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CONTAMINATION OF POPULATED ESTUARIES AND
ADJACENT COASTAL OCEAN--A GLOBAL REVIEW

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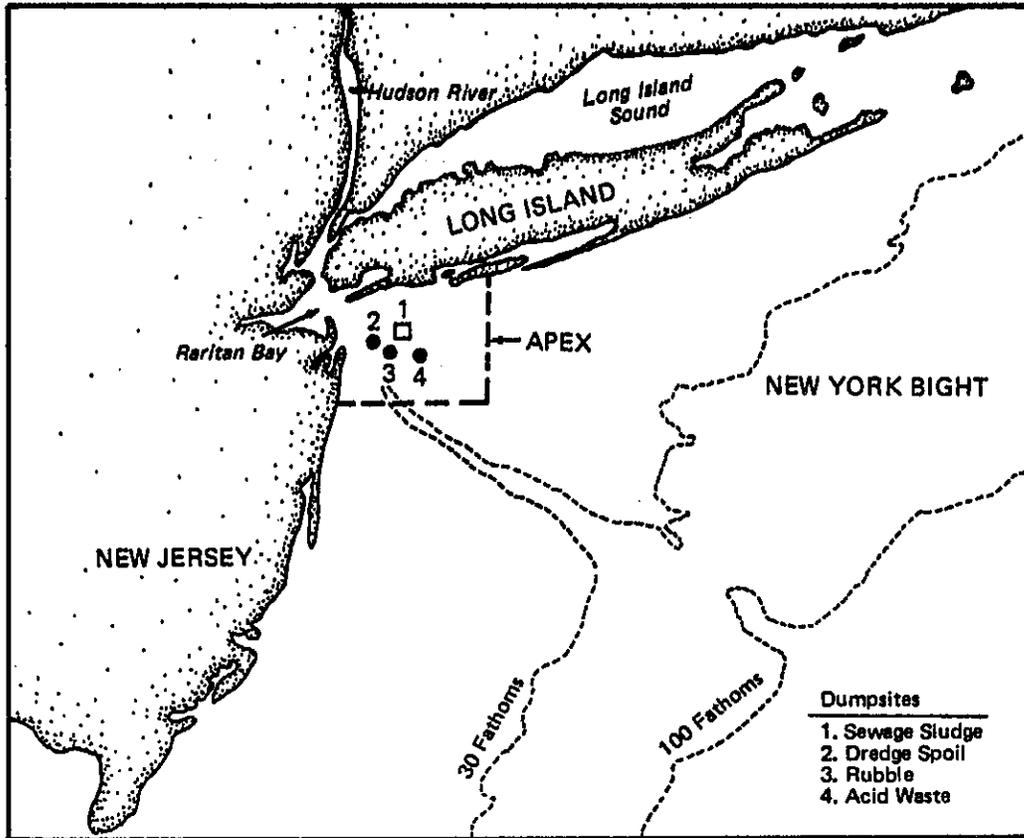
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The New York Bight

Contamination of Populated Estuaries and Adjacent Coastal Ocean--A Global Review

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ABSTRACT. A comparison of the contamination of coastal marine waters in the New York Bight with that of other populated coastal and estuarine marine areas demonstrates that the New York Bight does not appear to be more polluted, nor suffer from pollution impacts to any greater degree, than many other studied regions. Toxic contaminant levels, including concentrations of heavy metals and PCBs in sediments and water, found in the New York Bight are similar to those in many of the world's heavily populated coastal areas.

Based on population densities, the degree of industrialization, discharge volumes, and coastal hydrography, the New York Bight is not among the most pollution-susceptible regions of the world ocean (e.g., Saronikos Gulf, Guanabara Bay, Brazil, and New York, Singapore, and Hong Kong harbors), but, instead, is comparable to the large, but poorly flushed, semi-enclosed seas (e.g., the Baltic, Mediterranean, Irish, North, and Seto Inland seas). The total annual inputs of contaminants to the New York Bight are within the range of annual mass contaminant loadings to these and other comparable regions. While contaminant inputs may be similar, differences in hydrographic characteristics of these regions strongly influence the pollutant distributions in the ocean environment. For example, the Rhine-Meuse and Hudson-Raritan contribute similar quantities of contaminants to the ocean, but the much stronger tidal currents and dispersive forces of the Rhine-Meuse discharge area effectively reduce the ambient concentrations of contaminants. In contrast, contaminant inputs to the Saronikos Gulf in the Mediterranean are much smaller than those to the New York Bight, but contaminant concentrations are higher than the highest found in the New York Bight due to the restricted circulation of the Gulf.

Biotic concentrations of heavy metals and organic contaminants in the oceans, including New York Bight, do not appear to pose any substantive risk to human health, since they are rarely found to exceed established regulatory levels for these contaminants. Exceptions occur in localized areas, which receive large inputs of industrial and sewage discharges, such as such as portions of the Hudson-Raritan estuary.

Observed contaminant concentrations are universally below lethal toxicity levels for marine organisms except in extremely limited areas adjacent to contaminant discharges or ocean outfalls, or for short periods of time within an ocean dumping discharge plume. Sublethal effects on marine organisms caused directly by toxic chemical contaminants have rarely been demonstrated in the oceans themselves. Serious effects of pollutants on the world's oceans appear to be highly isolated and localized, and marine systems have demonstrated a high degree of recuperative ability when inputs of contaminants to a degraded area are reduced. Most ecological change, including change in species compositions of finfish and benthic fauna, is more likely caused by natural climatic variations, anthropogenic inputs of nontoxic organics or nutrients, and over-fishing. Anoxic events observed throughout the world are natural phenomena which in some areas, including the New York Bight, may be significantly exacerbated by nutrient-induced eutrophication, but not by input of sewage sludge.

Under appropriate conditions of introduction and dilution, the oceans are capable of accommodating large anthropogenic inputs with no significant effects on biota and human health, and without causing oxygen depletion. Southern California coastal waters have dispersive conditions which are perhaps ideal; however, contaminant concentrations immediately surrounding a Southern California sewage sludge outfall exceed those found in the New York Bight in the vicinity of a sewage sludge dumpsite. It is apparent that the method of introducing waste material to the ocean critically controls contaminant distributions in the discharge environment.

The New York Bight, including its apex region, is a moderately effective dispersive ecosystem, and contaminant concentrations are not extreme compared to other populated coastal areas of the world. The New York Bight appears to have suffered no irreversible ecological damage. However, more frequent anoxic events are likely if nutrient inputs to the sea should continue to increase.

I. INTRODUCTION

The urban-industrial complex of New York/New Jersey is but one of many centers of population and industrial development throughout the world that is located at the edge of the oceans. This concentration of the human race at the ocean edge was and still is motivated mainly by the demands of commerce bringing diverse raw materials to the centers of industry and distributing the finished products from these specialized centers of manufacture.

In addition to providing a means of commerce, the nearby ocean has often been a major source of food for coastal communities and a convenient place to dispose of the various waste products of an industrial society. Therefore, it is not surprising that many of the coastal regions of the world ocean have suffered from environmental problems and crises as the quantity of wastes discharged into the ocean has approached or exceeded the ability of certain restricted coastal ocean areas to assimilate these wastes.

The dramatically increased use of estuaries and coastal oceans as recreational resources in the past several decades has led to new concern about the quality of the ocean environment and the need for greater control over discharges that might affect human health and amenities. As a consequence of this increased public concern throughout many parts of the world, the last decade has seen a rapid intensification of the study of human impact on the coastal marine ecosystem. In the United States, the New York Bight Project of the National Oceanic and Atmospheric Administration (NOAA) is the largest of such studies.

The New York Bight project has dramatically improved our understanding of

the New York Bight ecosystem and the impacts that human activities have had upon that ecosystem. However, many other similarly impacted areas of the world ocean have been the subject of intensive study during this same period. It is the intent of this chapter to briefly review what has been found in some of these other studies and to compare those results with what has been found in the New York Bight (Fig. 1). Such comparisons reveal that there is a considerable commonality of problems in many regions of the coastal ocean and that the New York Bight, although heavily impacted, does not suffer more seriously from pollution than a variety of other coastal regions. In addition, comparisons between what has been observed in the New York Bight and elsewhere provide a better understanding of the observations made in the Bight and the probable consequences of alternative pollution management actions on the various perceived, present or future, problems of the Bight.

II. MARINE POLLUTION SUSCEPTIBILITY OF COASTAL OCEAN REGIONS

To identify the regions of the world ocean that are significantly impacted, three basic parameters involving any given stretch of coastline must be considered: 1) the population of the watershed region; 2) the degree of industrialization; 3) and the volume, flushing, and mixing characteristics of the affected segment of ocean. The first two factors are relatively easy to understand: the larger the population discharging domestic wastes and storm drain runoff into the ocean either directly or through rivers, the greater is the potential for the build-up of contaminants to levels at which adverse impacts can occur. Similarly, the greater the extent of industrialization, the greater is the quantity and diversity of toxic wastes released into the aquatic environment. These two relationships are found to be valid throughout the world, despite substantial recent efforts by many

countries to control the discharges of municipal and industrial wastes into the aquatic environment. Control efforts have been more extensive in the United States than in most other countries (Council on Environmental Quality, 1979), and yet the amounts of contaminants discharged to the New York Bight and other parts of the United States coastal ocean remain prodigious.

The third consideration determining the extent of human impact on the oceans is more subtle. Contaminants, including floating rubbish and microbes, pose a problem to man or to the marine ecosystem only if their concentrations in the environment reach certain levels. These critical concentrations differ for each contaminant and are somewhat different for each geographical location, since ecological sensitivity varies among different biological communities and human uses of the ocean vary from place to place. The critical concentration even varies seasonally, for example: 1) little or no beach bathing takes place in winter in some localities, and concern for pathogenic organisms in the surf zone is correspondingly less; and 2) larval forms of most marine animals occur only seasonally and often seem to be more sensitive to contaminants than their adult forms.

When contaminants are introduced into the coastal ocean, their concentration in that ocean is determined not only by how much of each contaminant is put in, but also by how rapidly they are dispersed in the coastal water into which they are discharged and how rapidly that segment of coastal ocean is mixed and flushed into the larger volume of the deep, open oceans. This consideration is true for both dissolved components and materials in the solid phase, although the latter would ultimately be mixed with the ocean bottom sediments.

As a consequence of the relationships described in the preceding paragraphs, the aquatic environments most likely to be adversely affected by pollution are those with large populations and neighboring industries, and those with restricted circulation and flushing. In Table 1, a representative selection of aquatic environments is listed with some indication of their potential for pollution problems based on these factors. Although arguments could be made to change an area's factor ranking as assigned in Table 1, the overall picture is clear: the regions of greatest concern are enclosed bays, harbors, estuaries, or gulfs sustaining large population and industry. These are followed by the large enclosed seas such as the Baltic and the Mediterranean, and then by open coastlines with poorly flushed bottom waters such as Southern California and by other dynamically-flushed, semi-enclosed seas such as the Irish Sea and the North Sea, all of which sustain heavy populations and extensive industry. Finally, the areas of least concern include the semi-enclosed seas such as the Red Sea and Persian Gulf with small populations and industry, and open coastal ocean areas with low populations. However, it should be noted that extreme events can lead to severe pollution problems even in regions with low susceptibility: for example, the 1983 oil spills within the Persian Gulf, uncontrolled because of the Iran-Iraq war.

The New York Bight is not included in the "most vulnerable" category composed of highly enclosed areas receiving heavy pollution loads, although Raritan Bay and New York Harbor are in this designation. Therefore, comparisons for the remainder of this study will be drawn mainly from the large, but poorly flushed, semi-enclosed seas such as the Baltic, Mediterranean, and Seto Inland Seas and the more dynamically flushed Irish and North Seas. The physical setting of the New York Bight is closer to the latter group, but, in view of the very large population and industry of the

Bight and the relatively restricted flushing of its apex, the potential of the Bight for contaminant concentration build-up may be more nearly that of the semi-enclosed seas of the Baltic and Mediterranean.

III. SELECTED POLLUTION-IMPACTED COASTAL OCEAN REGIONS

Before considering the observed impacts of various contaminant classes on selected regions of the world ocean, a general background must be set to explain the physical nature of these selected regions. It is against this background that the extent of marine pollution and its impact in New York Bight and other ocean areas can be brought into focus.

A. The Mediterranean

The Mediterranean (Fig. 2) is a very large semi-enclosed sea whose borders support a total population of about 100 million people (Table 2). The largest city is Cairo, which numbers an estimated 10 million people. No other city has more than 5 million population. The majority of the population is spread along the northwestern coastlines in smaller cities and towns (Henry, 1977). Four major rivers, the Rhone, the Danube, the Nile, and the Dneiper, flow directly or indirectly into the Mediterranean. Riverine and other coastal inputs of water and an inflow of surface North Atlantic water through the very narrow Strait of Gibraltar are balanced by an outflow of water through the straits and evaporation. Because there is an excess of evaporation relative to precipitation, Mediterranean water increases in density as it circulates from the Strait of Gibraltar along the North African and Levantine coasts to the Aegean Sea. From the Aegean Sea, heavy, highly saline water flows northerly through the Dardanelles, the Sea of Marmara, and the Bosphorus to the anaerobic lower portion of the Black

Sea. Flowing south in return from the Black Sea are the less dense, brackish surface waters which carry runoff from the many rivers entering the Black Sea, including the Danube.

The surface currents in the Mediterranean are slow and, as the tidal range is also very small, mixing within the Mediterranean is regionally very poor. Particularly poorly flushed regions include the Northern Adriatic and the Saronikos Gulf (Fig. 1), two regions where population and industry are also concentrated.

B. The Baltic

The Baltic (Fig. 3) is a much smaller sea than the Mediterranean (Table 2) and is separated from the North Sea by a shallow sill at the narrow passage between Denmark and Sweden. Because the volume of freshwater runoff is large, surface salinities are low (approximately 7 parts per thousand in the central portion of the Sea). Surface water leaves the Baltic through the Kattegat Strait, while water in the deep basins of the Baltic is only replaced by periodic inflows of saline water over the Kattegat sills. The halocline between surface and lower layers is permanent, and the flushing rate of the bottom waters is slow. These bottom waters suffer from anoxic periods of several years duration between periodic flushings with North Sea waters.

The Baltic shores and the banks of rivers entering the Baltic are heavily populated and industrialized, particularly in the southern half of the sea.

C. The Irish Sea

The Irish Sea (Fig. 4) is a semi-enclosed sea open at the north and south to

the Atlantic Ocean. Its area and volume are similar to that of the New York Bight (Table 2). However, unlike the New York Bight inner shelf and apex, it has little density stratification during summer. The population surrounding the Irish Sea is less than half that in the New York region and is spread among several estuaries, the two largest of which empty into the Firth of Clyde and into Liverpool Bay. The entire region, particularly the Liverpool Bay area, is heavily industrialized.

D. The North Sea

The North Sea (Fig. 5) is approximately the same size and volume as the Baltic (Table 2), but it is well flushed through its northern boundary with the Atlantic Ocean and Norwegian Sea and its narrow but dynamic passage into the English Channel. The southern North Sea coast is heavily populated and industrialized and receives discharges from several major rivers draining industrial areas, including the Rhine and the Thames. Like the Irish Sea, the southern North Sea has weak or nonexistent density stratification during summer. However, the northern part of the North Sea is more like the New York Bight: well mixed in winter with a seasonal summer thermocline.

E. Seto Inland Sea

The Seto Inland Sea (Fig. 6) is a narrow body of water surrounded by islands of the Japanese chain. It is open to the Pacific Ocean through narrow passages at its eastern end and to the Pacific Ocean and the Sea of Japan at its western end. The Inland Sea contains many small islands and semi-enclosed bays and is one of the major industrial areas of Japan. The heaviest concentration of population and industry is around Osaka at the

more poorly flushed eastern end of the Inland Sea.

F. Southern California Coastal Zone

The Southern California coastal zone, offshore from Los Angeles (Fig. 7), is unlike any of the other areas considered above. It is totally open to the Pacific Ocean, although the principal nearshore currents move north and south such that advection offshore is not favored and there are semi-enclosed deep basins offshore where the near-bottom waters are poorly flushed. The Los Angeles area is not heavily industrialized, although the specialty industries that are located there do generate waste products including some of the rarer trace metals. The southern California coastal ocean has no significant river input or other discharge, and no significant ocean dumping. This area is useful in our comparisons because it offers the chance to look at the effects of essentially a single contaminant source on an oceanic ecosystem. This source is the municipal waste discharged through submarine pipelines, the dominant source of contaminants entering these waters (Schafer, 1982).

IV. TOXIC CONTAMINANTS IN THE COASTAL MARINE ENVIRONMENT

Perhaps the central public concern regarding the effects of man on the New York Bight ecosystem is the fear that toxic substances, trace metals, and synthetic organics, introduced by dumping and other pathways, may be concentrated in fish and shellfish to levels that constitute a human health hazard, or that these substances may be present in concentrations high enough to harm natural ecological functions.

Three basic questions need to be answered in order to understand and control impacts of toxic substances on the marine environment and on humans through

seafood ingestion. First, where do the toxic substances come from and in what quantities are they introduced? Second, how are these toxic substances transported, degraded, and distributed in the estuarine/coastal ecosystem? Third, given the observed distribution of contaminants in an ecosystem, what are the ecological effects and the human health effects through intake of toxics in seafood? Nowhere in the world have these questions been completely answered even for a single toxic compound, much less for the entire spectrum of compounds discharged by man's activities. Nevertheless, substantial progress has been made in many parts of the world in estimating inputs of some toxic substances to the oceans. Although knowledge of transport routes, particularly transfers within the biosphere, is poor, many studies also have been performed to delineate the distribution of toxic substances in sediments, water, and biota of various ocean areas. In addition, a number of ecological changes have been observed that have been ascribed tentatively to toxic substance effects.

Human health effects due to marine pollution have been observed infrequently and have been caused almost exclusively by sewage-associated microbial pathogens. Adverse human health effects due to chemical contamination of marine waters have been documented only in rare instances such as Minimata Bay, Japan, where a single contaminant source led to extremely high levels of mercury in fish consumed by the local population. However, perceived or potential risk to human health has resulted in the closure to fishing of certain estuarine areas due to chemical contamination. For example, portions of the Hudson-Raritan estuary and New Bedford Harbor have been closed to fishing due to PCB contamination (Belton et al., 1983; Massachusetts Office of Coastal Zone Management, 1982).

A. Inputs of Toxic Contaminants to the Coastal Ocean

Information concerning inputs of various toxic substances to specific local areas of the ocean is extremely valuable in assessing the possible extent of contamination within that region. In Table 3, such information has been collected for several trace metals and, where possible, for some synthetic and petroleum organics. It should be pointed out that many of the sources of information used in arriving at the numbers in Table 3 are incomplete and possibly subject to large errors. Despite this uncertainty with respect to individual data points in Table 3, some general observations can be made that place the New York Bight problem in perspective.

The total inputs (not including atmospheric contribution) of toxic metals to the New York Bight are comparable to, or less than, the total inputs to the Mediterranean, Baltic, North Sea, and Irish Sea. Inputs to some of these other areas are widely distributed within the region and are not concentrated within a very small area such as in the Bight apex. However, there are many other areas throughout the world where inputs are significantly concentrated, including the Mississippi discharge zone, the Rhine-Meuse estuarine discharge area off the Netherlands (Fig. 5), the Thames estuary in the North Sea (Fig. 5), and the Liverpool Bay area of the Irish Sea (Fig. 4). Estimates of contaminant inputs for these selected regions are shown in Table 4. Although the Rhine-Meuse discharge region has inputs of the same order of magnitude as that of the New York Bight apex, the Rhine-Meuse discharge area is dramatically different. Tides in the discharge area offshore from the mouth of the Rhine-Meuse range up to several meters in amplitude with tidal currents up to several knots; this is compared to tides of about 1 meter and only a few tenths of a knot in the New York Bight. The effects of the larger tidal currents on mixing are

dramatic. Although the climate and depth of water off the Dutch coast are similar to that in the New York Bight apex, the entire southern North Sea (from the Dutch coast to the English coast and north for several hundred kilometers into the central part of the North Sea) remains vertically homogeneous throughout the year with neither thermal nor haline stratification.

Atmospheric fluxes of contaminants into the coastal marine environment may be large. While not reflected in Table 3, recent estimates suggest that atmospheric contributions (which include both anthropogenic and natural material) of Cd, Zn, and Pb to the New York Bight are 3 to 20 times greater than riverine contributions (Windom, 1981). Atmospheric contributions to the South Atlantic Bight of the U.S. (coasts of North and South Carolina and Georgia) and the North Sea may also exceed riverine inputs for these trace metals, as well as for As, Cu, Hg, and Ni (Windom, 1981). However, it must be noted that, unlike river inputs, atmospheric inputs are not localized and would not be expected to cause high contaminant concentrations in local areas of the ocean ecosystem.

From these considerations it might be concluded that the New York Bight has greater potential for significant accumulation of anthropogenic toxic metals, both in the water column and sediments, than any of the European seas because its inputs are high, the discharge area small, and the rate of dispersion and removal relatively low. In the European waters any significant toxic metal accumulations in either water or sediments must be more localized than accumulations in the Bight and take place only in small areas of poor circulation. The primary exception to this rule is the potential long-term pollution problem in the Baltic, where inputs

(particularly of Hg) are high and flushing is poor.

It is interesting to note that, contrary to popular belief, the possibility of serious damage to the Mediterranean as a whole from toxic metal pollution during the next few years is very unlikely because of the moderate input levels and the very large size of the Mediterranean itself. Nevertheless, certain areas of the Mediterranean are substantially more at risk, notably the northern Adriatic and Saronikos Gulf (Fig. 2), which sustain large populations, have large inputs, and are very poorly flushed.

Input data for metals to the Seto Inland Sea are not currently available. However, from Table 5, it can be seen that there is a rough proportionality between the population of a region and the quantity of metals discharged to the oceans. If these relationships hold true for the Japanese population, the toxic metal inputs to the Seto Inland Sea are roughly equal to the inputs to the New York Bight. The restricted volume for dispersion and the poor flushing of the Seto Inland Sea would indicate a high probability that accumulation levels of toxic metals are significantly higher in the Seto Inland Sea than in the New York Bight. The Seto Inland Sea may well provide a good model for the future of the Bight, if inputs of toxics to the Bight increase. However, comparison of the estimates of Mueller et al. (1976) and New York City (1983) suggests that the total inputs of many toxic contaminants to the New York Bight have decreased in the last decade.

Despite the high priority that has been placed in recent years on research into the fate and effects of toxic organic compounds, both naturally occurring (primarily petroleum hydrocarbons) and synthetic (primarily organochlorine compounds, including DDT and analogous insecticides, and PCBs), there is very little data from which inputs of these compounds to

various coastal marine ecosystems can be estimated.

Le Lourd (1977) has estimated that approximately four million tons of oil per year enter the ocean environment worldwide and that about one quarter of this amount is discharged into the Mediterranean. Because no surface water flows out of the Mediterranean, it is likely that all surface slicks and tar balls released into the Mediterranean will remain there until dissolved, decomposed, or washed up on beaches. These considerations have led Le Lourd (1977) and Osterberg and Keckes (1977), among others, to suggest that the Mediterranean may be more affected by oil pollution than any other body of ocean water in the world. However, even though input data for the New York Bight are far from complete, Mueller et al. (1976) have estimated that 320,000 tons of oil and grease annually enter the New York Bight mostly in municipal wastewaters and dredged materials, while New York City (1983) has estimated a total input of only 206,000 tons annually. If approximately one-half of this was petroleum-derived, the amount of oil entering the Bight would be 10-15% of that discharged into the Mediterranean, a very large figure considering that atmospheric inputs and spills are not included. Another way of illustrating the potential problem in the New York Bight is to compare the 180 million tons/yr of petroleum products passing through the Port of New York (U.S. Army Corps of Engineers, 1975) with the 370 million tons/yr crossing the Mediterranean and the worldwide marine transport of approximately 1700 million tons/yr.

Much better input information for petroleum organics and synthetic organics is needed for all impacted regions of the world ocean. Such data could be used effectively to aid assessment of the vulnerability of a given marine ecosystem, as attempted for the toxic metals (above) and nutrients (see Section V). Without adequate input data, no estimate of future accumulation

(or loss) rates of contaminants in the marine ecosystem can be made, even if the transport and decomposition of the contaminants within the environment are well understood.

B. Distributions of Toxic Contaminants in Coastal Ecosystems

Although the analytical methodology for determination of many contaminants in the marine environment is in some instances difficult, studies of contaminant distribution in a given ecosystem are simpler to perform than studies that seek to establish the rates of transport or transformation, or the effects of contaminants. Therefore, there is a massive body of data and literature concerning the concentrations of contaminants, particularly trace metals, in marine environments throughout the world. Selected data are useful in illustrating how contaminant levels in the Bight compare to those in other regions.

1. Trace Metals

The ocean has been found to contain measurable concentrations of almost all of the stable elements. More than 30 of these elements are metals or metalloids present only in trace quantities in sea water, sediments, and biota. All of these trace elements are known to be ultimately toxic if present in high enough concentrations, while many of these same elements are absolutely essential for the growth of plants and animals. For example, it has been shown that the nonavailability of certain trace metals in some parts of the ocean limits the rate of phytoplankton growth (Strickland, 1965), and several elements (e.g., Cu, Zn, Cr, Se), which are known to be toxic or inhibitory to marine organisms at certain concentrations, are also known to be essential to life, including human life, at lower

concentrations.

Three elements have been consistently selected as those of special concern in environmental pollution: Cd, Hg, and Pb. Two others, Zn and Cu, have been used consistently as indicators of pollution because of the relative ease of their analysis in environmental samples. We will focus on these five elements in the following discussion.

a. Water Concentrations

Analysis of ocean water samples for trace metal concentrations is a difficult undertaking, prone to sample contamination and other errors. Therefore, intercomparison of data sets from different regions of the world obtained by different techniques is not always reliable. However, in Table 6, some information is collected for various nearshore and estuarine areas. Careful examination of the data in Table 6 reveals that, while considerable variability exists, there are no obvious differences apparent among different regions of the coastal oceans of the world (including the New York Bight) with respect to these and probably other trace metals. No area stands out as significantly more contaminated than any other. However, in all other areas, just as in the New York Bight, there is a tendency for the highest metal concentrations to be found in and near the estuarine discharges, particularly those estuaries that are subject to industrial discharges.

It would require enormous effort to standardize methodology, intercalibrate, and intensively sample different regions to eliminate the variability due to sampling and analytical differences and the temporal and spatial heterogeneity of dissolved trace metal concentrations. Without such

efforts, little is to be gained by drawing detailed conclusions based on comparisons of concentration ranges or means that show only small differences between different regions of the world ocean. However, it is important that the New York Bight does not seem to have unusually high dissolved trace metal concentrations, according to the limited data available.

b. Sediment Concentrations

Metal concentrations in marine sediments are more reliably determined than dissolved concentrations in seawater. Since the sediments may act to integrate contaminant loadings, sediment concentrations are often used to assess the level of environmental metal contamination. Some sediment metal concentrations from selected coastal and estuarine regions of the world are listed in Table 7. The highest trace metal concentrations in the New York Bight sediments are found in sediments from the Christiaensen Basin and the Hudson Shelf Valley (Camody et al., 1973; Greig et al., 1974).

Concentrations in these sediments are significantly higher than those found in many other coastal regions, including some of those thought to be heavily polluted. However, there are certainly other areas not necessarily thought of as heavily polluted (e.g., the Saronikos Gulf, the Solent, the Severn, the Firth of Clyde, Tokyo Bay, the southwest Spanish and Portuguese coast, and the area close to the 7-mile Hyperion sewage sludge outfall in Southern California), where trace metal concentrations are as high as, or higher than, those found in the New York Bight (Table 7).

Many factors other than contaminant loading can affect the metal concentrations of marine sediments at any given location in the ocean. For

example, it is well known that natural fine-grained sediments contain higher trace metal concentrations than coarse sands. The mid-ocean regions, far from supplies of coarse sands, thus have high trace metal concentrations (Table 7) despite their low organic content. In the coastal regions, the lowest metal concentrations are found in clean, coarse sands and the highest generally in areas where fine-grained material accumulates. Therefore, the coastal regions that are most sensitive to anthropogenic inputs in terms of elevated concentrations in sediments are those regions where the sediments are naturally fine-grained, including topographic depressions, wetlands, and regions of very poor circulation. It is not surprising, therefore, that the metal-rich sediments of the New York Bight are located in the Christiaensen Basin and the Hudson Shelf Valley, where fine-grained sediment tends to accumulate. Sediments within and immediately to the east of the sewage sludge dumpsite have much lower metal concentrations (Carmody et al., 1973), well within the normal range of values found in other coastal regions not subject to severe pollution.

Sediment metal concentrations are controlled both by natural and anthropogenic inputs and by the transport dynamics of fine sediments within the region. By examining in a semiquantitative fashion the nature of these factors for some of the regions in Table 7 that have sediments with high metal concentrations, we can gain some insight as to the relative severity of trace metal pollution in the New York Bight. Each of the areas that have sediments with high trace metal concentrations is also an area where contaminant inputs are high. In each of these locations, the inputs include various forms of sewage (raw sewage, primary or secondary effluents, or sewage sludge), and a variety of industrial inputs largely discharged through the municipal sewage system. In several of the areas,

there are also large inputs of dredged material, much of which is contaminated with industrial and domestic wastes. Therefore, the sources of substantial proportions of the metals involved are industrial, although one major means of delivery to the marine environment is sewage discharge. For example, on the Firth of Clyde there are extensive iron, steel, and shipbuilding industries; in the Saronikos Gulf, petroleum and petrochemicals, leather tanning and finishing industries; in the New York region, petroleum, and petrochemicals and metal finishing industry; in the Solent, shipyard, petroleum and petrochemical industries; and in the Los Angeles area, the electronic component industry.

The few areas without major population densities and municipal discharges where high sediment metal concentrations are found are generally the result of inputs of a single specific industry. In most instances, this industry is the chlor-alkali industry, the primary contaminant is mercury, and the mercury-contaminated sediments are limited to a small area close to the discharge point.

The effect of flushing on the accumulation of metals in sediment can be seen by contrasting the metal concentrations in the New York Bight sediments with those in the Saronikos Gulf. The highest sediment metal concentrations in the Saronikos Gulf are as high as, or higher than, those observed in the most contaminated areas of the New York Bight, despite an Athens-area population of about one-tenth that of the New York City region, a correspondingly smaller industrial concentration, and a somewhat more dispersed industrial discharge than in the New York region (where most such discharges enter the Bight through Raritan Bay). However, the area of contaminated sediments in the Saronikos Gulf is only about one-third to one-half as large as that in the New York Bight. The critical differences

between the two regions are the poorer circulation and dramatically smaller tidally-induced dispersion in the Saronikos Gulf which ensure that a greater fraction of the discharged metals remain in the sediments near the disposal site.

Another aspect of the complex of physical factors determining metal concentrations in sediments is illustrated by the differences between the discharge mechanisms and resultant sedimentary metal concentrations in the New York Bight sewage sludge dumpsite and the 7-mile Hyperion sewage sludge outfall off Southern California (Fig. 7). The 7-mile Hyperion outfall receives sewage sludge and an associated trace metal load equivalent to about one-quarter of that dumped into the New York Bight, but diluted in a volume approximately ten times larger. Dispersion and transport of the metal-containing sewage solids away from the discharge area in Southern California are excellent. Galloway (1979) has estimated that "less than 10% of the metals injected from the outfall are present in the marine sediments of the study area" (some hundreds of square kilometers). Nevertheless, the highest metal concentrations (Table 7) found within a few hundred meters of the outfall exceed the highest values found in the New York Bight. While there clearly is no "containment" of the metals discharged from the Hyperion outfall, the placement of the discharge pipe at the head of a marine canyon, with its high potential for short-term, fine-sediment accumulation between major flushing events (including turbidity currents), and the confinement of all discharges to the same fixed location has concentrated the impacts of the discharge on the sedimentary environment of the discharge zone. By contrast, successive barge dumps of sewage sludge take place over a fairly large area in the New York Bight and result in good initial dispersion of fine particulate material. Accordingly, trace metal concentrations in New York Bight sediments

do not exceed those off Southern California, even though the long-term flushing efficiency of the Bight apex may well be poorer than that of the California discharge zone, and despite the very large additional loading to the New York Bight of trace metals from other sources, including dredged material and river runoff.

These considerations have a bearing on arguments concerning the desirability of concentration and containment, as opposed to maximized dispersion during sewage sludge and other ocean dumping. The restriction of dumping to one site tends to maximize impacts within a small region, while not ensuring containment of the toxic components of the dumped material at that site. Trace metals, nutrients, and organic matter (although not certain synthetic organic chemicals) are all introduced naturally by rivers and through the atmosphere to the coastal ocean. The anthropogenic loads of these chemicals are small compared to the quantities naturally mobilized and/or present in the oceans. For example, natural inputs to the ocean substantially exceed anthropogenic inputs for such metals of concern as Hg, Zn, and Cu (Goldberg, 1974). Under the proper conditions of introduction and dilution, even large anthropogenic loads can be accommodated by the ocean with no significant effects on trace metal concentration within the ocean ecosystem. For example, if the sludge discharged through the Hyperion outfall were instead barged the same 7 miles to sea and dumped over an area of several hundreds of square kilometers, changes in bottom sediment metal concentrations would be small and within the natural variability of coastal sediments. This has been the experience in other locations such as Liverpool Bay, the Thames estuary, and the English Channel where dispersion and flushing are good (Department of the Environment, 1972, 1978; Eagle et al., 1979).

In summary, a significant area of sediments in the New York Bight have trace metal concentrations which are measurably higher than those found in similar uncontaminated sediments. However, these concentrations are by no means unusual or extreme compared to those found in fine-grained sediments of many other populated coastal regions, nor are they different from concentrations observed in deep-sea clays, particularly those on or near active mid-ocean ridges (Table 7).

c. Biotic Concentrations

The concentrations of metals in organisms, particularly fish and shellfish used for human consumption, are of more direct relevance in considerations of both human and ecological health than are either dissolved or sediment metal concentrations. It is difficult to compare metal concentration data for organisms from one marine region with those from another region, as the species in different areas vary. Grieg et al. (1977 a,b), Grieg and Wenzloff (1977), Grieg et al. (1978), Wenzloff et al. (1979), and Roberts et al. (1982) have reported on the metal concentrations in phytoplankton, zooplankton, finfish, and shellfish from the New York Bight and adjacent coastal regions. With only a few exceptions, all the samples analyzed showed metal concentrations in these organisms were not significantly higher in the contaminated apex region than in other parts of the Middle Atlantic seaboard. Additionally, Grieg et al. (1978) reported metal concentrations in the species studied comparable to those found in similar species in other regions of the ocean. Wenzloff et al. (1979) did note a slight decrease from the Long Island to North Carolina coasts in silver, arsenic, and copper concentrations in certain clams, but no trend was apparent for other heavy metals. The levels of heavy metals in the surf clam and ocean quahog were considerably below the U.S. Food and Drug Administration (USFDA) action

level for mercury and also below the National Health and Medical Research Council (Australia) recommended maximum for cadmium, copper, lead, and zinc (Wenzloff et al., 1979). Carpenter (1978) has shown some evidence for somewhat higher average mercury concentrations in winter flounder from the New York Bight apex than from areas farther offshore, but all levels were well below the USFDA action level.

Significant differences between trace metal concentrations in contaminated areas and those in uncontaminated areas have been reported for other regions. For example, Grimanis et al. (1978) have reported elevated concentrations of arsenic (As), but not other metals, in one of two species of fish from the Saronikos Gulf near the Athens sewer outfall. Also, Halcrow et al. (1973) have reported that the whelk, Buccinum undatum, from the sewage sludge dumping area off the Clyde estuary had higher Zn, Ni, and Pb than did those from a coastal area. However, many of these reports are of questionable value because of the large, natural temporal and geographical variability in metal concentrations in marine organisms and the limited data sets involved. The metal concentration in an individual organism has been shown to depend not only on the concentration of metal in its habitat, but also on other factors, such as its age, sex, weight, the season, food availability, salinity, and amount of contaminated sediment in the gut during analysis (Phillips 1977a). In addition, most metals are present in marine waters and sediments in a variety of physical and chemical forms, only some of which are biologically available.

This multiplicity of variables controlling the concentration of metals in an organism can pose serious problems when comparing contamination in different areas, as illustrated by the experiences of the Mussel Watch program

(Goldberg et al., 1978). The idea behind this program was that the bivalve mollusc, *Mytilus edulis*, the common blue mussel which is ubiquitous throughout the North Atlantic and its adjacent seas, could be used as a "sentinel" organism to indicate changing levels of pollutants in coastal waters. The mussel was chosen as the sentinel organism because of its wide range and static lifestyle. It was postulated that data from mussels from different areas would serve as measures of the differences between these areas in pollutant loads and bioaccumulation. Where the blue mussel was not found, similar bivalves could be used whose ranges overlapped those of the blue mussel. Initial results from the Mussel Watch and similar programs have been promising. Metal concentrations in mussels from areas known to be contaminated have been found generally to be higher than those from non-contaminated areas (e.g., Phillips, 1977a,b, 1979; Capelli et al., 1978; Farrington et al., 1983). However, in other studies, the variability due to factors other than the environmental contamination level has obscured the regional differences (e.g., Karbe et al., 1977).

A technique developed by Davies and Pirie (1978) may eliminate many of the problems associated with the use of natural populations as indicators of the degree of bioavailable contaminants. In this technique, genetically uniform specimens of *Mytilus edulis*, carefully bred and grown in a controlled environment, are used for in situ bioassays of metal concentrations. Cages with these mussels inside are moored in the environment under study for a period of 20 days and then sampled. After this period of time, mercury concentrations of the mussels were directly proportional to the total mercury concentration of the estuarine water in which they were immersed. This method has proven to be applicable to other metals and organic contaminants and can be extended to other species (Phelps et al., 1980;

Martin et al., 1982; Phelps et al., 1983). It, therefore, provides a simple, relatively inexpensive method of estimating the time-averaged concentration of biologically available contaminants in a given environment and of monitoring the long-term changes in these concentrations.

The metal mercury deserves special attention with regard to its concentration in seafood. It is the only metal known to have caused human health problems (Minamata disease) contracted through ingestion of contaminated seafood (Smith and Smith, 1975). Therefore, many surveys of Hg concentrations in edible fish and shellfish have been performed (e.g., Anonymous, 1978; Yonnai and Sachs, 1978; Aubert, 1975; Roberts et al., 1982; Sherlock et al., 1982). Except in small areas subject to large inputs of bioavailable mercury (primarily from the chlor-alkali and pulp and paper industries), the mercury concentrations in marine organisms are comparable in similar species throughout the world ocean. Carnivores and larger fish tend to have higher mercury concentrations than other species. Tuna and swordfish, two species living in deepwater far from coastal pollution, tend to have the highest concentrations (Barber et al., 1972). Fisheries in some Japanese coastal waters (FAO, 1976) and certain coastal areas of the Baltic (Somer, 1977) in the vicinity of industrial mercury discharges have been closed to protect human health. Although Hg analyses of marine organisms from the New York Bight are relatively few, it appears that no current threat to human health exists from even abnormally large consumption of any seafood sampled from the Bight (Saila and Segar, 1979; Roberts et al., 1982). However, since natural mercury concentrations in marine organisms may be high and approach levels that can constitute a health hazard, continued minimization of mercury discharges and monitoring of Hg in seafood in the Bight should be supported. In this regard, New York City (1983) has

reported an estimated 65% decline between 1973 and 1981-82 in the total annual mercury load entering the New York Bight through sewage sludge dumping. This reduction is presumably due to controls on industrial discharges of mercury to the municipal sewers.

2. Synthetic Organics and Oil

Studies of the distributions of synthetic organics and oil in the marine environment are much more limited than studies of the distribution of trace metals. O'Connor and Stanford (1979) have named several classes of organic compounds as potential or perceived threats to the New York Bight ecosystem: polynuclear aromatic hydrocarbons (PNAHs), polychlorinated biphenyls (PCBs), chlorinated pesticides (aldrin, dieldrin, chlordane, DDT, endosulfan, endrin, heptachlor, hexachlorobenzene and toxaphene, and these compounds' various metabolites), chlorobenzenes, chlorophenols, diphenylhydrazine, halogenated diphenyl ethers, isophorone, low molecular weight halogenated hydrocarbons (carbon tetrachloride, chloroform, etc.), and petroleum hydrocarbons other than PNAHs.

Of these compounds only the concentrations of chlorinated insecticides and PCBs have been determined in a wide range of samples from a variety of locations throughout the world. The PNAHs and petroleum hydrocarbons other than PNAHs have, perhaps surprisingly, not been extensively determined. The primary reason for this dearth of information on petroleum compounds, despite intense worldwide concern about oil pollution, is that petroleum is an exceedingly complex mixture of individual hydrocarbons. Although simple, cost-effective analytical techniques exist that can determine total hydrocarbon concentrations in environmental samples (Maher, 1983), they have not been adopted worldwide. In any event, such measurements of total

hydrocarbons are difficult to interpret, since only some subgroups or individual compounds within the complex mixture affect the overall toxicity of crude or refined petroleum.

Considerable variation exists as to hydrocarbon fractions measured in individual studies. For example, Zsolnay (1979) used a complex calculation based upon gas chromatographic analyses of extracted water samples to estimate both "biogenic hydrocarbons" and "petroleum hydrocarbons"; Brown and Huffman (1979) determined "total hydrocarbons"; Portman and Preston (1975) measured "total C14 to C24 n-alkanes"; and Zsolnay et al. (1978) measured "tar" and "aromatic hydrocarbons." Therefore, it is simply not possible from the data presently available to make detailed comparisons of petroleum hydrocarbon or PNAH contamination in different regions of the world ocean. However, several limited comparative observations have been made that are worthy of note. Zsolnay et al. (1978) report that the concentrations of PNAHs (expressed as "aromatic hydrocarbons") in Baltic Sea water are almost twice as high as those in the Mediterranean and almost 10 times higher than those in the northwest Atlantic, including the Sargasso Sea. By contrast, the Mediterranean is reported to be more heavily polluted with surface-floating tar balls than both the Baltic and the northwest Atlantic. It would seem that there is little or no correlation between the quantity of tar derived from oil spills (and, by inference, the quantity of oil spilled) and the degree of contamination by PNAHs, the most toxic components of petroleum. Indeed, Zsolnay et al. (1978) found no such correlation in a group of samples taken from throughout the Mediterranean. In view of this lack of correlation, the finding that tar ball and total hydrocarbon concentrations of surface seawaters are highest along tanker routes, particularly in the Mediterranean (Myers and Gunnerson, 1976), is perhaps of only limited ecological significance.

Municipal waste treatment plant effluents (Van Vleet and Quinn, 1977b, 1978; Whipple et al., 1976; and Tanacredi, 1977), as well as a variety of land-based, non-point sources (Kneip et al., 1982), provide major inputs of petroleum hydrocarbon contamination to coastal waters, and a substantial proportion of the total hydrocarbon load is probably PNAHs. This, coupled with the apparent lack of correlation between spilled oil and PNAHs described above, would indicate that more attention ought to be focused on dispersed input of petroleum to the marine environment, rather than the admittedly more dramatic oil spills.

Comparatively little is known about the occurrence of petroleum hydrocarbons in fish and shellfish. However, it is noteworthy that both aliphatic and aromatic petroleum hydrocarbons could not be detected in lobster muscle tissue from the New York Harbor (Gravesend Bay) and New York Bight (Roberts et al., 1982).

PCBs and chlorinated insecticides, such as DDT, are readily determined in environmental samples, and there is a profuse literature concerning the concentrations of these substances in seawater, marine organisms, and sediments. Tables 8 and 9 show results of PCB analysis of seawater and sediment samples, respectively, from various locations. It can be seen that the variation in concentrations from one region to another is quite large, suggesting that the Baltic, Southern California, New York Bight, Japan, and the French Mediterranean coasts may be more heavily contaminated than the other areas studied. However, comparisons like this must be made with care, since intercalibration of the techniques used by individual investigators has not been performed and, at the very low concentrations of PCBs in seawater, there is a strong possibility that much of the data available is

unreliable due to sample contamination or other analytical errors. The same considerations hold for data concerning chlorinated insecticide concentrations in seawater and, therefore, no data are compiled here.

However, two general findings should be noted concerning the distribution of PCBs and chlorinated insecticides in the ocean: both chlorinated pesticides and PCBs are reported to be concentrated in the surface microlayer of the sea (surface slick) compared to the underlying water column (Stadler, 1977), and both vary widely in sediment depending on location and upon the sediment grain size (Dawson and Riley, 1977). Both PCBs and chlorinated insecticides are found in high concentrations in fine-grained sediments in the New York Bight apex (MacLeod et al., 1981) and in fine-grained sediment and the water column near British sewage sludge dumpsites (Dawson and Riley, 1977).

Additionally, high concentrations are reported in sediments near municipal wastewater outfall pipes, such as in the Saronikos Gulf (Dexter and Pavlou, 1973) and off Southern California (Young and McDermott-Ehrlich, 1976).

Elevated DDT and especially PCB concentrations have also been observed in the Baltic, and the possible source was identified as sewage sludge dumping (Oden and Ekstedt, 1976).

The highest concentrations of PCBs in marine sediments have been reported for New Bedford Harbor, Massachusetts, where concentrations up to 19% by weight (190,000 ug/g) have been observed (Massachusetts Office of Coastal Zone Management, 1982). The average PCB concentration in lobsters from this area was found to be only marginally below the USDA action limit (Massachusetts Office of Coastal Zone Management, 1982). This severe contamination resulted from a single, known input source. High concentrations of PCBs have also been reported in water and sediments from various regions of Japan (FAO, 1976). Concentrations of PCBs in sediments

of up to 390 mg/Kg are reported (Table 9). The PCB concentrations in seawater from Japan appear disproportionately high (Table 8) compared to other reported values and may be suspect even though high sediment concentrations imply high levels in the overlying waters. It is not entirely clear which of the grossly contaminated sediment samples referred to in this study were of marine origin and which were from rivers and harbors, although it is likely that the most grossly polluted sediments were from the latter, nearer the sources of contamination. Nevertheless, the lowest sediment concentrations reported in this Japanese survey are high compared to other marine regions of the world. High values were also reported from certain "hot spots" in the Hudson River, although values are much lower in the New York Bight apex. Apart from the New Bedford and Japanese marine sediments that are so grossly contaminated that many of them are being removed by dredging, the marine sediments in Escambia Bay, Florida, and those adjacent to the municipal sewage treatment plant outfalls off Southern California appear to have the highest concentrations of PCBs reported. The California sediments also have very high chlorinated insecticide (total DDTs) concentrations (Young and McDermott-Ehrlich, 1976). The decreased production and use of DDT and PCBs by the United States is apparent in the decreased concentrations of these contaminants in Southern California municipal wastewater discharge (Bascom, 1982; Schafer, 1982), which was reflected in sediments for DDT as early as 1975 (Young and McDermott, 1976). Comprehensive data are lacking for the PCB contents of sewage sludges dumped in the New York Bight, but the results of different researchers suggest that PCB concentrations have decreased over 50% in about 5 years (Bopp et al., 1981; West and Hatcher, 1980; MacLeod et al., 1981; New York City, 1983).

Chlorinated insecticides and PCBs are taken up by organisms and stored primarily in the fatty tissues. Concentrations tend to be highest in those species at the top of the food chain or those whose fatty tissue content is high; this makes the marine mammals and birds particularly capable of concentrating these compounds. Because of the varying ability of different species to bioconcentrate chlorinated organic compounds, it is difficult to compare biotic concentrations from widely separated regions of the oceans where the species composition may be entirely different. Within the range of a single species, geographical variations in the species' chlorinated insecticide concentrations have been used successfully in identifying the most contaminated areas (Anonymous, 1978; Bernhard, 1978). Unfortunately, it is difficult to extrapolate this information to the New York Bight, because insufficient information exists for species from this area of the northwest Atlantic coastal ocean. However, the few reported PCB analyses of fish and shellfish from the New York Bight (O'Connor et al., 1982) show concentrations that are within the ranges observed in the Baltic and Mediterranean, somewhat higher than the ranges observed in the North Sea and Irish Sea, and somewhat below the range observed off Southern California. Care must be exercised in interpreting these comparisons as it appears that concentrations of chlorinated organic compounds vary temporally over a small number of years (Young and McDermott-Ehrlich, 1976).

The concentrations of PCBs in organisms living within the Hudson-Raritan estuary are considerably higher than those in the Bight itself or in Raritan Bay. O'Connor et al. (1982) found higher PCB levels in striped bass from the Hudson River than in those from the New York Bight, and Roberts et al. (1982) were able to demonstrate elevated muscle tissue concentrations of PCBs in lobster in New York Harbor compared to samples

from Long Island Sound and the New York Bight.

C. Effects of Toxic Contaminants in Coastal Marine Ecosystems.

The trace metals and synthetic organic compounds considered in this section are of concern in the New York Bight because they are introduced into the environment in significant quantities, they are persistent in the environment, and, most important, they are toxic. If ingested or otherwise assimilated by organisms at a high enough intake level, these compounds can cause death. If encountered at lower concentrations, they can cause sublethal effects.

Concentrations of trace metals and synthetic organics in the marine environment, with very few exceptions, are considerably below the concentration at which lethal toxicity occurs in even the most sensitive of organisms. Exceptions to this rule have occurred, and continue to occur, in very limited areas when spills or accidental discharges take place or where there is gross pollution by a specific industrial discharge. Also, the concentration of some toxic substances may exceed the lethal toxicity limit for some species a small area within the discharge plume of a submarine pipeline discharge, and in a small area for a short period of time immediately following dumping. However, in each of these instances, the number of exposed organisms is small because of the limited temporal and spatial extent of the toxic levels. There are only a very small number of reported mortalities of marine species known to be caused by anthropogenically introduced toxic chemicals, and each of these is related to a major spill or to uncontrolled industrial discharge. Therefore, lethal toxicity of materials in the coastal ocean is not an important consideration unless major spills or uncontrolled discharges occur. Even in these circumstances, some marine organisms are able to detect such environmental disturbances and to successfully avoid contaminated areas (Johnston and Wildish, 1981).

Mortalities have, of course, taken place where oxygen has been depleted or where red tides have occurred. Both oxygen depletion and red tides are known to be aggravated by pollution, but their prime cause is not the anthropogenic input of a toxic chemical. In fact, the toxin responsible for fish mortality during red tides is synthesized by the bloom organism itself (Ballantine and Abbott, 1957; Boalch, 1979).

Sublethal pollution effects that could possibly occur in the marine environment include reduced organism viability, increased susceptibility to secondary disease, and mutagenic or teratogenic effects. It has been hypothesized that exposure of fish to polluted sediments causes fin erosion in teleosts (Sherwood, 1982; Murchelano and Ziskowski, 1982), although the damaged fins tend to regenerate and there is no evidence that this hypothesized pollutant effect adversely affects fish viability.

Concentrations of contaminants exceeding the maximum observed values in the New York Bight (Table 9) have been found to affect the productivity of the microbial plankton. O'Connors et al. (1982) observed reduced photosynthesis and biomass of phytoplankton exposed to less than one part per million levels of PCBs in laboratory experiments (approximately 3 orders of magnitude greater than New York Bight concentrations), and similar laboratory effects have been observed for planktonic bacteria (Blakemore, 1978; Blakemore and Carey, 1978). However, isolates of natural populations of phytoplankton taken from areas receiving anthropogenic inputs of contaminants tend to be more resistant to contaminants (Murphy et al., 1982) and, therefore, the reduced productivity observed in laboratory experiments may not occur in areas receiving continual anthropogenic inputs. In addition, natural assemblages of aquatic bacteria have the capacity to biodegrade PCBs (Cary and Harvey, 1978; Shiaris and

Sayler, 1982).

Sublethal effects on mackerel eggs that may be caused by contaminants have been noted in the New York Bight. Longwell and Hughes (1980, 1982) found decreased fish egg development rates and viability and increased chromosome or mitotic abnormalities associated with samples collected closer to shore and to contaminant sources in the New York Bight. These eggs develop at the air-water interface and are, therefore, particularly susceptible to elevated concentrations of hydrophobic organic contaminants, which tend to concentrate in the surface microlayer. However, low frequencies of abnormalities were noted in the immediate area of the sewage sludge dump site. While contaminants may decrease the viability of the mackerel fish embryos in limited areas, eggs are produced in huge numbers and over large areas in expectation of high natural mortality, and it is reasonable to conclude that any geographically limited contamination will not significantly affect the total adult mackerel fish stocks. Detection of any contaminant effect on adult mackerel stocks, if it were present, would be difficult since fish stocks are highly variable due to other environmental factors, including overfishing.

Hays and Risebrough (1972) have noted the incidence of abnormal young terns from Long Island Sound and attempted to associate these abnormalities with the teratogenic effects of DDE, PCBs, and mercury. While some of the abnormal birds were found to have relatively high tissue levels of these contaminants, caution must be exercised in interpreting these results. The body burdens of normal terns were not determined, and abnormal terns were found in less than one percent of the population during a two-year study (Hays and Risebrough, 1972).

It has been suggested that pollutants may cause teratogenic effects and

decreased fecundity in marine mammals. Reijnders (1980) has suggested that the harbour seal population in the Dutch Wadden Sea has decreased as a result of PCB and DDT introduction to the marine environment since by these compounds may interfere with mammalian reproduction (Jensen et al., 1977) and immune response (Vos, 1977). While there is no direct evidence for an effect of PCBs and DDT on marine mammals, relatively high levels of these and other compounds have been found in the tissues of dead stranded seals (Reijnders, 1980). Similarly, DeLong et al. (1973) have noted an association of premature births of California sea lions with relatively high concentrations of PCBs and DDT. As with the Long Island terns (Hays and Risebrough, 1972), however, the concentrations of these compounds in normal marine mammals are not well known (Wagemann and Muir, 1981) and no definitive conclusion can be made as to the possible effects of PCBs and DDT.

Reported sublethal effects of pollutants on marine organisms are diverse. However, improper experimental design has often led to invalid results (Eagle, 1981). Of critical importance in sublethal effects studies is knowledge of the contaminant concentration in the organism, not in the medium (water). As concluded by Eagle (1981), most laboratory studies report environmental concentrations, while monitoring programs tend to measure organism concentrations. In addition, organisms have variable resistance to toxicants depending upon the degree of prior acclimatization to the toxicant (Simkiss and Taylor, 1981), which further confounds results of sublethal effect. Marine mammals, birds, fishes, crustaceans, molluscs and microorganisms are capable of binding and detoxifying heavy metals in low-molecular-weight proteins called metallothioneins (Fischer, 1980), and heavy metals may be sequestered in relatively inert tissue compartments (e.g., cell walls, cartilage, scales) which are not easily digested and, therefore, limit metal

transfer through the food chain (Hodson, 1980). In addition, a variety of potentially toxic organics may be detoxified by mixed function oxidases, such as cytochrome P-450, and these enzymes appear in much greater concentration in marine organisms from polluted environments (Parke, 1981).

While sublethal effects of trace metals and toxic synthetic organics require additional study, it may be concluded from a summary of available data (O'Connor and Stanford, 1979) that there is no present reason for major concern about their lethal toxicity in the marine environment, including in the New York Bight, if these chemicals remain at or about present concentration levels. There remain two areas of concern for the potential or actual impact of these toxic chemicals in the Bight: the potential of human health effects through ingestion of contaminated seafood from the Bight, and the possibility that contaminant concentrations of these chemicals in the New York Bight ecosystem contribute to, or directly cause, sublethal biotic effects leading to adverse changes in population numbers or ecosystem structure.

1. Human Health Concerns

The concern for human health related to toxic chemicals in seafood results from the much publicized Minamata incident. The first case of mercury poisoning appeared in Minamata, Japan, in 1953, and by 1956 the "disease" had become quite widespread. By 1971 the number of cases reported had reached 121, with 43 deaths. In 1965 an outbreak took place in another area of Japan (the Agano River estuary) where 49 people were affected, 6 of whom died. All of the victims ate fish as their principal food. The problem was finally traced to mercury poisoning from fish contaminated by discharges from factories using mercury in catalysts for acetaldehyde manufacture (Smith and Smith, 1975).

This serious incident fueled a world-wide concern about the possible health effects of toxic metals and synthetic organics released to the marine environment and ingested in seafood. However, apart from the Minamata incident, there has been no documented case of any clinical or subclinical disease among the world's human population that can be attributed to a toxic chemical in seafood taken from non-estuarine marine waters. Clinical problems have of course been caused by a variety of toxic chemicals released to the environment (e.g., itai-itai disease caused by ingestion of Cd-contaminated rice and Pb poisoning in chemical industry workers). However, all instances of such poisoning, except Minamata, have apparently been caused either by ingestion of food grown in contaminated soil or by direct human exposure to toxic substances in the work or living environment.

The United States Food and Drug Administration (U.S. FDA) has set an "action limit" of 0.5 ppm for Hg in fish tissues: fish having Hg concentrations higher than this cannot be sold for human consumption. In an extensive survey of edible tissues of various fishes and shellfishes from the New York Bight, the highest concentration of Hg found was 2.3 ppm in the muscle of one lobster sample. However, concentrations in almost all of the over two hundred samples were less than 0.5 ppm (Hall et al., 1978). A survey by Roberts et al. (1982) also found concentrations consistently below 0.5 ppm. Saila and Segar (1979) have shown that even if a resident of the New York area were to consume lobster with 2.3 ppm of Hg as 20% of his diet (a very high and unlikely percentage), his total mercury intake would be only about one-half the minimum level thought to be toxic and about one-tenth of the intake level of Minamata disease sufferers. Therefore, a considerable safety margin exists in the ingestion of seafood from the Bight. Moreover, no fishery in the Bight region has been closed due to Hg contamination (i.e., the average value in no species

exceeds 0.5 ppm), although fisheries in several other parts of the world, including sections of Japanese, Swedish and Italian coastal waters, have been closed to fishing because of high Hg concentrations in fish (Goldberg, 1976). At present it appears that there is no threat to human health from mercury in the New York Bight. However, continued monitoring and continued control of industrial mercury discharges are necessary to ensure that this situation is maintained.

A recent study examined human mercury levels resulting from the consumption of seafood from Liverpool Bay and the Thames Estuary, both of which receive mercury inputs due to the chlor-alkali industry and sewage sludge discharge (Sherlock et al., 1982). Since the average consumption of fish was 0.36 kg per person per week, it was determined that people in the United Kingdom were unlikely to be adversely affected by the presence of methyl mercury in the consumed fish. Total mercury concentrations in fish muscle tissue averaged 0.34 mg/kg (range, 0.03 to 1.66 mg/kg), close to the U.S. FDA 0.5 ppm action level. However, even at the highest levels of fish consumption and, therefore, of mercury intake, human blood levels of mercury never exceeded the World Health Organization's estimated effect level of 200 mg/Hg/liter and rarely exceeded one-tenth of this value (Sherlock et al., 1982).

Saila and Segar (1979) have shown that toxic metals other than mercury are even less likely to constitute a hazard to human health through ingestion of seafood from the New York Bight, or other parts of the world ocean, except immediately adjacent to grossly contaminated industrial discharges.

Presumably because of the publicity surrounding itai itai disease (Friberg et al., 1974), cadmium is often thought to constitute a marine environmental problem comparable to mercury. If a person were to consume seafood with the highest Cd levels observed in New York Bight biota instead of other meats in

the average diet, his overall intake of cadmium would probably decrease (Salla and Segar, 1979). However, safe intake limits of cadmium may be exceeded in Hong Kong, where research suggests that both seafood concentrations of metals and seafood consumption are relatively high (Phillips et al., 1982).

Inputs of potentially toxic organics have resulted in fishery closures of several source streams to the New York Bight. On December 15, 1982, the New Jersey Department of Environmental Protection issued a fishery closure notice for portions of Raritan Bay and the Hudson River for striped bass, white catfish, white perch, and American eel based on evidence of PCB contamination of fish from these waters. Closures were not issued for the New York Bight since only a single bluefish (among 45 sampled) from the Bight was found to have a PCB concentration exceeding the 5 ppm U.S. FDA action level (Belton et al. 1982). In a more recent study (Belton et al., 1983), PCB concentrations in 6 of 28 bluefish samples from "marine waters" of New Jersey were found to be in excess of 2 ppm. As a result of these observations and observations of substantially higher concentrations of PCBs and chlordane in several species of fish from estuaries, an advisory was issued for limited consumption of five fish species from New Jersey waters including those marine areas from Sandy Hook to Barnegat Inlet, from the coast to 30 miles offshore (Belton et al., 1983). As a result of dioxin contamination, advisories have been issued for much of the lower portion of the Hudson River to limit human consumption of finfish (New York, Department of Environmental Conservation, 1982), and portions of the Passaic River feeding into Raritan Bay have been closed to fishing due to localized dioxin contamination (Newsday, 1983). The only contamination-related fishery closures in the New York Bight are precautionary shellfish closures to protect against possible contamination due to the input of sewage-associated microorganisms from the Hudson-Raritan estuary, as well

as from the disposal of sewage sludge. There have been no closures to fishing due to toxic chemical contamination in the New York Bight.

Little is known about the effects of human ingestion of chlorinated insecticides, petroleum hydrocarbons, and PCBs at subtoxic levels; many of these compounds are known or suspected carcinogens and have been associated with steroid hormonal imbalances (Jensen et al., 1977). However, since the New York Bight is not among the most contaminated regions of the world with these compounds (Raritan Bay and parts of the Hudson River may be among the most contaminated with these compounds), and since the U.S. FDA "action limits" are exceeded only by a small proportion of individuals of a few seafood species from the Bight (which were probably contaminated in the estuaries), there seems to be little reason for current concern or for drastic actions to reduce input levels of these contaminants to the Bight, especially for DDT and PCBs, inputs of which appear to be decreasing. Nevertheless, as further research reveals the principal sources of these compounds, prudent management to reduce their inputs to even safer levels should be exercised. Because it is probable that concentrations of these compounds in the New York Bight ecosystem are in approximate equilibrium with inputs (Swanson et al., in press), the environmental concentrations should not increase if inputs do not increase. However, monitoring should be initiated as soon as possible to ensure that action can be taken to reduce inputs if equilibrium has not been achieved and if the concentrations are still increasing in response to the increase in inputs which occurred during the last decade or two. Monitoring will also provide information concerning the degree of success of source control efforts. For example, in Southern California, DDT levels in the environment around the municipal sewage treatment plant effluent pipe have dropped, albeit slowly, after the use of DDT was halted (Bascom, 1977). In

the New York Bight region, the decline and recovery of the osprey population have been attributed to high, and then declining, inputs of DDT metabolites to the environment (Spitzer et al., 1978). Similarly, reductions in PCB loadings to the environment have been observed (Schafer, 1982).

Based on knowledge of the existing levels of contaminants in edible species, it may be concluded that any present adverse impacts on human health by toxic chemical contaminants discharged to the New York Bight are, at most, small compared to impacts from toxic contaminants in food organisms from polluted estuaries and rivers. There has been no indication of any acute poisoning, such as occurred in Minamata, and no other evidence exists to suggest that subclinical pathologies have occurred. There does remain the concern for the carcinogenicity of some contaminant chemicals. However, chemicals contaminating the New York Bight are ubiquitous, and it is very unlikely that seafood ingestion or direct body contact with seawater, compared to other sources, provides a significant fraction of the intake of any carcinogenic contaminant for any human being, unless perhaps seafood dominates their diet.

These conclusions are borne out by observations in areas of the world other than the New York Bight. Apart from Minamata and a very few similar incidents where grossly contaminated fish or shellfish have been consumed from inland waters close to major uncontrolled industrial discharges, no adverse effect on human health has been observed that can be ascribed to the contamination of the marine environment with toxic chemicals. Additionally, many heavily fished regions of the ocean appear to be more seriously contaminated than the New York Bight (see section IV, A and B). Therefore, unless a dramatic increase in contaminant loadings were to occur, or unless there are unidentified contaminants present in the New York Bight in higher

concentrations than in other parts of the world (Goldberg, 1974), there is no cause for apparent concern about effects on human health that might be caused by chemical contaminants present in the New York Bight.

2. Ecosystem Health Concerns

Unfortunately this reassuring conclusion concerning human health effects cannot be so easily reached concerning the effects of toxic contaminants on the marine ecosystem of the New York Bight and other impacted regions. Various changes or suspected changes in the ecology of the New York Bight have been ascribed at least in part to the toxic contaminant loading of the ecosystem (Wolfe et al., 1982). These changes include population reductions in some commercially important finfish and shellfish; altered benthic fauna, particularly in the vicinity of the dredged material and sewage sludge dumpsites (Carriker et al., 1982); and fin rot and other fish diseases (Sindermann, 1976; Murchelano, 1982). These and other changes have been observed in each of the other contaminated regions we have considered: Southern California (Bascom, 1977), the Mediterranean (Charbonnier, 1977, Peres, 1978), the Baltic (Akerblom, 1976, 1977), and particularly the Seto Inland Sea (FAO, 1976). These changes have not always been detrimental. For example, there is strong evidence that the productivity of the fisheries outside the Thames estuary was enhanced by the nutrients derived from London sewers (Sindermann, 1976).

In each of these ecosystems, it cannot be easily proven that biological changes ascribed to marine pollution are caused by toxic chemicals associated with the pollution or that the presence of toxic contaminants has even been a major contributor to such changes. The ocean environment, just like the land, is highly complex and subject to

climatic variables, and the terrigenous and anthropogenic inputs which could conceivably affect ecosystem structure are equally variable. Most of the changes in fish and shellfish population abundances that have taken place are more likely to have been caused by natural "climatic" fluctuations or by overfishing of certain populations, rather than by toxic chemicals (Grosslein and Azarovitz, 1982). The same considerations hold true for other observed ecological changes, including fish diseases.

The traditional response to this dilemma has been to rely on experimental studies of the effects of contaminants on organisms in "controlled ecosystems" in the laboratory environment. However, most such studies are of limited value. In the laboratory environment, many of the primary variables (e.g., climate and boundary conditions; the synergism, parasitism, and antagonism within natural populations; and the chemical composition of the natural environment) cannot be duplicated. Further, the concentrations of toxic contaminants used are often many times those found in even the most polluted natural ecosystem. The pitfalls of this experimental approach become apparent when we consider the case of the cupric ion (Cu^{++}), which is toxic to aquatic algae at high concentrations, innocuous at lower concentrations, and inhibitory or prohibitive at even lower concentrations which are below the nutritionally required level. Even the extreme toxicity of very high concentrations of copper can be drastically reduced by adding a chelating agent (e.g., Morris and Russell, 1973).

Although questionable laboratory toxicity studies have been used to infer that ecological damage may take place in certain contaminated ocean ecosystems, virtually nowhere has there been an unequivocal demonstration that the ecological changes that have been observed in the marine environment are caused by toxic chemicals. The only exceptions to this rule are where

discharges of extraordinarily large quantities of a particular toxic chemical contaminant took place, either through accidental discharges or ocean dumping.

It is instructive to look at what is perhaps the most intensively researched impact on the marine environment: the impact of sewage and sewage sludge and other organically rich materials on sedimentary infauna. Such studies have been carried out in many areas including the New York Bight (Pearce et al., 1976), the Saronikos Gulf (Sheppard, 1977), the Gulf of Napoule, France (Bellan, 1979), the Wadden Sea, Netherlands (Essink, 1978), the North Sea Island of Sylt (Otte, 1979), Vancouver, Canada (Otte, 1979), the Thames Estuary (Shelton, 1971), the English Channel (Jenkinson, 1972), the Firth of Clyde (MacKay et al., 1972; Halcrow et al., 1973), the Firth of Forth, Scotland (McLusky et al., 1978; Read et al., 1982, 1983), Southern California (Bascom, 1977), the German Bight (Rachor, 1977), and the German North Sea coast (Caspers, 1976). Each of these studies has established the distribution of benthic fauna in an area where organically rich sediments are found at or near a point of discharge or near dumping of an organically rich waste material, usually sewage and sewage sludge.

In each of the studies listed above, the observed distributions are essentially similar. In the sediments with the highest organic carbon contents, the diversity of benthic infaunal populations is reduced although usually the biomass is increased. In these organically rich sediments, the species that dominate are usually deposit feeders, particularly polychaetes and bivalve molluscs. In an area fringing the central portion of the highly organic sediments, there is always a region where diversity of species is normal, but the biomass is higher than areas farther removed from the high organic sediments. In these transition zones, filter-feeding species and

their predators dominate.

In addition, most of the studies listed, as well as several others, have found that trace metals and synthetic organic compounds are present in much higher concentrations in the organically rich sediments than in the surrounding areas where sediments are usually coarser grained and contain less organic matter. This has often led to the tentative conclusion that the ecological changes observed in sediments accumulating organic matter under the influence of waste discharges may have been caused by the toxic action of the chemicals.

However, from all of the available evidence, it would seem that the dominant factors controlling the changes in benthic ecology are the increased sedimentation rate and the greater availability of organic detritus, which cumulatively lead to competitive advantages for organisms with different feeding strategies than those present in more slowly accumulating, lower organic sediments (e.g., Caspers, 1976). Another important factor is certainly the lowered redox potential caused by the greater consumption of oxygen within the sediments. Therefore, the role of toxic chemicals, if any, in these benthic ecological changes is uncertain.

The ecological changes observed in organically enriched sediments may be either beneficial or detrimental, depending upon the degree of enrichment, and the food chain value of species whose populations are enhanced by this enrichment (Segar et al., 1984). However, observation of ecological change has been often interpreted by the scientific community and/or regulatory bodies as a demonstration of "damage" to the ecosystem. In fact, moderate organic enrichment of marine sediments in many areas may be an environmental enhancement of potential benefit to man.

Regardless of whether they are "damaging," changes in the ecology

of sedimentary environments caused by discharges of organic-rich wastes are apparently not permanent. For example, within four years after removal of a municipal sewage discharge from Cannes, France, bottom sediments of parts of the Gulf of Napoule that had been "severely altered" by the discharge had recovered such that the infaunal assemblages were entirely normal (Bellan, 1979). A Swedish estuary, which for 12 years had been impacted by organic waste discharged from a sulfite pulp mill, and whose infaunal species were badly impoverished over a large area, took only eight years for full recovery to a status indistinguishable from the pre-existing natural condition (Rosenberg, 1976). A long-term benthic study in the Firth of Forth showed a decline of certain pollution indicator species and the appearance of a recovered benthos following the reduction of sewage suspended solids in Edinburgh effluent (Read et al., 1982, 1983). Interestingly, even though phosphate, biological oxygen demand (BOD), and bacterial numbers decreased markedly in this area, there was no effect on the levels of oxidized nitrogen in the water column (Read et al., 1983), indicating that nitrogen levels were not related to sewage inputs.

In summary, although ecological changes have been observed in the vicinity of many marine outfalls and ocean dumpsites, and concentrations of many toxic chemicals have risen in the marine environment over pre-industrial conditions, it is not established that these elevated concentrations have exerted any significant effects, deleterious or beneficial, on the marine ecosystems into which they are discharged, except in a small number of restricted areas where discharge rates are extremely high, and flushing and dispersion are exceptionally poor.

A few ecological effects have been observed that can only reasonably be

ascribed to toxic chemicals released into the marine environment. Perhaps the best known of these is the decline and subsequent restoration of pelican populations in California over a period of several years during which the use of DDT and other chlorinated insecticides rapidly increased and later declined (Anderson et al., 1975). Other such subtle effects of toxic contaminants may well be occurring in the marine ecosystem. However, our ability to establish any cause-and-effect relationships between a given spectrum of contaminants at a particular location and the often subtle ecological changes observed is simply not adequate to delineate the effects of toxic chemicals in the marine environment. We are not yet able even to understand, explain, or predict the natural ecological changes occurring due to climatic variations or the changes occurring due to the impact of pollution other than by toxic chemicals. However, only in a few limited instances and in small geographical areas has pollution of the marine environment by toxic chemicals produced measurable human health impacts or significant ecological degradation. Regarding the New York Bight (excluding the adjacent river and estuarine areas), no known hazard to human health exists from toxic chemicals, and overall toxic chemical concentrations in the New York Bight are below minimum levels that are thought to be lethally toxic to marine organisms.

V. ANOXIA, ORGANIC MATTER, NUTRIENTS, AND MUNICIPAL WASTES

The most pervasive contaminating material entering the oceans is municipal sewage. In most coastal communities of the world, sewage is simply discharged without treatment into rivers, estuaries, or the ocean. However, in many countries, including the United States, this practice is undergoing major changes due to two factors: 1) increased public concern for the health hazards caused by sewage discharged into waters used for recreational activities, including bathing, and 2) declining fish populations, diminished oxygen concentrations, and black putrid bottom sediments of many tidal estuaries into which sewage is discharged. To meet these concerns, municipal waste managers have concentrated on reducing the oxygen demand on estuarine waters and the microbial contamination of recreational waters.

In most countries throughout the world, the favored solution to these problems has been to build long discharge pipes to carry the sewage sufficiently far out to sea that it no longer is discharged into the estuary or close to recreational areas. In the United States and a few other nations, the strategy adopted was entirely different. Clean-up of United States lakes and rivers has been achieved by secondary treatment of sewage (i.e., removal of suspended solids and biological oxidation of some of the organic matter before discharge), while the locations of discharges into estuaries or semi-enclosed harbors have usually remained unchanged. Secondary treatment causes a reduction in the concentration of pathogens in the treated effluent, but further treatment by chlorination is often practiced to effect an even greater reduction. The United States has required such secondary treatment on all discharges of municipal sewage into natural waters including the ocean, although a few coastal communities will soon be permitted to discharge sewage

effluent with less than secondary treatment into the ocean.

One important consequence of primary or secondary treatment is the generation of large volumes of sewage sludge at the treatment plant. Various disposal methods for this sludge are utilized (NAS, 1978). The methods utilized by coastal communities have included disposal by ocean dumping, discharge through ocean outfalls, and various land disposal systems.

The dumping of sewage sludge into the ocean has become a controversial subject mainly due to the public concern shown for possible health effects in the New York/New Jersey area during the mid-1970's. The environmental events that reached high levels of public visibility during this period included: beach pollution with sewage derived artifacts; the finding of organically rich, biologically-altered sediments offshore from the Long Island and New Jersey coasts; and anoxic bottom water events and fish kills, particularly that of 1976 (Swanson and Sindermann, 1979). The ocean dumping of sewage sludge was the most publically visible and, therefore, the easiest candidate for the public and legislators to blame for these events. As a result, a series of decisions followed that, if fully implemented, would have eliminated the disposal of sewage sludges into the ocean. Few nations in the world have adopted such a prohibition on ocean disposal of sewage sludges, while several nations, including the United Kingdom, are adopting the ocean disposal alternative for sewage sludges as their preferred strategy for cleaning up river and estuary pollution problems and avoiding land disposal environmental impacts.

Against this background, the observations made in other parts of the world ocean on the effects of organic and nutrient loading (including sewage and sewage sludge) on the oxygen depletion in the water column below a density

discontinuity (thermocline or halocline) can be better understood. Such an oxygen sag is a normal natural feature of areas where there is high productivity and restricted vertical mixing and flushing of the bottom water. The primary examples of this occurrence under natural conditions unaffected by man are the Cariaco trench and the Black Sea (Richards, 1965), where restricted bottom water flushing permits the build-up of oxygen-demanding substances over a period of years; and off the west coast of Africa, where upwelling of nutrient-rich water can cause sudden intense blooms of the phytoplankton (Harvey, 1963) with resultant large oxygen demands due to the decay of unconsumed dead cells. In addition to these natural occurrences, anoxic, or nearly anoxic, bottom or subthermocline waters have been found in the Baltic Sea (Fonselius, 1970a; Kremling, 1973; Sen Gupta, 1973; Kaleis, 1976); in several coastal locations in the Mediterranean Sea, including Elefsis Bay and the Saronikos Gulf, Greece (Sheppard, 1977; Zarkanellas, 1979); in the Gulf of Mexico (Harper et al., 1981); in Chesapeake Bay (Taft et al., 1980); and in the northern Adriatic Sea (Degobbis et al., 1979; Maretic et al., 1978). Anoxia has also been observed in several locations in Japanese waters including Tokyo Bay (Tsuji, 1974; Seki et al., 1974), Omura Bay (Hirayama and Iizuka, 1975), and large areas of the Seto Inland Sea (Murakami, 1977, Shiozawa et al., 1977). In each of these instances, the affected area is subject to discharges of municipal sewage, most of which is untreated and discharged through rivers or short seafloor pipelines. In addition, the flushing of each area is restricted, and a short-term, dense phytoplankton bloom, or a long-term increase in primary production, has been implicated. Simple mass balance calculations and observations of primary production changes have indicated for some of these regions that the nutrient loading (nitrogen and phosphorus forms) by municipal and other waste discharges is primarily responsible for the increased primary production and subsequent

bottom oxygen demand, and that only a small percentage of the oxygen demand is contributed directly by the organic matter of the municipal waste itself (see Segar and Berberian, 1976, Okaichi, 1975, and Sen Gupta 1973 for calculations relative to the New York Bight, Seto Inland, and Baltic Sea, respectively).

No oxygen depletion has been observed in either the North Sea or the Irish Sea (Department of the Environment, 1978) despite the fact that the organics (BOD), nitrogen, and phosphorus loadings are high (Table 3) and often localized at the mouth of estuaries. However, both of these seas have active tidal current regimes and are only weakly stratified in the summer, such that nutrients, primary production, and oxygen demand are spread over a much larger area than, for example, in the semi-enclosed areas of the Mediterranean.

Relative to the volume of the area into which contaminants are discharged, the Seto Inland Sea and the New York Bight each have higher contaminant loadings than any of the other areas considered (Table 2, 3). This is a somewhat artificial comparison as discharges to the North and Irish Seas, for example, are not evenly distributed or instantaneously mixed throughout the seas. Nonetheless, the inputs to the New York Bight are generally comparable to those of the Irish Sea and lower than inputs to the North Sea. Yet the New York Bight inputs take place essentially at a single point, the Hudson-Raritan estuary mouth (Mueller et al., 1976). In addition, exchange between the water of this apex region of the Bight and the waters outside the apex is somewhat restricted, especially during the critical summer period (Charnell and Hansen, 1974). The Seto Inland Sea is a very small, restricted sea with only slow exchange with water from the Pacific Ocean, and yet it receives organic and nutrient loads comparable to the New York Bight. Because of its restricted circulation, the eutrophication and anoxia problem in the Seto Inland Sea is

correspondingly worse than that in the New York Bight. The Seto Inland Sea could then be a model for the future of the New York Bight if the level of nutrient loadings continues to rise. In addition to its bottom water anoxia problem, the Seto Inland Sea has suffered from numerous and extensive red tide blooms of flagellate phytoplankton during the summer months of many recent years. These red tides have caused tremendous losses of Seto Inland Sea shellfish and finfish due to the production of toxic metabolites by the flagellates and the decay of these organisms which further contribute to the anoxia (Okaichi, 1975). Consistent with the parallelism in contamination and restricted flushing between the Seto Sea and the New York Bight, minor red tide blooms have already been reported in the New York Bight over the past several years.

The major anoxia of 1976 in the New York Bight has been attributed to a combination of natural events including reduced flushing of the water masses and the extensive bloom of Ceratium tripos (Swanson and Sindermann, 1979; Falkowski et al., 1980). This attribution has led some investigators to conclude that, because nutrient-induced eutrophication was not a primary cause of the 1976 anoxia, such eutrophication was not occurring in the New York Bight or was insignificant. However, more limited anoxic or near-anoxic areas have been observed particularly in the region influenced by the Hudson nutrient discharge in several recent years (Swanson and Sindermann, 1979). Therefore, it may be concluded that 1) there is a chronic low-oxygen problem in the New York Bight, particularly in the apex and New Jersey shelf regions, which is aggravated by nutrient inputs (Segar and Berberian, 1976); and 2) there exists in the New York Bight a tendency toward oxygen depletion on a larger scale, which is dominated by natural events as in 1976. The susceptibility to anoxia of the New York Bight ecosystem exists particularly

under conditions where bottom oxygen demand is increased above normal average values, whether this increase is caused by anomalous phytoplankton blooms or by eutrophication. Finally, based on the findings of Segar and Berberian (1976) and Falkowski et al. (1980), it may be concluded that, even if the quantity of material leading to oxygen demand (BOD and nutrients) in ocean-disposed sewage sludge was drastically increased (by a factor of at least ten) from its current level, this would have little or no incremental effect on the tendency of the New York Bight to become anoxic.

Although the nutrient loading of the New York Bight from domestic sewage discharges may not have been the primary cause of the 1976 anoxia, and although the 1976 anoxia may still have occurred without such inputs, available knowledge strongly supports a conclusion that the nutrient-induced eutrophication of the New York Bight significantly increases the probability that anoxic conditions, on some scale, will occur in any given year.

Elimination of anthropogenic nutrient discharges would not reduce the probability of anoxia in a given year to zero; however, increased nutrient inputs will increase the probability and, judging by experiences elsewhere in the world, that increase is likely to be significant.

In order to illustrate the interaction between increasing nutrient input, natural year-to-year ecological and physical variability, and the probability of anoxia, it is instructive to look at the Baltic which has been monitored extensively for several decades. The bottom water of the Baltic is flushed only during periods of unusual currents over the Kattegat sills, which occur irregularly every several years (Fonselius, 1972). In between periods of flushing, the naturally occurring primary production of the Baltic reduces the oxygen concentration in the near-bottom water of the deeper central basins, but, in the past, the area of anoxic water usually has been small. This is

known to be true based upon records for the early part of this century pertaining to hydrography, sedimentary infauna (Zmudzinski, 1977), and sediment chemistry of this region. However, during the past 60 years, there has been a deterioration in the oxygen concentrations, with much larger areas now being affected by anoxia than previously. This deterioration has been attributed to nitrogen- and phosphorus-induced eutrophication. During this period of decline, there were a number of periods of reoxygenation when conditions improved during unusual (but not extreme or anomalous) physical conditions which temporarily increased the flushing rate of the bottom waters. The steady decline due to increased nutrient inputs (and perhaps other factors such as reduced freshwater inflow due to industrial water use) is superimposed on a naturally variable system (Fig. 8). The extremes of oxygen depletion and reoxygenation within the system are caused by physical factors, while the probability of anoxia in any given year has become progressively larger during the period monitored due to the chronic pollution problem (Fig. 8).

This deterioration in the Baltic is almost certainly matched by a similar deterioration in many parts of the world ocean, including the New York Bight. However, in the absence of the long time series of observations over several decades, such as are available for the Baltic, this steady deterioration will be obscured by natural year-to-year variability. For example, the steady deterioration in the Baltic would certainly be difficult, if not impossible, to identify if only one decade of monitoring data were available (Fig. 8).

It is likely, therefore, that anoxia will continue to occur in the New York Bight episodically. The frequency with which anoxia as widespread and intense as that in 1976 will recur will be determined by the probability of recurrence of the physical and biological conditions that occurred in 1976 or their

equivalent. Nevertheless, if nutrient inputs increase, then large-scale anoxia events will occur more often and under progressively less unusual natural conditions than occurred in 1976.

Although this paper has focused primarily on nutrient inputs, which are thought to be the real problem leading to anoxia, the hypothesized linkage of sewage sludge with anoxia should be laid to rest once and for all. In several areas of Europe (e.g., the Kiel Bight, the Irish Sea, the Thames Estuary, and off Garroch Head in the Firth of Clyde), sewage sludge is dumped in precisely the same manner as in the New York Bight. Most of the dumping locations are remote from the influence of nutrients in municipal sewage effluent discharges, and in no instance has significant oxygen depletion of the water column been observed at or near the dumpsite. Observations off Garroch Head are particularly relevant to the New York Bight, since sewage sludge dumping has taken place there since 1904, flushing is somewhat restricted, and the quantity of sludge dumped there approaches that dumped at the New York site. Dispersion rates for the two areas are considered to be roughly equivalent (Champ and Park, 1981), and the sediments underlying the Garroch Head site are high in organics and are anoxic like those near the sewage sludge dumpsite in the New York Bight. Despite these similarities, no significant oxygen depletion has been reported in the water column at or near the Garroch Head dumpsite (Norton, 1983). These observations support the conclusion that dumping of sewage sludge in the New York Bight has a quantitatively insignificant impact on the oxygen depletion problem (Segar and Berberian, 1976).

In summary:

- 1) Increased input of nutrients from sewage treatment plant effluents will significantly increase the probability of anoxia and "red tides" in

the New York Bight. This conclusion is supported by evidence gathered from a variety of coastal areas throughout the world which do not receive sewage sludge through dumping, but which do receive anthropogenic nutrient inputs.

2) The evidence concerning the causes of anoxia and "red tide"-like blooms in New York Bight indicates that eliminating sewage sludge dumping will not significantly reduce the probability of occurrences of anoxia (Segar and Berberian, 1976; Falkowski et al., 1980). This conclusion is supported by evidence gathered from sewage sludge dumpsites in other parts of the world.

VI. THE NEW YORK BIGHT DIAGNOSIS AND PROGNOSIS

Comparisons drawn between the New York Bight and other regions of the coastal ocean which are also affected by industrial and urban contaminants permit diagnosis of the relative health of the Bight and analysis of the impacts caused by the contaminant loads. Geographic comparisons also allow us to estimate the probable impacts of changing pollutant loads on the New York Bight.

While contaminant inputs to the Bight are large and concentrated in the small apex region, they are not extreme compared to many other areas of the world ocean. Because the New York Bight is a moderately effective dispersive ecosystem and because it is effectively flushed by the Atlantic Ocean as compared to many other more restricted seas (e.g., Adriatic, Saronikos Gulf, and Baltic), the concentrations of toxic contaminants in the Bight are, at present, not extreme. Concentrations of some toxic chemicals in seafood from the New York Bight are undoubtedly higher than would be the case absent anthropogenic contaminant inputs. However, contaminant concentrations in the

seafood are within safe limits (with the exception of a small proportion of individuals of some seafood species probably contaminated in the rivers or estuaries), and the increased risk to human health due to the elevated concentrations is certainly very small (Plugge et al., 1983); consequently, no current threat to human health is posed by these toxic compound concentrations within the Bight ecosystem per se. Although ecological changes, some of which are deleterious, have been observed in limited areas of the Bight, these changes are not exceptional compared to some other coastal ocean regions and are probably caused primarily by factors other than the elevated toxic metal or synthetic organic concentrations.

The reduction in oxygen concentrations in Bight bottom waters, particularly in the apex and northern New Jersey coastal regions, is a chronic situation similar to that occurring in many of the world's coastal regions receiving municipal discharges. This chronic situation is apparently brought about by nutrient-induced eutrophication. Although primarily a natural event and not caused by the chronic oxygen depletion problem, the 1976 anoxia highlights the susceptibility of the New York Bight to summer oxygen depletion and anoxic conditions.

What then is the prognosis for the future? Because the concentrations of toxic components in the New York Bight ecosystem are probably in approximate equilibrium with the annual level of inputs (Swanson et al., in press), there would need to be a several-fold increase in input rates of toxic metals or synthetic organics before concentrations would exceed those levels observed in other coastal ecosystems of the world ocean (e.g., Southern California outfall regions, the Seto Inland Sea) or would be likely to pose a threat to human health. Even these more heavily polluted regions are not significantly more

"degraded" than the New York Bight and several such regions have shown the ability to recover in a matter of years after inputs are removed. Therefore, with prudent management to prevent increased input levels of toxics and, preferably, to promote reductions, it can be concluded that there should be no future major threat to human health or irreversible damage to the ecosystem. However, it also should be remembered that not all toxics have been studied, and this conclusion does not absolutely rule out the surprise factor due to some metal or compound not yet detected or introduced into the oceans in significant quantities.

The prognosis concerning oxygen depletion is not as good. If the population of the New York region increases, the nutrient inputs will inevitably also rise if current sewage treatment methods are maintained. In order to prevent nutrient-induced eutrophication problems from increasing in the New York Bight (and in some other coastal regions both in the United States and other countries), the nutrient loading of sewage treatment effluents will have to be controlled. Such control will be extremely costly, although it may be needed only for part of the year. Further study is needed to develop criteria that would establish the need for control and the degree of nutrient removal required. The further analysis of data from other ocean areas such as the Seto Inland Sea would aid this criteria development process.

For this report a broad selection of the readily available scientific literature related to coastal pollution in other regions of the world was searched; additional relevant information exists that is not readily available. Every effort should be made to make use of these other data to further refine our understanding of the impacts of marine pollution and to improve the timeliness and effectiveness of ocean pollution/waste management decisions.

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APPENDIX

Table 1. Pollution susceptibility of selected ocean regions

Area	Population	Industry	Discharge Region Volume	Dispersion and Flushing
New York Bight	3	3	1	2
Lake Erie	3	3	1	2
Seto Inland Sea (Japan)	3	3	1	2
Baltic Sea	2	3	1	3
Mediterranean Sea:	2	2	1	2
Adriatic	3	3	1	3
Saronikos Gulf	3	3	2	3
North Sea	3	3	1	1
Irish Sea	3	3	1	2
Southern California Coastal Region	3	2	1	1
Puget Sound	2	2	2	2
Chesapeake Bay	2	2	2	2
Guanabara Bay, Brazil	3	3	3	2
Hudson-Raritan Estuary, NY	3	3	3	2
Firth of Clyde, UK	3	3	2	2
Singapore: Harbor	3	3	3	3
Straits of Mallaca	3	2	2	1
Hong Kong: Harbor	3	3	3	2
Adjacent South China Sea	3	3	1	1
Red Sea	1	1	1	3
Persian Gulf	1	2	1	3
Black Sea	2	2	1	3

Higher numbers indicate a higher susceptibility. Densely populated or highly industrialized areas receive a "3", a high susceptibility. Areas with limited discharge volumes and restricted flushing also rate a "3" for their high susceptibility.

Table 2. Approximate areas, volumes, and coastal populations of selected coastal regions

	Area (Km ²)	Volume (Km ³)	Approximate Population (x10 ⁶)
New York Bight ¹	3.8 x 10 ⁴	2.0 x 10 ³	17
Southern California, Los Angeles	NA	NA	7
Mediterranean	2.5 x 10 ⁶	3.8 x 10 ⁶	100
Seto Inland Sea, Japan	2.2 x 10 ⁴	8.4 x 10 ²	>20
Baltic Sea	3.7 x 10 ⁵	2.1 x 10 ⁴	18
North Sea	5.8 x 10 ⁵	5.4 x 10 ⁴	22
Irish Sea	1.0 x 10 ⁵	6.0 x 10 ³	8

Note: NA= not applicable

¹ O'Connor, 1982

Table 3. Estimated contaminants and natural inputs to selected regions**
(metric tons/yr)

	BOD (x10 ⁴)	COD (x10 ⁴)	N (x10 ⁴)	P (x10 ³)	Hg	Pb	Cr	Zn	Cd	Oil & Grease		Total PCB
										(x10 ³)	Total DDT	
New York Bight ¹	A	55	370	14	19	53	2,100	1,600	6,800	110	210	10
	B	39	-	14	17	35	1,100	820	3,600	63	160	4.6
	C	6.9	-	2.5	0.4	-	25	0.16	34	0.29	-	0.2
South Atlantic (U.S.) Bight ²	B	-	-	-	-	1.0	65	-	310	3.0	-	-
	C	-	-	-	-	22	620	-	710	8.0	-	-
Southern California Hyperion 5 mile Hyperion 7 mile		25	-	-	>10	2.8	150	370	840	42	49	880
		6.4	-	-	5.5	0.9	13	57	140	8.8	8.4	2,300
		-	-	-	1.2	0.8	14	81	180	8.3	-	940
Mediterranean ³ Adriatic		330	860	100	360	130	4,800	2,800	25,000	-	1,000 ⁺	-
		80	170	27	85	41	1,400	200	8,600	-	-	6,000 ⁺⁺
Seto Inland Sea	91	80	16	17	-	-	-	-	-	-	-	-
Baltic Sea ⁴	41	-	59	17	29	6,200	900	19,000	250	-	-	-
Gulf of Bothnia ⁵	6.8	-	0.4	0.5	-	-	-	-	-	-	-	-
North Sea	A	110	-	92	110	47	4,900	4,000	24,000	330	-	-
	B	-	-	-	-	38	2,900	1,800	20,000	280	-	-
	C	-	-	-	-	5.6	5,600	720	14,000	530	-	-
Sorfjord, Norway ⁶	-	-	-	-	11	1,600	-	2,200	11	-	-	-
Irish Sea	A	37	-	8	22	35	2,400	1,100	10,000	110	-	-
	B	-	-	-	-	11	710	500	6,700	90	-	-

Table 3. (continued)

	BOD x10 ⁴	COD x10 ⁴	N x10 ⁴	P x10 ³	Hg	Pb	Cr	Zn	Cd	Grease x10 ³	Total DDT	Total PCB
Gulf of Mexico, Mississippi Delta 7,78	-	-	-	-	40	13,000	22,000	60,000	500	20	10	-

**data used to construct the table were obtained from various sources including unpublished International Council for the Exploration of the Sea (ICES, 1978) reports.

A = total of all inputs except atmosphere

B = inputs less atmospheric and dumping

C = atmospheric inputs

+ Does not include oil and grease in sewage (LeLourd, 1977)

++ From Kihlstrom et al., 1978

- Data not available

- 1 Meller et al., 1976, 1982; New York City, 1983. (Includes inputs to Hudson-Raritan estuary)
- 2 Windom, 1981
- 3 Helmer, 1972
- 4 Bruggmann, 1981
- 5 Baveriren and Vuoristo, 1978
- 6 Skei, 1981
- 7 Sackett, 1981. Data includes only Mississippi inputs, ocean dumping, and contaminants from oil and gas operations.
- 8 Trefry and Shokes, 1981

Table 4. Contaminant inputs at selected major estuary mouths* (metric tons/yr)

		BOD (x10 ⁴)	N (x10 ⁴)	P (x10 ³)	Hg	Pb	Cr	Zn	Cd
Hudson-Raritan ^{1,2,3}	A	39	14	17	35	1,050	820	3,600	63
	B	-	14	19	53	2,100	1,600	6,800	100
Rhine-Meuse	A	36	41	40	30	1,200	920	7,800	88
	B	-	-	-	-	1,900	2,100	9,500	120
Thames ³	A	28	16	29	1.8	200	200	1,800	98
	B	-	18	33	3.1	480	380	3,000	110
Humber	A	8.7	5.2	9	0.29	99	60	3,700	16
Mersey	A	11	2.3	3	4.3	56	15	700	22
	B	-	-	-	24	1,600	390	3,500	40
Mississippi ^{2,4}	A	-	-	-	40	13,000	22,000	60,000	500
Severn ³	A	30	63	-	-	970	1,300	3,600	100
	B	32	-	-	-	1,000	1,300	3,600	100
Firth of Forth ³	A	20	2.0	1.1	-	1,200	420	2,000	53
	B	-	2.2	1.3	-	1,200	440	2,100	54

A = River and other runoff only

B = Includes ocean dumping

- Not available

*Data used to construct the table were obtained from various sources including unpublished International Council for the Exploration of the Sea (ICES 1978) reports.

1 Mueller et al., 1976, 1980; New York City, 1983

2 Sackett, 1981

3 Department of the Environment, 1979

4 Trefry and Shokes, 1981

Table 5. Contaminant inputs to selected ocean regions normalized to population (metric tons/yr/10⁶ persons)

	BOD (x10 ⁴)	N (x10 ⁴)	P (x10 ³)	Hg	Pb	Cr	Zn	Cd
New York Bight								
A	3.2	0.84	1.1	3.1	220	96	400	6.2
B	2.3	0.80	1.0	2.1	62	48	210	3.7
Southern California Bight	3.6	-	1.4	0.4	21	53	120	6.0
Mediterranean	3.3	1.0	3.6	1.3	48	28	250	-
Seto Inland Sea	4.6	0.8	0.9	-	-	-	-	-
Baltic Sea	2.3	3.3	0.9	1.9	>55	-	>500	-
North Sea*								
A	2.6	1.7	2.6	0.8	160	140	690	11
B	-	-	-	0.3	90	55	590	10
Irish Sea								
A	4.6	1.0	2.8	4.3	300	140	1300	14
B	-	-	-	1.4	83	62	840	11

*Data for Netherlands not included due to uncertainty of contributing population size (ICES, 1978).

A = total of all inputs except atmospheric
 B = excludes ocean dumping, and atmospheric inputs

- Data not available

Table 6. Selected values of trace metal concentrations in coastal and estuarine waters.*
All concentrations in ug/l except Hg in ng/l.

	Zn	Cd	Pb	Cu	Hg	References
GREAT BRITAIN REGION						
Irish Sea	-	0.03-0.25	-	8.7-17	-	Mullin and Riley, 1956
English Channel	-	0.02-0.26	-	-	-	Mullin and Riley, 1956
5 Areas U.K. Coast	0.8-20	0.01-0.62	0.05-1.2	0.18-3.7	-	Preston et al., 1972
English Channel	2.0	0.06	0.17	0.46	-	Bryan, 1973
Straits of Dover	12	1.6	-	2	-	Dutton et al., 1973
Irish Sea	-	-	-	-	4.8-22	Baker, 1977
English Channel	-	-	-	-	2.4-5.4	Baker, 1977
Irish Sea	-	-	-	-	10-200	Gardner and Riley, 1973a
Irish Sea	-	-	-	-	<5-400	Gardner, 1978
Severn Estuary	-	-	-	-	<10-<60	Gardner and Riley, 1973b
Severn Estuary	12-52	0.3-5.8	0.4-2.5	-	-	Butterworth et al., 1972
Coast of U.K.	1.2-26	-	-	0.4-5.9	-	Bryan, 1969
Corway Estuary	9	0.8	-	3.0	-	Elderfield et al., 1971
Liverpool Bay	12	0.3	1.7	1.5	-	Abdullah et al., 1972
Cardigan Bay	7.5	1.1	2.2	1.7	-	Abdullah et al., 1972
Bristol Channel	10	1.1	1.2	2.1	-	Abdullah et al., 1972
Irish Sea Shoreline	3.8-49	0.03-1.4	0.6-2.9	0.9-3.1	-	Preston et al., 1972
Tamar Estuary	4.1	-	-	2.7	-	Bryan and Hammerstone, 1973
Clyde Estuary	6-25	0.2-1.2	0.5-1.9	0.5-5.0	-	Halcrow et al., 1973
Cardigan Bay	40-88	-	-	-	-	Ireland, 1973
Bristol Channel	3-44	0.4-9.4	0.3-13	0.5-5.4	-	Abdullah and Royle, 1974
Bristol Channel	200-320	1.8-7.5	60-240	16-43	-	Boyd and Romeril, 1974
Menai Straits	5-50	-	-	0.5-3.0	-	Morris, 1974
Poole Harbor	2-68	0.1-0.74	-	1-18	-	Boyd, 1975
Poole Estuary	26	0.8	-	6	-	Darrascott and Watling, 1975

Table 6. (continued)

	Zn	Cd	Pb	Cu	Hg	References
NORTH SEA						
North Sea	1.6	6.2	-	2.8	-	Dutton et al., 1973
North Sea	-	-	-	-	3.4-11	Baker, 1977
North Sea	0.8-12	<0.1-1.6	-	0.3-3.0	-	Dutton et al., 1973
North Sea Coast	31	0.9	-	21	100	Groot de and Allersma, 1975
Wadden Sea	30	<0.1	2	2	-	Groot de and Allersma, 1975
Rhine River	164	1.8	10	11	600	Groot de and Allersma, 1975
North Sea	-	-	1.2-3.8	2.4-18	-	Petrov et al., 1976
Wesser Estuary (Wadden Sea)	4-20	0.05-0.6	-	2.5-20	-	Duinker et al., 1982a
Elbe Estuary	8-28	0.1-0.6	-	5-13	-	Duinker et al., 1982b
NORWEGIAN SEA						
Inner Oslofjord, Norway	41-72	0.67-0.92	1.0-2.8	8.4-10	-	Anderson et al., 1973
Hardangerfjord, Norway	27-3600	0.01-85	2.9-13	3.7-58	-	Stenner and Nickless, 1974
Skjerstadfjord, Norway	14-110	0.01-0.3	1.6-27	5.2-77	-	Stenner and Nickless, 1974
Norwegian Sea	-	-	0.9-5.8	2.2-8.1	-	Petrov et al., 1976
BARENTS SEA						
Barents Sea	-	-	0.5-4.6	2.0-15	-	Petrov et al., 1976
BALTIC SEA						
Baltic	-	-	-	-	<2-23	Somer, 1977
Oresund	-	-	-	-	<78	Somer, 1977
Western Baltic	-	0.05-0.3	-	1-22	-	Schmidt, 1976
Central Baltic	-	0.05-0.2	-	0.5-2	-	Schmidt, 1976
Baltic	-	-	0.6-15	2.8-23	-	Petrov et al., 1976
Baltic Sea	1.5-3.5	3.0-6.0	0.05-0.2	0.6-1.0	-	Magnusson and Westerlund, 1980

Table 6 (continued)

	Zn	Cd	Pb	Cu	Hg	References
NORTH AMERICAN WEST COAST						
Southern California	1.1-14	-	0.6-5.2	1.6-3.0	-	Brooks et al., 1967
Monterey Bay	6.5	0.15	0.9	-	-	Knauer and Martin, 1973
Hawaii-						
California transect	2.0	0.02	-	-	-	Knauer and Martin, 1973
Puget Sound						
Tacoma Bay	9.5-110	0.11-6.7	-	3.0-100	2.3-6700	Roesijadi, 1982
Sequim Bay	0.5-1.4	0.05-0.12	-	0.41-0.83	0.4-1.7	Roesijadi, 1982
NORTH AMERICAN EAST COAST						
Gulf of Maine	3.1-6.9	-	-	1.2-4.0	-	Spencer and Brewer, 1969
Long Island Sound	0.8-6.4	<0.01-0.6	-	0.8-8.7	-	Fitzgerald et al., 1974
Beaufort N.C. (Estuary)	2.8-15	-	-	-	-	Chipman et al., 1958
New York Bight Area						
Apex: dissolved	0.50-0.61	0.003-0.27	0.01-2.0	0.012-0.380	10-90	EPA, 1982
particulate	-	0.003-0.67	-	0.012-0.380	-	
Apex: dissolved	1-120	<0.1-0.8	<0.5-8	<0.2-20	-	Alexander and Alexander, 1977
particulate	-	<0.1	<0.5	0.29-3.0	-	Alexander and Alexander, 1977
Apex						
	2.1-190	0.11-46	-	0.6-47	-	NMFS, 1972; EPA, 1978
	2-150	0.05-4.3	-	<0.6-25	-	Segar and Cantillo, 1976
	3.1-74	<0.05-12	-	0.32-53	-	Cantillo et al., 1982
Raritan Bay	130	2.4	6	73	-	Mytelka, 1973 (in Alexander and Alexander, 1977)
	-	-	1.3-14	2.7-65	-	Walchauer et al., 1978
Sandy Hook Bay	44	0.9	4	64	-	Mytelka, 1973 (in Alexander and Alexander, 1977)
Hudson River: Estuary	13-20	0.31-0.36	-	5.1-6.5	-	Klinkhammer and Bender, 1981
Lower Bay	9.6-21	0.18-0.32	-	2.6-5.8	-	Klinkhammer and Bender, 1981

Table 6. (continued)

	Zn	Cd	Pb	Cu	Hg	References
NORTH AMERICAN EAST COAST (con't.)						
New York Bight Area (con't.)						
Mid-Continental Shelf	1.8-38	0.006-19	0.69-2.4	0.23-18	50-240	EPA, 1978
Mid-Continental Shelf	8.8-150	0.27-19	-	0.8-16	-	NOAA, 1976
Mid-Continental Shelf	15-40	0.3-1.5	-	2-8	-	Segar and Cantillo, 1976
Continental Slope	1.6-21	0.05-0.6	0.8-6.1	<0.2-1.7	40-400	Brezenski, 1975
Continental Slope	-	<0.004-0.051	0.02-0.24	0.1-0.45	-	Kester et al., 1977
Continental Slope	0.2-38	0.4-2.8	<0.2-14.0	<0.1-7.0	90-720	Hauskrecht, 1977
EASTERN ASIA						
Hong Kong	30-810	10-100	40-1400	20-160	-	Chan et al., 1974
Vostak Bay, Sea of Japan	67	-	-	2.5-2.7	-	Saenko et al., 1976
Minamata Bay	-	-	-	-	56-290	Kumagai and Nishimura, 1978
Beppu Bay (Japan)	<0.1-18	-	-	0.4-7.7	-	Shiozawa et al., 1977
MEDITERRANEAN						
Adriatic	-	-	-	-	10-230	Kosta et al., 1978
Gulf of Fos	3.2-400	0.05-12	0.4-14	2.2-17	-	Benon et al., 1978
N.W. Mediterranean	0.8-7.7	<0.05-0.5	-	0.1-4.9	-	Fukai and Huynh-Ngoc, 1976b
N.W. Mediterranean	0.1-11	0.01-0.8	-	0.1-22	-	Fukai and Huynh-Ngoc, 1976b
W. Ligurian Sea	0.5-4.8	0.05-0.43	-	0.06-0.93	17-30	Fukai and Huynh-Ngoc, 1976a
S.W. Mediterranean	0.1-2.6	<0.05-0.51	-	<0.06-0.62	16-30	Fukai and Huynh-Ngoc, 1976b
Ligurian Sea	4-86	-	-	0.1-13	-	Fonselius and Koroleff, 1963
Tyrrhenian Sea	0.84-13	-	-	-	-	Bernhard et al., 1975
Tyrrhenian Sea	0.2-2.3	0.08-0.33	-	0.08-0.74	-	Fukai and Huynh-Ngoc, 1976b
Taranto Gulf	5.8-29	-	-	-	-	Bernhard et al., 1975
Ionian Sea	0.7-2.4	0.06-0.12	-	0.08-0.74	-	Fukai and Huynh-Ngoc, 1976b

Table 6. (continued)

	Zn	Cd	Pb	Cu	Hg	References
MEDITERRANEAN (con't.)						
Adriatic	-	-	-	-	10-210	Kosta et al., 1978
Adriatic	1.1-310	0.05-0.75	-	0.1-130	-	Branica et al., 1976
Northern Adriatic	0.8-24	<0.05-0.60	-	0.1-1.6	-	Bubic et al., 1977
Cretan Sea	0.1-29	-	-	<0.03-2.9	-	Fonselius, 1970b
Aegean & Cretan Sea	1.5-5.8	<0.05-0.12	-	0.08-0.74	-	Fukai and Hynh-Ngoc, 1976a
Cyprus Sea	-	-	-	-	9-140	Robertson et al., 1972
Mediterranean coast, San Remo-Livorno	-	0.1-0.28	-	0.08-4.0	-	Capelli et al., 1976
Mediterranean coast, Ventimiglia-Anzio	-	0.005-0.45	0.018-2.4	0.13-3.6	-	Murnberg et al., 1977
Adriatic Coast, Flonim-Nin	1.5-360	<0.05-6.9	0.02-2.8	0.2-3.6	-	Bubic et al., 1977
Mediterranean coast, Ros Hanigra-Haifa	1-260	0.6-2.9	2.1-11	0.8-31	10-180	Roth and Hornung, 1977
Gulf of Trieste	-	-	-	-	25-100	Majori et al., 1976a
Mediterranean waters	1.0-256.9	<0.05-6.9	0.02-13.3	0.13-31.2	10-180	Bernhard, 1981
ATLANTIC COAST - EUROPE						
Atlantic Coast:Spain/Port.	17-530	0.08-6.0	2.3-38	1.9-110	-	Stenner and Nickless, 1975

*Some Data obtained from Phillips (1977a) and Bernhard (1978)

Table 7. Trace metals in selected marine sediments*
(ug/g dry weight)

	Zn	Cd	Pb	Cu	Cr	Hg	References
GREAT BRITAIN REGION							
Garroch Head	89-590	0.6-8.5	61-210	14-170	-	-	Steele et al., 1973
Firth of Clyde	34-100	0.4-1.5	12-45	2.7-17	-	-	Steele et al., 1973
Swansea Bay	128	-	126	81	-	-	Bloxam et al., 1972
Severn Estuary	470	-	163	-	-	-	Butterworth et al., 1972
Firth of Clyde	140-830	-	77-320	38-210	48-310	-	Mackay et al., 1972
27 estuaries in England	97-3000	<0.02-9.3	-	-	-	-	Bryan and Hummerstone, 1973
The Solent	-	-	-	-	-	0.19-5.7	Raymont, 1972
Firth of Clyde--Sludge Disposal Area	440-680	4-8	270-400	250-300	87-180	2.2	Halcrow et al., 1973
Firth of Clyde--Control Area	60-130	1-3	24-67	9-20	10-65	0.04-0.15	Halcrow et al., 1973
Solway Firth	36-110	<1	12-66	5-16	15-62	-	Halcrow et al., 1973
The Solent	110-800	0.55-4.2	-	-	-	0.26-5.6	Leatherland and Burton, 1974
Poole Harbor	3-220	<1.7	5-190	1-60	-	-	Boyden, 1975
Severn Estuary	280	-	120	38	71	-	Chester and Stoner, 1975
Clyde Estuary	-	-	-	-	-	<0.05-5.3	Bartlett and Craig, 1981
Plymouth Estuary	-	-	-	-	-	0.018-2.6	Millward and Herbert, 1981
Lough Neagh, Northern Ireland	14-270	-	10-260	8-690	-	-	Rippey, 1982
Belfast Lough	46-800	-	52-460	10-200	120-360	-	Parker, 1982
Northern Irish Sea	-	-	-	-	-	0.07-3.3	Rae and Aston, 1981
NORTH SEA							
North Sea	40	0.29	21	6.2	-	0.04	Nicholson and Moore, 1981
North Sea (near Tyne, Tees, Humber estuaries)	20-200	0.1-0.8	17-240	2-50	4-41	0.03-1.0	Taylor, 1979
North Sea	<20->200	-	<10->30	<5->30	-	-	Eisma, 1981

Table 7. (continued)

	Zn	Cd	Pb	Cu	Cr	Hg	References
NORWEGIAN SEA							
Serfjord, Norway	2000	-	11000	2400	-	-	- Skei et al., 1972
Hardangerfjord, Norway	27-64	0.3-0.8	7.7-13	3-10	-	-	- Stenner and Nickless, 1974
Skjerstafjord, Norway	24-58	0.7-1.5	5.0-66	2-18	-	-	- Stenner and Nickless, 1974
Norwegian Fjord	-	-	-	-	-	90-350	Skei, 1978
BALTIC SEA							
Baltic Sea	110	-	25	78	90	-	- Manheim, 1961
Bothnia Gulf	130	-	67	33	-	-	- Hallberg, 1979
Central Baltic	270	-	84	60	-	-	- Hallberg, 1979
NORTH AMERICAN WEST COAST							
California							
7 mile Hyperion outfall	-	0.22-65	7-600	13-1000	62-860	0.04-6.1	Schafer and Bascom, 1976
San Clemente Basin, (100 km offshore California)	190	1.0	10	-	-	-	- Warren, 1981
Saanich Inlet, British Columbia	80-88	-	20	38-45	35-86	-	- Gross, 1967
Palos Verdes, California	54-2900	1.1-6	32-130	14-940	52-1500	0.13-4.4	Hershelman et al., 1981
Southern California	44-2100	0.43-140	10-540	9.0-780	26-1300	-	- Katz and Kaplan, 1981

Table 7. (continued)

	Zn	Cd	Pb	Cu	Cr	Hg	References
NORTH AMERICAN EAST COAST							
Delaware Dumpsite	100	100	400	500	400	-	Watling et al., 1974
Raritan Bay	12-280	<1-15	<4-1000	<1.6-1200	<2-260	-	Greig and McGrath, 1977
New York Bight	4-520	-	4-270	<1.6-210	<2-300	-	Greig et al., 1974
New York Bight	4.6-540	<0.47-9.6	-	<1.9-260	-	0.12-4.9	Timoney et al., 1978
New York Bight	7-480	-	5-270	1-320	2-370	-	Carmody et al., 1973
Narragansett Bay	53-170	0.06-2.5	17-81	26-98	13-81	-	Eisler et al., 1977
EASTERN ASIA							
Tokyo Bay, Japan	-	0.2-3.5	10-110	-	5-180	<0.1-2	FAO, 1976
Minamata Bay, Japan	-	-	-	-	-	1-100	Kumagai and Nishimura, 1978
Yatsushiro Sea, Japan	-	-	-	-	-	0.1-2	Kumagai and Nishimura, 1978
Thailand coast	43-79	0.77-1.5	53-180	6.2-26	-	0.23-2.8	Menasveta and Cheevaparanapinot, 1981
Seto Inland Sea, Japan	110-240	0.14-0.88	14-43	23-54	-	-	Aoyama et al., 1982
Minamata Bay, Japan	-	-	-	-	-	28-710	Fujiki, 1973
AUSTRALIA							
Port Phillip Bay	3.3-280	0.15-9.9	4.6-180	2.2-87	-	-	Talbot et al., 1976
Princess Royal Harbor	3-1200	0.26-7.6	13-180	6.4-122	1.3-42	-	Talbot, 1983
MEDITERRANEAN							
Saronikos Gulf	45-1800	-	-	-	35-1000	0.38-3.1	Papakostidis et al., 1975
Gulf of Catania, Italy	25-100	2.5-4.6	4.5-17	3.8-25	6.1-30	-	Castagna et al., 1982
Mediterranean Sea	2.1-2500	0.02-2.3	2.8-95	-	1.7-1100	0.04-47	Bernhard, 1981
Ligurian Coast, Italy	-	0.3-7.0	36-180	5.0-50	8.2-210	-	Cosma et al., 1982
Mediterranean, Nice-Monte Carlo	45-110	0.1-2.3	35-95	-	12-30	-	Renfro and Oregioni, 1974
Adriatic	31-130	-	8-24	-	10-220	-	Paul and Meischner, 1976
Adriatic	24-120	0.020-0.95	-	-	94-190	0.01-0.16	Kosta et al., 1978

Table 7. (continued)

	Zn	Cd	Pb	Cu	Cr	Hg	References
MEDITERRANEAN (con't.)							
Gulf of Trieste	24-150	0.6-1.8	2.8-67	-	-	-	Majori et al., 1967b
Mediterranean, Po - Rimini	66-110	0.44-1.5	10-92	-	24-87	-	Selli et al., 1977
Saronikos Gulf	50-2500	-	-	-	80-1100	0.5-3	Grimanis et al., 1977
Saronikos Gulf	40-3000	-	-	-	50-1200	0.3-10	Griggs et al., 1978
Mediterranean Rosh	2.1-18	0.3-2.2	4.3-20	-	1.7-12	0.01-0.57	Roth and Horning, 1977
Hanigra - Haifa	-	-	-	-	-	0.05-0.24	Selli et al., 1972
Tyrrhenian Sea	-	-	-	-	-	<0.1-11	Baldi et al., 1979
Tyrrhenian Sea	-	-	-	-	-	0.04-1.3	Renzoni et al., 1973
Mediterranean, Tuscany Coast	-	-	-	-	-	0.05-0.1	Selli et al., 1972
Adriatic, Iserzo- Trieste	-	-	-	-	-	0.2-47	Majori et al., 1976b
Adriatic, Iserzo river mouth	-	-	-	-	-	24-57	Kosta et al., 1978
Bay of Naples, Italy	-	-	5-200	9-130	20-700	-	Griggs and Johnson, 1978
Alexandria, Egypt	-	-	-	-	-	0.1-16	El-Sayed et al., 1979
Israeli Coast	20-80	-	15-28	5-40	30-60	0.2-0.5	Amiel and Navrot, 1978
OTHER							
Gulf of Paria, Venezuela	-	-	20	19	100	-	Hirst, 1962
Atlantic Coast of Spain and Portugal	6-3100	0.9-4.1	6-1600	2-1400	-	-	Stenner and Nickless, 1975
Bermuda, Hamilton Harbor, Hungry Bay	16-65	<0.25-0.99	6.4-230	7-130	15-480	-	Lyons et al., 1983
Kuwait, Persian Gulf	24-90	0.75-3.0	10-40	12-30	35-95	-	Anderlini et al., 1982
Argentina, Blanco Bay	26-64	1.9-3.1	-	7.5-24	-	-	Sericano and Pucci, 1982
Deep sea clays	160	0.4	80	250	-	-	Segar and Pellenberg, 1973

*Some data obtained from Phillips (1977a) and Bernhard (1978).

Table 8. PCB concentrations in seawater

Area	Concentration Range (ng/l)	Reference
Mediterranean Sea, open sea	0.2-8.6	Elder and Villeneuve, 1977
French Mediterranean coast	1.5-38	Elder et al., 1976
North Atlantic	1-150	Harvey et al., 1973
Atlantic North of 30°N	0.4-7.1	Harvey and Steinhauer, 1976
Atlantic (30°N-30°S)	4.0-8.0	Harvey and Steinhauer, 1976
Sargasso Sea	0.9-3.6	Bidleman and Olney, 1974
Southern California coast	2.3-36	Scura and McClure, 1975
Baltic Sea	0.3-140	Ehrhardt, 1981
Japan coastal waters	600-900	FAO, 1976
Puget Sound, Washington	3.0-22.0	Pavlou and Dexter, 1979
Gulf of Mexico	0.8-4.1	Nisbet, 1976
New York Bight	0.08-3.5	Boehm, 1981
New York Bight	1-3	EPA, 1982
	10-50	MacLeod et al., 1981
	1-44	Pearce 1980, (in EPA, 1982)
	20-80	West and Hatcher, 1980

Table 9. PCB concentrations in sediments

Area	Concentration Range (ug/g)	Reference
New Bedford Harbor, Massachusetts	<190,000	Massachusetts Office of Coastal Zone Management, 1982
Massachusetts Bay	0.001-0.030	New England Aquarium, 1976 (from Boehm, 1983)
North Atlantic, U.S. (excluding New York Bight)	0.0003-0.044	Boehm, 1983
Housatonic River, Connecticut	0.1-2.8	Sawhney et al., 1981
Hudson River, New York	13-160	Carcich and Tofflemire, 1982
	0.5-140	Bopp et al., 1981
New York Bight Apex	0.0005-2.2	West and Hatcher, 1980
	0.05-0.15	Boehm, 1980a
	0.006-0.29	Boehm, 1983
Chesapeake Bay, Maryland	0.07-0.13	Bopp et al., 1981
	0.004-0.4	Sayler et al., 1978 (from Boehm, 1983)
Delaware River, Delaware	0.14-0.32	Bopp et al., 1981
Escambia Bay, Florida	0.19-61	Nisbet, 1976
Gulf of Mexico	<0.002-0.035	Nisbet, 1976
Columbia River, Oregon	0.04-0.06	Bopp et al., 1981
San Francisco Bay, California	0.03-0.05	Bopp et al., 1981
Elliot Bay, Puget Sound	<0.4->1.6	Pavlou et al., 1982
Palos Verdes coastal sediments, California	0.08-13	Young and McDermott-Ehrlich, 1976
Puget Sound, Washington	0.08-0.64	Pavlou and Dexter, 1979
Japan, coastal and estuarine sediments	0.01-390	FAO, 1976

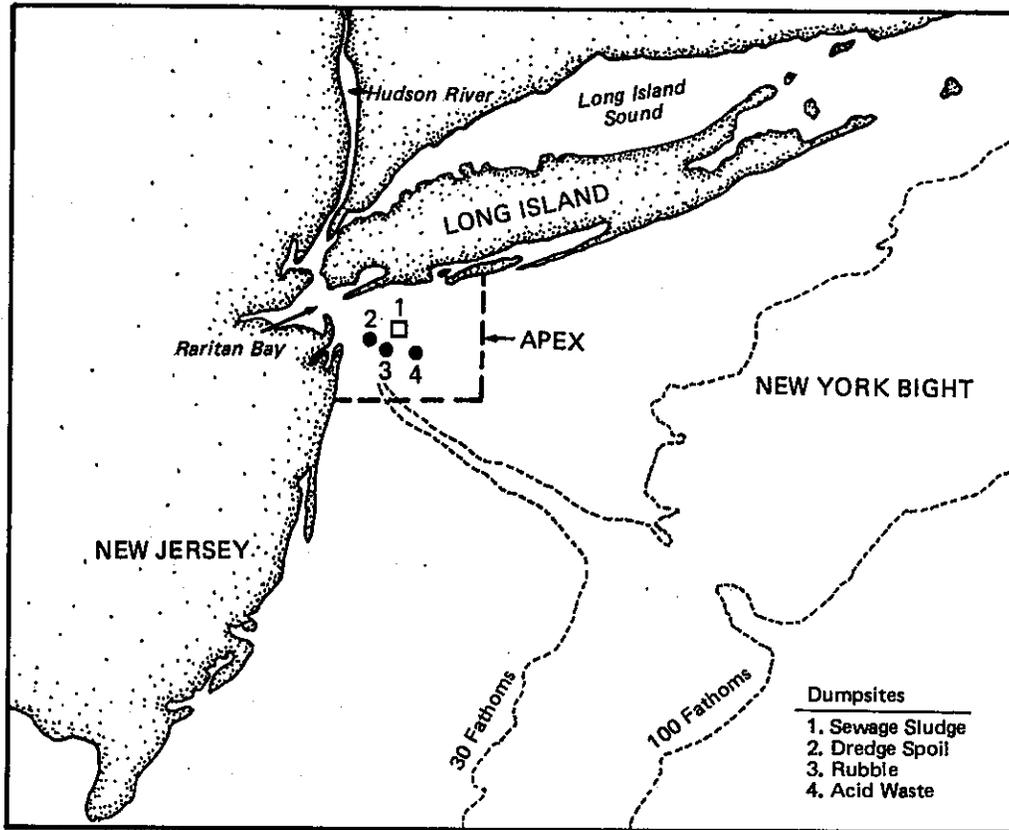


Figure 1. The New York Bight.

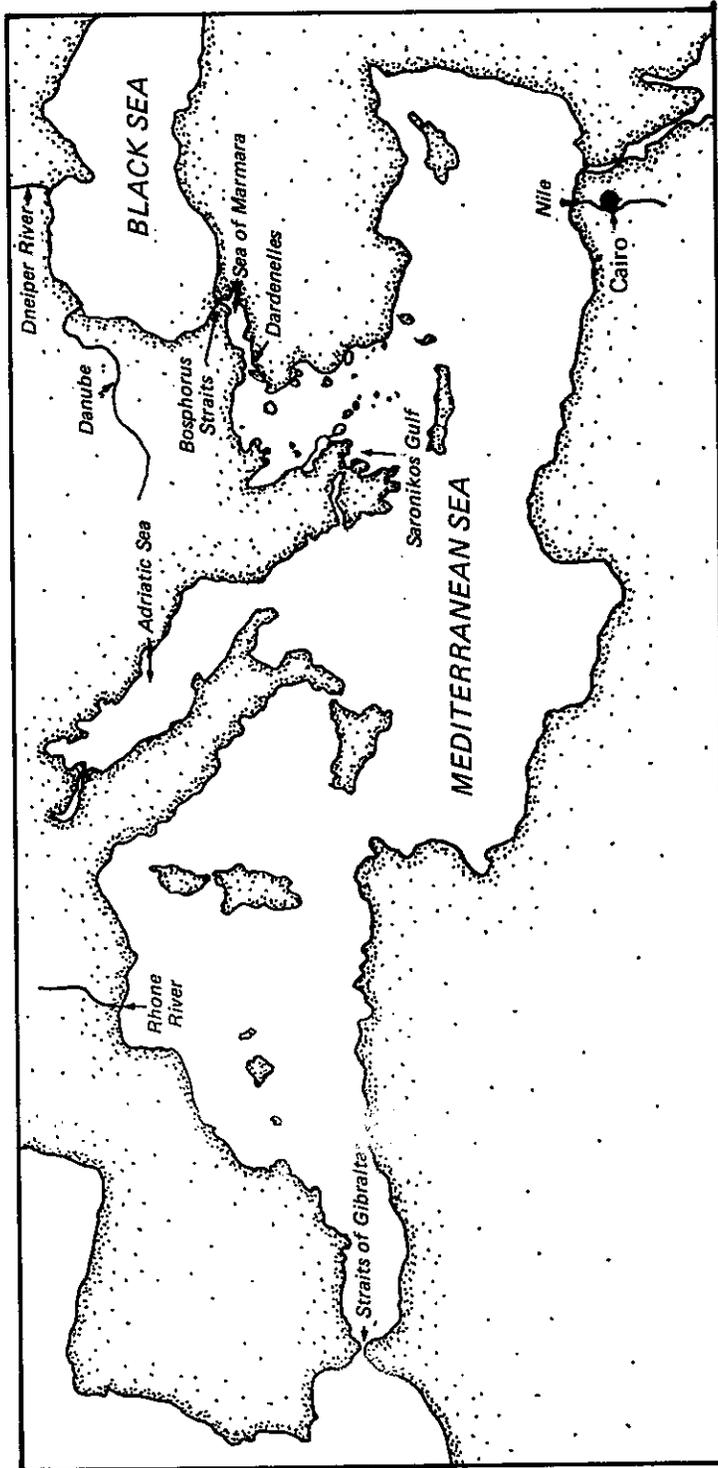


Figure 2. The Mediterranean Sea showing the Adriatic and Saronikos Gulf.

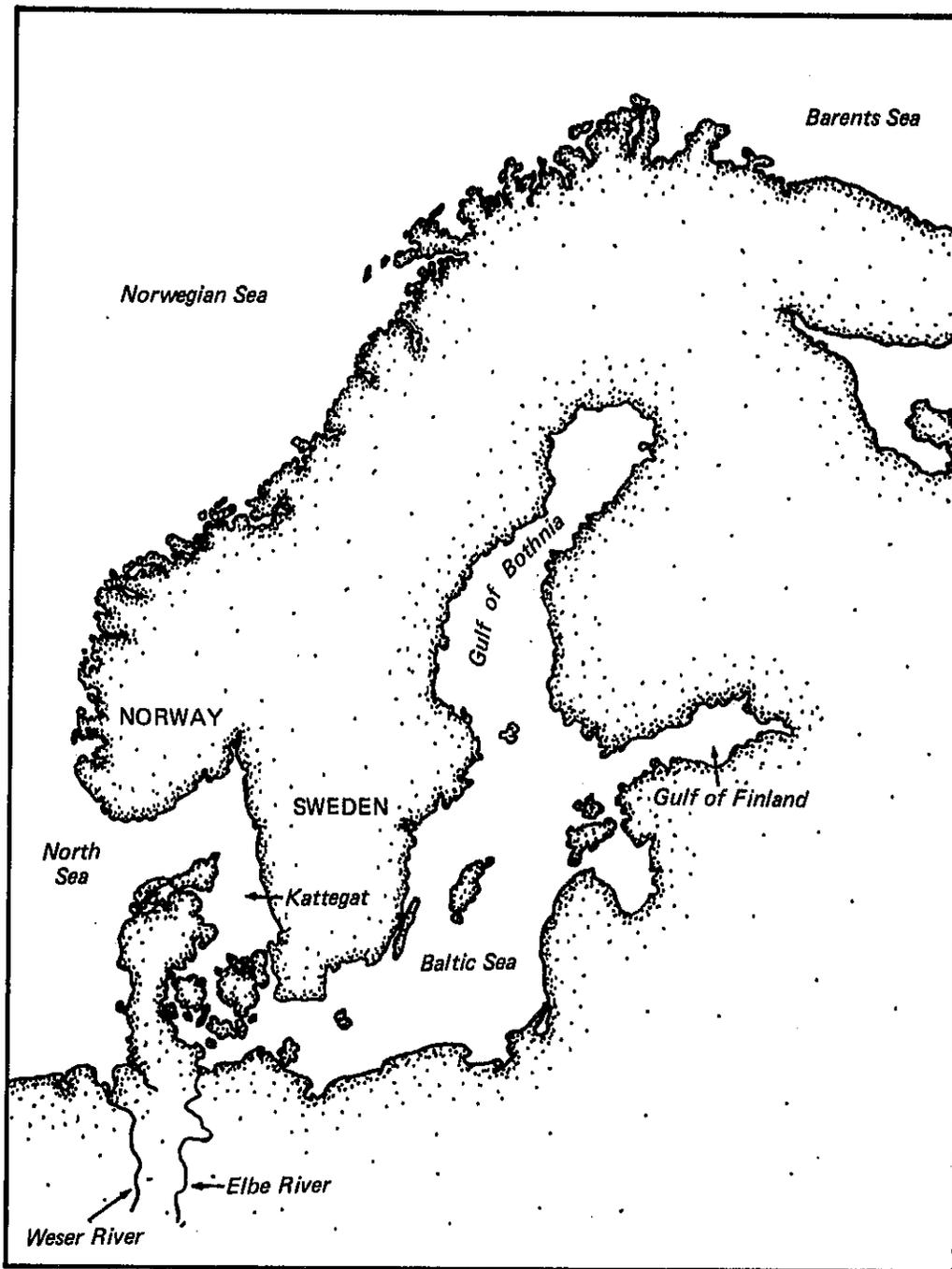


Figure 3. The Baltic Sea.

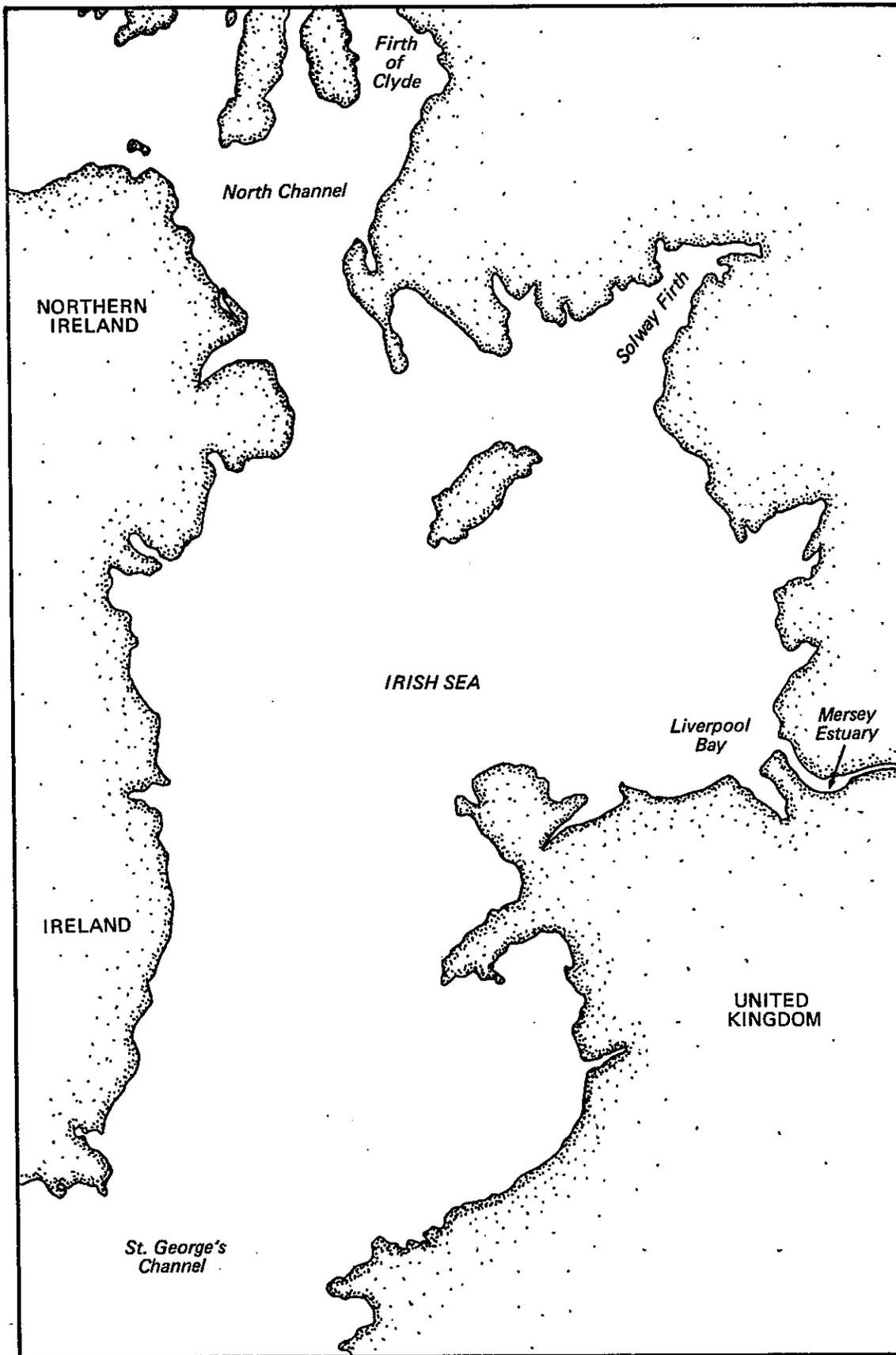


Figure 4. The Irish Sea showing the Mersey estuary and Liverpool Bay.

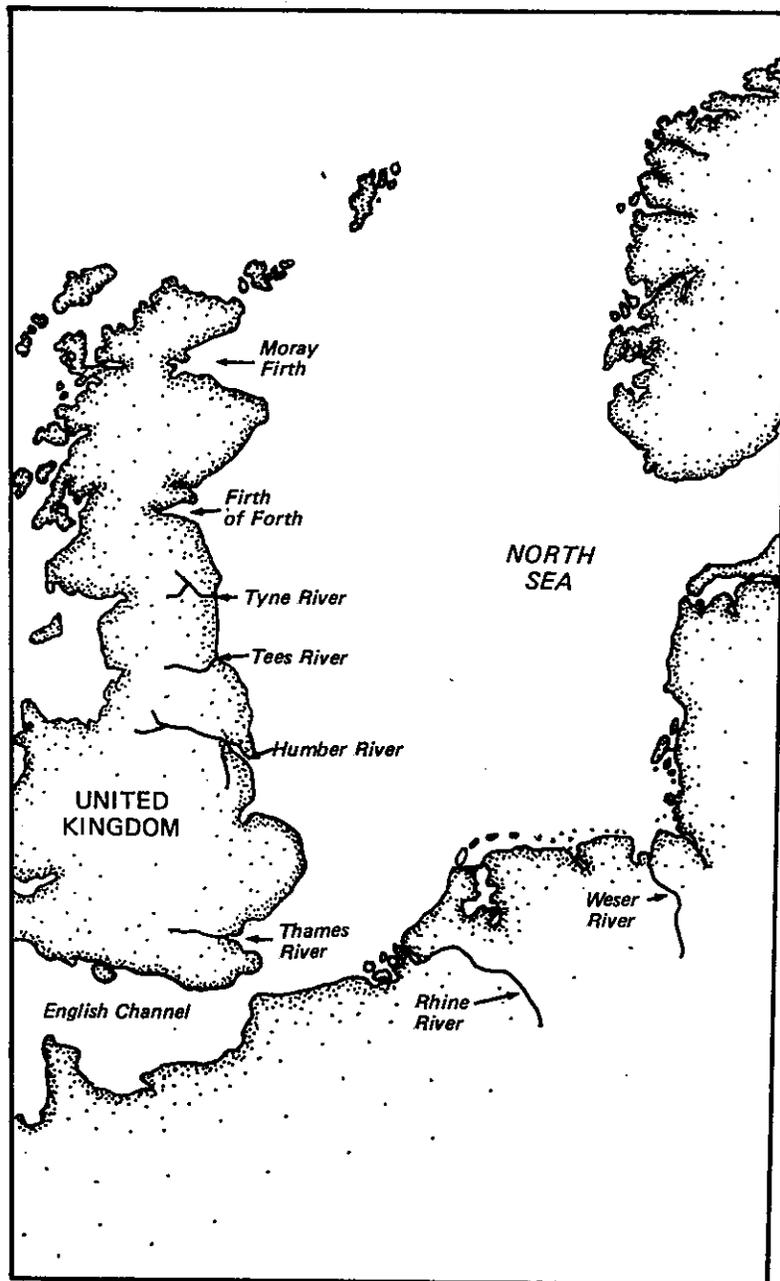


Figure 5. The North Sea showing the Rhine, Thames, Tyne, Tees and Humber rivers.

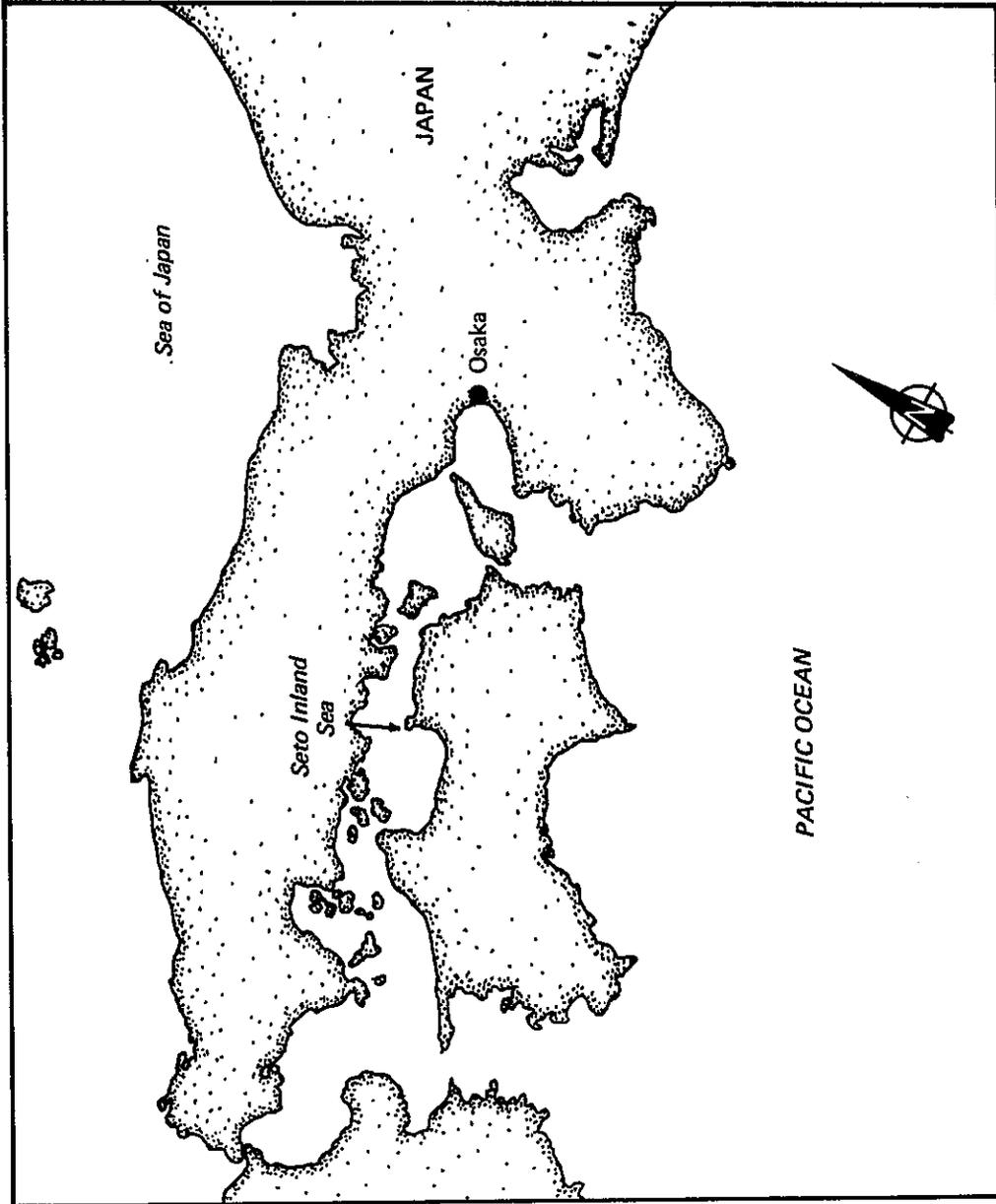


Figure 6. The Seto Inland Sea, Japan.

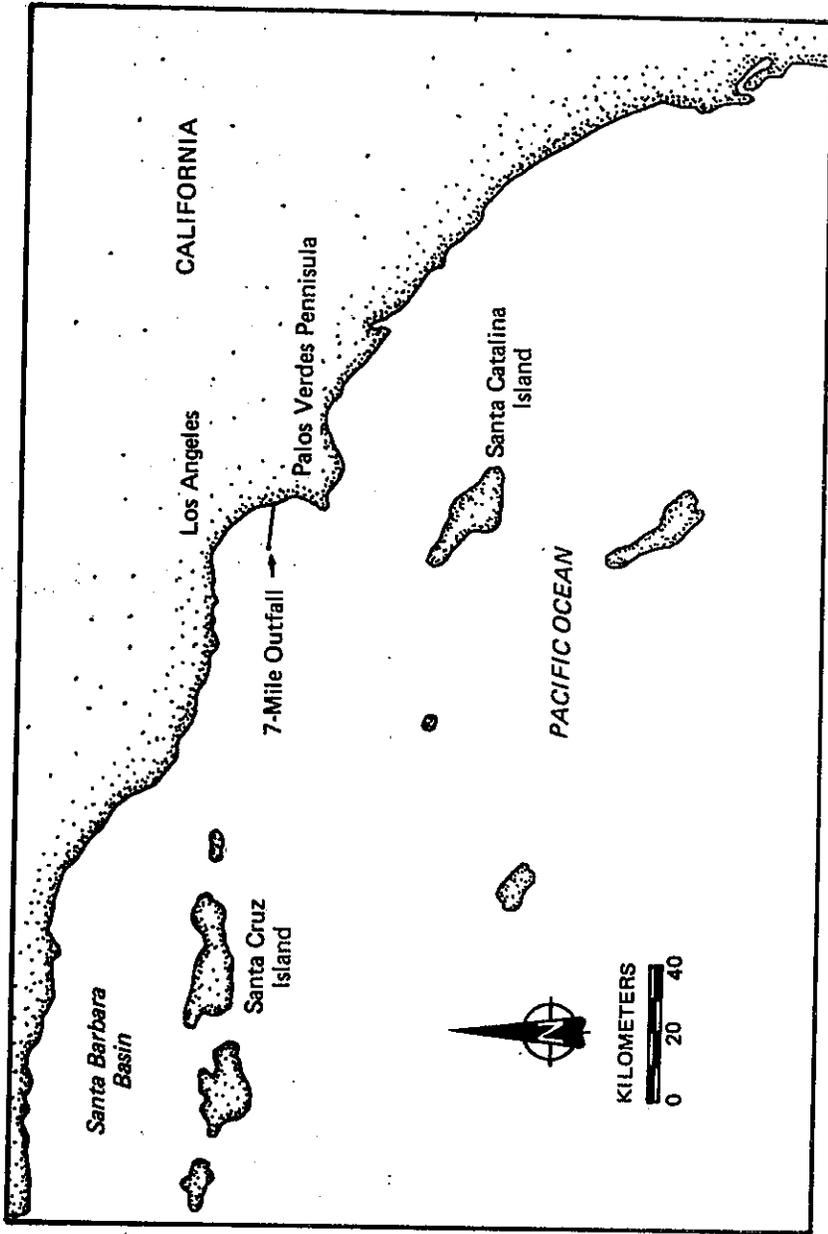


Figure 7. Southern California showing location of 7-Mile outfall.

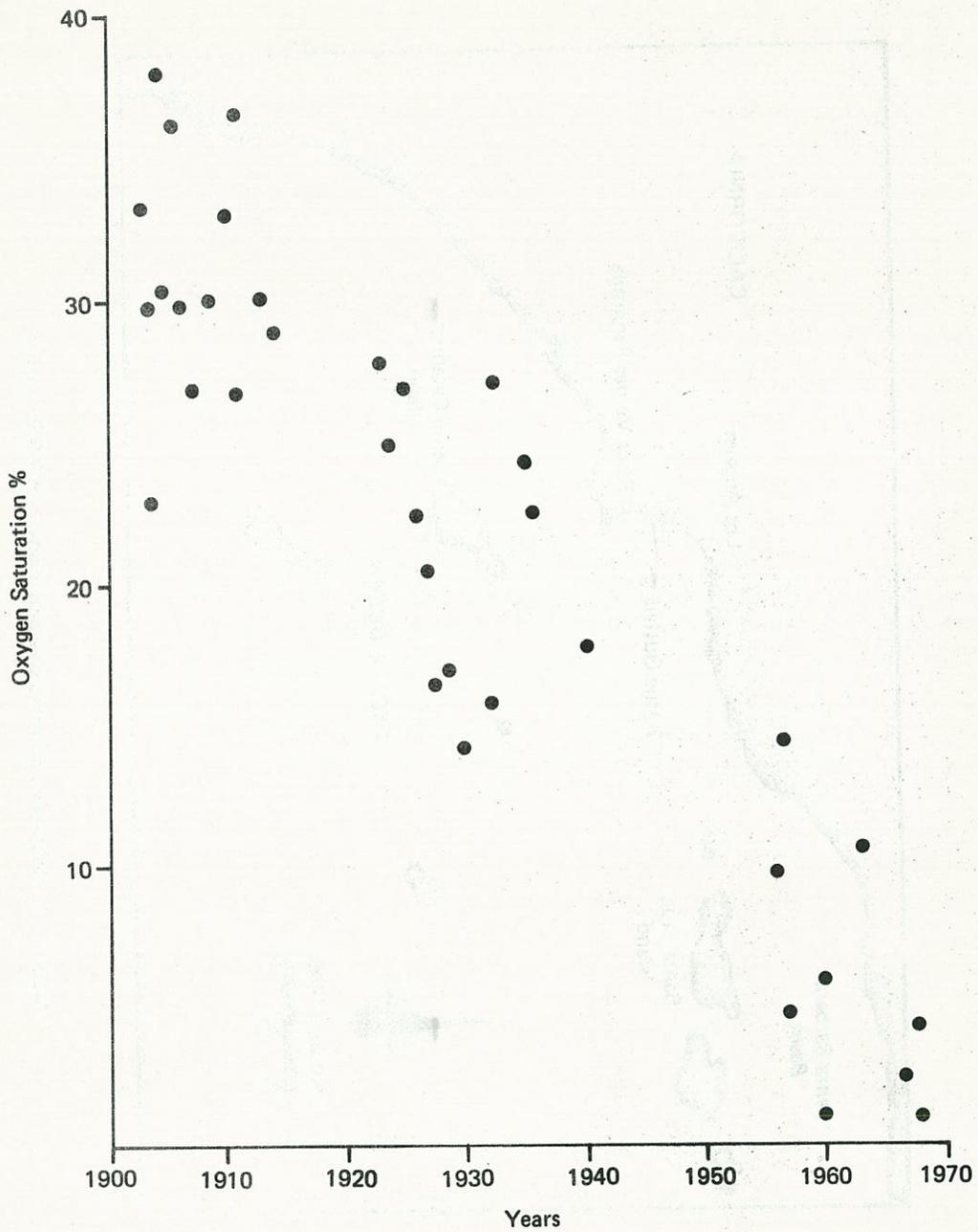


Figure 8. Oxygen saturation of sub-halocline waters in the Baltic 1900 – 1970 (from Kremling 1973).