no detectable concentrations of various pesticides and organic compounds. The Delaware River Basin Commission (1984) considered it unfortunate (as we do) that "there are very few data taken in sediments in near-shore areas."

There were a few reports of Delaware Bay sediment analyses for hydrocarbons in the 1970's. Hydrocarbon data from analysis of sediments from stations from Philadelphia to the bay mouth by Wehmiller and Lethan (1975) indicated a contaminated nature of up-river sediments compared to lower bay sediments (Mauer, 1979). Fiest (1977) analyzed Delaware Estuary bottom sediment grab samples for saturated hydrocarbons. He collected the samples from the mouth of Delaware Bay, throughout the Bay, and upriver about 150 km from January, 1975 to March, 1976. His study recorded the impact on bottom sediments of the <u>Corinthos</u> oil spill on January 31, 1975 which released approximately two million liters of crude oil into the Delaware River at Marcus Hook, Pennsylvania, about 129 km upriver from the Bay mouth. He noted type of sediment and presence of new petroleum, drawing a picture of the oil being carried downstream in moderately sandy sediments with the Spring freshet. Fiest (1979) summarized that "the sediments of the upper and middle Estuary recorded the impact of contamination with petroleum hydrocarbons following the <u>Corinthos</u> spill, [but] lower Estuary sediments showed no noticeable influx over a fourteen month monitoring period" and that "petroleum hydrocarbons are preferentially accumulating in areas of rapid sediment accumulation, i.e. shoaling areas." He noted that "the dynamic nature of estuarine circulation processes makes the determination of baseline concentrations of hydrocarbons in bottom sediments a complicated procedure." Sediment data from Fiest (1979) show a concentration range for total aromatics of 0 - 1,111,000 ppb with a mean value from all samples in which aromatics were detected of 165,000 ppb, and a median value of 75,000 ppb.

Organic Contaminants in Biota

Delaware River Basin Commission staff (1984) discussed data from analysis of fish tissue from the Delaware River. They reported only two sources of this data: (1) routine fish tissue sampling begun in 1979 by the four Delaware River Basin States as part of EPA's Basic Water Quality Monitoring Program (They sampled up to ten locations in the river per year, but the data was not all available then.), and (2) a 1983 special study by the U.S. Department of Interior, Fish and Wildlife Service which contained only limited fish tissue data. Some chlordane values from these sources were a concern (1980 and 1981 data from Trenton, New Jersey and data from the Delaware Estuary), but they reported only small (below action level) quantities of other contaminants such as DDT and PCB in fish tissues. They found concentrations of most toxics they measured from biota in the lower Estuary and upper Delaware Bay even below detectable limits. The Commission made clear, however, that "sufficient data have not been collected from which to draw scientifically valid conclusions.

Butler (1973) reported pesticide residues in tissues of clams, oysters, and mussels from six locations along the Delaware side of the Delaware Bay from sampling 1967-1969. He also reported pesticide and PCB residues in tissues of oysters from five locations along the New Jersey side of the Bay from sampling 1966-1972. Following is a summary of the eastern oyster, Crassostrea virginica, maximum residue data, with values shown in ppb, dry weight (Stations are listed in geographical order going down river to the mouth of the Bay.):

Delaware Stations	DDT	Dieldrin	PCB
3	860	125	
4	450	50	
5	450	---	
New Jersey Stations			
5	1,225	115	5
4	1,390	130	10
3	625	60	---
2	715	25-50	5
1	1,065	60	10

Butler commented that the "magnitude of DDT residues in Delaware clams and oysters showed no trend toward increased or decreased levels during the 3-year monitoring period. His New Jersey data show a decreasing residue trend in 1971 compared to 1968-1969 data. The DDT tissue residues in New Jersey oysters was clearly higher than residues in Delaware oysters from the opposite shore of the same Delaware Bay.

Effects: Populations

O'Conner (1986) discussed results of a NOAA project searching for correlations between historical records of fishery stock size and contaminant inputs. While data was too "sketchy" and methods of extraction too imprecise to test for correlation between contaminant input and stock sizes, O'Conner summarized some results indicating a negative correlation between at least one increasing trend in human population or industrial activity over the fifty year period 1928-1978 and stock sizes of blue crabs, butterfish, scup, and shad.

There is evidence from 1982 and 1983 sampling upstream of the tidal Delaware River by the Delaware River Basin Commission (1984) that these sediments are not toxic. They found healthy, diverse, macroinvertebrate populations there.

Maurer (1979), after reviewing some of his own earlier studies and those of other investigators regarding pollutants and benthic invertebrate populations in Delaware Bay, felt there was "little evidence to suggest that levels of pesticides and hydrocarbons recorded in lower Delaware Bay have any measurable, harmful effects on adult invertebrates."

Biggs, et al. (1984) studied and reported on the effects of coal transfer on the aquatic environment of lower Delaware Bay. They expected no measurable degradation to occur in surface waters of the Bay at any likely annual transfer by any loading methodology and no measurable degradation in bottom waters by use of conveyor/auger systems for coal transfer. "Measurable environmental degradation will occur in the bottom waters from dust emissions associated with five million tons per year loaded by clamshell bucket" (Biggs, et al., 1984). They discussed composition of coal including the existence of carcinogenic PAHs and the potential for leaching of toxic organics from the coal into the aquatic environment. They considered pH and dissolved oxygen concentration the parameters having the greatest influence on the leaching of materials into seawater. They suggested that coal dust could "capture" natural and anthopogenic organics which would otherwise flow through the estuary into the ocean. The organics adherring to the coal dust could then enter the food chain or become incorporated into sediments.

Zooplankton could be adversely affected by the increased particulate load from coal dust particles and that could lead to expanding detrimental effects. Mysids comprise a large portion of the bottom-dwelling invertebrates in the Delaware Bay and are important along with other zooplankton and small benthic invertebrates as food for fish and crabs. The larvae and juvenile of weakfish in the Delaware Bay feed primarily on copepods and mysids. "Reduction in food resources of these fish would adversely affect their populations" (Biggs, et al., 1984).

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NEW YORK: HUDSON RIVER - RARITAN BAY ESTUARY

The consolidated metropolitan area associated with New York City is the most heavily populated in the United States, at 17,539,344 in 1980, according to the 1986 World Almanac. Certainly we would expect high production of anthropogenic pollutants to be reflected in high concentrations in sediments and biota there, and in adjacent bodies of water, unless some very effective controls have been used. We have used over fifty sources of information regarding organic contaminants in the New York Harbor, Hudson River - Raritan Bay, and New York Bight Apex areas to form an image of the type and magnitude of a problem that may exist in this aquatic environment. Figures 1 and 2, maps of the area, are included to identify sites discussed.

PAHs in Sediments

Boehm (1981), Macleod, et al. (1981), O'Conner, et al. (1982), Reid, et al. (1982), and Stainken (1984) all reported concentrations of PAHs in sediments from locations in the Hudson - Raritan estuaries and in the New York Bight.

Boehm (1981) sampled surface sediments from the New York Bight sewage sludge dumpsite in the summer of 1980 and sampled from water column stations, and sediments in the Bight and in outer and inner New York Harbor in the summer of 1981. He particularly noted differences between the sewage sludge and the dredge spoil sediment composition. He reported total PAH concentration in sewage sludge as 47,300 ppb; in the top 4.5 cm of dredge spoil as 1,800 ppb; and in the layer 4.5 - 9 cm deep of dredge spoil as 27,000 ppb total PAH. Results of analysis of sediments for concentrations of individual PAHs led to the estimate that 85 to 90 percent of the PAH found in the sewage sludge samples originated from petroleum sources. Dredge spoil samples, on the contrary, were rich in pyrogenic PAHs.

O'Connor, et al. (1982) reported concentrations of PAHs in sediments of various sites in the Hudson - Raritan estuary, in the New York Bight region, and in sewage sludge. Sampling was conducted in 1978 and 1979 and originally reported in MacLeod, et al. (1981) and in O'Connor, et al. (in Press, Environmental Pollution). O'Connor, et al. (1982) said that the major source of PAHs to the estuarine/coastal system is the New York metropolitan region. PAHs in harbor sediments come from oil spills, sewer overflows, municipal wastewater discharges, and atmospheric fallout. A summary of their data for mean values of total PAH in sediments follows:

Hudson - Raritan estuary = 42,700 ppb (dry wt.)
New York Bight region (range) = 22 - 6,000 ppb (dry wt.)
Sewage Sludge = 20,400 ppb (dry wt.)

O'Connor, et al. (1982) said that of the three classes of organic compounds they studied - PAHs, PCBs, pesticides - "the PAHs represent the majority of organic contaminants discharged into the Bight apex each year."

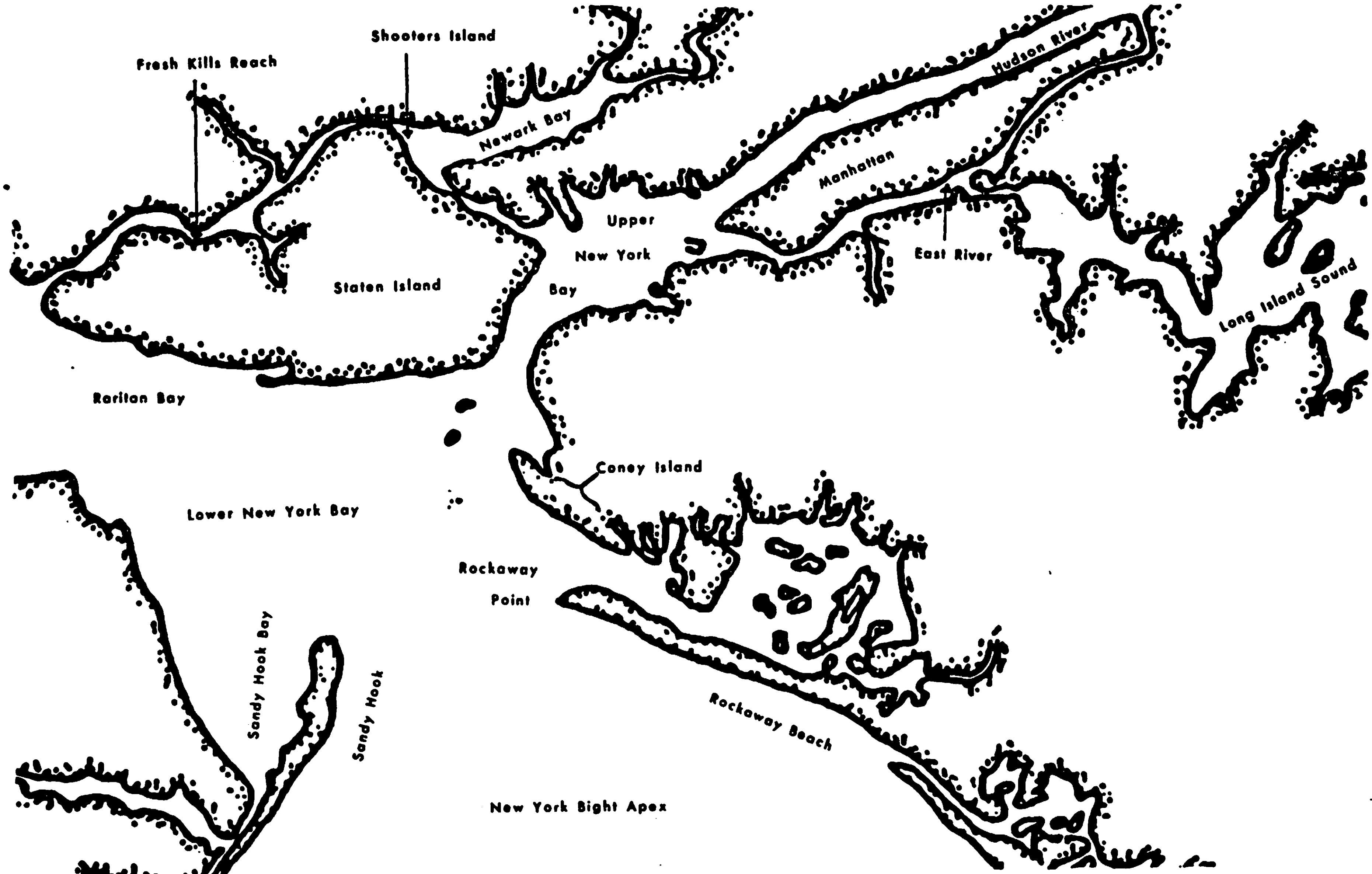

Figure 1. New York Harbor, Hudson-Raritan Estuary, New York Bight Apex, modified from Boehm (1981).

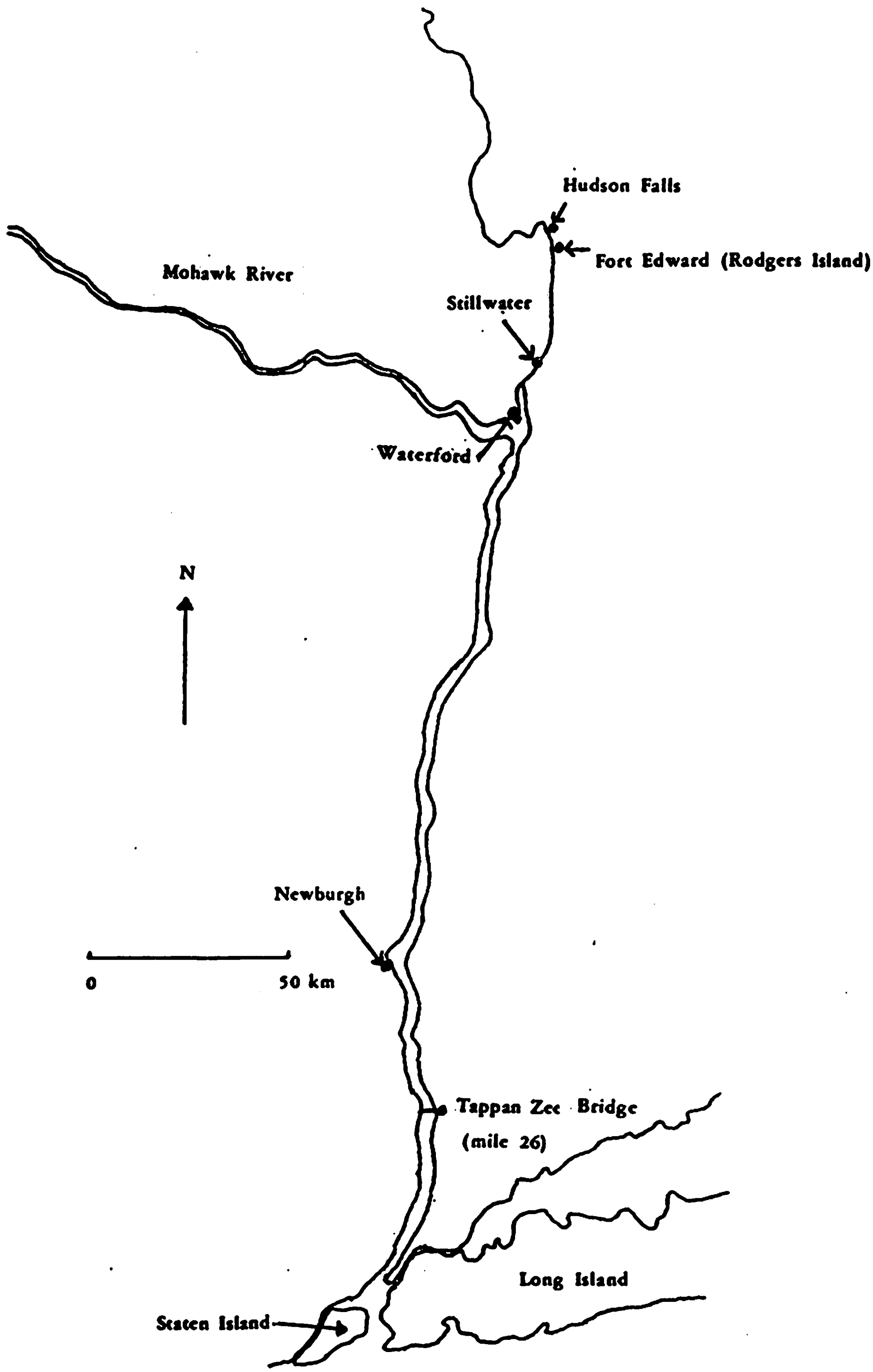

Figure 2. Hudson River

Reid, et al. (1982) observed "PAHs . . . were widely distributed but patchy, . . . with highest concentrations in the inner bight."

MacLeod, et al. (1981) gave results of analysis of sediments collected from sites in various areas in New York Harbor and the New York Bight from 1977-1980. Following are some of these data for mean concentrations of total PAH, shown in ppb (dry wt.):

Location	Date	PAH
Lower Bay	Oct. '77	5,442
Sandy Hook Bay	Aug. '77	3,055
N.Y. Bight Apex	Aug. '77	2,640
N.Y. Bight Apex	Oct. '77	3,928
Mamaroneck River, empties into N. Long Is. Sound	Aug. '78	8,860
Hudson River at W. Manhattan Is.	Oct. '78	1,940
Shooters Is.	Oct. '78	12,710
Hudson River, N. of Yonkers	May '80	294
Upper Bay	May '80	2,012
Ch. N. of Shooters Is.	May '80	6,423
Fresh Kills Reach	May '80	10,529

Stainken (1984) reported the mean value for PAH in his Raritan Bay sediment samples as 911 ppb. He sampled in October and November of 1977. Stainken's investigation indicated that circulation in the Bay deposited PAHs, especially the high molecular weight 3-6 ring ones, which are less soluble than others, in sediments along the southern shore of Staten Island.

Carriker, et al. (1982) stated that concentrations of PAHs in the New York estuarine system and Bight were very high in sewage sludge samples (1,000,000 ppb) and in estuarine sediments (2,000,000 ppb) and were low in outer Bight samples (30 ppb).

PCBs and Pesticides in Sediments

Bopp (1979) called the New York Harbor "the largest area of rapid sediment deposition in the tidal Hudson." He said that a million tons of fine grained sediment are removed from the harbor each year by maintenance dredging.

Bopp, et al. (1981) concluded from their study of PCBs in sediments of the Hudson River that the system had been contaminated during the previous three decades by discharges from two General Electric capacitor plants on the upper Hudson to an average level of about 10,000 ppb PCBs in contemporary tidal Hudson sediments. They found that the level of contamination decreased with distance downstream from these manufacturing plants. Carriker, et al. (1982) also called the capacitor manufacturing plants the largest point sources of PCBs to the Bight/estuary region and said the PCBs continue to be transported downstream with sediments even though the discharges have stopped.

Data concerning concentrations of PCBs in sediments from the Hudson Raritan Bay, the New York Harbor, upper and lower Bays, and the New York Bight Apex as reported by several investigators is shown in Table 1.

Dieldrin and chlordane concentrations reported by Bopp, et al. (1982) are shown in Table 1. Pesticides enter the Hudson-Raritan estuary via Hudson River transport and municipal wastewater. Major sources of the pesticides are agricultural use in the Hudson/Mohawk and Raritan watersheds, agricultural zones of eastern Long Island, and application of pesticides in the metropolitan area for control of cockroaches and other vermin (O'Conner, et al. 1982).

Belton, et al. (Oct., 1985) described a study by the Office of Science and Research of the New Jersey Department of Environmental Protection in which they discovered the highly toxic synthetic toxin, 2,3,7,8-TCDD, in sediments and biota of the Passaic River-Newark Bay system. They found the highest level of this dioxin in sediments at a site adjacent to the former Diamond Alkali site in Newark, N.J., even though the plant had not been operating for twelve years. They also discovered TCCD contaminated sediments upstream adjacent to the Givauden Chemical Company. TCCD is a chlorinated tricyclic aromatic compound which is an impurity produced when certain other chlorinated chemicals such as chlorophenols, widely used as pesticides, are manufactured.

PAHs in Water

Stainken and Frank (1979) sampled and analyzed bottom water (2-3 feet above bottom) from 18 sites in the Lower New York Bay-Raritan Bay Complex in 1977. Their analysis identified various PAHs present in water from some sites, but most samples had concentrations of PAHs below their detection level. A sample from near the Oakwood Sewage Treatment Plant (OKWD) was an exception. Concentrations of 1, 2, 3 and 4 ring PAHs were approximately 5-10 ppb each in the OKWD samples. "Although relatively high concentrations of polynuclear aromatic hydrocarbons are found in sediments and animals, very low concentrations occur in the water column" (Stainken and Frank, 1979).

Boehm (1983) reported mean total PAH concentration within the estuary, landward of Rockaway Point as 13 parts per trillion (.013 ppb) and in the New York Bight seaward of Rockaway Point as 1.9 parts per trillion (.0019 ppb) in surface waters and 55 parts per trillion (.055 ppb) in mid and bottom waters.

Stainken (1984) found substantially lower mean values of PAHs in water than in sediments and in bivalves of Raritan Bay. He reported a range of 5-20 parts per trillion in Raritan Bay water samples.

Table 1. Mean values for contaminant concentrations in New York sediments (ppb, dry wt.).

Location (depth in cm if known)	Date	Conc.	Contaminant	Reference
Raritan Bay (0-7)	73-76	200	PCB 1242	Bopp (1979)
Hudson Estuary at Manhattan Island (0-53)	74-77	4,425	PCB 1242	Bopp (1979)
(53-60)	74-77	750	PCB 1242	Bopp (1979)
Hudson River off Manhattan Island (0-50)	1975	4,950	PCB 1242	Bopp, et al. (1982)
Hudson River off Manhattan Island (0-50)	1975	1,949	PCB 1254	Bopp, et al. (1982)
Hudson River off Manhattan Island (0-5)	1975	6,900	PCB 1242	Bopp, et al. (1982)
Hudson River off Manhattan Island (0-5)	1975	1,950	PCB 1254	Bopp, et al. (1982)
Upper Bay (0-260)	1975	2,543	PCB 1242	Bopp, et al. (1982)
Upper Bay (0-260)	1975	1,079	PCB 1254	Bopp, et al. (1982)
Upper Bay (0-10)	1975	1,630	PCB 1242	Bopp, et al. (1982)
Upper Bay (0-10)	1975	610	PCB 1254	Bopp, et al. (1982)
Raritan Bay	1977	110	ΣPCB	Stainken (1984)
Raritan Bay (0-12)	1977	125	PCB 1242	Bopp + Simpson (1984-85?)
Raritan Bay (0-12)	1977	55	PCB 1254	Bopp + Simpson (1984-85?)
Raritan Bay	1977	232	ΣPCB	Stainken + Rollwagen (1979)
Sandy Hook Bay	1977	700	ΣPCB	MacLeod, et al. (1981)
Lower Bay	1977	75	ΣPCB	MacLeod, et al. (1981)
N.Y. Bight Apex	1977	1,035	ΣPCB	MacLeod, et al. (1981)
Sandy Hook	1977	72	ΣPCB	Stainken + Rollwagen (1979)
Lower Bay	1977	10	ΣPCB	Stainken + Rollwagen (1979)
Staten Is., intertidal	1977	467	ΣPCB	Stainken + Rollwagen (1979)
Mamaroneck River (empties into Long Is. Sound)	1978	450	ΣPCB	MacLeod, et al. (1981)
Hudson River, Manhattan Harbor, dredged material	1978	500	ΣPCB	MacLeod, et al. (1981)
Shooters Is. dredged material	1978	3,500	ΣPCB	MacLeod, et al. (1981)
Raritan Bay (0-12)	1980	488	PCB 1242	Bopp + Simpson (unpublished)
Raritan Bay (0-12)	1980	333	PCB 1254	Bopp + Simpson (unpublished)
Raritan Bay (12)	1980	160	PCB 1242	Bopp + Simpson (unpublished)
Raritan Bay (12)	1980	179	PCB 1254	Bopp + Simpson (unpublished)
Hudson River, N. of Yonkers	1980	62	ΣPCB	MacLeod, et al. (1981)
Upper Bay	1980	130	ΣPCB	MacLeod, et al. (1981)
Ch. N. of Shooters Is.	1980	3,200	ΣPCB	MacLeod, et al. (1981)
Fresh Kills Reach	1980	2,000	ΣPCB	MacLeod, et al. (1981)
Hudson River, off Manhattan Island (0-5)	1975	65	chlordane	Bopp, et al. (1982)
Hudson River, off Manhattan Island (5-50)	1975	137	chlordane	Bopp, et al. (1982)
Upper Bay, (0-10)	1975	165	chlordane	Bopp, et al. (1982)
Upper Bay (10-250)	1975	178	chlordane	Bopp, et al. (1982)
Hudson River, off Manhattan Island (0-5)	1975	17	dieldrin	Bopp, et al. (1982)
Hudson River, off Manhattan Island (5-50)	1975	37	dieldrin	Bopp, et al. (1982)
Upper Bay (0-10)	1975	37	dieldrin	Bopp, et al. (1982)
Upper Bay (10-250)	1975	51	dieldrin	Bopp, et al. (1982)

PCBs and Pesticides in Water

Boehm (1983) remarked that average PCB concentrations in the estuary landward of Rockaway Point are much higher than outside the estuary. He also noted that PCB levels are higher in bottom waters than in surface water.

Bopp and Simpson (unpublished) gave values for total PCBs they measured in Hudson River water in 1979 and 1980. The average of PCB concentrations from stations in the lower Hudson below the Tappan Zee Bridge and including an East River station was 15 parts per trillion (.015 ppb). The range was very small, from 14 to 17 parts per trillion. In Hudson River stations above the Tappan Zee Bridge but only up to milepost 82 (82 miles north in the Hudson from the Battery at the tip of Manhattan Island), concentrations were higher, averaging 37 parts per trillion (.037 ppb).

MacLeod, et al. (1981) detected .01 ppb total PCBs in subsurface water from a New York Bight station sampled in 1978. A surface microlayer sample from the same station contained .2 ppb total PCBs, .003 ppb Dieldrin, and .01 ppb Heptachlor.

Sloan, et al. (1985) reported PCB concentrations from two locations, Stillwater and Waterford in the upper Hudson from sampling 1977-1983. These data shown below basically indicate progressive decrease in PCB levels through time; also levels are slightly higher in water from the more northern location of Stillwater:

Hudson:	at Stillwater	at Waterford
1977	.69 ug/l	.37 ug/l
1978	.41 "	.34 "
1979	.82 "	.47 "
1980	.33 "	.22 "
1981	.34 "	.26 "
1982	.24 "	.16 "
1983	.25 "	.15 "

PAHs in Biota

O'Conner, et al. (1982) stated that "Contamination of biota with pesticides, PCBs, and PAHs is essentially ubiquitous in the system (New York Bight), but is at relatively low levels." They noted that PAH concentrations in biota of the Bight were highest in blue mussels and lobsters and lowest in fishes, and that concentrations were higher in hepatopancreas and livers of lobsters and winter flounder than in edible flesh.

Reid, et al. (1982) noticed a slight tendency for total PAH, minus phenanthrene, for all species combined to be highest in and near Christiaensen Basin (near the New York Bight apex, betwen sewage sludge and dredged material disposal sites). They also ranked species they analyzed according to overall PAH values - highest in rock crabs (almost all phenanthrene), lobster next, and red hake next.

Pancirov and Brown (1977) reported measurements of PAH concentration in fish, mollusk, and crustacean tissue from a variety of locations in North America. Oysters from Long Island Sound and crabs from Raritan Bay had higher than 2 ppb, wet wt. (approx. 10 ppb, dry wt.). They explained that these waters "are exposed to municipal and industrial wastes." They reported about 14 ppb (dry wt.) for total PAHs in their Raritan Bay crabs.

Ecological Analysts, Inc. and SEAMOcean, Inc. (1983) observed from results in MacLeod, et al. (1981) that replicate plankton/egg samples at their station closest to the Sandy Hook - Rockaway Point transect collected on the same date showed up to 1,000 fold variability in PAH concentrations.

MacLeod, et al. (1981) sampled fish and invertebrates from the New York Bight area in 1977-1979 and reported tissue levels of individual PAHs for some; though they found that most PAHs were not present above detection levels. Following are some of the means of detectable concentrations of PAHs (shown in ppb, dry wt.) for biota reported by MacLeod:

Lobster, Raritan Bay, 1978	
methynaphthalene	40
phenanthrene	30
fluoranthene	20
pyrene	30

Blue mussel, Sandy Hook, 1977	
naphthalene	50
phenanthrene	50
fluoranthene	400
pyrene	400
benz(a)anthracene	700
chrysene	300

Blue mussel, Staten Island, 1979	
naphthalene	100
1-methylphenanthrene	50
fluoranthene	300
pyrene	50

Blue mussel, Coney Island, 1979	
naphthalene	40
1-methylphenanthrene	400
fluoranthene	1,000
pyrene	1,000

Winter flounder, Raritan Bay, 1977	
naphthalene (in flesh)	17
naphthalene (in liver)	270

Windowpane flounder, Raritan Bay, 1977	
naphthalene	23

Macleod, et al. (1981) reported that PAHs were not found above detection levels in mackerel sampled from the New York Bight apex in 1977 or in surf clams sampled from Rockaway Beach 1978-79. Stainken's (1984) mean value of PAHs in bivalve tissue of the Raritan Bay - Lower New York Bay complex sampled in 1977 was 387 ppb.

PCB & Pesticides in Biota

Stainken (1984) referring to bivalves sampled from the Raritan Bay - Lower New York Bay complex in 1977 commented that "The tissues at all sites generally contained more PCB than did the sediments." Stainken and Rollwagen (1979) reported a range of PCB values in bivalves of 12-360 ppb, wet wt. (approx. 60 - 1,800 ppb, dry wt.). Their values for PCBs in tissues of oysters, soft and hard clams, are shown in Table 2 along with PCB levels in biota reported in other studies of the Raritan Bay - Hudson River Estuary and New York Bight apex.

Ecological Analysts, Inc. and SEAMOcean, Inc. (1983) noted that results in MacLeod, et al. (1981) showed a high variability for PCB concentration in plankton/egg samples from the New York Bight. Duplicate samples from their site at the Sandy Hook - Rockaway Point transect had both the highest and lowest values of PCB concentrations. Variability was in the range of 2-40 fold.

In 1975 Skea, et al. (1979) collected fish from a lake in New York and placed them in cages in the Hudson River near Fort Edward and Rogers Island for two weeks, then analyzed whole fish composite samples by species. They also analyzed uncaged fish. PCB residues in the four species of caged fish were similar, suggesting to them that bioaccumulation of PCBs from water "does not differ substantially among species of similar lipid content and size." Apparently the quantity of PCBs accumulated is closely related to the fat content and weight of fish. The heaviest fish with the highest fat content accumulated the largest amount of PCBs per fish. PCB concentrations for each of the four species of fish caged and analyzed by Skea, et al. (1979) are shown in Table 2. The uncaged fish (fish before exposure to Hudson River water) contained less than 20 ppb Aroclor 1016.

Sloan, et al. (1985) considered pumpkinseed, Lepomis gibbosus, a fish sensitive to PCB environmental contamination and useful for monitoring trends in PCB contamination of Hudson River biota. Referring to data from pumpkinseed and water analysis they noted a decline in PCB concentrations throughout the river in 1980. This decline was probably a response to the 1977 cessation of major PCB discharges at Fort Edward and Hudson Falls. In 1983 following a flood there was an increase in mean PCB concentration in yearling pumpkinseed in the lower Hudson River, but there were no significant changes in PCB concentrations in pumpkinseed collected in the upper river. After comparing concentrations among other species and age classes of fish through 1983 they found that "increases were not uniform, neither was there evidence of continued decline." Given the relatively high sediment concentrations along with fish concentration data they concluded that "this system has reached a quasi-stable state with respect to PCB levels in the biota." An October 9, 1981 press release included in New York State DEC (Dec., 1981) report also stated that the decline in PCB levels in Hudson River fish following termination of PCB discharges had stabilized. It warned that higher spring flows could contaminate the lower Hudson fisheries by washing PCB contaminated sediment down river. See Table 2 for some of the data from Sloan, et al. (1985).

Belton, et al. (1983) discussed results of investigations of PCB contamination in fish by the New Jersey Department of Environmental Protection. The primary type PCB detected during these investigations was Aroclor 1254. Most heavily contaminated fish included striped bass and bluefish which "are quite important to both the recreational and commercial fisheries in New Jersey." American eel, white perch and white catfish also showed elevated PCB levels for both the sampling periods 1975-1980 and 1981-1982. Atlantic sturgeon also had high PCB tissue concentrations, but sampling size was too small to use results for managerial decisions (Bruce Ruppel, Personal Communication, April 15, 1986).

Table 2a. Mean values for ΣPCBs in fishes of New York Harbor and surrounding area in ppb, dry wt.

Organism	Location	Date	Conc.*	Reference
Striped bass	Raritan Bay	1975	640-1,820[1, 2]	Pearce, 1979a,b.
Spot	Raritan Bay	1975	530[2]	Pearce, 1979a,b.
Winter flounder	Raritan Bay	1975		Pearce, 1979a,b.
60-65 mm			370[2]	
220-495 mm			140[2]	
Snapper bluefish,	Raritan Bay	1975		Pearce, 1979a,b.
"30 cm			3,090[2]	
Weakfish, (26 cm)	Raritan Bay	1975	630[2]	Pearce, 1979a,b.
Eels	Raritan Bay	1975	2,193[2]	Pearce, 1979a,b.
caged Creek chubsucker	Hudson near Rogers Is.	1975	11,000[3]	Skea, et al. 1979.
caged Yellow perch	Hudson near Rogers Is.	1975	9,000[3]	Skea, et al. 1979.
caged Pumpkinseed	Hudson near Rogers Is.	1975	12,500[3]	Skea, et al. 1979.
caged Brown bullhead	Hudson near Rogers Is.	1975	16,500[3]	Skea, et al. 1979.
Striped bass	Raritan/Sandy Hook Bays	75-80	4,150	Belton, et al. 1982.
American eel	Raritan/Sandy Hook Bays	75-80	5,750	Belton, et al. 1982.
Bluefish	Raritan/Sandy Hook Bays	75-80	4,900	Belton, et al. 1982.
Striped bass	Lower Hudson River (0-15 miles)	75-80	17,450	Belton, et al. 1982.
Bluefish	Lower Hudson River (0-15 miles)	75-80	17,200	Belton, et al. 1982.
Winter flounder, with fin erosion, muscle	Sandy Hook/Raritan Bay	76-77	365-700[1]	Sherwood 1982.
Winter flounder, without fin erosion, muscle	Great Bay, NJ (control area)	76-77	25-700[1]	Sherwood 1982.
Winter flounder, with fin erosion, muscle	N.Y. Bight Apex	76-77	395-550[1]	Sherwood 1982.
Winter flounder, with fin erosion, liver	Sandy Hook/Raritan Bay	76-77	7,000-37,000[1]	Sherwood 1982.
Winter flounder, without fin erosion, liver	Great Bay, NJ (control area)	76-77	1,250-12,000[1]	Sherwood 1982.
Winter flounder, with fin erosion, liver	N.Y. Bight Apex	76-77	10,000-50,000[1]	Sherwood 1982.
Brown bullhead, muscle	Hudson at Stillwater	1977	532,500	Sloan, et al. 1985.
Yellow perch, edible flesh	Hudson at Stillwater	1977	60,000	Horn + Skinner 1985.
Brown bullhead, edible flesh	Hudson at Stillwater	1977	550,000	Horn + Skinner 1985.
Striped bass, edible portion	Staten Island	1978	26,950	NYSDEC (June) 1981.
Atlantic tomcod	Hudson River estuary	1978		Klauda, et al. 1981.
body tissue			850	
liver			187,600	
Brown bullhead, muscle	Hudson at Stillwater	1979	44,850	Sloan, et al. 1985.
Brown bullhead, muscle	Hudson at Stillwater	1979	45,700	NYSDEC (June) 1981.
Pumpkinseed, whole	Hudson at Stillwater	1979	99,550	Sloan, et al. 1985. NYSDEC (June) 1981.
Pumpkinseed, whole	Hudson at Newburgh	1979	15,150	Sloan, et al. 1985.
Pumpkinseed, whole	Hudson at Newburgh	1979	14,950	NYSDEC (June) 1981.
Striped bass, muscle	Tappan Zee Bridge	1979	34,950	NYSDEC (June) 1981.
American shad, muscle	Tappan Zee Bridge	1979	6,850	NYSDEC (June) 1981.
American eel, edible portion	East River	1979	14,225	NYSDEC (June) 1981.

American eel, edible portion	Upper Bay	1979	18,700	NYSDEC (June) 1981.
Brown bullhead, muscle	Hudson at Stillwater	1980	61,700	Sloan, et al. 1985. NYSDEC (June) 1981.
Pumpkinseed, whole	Hudson at Stillwater	1980	100,600	NYSDEC (June) 1981.
Pumpkinseed, whole	Hudson at Stillwater	1980	108,650	Sloan, et al. 1985.
Pumpkinseed, whole	Hudson at Newburgh	1980	23,150	Sloan, et al. 1985. NYSDEC (June) 1981.
Striped bass	Tappan Zee Bridge	1980	38,400	NYSDEC (Dec.) 1981.
Striped bass, muscle	Tappan Zee Bridge	1980	29,900	NYSDEC (June) 1981.
American shad, muscle	Tappan Zee Bridge	1980	7,750	NYSDEC (June) 1981.
American eel, muscle	Manhattan (Pier 40)	1980	29,450	NYSDEC (June) 1981.
Yellow perch, edible flesh	Ft. Edward/Thompson Is.	1980	105,000	Horn + Skinner 1985.
Brown bullhead, edible flesh	Ft. Edward/Thompson Is.	1980	50,000	Horn + Skinner 1985.
Yellow perch, edible flesh	Hudson at Stillwater	1980	5,000	Horn + Skinner 1985.
Brown bullhead, edible flesh	Hudson at Stillwater	1980	60,000	Horn + Skinner 1985.
Pumpkinseed, whole	Hudson at Stillwater	1981	68,250	Sloan, et al. 1985.
Pumpkinseed, whole	Hudson at Stillwater	1981	71,700	Sloan, et al. 1985.
Pumpkinseed, whole	Hudson at Newburgh	1981	11,625	Sloan, et al. 1985.
Pumpkinseed, whole	Hudson at Newburgh	1981	13,900	Sloan, et al. 1985.
American eel, muscle	Tappan Zee Bridge	1981	54,150	NYSDEC (Dec.) 1981.
Striped bass	Tappan Zee Bridge	1981	22,200	NYSDEC (Dec.) 1981.
American shad, muscle	Tappan Zee Bridge	1981	7,000	NYSDEC (Dec.) 1981.
Pumpkinseed, whole	Hudson at Stillwater	1981	69,100	NYSDEC (June) 1982.
Pumpkinseed, whole	Hudson at Newburgh	1981	12,465	NYSDEC (June) 1982.
Striped bass	Raritan Bay	1981	19,650[4]	Belton, et al. 1983.
American eel	Raritan Bay	1981	5,300[4]	Belton, et al. 1983.
Bluefish	Raritan/Sandy Hook Bays	1981	6,450[4]	Belton, et al. 1983.
Bluefish	Raritan/Sandy Hook Bays	1981	6,413	Ruppel 1983.
Bluefish	Hudson/Upper N.Y. Bay	1981	8,900	Ruppel 1983.
Striped bass	Lower Hudson River (0-15 miles)	1981	7,800[4]	Belton, et al. 1983.
American eel	Lower Hudson River (0-15 miles)	1981	17,500[4]	Belton, et al. 1983.
Striped bass	Hudson River, km 30	?	60,000	O'Conner, et al. 1982 In: O'Conner, et al. (in press).
Winter flounder	Raritan Bay	?	350	O'Conner, et al. 1982 In: O'Conner, et al. (in press).
Pumpkinseed, whole	Hudson at Stillwater	1982	45,183	Sloan, et al. 1985.
Pumpkinseed, whole	Hudson at Stillwater	1982	46,050	Sloan, et al. 1985.
Brown bullhead, muscle	Hudson at Stillwater	1982	48,850	Sloan, et al. 1985.
Pumpkinseed, whole	Hudson at Newburgh	1982	8,017	Sloan, et al. 1985.
Pumpkinseed, whole	Hudson at Newburgh	1982	11,100	Sloan, et al. 1985.
Bluefish	Hudson/Upper N.Y. Bay	1982	8,260	Ruppel 1983.
Bluefish	Raritan/Sandy Hook Bays	1982	3,394	Ruppel 1983.
Bluefish	Raritan/Sandy Hook Bays	1982	2,300	Belton, et al. 1983.
Striped bass	Lower Hudson River (0-15 miles)	1982	15,900	Belton, et al. 1983.
American eel	Lower Hudson River (0-15 miles)	1982	24,000	Belton, et al. 1983.
Yellow perch, edible flesh	Hudson at Stillwater	1982	25,000	Horn + Skinner 1985.

Brown bullhead, edible flesh	Hudson at Stillwater	1982	50,000	Horn + Skinner 1985.
American eel, edible flesh	Hudson at Stillwater	1982	150,000	Horn + Skinner 1985.
American eel, edible flesh	Newark Bay at Shooters Island	82-83	12,425	Belton, et al., 1985 (Sept.)
Striped bass, edible flesh	Newark Bay at Shooters Island	1983	23,085	Belton, et al., 1985 (Sept.)
Pumpkinseed, whole	Hudson at Stillwater	1983	45,450	Sloan, et al. 1985.
Pumpkin seed, whole	Hudson at Stillwater	1983	54,250	Sloan, et al. 1985.
Brown bullhead, muscle	Hudson at Stillwater	1983	84,000	Sloan, et al. 1985.
Pumpkinseed, whole	Hudson at Newburgh	1983	13,450	Sloan, et al. 1985.
Pumpkinseed, whole	Hudson at Newburgh	1983	12,417	Sloan, et al. 1985.
Yellow perch, edible flesh	Ft. Edward/Thompson Is.	1983	165,000	Horn + Skinner 1985.
Brown bullhead, edible flesh	Ft. Edward/Thompson Is.	1983	75,000	Horn + Skinner 1985.
Brown bullhead, edible flesh	Hudson at Stillwater	1983	85,000	Horn + Skinner 1985.
Yellow perch, edible flesh	Hudson at Stillwater	1984	25,000	Horn + Skinner 1985.
Brown bullhead, edible flesh	Hudson at Stillwater	1984	55,000	Horn + Skinner 1985.

* Wet wt. converted to Dry wt. via Dry wt. = .20 (Wet wt.); therefore, ppb, Dry wt. = 5 (ppb, Wet wt.)

1. Range shown
2. No indication of wet/dry weight basis
3. PCB_{1016}
4. PCB_{1254}

Fish species, Table 2a:

Striped bass ... <u>Morone saxatilus</u>
Spot ... <u>Leostomus xanthurus</u>
Winter flounder ... <u>Pseudopleuronectes americanus</u>
Bluefish ... <u>Pomatomus saltatrix</u>
Weakfish ... <u>Cynoscion regalus</u>
American eel ... <u>Anguilla rostrata</u>
Atlantic tomcod ... <u>Microgadus tomcod</u>
Brown bullhead ... <u>Ictalurus nebulosus</u>
Pumpkinseed ... <u>Lepomis gibbosus</u>
American shad ... <u>Alosa sapidissima</u>
Yellow perch ... <u>Perca flavescens</u>
Creek chubsucker ... <u>Erimyzon oblongus</u>

Table 2b. Mean values for ΣPCBs in invertebrates of New York Harbor and surrounding area in ppb, dry wt.

Organism	Location	Date	Conc.	Reference
Mercenaria mercenaria	Raritan Bay	1977	550[1]	Stainken + Rollwagen 1979.
Crassostrea virginica	Raritan Bay	1977	405[1]	Stainken + Rollwagen 1979.
Mya arenaria	Raritan Bay	1977	455[1]	Stainken + Rollwagen 1979.
Mercenaria mercenaria	Sandy Hook	1977	550[1]	Stainken + Rollwagen 1979.
Mercenaria mercenaria	OKWD outfall in Lower Bay	1977	1,800[1]	Stainken + Rollwagen 1979.
Mya arenaria	Staten Is., intertidal	1977	1,315[1]	Stainken + Rollwagen 1979.
Mytilus edulis	Sandy Hook Bay	1977	1,200	MacLeod et al. 1982.
Homarus americanus flesh	Raritan Bay	1978	1,350	MacLeod et al. 1982.
Surf clams	Rockaway Beach, Long Is.	1978	167	MacLeod et al. 1982.
Mytilus edulis	Staten Island	1979	2,500	MacLeod et al. 1982.
Mytilus edulis	Coney Island	1979	3,200	MacLeod et al. 1982.
Surf clams	Rockaway Beach, Long Is.	1979	150	MacLeod et al. 1982.
Callinectes sapidus, whole without shell	Tappan Zee Bridge	1979	14,750	NYSDEC 1981.
Callinectes sapidus, hepatopancreas	Tappan Zee Bridge	1979	33,750	NYSDEC 1981.
Homarus americanus	Raritan Bay	?	1,500[1]	O'Conner, et al. 1982. In: O'Conner, et al. (in press).
Mytilus edulis	Sandy Hook	?	1,000[1]	O'Conner, et al. 1982. In: O'Conner, et al. (in press).
Mytilus edulis	Coney Island	?	2,000[1]	O'Conner, et al. 1982. In: O'Conner, et al. (in press).
Mercenaria mercenaria	Raritan Bay	?	500[1]	O'Conner, et al. 1982. In: O'Conner, et al. (in press).
Callinectes sapidus, whole	Newark Bay	1982	7,963[1]	Belton, et al. 1985 (Sept.)
Homarus americanus	Raritan Bay	1984	3,600[1]	Belton, et al. 1985 (Oct.)

1. Wet wt. converted to Dry wt. via Dry wt. = .20(Wet wt.); therefore, ppb, Dry wt. = 5(ppb, Wet wt.)

Invertebrate species, Table 2b: Mercenaria mercenaria = hard clam
Crassostrea virginica = Eastern oyster
Mya arenaria = soft clam
Mytilus edulis = blue mussel
Homarus americanus = American lobster
Callinectes sapidus = blue crab

Bruce Ruppel, of the Office of Science and Research within New Jersey's Department of Environmental Protection (DEP) wrote in a recent but undated flier about results of the DEP's PCB survey project. He reported that they found detectable levels of PCBs in edible flesh of 75 percent of the finfish and 50 percent of the shellfish. Only 2.4 percent of those finfish and none of the shellfish contained levels at or above the Food and Drug Administration "action level" for contaminated food of that time. The FDA in 1977 proposed a lower PCB tolerance level of 2 ppm (2,000 ppb), then adopted that level in 1984. Levels of PCBs in 11.1 percent of the finfish the DEP surveyed exceeded the proposed level.

Belton, et al. (1983) noted that temporal trends in PCB contamination varied among fish species. Aroclor 1254 concentrations remained relatively constant in American eel and striped bass samples for the three sampling periods 1975-80, 1981, and 1982. Concentrations declined steadily in white catfish for each period. Levels decreased in white perch in 1981, then increased in 1982. Levels in bluefish were lower in 1982 than in 1975-81. They found that the larger bluefish (≥60cm.) contained mean ΣPCB levels above the proposed FDA tolerance of 2 ppm (2,000 ppb). Generally results indicated elevated PCB levels in all fish collected from the Hudson-Raritan estuary in both 1981 and 1982. Ruppel (undated flier) said that "the Hudson River still appears to be the most severely contaminated drainage within the state's (New Jersey) water."

Reid, et al. (1982) reported that PCB values in biota were not consistently related to levels in sediments. The rock crab was an exception. The difference between concentrations in the relatively contaminated inner New York Bight area and concentrations for rock crabs outside this area was statistically significant ($p<0.01$). Tissue level data from windowpane flounder and lobsters appeared to be related to sediment levels, but they could not establish definite statistical significance. They had only one outer Bight sample for lobsters. They thought one reason for the apparent inconsistency in relation of PCB body burdens to contamination of sediments might be mobility of the species they sampled.

Reid, et al. (1982) compared their values for PCB concentrations in biota to those of MaLeod et al. (1981). Theirs were similar for scallops (<40 ppb wet or < 200 ppb based on dry wt.), and equal to or higher than those of MacLeod, et al. (1981) for lobster, rock crabs, and windowpane flounder.

Belton, et al. (Oct., 1985) reported concentrations of the highly toxic dioxin, TCDD, in tissues of finfish and crustaceans from New Jersey waterways. The following are mean concentration data from 1982 - 1984 samples from Newark Bay that contained detectable TCCD levels:

Blue crab, composite hepatopancreas	1982	503 parts per trillion
Striped bass, whole fish composite	1983	40 parts per trillion

Following are some Raritan Bay dioxin data:

Striped bass, (one sample)	1983	20 parts per trillion
American lobster, muscle and hepatopancreas mixture	1984	44 parts per trillion

Table 3a. Mean pesticide concentrations in fishes of New York Harbor, Hudson River, Raritan Bay, and New York Bight Apex, shown in ppb, dry wt.

Organism	Location	Date	Pesticide	Conc.	Reference
Mackerel flesh	New York Bight Apex	1977	Dieldrin	46	MacLeod, et al. 1981.
Mackerel flesh	New York Bight Apex	1977	DDT and metabolites	222	MacLeod, et al. 1981.
Winter flounder	S. of Montauk, Long Island	1978	Dieldrin	4	MacLeod, et al. 1981.
Winter flounder	S. of Montauk, Long Island	1978	DDT and metabolites	690	MacLeod, et al. 1981.
Striped bass	Tappan Zee Bridge	1980	DDT	1,200	NYDEC (Dec.) 1981.
Striped bass	Tappan Zee Bridge	1980	chlordane	250	NYDEC (Dec.) 1981.
Striped bass	Hudson River, km 15	?	ΣDDT	1,950	O'Conner, et al. 1982 In: O'Conner, et al. (in press).
Winter flounder	Raritan Bay	?	ΣDDT	45	O'Conner, et al. 1982 In: O'Conner, et al. (in press).
Bluefish	Raritan/Sandy Hook Bays	1981	chlordane	551	Belton, et al. 1983.
Bluefish	Hudson/Upper N.Y.Bay	1981	chlordane	1,030	Ruppel 1983.
Bluefish	Sandy Hook Bay	1981	chlordane	1,069	Ruppel 1983.
Bluefish	Hudson/Upper N.Y.Bay	1982	chlordane	198	Ruppel 1983.
Bluefish	Raritan/Sandy Hook Bays	1982	chlordane	224	Ruppel 1983.
Bluefish	Raritan/Sandy Hook Bays	1982	chlordane	139	Belton, et al. 1983.

Table 3b. Mean pesticide concentrations in invertebrates of New York Harbor, Raritan Bay, and New York Bight Apex, shown in ppb, dry wt.

Organism	Location	Date	Pesticide	Conc.	Reference
Blue mussel	Sandy Hook Bay	1977	DDT and metabolites	221	MacLeod, et al. 1981.
Lobster flesh	Raritan Bay	1978	Dieldrin	35	MacLeod, et al. 1981.
Blue mussel	Staten Island	1979	Dieldrin	10	MacLeod, et al. 1981.
Blue mussel,	Staten Island	1979	DDT and metabolites	340	MacLeod, et al. 1981.
Blue mussel	Sandy Hook	?	ΣDDT	325	O'Conner, et al. 1982 In: O'Conner, et al. (in press).
Lobster	Raritan Bay	?	ΣDDT	130	O'Conner, et al. 1982 In: O'Conner, et al. (in press).
Blue mussel	Coney Island	?	ΣDDT	160	O'Conner, et al. 1982 In: O'Conner, et al. (in press).
Surf clam	Rockaway Beach	?	ΣDDT	20	O'Conner, et al. 1982 In: O'Conner, et al. (in press).

O'Conner, et al. (1982) remarked that flesh concentrations of DDT were higher in striped bass from the estuary (Hudson and New York Harbor) than in fish from the ocean (off Montauk Point) by a factor of between four and ten. Table 3 shows mean pesticide concentrations in fish and some invertebrates as reported by investigators from 1977 - 1982 sampling.

Effects: Populations

Jeffries (1964) studied distribution of estuarine zooplankton species in Raritan Bay and in two other areas, Narragansett Bay, Rhode Island, and the York River in Virginia in 1957 and 1958. He noted differences among the three areas mostly in timing of seasonal changes, presumably due to their climatic differences. He also noted differences in dominance. "Polychaete larvae dominated the temporary plankton at both stations in Raritan Bay, codominated with larval gastropods in Narragansett Bay, and shared secondary overall relative importance with lamellibranch veligers in the York River." Jeffries was able to separate local influences on estuarine zooplankton from normal climatic variation. Consequently he concluded that in Raritan Bay the relative proportion of major taxonomic groups indicated artificial complications arising from waste materials. He noted an irregularity peculiar to Raritan Bay. The copepod, Acartia tonsa, disappeared during summer. It's disappearance paralleled drastic flucuations in phytoplankton which respond to nutrients from discharged waste and land runoff with dense, persistent spring blooms.

Pearce (1974) called the waters of Raritan Bay a "highly impoverished" environment and discussed the "deterioration" of the Hudson River estuary. He also noted that benthic invertebrate communities of Lower Bay and of the area between Sandy Hook and the Rockaways vary tremendously by virtue of variations in sediment type, water depth, salinity, channel dredging, waste disposal, etc.

As example, of the deterioration of the estuary, Pearce (1974) said that commercially important species, particularly the American oyster, which once were harvested from Raritan Bay, "are no longer present in sufficient numbers to warrant commercial activities." He found standing crops, diversity, and productivity of benthic invertebrates greatly diminished at sampling stations, particularly at the western end of Raritan Bay, compared to "the era before man began to impinge upon the waters." He attributed the deterioration to physical disturbances and pollutants in Raritan Bay.

Pearce (1974) considered the significant effect of ocean dumping on bottom dwelling invertebrates the important point of his presentation. Frequently bottom samples from sludge and dredge spoil beds contained none of the normal benthic fauna expected there. When samples from waste disposal sites did contain benthic organisms, the divesity of species was often greatly reduced. Pearce hypothesized that several factors including toxins in sediments, reducing conditions with associated reduced dissolved oxygen, and direct burial of the invertebrates were responsible.

Ristich, et al. (1977) reported on results of a 1972 benthic survey of invertebrate fauna of the 80-mile area of the Hudson River from New York Harbor to Poughkeepsie. They observed that salinity "was one of the most important measurable factors controlling species range and community boundaries." Lowest number of species was in fresh water. They reported the highest number of animals in stations south of the Tappan Zee Bridge (mile point 26). The data they presented indicated that the Hudson estuary was healthy. High density of organisms and dominance of organisms such as Cyathima polita, which are intolerant to pollution and enrichment were signs of health. They did, however, find obviously stressed areas near sewage treatment plants, sewage outfalls, and heavy industry, where they collected few organisms.

Results from a study of the Hudson estuary in 1974-1975 reported in Weiss, et al. (1978) agree with bivalve distributions reported in Ristich, et al. (1977). Weiss, et al. (1978) also studied distributions of foraminifera and diatoms and found them clearly salinity related. Their descriptions of composite "assemblages" of mollusks, diatoms and foraminifera should be useful to subsequent investigations attempting to determine effects of factors other than salinity, such as organic pollution, on communities in the Hudson estuary.

Wolfe, et al. (1982) agreed with Pearce's (1974) assessment of the Hudson-Raritan estuary. They believed pollutants had degraded the environments, causing most severe effects in the western portions of Raritan Bay and the New York Bight Apex. Although ecological changes in the New York Bight can be documented, they noted the area had been subjected to such a wide variety of pollutants for such a long time, they could not distinguish which pollutants were responsible for which changes.

Franz (1982) reported the disappearance of molluscan species from Staten Island by the 1920's. Historical accounts of New York's oyster industry provided evidence that the major environmental deterioration of New York Harbor occurred from 1890-1920. Records showed that "by the 1920's, DO levels over much of the harbor had declined to critical levels..." (Franz, 1982). He said the number of species remaining in the 1920's is near the present number. Franz compared species richness of mollusks in muddy sand assemblages of Raritan Bay with analagous assemblages of other Middle Atlantic coastal areas. The comparison indicated an impoverished condition in Raritan Bay. Following are examples of molluscan species richness in a few muddy sand habitats reported in Franz (1982):

Location	Total	Reference
Buzzards Bay	23	Sanders (1960)
Long Island Sound	14	Franz (unpublished, 1972 data)
N.Y. Bight Apex	9	Franz (unpublished, 1973 data)
Raritan Bay	4	McGrath (1974)

Michael (1982) assessed the probable relative importance of petroleum hydrocarbons in causing changes in benthic communities of the New York Bight. He observed that changes in benthic community patterns in parts of the New York Bight and in the Hudson River estuary were similar to those caused by oil spills in locations where other contaminant levels were very low. Although petroleum hydrocarbons include toxic and persistent PAH components, Michael considered it most likely impossible to identify individual compounds responsible for changes in the benthic communities, just as Wolfe, et al. (1982) thought.

Reid, et al. (1982) identified a small, highly impacted area just west of the sewage sludge dumpsite dominated by pollution tolerant polychaetes Capitella sp. and an "enriched" zone surrounding that area. Crustaceans were scarce, but density of several polychaetes, a small bivalve and an anthrozoan were higher in the enriched zone. They found the lowest number of species in the New York Bight at their station in the sewage sludge deposition area. (Number of species is generally lower in stressed areas.)

Stainken, et al. (1984) reported results of sampling Raritan Bay macrobenthos during the four seasons of 1979-80. Seasonal species data indicated presence of different species between summer and fall, and between spring and summer; but the head of the bay and mid bay consistently contained the most species. They also reported a lower density of Mercenaria mercenaria than earlier studies, but still judged from their macrofaunal survey results that there had not been major changes in the quality of the environment in the last six years.

Stainken (1984) sampled sediments at 14 sites in the Raritan Bay - Lower New York Bay complex for benthic fauna in June and October - November, 1977. He examined the relationships of sediment PAH and PCB to species diversity in June and to the dominant infauna, polychaetes and mollusks. He discovered a correlation of -0.61 with numbers of polychaetes, indicating a possible influence from the PAHs and PCBs in the sediments. Correlations for the other relationships examined were indistinct; however, Stainken did note an apparent trend toward decreasing numbers of species as total organic loads (hydrocarbons, PCBs, PAH,) exceeded 300-400 ppm (300,000-400,000 ppb). His study revealed "a trend toward decreasing diversity and density or richness of fauna as the total sediment hydrocarbons and xenobiotics increased" (Stainken, 1984). His results also showed decreasing numbers of species with increasing percent silt-clay composition of sediments. He considered the sediment characteristics and prevailing local currents the dominant influences on distribution and abundance of fauna, but also considered sediment xenobiotic loading a modifying factor.

Effects: Fish Abnormalities

Chang and Longwell (1984) studied abnormality and mortality in planktonic Atlantic mackerel embryos in the New York Bight. They sampled in May, 1977 and 1978. Results of multivariate analysis revealed that aromatic hydrocarbons and salinity associated with the early development stages of the mackerel as primary variables. Chlorinated compounds and temperature associated with later development stages of the embryos. "However, in the case of temperature and PCBs, the correlation of abnormality, mortality and development delay with temperature is weaker than it is with PCBs" (Chang and Longwell, 1984).

Sinderman, et al. (1982) declared that early life stages (early embryo and late larval stages) and gonadal tissues of adult fishes and shellfishes are particularly sensitive to pollutants. The organic contaminants of particular importance are PCBs, DDT and its metabolites, and petroleum hydrocarbons. They were concerned about the potential impact of pollutants within New York Bight and adjacent estuarine areas on reproduction in the large number of fish species which spawn there.

Smith, et al. (1979) observed grossly visible abnormalities in livers of Atlantic tomcod (Microgadus tomcod) they collected from the Hudson River estuary. Histopathological examinations revealed a 25 percent frequency of hepatocellular carcinomas in the 1977-78 spawning population.

Klauda, et al. (1981) measured PCB residues in liver, gonad, and body tissue of adult Atlantic tomcod they collected in January and February, 1978 from the same spawning populations Smith, et al. (1979) studied and investigated the speculation by Smith, et al. (1979) that PCBs may play a role in the development of hepatomas in tomcod. Klauda, et al. (1981) compared PCB levels from four types of livers - normal, and with each of three categories of abnormality. They discerned no association between PCB concentration and type of liver abnormality. They learned of evidence, however, that tomcod livers which externally appear normal or "hemorrhagic" (one of their three categories of abnormality) may eventually develop neoplastic nodules and hepatocellular carcinomas. Thus they suspected "normal livers may be rare in the Hudson River population of Atlantic tomcod."

Pearce (1979b) described reports from several studies of fish disease in Raritan Bay and the New York Bight Apex. In July-August, 1967, 70 percent of 1,152 bluefish examined were diseased. Scientists found a 15 percent prevalence of fin erosion in 451 winter flounder from Raritan Bay in March-May, 1973; they found only 2.2% of 480 winter flounder from Great Bay in central New Jersey afflicted with fin erosion.

Murchelano and Ziskowski (1982) presented results of research on the prevalence of fin rot disease of New York Bight winter flounder, Pseudopleuronectes americanus during the five year period 1973-1977. They found significant differences ($P<0.05$) in disease prevalence among three areas - the Apex, Sandy Hook/Raritan Bays, and the control sites (Great Bay, N.J. and offshore waters of the Bight) - in three of the years (1973, 1974, 1977). "The Apex had the highest prevalence of the disease (7.4%) for the five year period..." the year of highest disease prevalence was 1973. Following are percents of winter flounder collected having fin rot in 1973 in each area:

Apex	15.0%
Sandy Hook/Raritan Bay	8.0%
Control area	2.1%

Study results showed that disease prevalence in all areas declined after 1973 even though the volume of sewage sludge and dredged materials dumped into the Bight did not change much during the study period. Murchelano and Ziskowski felt that fin rot would not reduce the marketability of the winter flounder since the disease does not affect muscle tissue. They considered presence of fin rot disease in Bight winter flounder an indicator of environmental stress.

Results of a study of fin erosion of winter flounder from the New York Bight Apex and Sandy Hook/Raritan Bay region and of flatfishes from Puget Sound and southern California reported by Sherwood (1982) suggest that exposure of susceptible species to PCB contaminated sediments contributes to the development of fin rot. In the study conducted in 1976-1977, they compared levels of contaminants in diseased flatfishes with levels in apparently healthy flatfishes from New York, southern California, and Puget Sound. In all three regions "total PCBs in muscle, liver, and brain tissue were higher in fishes with fin erosion from the contaminated areas than in apparently unaffected specimens from the control areas." Sherwood (1982) reiterated the point from Murchelano and Ziskowski (1982) that fin rot disease signals the existence of a degraded environment. Sherwood did not consider the fin condition a threat to public health, but she did caution that consumption of fishes with eroded fins may be a problem in that they are likely to have elevated levels of contaminants, possibly exceeding FDA guidelines for total DDT and/or PCBs.

Effects: Mollusk and Crustacean Abnormalities

C. Austin Farley of the NMFS Northeast Fisheries Center in Oxford, Maryland gave a presentation on October 30, 1985 at the 1985 Interstate Seafood Seminar in Ocean City, Maryland entitled "Epizootiology of Presumed Infectious Sarcomas in Chesapeake Bay Soft Clams, *Mya arenaria*." Sarcomas, he defined in his presentation, are non-epithelial cancer type lesions. Sarcomas, carcinomas (epithelial lesions), and leucemia are all generally malignant neoplasms. Farley stated in his presentation, "There doesn't seem to be any relationship with pollution and this disease." He defended that statement with evidence from Boston Harbor and Raritan Bay where there are high levels of benzo(a)pyrene. Farley looked at thousands of mollusks from those areas and from other east coast areas and found no sarcomas in them. He later elaborated on research results concerning mollusks in Raritan Bay (Farley, Personal Communication, November 1, 1985), reporting results of sampling at Ward's Point in Raritan Bay, near Staten Island. Even though there is a lot of pollution there, he found no neoplasms in the mollusks. When Farley has found mollusks with sarcomas, such as in soft clams from Chesapeake Bay, he has not seen a relationship to pollution, though he is willing to leave the possibility open.

Gopalan and Young (1975) collected shrimp samples from stations in the New York Bight and from one station in western Long Island Sound from May-October, 1972 and 1973. They found the incidence of shrimp shell disease as high as 30 percent in certain localities in the Bight. They found that 30.8 percent of the Raritan Bay shrimp, 25 percent of the shrimp from the New York Bight apex, about 9.4 percent of the shrimp from stations in Sandy Hook Bay, and only 7.0 percent of the shrimp from the long Island Sound station were afflicted with shell disease. "Diseased specimens were rarely encouuntered in 48 shrimp collected at Beaufort, North Carolina and 200 at Woods Hole, Massachusetts, areas without substantive pollution" (Gopalan and Young, 1975). Although these data suggest a possible relationship between pollution and shell disease, results of laboratory experiments by Gopalan and Young suggest a bacterial origin and contagious nature for the shell disease. An antibiotic medium (artificial sea water with tetracycline hydrochloride) greatly inhibited the rate of infection. In natural sea water in an aquarium, half of the initially healthy shrimp became infected; yet none of the healthy shrimp reared in the artificial medium became infected. Also the antibiotic in the artificial sea water appeared to retard the progression of the disease in infected shrimp.

Young and Pearce (1975) discussed the effect of solid waste disposal in the New York bight on lobsters, <u>Homarus americanus</u>, and crabs, <u>Cancer irroratus</u>, inhabiting the vicinity of the dump grounds. The animals they collected from the vicinity of sewage sludge and dredge spoil areas most frequently showed pathological conditions such as exoskeleton erosions and gill abnormality. Young and Pearce also collected crabs and lobsters from clean areas and held equal numbers for six weeks in aquaria containing clean sand and in aquaria containing sludge and spoils bottoms. Crustaceans in aquaria containing sediments contaminated with sewage sludge and dredge spoils all developed similar erosions. Gills of the crustaceans in those aquaria became abnormal. Lobster gills were fouled with granular material and filaments had a dark brown coating. Crab gills developed a brown coating and distorted cuticle during the six week exposure to contaminated sediments. These abnormal conditions did not appear in animals held over clean sand, or in animals collected from clean areas. Disease in these crustaceans, Young and Pearce think, may develop in response to low oxygen concentrations and/or high bacterial concentrations in the contaminated sewage sludge and dredge spoil sediments.

Effects: Sediment Toxicity

Zitko and Boehm present "threat ratings" of various organic compounds in Chapter 5 of Breteler (1984) which are akin to sediment toxicity data, but were apparently not calculated from results of experiments exposing biota to New York Harbor sediments. Rather they used toxicity data from bioassays establishing toxicity of individual compounds to marine biota and information from the literature estimating toxicity of these compounds to humans. They multiplied their "toxicity rating" value (using a scale of 0-5, with 5 representing highest toxicity) by their sediment concentration rating (also using a scale of 0-5) to determine "threat ratings." They calculated sediment concentration rating, also called "abundance rating" from data in MacLeod, et al. (1981) about the New York Bight including Lower New York Bay sediments. Resulting ratings for PCBs, dioxins and some pesticides follow:

Compound	Toxicity rating		Sed. conc. rating	Threat rating	
	Humans	Marine Biota		Humans	Marine Biota
PCB 1016 + 1242	4	3	4	16	12
PCB 1254	4	4	5	20	20
PCB 1260	4	4	4	16	16
Polychlorinated naphthalenes	4	4	1	4	4
Chlorinated dibenzodioxins (2, 3, 7, 8 TCDD)	5	5	1	5	5
Heptachlor	4	3	2	8	6
Heptachlor epoxide	4	3	3	12	9
Aldrin	4	4	1	4	4
DDT and metabolites	3	4	4	12	16
Chlordane	3	4	4	12	16
Trans-nonachlor	3	4	4	12	16
Dieldrin	4	4	3	12	16
Endrin	4	5	2	8	10
Mirex	4	4	1	4	4

Kepone	4	4	1	4	4
Toxaphene	4	4	1	4	4
Endosulfan	4	4	1	4	4
2,4-D	2	1	1	2	1
2,4,5-T	3	1	1	3	1

They classified the following halogenated hydrocarbons which received a "threat rating" of 10 or above in the category of "major perceived threat":

To humans	To Aquatic Invertebrates
PCB-1016	PCB-1016
PCB-1254	PCB-1254
PCB-1260	PCB-1260
DDT and metabolites	DDT and metabolites
Chlordane	Chlordane
Trans-nonachlor	Trans-nonachlor
Dieldrin	Dieldrin
Lindane	Endrin
Heptachlor epoxide	

Zitko and Boehm cautioned against exclusive use of acute toxicity values which may be misleading. As an example they mentioned investigations demonstrating the chronic toxicity of PCBs to a wide variety of organisms. Primarily these investigations indicated significant decrease in reproductive success resulting from prolonged exposure to PCBs. They considered acute and chronic toxicity as well as carcinogenicity in assigning toxicity ratings to compounds.

J.M. Neff in Chapter 7 of Breteler (1984) presents "threat ratings" for total hydrocarbons, alkanes, and PAHs of the Hudson-Raritan estuary, calculated in a manner similar to calculations for "threat ratings" described for halogenated hydrocarbons in Chapter 5 by Zitko and Boehm, i.e. the product of a "toxicity rating" and an "abundance rating." His "threat rating" to aquatic biota for total hydrocarbons was 8-12 placing them in the "major perceived threat" category. "Threat rating" of total hydrocarbons to humans was 4-8, classifying them a "potential significant threat" for humans. Alkanes received a "threat rating" of 2-3 indicating "no perceived threat" to either aquatic organisms or humans. Based on rather high concentrations of PAHs in sediments and existance of some commercially important benthic invertebrates in the Hudson-Raritan estuary along with a toxicity rating of 3, based on acute toxicity of low molecular weight PAHs, "threat rating" of PAHs to estuarine biota was placed at 9-12, thus considered "a major perceived threat" to them.

"The main threat to human health of PAH is their well-known mammalian carcinogenicity. Because carcinogenic PAH occur at relatively high concentrations in the estuarine system, including the tissues of exploited fish and shellfish species, the human toxicity rating was placed at 4" (Breteler, 1984). The resulting "threat rating" for PAH in the estuary to man was 12-16, considered "a major perceived threat."

Impacts: History

Pearce (1979a) gave historical examples of pollution problems. He said Raritan Bay was the most heavily polluted of the major embayments of the northeasterm United States and that it had been polluted along with estuarine portions of the rivers for decades. His examples included a problem of coal oil tainted oysters and shad at the time of the Civil War, water quality problems necessitating closure of the New York Aquarium in 1912 and a problem of industrial wastes threatening the existance of commercial oysters during World War I. He discussed disease and low dissolved oxygen as two causes of decreases in populations. These parameters in some cases may be related to organic pollution. Pearce noted "accumulations of organic materials... can be suspected to have locally resulted in reduced DO in Raritan Bay and the contiguous coastal zone." He further directed attention to the double threat of organic matter which accumulates in the sediments off the Hudson-Raritan estuary. This matter is dredged from shipping channels, then dumped at offshore sites. It affects the benthic habitat first during accumulation in harbors and estuaries and second when it is dumped in offshore coastal environments.

Impacts: Fisheries and Human Health

Hydroscience, Inc. (1978) conducted a project to estimate possible effects of remedial action, primarily of various upstream dredging schemes for reducing PCB sources in the Hudson, on the Hudson River ecosystem, and especially the fishery there. They reported that effects would vary with species, location, and age of fish. For example, older fish may retain higher body burdens due to their lower excretion rates. They estimated extensive upstream reductions of PCB inputs to the estuary might reduce body burdens on larger fish by about 30-50 percent; but the levels might still be close to the 5,000 ppb FDA action level at the time and above the proposed 2,000 ppb action level. (That proposed FDA action level for PCBs in fish went into effect in 1984). So, the fishery might be in a marginal condition. Their analysis indicated that if no action were taken, more than a decade might be required before substantial improvement would be noticed in PCB concentrations of the fishery especially the striped bass. "Under extensive upstream dredging, reduction in fish PCB body burden may result after a period of several years to about less than a decade" (Hydroscience, Inc. 1978).

Larry Skinner from New York State Department of Environmental Conservation's Division of Fish and Wildlife spoke about policies in New York State concerning chemical contaminants in fish (Skinner, October 30, 1985). Impacts are seen in New York in the form of health advisories, fisheries closures, and fisheries regulations. Some closures resulting from knowledge of high levels of organic contaminants in fish include the following: upper Hudson within a 50 mile reach (PCB), lower Hudson, commercial striped bass (dioxin), and marine waters of western Long Island and New York Harbor (PCB). Actually fish in the eastern portion of Long Island have PCB levels nearly as high as those in the western portion. Levels of 3,000 ppb (above the new FDA action level) are found there, yet fisheries are still open. The decision regarding drawing the line between east and west Long Island was arbitrary and residents of the area are confused and some (those that live and like to fish on the side in which closure is in effect) are angry about it .

Belton, et al. (Oct., 1985) reported on a study of dioxin contamination of sediments, fish and crustaceans in waterways of New Jersey by the Office of Science and Research within the New Jersey Dept. of Environmental Protection. Results of this study led Commissioners of the New Jersey Department of Environmental Protection and Health in August of 1984 to order "prohibition on the sale and consumption of all fish and shellfish taken from the tidal Passaic River." They extended that ban to include striped bass and blue crabs taken from Newark Bay and some adjoining waterways. They decided information about lobsters was not sufficient to make a conclusion about health risks from their consumption.

Using an EPA method of human cancer risk assessment based on results of a study of effects of PCB 1260 on rats along with their own data on tissue concentrations of PCB in fish and crabs collected in New Jersey water, Belton, et al. (Sept., 1985) calculated that lifetime daily consumption of 36.8 grams of fish contaminated with 2 ppm (2,000 ppb) PCB would result in approximately 450 cancers per 100,000 people. More specifically they concluded that "37 persons in a population of 10,000 who ate 15.7 grams of Hudson River or Newark Bay striped bass per day over a lifetime might develop some form of liver cancer." This excessive risk, over the typical benchmark acceptable risk of one cancer/million people, was the basis for New Jersey's regulatory responses of commercial fisheries closures for striped bass in affected areas and of issuing warnings to the public against consumption of contaminated fish. Belton, et al. (Sept., 1985) also gave results of a "Fisherman Survey." Fishermen's risk perception revealed in survey results were interesting. Only 35 percent thought fish in the New York-Newark Bay area were not safe to eat and 31 percent of those who consumed the fish thought their catch was contaminated, but ate it anyway!

Ruppel (undated flier) discussed adverse health effects of PCBs including nervous system disorders, reproductive difficulties, skin rashes, swollen joints, gum discoloration, eye discharges and lethargy (symptoms that developed in people ingesting PCB contaminated rice oil in Japan in 1968) and illustrated fish filleting, skinning, and cooking technique designed to reduce health risk in fish consumption. He added that the FDA recently determined that "the major source of PCB exposure is through food ingestion, with fish being the singly most important dietary contributor.

In Chapter 6 of Breteler (1984) M.S. Conner, C.E. Werme, and K.D. Rosenman discussed the public health consequences of organic compounds in the Hudson-Raritan estuary. Table 4 containing Tables 27 and 28 of Breteler (1984) shows carcinogenicity of PAH, PCB, and certain pesticides, concentrations of these contaminants in fish and invertebrates and human cancer risk from consumption of winter flounder, windowpane flounder, lobster, and mussels from Raritan Bay and striped bass from the Hudson River.

Table 27. Carcinogenic potency factor and wet weight concentrations of contaminants in individual fish or shellfish from the Hudson-Raritan Estuary [a]

Compound	Carcinogenic potency factor	Wet weight concentrations (ppb) from muscle: Raritan Bay winter flounder			Raritan Bay windowpane flounder			Hudson River striped bass			Raritan Bay lobster					Raritan Bay mussels
Chlordane	1.61	8.0	8.0	8.0	6.0	3.8	6.3	144	144	168	4.8	1.7	2.3	1.1	1.1	10.4
DDT and metabolites	8.42	16.0	10.0	9.0	22.0	13.1	18.9	792	1392	504	11.2	10.2	29.9	20.2	20.2	66.3
Dieldrin	30.40	0	4.0	4.0	0	3.8	4.2	0	0	0	6.4	5.1	0	2.2	2.3	0
Hexachlorobenzene	1.69	0	0	0.6	0	0.4	0.4	0	0	0	0.3	0.3	1.1	1.1	0.9	0
Heptachlor	3.37	0	0	0	0	0	0	0	0	0	0.8	0.7	0	0.2	0.1	0
Lindane	0.78	0	2.0	1.2	0	0	0.8	0	0	0	0	0	0	0	0.2	0
Nonachlor	0 [b]	4.0	6.0	6.0	4.0	1.9	4.2	192	192	144	3.2	3.4	4.6	2.2	2.3	6.5
PAH [c]	0	6.0	8.0	10.0	12.0	14.4	13.4	10	5	0	12.8	25.5	77.7	6.6	36.8	250.9
PCB	4.34	100	80	80	160	76	126	12960	8880	3120	96	357	230	220	207	156

[a] Data from MacLeod et al. (1981).
[b] No experimental evidence of carcinogenicity.
[c] PAH included naphthalene, 1-methylnaphthalene, biphenyl, phenanthrene, fluoranthrene, pyrene, chrysene, and benz(a)anthracene. No experimental evidence has linked any of these compounds to cancer.

Table 28. Lifetime increased risk of cancer ($x10^{-6}$) from consuming 6.5 grams of Hudson-Raritan fish or shellfish per day

Compound	Raritan Bay winter flounder			Raritan Bay windowpane flounder			Hudson River striped bass			Raritan Bay lobster					Raritan Bay mussels
Chlordane	1.2	1.2	1.2	0.9	0.6	0.9	22	22	25	0.72	0.25	0.34	0.16	0.17	1.6
DDT and metabolites	13	7.8	7.0	17	10	15	620	1100	390	8.8	8.0	23.39	16	16	52
Dieldrin	0	11	11	0	11	12	0	0	0	18	14	0	6.2	6.5	0
Hexachlorobenzene	0	0	0.1	0	0.1	0.1	0	0	0	0.05	0.05	0.18	0.17	0.14	0
Heptachlor	0	0	0	0	0	0	0	0	0	0.25	0.21	0	0.07	0.04	0
Lindane	0	0.14	0.09	0	0	0.1	0	0	0	0	0	0	0	0.02	0
Nonachlor	0	0	0	0	0	0	0	0	0	0	0	0	0	.00	
PAH	0	0	0	0	0	0	0	0	0	0	0	0	0	.00	0
PCB	40	32	32	64	31	51	5200	3600	1300	39	140	93	89	84	63
Additive risk	54	53	52	83	52	78	5900	4700	1700	67	170	120	110	110	120

Table 4. Tables 27 and 28 (Breteler, 1984).

Reid, et al. (1982) did not seem as concerned about the impact of organic contaminants in their reported area of study, the New York Bight and Long Island Sound, on human health as other investigators (Breteler, 1984; Belton, et al., Sept., 1985 and Oct., 1985; Skinner, Oct. 30, 1985) who reported on conditions in Newark Bay and the Hudson River - Raritan Bay estuary. "Contaminants in demersal biota did not appear to be in concentrations sufficient to cause concern for human health; [although] body burdens were elevated compared to reported concentrations outside the Bight" (Reid, et al., 1982). They added, though, that there is uncertainty about ecological effects.

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NARRAGANSETT BAY

Narragansett Bay is a relatively large estuary in our smallest state, Rhode Island. It is "a temperate, well-mixed, plankton-based estuarine ecosystem" (Wakeham, et al., 1982) and an "area of high recreational and commercial activity" (Lopez-Avila and Hites, 1980). The state capital, Providence, is located on the Providence River at the head of Narragansett Bay. Sewage treatment effluents from outfalls at Fields Point and two other locations are a constant source, and oil spills are occasional sources of organic contamination of Narragansett Bay (Van Vleet and Quinn, 1978; Quinn, 1979). Urban runoff is a major source of organic pollution (Hoffman, et al., 1983, 1984). According to Hoffman (1985) oil spills were a minor part of the total oil pollution budget for Narragansett Bay and its tributaries. A map of Narragansett Bay is shown in Figure 1.

Organic Contaminants in Water and Sediments

For two summers (1979 and 1980) and two winters (1980 and 1981) Wakeham, et al. (1982) sampled water from Narragansett Bay, then analyzed the samples for volatile organic compounds. They identified fifty seven of these major compounds. Benzene and toluene were the dominant aromatic hydrocarbons they found. Distribution along the north-south transect in the bay indicated that industrial and municipal effluents discharged into the head of the bay were the major sources, especially of chlorinated and aromatic hydrocarbons. Table 1 shows data for the combined concentrations of the PAHs naphthalene, 2-methyl naphthalene, and 1-methyl naphthalene in various surface and subsurface water samples from sites in Narragansett Bay. The only apparent geographical trend is that highest concentrations came from the outfall site. "High concentrations [of aromatic hydrocarbons] in the Fields Point outfall area and at Conimicut Point in winter are consistent with uses as industrial solvents and subsequent incorporation into industrial wastewaters" (Wakeham, et al, 1983). No long term temporal trend is obvious from these data, but concentrations tended to be higher in winter than in summer. Wakeham, et al. (1983) explained that in summer aromatic hydrocarbons are degraded in a few days.

McGregor (1983) found evidence suggesting that volatilization was the main process removing volatile organic compounds from Narragansett Bay water, followed by "a small but recognizable contribution" by biodegradation and an insignificant removal by sedimentation or chemical degradation. Results of McGregor's sampling in 1980 and 1981 and analysis of water samples disclosed the following major components of the volatile organic compounds in Narragansett Bay: benzene, trichloroethylene, toluene, tetrachloroethylene, chlorobenzene, ethylbenzene, xylenes, chlorotoluene, C3 alkyl benzenes. McGregor's data also indicated maximum values for these contaminants "were more likely to be found in the winter than in the summer." He noted a steep decline in volatile organic compound concentrations in the water in the spring of 1981 coinciding with a series of strong storms, and suggested that volatilization was responsible. Figure 2 illustrates seasonal trends in monocyclic aromatic hydrocarbon concentrations in various locations in Narragansett Bay and their

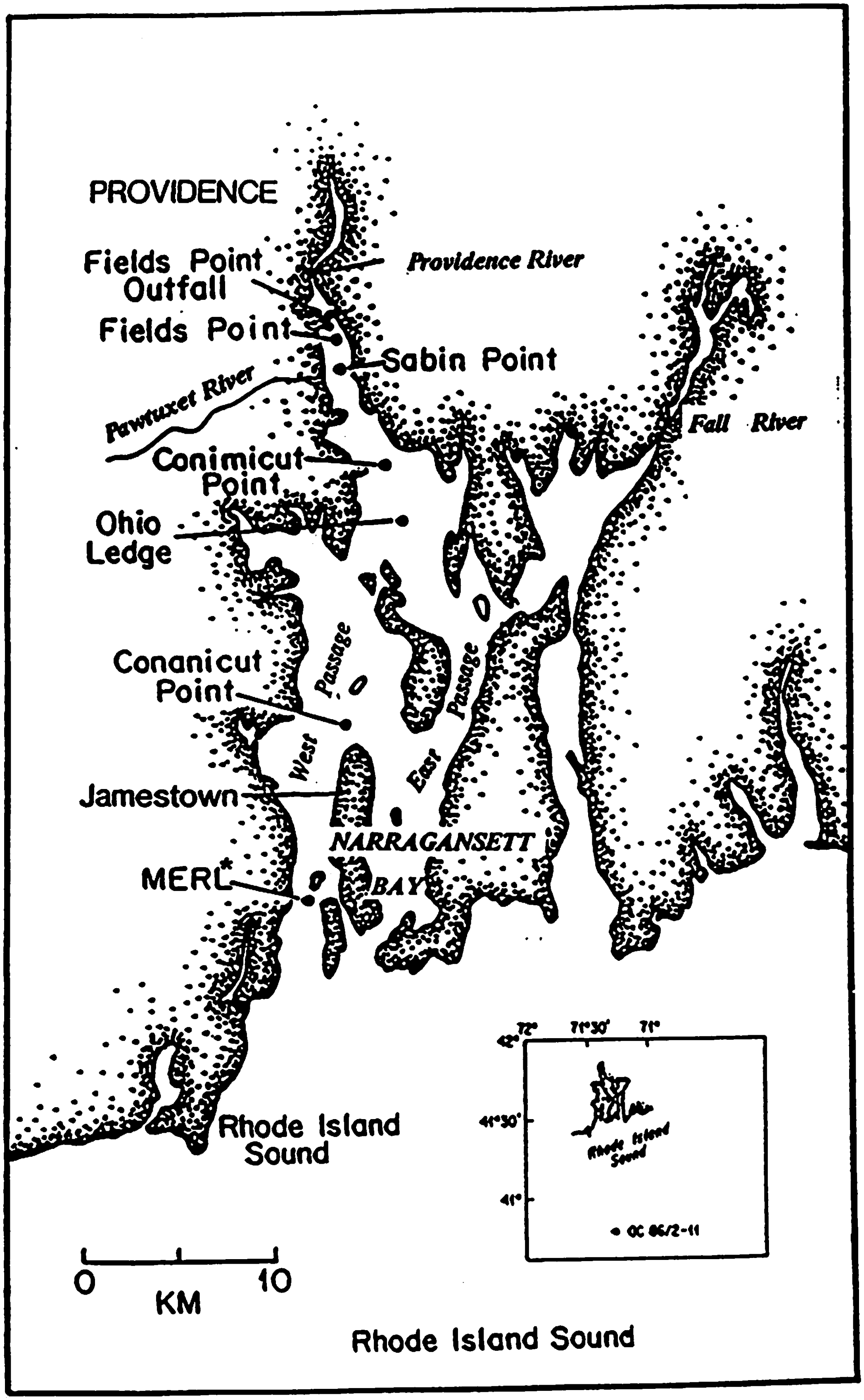

Figure 1. Narragansett Bay, Rhode Island (Wakeham, et al., 1982).
*Marine Ecosystems Research Lab, URI

Table 1. Mean concentrations (ng/Liter, or parts per trillion) of Σ naphthalenes (naphthalene, 2-methyl naphthalene, and 1-methyl naphthalene) in water samples from Narragansett Bay sites from Providence River to the mouth (Wakeham, et al., 1982).

Location	Concentration	Sampling Date
Fields Point, 0m	4	Aug., 1979
Sabin Point, 0m	3	Aug., 1979
Sabin Point, 4-10m	2	Aug., 1979
Conimicut Point, 3-3.5m	2	Aug., 1979
Conanicut Point, 0m	1	Aug., 1979
Conanicut Point, 6-8m	1	Aug., 1979
MERL, 0m	1	Aug., 1979
MERL, 8-13m	2	Aug., 1979
Providence River, 0m	32	Feb., 1980
Providence River, 4m	53	Feb., 1980
Fields Point Outfall	590	Feb., 1980
Fields Point, 0m	70	Feb., 1980
Fields Point, 3-7.5m	40	Feb., 1980
Sabin Point, 0m	87	Feb., 1980
Sabin Point, 4-10m	19	Feb., 1980
Conimicut Point, 0m	82	Feb., 1980
Conimicut Point, 3-3.5m	48	Feb., 1980
Conanicut Point, 0m	36	Feb., 1980
Conanicut Point, 6-8m	17	Feb., 1980
MERL, 0m	20	Feb., 1980
MERL, 8-13m	13	Feb., 1980
Providence River, 0m	6	July, 1980
Fields Point Outfall	650	July, 1980
Fields Point, 0m	17	July, 1980
Fields Point, 3-7.5m	4	July, 1980
Sabin Point, 0m	5	July, 1980
Sabin Point, 4-10m	6	July, 1980
Conimicut Point, 0m	8	July, 1980
Conimicut Point, 3-3.5m	4	July, 1980
Conanicut Point, 0m	<3	July, 1980
Conanicut Point, 6-8m	<3	July, 1980
MERL, 0m	1	July, 1980
MERL, 8-13m	<3	July, 1980
Providence River, 0m	13	Feb., 1981
Fields Point Outfall	1,100	Feb., 1981
Fields Point, 0m	28	Feb., 1981
Fields Point, 3-7.5m	48	Feb., 1981
Sabin Point, 0m	29	Feb., 1981
Sabin Point, 4-10m	43	Feb., 1981
Conimicut Point, 0m	64	Feb., 1981
Conimicut Point, 3-3.5m	14	Feb., 1981
Conanicut Point, 0m	35	Feb., 1981
Conanicut Point, 6-8m	15	Feb., 1981
MERL, 0m	15	Feb., 1981
MERL, 8-13m	17	Feb., 1981

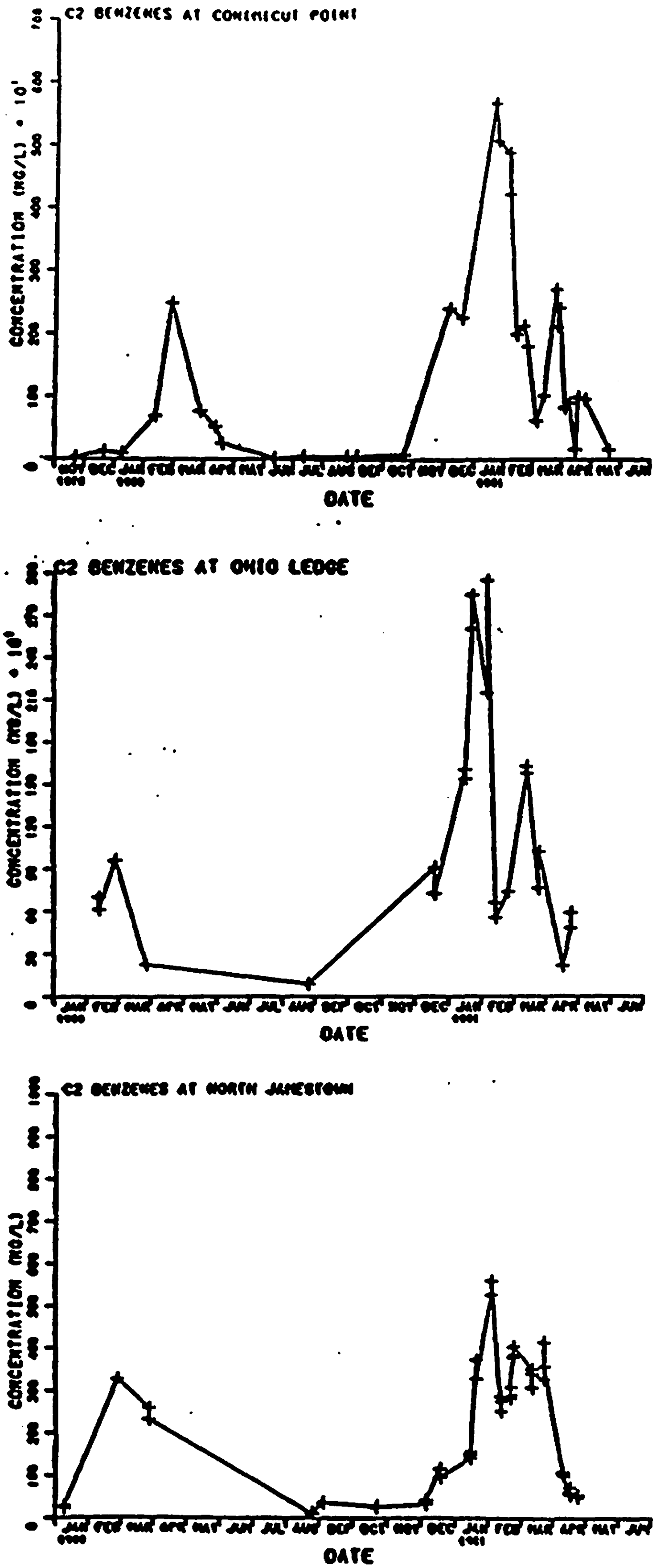

Figure 2. Total C2 Benzenes from 1 meter depth H_2O at Conimicut Point, Ohio Ledge, and North Jamestown in Narragansett Bay (McGregor, 1983).

fluctuations during McGregor's sampling period. It contains McGregor's plots of total C2 benzenes from November, 1979 to June, 1981 at Cominicut Point, Ohio Ledge, and North Jamestown.

Hoffman, et al. (1983) collected urban runoff samples during several storm events with different rainfall amounts from storm drains serving different land uses in the Narragansett Bay drainage basin, and analyzed them for petroleum hydrocarbons. They used those hydrocarbon load data and rainfall records to calculate the load for each storm and were able to predict an annual input rate of hydrocarbons to receiving waters in terms of mass hydocarbon per area for each land use. They predicted an annual input of 665 tons of hydrocarbons to Narragansett Bay from urban runoff; urban runoff is obviously a major source of hydrocarbons. The highest loads, ~445 tons or 66.8% of the total, came from industrial land. Interstate highway land gave ~153 tons or 22.9% of the total; commercial and residential land combined followed giving ~68 tons or 10.2%.

Hoffman, et al. (1984) compared urban runoff to other sources of PAHs entering Narragansett Bay and determined that urban runoff accounted for 71% of the total inputs for higher molecular weight PAHs and 36% of the total PAHs. "The loads of PAHs. . .in urban runoff were higher at the highway and industrial land uses in comparison to the commercial and residential areas" (Hoffman, et al., 1984).

Quinn (1979) reported results of a five year study of sources of hydrocarbon pollution in Narragansett Bay. "Sewage effluent discharged into the Providence River at the head of the bay proved to be the major source of such [anthropogenic] hydrocarbons in the water, sediment and hard clams of Narragansett Bay" (Quinn, 1979). Values predicted via a hydrodynamic model developed by University of Rhode Island investigators and values obtained via actual water sample analysis both showed decreasing particulate hydrocarbon concentration from Providence River to the mouth of Narragansett Bay.

Quinn also reported results of sediment core analysis from stations from Providence River to the mouth of the bay. Just as with particulate matter, hydrocarbon concentration in surface sediments decreased by a factor of 100 from the river to the lower bay. At seventeen of their twenty stations anthropogenic hydrocarbons decreased with depth of the core. "At a station adjacent to the Fields Point outfall, the anthropogenic hydrocarbons in the core were very concentrated in the surface section. . .relatively high values were found down to the deepest section of the core (40cm) which represented a time period of at least 40 years" (Quinn, 1979).

Hurtt and Quinn (1979) also reported "a decrease in surface sediment hydrocarbons from Providence River to the mouth of the bay and the concentrations also decreased with depth in the cores,. . .leveling off at 20-25 cm." They thought this depth was probably related to increased use of petroleum at the end of the 19th century. The data reported in Quinn (1979) indicate that the Providence River is the major source of anthropogenic hydrocarbons in Narragansett Bay sediments.

Lopez-Avila and Hites (1980) investigated contamination of water and sediment in the Pawtuxet River, Pawtuxet Cove, the Providence River and the Narragansett Bay by wastewater from a small specialty chemicals manufacturing plant on the Pawtuxet River. They made several hundred quantitative measurements from field sampling in 1977 and reported summary data for eleven organic compounds that occurred throughout the river and bay system or were present at relatively high concentrations. They found that aqueous concentrations of the various compounds followed rules of simple dilution. They also made some generalizations about prediction of sedimentary behavior of these industrial organic compounds. They calculated the maximum distance which compounds discharged from the plant would reach. Using an hypothetical case of (a) a compound concentration of <5000 ppb in the wastewater, (b) the detection limit of their analysis, and (c) a maximum octanol-water partition coefficient (log P) value of 8 of any discharged compound, the maximum distance from the discharge point would still be within Narragansett Bay.

Following is a summary of sediment concentration data for three of the eleven compounds reported by Lopez-Avila and Hites (1980) at various distances from the plant:

C_1-benzotriazole:	Pawtuxet River, near plant	500,000 ppb
	Pawtuxet River, mid river, (1)km from plant	800,000 ppb
	Pawtuxet River, near dam, (2)km from plant	40,000 ppb
	Pawtuxet Cove, (3)km from plant	200,000 ppb
	Providence River, (5)km from plant	10,000 ppb
	Narragansett Bay, (15)km from plant	not detected
C_{10}-benzotriazole:	Pawtuxet River, near plant	300,000 ppb
	Pawtuxet River, mid river, (1)km from plant	300,000 ppb
	Pawtuxet River, near dam, (2)km from plant	70,000 ppb
	Pawtuxet Cove, (3)km from plant	100,000 ppb
	Providence River, (5)km from plant	10,000 ppb
	Narragansett Bay, (15)km from plant	600 ppb
chlorobenzotriazole:	Pawtuxet River, near plant	300,000 ppb
	Pawtuxet River, mid river, (1)km from plant	400,000 ppb
	Pawtuxet River, near dam, (2)km from plant	20,000 ppb
	Pawtuxet Cove, (3)km from plant	80,000 ppb
	Providence River, (5)km from plant	20,000 ppb
	Narragansett Bay, (15)km from plant	500 ppb

Wade and Quinn (1979) studied distribution of PCBs and pesticides in sediments from mid-Narragansett Bay. They collected core samples at north Jamestown (near Conanicut Point) in November, 1976 and April, 1977. Concentrations of these synthetic chlorinated compounds decreased with depth in the sediment cores, but reached background levels at different depths. Also more hydrocarbons were associated with smaller size fraction sediments, at least in surface sections. "These trends suggest that over the time period covered by these cores the inputs of biogenic materials has remained relatively constant, while the input of anthropogenic hydrocarbons has increased dramatically during the last 100 yr. This increase is probably due to the expanded use of petroleum over this time period and subsequent chronic inputs to this estuarine environment" (Wade and Quinn, 1979). Following are concentrations in ppb, dry wt. of total PCBs, of pp-DDT, and of chlordane from various depths of one of Wade and Quinn's Narragansett Bay sediment cores:

sediment cores:	PCBs	ppDDT	Chlordane
0-4cm	39.2	0.14	1.16
4-6cm	8.4	0.05	0.29
6-8cm	3.4	0.00	0.01
8-10cm	5.7	0.04	0.00
10-12cm	1.9	0.00	0.00
12-14cm	0.7	0.00	0.00

Organic Contaminants in Biota

Farrington, et al. (1983) presented an overview from data for PCBs, aromatic hydrocarbons, and other contaminants in mollusks collected in the U.S. Mussel Watch program in 1976 - 1978 from U.S. east and west coast locations. They noted that "similarities in geographical distributions of concentrations were present in all 3 years with at least an order of magnitude elevation of concentrations of . . . PCBs, and fossil fuel hydrocarbons in bivalves sampled near the larger urban areas." Figure 3 shows temporal fluctuations in PCB and aromatic hydrocarbon concentrations in mussels, *Mytilus edulis*, from Narragansett Bay as reported by Farrington, et al. (1983).

Pruell, et al. (1984) gave results of analysis of hard clams, *Mercenaria mercenaria*, harvested from Narragansett Bay and found in seafood stores in May and August, 1979, and of hard clams collected from a relatively unpolluted site in lower Narragansett Bay which served as controls. Generally levels of total hydrocarbons and of PAHs in clams from the lower Narragansett Bay control location were lower than those in the clams from seafood stores. They reported a mean concentration value for total identified PAHs (phenanthrene, fluoranthene, benz(a)anthracene, pyrene, chrysene and triphanylene) in clams from seafood stores as 6.3 ± 5.0 ppb, wet weight (approx. 31.5 ppb, dry wt.). They concluded that their study illustrated that contaminants discharged to Narragansett Bay may make their way into commercial outlets.

Quinn (1979) discussed results of analysis of hard clams, *Mercenaria mercenaria*, from several areas of Providence River and Narragansett Bay. He observed a trend of decreasing concentration of anthropogenic hydrocarbons in the clams with increasing distance from the river. He predicted that at least a year of depuration would be required to reduce concentrations of anthropogenic hydrocarbons accumulated in river clams to levels found in clams from the lower bay.

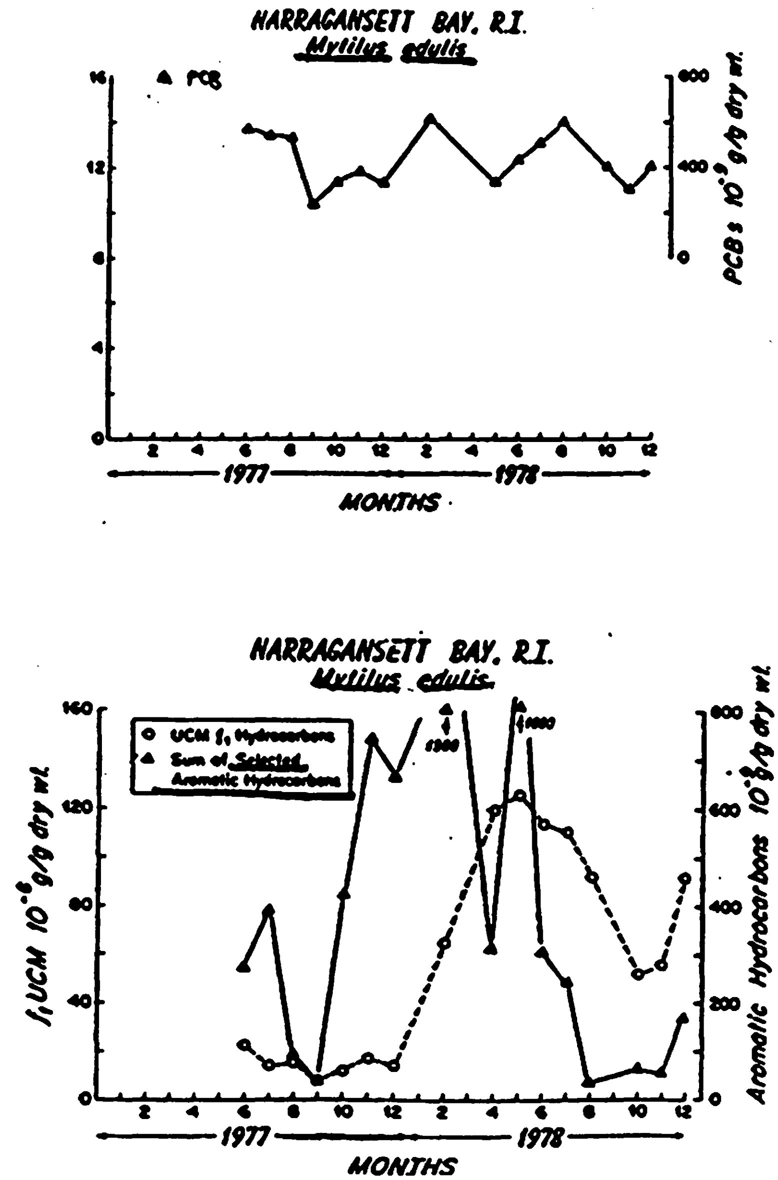

Figure 3. Temporal fluctuations of chemical concentrations in Mytilus edulis from Narragansett Bay, RI. 1977 and 1978: PCBs and hydrocarbons (Farrington, et al., 1983).

Effects of Contaminants: Populations

O'Conner (1986) summarized some results from one recently completed NOAA supported study (soon to be published) which sought correlations between fishery stock sizes (from historical records) and contaminant inputs to coastal and estuarine waters. The summary offered reason to suspect that human activities other than fishing have affected stock sizes of fish and shellfish. There was also an implication that some factors other than dissolved oxygen (DO) and biological oxygen demand (BOD) could have caused a response in stock sizes. The summary revealed a few cases of positive correlation with human activity in which stock sizs increased following dredging. Some stock sizes varied inversely with some measure of monotonically increasing human activity or trend over the fifty year period, 1928-1978, such as human population or industrial activity. In Narragansett Bay the summarized results showed such a negative correlation for stock sizes of lobsters, striped bass, summer flounder, winter flounder, tautogs, and white perch. O'Conner (1986) cautioned, however, that these results "do not indicate whether contaminants are ever at fault." Investigators in this study were not able to test whether inputs of any particular contaminant correlated with stock sizes in Northeast estuaries.

Saila, et al. (1971) discussed effects of pollution on benthic populations in Providence Harbor at the head of Narragansett Bay. They named important brackish water species which they would have expected to find in an undisturbed environment in the same salinity zone as Providence Harbor, but were missing from the harbor. Considering the absence of these - oysters (*Crassostrea virginica*), polychaetes (*Scolecolepides viridis* and *Pygospio elegans*), mollusks (*Gemma gemma* and *Hydrobia sp.*), gammarid amphipods, and all intertidal bivalves and barnacles - they concluded that "the fauna of Providence Harbor is limited not only by low oxygen concentrations and low and variable salinity, but also by toxic materials in the water and/or sediment."

NARRAGANSETT BAY

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GREAT LAKES

The Great Lake basin of North America drains about 300,000 square miles of land area and supports a population of 40 million people. Pollution problems in the basin gained national attention many years ago with rivers that caught fire and cries that Lake Erie is dead. Early pollution control efforts in the region focused on controlling oxygen depleting wastes and reducing nutrient inputs to control eutrophication. In the late sixties and early seventies concerns for the "health" of the Lakes turned to toxic chemicals, eg. PCBs, DDT and mercury (Nriagu & Simmons, 1984).

The size and complexity of the Great Lakes precludes a comprehensive discussion of the entire region in this report. Instead we will limit our analysis to the major problems within the lakes and a limited number of tributary streams where specific problems related to toxic chemicals have been identified.

Closure of fisheries due to contamination by chlorinated hydrocarbons are without doubt the major concern in the Great Lakes. In addition to these harvesting limitations biological effects on fishes and birds have been linked to these contaminants.

Figure 1 shows 42 areas of concern in the Great Lakes region which have been identified by the Great Lakes Water Quality Board of the International Joint Commission. In all but 4 of these areas one or more toxic substances (organics and/or heavy metals) are implicated as the major problem.

Organic Contaminants in Sediments

Chlorinated Hydrocarbons

Simmons (1984) reviewed the extent of sediment contamination by PCBs in the Great Lakes. Figure 2 shows the distribution of PCBs in surface sediments from the Lakes. The discussion which follows is quoted from the review of Simmons (1984). "In Lake Superior, the highest PCB levels in sediments reported by Frank et al. (1980) were from the Duluth subbasin (8.6 ng/g), the Marathon basin (6.4 ng/g), and Thunder Bay (5.7 ng/g). Eisenreich et al. (1979) also reported highest concentration in sediments from the extreme western end near Duluth-Superior (230 ng/g) and the central part of the lake between the Keweenaw Peninsula and Thunder Bay (290 ng/g)."

"In Lake Michigan, the most highly contaminated areas reported also contain some of the highest PCB levels reported anywhere in the Great Lakes. These areas include Waukegan harbor (500,000 µg/g) and North Ditch (250,000 µg/g), tributary to Lake Michigan. Milwaukee Harbor (6.42 µg/g), the Fox River Basin (190 ng/g), and southern Lake Michigan (188 ng/g) have also been designated as highly contaminated areas (U.S. EPA Region V; Armstrong, 1981; Kleinert, 1976)."

"In Lake Erie, the Western Basin had maximum concentrations of PCB (660 ng/g), and the Central Basin (330 ng/g) and the Eastern Basin (320 ng/g) had lower values (Frank et al., 1977)."

"In Lake Huron, Frank et al. (1979) indicated that Thunder Bay (Alpena, MI) and Saginaw Bay are the areas where PCB concentration was maximum. Burin and Robbins (1977) also reported 3-12 µg/g PCB in Saginaw Bay sediments."

"In Lake Ontario, several areas have been designated as problem areas. Frank et al. (1980) reported 260 ng/g as maximum PCB concentration in sediments from the Bay of Quinte. They also reported greater than 200 ng/g values along the southern shore of the lake implicating the Niagara River as the major source of PCBs to Lake Ontario. The Genesee and Oswego rivers may be important minor sources."

Frank, et al. (1977) reported on residues of PCBs and other chlorinated hydrocarbons in the sediments of Lake St. Clair (1970 and 1974) and in Lake Erie (1971). Residues of DDE, TDE and DDT were highest in sediments from the Western Basin of Lake Erie, 70 ng/g, while the Central and Eastern Basins showed residues of about 25 ng/g. Sediments from Lake St. Clair were least contaminated with levels of 6.6 ng/g in 1970 and 2.6 ng/g in 1974.

Table 1 (from Thomas and Frank, 1983) shows the mean levels of PCBs within the various sectors of the Great Lakes. The data for Lake Erie are from Frank, et al. (1977).

Shear (1984) surveyed the levels of Mirex in surface sediments of Lake Ontario in 1978. Mirex was first reported in Lake Ontario in 1974 in fishes from the Bay of Quinte (Kaiser, 1974). In addition to Mirex, Strachan and Edwards (1984) reviewed the data on a number of other chlorinated organics in Lake Ontario.

Total DDT in surface sediments from Lake Ontario in 1968 averaged 43 ng/g with a high of 218 ng/g (Frank, et al. 1974). Loading of DDT to the lake was estimated at 150 kg/year from atmospheric sources and 120 kg/year via the Niagara River.

Dieldrin and aldrin have also been identified as contaminants of Lake Ontario but at much lower concentrations than other chlorinated hydrocarbons (Great Lakes Water Quality Board, 1985).

Rice and Evans (1984) reviewed the data on toxaphene, another chlorinated hydrocarbon of concern in the Great Lakes. They found that no data were available for concentrations of this pesticide in sediments, with only limited information on levels in biota.

Polychlorinated dioxins and furans have recently been identified as trace contaminants in sediments from several areas of the Great Lakes, including the Niagara and Tittabawassee Rivers (Great Lakes Water Quality Board, 1985). However, information on levels of sediment contamination is too limited to draw conclusions at this time.

Sullivan and Delfino (1982) reviewed information on toxic chemicals in Wisconsin's Lower Fox River Basin. The lower Fox River is one of the largest rivers draining into Lake Michigan via Green Bay and is the most industralized in Wisconsin (Edgington, 1984). Control of conventional pollutants has led to a dramatic improvement in the lower reaches of the Fox River and in the southern end of Green Bay. However, concern over the potential effects of toxic chemicals has arisen from observations of abnormalities in fishes and birds from the river and Bay (Fox, 1982, Gilbertson, 1982).

Table 2 and Figure 3 show the concentrations and sampling locations for PCB levels in sediments from the Fox River and Lower Green Bay (Sullivan and Delfino, 1982).

Eadie (1984) and Hallett and Brecher (1984) reviewed the information concerning the distribution and cycling of PAHs in the Great Lakes.

Table 3 gives estimates of the atmospheric flux of PAHs into the Great Lakes for 7 individual compounds, while Table 4 presents residue data for several compounds in Great Lakes surface sediments. Table 5 provides a more detailed analysis for Lake Michigan sediments and Figure 4 shows the distribution of fluoranthene in surface sediments of Lake Michigan. Eadie (1984) observed that "Lake Superior sediment is an order of magnitude lower in PAH than the lower lakes, and Lake Michigan sediments are highest in PAH concentration."

Table 6 presents a detailed account of PAHs identified in sediments as a function of depth from a station near Toronto in Lake Ontario.

Organic Contaminants in Biota

Residues of pesticides, PCBs and polynuclear aromatic hydrocarbons reported for the region are presented by lake system.

Figure 5 shows the residue data for dieldrin, total DDT and Aroclor 1254 in lake trout from Lake Superior between 1977 and 1982. Figure 6 presents the residue data for organochlorine contaminants in herring gull eggs from two nesting colonies on the lake between 1974 and 1983. Variations in residues within years, i.e. among the samples for a given year, make judgements concerning trends over time difficult. However, some conclusions appear warranted: (1) average residues of total DDT have declined over time in both species; (2) dieldrin levels remained constant in both species, and (3) PCB levels generally decreased over time.

Residue data for Lake Michigan are presented in Table 7 and Figures 7, 8, and 9. Compared to Lake Superior residues of dieldrin, DDT and PCBs are almost an order of magnitude higher in Lake Michigan. Figure 7 shows the levels of dieldrin, DDT and PCBs in lake trout from 1970 to 1982. Dieldrin levels appear to increase from 1970 to 1979 and then decline, while DDT residues declined from 20 ppm in 1970 to 5 ppm in 1982. PCB levels declined in the late 70s but since 1979 the rate of decline appears to have slowed.

Residue data for bloater chubs are shown in Figure 8. These data are much less variable than those for lake trout. Significant declines in both total PCBs and DDT are evident. Dieldrin levels on the other hand increased from the early 1970s and appear to have stabilized.

Figure 9 presents the organochlorine data for herring gull eggs from a nesting colony in the lake. Significant declines are shown for all residues except dieldrin.

Data for Lake Huron are presented in Tables 8 & 9 and Figure 10. Residues of PCBs and DDT in lake trout (Table 8) are much lower than those found in Lake Michigans. Levels of PCBs and DDT appear to have peaked in 1981 and 1982 and declined in 1983. Residues in rainbow smelt (Table 9) remained relatively constant between 1979 and 1983. Figure 11 shows organochlorine levels in herring gull eggs from a colony on Lake Huron, significant declines between 1974 and 1983 occurred for all components except dieldrin.

Data for rainbow smelt from Lake Erie are presented in Table 10. Residues of DDT and PCBs for this species are similar to those observed in Lake Huron. Figure 11 presents organochlorine data for herring gull eggs from Lake Erie. Levels are similar to those found in Lake Huron and generally decline with time.

Organochlorine residues in Lake Ontario fishes are shown in Table 11, 12 and 13. Residues of DDT and PCBs show declines between the early 1970s and 1980.

Table 14 lists the polynuclear aromatic hydrocarbons identified in fishes from Lake Ontario and the Detroit River and Table 15 shows the levels of 4 specific compounds for carp and pike. Table 16 presents data for PAHs in herring gull "lipids". Hallett and Brecher (1984) concluded that the metabolism of PAHs by fishes accounts for the low levels of these compounds observed in fishes and their predators, i.e. herring gulls.

Residue data from Lower Fox River and Green Bay are presented in Table 17. PCBs in carp from Little Lake des Morts were extremely high in the mid-1970s. These elevated levels were caused by a point source (paper mill) and the effectiveness of a pollution control program in reducing residues in this species is shown in Figure 12.

Effects of Contaminants on Biota

Fitchko (1985) recently made assessment of the effects of persistent toxic substance on Great Lakes biota. He concluded in part: (1) that "persistent toxic chemicals have apparently had both localized and lakewide effects on the health of Great Lakes aquatic populations; however, our understanding of these effects is poor; (2) few studies have been undertaken to assess the effects of specific toxic substances on the structure and functional responses of Great Lakes biota; (3) and insufficient research has been conducted to permit an assessment of human health consequences of exposure to great lakes drinking water and contaminated fish."

Lake wide effects which appear to have been caused by chlorinated organic contaminants (DDT and PCBs) have in Lake Michigan caused reproductive failure of planted lake trout (Willford, et al. 1981), and in Lake Ontario these compounds have been implicated in the reproductive failure of fish eating bird colonies in the early 1970s (Weseloh, et al. 1983 and Mineu, et al. 1984).

Fishing restrictions and/or consumption advisories are in effect for the species and areas shown in Figure 13.

Shear (1984) presented preliminary data implicating the industrial effluents as the most likely cause of epidermal papillomas in white suckers along Canadian Great Lakes shoreline (Figure 14). Black (1983) demonstrated that PAH contaminated sediments could produce epidermal hyperlasia and pappillomas in brown bullheads. In a field study brown bullheads from eastern Lake Erie and the Niagara River where sediments contained high levels of PAHs were found to have epidermal neoplasms (Black 1982). Sonstegard (1977) also implicated contaminated sediments in causing gonadal tumors in carp and goldfish in "polluted" areas of Lake Ontario.

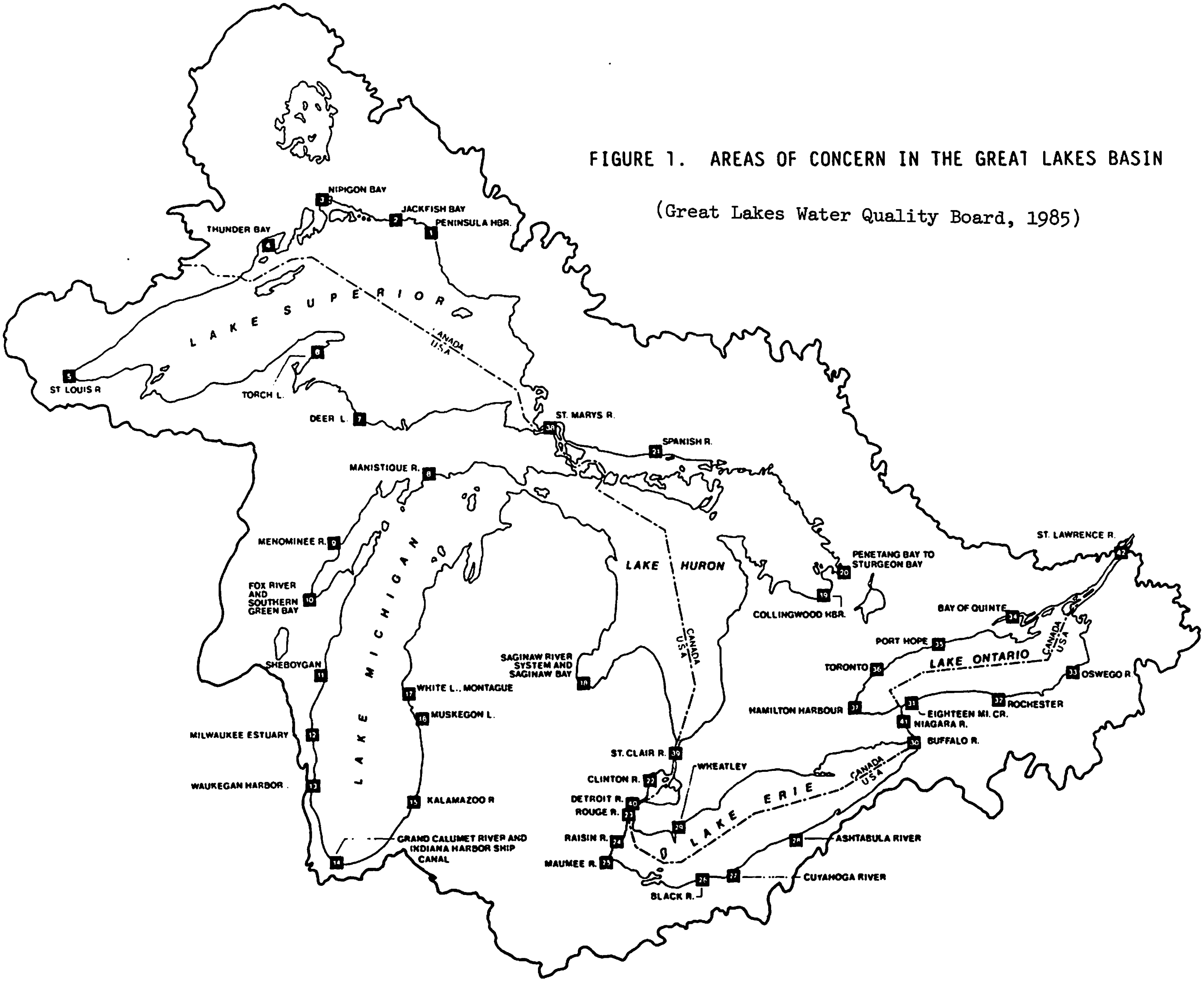


FIGURE 1. AREAS OF CONCERN IN THE GREAT LAKES BASIN

(Great Lakes Water Quality Board, 1985)

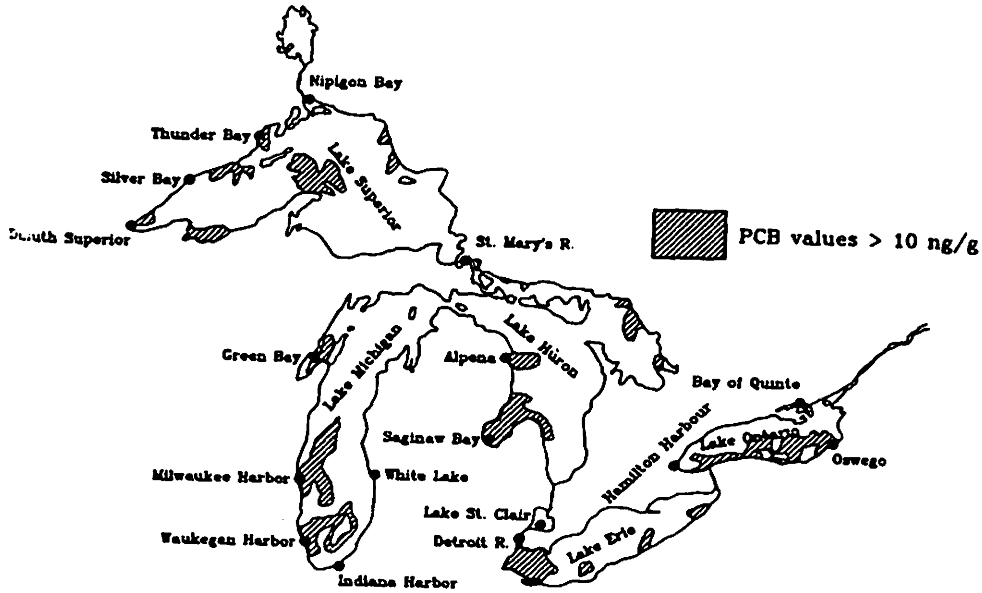
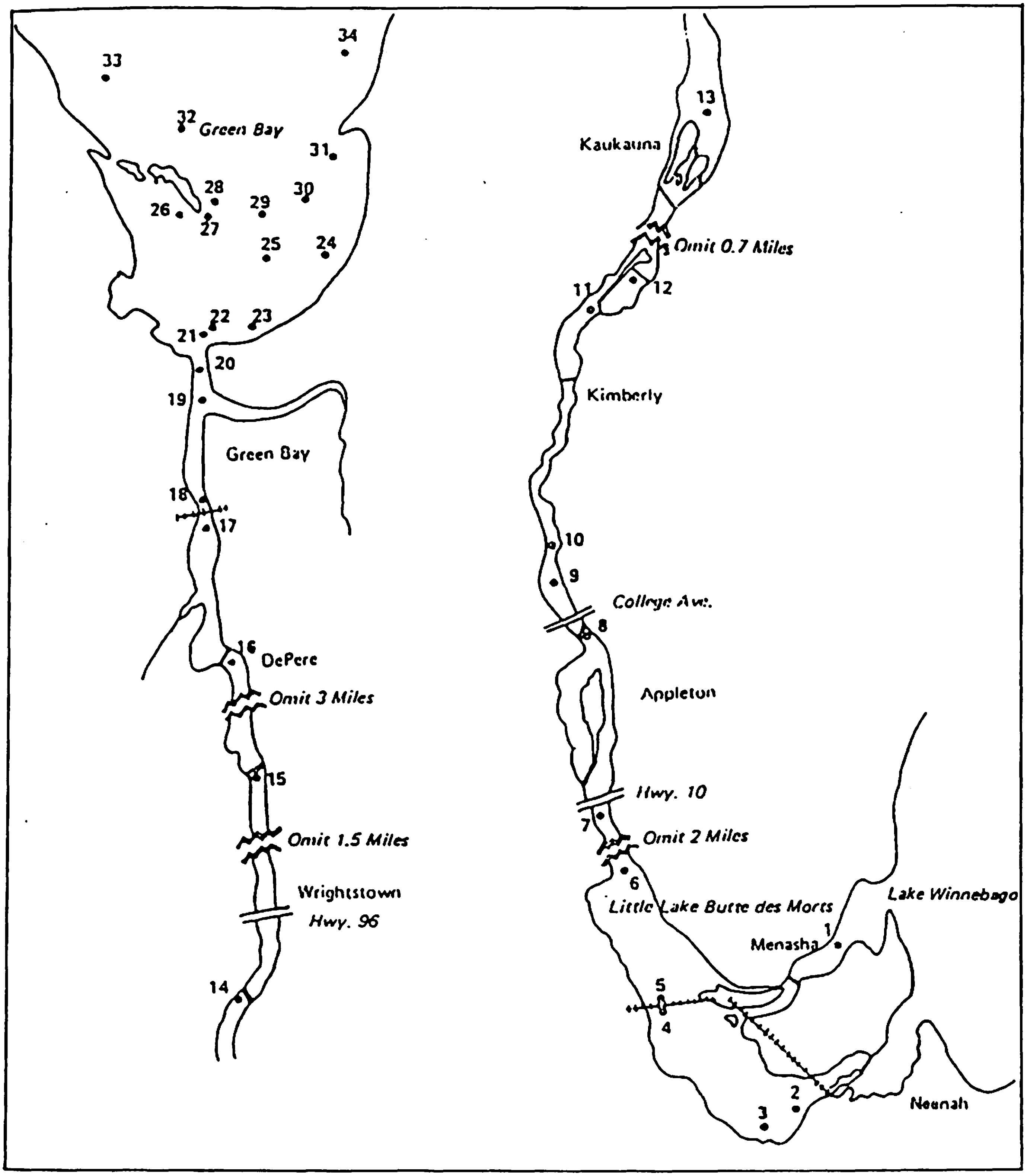


Figure 2

PCBs in sediments (Simmons, 1984)

Figure 3
Sediment Sampling Sites — Fox River and Lower Green Bay

(Sullivan and Delfino, 1982)



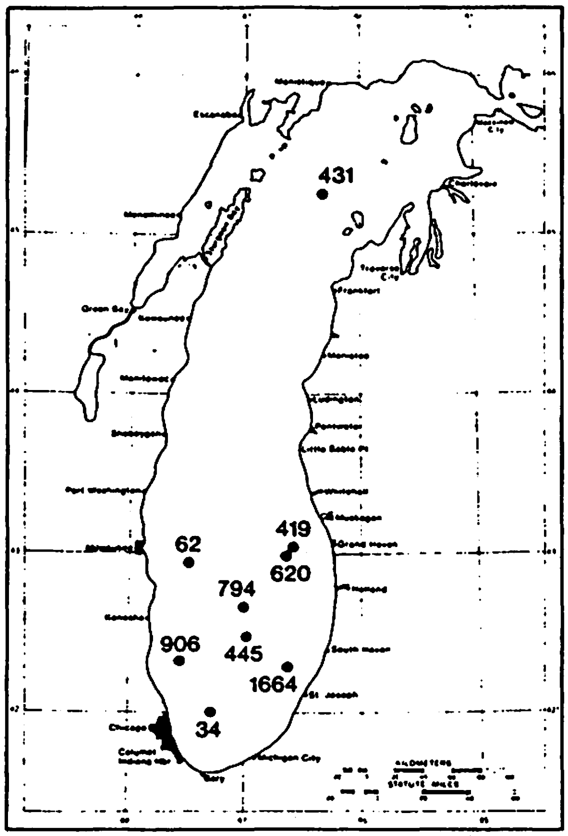


Figure 4 The distribution of fluorathene (ng/g) in the surfacial sediment of Lake Michigan.

(Eadie, 1984)

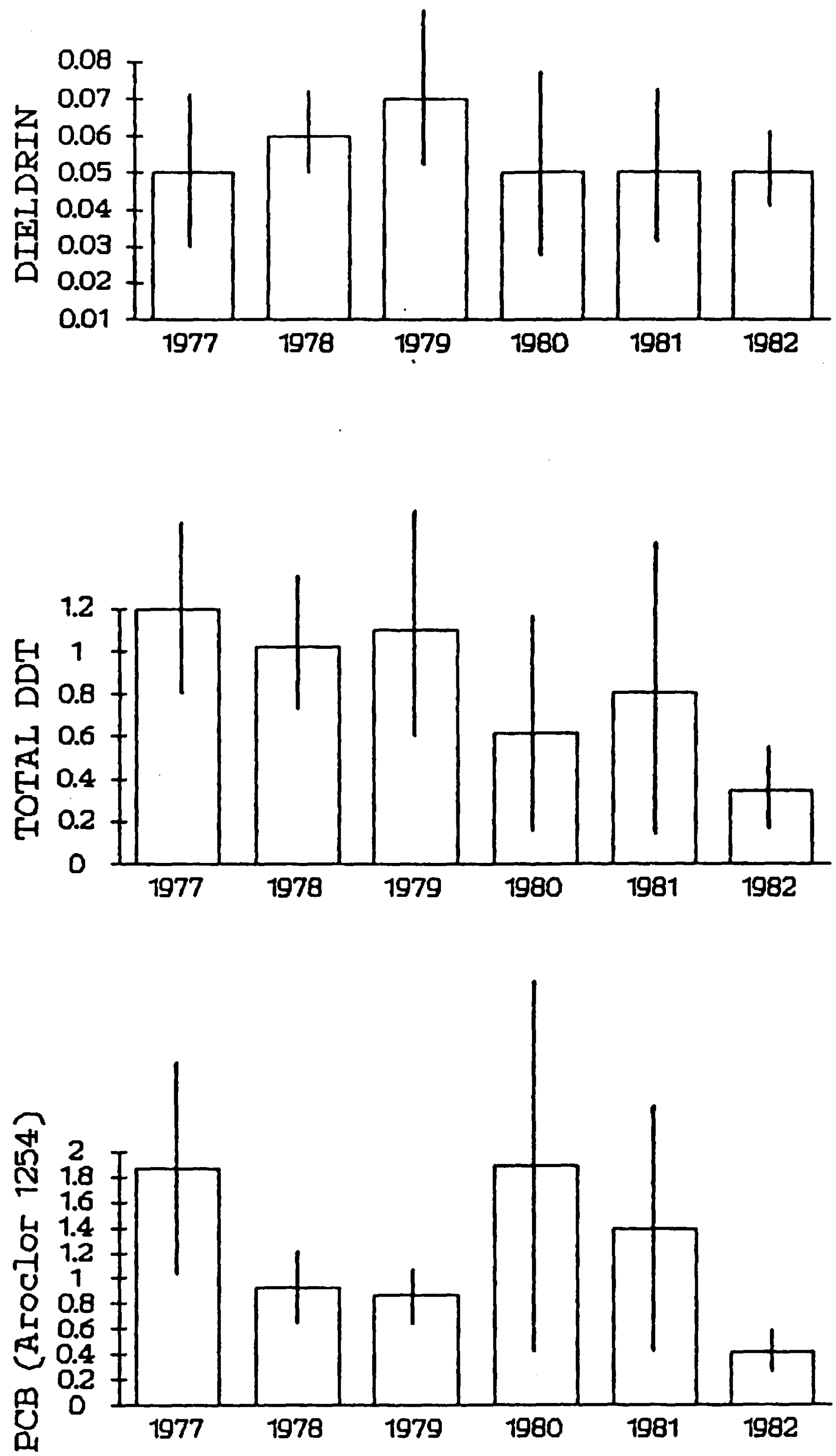


FIGURE 5. MEAN ANNUAL DIELDRIN, TOTAL DDT, AND PCB CONCENTRATIONS (mg/kg WET WEIGHT WITH 95% CONFIDENCE INTERVALS) IN WHOLE FISH SAMPLES OF LAKE TROUT COLLECTED FROM THE APOSTLE ISLANDS AREA OF LAKE SUPERIOR, 1977-1982.
Data from U.S. Environmental Protection Agency

(Great Lakes Water Quality Board, 1985)

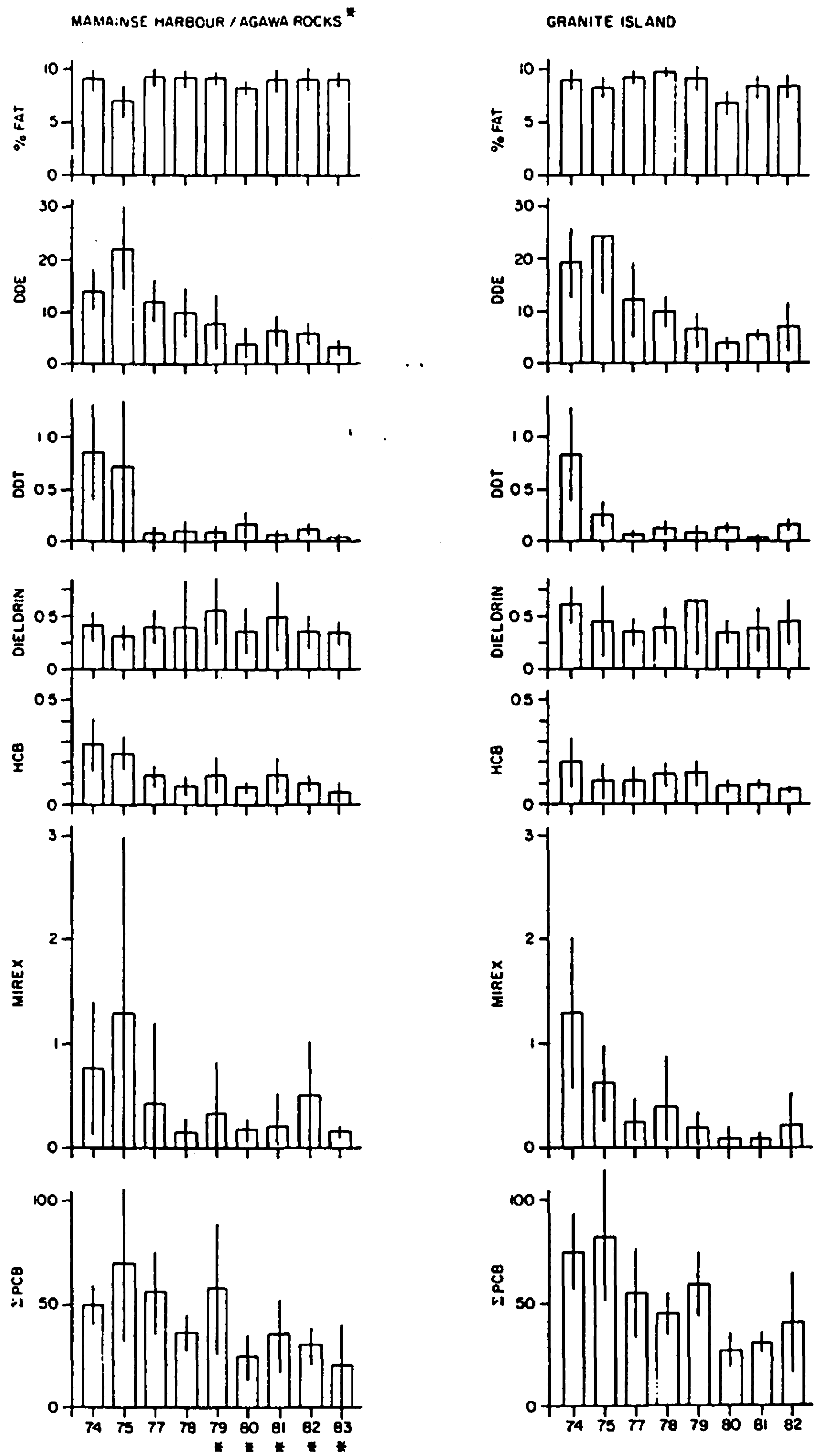


FIGURE 6. ORGANOCHLORINE CONTAMINANT CONCENTRATIONS (mg/kg WET WEIGHT ± S.D.) IN TWO COLONIES OF HERRING GULL EGGS ON LAKE SUPERIOR, 1974-1983.
Data from the Canadian Wildlife Service

(Great Lakes Water Quality Board, 1985)

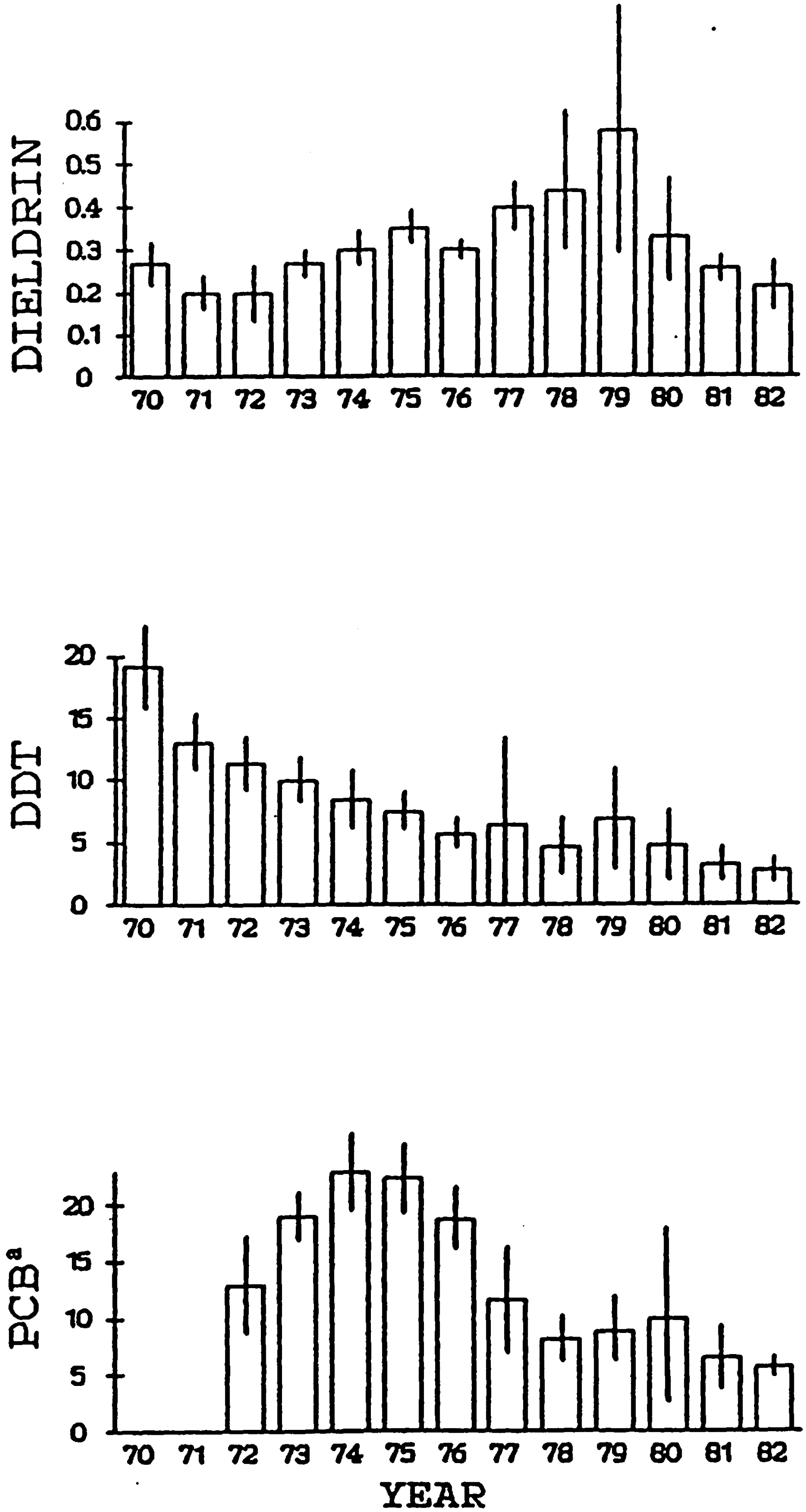


FIGURE 7 MEAN ANNUAL DIELDRIN, TOTAL DDT, AND PCB CONCENTRATIONS (mg/kg WET WEIGHT; WITH 95% CONFIDENCE INTERVALS) IN WHOLE FISH SAMPLES OF LAKE TROUT COLLECTED FROM EASTERN LAKE MICHIGAN. Data from U.S. Fish and Wildlife Service

a. From 1972-76 quantified using 1:1:1 Aroclor 1248, 1254, 1260 and beginning in 1977 only Aroclor 1254.

(Great Lakes Water Quality Board, 1985)

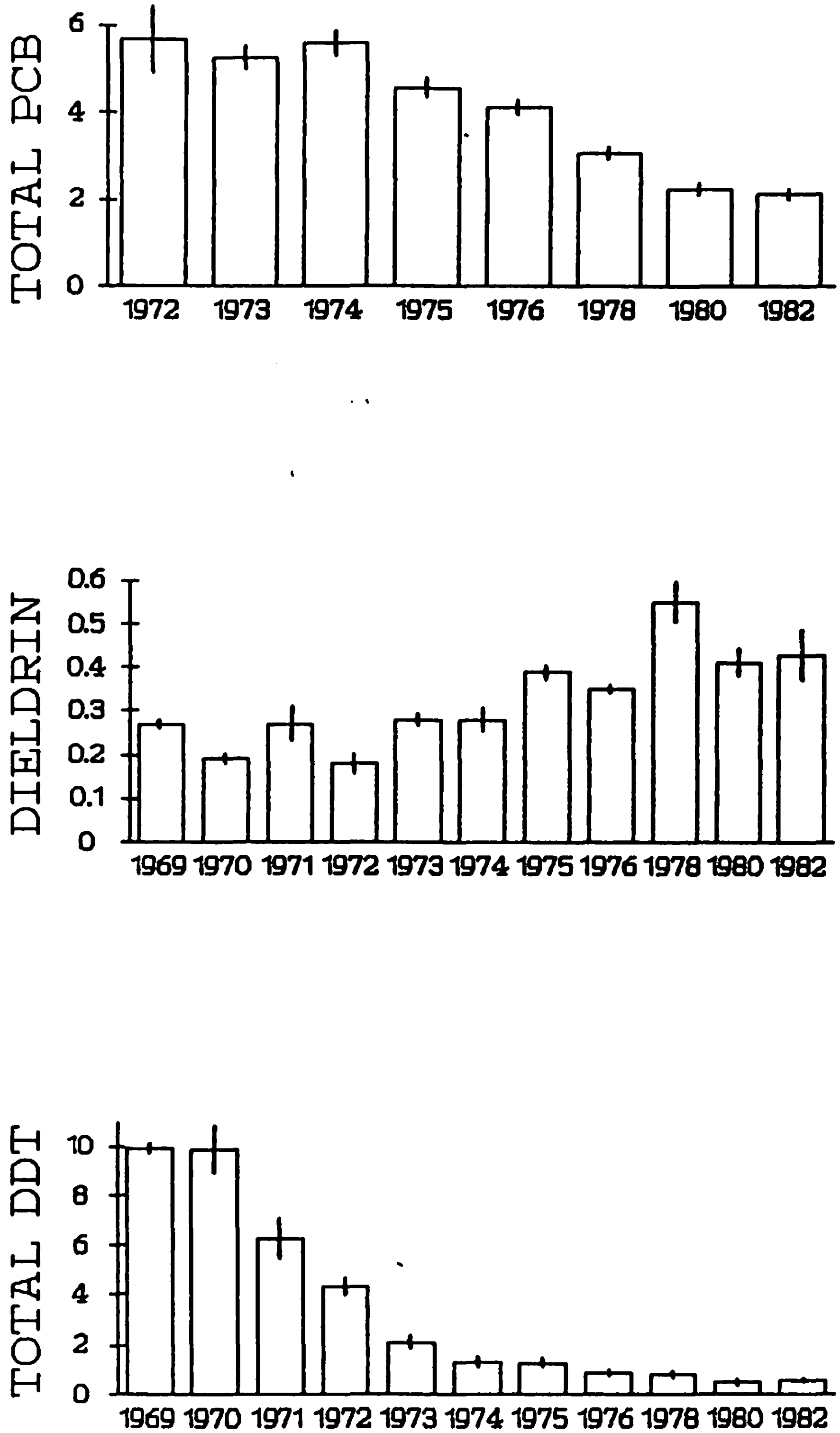


FIGURE 8. MEAN ANNUAL TOTAL PCB, DIELDRIN, AND TOTAL DDT CONCENTRATIONS (mg/kg WET WEIGHT; WITH 95% CONFIDENCE INTERVALS) IN WHOLE FISH COMPOSITE SAMPLES OF BLOATER CHUBS COLLECTED FROM EASTERN LAKE MICHIGAN.
Data from U.S. Fish and Wildlife Service

(Great Lakes Water Quality Board, 1985)

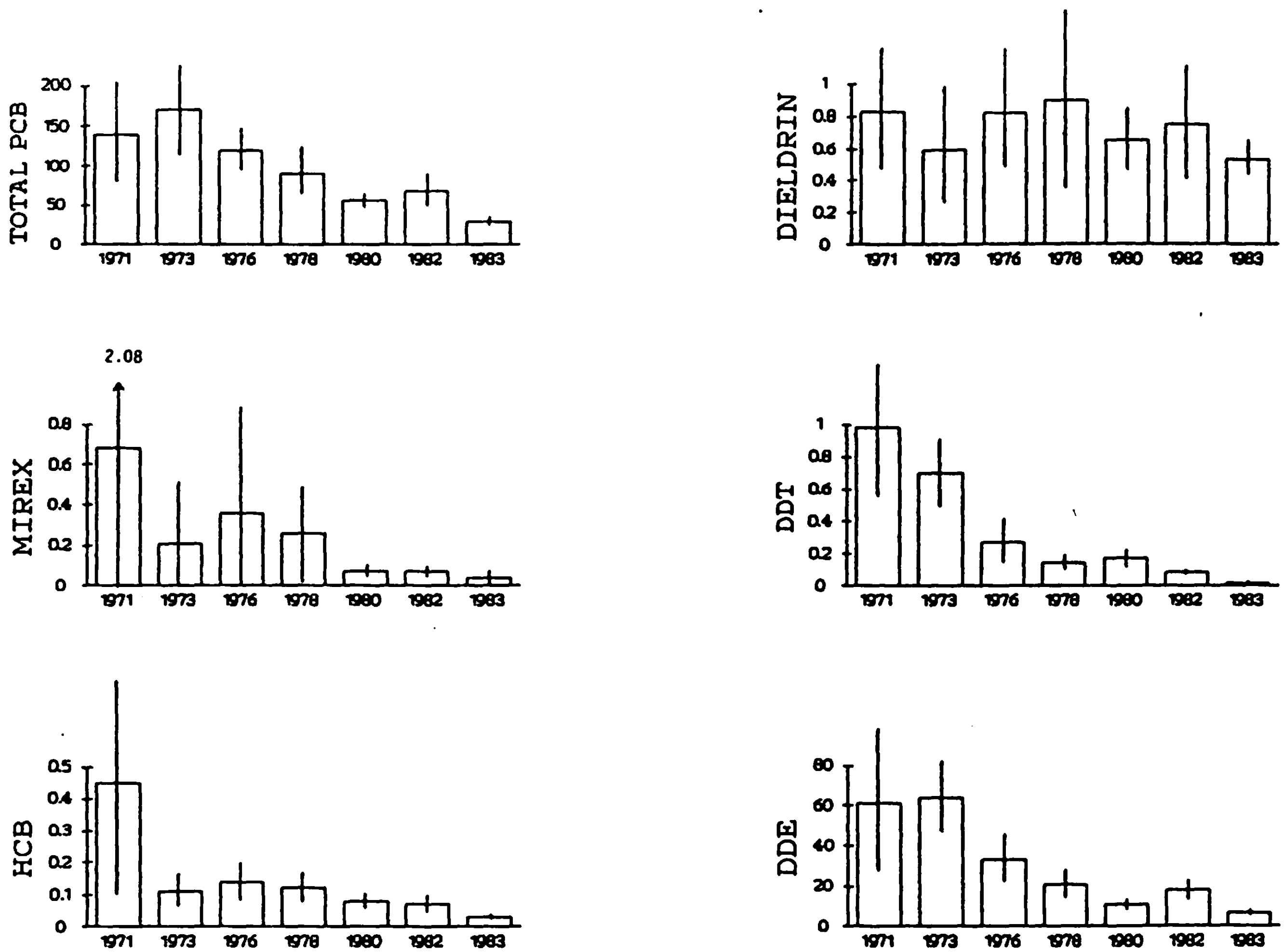


FIGURE 9 MEAN ORGANOCHLORINE CONTAMINANT CONCENTRATIONS (mg/kg WET WEIGHT ± S.D.) IN HERRING GULL EGGS FROM LAKE MICHIGAN'S BIG SISTER ISLAND, 1971-1983.
Data from Canadian Wildlife Service

(Great Lakes Water Quality Board, 1985)

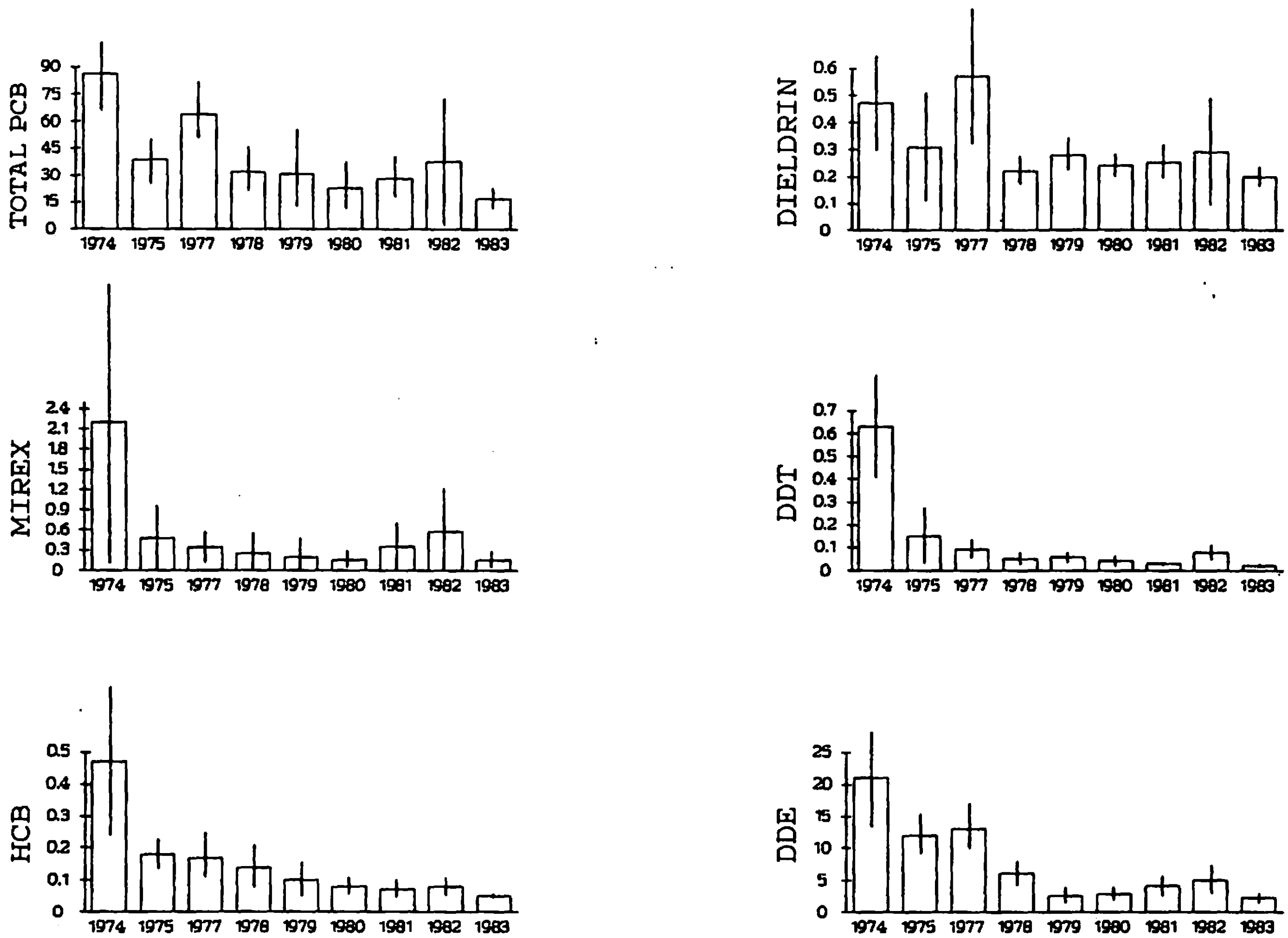


FIGURE 10 MEAN ORGANOCHLORINE CONTAMINANT CONCENTRATIONS (mg/kg WET WEIGHT ± S.D.) IN HERRING GULL EGGS FROM LAKE HURON'S CHANTRY ISLAND COLONY, 1974-1983.
Data from the Canadian Wildlife Service

(Great Lakes Water Quality Board, 1985)

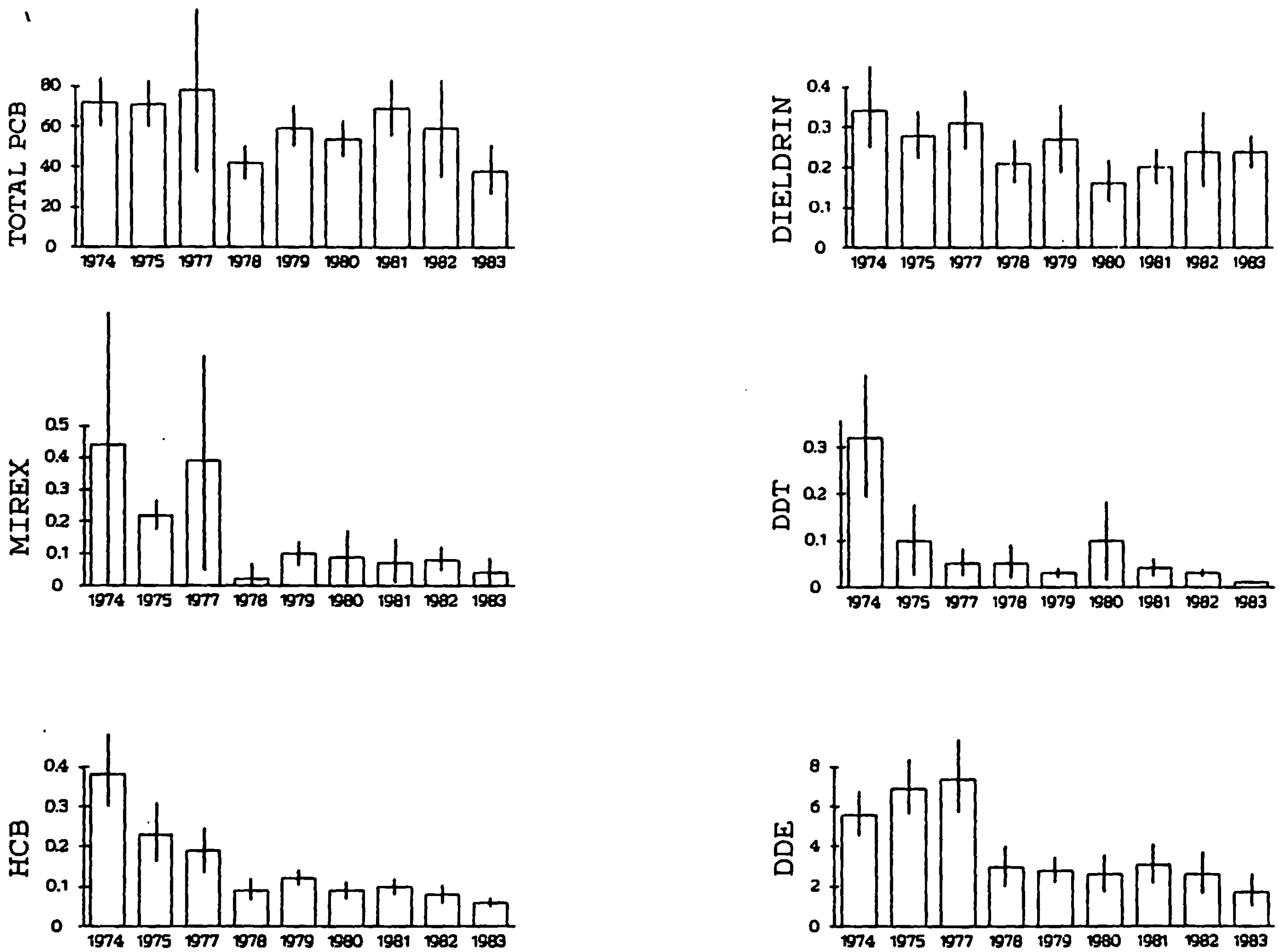
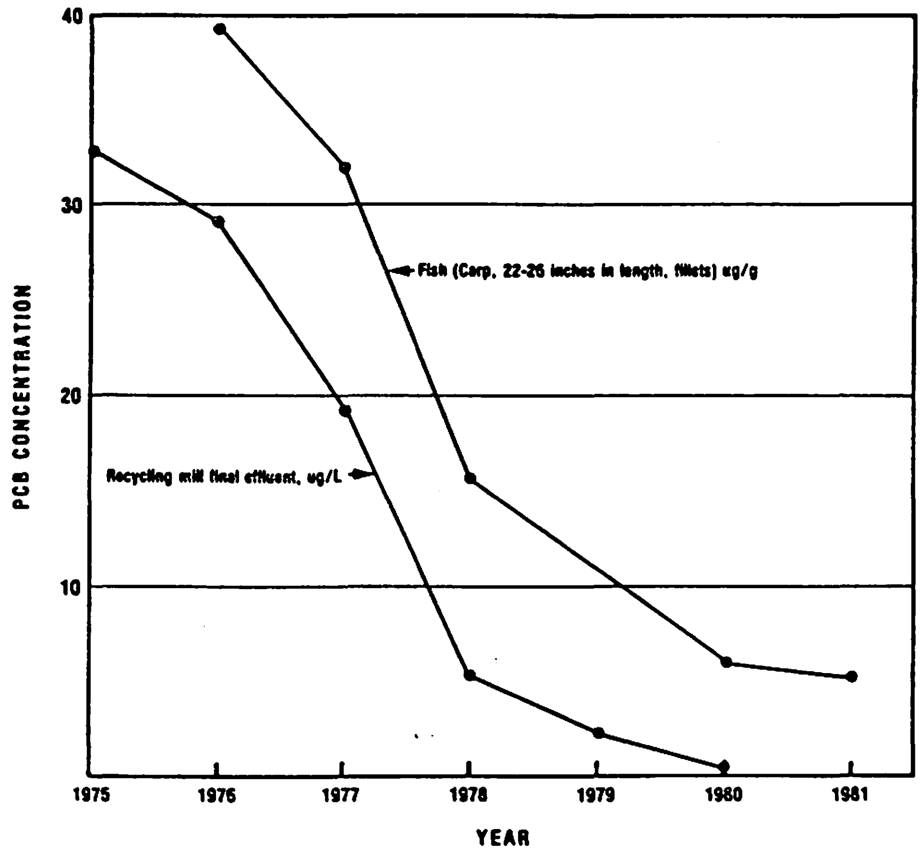


FIGURE 11 MEAN ORGANOCHLORINE CONTAMINANT CONCENTRATIONS (mg/kg WET WEIGHT $\pm$ S.D.) IN HERRING GULL EGGS FROM LAKE ERIE'S MIDDLE ISLAND COLONY, 1974-1983.
Data from the Canadian Wildlife Service (29).

(Great Lakes Water Quality Board, 1985)

Figure 12



PCB concentrations in a recycling paper mill final effluent and in carp (1975-1981 average concentrations) in Little Lake Butte des Morts.

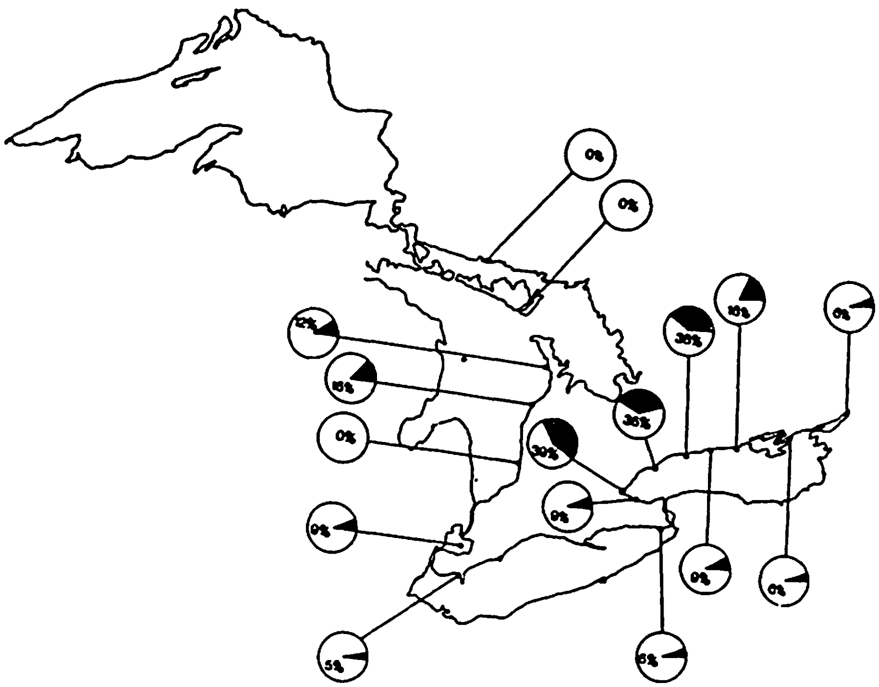
(Sullivan, et al., 1983)

Figure 13

PCB contamination in Great Lakes fish.

(Simmons, 1984)

Figure 14



Prevalence (%) of epidermal papilloma on white suckers.

(Shear, 1984)

Table 1 Mean values and standard deviations for PCB's (ng/g) by lake, non-depositional zones (NDZ) and depositional basins for the surficial 3 cms of sediment in the Great Lakes.

Lake Superior	Mean	S.D.
Whole Lake	3.3	5.7
Total Basin	4.8	5.5
NDZ	3.9	2.1
Sectors:-		
Duluth	8.6	13.7
Apostle	5.0	2.2
Chefswet	3.3	1.3
Thunder Bay	5.7	3.6
Trough	5.5	2.9
Isle Royale	4.5	2.2
Caribou	3.7	1.6
Marathon	6.4	7.3
Whitefish	4.4	3.0
Keweenaw	3.1	1.3
Lake Huron		
Whole Lake	12.8	10.3
Total Basin	15.4	12.8
NDZ	10.7	7.3
Sectors:-		
Mackinac	17.9	14.9
Manitoulin	12.2	5.7
Alpena	8.5	0.7
Saginaw	33.0	38.3
Port Huron	17.0	5.2
Goderich	18.6	14.0
Lake St. Clair		
Whole Lake 1970	19.1	8.9
Whole Lake 1974	9.9	6.3
Lake Ontario		
Whole Lake	57.5	56.2
Total Basin	85.3	57.0
NDZ	28.1	34.7
Sectors:-		
Niagara	89.1	68.3
Mississauga	77.1	50.6
Rochester	89.4	56.6

Lake Michigan	Mean	S.D.
Whole Lake	9.7	15.7
Total Basin	17.3	23.9
NDZ	6.3	8.1
Sectors:-		
Fox	73.5	78.9
Southern	17.1	11.1
Milwaukee	29.2	23.1
Waukegan	19.5	13.2
Grand Haven	17.1	23.1
Sarian	7.9	-
Algoma	10.1	10.6
Traverse	2.5	-
Georgian Bay		
Whole Lake	11.2	10.7
Total Basin	11.1	8.1
NDZ	11.2	13.2
Sectors:-		
Nottawasaga	19.6	12.3
Owen Sound Trough	9.1	6.1
Lion's Trough	18.0	8.5
Cabot	15.5	3.7
French River	24.0	24.0
Flowerpot	9.1	2.6
North Channel	8.2	4.0
Lake Erie		
Whole Lake	94.6	113.6
Total Basin	115.2	114.8
NDZ	64.0	105.1
Sectors:-		
Western	251.7	156.0
Sandusky	106.9	46.0
Central	74.1	55.7
Eastern	85.6	85.2
Bay of Quinte		
Whole Bay	50.6	53.6

(Thomas and Frank, 1983)

Table 2

PCB Concentrations in Fox River and Lower Green Bay Sediment[1]

Station No. and Location	Collection Date	PCB (mg/kg) (dry weight)	Other Compounds* (mg/kg)
1. Menasha Channel	05/23/77	<0.05	
2. Directly below Bergstrom	05/23/77	1.4	
3. 300 yards below Bergstrom	05/23/77	61.0	DHA - 2.7 PCP - 0.22
4. Little Lake Butte des Morts CNWRR Bridge	05/23/77	1.3	
5. Little Lake Butte des Morts CNWRR Bridge	11/24/76	5.5	
6. Little Lake Butte des Morts outlet	05/23/77	21.0	
7. Appleton Yacht Club	11/24/76	8.2	
8. Above lower Appleton Dam	05/23/77	9.0	
9. Below Consolidated	05/23/77	1.2	
10. One mile below Consolidated	06/22/77	3.6	
11. Above Kimberly Dam below Midtec	05/23/77	0.9	
12. Above lower Little Chute Dam	05/23/77	5.1	
13. Below Thilmany Paper	06/04/77	4.8	
14. Above Rapide Croche Dam	06/04/77	5.8	PCP - 0.22
15. Above Little Rapids Dam	06/04/77	5.0	PCP - 0.28
16. Above DePere Dam	06/22/77	0.18	
17. Across from Ft. Howard outfall	05/23/77	0.96	
18. Below Ft. Howard, CNWRR bridge	05/23/77	18.3	
19. Near mouth of East River	05/24/77	13.0	
20. Above Green Bay STP near mouth	05/24/77	2.1	
21. Below Green Bay STP outfall	11/24/76	38.0	
22. Green Bay	05/24/77	7.5	
23. Green Bay	05/24/77	7.2	
24. Green Bay	05/24/77	4.7	
25. Green Bay	05/24/77	0.12	
26. Green Bay	05/24/77	1.8	
27. Green Bay	05/24/77	0.46	
28. Green Bay	05/24/77	5.6	
29. Green Bay	05/24/77	< 0.05	
30. Green Bay	05/24/77	11.0	
31. Green Bay	05/24/77	2.6	
32. Green Bay	05/24/77	<0.05	
33. Green Bay	05/24/77	0.02	
34. Green Bay	05/24/77	0.075	

*DHA = Dehydroabietic Acid
PCP = Pentachlorophenol

(Sullivan and Delfino, 1982)

Table 3 Atmospheric Flux of PAH to the Great Lakes (metric tonnes/yr)

Compound	Lake					
	Superior[a]	Michigan[a]	Michigan[b]	Huron[a]	Erie[a]	Ontario[a]
Phenanthrene	4.8	3.4	2.1	3.5	1.5	1.1
Anthracene	4.8	3.4	2.1	3.5	1.5	1.1
Fluoranthene	—	—	3.6	—	—	—
Pyrene	8.3	5.9	4.0	6.1	2.6	1.9
Benzo(a)anthracene	4.1	2.9	3.3	3.0	1.5	1.1
Benzo(a)pyrene	7.9	5.6	4.0	5.8	2.5	1.8
Perylene	4.8	3.3	2.1	3.4	1.5	1.1

[a] Eisenreich et al., 1981.
[b] Andren and Strand, 1981.

(Eadie, 1984)

Table 4 Lake Michigan Sediment PAH (ng/g dry)

Station	Phen	Fl	Py	C + T	BaP
T1	28	34	30	36	33
T2	809	906	733	—	450
T3[a]	1268	1664	1430	1128	944
T4	308	445	363	319	251
T5	537	794	665	522	248
T6	19	62	38	41	26
T7	298	620	421	554	572
T8	245	419	319	348	571
T11	263	413	318	543	324

[a] Coefficients of variation (%) for replicate analyses of replicate extracts from station T3 are: phen., 27.6; Fl., 13.0; Py., 11.6; C + T, 19.4; BaP, 18.5.

(Eadie, 1984)

Table 5 Concentration of PAH in Great Lakes Surficial Sediments (ng/g dry)

Compound	Lake				
	Superior[a]	Michigan[b]	Huron	Erie[c]	Ontario[d]
Phenanthrene	34	533 ± 382	272	346 ± 92	58.5
Fluoranthene	88	754 ± 444	487	569 ± 442	615 ± 394
Pyrene	53	607 ± 399	356	391 ± 91	647 ± 594
BaP	28	480 ± 246	294	255 ± 152	—
n	1	9	1	3	2–4

[a] Gschwend and Hites, 1981.
[b] Eadie, 1983; depositional basins.
[c] Eadie et al., 1982.
[d] IJC, 1977.

(Eadie, 1984)

Table 6 Abundances of Polycyclic Aromatics in Lake Ontario Sediment (LAT. 43°39′, LONG. 78°12′) μ/g Dry Sediment [a]

Aromatic	0–5 cm	10–15 cm	20–25 cm	30–35 cm	55–60 cm	70–75 cm
Biphenyl	0.014	0.007	0.009	0.004	0.004	0.004
Tetrahydropyrene	0.056	0.029	—	—	—	—
Fluoranthene	0.281	0.058	—	—	—	—
Pyrene	0.056	0.029	—	—	—	—
1.2-Benzanthracene						
Chrysene	0.225	0.088	0.052	—	—	—
Triphenylene						
Dimethyl chrysene	0.112	—	—	—	—	0.018
2.3-Benzofluoranthene	0.450	0.029	0.017	0.017	0.020	0.009
Methyl benzofluoranthene	0.056	—	—	—	—	—
Benzpyrenes	0.337	—	0.017	0.034	0.010	0.009
Perylene	0.056	0.029	0.017	0.034	0.30	0.046
Methyl benzpyrene	0.056	—	—	—	—	—
Methyl perylene	0.112	—	—	—	0.010	0.027
20-Methyl cholanthrene	0.337	—	—	—	—	0.018
Benzperylene	0.225	—	—	—	—	—
Coronene	0.562	—	—	—	—	—
Total aromatics	2.935	0.269	0.112	0.089	0.084	0.131

[a] IJC, 1976.

(Strachan and Edwards, 1984)

Table 7

Results of PCB Analysis from 1971 to 1981 in Seven Species of Fish from Lake Michigan. PCB and percent fat were determined from fillet portions.

	Lake Trout[a]				Chinook Salmon				Coho Salmon				Walleye				Lake Whitefish				Chub				Alewife			
Year	n	l	f[b]	PCB[b]	n	l	f	PCB	n	l	f	PCB	n	l	f	PCB	n	l	f	PCB	n	l	f	PCB	n	l	f	PCB
1971	29	23.2	—	16.7																								
1972	10	21.4	—	22.4																								
1973																												
1974	30	24.5	15.2	15.9	8	33.1	2.3	11.7	18	23.5	3.4	5.3										11.6						
1975	54	21.9	10.3	8.5					2	20.1	5.8	4.8					18	19.7	16.9	4.4	2		9.0	2.3				
1976	26	23.9	11.7	9.4	7	30.8	2.0	7.4					4	14.6	0.9	0.6	3	15.9	3.3	2.0		10.6			2	7.0	4.8	4.7
1977	3	13.7	4.3	1.2									8	17.7	2.4	1.4	31	19.7	13.4	3.6	26		17.6	3.8	1	7.0	3.0	2.4
1978	30	22.7	12.3	7.8	24	30.9	4.1	8.9	5	21.9	5.3	6.1					15	19.9	11.2	3.4	5	10.2	15.9	2.1	4	6.1	5.3	3.4
1979	3	23.9	16.3	8.5	10	33.2	2.7	6.1	10	21.6	3.4	3.1					11	18.2	12.9	1.7	13	8.3	16.6	1.2				
1980	7	21.4	13.8	2.6	21	35.4	3.5	4.4	10	23.5	3.8	1.7	1	15.9	9.0	8.1												
1981					30	30.7	4.5	3.8	1	26.0	4.1	1.6	7	21.3	5.2	3.4												
Total	192				97				46				20				78				46				7			

[a] n = number of samples; l = ave. length in inches; f = ave. % fat; PCB = ave. PCB in mg/kg (ppm).
[b] PCB and percent fat were determined from fillet portions.

(St. Amant, et al., 1984)

TABLE 8

MEAN CONTAMINANT CONCENTRATIONS (mg/kg WET WEIGHT ± S.E.) IN WHOLE FISH SAMPLES OF LAKE HURON LAKE TROUT, 1979-1983[a]

	1979[b]	1980[b]	1980[c]	1981	1982	1983	1983[b]
Number	47	50	47	50	79	50	49
Weight (g)	714.8 (70.6)	905.3 (62.8)	742.3 (106.7)	1,949.4 (61.2)	2,042.5 (101.1)	1,454.9 (68.7)	913.5 (105.3)
% Lipid	8.77 (0.58)	10.35 (0.58)	10.33 (0.63)	18.05 (0.47)	15.70 (0.51)	13.58 (0.48)	8.98 (0.74)
PCB	0.78 (0.08)	0.42 (0.03)	0.92 (0.08)	2.26 (0.13)	2.44 (0.17)	1.24 (0.06)	0.53 (0.03)
DDE	0.15 (0.02)	0.22 (0.01)	0.34 (0.03)	0.60 (0.03)	0.48 (0.73)	0.39 (0.02)	0.19 (0.02)
ΣDDT	0.20 (0.02)	0.25 (0.01)	0.49 (0.04)	1.06 (0.05)	0.73 (0.04)	0.68 (0.04)	0.26 (0.02)
Mercury	0.16 (0.01)	0.18 (0.01)	0.14 (0.01)	0.24 (0.01)	0.19 (0.01)	-	-
Arsenic	0.15 (0.01)	0.18 (0.01)	0.27 (0.02)	0.43 (0.02)	0.43 (0.02)	-	-
Selenium	0.70 (0.02)	0.75 (0.01)	0.81 (0.01)	0.48 (0.02)	0.62 (0.01)	-	-

[a]Data from the Canada Department of Fisheries and Oceans
[b]Splake.
[c]Splake-Backcross.

(Great Lakes Water Quality Board, 1985)

TABLE 9

MEAN CONTAMINANT CONCENTRATIONS (mg/kg WET WEIGHT ± S.E.) IN WHOLE FISH SAMPLES OF LAKE HURON RAINBOW SMELT, 1979-1983[a,b]

	1979	1980	1981	1982	1983
Number	12	36	36	24	24
Weight (g)	26.33 (1.72)	20.68 (1.51)	27.69 (1.85)	27.16 (2.53)	23.44 (1.67)
% Lipid	4.24 (0.16)	3.46 (0.14)	4.30 (0.14)	5.17 (0.25)	3.99 (0.10)
PCB	0.19 (0.02)	0.11 (0.01)	0.13 (0.01)	0.29 (0.02)	0.18 (0.01)
pp'DDE	0.05 (0.01)	0.05 (0.01)	0.07 (0.01)	0.08 (0.01)	0.07 (0.01)
ΣDDT	0.07 (0.02)	0.07 (0.01)	0.10 (0.01)	0.12 (0.01)	0.10 (0.01)
Mercury	0.06 (0.01)	0.07 (0.01)	0.06 (0.00)	0.05 (0.00)	-
Arsenic	0.27 (0.02)	0.26 (0.01)	0.31 (0.01)	0.36 (0.02)	-
Selenium	0.64 (0.01)	0.69 (0.02)	0.68 (0.02)	0.54 (0.01)	-

[a]Data from the Canada Department of Fisheries and Oceans
[b]Each sample represents a composite of five fish.

(Great Lakes Water Quality Board, 1985)

TABLE 10

MEAN CONTAMINANT CONCENTRATIONS (mg/kg WET WEIGHT ± S.E.) IN WHOLE FISH SAMPLES OF LAKE ERIE RAINBOW SMELT, 1977-1983[a]

YEAR	N[b]	WEIGHT	% LIPID	PCB	pp'DDE	ΣDDT	Hg	Pb	As	Se
1977	78	18.45 (1.69)	2.74 (0.10)	0.18 (0.01)	0.04 (0.00)	0.06 (0.00)	0.05 (0.00)	c	-	0.29 (0.01)
1978	44	30.24 (1.90)	4.45 (0.18)	0.27 (0.03)	0.04 (0.00)	0.06 (0.01)	0.05 (0.00)	c	0.15 (0.01)	0.36 (0.01)
1979	35	32.38 (3.15)	4.65 (0.25)	0.38 (0.04)	0.05 (0.00)	0.10 (0.01)	0.04 (0.00)	c	0.23 (0.01)	0.31 (0.01)
1980	42	25.56 (2.10)	3.48 (0.11)	0.26 (0.02)	0.06 (0.00)	0.12 (0.01)	c	0.21 (0.03)	0.16 (0.01)	0.37 (0.01)
1981	36	31.55 (2.85)	4.76 (0.22)	0.23 (0.02)	0.03 (0.00)	0.06 (0.00)	0.04 (0.00)	c	0.23 (0.01)	0.35 (0.01)
1982	34	29.78 (2.05)	5.17 (0.18)	0.30 (0.01)	0.03 (0.00)	0.07 (0.01)	0.03 (0.00)	c	0.26 (0.01)	0.35 (0.01)
1983	29	21.44 (2.31)	3.84 (0.22)	0.32 (0.03)	0.02 (0.00)	0.04 (0.00)	-	-	-	-

[a]Data from the Canada Department of Fisheries and Oceans (15).
[b]Each sample consists of a composite of five fish.
[c]>50% of results below detection limit.

(Great Lakes Water Quality Board, 1985)

Table 11 Levels of Selected Organic Compounds (μg/g) in Whole Fish from Lake Ontario, 1975–1981 (Shear, 1984)

Year	Organic Compound ΣDDT	PCB	Dieldrin	Mirex
		Lake Trout		
1977	2.66	4.95	0.04	0.27
1978	1.16	7.10	0.18	0.21
1979	1.58	3.79	0.20	0.23
1980	0.62	4.79	0.10	0.18
1981	1.39	2.82	0.15	0.15
		Coho Salmon		
1977	1.43	3.03	0.07	0.16
1978	0.64	3.00	0.10	0.08
1979	0.81	1.21	0.10	0.05
1980	0.74	2.30	0.07	0.10
		Smelt		
1977	0.60	1.50	0.02	0.11
1978	0.44	1.82	0.05	0.06
1979	0.39	0.80	0.04	0.06
1980	0.25	1.12	0.04	0.08
		Spottail Shiners		
1975	0.24	0.69	—	—
1976	—	—	—	—
1977	0.16	0.65	—	0.013
1978	0.099	0.32	—	0.029
1979	0.026	0.15	—	0.001
1980	0.041	0.27	—	0.011

Table 12 DDT Residues (μg/g) in Selected Lake Ontario Biota (Strachan and Edwards, 1984)

Sample Year	Lake Trout		Smelt			Coho Salmon			Spottail Shiner	Herring Gull Eggs
	DEC[a]	DFO	DEC	DFO	EPA	OME	DEC	DFO	OME	CWS
1972				1.8e				0.9e		34
1973				1.5e				1.7e		
1974	7.7e[b]			0.4e						23
1975	1.3e				1.4w		0.93e		0.17w	22
1976	0.91e			0.25e			0.93e	0.69f	0.28w	18
1977		2.3e		0.6e		1.4w		1.6e	0.21w	
		2.7w		0.6w						
1978		1.3w		0.44w		0.64w			0.18w	
1979		1.6w		0.39w		0.81w			0.06w	
1980		0.62w		0.25w		0.74w			0.05w	7.7

[a] CWS = Canadian Wildlife Service (Canada); DEC = Department of Environmental Conservation (New York State); DFO = Department of Fisheries and Oceans (Canada); EPA = Environmental Protection Agency (United States); OME = Ontario Ministry of the Environment (Province of Ontario).
[b] e = edible portion; f = fillet; w = whole fish.

Table 13 PCB Residues (µg/g) in Selected Lake Ontario Biota (Strachan and Edwards, 1984)

Sample	Lake Trout			Smelt			Coho Salmon			Spottail Shiner	Herring Gull Eggs
Year	DEC[a]	DFO	OME	DEC	DFO	OME	DEC	DFO	OME	OME	CWS
1970				2.2f			7.9				
1971							6.7				
1972					5.0e		4.7	4.1e			204
1973					7.3e			6.7e			
1974	7.7				1.7e		6.3				155
1975	9.4			2.1f			8.4			0.69w	145
1976	7.1				1.2e	2.6	6.1	3.7e	11.6f	1.3w	138
1977		5.0w	4.9		1.4e	1.5w		3.2e	3.0w	1.5w	104
					1.5w			3.0w			
1978		7.1w	6.1		1.8w	1.6		3.0w		1.1w	61
1979		3.8w	3.8		0.8w	0.75		1.2w	2.8w	0.46w	
1980		4.8w			1.1w			2.3w		0.31w	41

[a] CWS = Canadian Wildlife Service (Canada); DEC = Department of Environmental Conservation (New York State); DFO = Department of Fisheries and Oceans (Canada); OME = Ontario Ministry of the Environment (Province of Ontario).

[b] e = edible portion; f = fillet; w = whole fish.

Table 14 Polynuclear Aromatic Hydrocarbons Identified by Mass Spectrometry in Great Lakes Fish[a,b] (Hallett and Brecher, 1984)

PAH	Hamilton Harbour		Detroit River	
	Carp	Pike	Carp	Pike
Naphthalene	X	X		X
2-methyl Naphthalene	X	X		X
1-methyl Naphthalene	X	X		X
Biphenyl	X	X		X
Acenaphthene		X		X
Dimethyl naphthalene		X		X
Fluorene		X		X
Anthracene	X	X		X
Phenanthrene	X	X		X
1-phenyl Naphthalene	X	X		X
1-methyl Phenanthrene	X	X		X
1-methyl Anthracene	X	X		X
2-methyl Anthracene	X	X		X
2-methyl Phenanthrene	X	X		X
9-methyl Anthracene				X
Fluoranthrene	X	X		X
Pyrene	X	X		X
1,2-benzofluorene		X		X
2,3-benzofluorene		X		X
Chrysene	X	X		X
Benzo(a)pyrene		X		X
Perylene		X		X
Dibenz(a,h)anthracene	X	X		X
Coronene	X	X		X

[a] Hallett et al., 1978.
[b] X = detected.

Table 15 Polynuclear Aromatic Hydrocarbons (ng/kg) Fresh Weight Fillet[a] (Hallett and Brecher, 1984)

Fish	Perylene	Benzo(k)fluoranthene	Benzo(a)pyrene	Coronene
		Hamilton Harbour		
Carp				
1	46	8	108	300
2	140	40	160	360
3	26	12	96	210
4	160	40	200	400
5	40	16	108	220
6	74	12	160	60
7	40	8	144	300
8	142	68	268	320
9	nd[b]	nd	nd	20
Pike				
1	90	48	154	240
2	64	32	128	200
3	40	12	70	200
4	32	10	54	220
5	58	20	74	200
6	40	12	64	170
7	34	8	60	152
8	40	12	54	140
9	nd	nd	nd	nd
10	20	12	34	100
		Detroit River		
Carp				
1	16	10	40	80
2	nd	nd	nd	60
3	40	14	40	nd
4	26	10	40	40
5	nd	nd	nd	nd
6	nd	nd	nd	nd
7	nd	nd	nd	120
8	nd	nd	nd	80
9	nd	nd	nd	nd
10	nd	nd	nd	nd
Pike				
1	34	26	40	20
2	20	14	14	40
3	18	8	34	40
4	20	8	20	44
5	68	26	128	290
6	18	10	24	40
7	20	6	30	30
8	nd	nd	nd	nd
9	46	24	70	120
10	52	26	100	120

[a] Hallett et al., 1978.
[b] nd = nondetectable.

Table 16 Identification of Polynuclear Aromatic Hydrocarbons in Great Lakes Herring Gull Lipid[a,b] (Hallett and Brecher, 1984)

Compounds	Concentration (μg/kg) Pigeon Island	Kingston	Mass Spectral Confirmation
Naphthalene	0.050	0.054	+
2-methyl Naphthalene	0.036	0.005	+
1-methyl Naphthalene	0.043	0.009	+
Biphenyl	0.151	0.017	+
Acenaphthene	0.038	0.007	+
4-methyl Biphenyl	0.061	0.010	+
Fluorene	0.044	0.003	+
Anthracene	0.152	0.024	+
Phenanthrene	nd	0.002	+
1-phenyl Naphthalene	0.008	0.008	
2-methyl Phenanthrene	0.021	0.007	+
1-methyl Phenanthrene	0.010	0.015	+
9-methyl Anthracene	0.011	0.025	+
3,6-dimethyl Phenanthrene	nd	0.012	+
Fluoranthrene	0.082	0.017	+
Pyrene	0.076	0.015	+
1-aza Pyrene	a	a	
9-acetylanthracene	a	a	
DDE	—	—	+
1,2-benzofluorene	a	a	+
2,3-benzofluorene	a	a	+
1-methyl Pyrene	a	a	+
2-acetyl Phenanthrene	a	a	
1,1-binaphthyl	a	a	
Chrysene	0.053	a	+
Benz(e)pyrene	0.026	0.021	+
Benz(a)pyrene	0.038	0.030	+
Perylene	0.053	0.026	
9-dichloromethylene Fluorene	b	b	+b
Dimethyl biphenyl	b	b	+b

[a] Hallett et al., 1977.

[b] a = PCB interference; b = Standards of compounds unavailable, compounds identified by mass spectra; nd = not detected.

Table 17
Lower Fox River and Lower Green Bay Fish Contaminant Data

Sample Location	Date	Species	Quantity	Form	Length (mm)	% Fat	PCB** (mg/kg)	Other Chloro-organics or metals (mg/kg)
Little Lake Butte Des Morts	2/76	Yellow Perch	05	F	133	0.4	0.8	
			05	F	152	0.2	0.6	
			05	F	152	0.4	0.8	
			05	F	203	0.2	0.9	
	6/76	Carp	03	F	508	7.9	26.0	
			02	F	279	5.9	12.0	
			03	F	391	6.7	24.0	
			03	F	615	15.9	39.0	
	8/76	Brown Bullhead	04	F	257	5.9	5.2	
			04	F	213	13.6	6.0	
		Green Sunfish	02	EP	170	3.1	2.4	
		Yellow Perch	05	F	203	0.7	1.4	
			05	EP	147	1.1	1.3	
		Walleye	05	F	368	0.6	1.7	
			05	F	262	0.3	1.2	
		White Bass	03	EP	325	6.5	9.8	
			03	EP	193	3.2	9.3	
	4/77	*Northern Pike	01	F	688	0.5	2.4	Pentachloroanisole<0.005
			01	F	770	0.8	2.3	Pentachloroanisole 0.020
		*Walleye	01	F	412	1.5	1.8	Pentachloroanisole 0.060
		*Carp	01	F	424	2.4	5.2	Pentachloroanisole 0.036
		Carp	01	F	406	1.2	2.7	
			01	F	445	1.7	10.0	
			01	F	406	2.0	4.3	
			01	F	470	2.3	13.0	
			01	F	546	4.6	16.0	
			01	F	584	5.4	28.0	
			01	F	558	5.8	30.0	
			01	F	508	6.1	18.0	
			01	F	533	6.1	20.0	
			01	F	570	9.0	39.0	
			05	WF	519	12.5	50.0	
	5/77	Carp	05	WF	519	10.6	41.0	
			05	WF	493	9.2	35.0	
			01	F	570	9.0	39.0	
	8/78	Brown Bullhead	05	WF	203	5.8	4.4	
			05	WF	229	5.0	4.1	
			05	WF	191	5.0	3.8	
			05	WF	216	5.2	3.1	
			05	WF	203	4.7	3.6	
		Carp	01	F	640	6.3	16.0	
			01	F	457	8.1	8.9	
			01	F	356	3.3	3.3	
			01	F	447	4.2	3.4	
			01	F	325	2.2	1.8	
		Northern Pike	01	F	610	2.2	6.0	
			01	F	541	1.7	2.8	
			01	F	432	2.9	1.6	
		White Sucker	01	F	455	2.9	3.5	
			01	F	417	5.5	3.6	
	8/78	White Sucker	01	F	419	4.7	2.7	
			01	F	399	10.0	5.0	
			01	F	386	5.2	9.2	
		Walleye	01	F	340	6.4	2.7	
			01	F	318	2.1	3.6	
		Yellow Perch	05	WF	173	3.1	1.8	
			05	WF	165	3.9	3.5	
			05	WF	203	3.3	2.0	
			05	WF	191	3.4	2.2	
			05	WF	178	3.2	1.8	
	9/78	Carp	03	WF	533	11.4	17.0	
		White Sucker	05	WF	406	7.4	9.6	
		Walleye	05	WF	404	6.9	7.9	

FORM		
F — Fillet WF — Whole Fish EP — Edible Portion		*Fish samples were screened for compounds other than PCBs. **Quantitated by matching to the nearest Aroclor or Aroclor mixture.

Table 17 (cont.)

Lower Fox River and Lower Green Bay Fish Contaminant Data

Sample Location	Date	Species	Quantity	Form	Length (mm)	% Fat	PCB** (mg/kg)	Other Chloro-organics or metals (mg/kg)
Little Lake Butte Des Morts (continued)	8/79	Brown Bullhead	10	F	236	4.7	1.3	
		Northern Pike	02	F	597	0.9	1.0	
		Walleye	03	F	389	2.7	1.4	
		Yellow Perch	10	F	224	3.0	0.3	
			15	F	180	1.1	0.6	
		*Northern Pike	03	WF	592	3.1	6.8	Mercury 0.22 Copper 0.70 p,p'DDE 0.06
		*White Sucker	04	WF	394	3.2	2.5	Chromium 0.50 Mercury 0.08 Copper 2.90
Kaukauna	8/78	Carp	01	F	432	10.5	57.0	
			01	F	450	6.7	26.0	
			01	F	485	5.2	17.0	
			01	F	523	1.3	38.0	
			01	F	429	6.1	22.0	
		Northern Pike	01	F	584	4.7	11.0	
		Yellow Perch	03	WF	152	0.7	3.5	
	6/79	Brown Bullhead	03	F	224	4.2	2.7	
		Carp	05	F	439	3.0	11.0	
		Northern Pike	02	F	490	0.6	1.0	
		White Sucker	04	F	333	1.8	1.4	
		Walleye	04	F	394	4.9	8.0	
		Yellow Perch	01	F	277	0.4	<0.2	
DePere (below dam)	5/77	Bowfin	01	F	648	0.4	0.5	
		Carp	01	WF	259	6.9	4.4	
			01	WF	325	1.6	6.6	
			01	WF	376	8.8	90.0	
			01	F	439	0.7	2.5	
		Northern Pike	01	F	498	1.0	3.0	
			01	F	531	0.7	3.2	
			01	F	455	0.5	2.5	
		White Sucker	01	F	432	2.3	4.2	
			01	F	429	0.6	1.4	
			01	F	483	0.6	2.5	
		*White Sucker	01	F	483	1.0	2.3	Dieldrin 0.008
		White Sucker	01	F	452	1.8	3.2	
			01	F	381	1.0	4.4	
		Walleye	01	F	330	4.9	4.5	
			01	F	452	2.6	6.8	
		Yellow Perch	05	F	203	1.0	1.0	
			04	WF	173	2.8	6.6	
			05	WF	185	3.2	5.4	
			05	F	196	2.6	5.3	
	8/78	Carp	05	WF	483	13.2	65.0	
			05	WF	406	6.2	14.0	
		Walleye	05	WF	457	10.0	25.0	
	9/78	Chinook Salmon Eggs				10.0	11.0	
		Chinook Salmon	01	F	871	4.2	9.4	
			01	F	914	5.8	12.0	
			01	F	876	4.3	9.1	
	4/79	*Carp	05	WF	518	9.0	17.0	Mercury 0.12 Copper 1.50 p,p'DDE 0.50
		*White Sucker	03	WF	387	3.6	5.9	Chromium 0.50 Mercury 0.12 Copper 2.10 p,p'DDE 0.14
		*Walleye	02	WF	401	10.0	16.0	Chromium 0.50 Mercury 0.14 Copper 1.80 Dieldrin 0.03 p,p'DDE 0.34

FORM		
	F — Fillet WF — Whole Fish EP — Edible Portion	*Fish samples were screened for compounds other than PCBs. **Quantitated by matching to the nearest Aroclor or Aroclor mixture.

Table 17 (cont.)

Lower Fox River and Lower Green Bay Fish Contaminant Data

Sample Location	Date	Species	Quantity	Form	Length (mm)	% Fat	PCB** (mg/kg)	Other Chloro-organics or metals (mg/kg)
DePere (below dam) (continued)		Walleye	01	F	386	1.3	3.7	
			01	F	434	2.6	3.3	
			01	F	290	1.4	3.3	
			06	F	239	2.6	1.5	
			02	F	305	3.3	3.2	
Fox River Mouth	10/79	*Carp	05	WF	460	11.0	5.8	Mercury 0.10 Alpha BHC 0.01 Copper 1.30 p,p'DDE 0.50
		*Walleye	05	WF	381	11.0	16.0	Mercury 0.14 p,p'DDE 0.06 Alpha Chlordane (Cis) 0.05 Trans-Nonachlor 0.05 Alpha BHC 0.01 Copper 0.80 Dieldrin 0.03 p,p'DDE 0.46
	4/79	*Yellow Perch	10	WF	178	3.2	8.4	Mercury 0.04 Copper 0.80 p,p'DDE 0.12
		*Brown Bullhead	05	WF	211	1.3	3.1	Mercury 0.06 Copper 0.90 p,p'DDE 0.10
			06	WF	213	2.3	5.4	Mercury 0.05 Copper 1.40 p,p'DDE 0.13
		*Carp	04	WF	439	11.0	6.1	Mercury 0.15 p,p'DDD 0.07 Alpha BHC 0.01 Copper 0.90 p,p'DDE 0.52
Lower Green Bay (Grid 1001)	4/77	Lake Whitefish	01	F	533	11.8	17.3	
		*Alewife	05	WF	187	4.4	7.4	Mercury 0.09 p,p'DDD 0.09 Copper 1.10 Dieldrin 0.05 p,p'DDE 0.63
		*Brown Bullhead	05	WF	231	2.9	8.5	Mercury 0.04 Copper 1.40 p,p'DDE 0.16 Gamma BHC 0.02
		*Carp	05	WF	366	8.0	5.9	Mercury 0.11 p,p'DDD 0.05 Copper 1.80 p,p'DDE 0.18
		*White Sucker	01	WF	269	6.0	4.4	Mercury 0.05 Copper 1.00 p,p'DDE 0.11
		*Yellow Perch	05	WF	165	7.5	8.5	Mercury 0.03 Copper 1.00 p,p'DDE 0.12 Gamma BHC 0.04

(Sullivan and Delfino, 1982)

FORM		
	F — Fillet WF — Whole Fish EP — Edible Portion	*Fish samples were screened for compounds other than PCBs. **Quantitated by matching to the nearest Aroclor or Aroclor mixture.

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