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MIDDLE ATLANTIC CONTINENTAL SHELF AND THE NEW YORK BIGHT

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FOREWORD

As part of the world's largest urban region, the New York–New Jersey–Connecticut metropolitan area has long symbolized many of the attractive aspects of American society. But to many, it has become an unhappy example of the impact of man's wastes on the ocean. Since 1970 the New York Bight—the coastal ocean between Long Island and New Jersey—has been characterized in the newspapers as a "dead sea" where marine animals have been killed by decades of waste disposal and from which a blanket of sewage sludge moves inexorably toward the region's beaches. Each new story has triggered public outcries, legislative hearings, and agency investigations to check on the concerns of the group or agency involved.

From the individual research projects and from the several larger scale investigations supported by the Marine EcoSystems Analysis Program of the Environmental Research Laboratories of the National Oceanic and Atmospheric Administration and by the U.S. Environmental Protection Agency, an understanding is developing of the complex chemical, physical, geological, and biological processes that control the New York Bight as a productive marine ecosystem.

In the past, with smaller populations, lower waste loads, and fewer individuals able to enjoy the ocean, there was less concern about conflicts among these competing usages. But by the mid-1970s, more coastal cities discharge their wastes in the ocean, while the average citizen demands a healthier, more pleasant environment.

Such a complex of problems is new to most oceanographers and marine scientists. Oceanographers have long been accused of living in idyllic rural locations, taking long cruises to remote areas, and turning off their instruments when they encounter coastal or polluted waters. And a familiar preface to funding requests is: "As soon as we understand this unaltered environment, we can turn to polluted areas and hope to unravel the processes which control them." All too commonly, the transfer to polluted areas never happens. It is only in the past few years that most oceanographers have approached urban problems. But the New York Bight is one of the few areas where oceanographers have carried their research into dirty waters.

The New York Bight and the middle Atlantic shelf represent much more than a dumping ground. Rich and abundant commercial and recreational fisheries occupy these waters, and wildlife sanctuaries dot the less-populated shores. It is the setting for the Gateway National Recreation Area project, which seeks to provide oceanic recreation opportunities in a relatively natural environment for the millions of area residents. It is also a busy conduit for marine commerce, crossed by some of the most heavily traveled shipping lanes in the world. The problems arising from the often conflicting uses are severe enough, but the area's future is further complicated by a host of proposed major developments—offshore nuclear power plants, superports, deep-water oil terminals, and artificial islands are all in various planning stages.

As scientists, we have much to learn about polluted marine environments and how to research them. This symposium was organized as a step in that learning process.

Those who organized the symposium hope they have made such a step and that as a result research programs will be improved in comparable coastal ocean or lake areas.

M. Grant Gross

OPENING REMARKS

It is a pleasure for me, on behalf of President Dugdale and the American Society of Limnology and Oceanography, to welcome you to this symposium concerning the continental shelf and the New York Bight. As you know, several years ago the society sponsored Special Symposium 1 which dealt with what was then a highly contentious subject: The Limiting-Nutrient Controversy. That symposium resulted in a more balanced perspective concerning the relative importance of certain nutrients, such as phosphorus, nitrogen, and carbon, in the process of eutrophication.

The purpose of these special symposia is to provide a forum within which interested parties—scientists, engineers, and others from academic, state, and Federal institutions—can communicate their special insights with regard to broad-scale problems in the aquatic environment. A further purpose is that these insights will be distilled and disseminated to the nontechnical arenas of society in such a way that the issues will be balanced and understandable and the conclusions useful.

We come now to Special Symposium 2. Again we are dealing with the impact of human activities on a broad and important area of the marine environment. I don't know how controversial the issues are but certainly there are issues. Perhaps our discussions will help to clarify these issues.

It is our hope that you will not only describe this marine environment in its present state but that you also will tell us what the problems are now and what they might be in the future. We also hope that from these discussions a sense of priorities and a suggestion for sound management will emerge to ameliorate, or minimize, or even prevent significant future damage to this important marine resource area.

I would like to thank the Chesapeake Bay Institute, the New York Sea Grant Institute, and the Marine EcoSystems Analysis Program Project, National Oceanic and Atmospheric Administration, for helping to organize this meeting. In particular, I would like to thank M. Grant Gross who has so ably and efficiently chaired the organizing activities.

The American Museum of Natural History is an interesting and delightful place to hold a large meeting such as this. I would like to end my brief remarks by wishing you an interesting, successful, and useful 3-day meeting.

George W. Saunders, Secretary
American Society of Limnology
and Oceanography

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Section 1 Overview

Man's impact on the middle Atlantic continental shelf and the New York Bight—Symposium summary

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Human activities in coastal ocean areas are causing ever-increasing concern, especially near urban regions. Serious questions have been raised about the impact on this coastal ecosystem of releasing wastes and contaminants, siting power plants, and drilling for oil on the continental shelf.

The symposium considered the environmental quality of the middle Atlantic continental shelf and New York Bight and assessed man's impact on this continental shelf ecosystem. Particular attention was given to applications of research results to questions of public policy and resource allocation in addressing man-related environmental changes. Here, we summarize these results and attempt to place them in a regional and national perspective.

The Middle Atlantic Bight, between Cape Cod and Cape Hatteras, borders the center of the Atlantic Urban Region, the world's largest urban cluster, which extends from southern New Hampshire to northern Virginia (Pushkarev 1969). The region housed about 45 million persons in 1970, and is expected to exceed 70 million persons in 2000.

The New York Bight extends to the edge of the continental shelf, about 150 to 180 km offshore and is bounded by Long Island on the north and New Jersey on the west (Fig. 1).

Man's impact on the marine ecosystem of the middle Atlantic continental shelf is directly related to the variety and intensity of human activities there. New York Bight waters, particularly in the apex, are heavily used for a variety of activities by the dense population on its edges—transportation, commercial and sport fishing, recreation, boating, and waste disposal. Wastes from about 19 million persons are carried by the Hudson and other rivers into the bight. Newly built sewage treatment plants on Long Island's south shore and in the New Jersey coastal region use pipelines and offshore discharge to handle the waterborne wastes discharged from the heavily industrialized and densely populated region.

In 1974, the New York Bight received 93% (by volume) of wastes (excluding dredging) discharged through ocean dumping in the United States coastal waters (EPA 1975). The volume of wastes discharged increased from 4 to 20% between 1973 and 1974 (*see* Gross 1976).

Waste disposal in the New York Bight—Most of the concern about the New York Bight in the mid-1970s has involved waste disposal, primarily ocean dumping—the

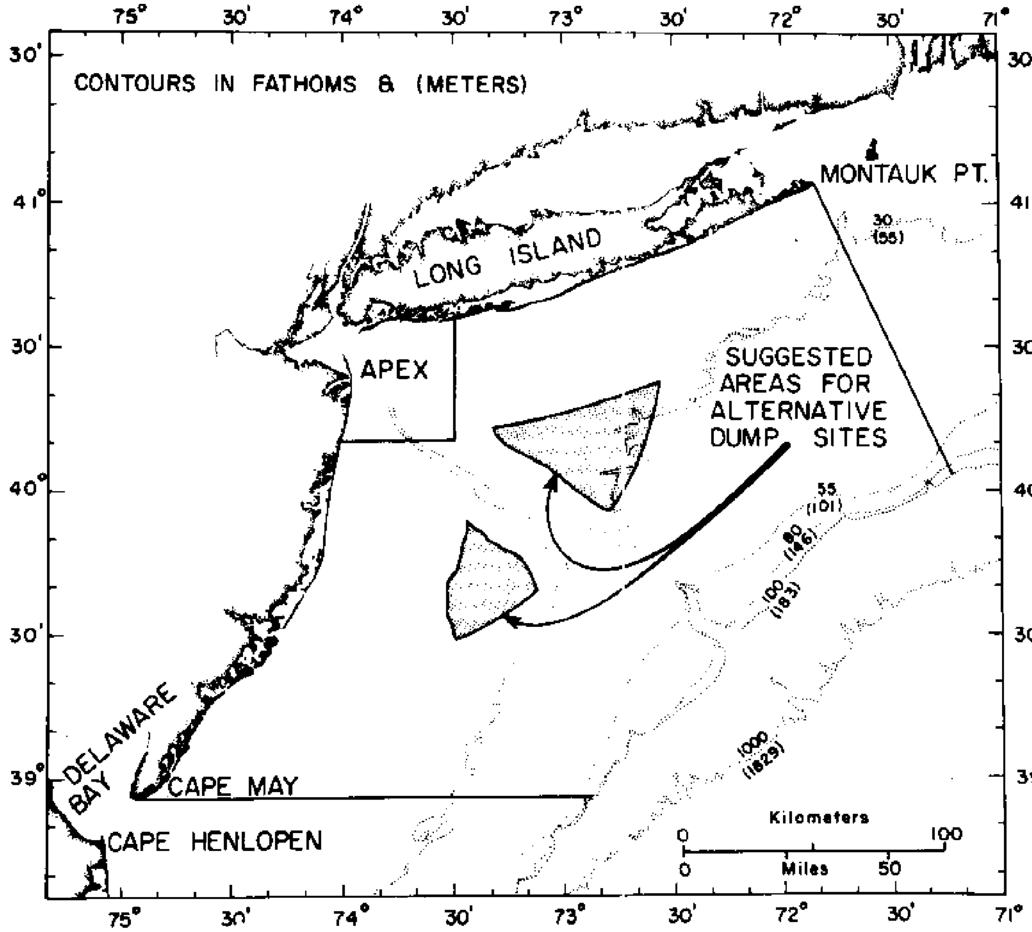

Fig. 1. Locations of suggested alternative dump sites in the New York Bight and apex.

disposal of wastes transported to sea aboard barges or ships. Disposal of wastes in the harbors, rivers, and offshore waters of the New York–New Jersey metropolitan region, however, is an old and recurrent problem. And efforts to regulate disposal activities have a history nearly as long. On 30 December 1675, Governor Edmund Andros, second English governor of the Colony of New York, forbade any person to "cast any dung, dirt, refuse of ye city or anything to fill up ye harbor or among ye neighbors under penality of forty shillings" (Rush 1920, p. 27). The State of New York attempted to regulate dumping and unloading of materials in the waters of the port (Laws of New York 80th Sess., c. 671, vol. II, p. 487, 16 April 1857). New Jersey enacted similar regulations in 1882 (New Jersey Laws, Session of 1882, c. CXVIII, p. 155, 22 March 1882). Because of a lack of funds and divided responsibility between the two states, the regulatory efforts were unsuccessful (H. C. Miller unpublished manuscript) and the federal government assumed jurisdiction under the Supervisor of the Harbor Act (Act of 29 June 1888, c. 496, 25 Stat. 209).

Wastes dumped in the bight include dredged material, sewage sludge, cellar dirt, construction debris, acid wastes, and toxic chemicals (not addressed in this symposium). Four disposal areas are used (Fig. 2).

Dredged materials from vessel berths, anchorage grounds, and channels and coal

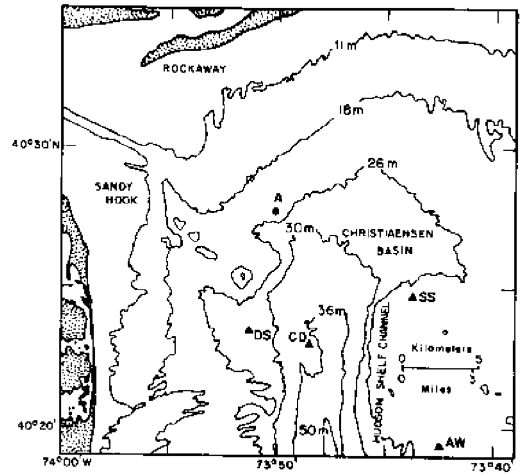

Fig. 2. Bottom topography of the New York Bight apex and major features. A—Ambrose Light; SS—sewage sludge site; DS—dredge material site; CD—cellar dirt site; AW—acid waste site. The toxic chemical site lies outside the area shown.

ash from steam-electric power generating plants are dumped at one site (Gross 1972). Between 1965 and 1970 an average of 150× 10^6 ft^3 (4.3×10^6 m^3) of dredged material was dumped annually (Pararas-Carayannis 1973).

The bight is one of two offshore areas of the United States used in 1975 for sewage sludge dumping. (The other is offshore from the entrance to Delaware Bay, where sludge from Philadelphia and Camden is dumped.) The sewage sludge dump site, selected in 1924, is located in waters 90 ft (27 m) deep. Approximately 150×10^6 ft^3 (4.3×10^6 m^3) of sewage sludge were discharged at or near this site in 1973 (EPA 1975).

Sewage sludge is usually about 5% solids, consisting of three major fractions. The denser solids sink to the bottom near the disposal site. The liquid fraction and low density solids remain suspended in the water for some time after dumping, depending on sludge composition and on currents and water density in the disposal area. This second fraction is composed of dissolved and suspended solids in the water column. The third fraction is that which remains at or near the surface—floatables and surface-active material. All fractions contain hydrocarbons, industrial metals, pathogenic organisms, and treatment chemicals.

Other sources of sewage materials entering the bight include river outflows, sewage treatment plants, industrial outfalls, and storm water runoff and overflows (Klein et al. 1974).

Waste acid and extracted ores from titanium dioxide production have been disposed of at the acid waste site since 1948. Between 1965 and 1970 the average annual amount dumped was 72×10^6 ft^3 (2.0×10^6 m^3) (Pararas-Carayannis 1973).

"Cellar dirt" is the debris from construction and demolition such as stone, tile, brick, concrete, masonry material, and pipe. The cellar dirt site has been used since about 1945. The average yearly volume dumped during 1965–1970 was 16×10^6 ft^3 (4.5×10^5 m^3) (Pararas-Carayannis 1973).

Previous studies—The New York Bight was studied originally as part of regional investigations in the Middle Atlantic Bight. Bigelow (1933) and Bigelow and Sears (1935) included the New York Bight in their studies of the annual cycle of temperature and salinity of continental shelf waters between Cape Cod and Chesapeake Bay. In studying the accumulation of river discharges in the shelf waters, Ketchum and Keen (1955) included the bight as did Bumpus (1973) in his description of circulation on the continental shelf. Williams and Duane (1974) provided information on the shelf geology and marine sand and gravel resources.

Studies focused on the New York Bight have generally been made in conjunction with marine waste disposal problems. The earliest complete published oceanographic survey of the region before the recent initiation of the work on the disposal sites was done by Redfield and Walford (1951) in their studies of the effects of the disposal of titanium dioxide processing wastes. Ketchum et al. (1951) summarized observations from these early studies, and Ayers (1951–1952) surveyed New York Harbor and surrounding waters.

Since the late 1960s, several studies have been primarily concerned with various aspects of ocean disposal practices. Buelow

(1968) discussed the effects of sewage sludge disposal on surf clam production near New York Harbor and Delaware Bay. Gross (1970) noted that waste solids disposal in the New York Bight was the largest sediment source in the Middle Atlantic Bight. Gross (1972) also showed that waste solids formed deposits on the ocean floor, easily differentiated by their anomalously high carbon and metal contents.

Pearce (1972) showed that waste disposal operations had altered distributions and abundances of bottom dwelling marine organisms. Pararas-Carayannis (1973) summarized results of U.S. Army Corps of Engineers research programs. Results obtained by the Marine EcoSystems Analysis (MESA) Program of the National Oceanic and Atmospheric Administration (NOAA) have also been summarized (NOAA 1975). These results were evaluated in recommending against moving the sewage sludge disposal site (NOAA 1976).

The New York Bight environment

The environment of the New York Bight—its physical, chemical, and geological makeup—and man's impact on it depend on the larger scale processes of the entire Middle Atlantic Bight. Many of the driving forces and processes that most effectively control the bight ecosystems are determined by events that affect the entire Middle Atlantic Bight.

Water circulation and properties—Air-sea interactions involving the waters of the Middle Atlantic Bight are typical of mid-latitude east coast areas (Laevastu et al. 1976). In autumn and winter, relatively dry continental air blows across the coastal ocean causing intense evaporation and cooling of surface waters. The large heat and moisture inputs to the atmosphere modify coastal winds and storm systems. Storms cause rapid deepening of the pycnocline and cooling of surface waters.

Beardsley et al. (1976) showed that continental shelf waters flow to the south at average speeds between 5 and 10 cm/s. Storm winds, especially northeasters, can move shelf waters over the entire Middle Atlantic Bight. These storm-caused movements reach speeds of 25 to 50 cm/s with the waters moving in a slablike flow. There is also exchange of water associated with these storm events. When the waters are unstratified, the return occurs in a near-bottom layer. During stratified conditions, return flows occur at middepth above the cold-water lens at the shelf break. These exchanges take place between slope and shelf waters at speeds of 3 to 6 cm/s. The time required for these currents to exchange waters in the Middle Atlantic Bight is about 9±3 months.

Waters of the inner New York Bight exhibit estuarine circulation typical of coastal areas where discharges of river water exceed evaporation. Surface waters move generally seaward above the pycnocline while near-bottom waters move generally landward. Gordon et al. (1976) analyzed conditions in October 1974, during the transition between stratified (summer) and vertically homogeneous (winter) water columns. The salinity distribution is complicated over the mid- and outer shelf because of intrusions of relatively saline and warm slope waters. Such intrusions may be important in exchanging waters between shelf and slope regions.

Bowman and Wunderlich (1976) discussed seasonal variations in water properties in the bight apex. Water movements are dominated by river discharge and local winds During fall and winter, the density structure is dominated by salinity effects. In late spring and summer, density distributions are dominated by local heating and the water column is strongly stratified. In spring, the Hudson River outflow plume is distinctly marked by low surface salinities and spreads over most of the bight apex, its position and movements determined by local winds. In fall and winter, the density field is essentially vertical. During late spring and summer, it is distinctly horizontal.

Sediment distribution and transport—Major topographic features of the middle Atlantic continental shelf were formed during the low stands of the sea in the Pleistocene Ice Ages (Swift et al. 1976). The dominant bottom feature of the New York Bight is the Hudson Shelf Channel, apparently cut by the ancestral Hudson River during times of low sea level. Reworking of shoreline deposits as sea level rose left sand deposits (up to 10 m) over most of the bight shelf (Williams and Duane 1974).

Intense storms generate bottom currents strong enough to cause sediment transport and erosion forming wave ripples, sand waves (up to 1 m high), and sand ridges (up to 10 m high and extending tens of kilometers obliquely to the shoreline). Present sediment distributions (except for waste deposits) are the result of sea level fluctuations during the past 15,000 years (Freeland et al. 1976). Discontinuous deposits of fine sand, up to 10 m thick, were deposited when the shoreline moved across the continental shelf, between 15,000 and 5,000 years ago. Fine-grained sediments are typically resuspended and deposited in estuaries and salt marshes or carried beyond the shelf edge into the open ocean. There is no compelling evidence that large volumes of sediment from the region's rivers are now accumulating on the middle Atlantic continental shelf.

Biscaye and Olsen (1976) found that concentrations of suspended particles (<0.4 μm) generally decreased seaward. Particle concentrations were higher in waters overlying fine-grained deposits in the Hudson Shelf Channel. The fine-grained deposits there were also a localized source of dissolved radon, a radioactive tracer (3.9-d half-life).

Man's impact on the New York Bight environment—Disposal of sewage sludge is not the primary contributor of contaminants to the bight. Major inputs of contaminants to the New York Bight were reported by Mueller et al. (1976) to be wastewater discharges, runoff from the Hudson River estuary, and barge discharges directly into the bight apex. Direct inputs from the Long Island and New Jersey coastal zones were less than 6% of the total. Groundwater discharges were found to be insignificant. Data on atmospheric inputs and urban runoff were shown to be seriously deficient. Metals (Pb and Cr) come primarily from dredge spoil discharges, except for Hg which comes primarily from wastewater discharges. Nitrogen inputs come primarily from wastewater discharges while phosphorus is derived primarily from dredged materials.

Segar and Cantillo (1976) reported on metal contaminants (Cr, Cd, Cu, Fe, Hg, Mn, Zn) released to the New York Bight. They concluded that the Hudson estuarine system was the dominant source of dissolved metals, particularly Mn. A large portion of these metals, particularly Fe and Mn, are precipitated from the water column during mixing of estuarine and oceanic waters. Metals, such as dissolved Zn, are released from dumped materials. As a result of such releases as well as the estuarine sources, concentrations of Cd, Fe, Hg, and Zn are considerably higher in waters of the bight apex than in waters offshore. Copper concentrations were found to be more uniform.

Callaway et al. (1976) studied dispersion of sewage sludges discharged from ships and barges in the New York Bight. The sewage sludges they studied averaged about 3.2% solids with a bulk density of 1.014 g/ml, significantly less than seawater. Thus, the liquid portions of the sludges tended to remain with and be transported by near-surface layers. Sludges were initially diluted by mixing (125- to 600-fold) about 15 min after dumping ceased. Densities of solid particles in the sludges averaged 1.8 g/ml and thus tended to settle to the bottom, at speeds up to 2 cm/s. Mass centroid depths shifted at rates of 0.01 to 0.003 cm/s. Particles had diameters of 2–20 μ (determined by Coulter counter). When the waters were well mixed, particles rapidly dispersed throughout the entire water column. When the waters are stratified, the liquid portion of the sludge tends to remain near the surface layers, moving with them, while undergoing further dilution.

Segar and Berberian (1976) analyzed sources of oxygen-consuming organic matter to the New York Bight apex to determine which were the major causes of the oxygen depletion there. They reported that locally produced carbon from phytoplankton accounted for much of the oxygen demand in the apex, especially in summer. Sewage sludge and riverborne organic matter were generally of equal importance. During stratified conditions, seabed oxygen consumption had little effect on dissolved oxygen concentrations in the region's lower water layer. Since the bulk of the dissolved inorganic nitrogen that supports these high levels of productivity is discharged with effluents from New York Harbor, Segar and Berberian argued that river input is the dominant factor causing low dissolved oxygen levels in the bight during late summer.

The water circulation in Raritan Bay, reported by Parker et al. (1976) controls nutrient and chlorophyll *a* distributions in the bay. The gyre formed in the bay tends to retain nutrients, chlorophyll, and suspended solids, thus locally increasing phytoplankton growth in the estuarine system. Surface waters, rich in chlorophyll *a* and nutrients, flow past Sandy Hook into the bight apex.

Physical effects of waste disposal are the most striking; topography and sediment types at disposal sites in the bight apex have changed markedly. Between 1936 and 1973, about 88 million cubic meters of dredged waste was deposited, causing up to 10 m of shoaling. Accumulations of sewage-derived solids may also have altered sediment properties in the head of the Hudson Shelf Channel, but there is no evidence of major accumulations (Freeland et al. 1976).

The Christiaensen Basin (landward extension of the Hudson Shelf Channel) contains fine-grained deposits mixed with coarser sands. Harris (1976) found temporal and spatial variations especially in the distributions of the fine-grained deposits and in the concentration ratios for certain metals, specifically Zn and Cr, which he believes can be used to determine the sources of these materials. Changes in the distributions of these deposits were most obvious near Long Island beaches and in the northernmost part of Christiaensen Basin. Near Long Island, mud patches were most extensive during late spring and summer; they were absent, covered by sands, or restricted to troughs between sand waves during early fall to early spring. Ages of organisms living in these muds indicated that individual patches existed within a few kilometers of the Long Island shore as early as 1972. In the Christiaensen Basin, mud accumulated there moved southward in some years and retreated in others. The metal concentration ratios suggest that some fine-grained muds came from sewage sludge deposits offshore and some from similar materials in western Hempstead Bay.

Hatcher and Keister (1976) analyzed organic matter in New York Bight sediments, which is derived from land-based sources, from sewage solids, or from local productivity of phytoplankton in response to the large nutrient inputs from local sewage treatment plants. They used the ratio of total carbohydrates (TCH) to total organic carbon (TOC) to study distributions of sewage-related organic matter.

	TCH:TOC
Terrestrial soils	20
Marine plankton	30–80
Sewage solids	30
Partially decomposed sewage solids	40–60

TCH:TOC values were around 40 in the sewage sludge disposal site and 50 or more in the axis of the Hudson Shelf Channel.

The New York Bight ecosystem

Malone (1976) reported on the effects of various nitrogen sources on plankton productivity in the New York Bight. Phytoplankton productivity ranged from a December low of about 0.1 gram of carbon per square meter per day (g C m^{-2} d^{-1}) to a June maximum of 6.4. Most of the dissolved inorganic nitrogen (nitrate+nitrite+ammonia) was consumed by phytoplankton within 20 km of the estuary mouth. Annual production by phytoplankton amounted to 370 g C m^{-2} yr^{-1} in the 600-km^2 area nearest the harbor entrance; such levels of phytoplankton production are typical of upwelling and productive areas of estuaries (Table 1). The supply of dissolved inorganic nitrogen from the estuary exceeded phytoplankton demands except during June, July, and August when estuarine discharge accounted for about 60% of the demand. Ap-

Table I. Primary productivity of major ocean areas, the New York Bight, and adjacent areas.

Ocean area	Mean productivity (g dry C m^{-2} yr^{-1})	References
Open ocean	50	Ryther 1969
Coastal ocean	100	Ryther 1969
Upwelling area	300	Ryther 1969
Peru	475	Guillén et al. 1973
New York Bight	100–160	Ryther and Yentsch 1958
New York Bight apex	370	Malone 1976
Long Island Sound	380	Riley 1956

parently phytoplankton productivity was light limited in the apex.

Walsh et al. (1976) examined behavior of lower trophic levels during the spring bloom (sudden increase in phytoplankton abundance) during late March–early April 1975. They reported a maximum in productivity and a minimum in grazing stress at midshelf locations. Diatoms, dominated by *Thalassiosira nordenskioldi*, were the most abundant. The number of cells per chain was greatest at midshelf locations and decreased at stations nearshore and near the shelf break, from which they concluded that grazing pressure was minimal at midshelf locations where productivity was highest. Increased grazing pressure may account for the shorter diatom chains in nearshore and shelf break stations. Mixing, caused by storms and breaking of internal waves, may be responsible for replenishing nutrients in surface waters of the continental shelf ecosystem in spring.

Litchfield et al. (1976) demonstrated substantial seasonal and spatial variability in microorganisms in sediment from Sandy Hook, Raritan Bay, and the Hudson Shelf Valley. A significant fraction of the microbial populations studied seemed to require organic nitrogen sources and even specific nitrogenous compounds. Such bacteria were found to depths of 45 cm in sediment cores. A large fraction of the bacteria was capable of utilizing urea as the sole nitrogen source.

A predictive model of the recovery of a perturbed benthic ecosystem was developed by Papadakis and Saila (1976). Required input parameters for the model were the immigration rate, death rate, and average biomass of organisms by species as well as an estimate of the environmental carrying capacity. Simulation runs with the model, using arbitrary parameters, predicted changes in biomass over time which resemble the limited empirical data available.

Rowe et al. (1976) analyzed coupling between benthic processes and those occurring in the surface waters. This is particularly important in shallow continental shelf waters where the waters are seasonally mixed, top to bottom, during the colder months of the year, and where there is substantial exchange between surface and subsurface waters owing to upwelling induced by winds and river discharges.

Nitrogen rather than phosphorus was found by Ryther and Dunstan (1971) to be the nutrient limiting phytoplankton production in coastal waters. Scarcity of nitrogen in the water seems to be related to its loss from the system during decomposition of organic matter on the bottom. Thus, in coastal systems nutrient regeneration and release from the sediment is a process of major importance to near-surface biological processes. Sediments in the New York Bight apparently oxidize about 20% of the daily input of sewage sludge. In a highly productive but unstressed coastal ocean system off Baja California, the daily oxygen consumption of the bottom could account for about half the input of organic matter (Rowe et al. 1976).

Edwards (1976) discussed data from groundfish surveys between Georges Bank and Cape Hatteras. These data show that about 22% of the available finfish resource was harvested during the years 1963–1965. In the period 1964 to 1967, standing crops decreased about 40% indicating that the fishery was probably being harvested at or near the maximum rate.

According to Franz (1976) the highest density of juvenile clams occurred closest inshore, within about 1 km of Long Island beaches. Abundance decreased rapidly offshore. Highest concentration of juveniles occurred in the western sector near the entrance to New York Harbor. Surf clam populations are dominated by relatively few year classes and most of the sites east of Fire Island Inlet are older than 10 years. Flourishing clam populations near New York Harbor are dominated by younger clams (5–7 years); few older clams are present.

Man's impact on the New York Bight ecosystem—Nitrogen in several forms is contributed to the bight by the estuarine discharge carrying nitrogen derived from upriver and harbor sources and by waste disposal operations, especially sewage sludge disposal. Malone (1976) has shown that ocean dumping could have provided a maximum of 10% of the nitrogen demand in summer and that disposal of sewage sludge and dredged wastes had no statistically significant effect on phytoplankton growth rates nor on the factors regulating the rates.

Thomas et al. (1976) reported on seabed oxygen consumption in the bight. Seabed oxygen consumption is a direct measure of the rate of biological and chemical activity of the sea bottom and is also a factor in determining the dissolved oxygen concentrations of near-bottom waters. Areas of high oxygen consumption were found in the Christiaensen Basin and upper Hudson Shelf Channel, and near Asbury Park, New Jersey, where there may have been some local release of sewage materials. Substantial seasonal variations were reported as well as year-to-year variations. Causes and magnitudes of the changes are not yet known. In August 1974, dissolved oxygen concentrations in near-bottom waters were as low as 1 ppm, well below the tolerance levels of many marine organisms. Oxygen consumption on the bottom seems to be directly controlled by the amount of organic matter that reaches the bottom where it is decomposed.

Pearce (1972) documented effects of waste disposal in the bight apex and the resulting changes in benthic populations in and near the disposal areas. He reported that two areas totaling about 50 km^2 were impoverished of the normal benthic fauna. Deposits in these areas had high metal and carbon contents, and a finer grain size, leading to significantly changed bottom characteristics. More detailed studies (Pearce et al. 1976) of quarterly surveys showed a reduction in the number of species and individuals in August 1974 relative to August 1973. Some species (such as the cancroid crab *Cancer irroratus*) were considerably reduced in number and distribution. But other species, such as the deposit-feeding bivalve *Tellina agilia,* remained essentially constant. Establishment of the natural variability of the area is essential to an understanding of the extent of changes due to environmental alteration.

Rosenfield (1976) reviewed disease problems of commercially important mullusc and crustacean food organisms. Oyster protozoan parasites have devastated oyster populations along the middle Atlantic coast, particularly in Chesapeake and Delaware Bays. Other infectious disease agents, including viruses, bacteria, protozoans, fungi, helminths, and crustacean parasites, cause varying amounts of pathological response in the organisms. Proliferative cell conditions in marine shellfish have been reported more frequently.

Some infestations are related to environmental modifications. For instance, expression of herpes-type inclusions in hemocytes may be enhanced in oysters exposed to thermal effluents. In the New York Bight, lobsters and crabs from sludge and dredged waste disposal sites had shell erosion and ulcers. Shellfish harvested from polluted waters are also known to contain hepatitis virus, *Shigella,* and *Salmonella.*

Waste deposits also affect microorganisms. Koditschek (1976) reported that about 1% of bacteria from deposits near the disposal sites were resistant to $HgCl_2$ (10^{-5} M) and/or tetracycline (40 mcg/m). Most isolates from these deposits showed multiple antibiotic resistance.

Not all of man's effects on the bight arise from waste disposal. Grosslein (1976) reported that since 1967 the standing crop of commercial finfish and squid has declined more than 50% due to intensive fishing in the region. Species exhibiting the largest declines include sea herring, flounders, hakes, and sea robins. These declines are attributed primarily to heavy fishing and especially to unselective bottom trawling in the winter and spring when many species congregate in the mid-Atlantic shelf region.

Grosslein further concluded that the environmental factors controlling natural fluctuation are still virtually unknown. Thus, the net effect of a degraded environment must be inferred from major population trends coupled with experimental data. For example, it will be necessary to understand food webs for major species and to know the sublethal or chronic effects of various pollutants to properly account for their effects.

Locating areas of minimum risk to finfish for new waste disposal sites or offshore petroleum production or transfer facilities is difficult because of wide overlaps in density distributions of most species. Grosslein concluded that it is virtually impossible to find a site in the bight where a significant aggregation of some major species does not occur at some time during the year.

Sindermann (1976) concluded that some of the region's estuaries and coastal areas are badly degraded—New York Harbor and the New York Bight are well known examples. There was little evidence of widespread damage to major fisheries resource populations resulting from pollution of coastal water, except for localized effects. Other factors such as repeated year-class failure (or success), shifts in population distributions, and overfishing may cause pronounced changes in fisheries.

Fin rot—progressive destruction of fin tissue—was evident in fish such as winter flounder, *Pseudopleuronectes americanus*, from the apex area. Fin rot occurred in about 14.1% of the specimens taken during surveys made in 1974 and 1975 as compared to 1.9% in fish caught in adjacent coastal waters (Murchelano and Ziskowski 1976). Fin rot was not restricted to the apex. Of the fish taken from Raritan Bay, 7.6% exhibited symptoms of fin rot as compared to 1.9% in fish taken from Great Bay, New Jersey, which is relatively uncontaminated. And even in the same small area, fin rot was found in 5.1% of the fish from areas underlain by carbon-rich deposits (apparently contamination by sewage deposits); 2.9% of fish taken from areas of low-carbon deposits had fin rot. Windowpane flounder, *Scophthalmus aquosus*, a fish with an abundant slime coating, rarely exhibited fin rot. Fin rot in winter flounder is apparently not a rapidly progressive disease which ultimately causes death. But it may modify behavior and render fish more vulnerable to predation or reduce their ability to capture prey.

Halogenated hydrocarbons and some common industrial metals (lead, silver, cadmium, chromium) are known chemical mutagens, and significant genetic damage can occur at subtoxic levels. Crosby Longwell (1976) studied chromosomes and mitoses during genetically sensitive and critical stages of the second half of meiosis, fertilization, cleavage, and embryo divisions to see if eggs (from a single population of mackerel) from the New York Bight showed any evidence of abnormalities. Of 19,000 chromosome and mitotic figures scored in 380 eggs collected in May 1974, a third showed abnormalities. In only a fifth of the eggs were all chromosome and divisions figures scored normal. The abnormalities showed the same sort of damage expected to follow irradiation. Comparable levels of abnormalities were observed in mackerel eggs from the periphery of the bight (13 to 16%). Samples from two stations in the toxic chemicals disposal site ("106 mile site") showed the highest level of abnormalities; it was the only location where a significant number of eggs were dead.

Public health

The earliest studies of the New York Bight arose from concerns about the potential public health risk involved in the increasing harvest of surf clams (*Spisula solidissima*) and the various wastes, especially sewage sludge, discharged by barges in the bight. These studies, begun in 1962, resulted in the closure to shellfish production of an area around the sludge disposal site. Verber (1976) discussed the growth of the sea clam industry and the closure of continental shelf areas off New York, Delaware Bay, and Boston Harbor, where sewage sludges or sewage treatment plant effluents are discharged to the coastal ocean in large quantities.

A prime objective of programs designed to protect people from exposure to waterborne diseases is to develop criteria, grounded in a data base, that relate illness, as measured by symptomatology, to some potential microbial or chemical indicator of water quality. Reliable data of this sort have been lacking for the United States.

Cabelli et al. (1976) reported on the effects of exposure of weekend swimmers to the relatively unpolluted waters at Rockaway Beach (Riis Park) and barely acceptable waters at Coney Island Beach near 22nd Street, both on Long Island's south shore. Among swimmers at the Coney Island Beach, gastrointestinal symptoms (vomiting, diarrhea, nausea, or stomachache) were significantly higher than among nonswimmers. At the Rockaway Beach there was no difference between swimmers and nonswimmers. Children and Latin Americans were most sensitive. *Escherichia coli* and enterococcus were found to be the most reliable indicators of water quality for this purpose.

Research needs

The symposium has documented the rapid increase in our knowledge of the New York Bight between 1970 and 1975. Distributions of sediment deposits and organisms have shown the extent of ocean bottom directly affected by long continued waste disposal operations in the bight. Uncertainty about sources of wastes and their composition and the lack of long time series of observations prevent definitive statements about natural, annual, and seasonal variability of organisms. Thus, assessments of human impacts on the bight and the marine life in it, unless they are nearly catastrophic, must be regarded as preliminary.

The studies reported in the symposium have shown that our knowledge about physical processes is also limited. For example, we do not know the circulation in the region well enough to permit us to make definitive budgets for many known pollutants. Nor do we know the effect on the circulation and renewal of near-bottom waters through the Hudson Channel.

Relatively high productivity of surface waters constitutes an oxygen demand to near-bottom waters. Some decomposition of unutilized organic matter occurs on the bottom, but apparently not enough to account for the low dissolved oxygen conditions in the bight during summer stratified conditions. Effects of disposal of dredged materials or sewage sludges in the Bight cannot easily be separated from the regional effects of wastes discharged in the estuarine outflows.

Among the points not covered in the symposium is airborne transport of materials to the bight. This is an important pathway for pollutants in southern California (S. Calif. Coastal Water Res. Project 1973) and doubtlessly is important in the New York region as well. Conversely, we might ask if any of the wastes dumped in the bight are concentrated at the air–sea interface and transported into the atmosphere, possibly reaching humans as aerosols taken into the lungs. Nor was there any study of marine birds, although they are severely affected by petroleum in the marine environment.

A major difficulty is our inability to distinguish waste products from similar metal-rich or carbon-rich materials brought to the bight by rivers or sewers, transported out of nearby marshes, or formed in the bight itself by phytoplankton.

Sources of the low-density, high-carbon deposits that occur seasonally offshore from Long Island remain unknown. Mud deposits are noted on the nautical charts dating from 1845. There is some evidence that their positions may vary seasonally but none that they are steadily moving onto nearby beaches in large quantities. There is no compelling evidence that these materials are principally derived from sewage solids dumped in the bight. Uncertainty about the possible movements and ultimate fate of the sewage sludges constitutes a problem for regulatory agencies at all levels of government in the region.

There are, however, immediate management issues that need to be addressed. Among the most important is the issue of utilizing the existing sewage sludge site or of relocating it farther out (about 100 km) on the continental shelf (Fig. 1).

Available data show that little contamination exists in the midshelf region of the bight and that the living marine resources there reflect a relatively pristine environment (NOAA 1976). Considering this, along with the widespread and overlapping ranges of organisms, one must question the advisability of contaminating a midshelf region with a new disposal area.

Public health aspects of ocean waste disposal clearly require more attention. For example, is there significant transfer of wastes or bacteria or viruses from the ocean surface back to land by winds? And is there any risk in recreational harvesting of shellfish from areas contaminated by waste discharge?

Finally there are unanswered questions about perceived environmental quality. What environmental conditions are unacceptable? Which are tolerated? Here we are probably dealing with surface films, discolored or turbid waters, odors, and fish kills.

The significant point is that most of these questions are not routinely included in environmental studies. They are thought to have little impact on marine ecosystems (aside from fish kills). But they may have a great impact on continued acceptance of waste disposal in coastal waters.

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The old and the new: New York Bight research in perspective

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Grant Gross invited me to comment on the papers presented in this symposium as one of the surviving members of a very small research team who carried out the first intensive oceanographic investigation, about a quarter of a century ago, of the area now called the New York Bight apex (Redfield and Walford 1951; Ketchum et al. 1951). He asked me to evaluate what has been accomplished in recent years and to comment on what needs to be done in the future. Although I had copies of the abstracts in advance, these are generally less informative than the actual presentation, so that most of my conclusions have been reached during the two and a half days of this symposium.

Let me first draw a few contrasts. About six of us actually participated at one time or another in our New York Bight studies. If we had planned a symposium 20 years ago when we first had some results, we might have been able to convene a single morning session of half a dozen papers, and we might have attracted an audience of about 10 or 20 people including the six participants. This is an obvious contrast with the more than 500 who have signed up in advance for this symposium and many others who have joined since without advance registration. Clearly interest in pollution problems in general, and in this area in particular, has expanded greatly in the interim.

Before our investigations, the only oceanographic information about this part of the continental shelf was based on Bigelow's observations of salinity and temperature in the 1920s and 1930s (Bigelow 1933; Bigelow and Sears 1935). The stations were widely spaced, and, although there was some seasonal coverage, the details that are only now beginning to appear were lacking. Looking at a system which had not been seriously evaluated before frankly gave us a considerable advantage.

Cost of research provides another contrast and the impact of inflation is obvious. Our program, including four seasonal surveys, was conducted under the auspices of a committee of the National Research Council, funded at a level of about $90,000 provided by the National Lead Company (now National Industries). Their interest concerned the effects of the disposal of the acid-iron wastes of titanium dioxide production, and we had fine cooperation from the group at National Lead throughout our studies which included periods before, during, and after their barging operation. On the first morning of this symposium we heard presentations of the studies in this area now being conducted by NOAA, by EPA, by ERDA, and by New York City and State. I don't know how far $90,000 would go in support of this program but I suppose it might provide for a few days of the present effort.

In listening to these papers, I have been touched with a bit of nostalgia, with a good bit of gratification and with some pleasure. The nostalgia, of course, comes because the various distributions of salinity and temperature across the continental shelf all look very familiar. Nature has not changed in 25 years, and this stretch of the continental shelf looks pretty much the same today as it did then. The gratification and pleasure come from the fact that some of the conclusions that we drew have been verified several times in the talks that have been presented. Beardsley et al. (1976) gave flushing times for this stretch of the continental shelf that are about the same as those calculated by Ketchum and Keen (1953). Their approach, however, gives promise of elucidating the mechanisms that control exchanges between the shelf water and the open sea.

There has also been a bit of disappointment mixed in with these pleasant reactions. A lot more data and information have been accumulated, but there do not seem to be many new ideas, and there are still many unanswered questions. This is now an intensive program, and I appreciate that it is only about 3 or 4 years old. There have been some very fine presentations, and the new data available today should help us understand the system in ways that were not possible before.

Many studies can be conducted significantly better today than was possible 25 years ago, giving a modern investigator some obvious advantages. For example, a modern, continuous-recording STD can be used to observe accurately the transition between the surface coastal water and the deeper, saltier water, which may occur in only a foot or so of depth. It is virtually impossible to sample within that transition zone using, as we were, Nansen bottles. On a T/S diagram of discrete samples, the data for all of the shelf water samples group together in the warmer, fresher area and are widely separated from the colder, higher salinity deep water by a large gap for which we are without information. Only about a dozen observations of the transition zone appear in hundreds of stations made with Nansen bottles in the New York Bight apex and throughout this coastal area. However several observations in the transition zone at each station would be desirable. The STD makes this possible. This instrument was invented before our studies (Ford 1949), and I worked with it in 1945. At that time, it was a cumbersome, awkward, and undependable gadget. There is simply no comparison between the old and the new STD equipment and measurement.

Continuous current measurements are also much more practical today. The idea of measuring currents directly is, of course, not new; Marmer's (1935) study of the currents in New York Harbor and its approaches is still outstanding, based on observations made in the 1920s and 1930s. But as Beardsley et al. (1976) pointed out, continuous long term current measurements are needed to evaluate the low frequency events which may have a very important effect on the conditions in a coastal area such as this. Swift et al. (1976) pointed out that these special, low frequency, high intensity events do more to affect the geology of the system in a day or two than the less dynamic conditions throughout the entire balance of the year.

Aerial and satellite surveys with direct photography, infrared imagery, and derived surface temperature distributions are also available today. This permits a synoptic picture which was absolutely impossible with a plodding research vessel like the *Balanus*. Surface fronts and even the plume of waste discharged from a moving barge have been clearly shown in some of these photographs. I would think that much more of value could be obtained from this source of data.

The opportunity for direct underwater observation by SCUBA diving and submersibles is another type of observation that is in an entirely different category today than 25 years ago. Many of the oceanographers who have spoken here in the last few days are trained SCUBA divers, and we now have submersibles that can take the scientist down to look at the bottom. For many problems, direct observation is unsurpassed.

Analytical techniques for detecting and measuring pollutants are also vastly improved—orders of magnitude better than anything that could conceivably have been done 25 years ago, or even 10 years ago for that matter. The development of modern chemical methods to measure heavy metals, petroleum, and chlorinated hydrocarbons makes it possible to evaluate these pollutants in ways that could not be done during our earlier studies. As a matter of fact we were not much worried about chlorinated or petroleum hydrocarbons at that time. Those were the early days of the use of DDT and the threat to the environment was yet to be recognized, and the rate of petroleum hydrocarbon pollution of the sea has increased exponentially since then. Oil was obvious in New York Harbor then, but not too far off the coastline the area appeared to be in pretty good shape. During the second world war, of course, many tankers were sunk along the coast and a lot of beaches were contaminated by oil, but it didn't seem to be a terrific problem for the open sea at that time. Today, I think it is. Farrington (1974) for example has shown that the petroleum hydrocarbon content of the sediments in the dumping area of the apex is three orders of magnitude greater than that in uncontaminated sediments nearby.

Our Chairman, Grant Gross, and Jack Pearce have both established the heavy metal contamination of sediments in the sludge and dredge spoil disposal areas (Gross 1970; Pearce 1969). Once when talking about the problems of heavy metal contamination of the New York dump site based on data that Jack Pearce had obtained, I used the phrase "heavily contaminated sediments." An engineer who was on the same program said these weren't strongly contaminated and not much of a problem. When asked to justify my use of the words "heavily contaminated," the only way that I could think to respond was, "If I were looking for an ore for the recovery of cadmium or lead, I would not go to the dump site for it; I'd go someplace else. But for the marine organisms that have to live in contact with these sediments, they are heavily contaminated." Pesch (1975) found various heavy metals concentrated in the meats of shellfish in and adjacent to the sewage sludge disposal site off the Delaware estuary, and this evidence confirmed EPA's edict that the city of Philadelphia should phase out ocean disposal of sewage sludge by 1981.

In terms of the biological studies that can be conducted now, there have been improvements in some areas, but not all. Certainly, direct measurement of primary productivity by the ^{14}C method (Malone 1976) has been a tremendous advance that has taken place in the past 25 years. It makes it possible to tie-in a direct instantaneous measurement of productivity with all of the nutrient cycling that is going on in the area. Lang (1975) made the point that his water is only "slightly used," but it's obvious to all, and I am sure that he realizes, that his "slightly used water" is greatly enriched with nutrients.

However, it was certainly gratifying to hear from two outstanding fisheries biologists (Sindermann 1976; Edwards 1976) that little or no effect of waste disposal on fish abundance could be determined. There have been clear changes in the abundance of some species, but this was attributed mainly to fishing intensity rather than to pollution, even though extensive dumping of sewage sludge and dredge spoils has taken place for 50 years or more. McHugh (1972) traced the general shift in species composition and decline of the fisheries of New York State over nearly the last century. He concluded that the reasons for some of the changes are known, but other trends remain unexplained. Establishing cause-and-effect relationships between pollution and a change in a natural population is, at best, a difficult task. However the results presented by Murchelano and Ziskowski (1976) suggest that fin rot disease of summer flounder is directly related to sewage sludge disposal in the New York Bight apex. Their data show that the incidence of fin rot is higher in the apex than in adjacent areas, but also that it may be even higher at times in Raritan Bay. The causative agent for the infection needs to be identified so that control measures can be initiated. If the dump site is the locus of the infection, which could then spread to adjacent areas, this would be a clear cause for concern.

What should be done in future studies of the New York Bight? Lang (1975) and Dewling et al. (1975) spelled it out loud and clear on the first morning of this symposium. They need the scientific basis to permit logical decisions about various engineering developments. Lang, for example, asked, "What do we buy with another billion dollar investment in additional sewage treatment plants?" Dewling raised the questions "Should the dumping in the New York Bight cease in 1981? Should it be moved now or left where it is? Should it be moved later?" The oxygen data presented by Segar and Berberian (1976) show that dumping sewage sludge and dredge spoils in the apex is not the major cause of oxygen depletion. Photosynthetically produced organic carbon and that in the river water contribute greater amounts than dumping. Oxygen of course is not the only basis on which to justify a decision, but excessive depletion is a clear indication of ecological damage. We also have to know the content of heavy metals, petroleum hydrocarbons, and chlorinated hydrocarbons; and we need to know the effects that these have on the marine populations of the area.

I do not think that we have yet learned how to proceed in evaluating effects and relating them to the cause. One of the problems in all pollution studies is that it is scientifically impossible to prove the negative. The scientist is asked to "prove to me that this does no harm." All that can be proven is that under certain conditions, no damage could be found or, possibly, that a known pollution loading does do damage. You can prove the positive effect if one is found, but there is no way that you can prove the negative. Thus, we need to know specific limits for specific materials which will do detectable damage to the ecosystem, and we must be able to tell the engineer that damage will result if these limits are exceeded. So far, this can be done in very, very few cases. Identifying these limits should be the main objectives of EPA-sponsored research and one of the objectives of the research of any scientist who is involved in such an investigation.

What else do we need to know? We need a useful, predictive model. I was looking forward to McLaughlin's presentation of the conceptual model earlier this morning. I'm sorry to say, he's left me looking forward to reading about it when he presents it in more detail (McLaughlin and Elder 1976). There's a trap in modeling, and I trust that this group is not falling into it. Modeling, of course, is the "in thing" today. If you aren't making a model of some sort, you're not in the running. Modelers sometimes assume that if they can draw enough boxes and enough arrows on a piece of paper, they suddenly know something about the system. Actually, it works the other way around. The boxes and arrows are nonsense until something is known about the system and then you can start to build a model based on what you know. If one builds an entirely conceptual model (obviously nobody does; whatever information is available is used), one can delude oneself and others that something is known about the system.

Some ecologists fell into a similar trap many years ago. Probably very few present here today remember when ecologists simply *had* to coin new words. They coined a new word for each phenomenon they could see, and some thought that when they had coined a name they understood something about it. That wasn't true. Some of these words are in our common language today; they were useful and did define a concept that we now understand something about. Others have been completely forgotten.

What can a model do? It can help define the gaps of knowledge where we need more precise information. If it fails in doing this, then it fails in the first purpose of a model Even more importantly, once enough is known of the forcing factors, of the flow factors, of the control factors, then a prediction can be made about the effects of a perturbation of some part of the system on some other part of the system. There may never be a complete model of the New York Bight, including the circulation, the geology, the biology, the ecology, the benthic biology, the phytoplankton, the zooplankton and all of their interactions. But that doesn't say that a model can't be useful; it doesn't have to be complete, but it does have to be realistic. Approximations or estimates can be used for some of the "unknowns" and the results checked against reality.

What other things do we need in the long range? I've already mentioned the need for long term current meter observations set out in arrays so that one can get an idea of how consistent circulation patterns are from place to place. The time span of these observations has to be multiples of the time frequency of the events investigated. In other words, it is satisfactory to have a month's observation for events of a tidal frequency, but this is useless for seasonal events. For these, the time series must extend for years. Beardsley pointed this out by comparing observations at station D, where we do have years of experience with the short time period current meter records on the continental shelf (Beardsley et al. 1976). We simply know nothing about the low frequency end of the power spectrum for the shelf area, and these infrequent events may be of great importance.

We also need studies of the recovery of an ecosystem after a stress has been removed. One possible reason for moving sewage sludge dumping to a new site would be to study the sequence recovery might follow and how long would be required, if, indeed, recovery should take place. Segar and Berberian (1976) evaluated the oxygen depletion of New York Bight water below the thermocline and concluded that the major part of the demand results from the photosynthetic production of organic matter in the area. For the summer period this would account for more than 95% of the oxygen depletion. The high productivity in the surface layers is, in turn, attributed to the nutrient enrichment, especially of available nitrogen, resulting from pollution carried primarily in the river flow. They further concluded that removal of the sludge and dredge spoil dumps from the apex would have little effect on the oxygen deficiency, but warn that increases in nitrogenous enrichment of the estuary and river could lead to disastrous anoxic conditions. Clearly any studies of recovery must include all parts of the ecosystem and cannot be limited to the removal of one of the multiple stresses that have been imposed.

It should also be mentioned that estimates by Segar and Berberian (1976) of primary production in the area are nearly an order of magnitude greater than those presented by Malone (1976). However, Segar and Berberian were integrating over an area four times as great as that used by Malone, so that it is not clear whether this is a real quantitative difference or one dictated by the spatial boundaries of the area considered. Malone (1976) also concluded that the supply of available nitrogen was essential for high productivity and that polluted river flow is the main source of this extra nitrogen. Similar conclusions about the supply of nitrogen were reached by Mueller et al. (1976) who pointed out, however, that the relative importance of different sources was not the same when considering other pollutants.

In conclusion, while it is gratifying that some of our earlier observations have been confirmed, it is even more gratifying to find that much new information is now available. The newer methods that I have mentioned are being used and are producing results that could not have been obtained during the course of our earlier investigations. It is also clear that we are approaching an understanding of some of the mechanisms of the delivery and distribution of various pollutants within the area. It seems apparent that the ecological conditions in the New York Bight apex have deteriorated considerably over the last 25 years. We are beginning to understand what must be done to improve conditions, and evidence is accumulating that will make it possible to evaluate the various options for corrective action. Hopefully, the essential information is arriving in sufficient time to direct appropriate action.

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Section 2 Physical processes

Physical oceanography of the Middle Atlantic Bight[1,2]

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Abstract

Kinetic energy spectra from moored current meters in the mid-Atlantic Bight reveal marked differences in current variability between the inner shelf and the outer shelf and slope regions. The nearshore subtidal current variability appears to be dominated by meteorological forcing. The amplitude of the semidiurnal and diurnal tidal peaks decreases in the offshore direction. Shallow water records show little or no inertial energy, while at the shelf break and over the slope, inertial motion contributes significantly to the current variance. A simple conceptual model is presented to explain how intense winter low pressure systems ("northeasters") drive strong alongshore currents which are coherent over much of the bight. A map of "mean" currents measured in recent moored array experiments demonstrates subsurface water flow along the shore toward the southwest. The average currents generally increase in magnitude offshore and decrease with closeness to bottom. At most sites, the mean current veers toward shore with increasing depth. The alongshore volume transport measured at three transects across the bight shows surprising uniformity, considering the possible sources for discrepancy. This transport (order $2.0 \times 10^5 m^3 s^{-1}$) of water within the 100-m isobath implies a mean residence time of the order ¾ year. Much of the shelf water observed flowing westward south of New England must originate in the Gulf of Maine–Georges Bank area.

Before 1970, information on the circulation of the mid-Atlantic Bight came mostly from temperature and salinity measurements and from drift bottles and seabed drifters. Bigelow (1933) and Bigelow and Sears (1935) first described seasonal temperature and salinity changes on the continental shelf, where vernal warming and freshwater runoff build a strong stratification which is subsequently destroyed in the fall by storms and cooling. Iselin (1939, 1955) postulated an offshore motion in the

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upper layers of the shelf water and corresponding shoreward flow in the lower layers because salinity generally increases with depth. He also noted that the circulation obeys the "rule of coastal circulation," whereby the average flow is parallel to the coast with land on the right-hand side of an observer facing downstream. Bumpus (1973) in his summary of a 10-year program of drift-bottle and seabed drifter releases and occasional drogue and drift-pole measurements, concluded that a mean alongshore flow of order 5 cm s^{-1} occurs from Cape Cod to Cape Hatteras, except during periods of strong southerly winds and low runoff (Bumpus 1969). Nantucket Shoals and Diamond Shoals appear to be oceanographic "barriers" which limit the alongshore flow. At the Cape Hatteras end, the alongshore flow turns seaward and becomes entrained in the Gulf Stream. Occasionally strong northeast winds drive a small amount of mid-Atlantic Bight water southward around Cape Hatteras (Bumpus and Pierce 1955).

The transition zone between shelf water and warmer, saltier slope water often occurs during winter as a sharp inclined front located near the shelf break. In summer, the front is less distinct but large temperature and salinity gradients still occur in the offshore direction below the seasonal thermocline. These gradients are due to a band of cold, low-salinity water located near the bottom on the outer shelf. Described by Bigelow (1933) as a remnant from the previous winter's cooling, these waters can have temperatures of 6°–8°C in August. Temperatures are around 16°C only 20 km offshore of the "cold pool." The mechanisms governing the movement of this frontal zone and exchanges of heat, salt, water, and momentum across it are not well understood.

Current variability, circulation, and water structure

Self-contained current meters, temperature and pressure gauges, and other instruments deployed in moored arrays in the mid-Atlantic Bight are now beginning to provide records of sufficient length to characterize the variability of the subsurface current field in this region. We will present here some preliminary results of these field programs with an emphasis on describing the "mean" circulation and subtidal current variability.

We begin by examining in Fig. 1 several kinetic energy spectra computed from 1-month or longer current records obtained at several different sites on the middle Atlantic continental shelf and rise. The four sites, labeled "A" through "D," and the location, water depth, depth at which the current record was taken, and other pertinent information for each site are given in Table 1. (Sites A through D correspond respectively to stations 18, 1, 4, and 11 shown in Fig. 2.)

The site A data has been taken and reported by EG&G (1975) under contract to the Public Service Electric and Gas Company of New Jersey. Flagg et al. (1976) obtained the site B and C data. The low-frequency cutoff for each estimated spectrum is inversely proportional to the length of the particular current record analyzed. The Woods Hole Oceanographic Institution has maintained moored arrays at site D for almost a decade and the very long current records obtained there allowed Webster (1969) and Thompson (1971) to make a reliable estimate of the kinetic energy spec-

Table 1. Location and other pertinent information for the current and wind spectra shown in Fig. 1.

Site	Sta. No. (Fig. 2)	Location	Time	Water depth (m)	Instr. depth (m)	Data source
A	18	39°28N, 75°15W	Dec 73–Feb 74	12	5	EG&G (1975)
B	1	40°54N, 71°04W	Mar 74	58	28	Flagg et al. (1976)
C	4	40°18N, 75°51W	Mar 74	112	30	Flagg et al. (1976)
D	11	39°20N, 70°00W	Several years	2,540	100	Webster (1969); Thompson (1971)

trum in the slope water over a seven-decade range frequency. The power density of the wind stress observed at site A is also shown in Fig. 1. Wind stress has been computed using the quadratic drag law $\tau = C_D|W_{10}|W_{10}$ where W_{10} is the observed wind vector at 10-m height and the assumed constant drag coefficient is $C_D = 3.2 \times 10^{-7}$ in c.g.s. units.

The spectra have been visually smoothed within the estimated uncertainties to simplify graphical presentation and our spec-

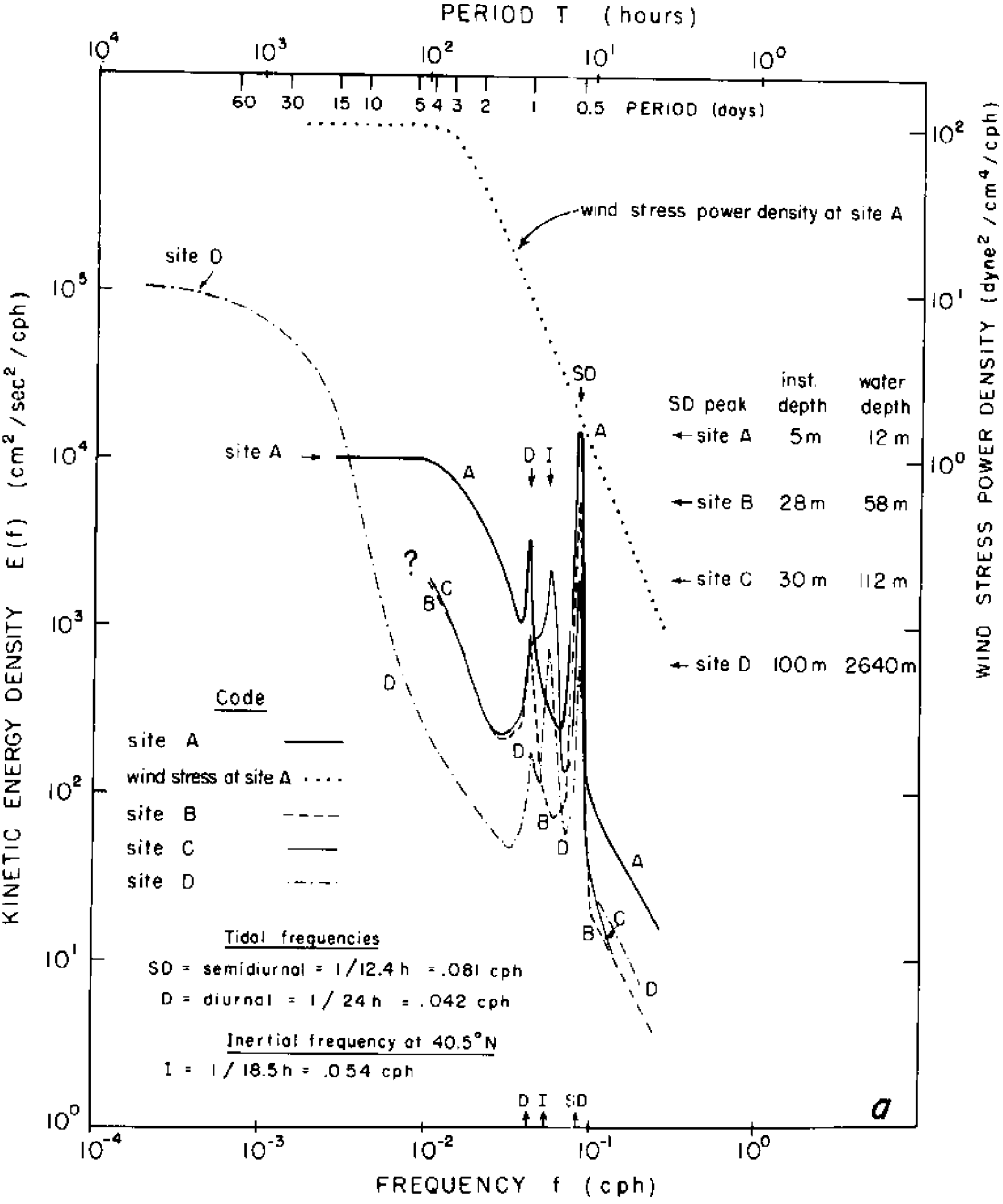

Fig. 1. Spectra of currents and wind in the Middle Atlantic Bight. Locations of current meter moorings are listed in Table 1. (*See text for explanation of different formats.*)

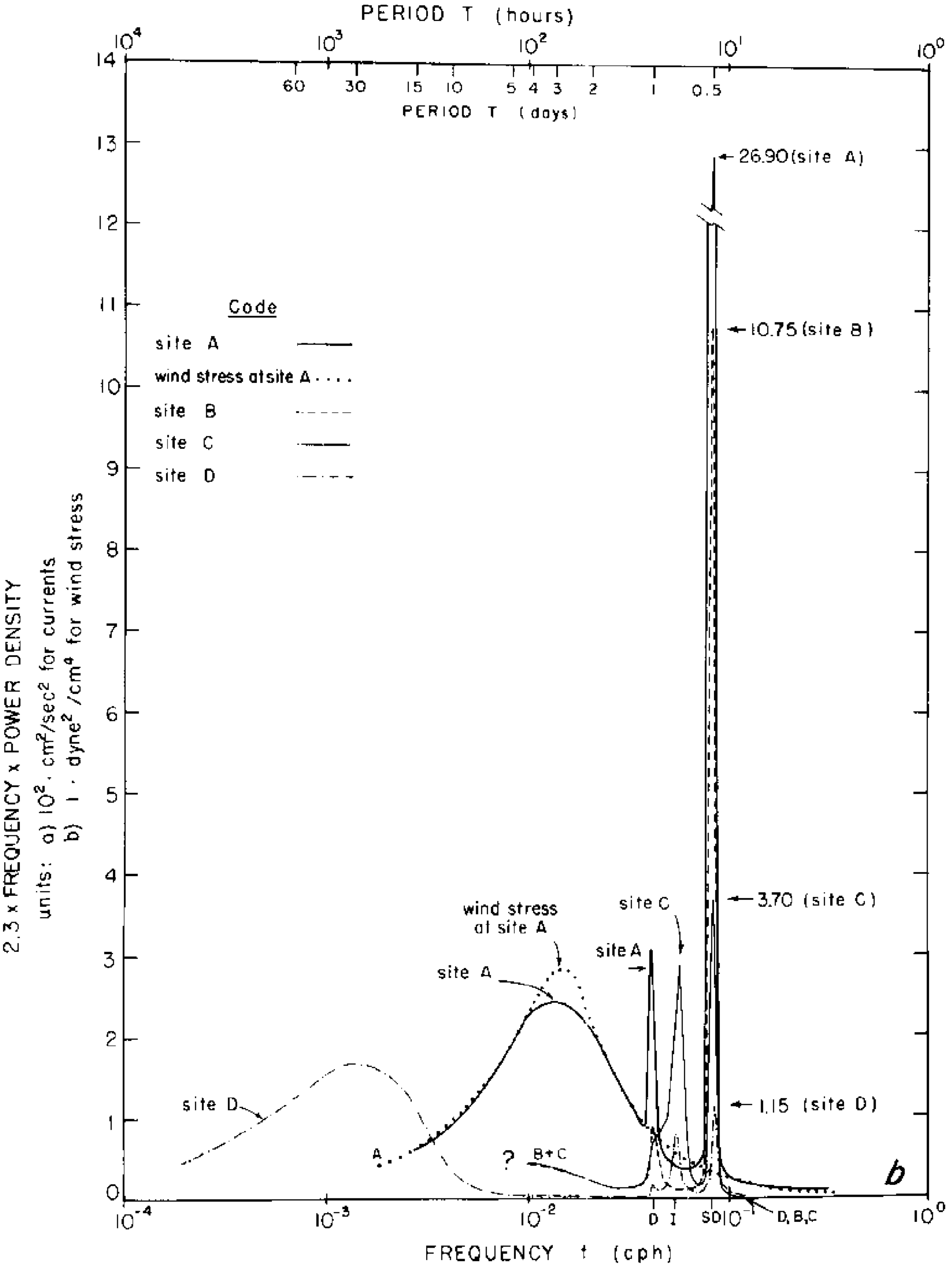

Fig. 1. Continued

tral characterization of wind and current variability over the continental shelf. The reader should remember that spectra obtained from *much longer* records will presumably show more structure than the smoothed *estimates* shown in Fig. 1. The spectra have been plotted in both the log $E(f)$ versus log f format (Fig. 1a) and the area-preserving linear $2.3 \times f \times E(f)$ versus log f format (Fig. 1b). The first format best displays the functional form of the energy density E as a function of frequency, e.g. $E(f) \propto 1/f^m$ corresponds to a straight line plotted in Fig. 1a with a slope of $-m$. The second format in Fig. 1b is used to illustrate how much different frequency bands contribute to the total variance of the current record. The total area under the $2.3 \times f \times E(f)$ curve is equal to the variance, and the area under the curve between two specific frequencies is the contribution from that frequency range to the variance.

The spectra shown in Fig. 1 illustrate several fundamental features of wind and current variability on and near the mid-Atlantic continental shelf. Wind stress and current spectra are inherently "red," with the power or kinetic energy density generally decreasing with increasing frequency. Wind stress power density at site A is approximately constant at lower frequencies with a transition occurring at periods between 2 and 4 days and a higher frequency falloff of about $f^{-5/2}$.

Most of the fluctuation in the wind stress at site A is caused by rather wideband meteorological transients which have characteristic periods between 1 and 8 days. The intense low pressure disturbances or cyclones which generally form over the southeast United States and intensify while propagating up along the eastern seaboard have characteristic periods of 2 to 4 days and cause the peak in the site A wind stress power density curve shown in Fig. 1b.

In addition to being red at lower frequencies, the four current spectra exhibit relatively sharp peaks at the semidiurnal (SD) and diurnal (D) frequencies. Amplitude of the semidiurnal peaks generally increases across the shelf toward shallower water; at sites B and C, the kinetic energy density at the semidiurnal frequency crudely follows the relationship $E \propto h^{-3/2}$ as predicted by shallow-water wave theory. The large semidiurnal peak observed at site A is probably caused by the proximity of Little Egg Inlet which can channel and intensify local tidal currents. Semidiurnal and diurnal tidal currents are weakest at site D on the continental rise. While the kinetic energy density at the diurnal frequency shows a general increase with decreasing depth across the shelf, the spatial structure of the diurnal tidal currents is not yet understood.

It is important here to note, however, that semidiurnal and diurnal tidal currents on the continental shelf are in part predictable since the astronomical forcing is deterministic and periodic. The accuracy of this prediction depends on the basic accuracy of the initial calibration of local tidal currents with the astronomical forcing, the degree of local nonlinearity (e.g. the phase shifting of the surface tide by strong storms), and the relative importance of baroclinic or "internal" tides, i.e. internal waves of tidal frequency. We expect baroclinic effects to be important perhaps all the time in the deeper water near the shelf break and over most of the shelf during the warmer months when a strong seasonal pycnocline has formed. Wunsch and Hendry (1972) observed bottom-intensified semidiurnal tidal currents in about 850 m of water on the New England continental slope. They described these observations as a train of internal waves of semidiurnal frequency generated at greater depth on the slope and propagating up the slope toward the shelf. How far these internal tides penetrate onto the shelf and how much mixing is caused by their dissipation is as yet unknown.

Tidal flow over topographic features can also generate higher frequency internal waves via nonlinear mechanisms. For example, trains of large-amplitude internal waves have been observed by remote sensing to propagate almost across the shelf during summer stratified conditions. Apel et al. (1975) believed such wave trains are formed near the shelf break by diurnal and semidiurnal tidal currents. The question of how much energy is really drained from the barotropic tides via topographic generation of internal waves remains unanswered.

Current spectra at sites C and D show an additional kinetic energy density peak near the local inertial frequency. The contribution of the near-inertial frequency band to the current variance is considerable at these two sites and especially so at site C near the shelf break (Fig. 1). The local generation of near-inertial currents by meteorological transients has been well documented at site D by Pollard and Millard (1970); fast moving fronts or strong veering winds which rotate clockwise with near-inertial frequency clearly excite nearly vertically propagating internal waves. The *absence* of near-inertial peaks in the kinetic energy spectra at sites A and B nearer shore is probably due to the existence of other "natural" modes like edge and shelf waves (*see* Reid 1958) which are preferentially excited during any transient adjustment period. The observed lack of strong near-inertial energy in shallow nearshore water should simplify the local current prediction problem.

We now turn to the lower frequency end of the current spectra. Long records at site D show that much of the current variance *at 100 m* in slope water is caused by low frequency motion with characteristic periods centered at about 30 days. Propagation of topographic Rossby waves up the continental rise (perhaps generated by the Gulf Stream), meandering of the Gulf Stream itself, and formation of anticyclonic (warm core) eddies can all generate strong low frequency currents at site D which cause the spectral shape shown. In contrast with site D, kinetic density spectrum at site A (Fig. 1b) shows that subtidal frequency currents in nearshore *shallow water* are strongly *wind driven* and cause most of the total current variance (*see* EG&G 1975). We thus suggest that the current prediction problem in shallow nearshore water further simplifies to the development of a model which relates the subtidal current to measurable meteorological forcings.

Current records obtained at sites B and C are too short for the computed spectra to indicate locations and magnitudes of lower frequency peaks; hence the large question mark shown in Fig. 1. It is *not* known at this time how much of the lower frequency end of the spectra is caused by local or regional meteorological forcing or by the transmission (or leakage) of lower frequency energy onto the continental shelf from the deeper ocean. We have used the model of Niiler and Kroll (in prep.) to estimate the possible transmission of topographic Rossby wave energy from the rise onto the shelf and find that this flux of energy across the shelf break is comparable with the direct kinetic energy input due to a surface wind stress of 1 dyne/cm^2 acting over the width of the continental shelf. Based on this and other preliminary observations, we suggest that the open ocean causes energetic low frequency motion on the outer continental shelf of the mid-Atlantic Bight. Longer current records (8 months or longer) are needed to quantify accurately the importance of low frequency energy transmission onto the shelf.

Having shown that much of the current variability in the shallower section of the bight is directly wind driven, we now describe a simple conceptual model for the dynamics of the response of this region to strong wind events. This model, suggested by Beardsley and Butman (1974), has been supported by other observations (Boicourt and Hacker 1976; Beardsley et al. in prep.). Intense winter lows, the "northeasters" which pass to the east of the mid-Atlantic Bight, produce strong wind stress fields toward the south and west over the shelf, generally paralleling the coast from Cape Cod to Cape Hatteras. The transient mass flux in the surface Ekman layer has a component to the right of the wind stress vector and a component parallel to the wind stress. During northeasters, the Ekman component directed to the right of the wind stress is onshore, causing sea level to rise along the coast. Wunsch (1972) and Brown et al. (1975) have shown that sea level over the deep ocean (and presumably the outer slope) is nearly constant over time scales of several days, so that the coastal rise in sea level creates a large onshore pressure gradient that is roughly in geostrophic balance with the strong alongshore flow. Since the wind stress field tends to parallel the coastline, the intense northeaster generates strong alongshore currents and cross-shelf pressure gradients which appear to be coherent over the entire shelf from Cape Cod to Cape Hatteras. Boicourt and Hacker (1976) observed that the more energetic subtidal current fluctuations (especially those associated with northeasters) oriented along the 35-m isobath off Maryland and Delaware are coherent and approximately in phase over distances of 230 km. They report typical maximum daily mean speeds of 40 cm/s at depths of 10 and 20 m, which can produce alongshore fluid particle excursions of 40–80 km during the several days of the storm. Beardsley et al. (in prep.) found that subsurface pressure gradients caused by sea level changes are coherent over the mid-Atlantic shelf from Cape May to Cape Cod. These observations suggest that the wind-driven component of the alongshore flow may be predicted from the more easily measured wind-stress and pressure fields and coastal sea level fluctuations.

We will now focus on the "mean" or very low frequency current field on the mid-Atlantic Bight. We have plotted in Fig. 2 the average currents which have been measured in recent moored array experiments. Only records of 1 month or longer duration have been used and information on the individual measurements (e.g. local water depth, instrument depth, time of measurement, current values, source of data, etc.) is given in Table 2. The mean currents are plotted as vectors with the magnitude equal to the average speed. The same current meter stations are numbered in Fig. 2 sequentially starting from the north and the same key is used in Table 2. The depth (in meters) of an individual measurement is indicated in Fig. 2 by a small number located near the head of the current vector. We have separated the measurements into winter (unstratified) measurements (denoted by solid vectors) and summer (stratified) measurements (dashed vectors). Measurements from several sites (5–11) on the continental rise and outer slope are included to show the mean westward flow of slope water. The mean position of the northern edge of the Gulf Stream is also shown, with the reminder that the actual position of the Gulf Stream in this region is highly variable (Hansen 1970).

These direct measurements of the mean current field on the shelf *demonstrate* subsurface water flow along the shore toward the southwest. The mean currents generally increase in magnitude offshore and decrease with closeness to the bottom. At most sites, the mean current veers toward shore with increasing depth. With the exception of station 21, a net southwestward transport is observed at all sites.

Measurements made along the three transects labeled I (New England), II (New York), and III (Norfolk) in Fig. 2 have been used to *estimate* mean alongshore volume transport. The transects cover the bulk of the continental shelf out to the 100-m isobath. Calculated transport values, cross-sectional area, and mean speeds for each transect are listed in Table 3. Although Wright and Parker (1976) estimated that roughly half of the volume of the shelf water from Cape Cod to Cape Hatteras lies in a thin surface wedge outside the 100-m isobath, there are essentially no direct measurements of mean current in the shelf water wedge beyond the 100-m isobath.

The estimated volume transports for the three transects are surprisingly consistent, considering that the northern transects (I and II) are early spring measurements in two different years, while the southern transect (III) value represents summer measurements. In addition, the transects were made at different depths, with different instruments, and with varying spatial resolutions. For these reasons, we hesitate to speculate about exchange of shelf water and slope water based on continuity arguments and assumed stationary flow through the transects. We are uncertain, for example, whether the higher mean alongshore speed shown in transect III is due to a continuity of transport within the 100-m isobath which forces the mean speed to increase through the smaller cross-sectional area, or whether it is due to a more consistent southward flow in summertime (for which there is some evidence). The consistency of the transports lead us to speculate that there may be little significant seasonal change in alongshore transport. Only

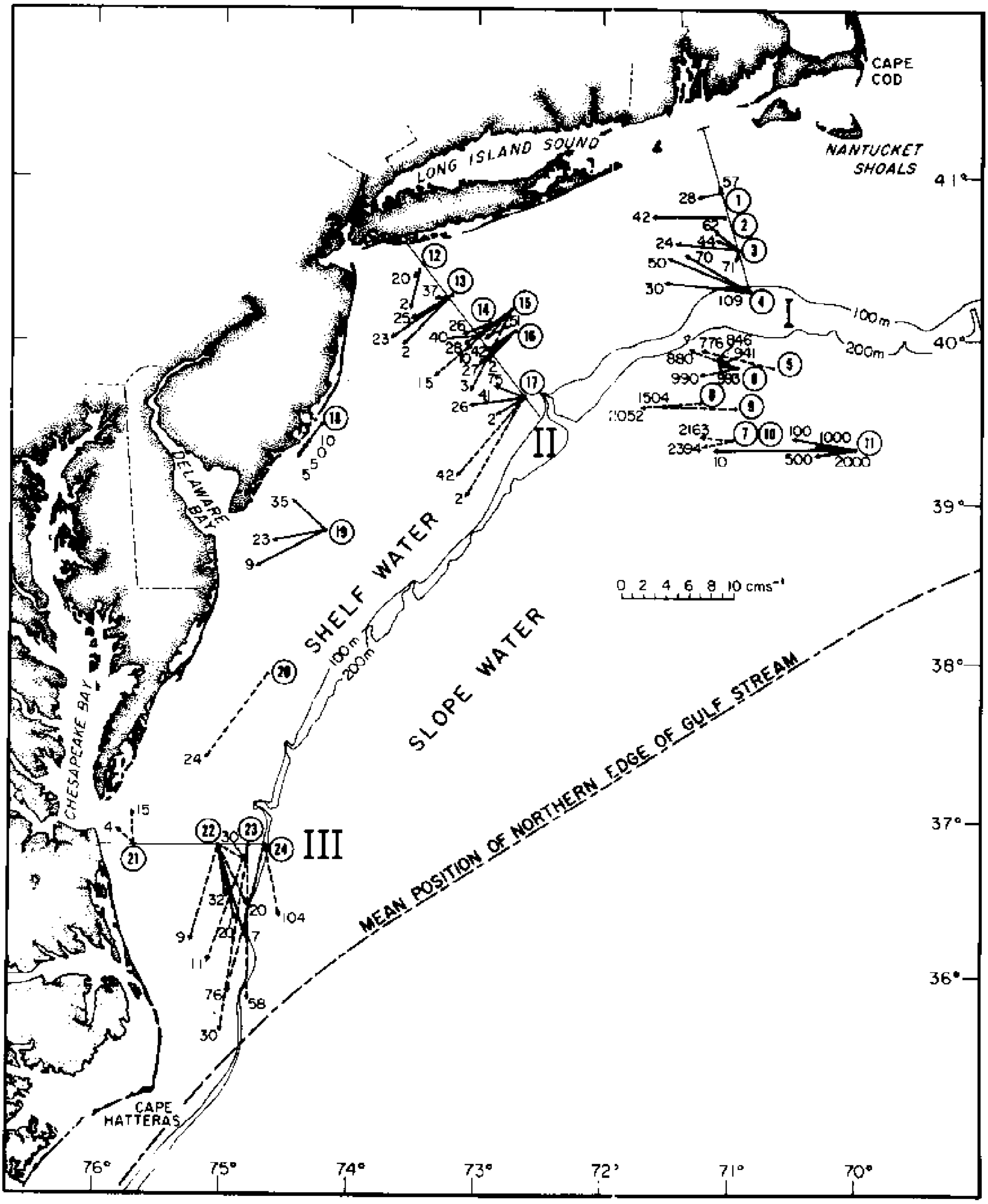

Fig. 2. Mean velocities as measured by moored current meters in the Middle Atlantic Bight region. Winter measurements are indicated by solid arrows, summer velocities by dashed arrows. Individual stations are numbered according to Table 2; station numbers are circled. Measurement depths (in meters) are shown near the head of the arrows.

Table 2. Tabulation of the recent direct measurements of sub- and near-surface mean currents shown in Fig. 2. (NA = not applicable.)

Sta. No.	Location	Start time	Record length (days)	Water depth (m)	Instr. depth (m)	E (cm/s)	N (cm/s)	Data source*
1	40°54N,71°04W	28 Feb 74	35	58	28	−2.1	−0.5	A
					57	−0.2	0.8	
3	40°33N,70°56W	28 Feb 74	35	72	24	−5.7	0.5	A
					44	−2.2	1.0	
					62	−2.2	1.8	
					71	−0.1	−0.5	
4	40°18N,70°51W	28 Feb 74	35	110	30	−7.8	0.7	A
					50	−7.4	3.0	
					70	−5.9	3.3	
					109	−0.8	0.0	
7	39°23N,70°59W	20 Aug 70	46	2,527	1,504	−2.8	0.3	B
9	39°35N,70°58W	20 Aug 70	111	2,263	2,163	−6.4	0.2	B
5	39°50N,70°40W	20 Aug 70	104	873	776	−6.4	1.6	B
6a	39°50N,70°56W	20 Aug 70	45	943	846	−2.0	1.0	B
			45		933	−2.0	0.4	
			45		941	−1.5	0.4	
b	39°50N,70°56W	20 Aug 70	451	993	880	−4.6	1.8	
			86		990	−3.6	−0.7	
8	39°37N,71°15W	20 Aug 70	111	2,150	2,052	−5.1	−0.4	B
10	39°23N,71°01W	20 Aug 70	56	2,509	2,394	−2.4	−0.5	B
2	40°45N,71°03W	8 Mar 73	33	60	42	−6.4	0.0	C
11	39°20N,70°00W	NA	NA	2,640	10	−13.0	−0.6	D
					100	−5.7	1.1	
					500	−3.7	−0.6	
					1,000	−3.5	0.3	
					2,000	−1.6	−0.1	
12	40°25N,73°28W	22 Mar 74	59	23	2	−0.7	−3.5	E
			89		20	−0.6	−0.7	
13	40°16N,73°13W	25 Feb 75	111	38	2	−4.7	−4.8	E
		25 Feb 75	37		25	−3.6	−2.2	
		29 Apr 75	48		23	−5.5	−3.9	
		25 Feb 75	111		37	−1.5	−0.3	
14	40°06N,72°54W	25 Feb 75	112	48	10	−2.4	−2.6	E
		25 Feb 75	62		26	−2.8	−1.1	
		25 Feb 75	112		40	−1.6	−1.3	
16	40°03N,72°42W	1 Mar 75	59	50	2	−2.8	−2.8	E
		1 Mar 75	59		27	−3.1	−2.8	
		1 Mar 75	108		42	−2.9	−1.9	
17	39°39N,72°38W	24 Feb 75	64	76	2	−2.6	−1.7	E
		23 May 75	25		2	−5.4	−9.1	
		24 Feb 75	64		26	−4.9	−0.7	
		24 Feb 75	64		41	−4.0	−0.6	
		30 Apr 75	42		42	−6.2	−7.4	
		24 Feb 75	64		75	−2.6	1.0	
18	39°28N,74°15W	1 Jul 72,73,74	60	12	5	−2.1	−2.9	F
		1 Dec 73,74	60		5	−2.0	−2.8	
		1 Jul 72,73,74	60		10	−1.7	−2.3	
		1 Dec 73,74	60		10	−1.3	−1.8	
15	40°07N,72°51W	18 Jun 74	35	50	2	−3.8	−6.9	E
		18 Jun 74			13	−6.8	−5.5	
		18 Jun 74			26	−4.1	−3.0	
		18 Jun 74			46	−1.7	−1.8	

Table 2. Continued

Sta. No.	Location	Start time	Record length (days)	Water depth (m)	Instr. depth (m)	E (cm/s)	N (cm/s)	Data source*
19	38°49N,74°12W	29 Oct 74	36	43	9	−6.2	−3.2	G
		29 Oct 74			23	−4.9	−1.0	
		29 Oct 74			35	−3.0	2.8	
20	37°55N,74°39W	26 Jun 74	22	35	24	−5.7	−7.5	H
21	36°50N,75°42W	21 Jul 74	29	16	4	−1.7	1.5	H
					15	−0.2	3.1	
22a	36°50N,75°02W	21 Jul 74	37	36	9	−2.4	−8.4	H
		21 Jul 74			20	2.6	−5.5	
		21 Jul 74			30	2.4	−1.3	
b	36°50N,75°02W	15 Jan 74	29	36	7	2.6	−8.6	
		15 Jan 74			20	1.6	−6.9	
		15 Jan 74			32	0.7	−4.7	
23	36°50N,74°48W	21 Jul 74	26	70	11	−3.7	−10.7	H
		21 Jul 74			30	−2.3	−16.6	
		21 Jul 74			58	0	−13.8	
24	36°50N,74°40W	21 Jul 74	18	70	76	−3.1	−12.6	H
		21 Jul 74			104	1.4	−6.6	

* A–Beardsley and Flagg (1976); B–Schmitz (1974); C–Beardsley and Butman (1974); D–Webster (1969); E–NOAA-MESA (in prep.); F–EG&G (in prep.); G–Boicourt (personal communication); H–Boicourt and Hacker (1976).

simultaneous measurements will provide conclusive evidence.

If the fluxes through the three transects are approximately the same, we postulate that there is little *net* flow between the shelf and slope regions. Wright (1976) estimated that as much as 2,000 km³/yr might leave the shelf region off New England via the "calving" process. This number, however, was determined on the basis of a much larger alongshore gradient in transport than we observed.

The various volume fluxes for the mid-Atlantic Bight are shown schematically in Fig. 3. For comparison, note that the alongshore transport of shelf water within the 100-m isobath, while 60 times the river runoff, is only about 0.3% of the northward transport of the Gulf Stream. The volume of the shelf water within the 100-m isobath is estimated to be $V_{Sh} \doteq 6{,}000$ km^3 (Ketchum and Keen 1955; Wright and Parker 1976) and the estimated alongshore mean flux of shelf water within the 100-m isobath is $T_{Sh} \doteq 8{,}000$ km^3/yr. The mean residence time is then $\tau = V_{Sh}/T_{Sh} \doteq \frac{3}{4}$ yr. This implies that the shelf water between Cape Hatteras and Cape Cod is removed from the shelf and entrained into the Gulf Stream in less than a year. This estimate is slightly less than the 1.3 years estimated by Ketchum and Keen (1955) who knew the freshwater inflow and the salinity distribution on the shelf and who assumed no flow entering or leaving the bight via the Nantucket Shoals and Cape Hatteras.

The Gulf of Maine and Georges Bank region must supply the low salinity water observed flowing westward through transect I, i.e. $T_{Sh} \doteq T_{GM} + T_{GB}$ following the notation in Fig. 3. This conclusion is implicit in the hydrographic structure of the shelf water in this region, namely that the shelf-slope water salinity front is a persistent and continuous feature from New York to the southern flank of Georges Bank and

Table 3. Alongshore transport to the 100-m isobath estimated through three transects across mid-Atlantic Bight. Position of individual transects shown in Fig. 2.

Transect	Cross-sectional area to 100-m isobath A (km²)	Transport to 100-m isobath T (km³/yr)	Mean speed $u = T/A$ (cm/s)	Period
I	6.4	5,300	2.7	Mar 74
II	7.6	8,800	3.7	Mar–Apr 75
III	3.6	8,200	7.2	Jul–Aug 74

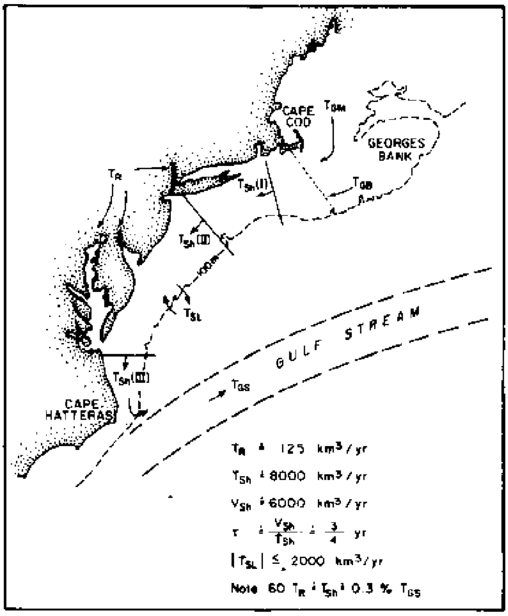

Fig. 3. Schematic diagram of the important volume transports for the Middle Atlantic Bight. T_R is the total annual freshwater runoff (of which over 50% occurs via the Chesapeake Bay), T_{Sh} is the alongshore transport over the shelf out to the 100-m isobath, T_{SL} is the net flux of slope water into the shelf water, T_{GS} is the transport of the Gulf Stream, and T_{GM} and T_{GB} are the unknown fluxes of shelf water from the Gulf of Maine and southern flank of Georges Bank. The volume of the shelf water mass out to the 100-m isobath is V_{Sh}, and the average residence time τ is simply V_{Sh}/T_{Sh}.

the "cold pool" is also continuous during spring and summer along this same section of the shelf (*see* Bumpus 1976). Any substantial flux of more saline slope water occurring across the 100-m isobath must be balanced by an increased flux of low salinity water (above T_{Sh}) from the Gulf of Maine and Georges Bank to maintain a steady salt balance. This conclusion is also dictated by simple continuity arguments which require a northern source region to maintain the observed westward flux of shelf water shown in Fig. 2.

The summer current measurements in transects II and III show that the alongshore currents in the cold pool water equals or exceeds the mean southward current of the surrounding warmer water. These measurements counteract the traditional impression that the cold pool, formed by winter cooling, remains stationary throughout the spring and summer seasons (Ketchum and Corwin 1964). There is good evidence (Ford et al. 1952; Boicourt 1973) that the cold pool moves southward and is entrained by the Gulf Stream. High alongshore velocities of the cold water, as measured in transects II and III, imply that the cold water found near Cape Hatteras in August must have formed by winter cooling near Cape Cod or perhaps in the Gulf of Maine.

Two large unknowns in the calculation of water and salt budgets in the mid-Atlantic Bight are fluxes of water and salt into the region from the north and amounts of water and salt exchanged across the shelf-slope boundary. Although we cannot yet quantify shelf-slope exchanges, we can describe some processes involved. In summer and winter much exchange appears to be wind controlled, with onshore-offshore flows in the upper Ekman layer compensated by opposite flows in the lower layer (Boicourt and Hacker 1976). In winter the cross-shelf flows driven by northeast winds enhance the thermal front at the shelf break and vertically mix the midshelf region. Winds from the south and southwest, on the other hand, cause offshore flows in the upper Ekman layer and intrusions of warm salty slope water along the bottom, thereby tending to stratify the outer shelf region.

Summertime cross-shelf circulation is larger and has a more complex vertical structure. Boicourt (1973) and Boicourt and Hacker (1976) found that southerly winds can drive an intrusion of high salinity slope water onto the shelf at middepths in the southern mid-Atlantic Bight. Because these intrusions have been commonly observed on the outer shelf, they may be an important process in shelf water–slope water exchange. Gordon et al. (1976) observed a high salinity layer at middepth in the New York Bight, indicating that such intrusions may occur widely in the bight. The cold pool and strong thermocline are evident in the water temperatures in the southern mid-Atlantic Bight (Fig. 4). The salinity distribution shows an intrusion of high salinity slope waters in the upper ther-

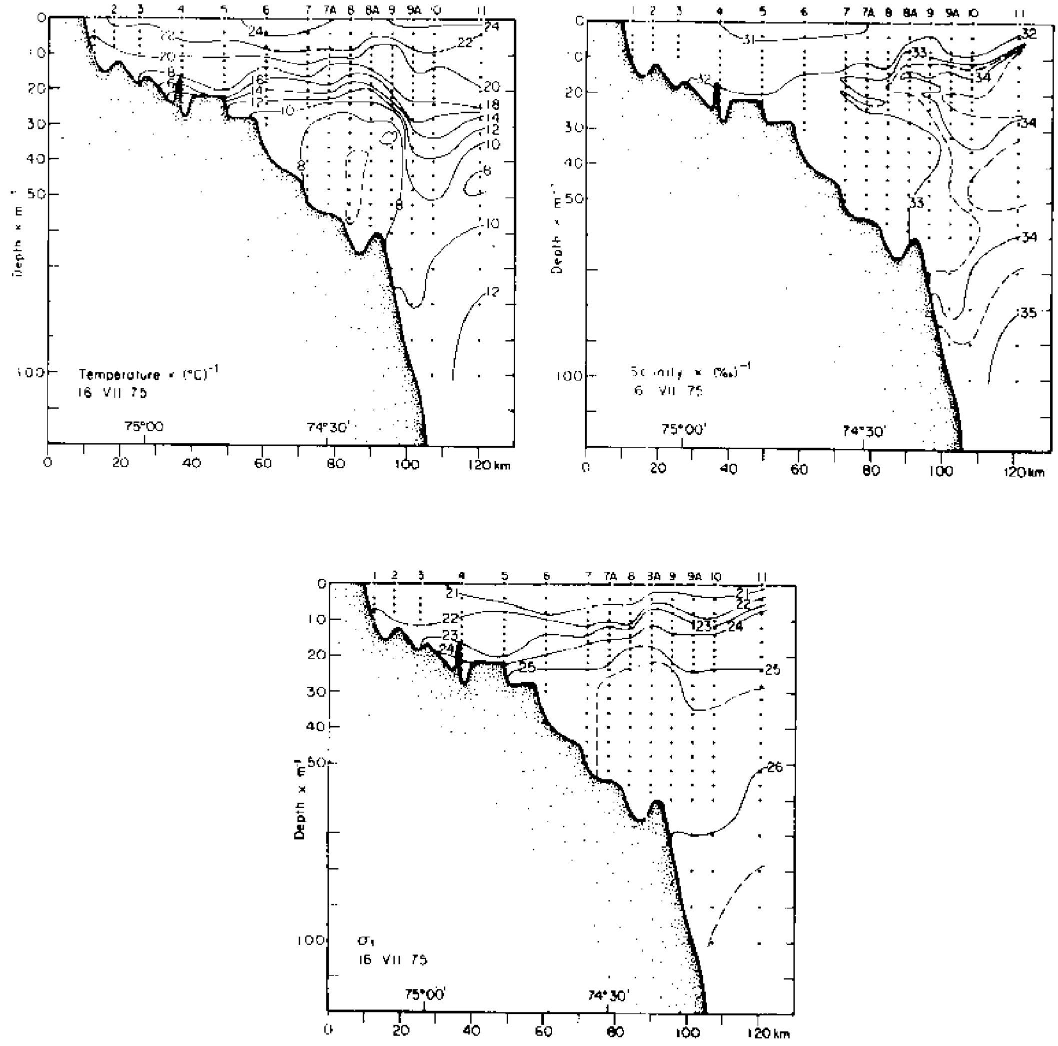

Fig. 4. Distributions of temperature, salinity, and σ_t in a cross-shelf vertical section off Ocean City, Maryland, July 1975.

mocline. This intrusion extends about 30 km inshore of the shelf break, apparently driven by southerly winds. A small parcel of cold (<8°C), low salinity water may have been detached from the cold pool and moved offshore. Because such parcels are commonly found in this position, however, the amount of water actually detaching is uncertain.

South of New England, Bigelow (1933) and Cresswell (1967) described calving of the cold pool with parcels or bubbles of shelf water moving into slope water. Wright (1976) suggested that significant interchange of shelf and slope water may occur via this mechanism. This process may be related to the formation of anticyclonic Gulf Stream eddies and their subsequent southwest drift along the edge of the slope. Satellite infrared photographs (e.g. Hughes 1975) suggest some exchange of shallow surface water, and Saunders' (1971) aerial temperature survey of one warm-core eddy suggests that some deep shelf water is pulled off the shelf and entrained into the trailing side of the eddy. How much shelf water is exchanged via these processes and with what frequency (i.e. the intermittency of these processes) is not known.

We conclude this section with a brief discussion of the physical processes that govern the mean circulation in the mid-Atlantic Bight. Stommel and Leetmaa (1972) have constructed a theoretical model (with linear dynamics) for the winter shelf circulation driven by a mean wind stress and a distributed freshwater source at the coast. They then applied this model to the bight and concluded that an alongshore sea level slope of about 10 cm drop from Cape Cod to Cape Hatteras must exist to drive the mean flow toward the southwest (as observed!) *against* the mean eastward wind stress. This same basic conclusion was also reached by Csanady (in prep.) who examined the influence of wind stress variability on the Stommel and Leetmaa model. This inferred alongshore pressure gradient can be either created by a succession of long, shore-trapped waves as suggested by Csanady (in prep.), who showed evidence for this process in Lake Ontario, or maintained by an upstream source of fresh shelf water, presumably here the St. Lawrence system and inshore Labrador Current. Sutcliffe et al. (1976) reported evidence that fluctuations in the transport of the St. Lawrence system can be traced down the Scotian Shelf and into the Gulf of Maine. This, together with our early point that most of the fresh shelf water observed flowing westward through transect I (Fig. 2) must be supplied by the Gulf of Maine and outer Georges Bank regions, suggests a continuous freshwater pathway from the St. Lawrence to Cape Hatteras. The alongshore pressure gradient inferred to occur over the mid-Atlantic Bight may be partially supported by a northward rise in sea level found by oceanic leveling in the slope water by Sturges (1974).

Special features of the New York Bight and adjacent nearshore zone

The New York Bight contains several features of general interest that have been intensely studied. Special topographic features of this region include a relatively deeply incised inner shelf region into which enters one of the major river systems of the region and the Hudson Shelf Valley and Hudson Canyon.

The Hudson-Raritan estuary has a characteristic circulation consisting of a seaward flow of relatively brackish estuarine water in the near-surface water, and a shoreward flow of more saline water near the bottom. The relatively great width and complicated channel system in the Sandy Hook-Rockaway transect allows inertial and Coriolis effects to further modify currents such that seaward flow tends toward the southern side of the entrance, and the inflow occurs mainly in the navigation channels and along the northern side of the entrance (*see* Parker et al. 1976). This mean flow of a few centimeters per second is a weak residual superimposed on stronger tidal flow but causes most of the material exchange between the estuary and shelf regions.

The Hudson Shelf Valley is the offshore expression of the Hudson estuary. Current measurements in this valley (30 km off the New Jersey shore) indicate that the average flow in the valley over intervals as long as a month can be shoreward with an average speed of a few kilometers per day. Such flows are more than ample, if coherent in space, to return suspended materials to the harbor entrance from far out on the continental shelf.

The combination of the Hudson estuary, the complex bottom topography, and the nearly right-angle bend in the shoreline produces quite complicated flow patterns over the inner shelf. There is evidence in the water properties that the near-surface flow from the estuary tends to move southward along the New Jersey shoreline. Recovery of seabed drifters suggests the statistical occurrence of a mean clockwise circulation within the inner bight, counter to the flow over the shelf farther offshore. This circulation is sometimes reflected in current measurements (Charnell and Mayer 1975), but the current regime is best described as more dispersive than advective, especially during spring and summer, the seasons of maximum stratification.

An interesting and significant aspect of the flow in the inner bight is a shoreward velocity component in the bottom boundary layer. Numerous current measurements have been made for the NOAA-MESA project at distances of 1–5 m above the bottom. Averages of such measurements over any significant time frequently show a distinctly shoreward component. In 19 out of 21 cases examined in which a clear distinction could be made, there was a shoreward component in the bottom boundary layer. Furthermore, subdividing the data into sets in which the flow is east or west along Long Island, for instance, yields the same result: flow in the bottom boundary layer is shoreward in both cases. It is not yet ascertained whether this shoreward veering is a result of surface winds or whether it may be a manifestation of estuarine circulation generally over the shelf, but in any case it suggests a tendency for near-bottom materials to be carried inshore. Such a process is a plausible explanation for the relatively high and constant rate of return of seabed drifters from bight waters (Charnell and Hansen 1974) and supports previous reports (Bumpus 1973).

Some remaining problems

Although progress has been made in determining the current variability and circulation pattern over the mid-Atlantic shelf, we are still unable to provide unambiguous answers to many questions of a basic engineering sort posed by environmental managers. Only a general estimate of the flushing rate of the shelf is available, and critical evaluation of the importance of the shelf-break exchange is not yet possible. Although a first-order description of flow to be expected can now be given for many parts of the bight, our ability to predict details and events remains poor. The dominant forces controlling the circulation are believed known but their relative importance and region of influence are not. Neither conceptual nor observational tools are adequate to the task for modeling of other than tides and tidal currents. Local models have useful applications but must be posed very carefully (especially boundary conditions) in the context of what is and what is not known about the physics of water movement over the shelf. It cannot be safely assumed that the way to solve a given management problem will be pointed by a mathematical model in any straightforward sense. Finally, there remain fundamental questions related to smaller scale phenomena, especially mixing and other dissipative processes. Smaller scale topographic features like the inner New York Bight embayment, the ridge and swale areas, and the shelf valleys and submarine canyons must exert some steering influences on the local flow. Some of these smaller scale problems will be immediately addressable when the physics of shelf circulation are better known; others must await improvement of observational instruments and techniques.

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Ocean–atmosphere interactions off the northeast coast of North America

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Abstract

The New York Bight–middle Atlantic continental shelf region is typical of midlatitude continental east coast areas in the intense sea–air interaction occurring during late autumn and winter. During these seasons dry but cool air moving over warm coastal water takes up a considerable amount of heat and moisture (latent heat). The daily uptake near the coast can exceed 0.5 g of water per cm^2 on a monthly mean basis.

The response of the surface layers of the atmosphere to the properties of the sea surface is relatively rapid so that quasi-equilibrium conditions are established after surface air has traveled about 6 h over water.

The main consequences of this large heat and moisture uptake on the atmosphere are deepening of lows that pass the coastline and frequent cyclogenesis off the coast. As the heat and moisture are transported upward, a trough forms at the 850-mb level. A thermally driven cyclonic surface wind component is created along the coast but it often escapes the attention of synoptic weather analysts.

The main effect of the atmosphere on the ocean is rapid cooling of already cool coastal waters during autumn and winter, resulting in increases in thermocline depth. The coastal southerly current is influenced by the surface cyclonic wind component.

The following review emphasizes sea–air interactions off the east coast of North America and summarizes data not given in past literature. Some coastal sea–air interaction phenomena that have not received much attention are also included.

Most sea–air interaction processes and their effects are mesoscale phenomena, but to keep this paper brief, only the large-scale effects are emphasized.

Characteristics of sea–air interaction in the New York Bight–middle Atlantic continental shelf region

Continental east coasts have a characteristic sea–air interaction process determined by the flow of continental air masses over the ocean. During winter, continental air is much colder than coastal waters; during summer, it is generally warmer than coastal waters. In both seasons, continental air is usually relatively dry. As a consequence of the temperature and water vapor pressure deficit of the near-surface air flowing from the continent over the ocean, intense sea–air interaction takes place off the east coast of the United States. Considerable evaporation occurs off the east coast year-around, but it reaches the maximum in autumn and winter when it may exceed, on a monthly average, 5 $mm/cm^2/d$ (*see Fig. 1*) (Larson and Laevastu 1975). The evaporation in Fig. 1 is computed using twice-daily surface pressure analyses from which surface wind is computed. The water vapor pressure deficit is computed using surface wind and sea surface temperature as outlined later. Finally, evaporation is computed using the Rohwer-Laevastu evaporation formula (Rohwer 1931; Laevastu 1960). The amount of evaporation decreases with increasing distance from the coast and depends to some extent on the stability of the lowest atmospheric layers. During summer the lower layers of warm air are cooled rapidly over the cold coastal waters and a stable stratification is soon established. However, some evaporation still continues due to the low water vapor pressure of the air, near-surface turbulence (winds, waves), and the absence of a sharp inversion in the atmosphere. During winter, cold, dry continental air is warmed and the resulting increased turbulence carries heat and moisture high up into the atmosphere. As a consequence, rapid offshore cyclogenesis often occurs because of the rapid uptake of heat and moisture and the release of latent heat from upward vertical motions promoted by heating from below.

A characteristic feature off these east coasts is the narrow, equatorward flow of cold water on the continental shelf near the coast, directly affected by sea–air interaction consequences, and the strong poleward flow (Gulf Stream) off the continental slope.

The sea–air interaction off the U.S. east coast is further complicated by the presence of the Gulf Stream front and warm Gulf Stream water farther offshore. A description of this consideration is not within the scope of this review but concerns the effects of oceanic fronts on the atmosphere in general (Laevastu and Hamilton 1974).

Sea–air interactions and their effects in coastal regions are usually synoptic, mesoscale phenomena and could be easily recognized if mesoscale synoptic analyses were made and if a sufficiently dense synoptic observation network existed off the coast. This is rarely the case: shipping lanes are some distance from the coast and ships near land are not required to make or transmit synoptic reports. However, mesoscale interaction effects are additive and a large-scale picture of the effects of sea–air interactions emerges as we go higher up into the atmosphere. This can, for example, be easily recognized when comparing typical monthly mean 500-mb patterns with sea surface temperature patterns, especially during winter. The typical east coast effects of the sea–air interaction along the coast start from the northern border of Florida and intensify northward, reaching a maximum off Nova Scotia. This can be seen in the annual change in the heat content of the upper layers of the ocean (Fig. 2). Off the Canadian coast north of Nova Scotia, the local sea–air interactions are strongly dominated by local oceanographic conditions created by the cold Labrador Current and large-scale anomalies in monthly mean conditions in various parameters in the ocean as well as in the atmospheric circulation off Newfoundland.

Marine fog, occurring especially during spring in the New York Bight, is a direct consequence of sea–air interaction processes.

Our knowledge and understanding of sea–air interaction processes and their consequences is changing. For example, the concept of heat and moisture uptake as a steady microscale process is no longer fully applicable, because the importance of the gusty "thermals" in this transfer is being emphasized. Offshore winds are usually much more gusty than onshore winds.

Response of surface air properties to sea surface properties

Basic to sea–air interaction studies is knowledge of the response speed of the properties—temperature and water vapor pressure—of the lower layer of the atmo-

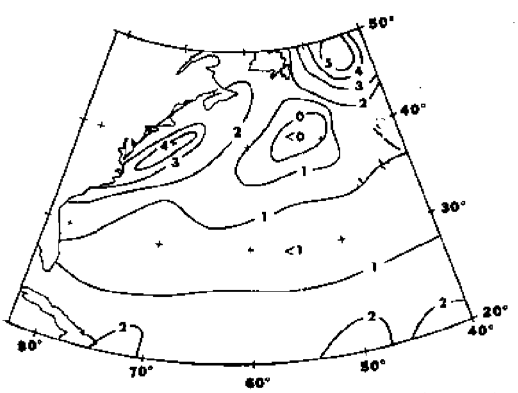

Fig. 1. Monthly mean evaporation in the northwest Atlantic in February 1967 (mm/d).

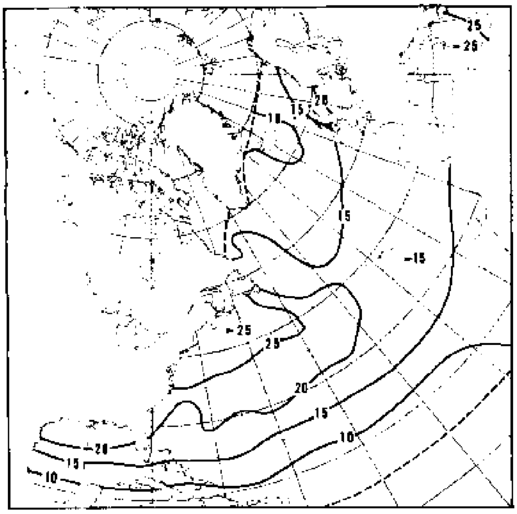

Fig. 2. Difference in heat content in upper 30-m layer between February and August (kcal/cm²).

sphere to the corresponding properties of the sea surface. Differences in these properties determine heat and moisture exchange rates. Usually connected to this problem is the turbulent transfer of heat and moisture upward into the atmosphere, a function largely of the wind speed but also dependent on stability. Properties of the surface air are referred to a standard height of 10 m. The routine synoptic meteorological measurements of air temperature and water vapor pressure (dew point temperature measurement) by vessels voluntarily observing and reporting are inaccurate, partly due to the influence of the ship. Furthermore, these properties vary over short distances and in synoptic time scale. Thus, methods must be found to compute these properties from other more reliable and generally available properties, such as the direction and speed of the surface wind and the sea surface temperature. Past investigations have shown that the evaporation and the transfer of sensible heat are functions of the rate of change of the temperature of the sea surface under the trajectory of the wind. The best results, which verify empirical observations, have been obtained by the Åmot-Mosby formula (Åmot 1944; Mosby 1957):

$$\frac{d\Delta T}{dt} = W \times \nabla T_w - \frac{dT_a}{dt}, \qquad (1)$$

where ΔT is $T_w - T_a$ (the difference between sea surface and air temperatures) and $W \times \nabla T_w$ is the rate of change of sea surface temperature along the trajectory. A similar equation applies to water vapor pressure. Observations have shown that the change of air temperature can well be represented as

$$\frac{dT_a}{dt} = K(T_w - T_a) - C. \qquad (2)$$

By substituting 2 into 1, rearranging, and assuming $W \times \nabla T_w$ to be constant, 1 can be transformed into integrable form:

$$\frac{d\Delta T}{dt} + K\Delta T = C + W\nabla T_w, \qquad (3)$$

which after integration gives

$$\Delta T = \Delta T_0 e^{-Kt} + \left(\frac{C}{K} + \frac{1}{K} V_r \frac{\partial T_w}{\partial r}\right)(1 - e^{-Kt}), \qquad (4)$$

where r is the distance and T_0 is at the time t_0. The empirical constants of C and K must be determined from observational data.

Using K as 0.28 and C as 0.13, assuming an initial sea–air temperature difference of 2.5°C, and using various rates of sea surface temperature change $(V_r \partial T_w / \partial r)$ designated hereafter as ΔT_w, we find that air moving over the constant sea surface temperature gradient reaches a constant temperature after about 6 h of travel (Fig. 3). Thus the response of the properties of the surface air to the properties of the sea surface is relatively rapid. The final equilibrium sea–air temperature difference depends on the rate of change of sea surface temperature (ΔT_w) under the trajectory of surface wind. The same formula is used for both stable and unstable conditions, although one would expect some differences in the responses of the two conditions. However, the response difference in stability conditions is indirectly included in the formula by the rate of change of the properties of the underlying surfaces because the equilibrium difference between the temperature of the air and the sea surface depends on this rate of change. Furthermore, the lowest layers of the atmosphere are always turbulent, except in relatively calm conditions.

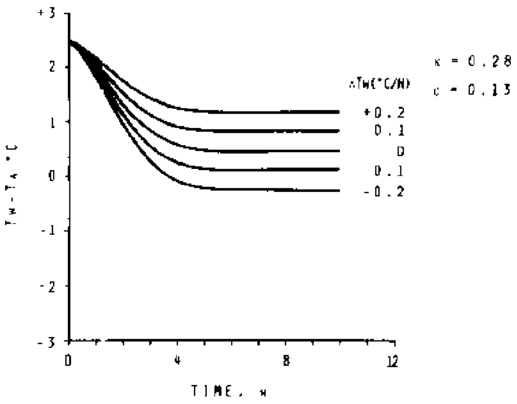

Fig. 3. Change of sea–air temperature difference with time; computations made with formula 4.

The same formulas are used for computing water vapor pressure of the surface air at a 10-m height (ΔT is replaced by Δe). The empirical constants for computation of Δe are different from those for computation of ΔT. The latest experimentally determined values are for ΔT, $C = 0.10$ and $K = 0.35$ and for Δe, $C = 0.40$ and $K = 0.50$ (Laevastu and Harding 1974).

The distribution of heat and moisture in the low layers of the atmosphere over the sea under various meterological conditions and at various latitudes (especially the vertical transports of these quantities) has not been sufficiently studied in the past. As a result, no sound, empirically verified, scientific basis exists for the various vertical transfer coefficients used in detailed boundary layer models.

Effects of energy feedback from ocean to atmosphere

Effects of feedback from ocean to atmosphere are most pronounced during fall and winter. During these seasons atmospheric surface lows moving off the east coasts deepen rapidly and change their course, usually to NNE, following the upper air flow. Also rapid cyclogenesis often occurs along the coast and along the oceanic frontal zone (the Gulf Stream front off the eastern U.S.).

Existing hemispheric numerical weather forecasting models with coarse grid nets usually contain treatment of the energy feedback from ocean to atmosphere in simplified form. Most boundary layer models, mainly experimental at present, use a number of constant empirical exchange coefficients, often variable in space and time and not verified experimentally. Satisfactory detailed initial conditions (analysis) for small-mesh primitive equation (PE) boundary layer models are nearly impossible to obtain due to sparse synoptic data.

A way to demonstrate and "tune" the effects of the oceanic feedback on the atmosphere is with three-parameter filtered

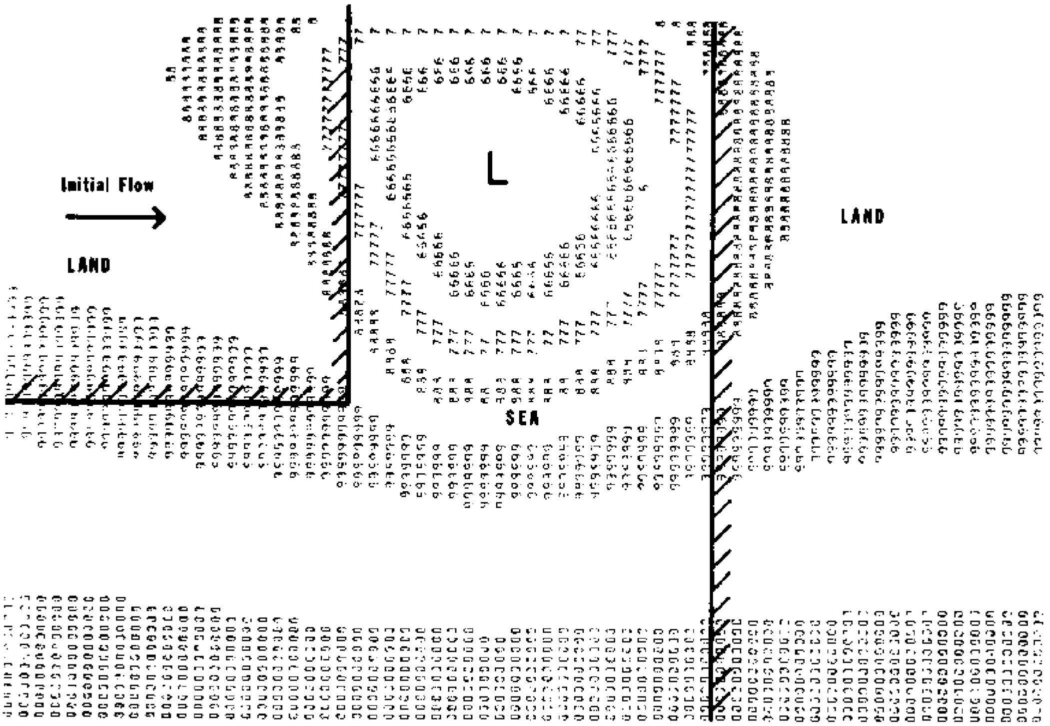

Fig. 4. Example of output from Bengtsson's three-parameter small-mesh model, depicting the development of a low in 12 h from an initially zonal flow of cold, dry air over warm water.

models, such as Bengtsson's (1974), which eliminate many shortcomings and difficulties of the PE models. Although we have not been running this model for the U.S. east coast, we have carried out some experimental work in a "cyclic channel" and in the Gulf of Alaska. An example of the cyclogenesis in 12 h from an initial zonal flow of cold air from the land toward the warmer sea is shown in Fig. 4. This model contains energy feedback from the ocean that we believe is physically correct. Details of the model as well as tests and verifications will be described elsewhere.

Examples for the east coast of typical sea–air interactions in synoptic scale are shown in Figs. 5 to 7. Figure 5 shows a small low developing over the east coast on 12Z, 18 February 1967; a portion of cold high is "bulging" over the Gulf of St. Lawrence. The computed evaporation patterns in Fig. 6 are largely influenced by the direction and speed of the surface wind in relation to sea surface isotherms. The great evaporation near the coast and near the Gulf Stream boundary is determined by the water vapor deficit in the lower air layers. Figure 6 also shows the direction and speed of 850-mb winds (shown by arrows). Most of the moisture (latent heat) is transported laterally between the surface and 750-mb level; thus, 850-mb winds indicate well the direction and speed of this transport. Considering the magnitude of latent heat uptake (evaporation) and its mean transport (850-mb winds), we can trace the energy

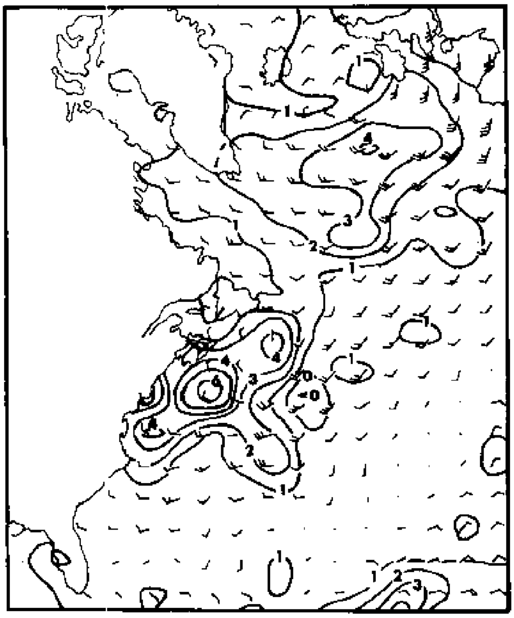

Fig. 6. Computed evaporation (mm/d) and 850-mb winds (arrows) over the North Atlantic on 0Z, 19 February 1967.

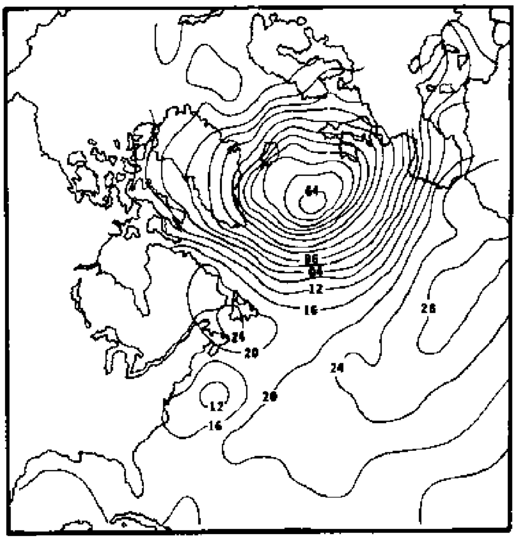

Fig. 5. Numerical surface pressure analysis over the North Atlantic on 12Z, 18 February 1967.

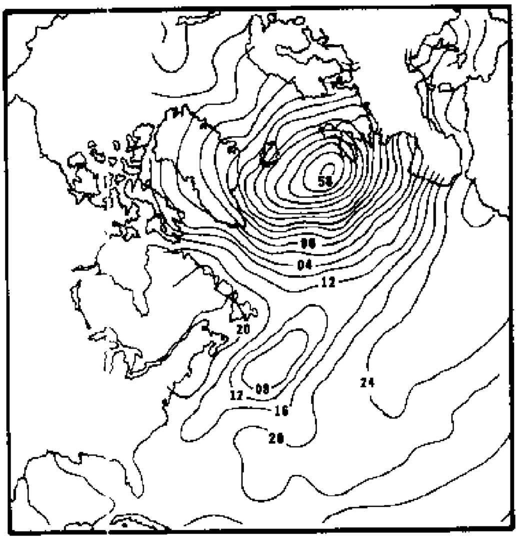

Fig. 7. Numerical surface pressure analysis over the North Atlantic on 0Z, 19 February 1967.

source for the deepening of the downstream lows (seen in Fig. 7 off the Gulf Stream).

The steering of surface systems might be affected by surface energy uptake patterns besides the well recognized upper air steering. In any case, the upper air flow patterns themselves are affected by surface energy uptake.

Other effects of the sensible and latent heat uptake off the east coast and in the Gulf Stream boundary region on the higher levels of the atmosphere above the sea can be recognized in Fig. 8 where the 850-mb *D*-value isopleths turn southward off the east coast (in relation to 500-mb *D*-value isopleths), indicating the lowering of the 850-mb level as a result of lower surface pressure. (The *D*-values indicate the deviation of 850- and 500-mb heights from their "standard" height.) Farther offshore the 850- and 500-mb monthly mean isopleths are parallel and follow roughly the sea surface temperature isotherms.

Additional effects of sea–air interactions off the east coast can be seen on monthly mean surface wind speed isotach charts (Figs. 9 and 10) (Larson 1975).

During winter (Fig. 9) the basic east–west trends of the isotachs suggest that in the long term monthly mean the wind speed is dominated by the north–south temperature gradient. The influence of the coastal sea–air interaction can be seen in the north–south trend of the isotachs immediately off the coast.

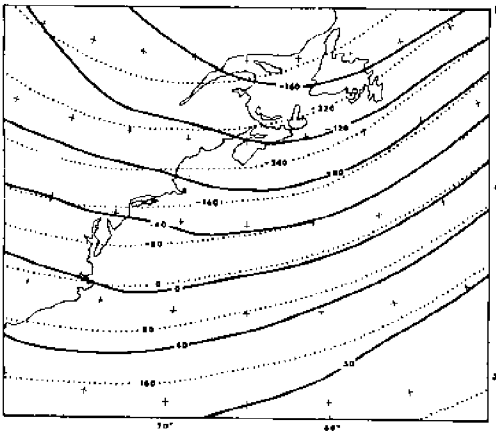

Fig. 8. Monthly mean 500-mb (dotted lines) and 850-mb (solid lines) *D*-values in February 1967 (m).

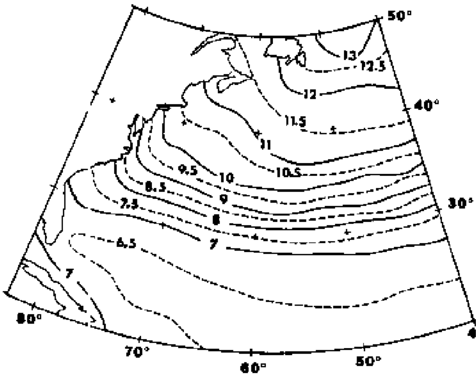

Fig. 9. Monthly mean surface wind speed during January 1971 (m/s).

In summer (Fig. 10) the north–south trend of the isotachs is farther offshore. In this season the highest wind speeds occur after the Gulf Stream is traversed. This suggests a greater effective temperature difference between the Gulf Stream and North Atlantic waters than between the continent and Gulf Stream during summer.

The coastal ocean off the New York Bight–middle Atlantic continental shelf region is ideal for studying local-, medium-, and large-scale effects of the ocean on the atmosphere. This area is in many ways more suitable for this purpose than the GATE area off North Africa. Of special interest would be case studies of rapid modification of surface atmospheric systems off the coast during the cooling season connected with numerical modeling (reproduction of the observed modifications). The

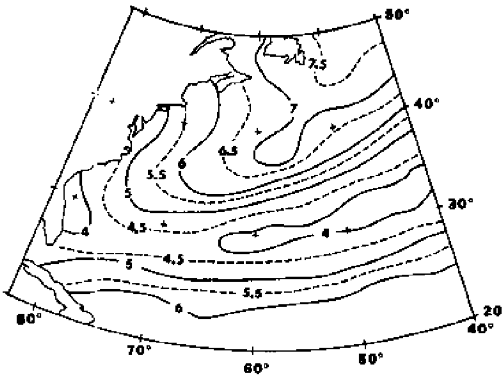

Fig. 10. Monthly mean surface wind speed during July 1971 (m/s).

ocean area off the U.S. east coast offers opportunities for studying vertical transport of heat and moisture in the atmosphere, where our quantitative knowledge (e.g. of the vertical eddy exchange) is still insufficient.

Effects of sea–air interactions on the ocean

Because of the near-infinite moisture content and the high heat capacity of the oceans, the ocean–atmosphere interaction effects are relatively small on the ocean and can be seen mainly on a monthly or seasonal basis; a few interaction aspects, such as waves and deepening of the thermocline, can be observed in synoptic conditions. One principal effect of the energy exchange between the ocean and atmosphere on the ocean is the heating and cooling of surface layers of the sea. Most intensive heating/cooling takes place in nearshore waters where the sea–air temperature and water vapor pressure differences are usually largest.

Most heating/cooling follows a seasonal cycle with timing and magnitude anomalies from year to year. Intensive cooling from evaporation and heat exchange in coastal water (and partly from the southward advection of cold water along the coast) keeps the coastal waters in the New York Bight–middle Atlantic continental shelf region considerably cooler than at the corresponding latitudes along the U.S. west coast, provided no upwelling occurs in the comparison area. In the Gulf Stream proper, heat is transported from low latitudes to high latitudes, and the annual heat cycle is essentially determined by changes in this transport and by cooling. The annual heat storage change (difference in temperature between the coolest and warmest months) is largest off the northeast coast (*see Fig. 2*).

Although the general surface current circulation pattern (net current) in the New York Bight–middle Atlantic continental shelf region is largely determined by ther-

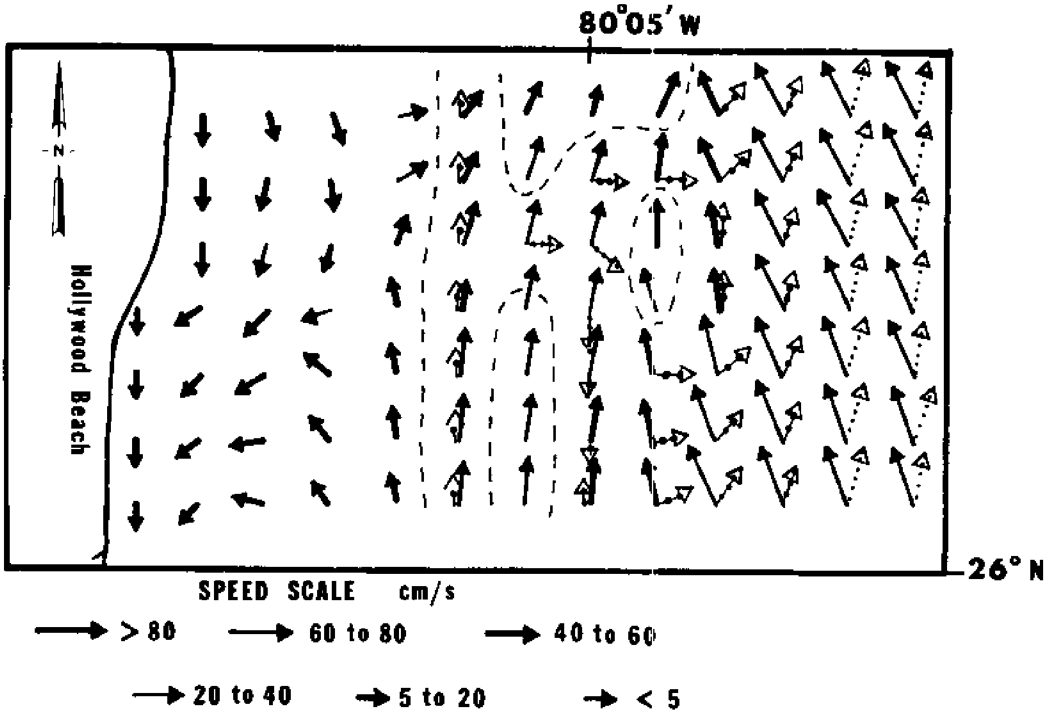

Fig. 11. Computed surface (solid arrows) and below thermocline (dotted arrows) currents off Hollywood Beach, Florida, 2 h after high water, wind 8 m/s from NNE. Currents computed with a two-layer hydrodynamical-numerical (HN) model, grid size 300 m.

mohaline gradients, the mean wind and the "modified" surface wind systems near the coast (caused by sea–air interactions and described briefly earlier) contribute considerably to the maintenance of this circulation.

The wind affects the southward flow in shallow water farther south along the Florida coast. An example of computed currents off Hollywood Beach using a two-layer model is shown in Fig. 11 (Laevastu and Stroud 1975). The computation was made with an input of NNE wind of 8 m/s, which yields the southward flow in shallow water. Other computations with SSE winds did not give this southward current.

In the synoptic time scale, tidal currents predominate on the continental shelf of the New York Bight–middle Atlantic continental shelf region. The general circulation is superimposed on these tidal currents, which are greatly affected by local bathymetry. An example of computed tidal currents at 3 h after low water at Sandy Hook, with winds of 17 m/s from 295°, is shown in Fig. 12 (Laevastu et al. 1974). The effect of the Hudson Submarine Canyon and the strong southward flow off the New Jersey coast can be recognized in this figure.

Sea–air interactions affect the mixing in the upper layers of the ocean, partly by mechanical mixing from wind action (waves) and partly by convective stirring during fall and winter. In the beginning of the cooling season the thermocline deepens rapidly due to the rapid cooling and evaporation from the surface, lowering the temperature as well as increasing slightly the salinity of the Gulf Stream water; dense waters are thus created at the surface.

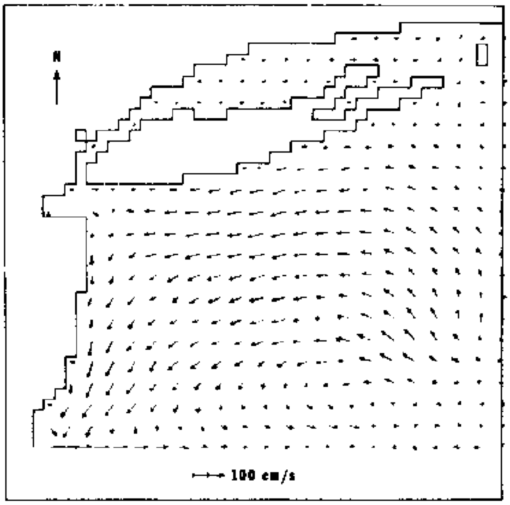

Fig. 12. Computed currents (with HN model) in New York Bight 3 h after low water at Sandy Hook, wind 17 m/s from 295°.

Waves in the New York Bight–middle Atlantic continental shelf region are usually created by local winds. Although swell does occur, it is considerably less than on the northwest coast, partly because of the prevailing offshore flow along the east coast.

Finally, most pronounced sea surface temperature anomalies (at times in excess of 3°C) occur off Nova Scotia and Newfoundland. These are caused by atmospheric circulation anomalies, both through advectional effects on currents: advection of cold (Labrador Current) or warm (Gulf Stream) water to the anomaly area and through heating/cooling by heat exchange. These atmospheric circulation anomalies usually reflect a dislocation of the climatological Icelandic Low. Thus, with these sea surface temperature anomalies, the ocean acts as an integrator of the passing surface systems. The sea surface temperature anomalies off Newfoundland can be related to weather types farther downstream to the British Isles and western Europe where these relations are being used for monthly weather outlooks.

Additional studies of various aspects of sea–air interaction off the U.S. east coast are needed in which the ocean and atmosphere are considered as a fully coupled system.

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Recent progress in the numerical prediction of ocean waves

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Three computer-based numerical wave specification and wave prediction models have been developed during the past 5 years or so. One is for the deep-water northern hemisphere oceans on a triangular grid with points about 350 km apart. Another is for the Mediterranean. The third is for tropical cyclones in the Gulf of Mexico with points about 37 km apart. The first two are presently operational at the Fleet Numerical Weather Facility at Monterey. The tropical cyclone model was just tested again on the recent hurricane in the gulf that made landfall on the Florida panhandle.

Various groups are evaluating how well these models both specify and forecast waves. One recent evaluation showed a 1.3-m RMS error in wave specification for waves varying from 2 to 17 m in significant wave height. The major source of error appears to be the lack of accuracy with which the wind fields over the ocean are specified and forecasted.

The development of Seasat-A, to be launched in 1978, should provide vastly improved wind information over the oceans and vastly improved wave forecasts.

The implication of these developments for specifying waves on the continental shelf of the U.S. east coast is that an improved higher resolution model should make it possible to describe the waves for this important area of the world.

New York Bight water stratification—October 1974[1,2]

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Abstract

Thermohaline stratification of New York Bight continental shelf water during October 1974 is basically of the summer regime. Salinity increases markedly with increased distance from the coast, yet a basic vertical structure is maintained: an upper isohaline layer; a salinity maximum at the top of the thermocline; a salinity minimum at the base of the thermocline; a deep isohaline layer associated with the cold near-bottom winter residual stratum; and (over the outer shelf) a bottom intrusion of relatively saline and warm slope water. Inversions in temperature and salinity are common within the thermocline.

The pycnocline is continuous over the shelf and slope, though some weakening and deepening occurs over the shelf break. Over the shelf it is mainly supported by the thermocline and over the slope by the halocline. The pycnocline may not be an effective barrier to isopycnal interchange of surface and deep layers in view of the relative slope of isopycnals to pycnocline.

In October 1974 oxygen distribution of the continental shelf was primarily two-layered, with a sharp division at the pycnocline. The lower cold layer has an oxygen concentration of near 60% of full saturation, with values near 3.6 ml/liter. This is low; if the deeper layer is principally a residue of the winter homogeneous condition with initial saturated oxygen values, it would represent oxygen consumption at a rate of 2.6 ml/liter during the six summer months after accounting for the low oxygen influx of slope water.

Between 16–31 October 1974, a physical-chemical oceanographic cruise was carried out in the New York Bight aboard RV *Vema*, cruise 32-01. There were 58 hydrographic stations (Fig. 1) obtained with a Plessey model 9040 STD (data are reported by Amos 1976). Water samples were collected for determination of dissolved oxygen and various geochemical properties (Biscaye and Olsen 1976). The STD profiles were standardized to reversing thermometer-water bottle derived temperature-salinity-depth data. The *Vema* station distribution comprises three transects across the shelf, slope, and upper rise of the bight, with emphasis on the Hudson Channel-Canyon system. Here we report the October 1974 thermohaline stratification as revealed by the STD hydrographic stations.

October is a transition month between the highly stratified summer regime of the water column and the more vertically homogeneous winter regime. The conversion from the summer to winter condition begins in September when stored heat begins to be liberated to the cooler atmosphere and incoming shortwave radiation is reduced. The heat flux accelerates during October, inducing the upper layer to deepen and erode into the lower layer by convective processes. By the end of November the two-layer stratification is destroyed. Vertical gradients remain small until warming begins toward the end of March.

Water mass divisions

There are two divisions in the *T-S* distribution (Fig. 2).

Deep water division—a well defined *T-S* curve extending toward low temperatures from the warm-saline end point (near 15.8°C, 36.09‰ at station 68). There are a number of gradient changes of the *T-S* curve within the deep water division which marks water mass components (*see "Deep stratification"*).

Surface water division—represented by *T-S* points on the low salinity side of the deep water *T-S* curve. The low salinity

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[1] The field program, data processing, and data interpretation were funded by U.S. Energy Research & Development Administration (ERDA) contract AT(11-1)2185.

[2] Contribution 2402 of Lamont-Doherty Geological Observatory.

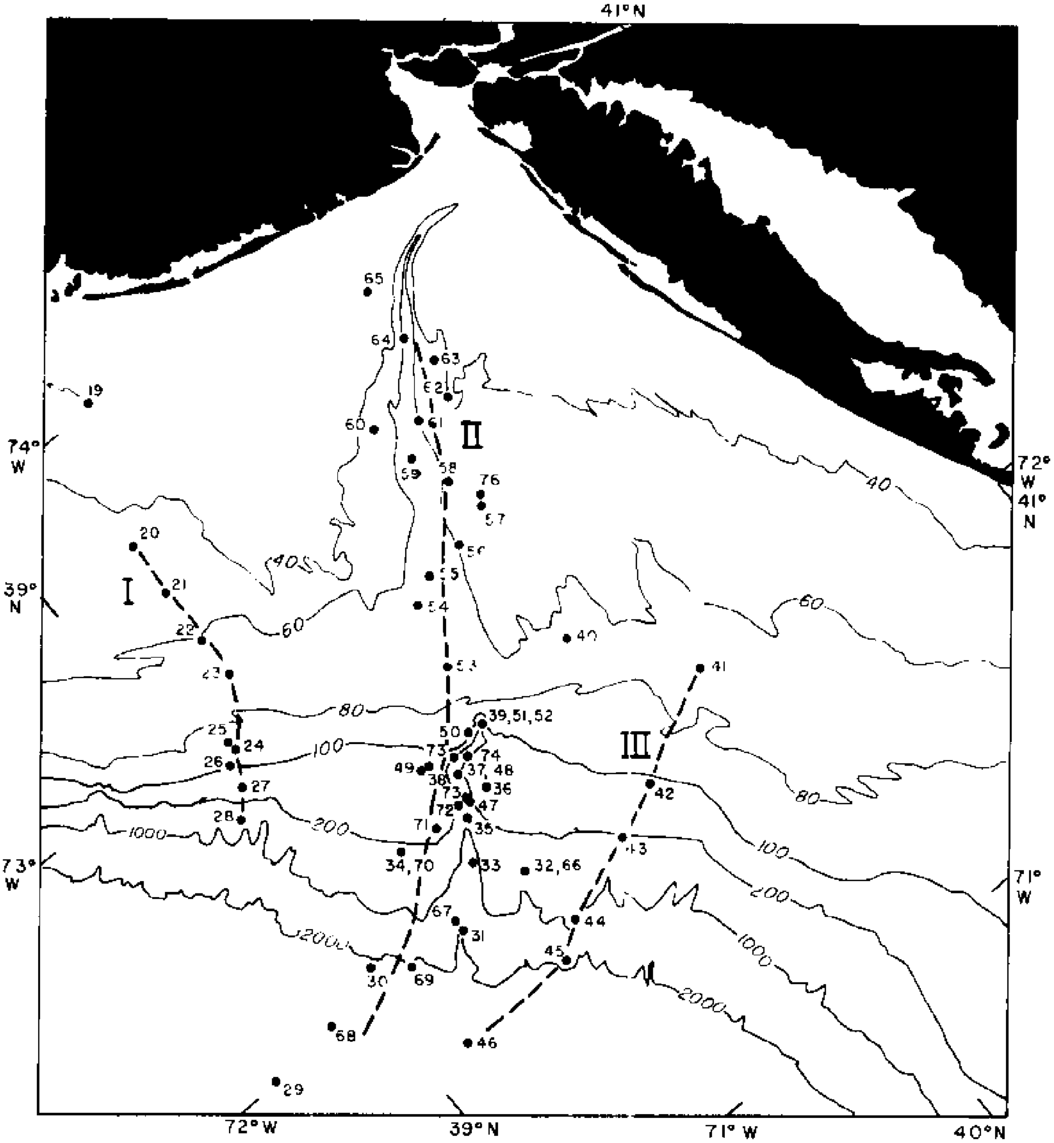

Fig. 1. New York Bight topography (Uchupi 1968), and position of RV *Vema* cruise 32–01 STD hydrographic stations 19–76, October 1974 (Amos 1976). Positions of three profiles referred to in other figures are shown.

character of the surface division is a product of river discharge.

It is possible to separate surface water into shelf water and slope water components near the shelf break where isohalines tend to near vertical. The 34.0‰ isohaline is a convenient boundary between shelf and slope surface water, since during October 1974 it is close to the average salinity over the shelf break.

Within the surface waters there is a three-layered stratification pattern; the inset in Fig. 2 shows station 50 near the shelf break as an example.

"*Clines*"

Thermocline—Thermohaline stratification over the continental shelf during October 1974 is multilayered, though significant cooling of the surface layer has apparently

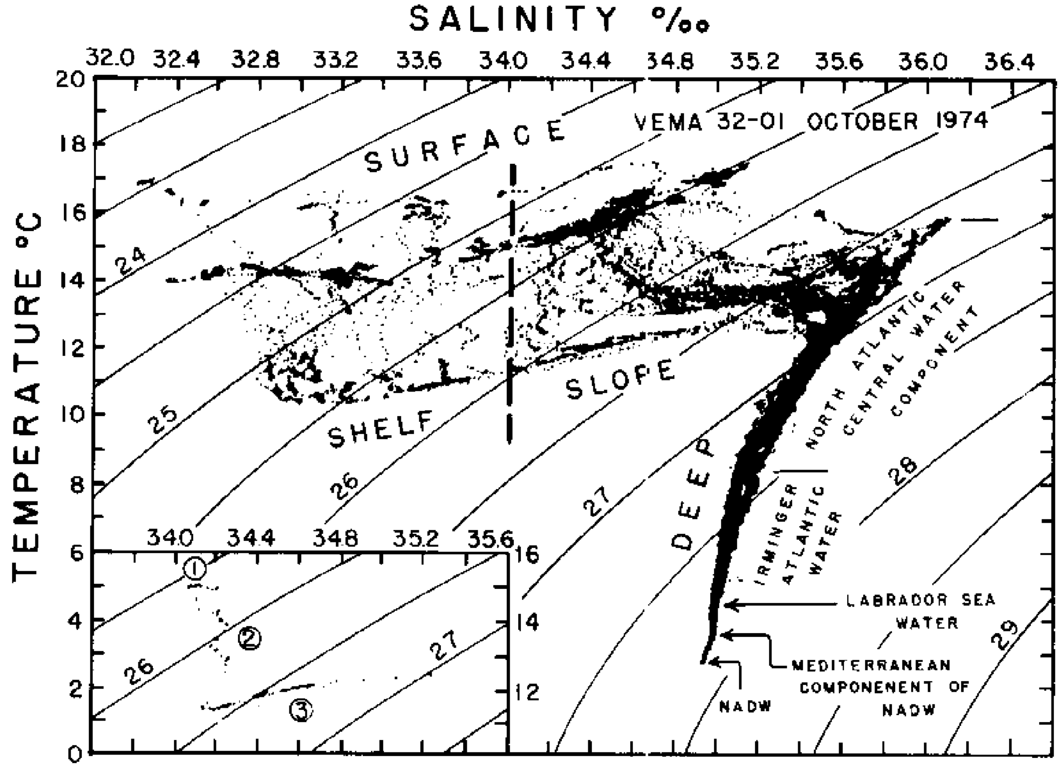

Fig. 2. Group *T-S* scatter of *Vema* 32–01 data. For the upper 100 m every data point is shown (about 1-m intervals), below which not all data points are plotted. The two water mass divisions, surface and deep, are subdivided as discussed in the text. The inset is the *T-S* scatter for station 50, with the *T-S* expression of the three basic stratification units indicated.

taken place, presumably deepening the surface layer. In July and August surface temperatures near 22–23°C are more typical (Walford and Wicklund 1968), or about 8° above the October 1974 surface temperature. Stations 19–24 are the earliest and southernmost stations, which may explain their warmer surface waters. Their *T*-S curves fall significantly above the general scatter of the shelf surface water (Fig. 2).

Stratification was strong in October 1974, especially over the inner shelf. At station 64 (Fig. 3), the thermocline is at 40–51-m depth, with a drop from 14.1°C to 11.1°C, which yields a Brunt-Väisälä period of 4.6 min. The thermocline (Fig. 4) extended seaward, becoming slightly deeper and weaker over the outer shelf, eventually disappearing (or becoming significantly weaker) in the vicinity of the shelf break.

Halocline—Salinity stratification over the inner shelf (e.g. stations 63 and 64) during October 1974 acted as a stabilizing influence, reflecting the estuarine character of coastal waters. River water was confined to the upper layers and spread with a seaward component, while deeper, more saline waters spread landward. This simple pattern for the inner shelf is shown by bottom drift measurements (Bumpus 1973).

Salinity stratification becomes more complex over the middle and outer shelf, and numerous inversions are observed. In these areas the salinity gradient acts as a destabilizing influence within the thermocline (e.g. station 50).

Over the shelf the halocline is weak (except for locally high gradients associated with thermocline features). Over the outer shelf and shelf break isohalines are vertically oriented, i.e. the predominant gradient is lateral. A significant halocline is established seaward of the shelf break. This halocline marks the transition from the surface

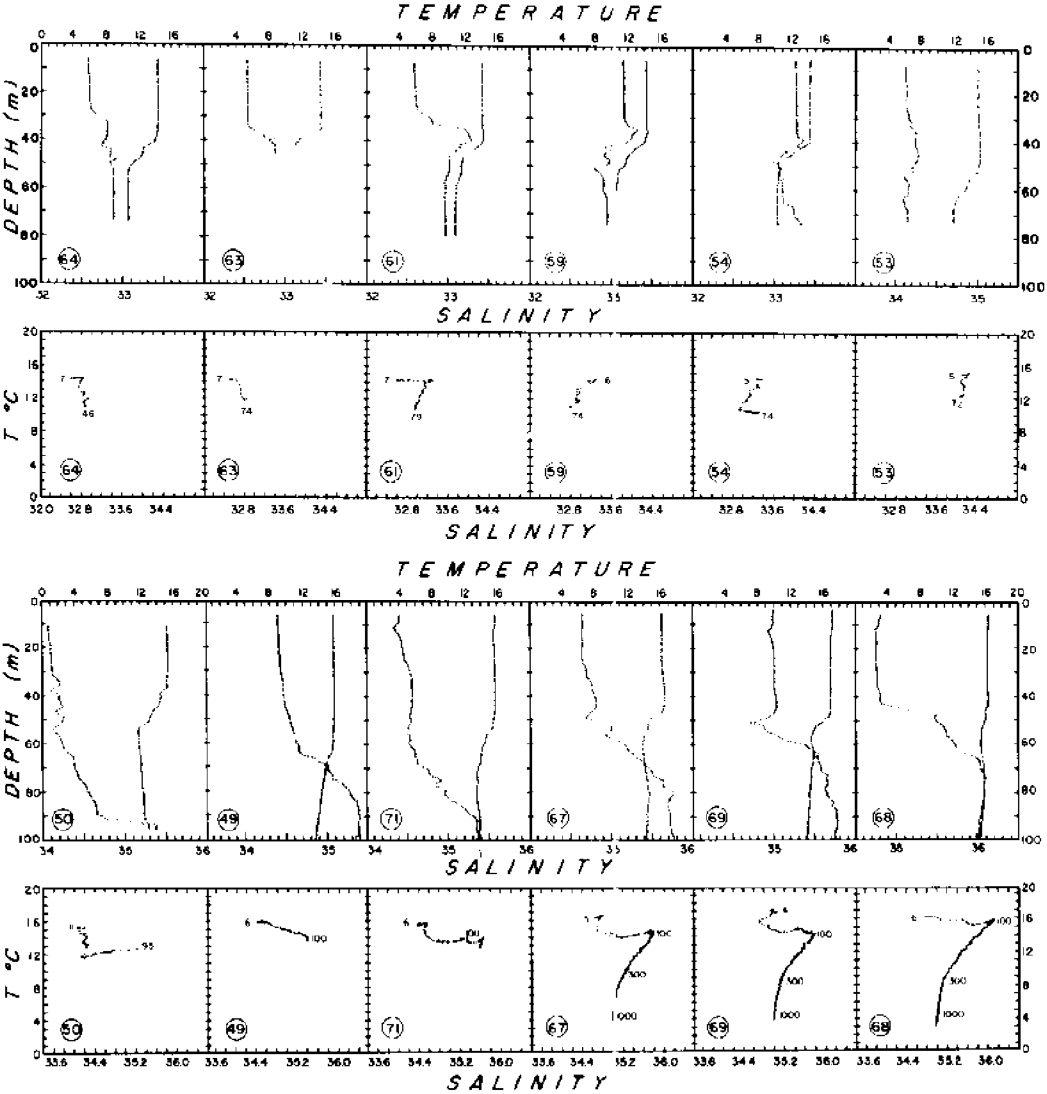

Fig. 3. Profile II individual station curves for temperature and salinity versus depth and the relation of temperature to salinity. The depth (m) of the STD point closest to the sea surface and sea floor is given on the *T-S* diagram.

to the deep water mass division and is a mixture of surface and deep components.

Pycnocline—The pycnocline is near 50 m inshore, near 70 m over the shelf break, and shallows again to near 50 m over the slope and rise (Fig. 4). The pycnocline is associated with the thermocline over the inner and outer shelf, but, supported by the salinity field, it continues seaward of the thermocline, over the slope and rise. Its deepest portion and weakest condition occurs in the transition between a thermally and a haline supported pycnocline. No intense density front is observed over the shelf break, through increased lateral gradient in σ_T is associated with the salinity front at the shelf break. The pycnocline is continuous across the area, but it is not made up of the same isopycnals. Isopycnals over the shelf and slope are inclined relative to the pycnocline. This situation may be conducive to significant transpycnocline

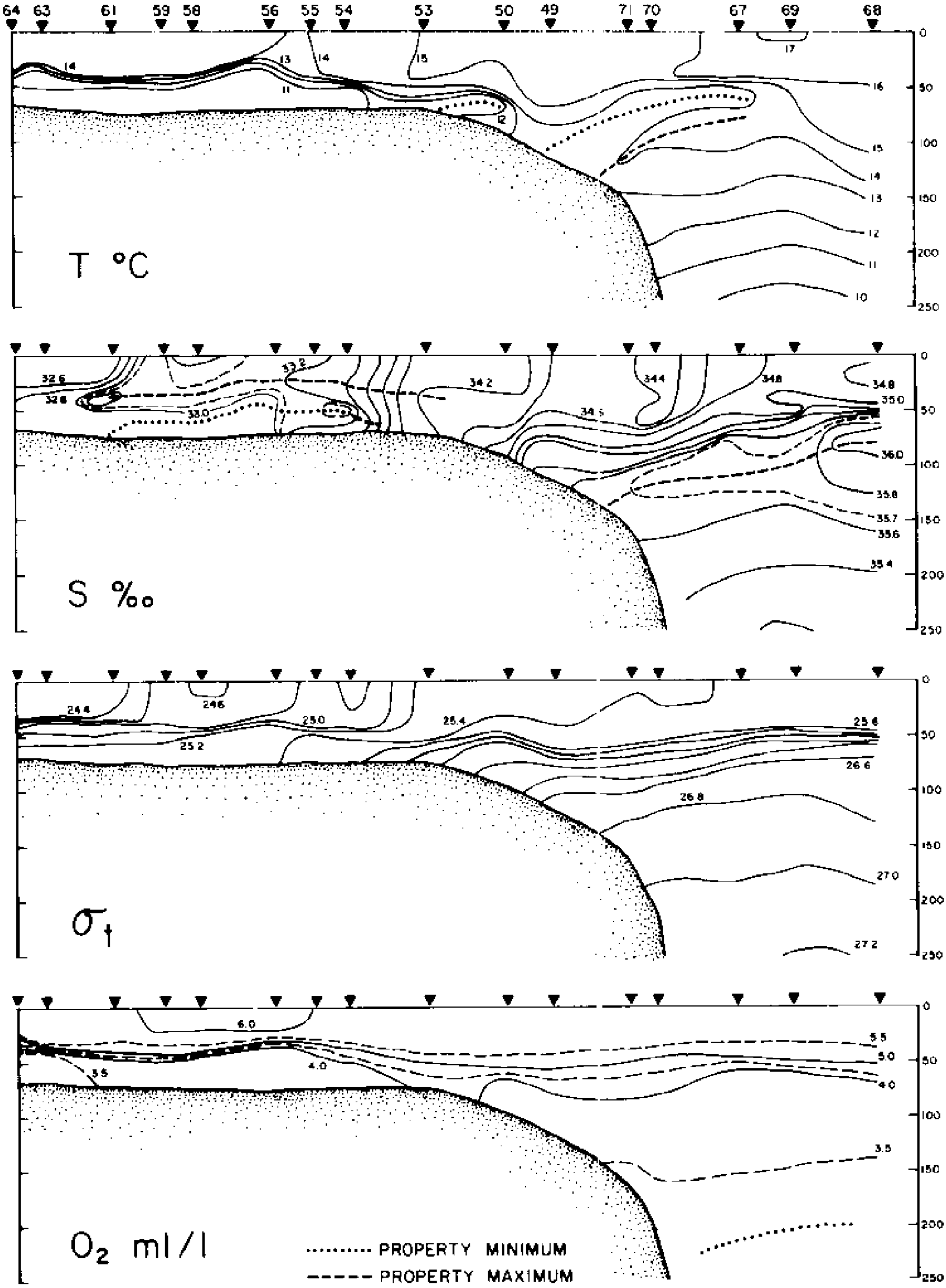

Fig. 4. Temperature, salinity, density, and oxygen distributions along profile II.

transport of properties by stirring along isopycnal surfaces. The relation between isopycnals and pycnocline is a product of the estuarine character of the surface water. The thermocline is strong and more or less horizontal, but the river discharge induces steeply inclined isohalines.

Oxygen-cline (*Fig. 4*)—Surface oxygen concentrations are near saturation values (5.5–5.7 ml/liter) (Fig. 5a) for the observed temperature. Below the pycnocline level, oxygen concentration decreases to values <4 ml/liter (below 60% saturation). The shelf water pycnocline is seasonal and cold water below the pycnocline is believed to be a residue of the winter period, so the relatively low subpycnocline oxygen level suggests significant changes of oxygen levels on the shelf floor. The saturation value of about 7.3 ml/liter (saturation for 5°C water, typical March bottom temperature) is expected for winter water; a decrease of nearly 5 ml/liter occurs from late March (when the stratified summer regime begins) to October—about 6 months.

Subpycnocline water over the slope is also low in oxygen relative to the surface, but it is associated with the upper layers of the deep water division. Below 200 m, oxygen concentrations increase with depth.

Thermohaline stratification of surface waters

The family of temperature and salinity traces versus depth and their corresponding *T*-*S* curves (Figs. 3, 6, 7) reveal a complex stratification within surface waters over the shelf and slope. Inspection of these curves shows that, once removed from the inner shelf, a simple estuarine stratification is altered by an intermediate salinity extrema. The transition from estuarine character is often quite abrupt, as can be seen for station pairs 61 and 59 (Fig. 3) and 31 and 22 (Fig. 6).

The absolute values of salinity increase markedly with increasing distance from the coast, yet a basic vertical structure of the surface water division is maintained to varying degrees of complexity (Figs. 2 inset, 3, 6, and 7).

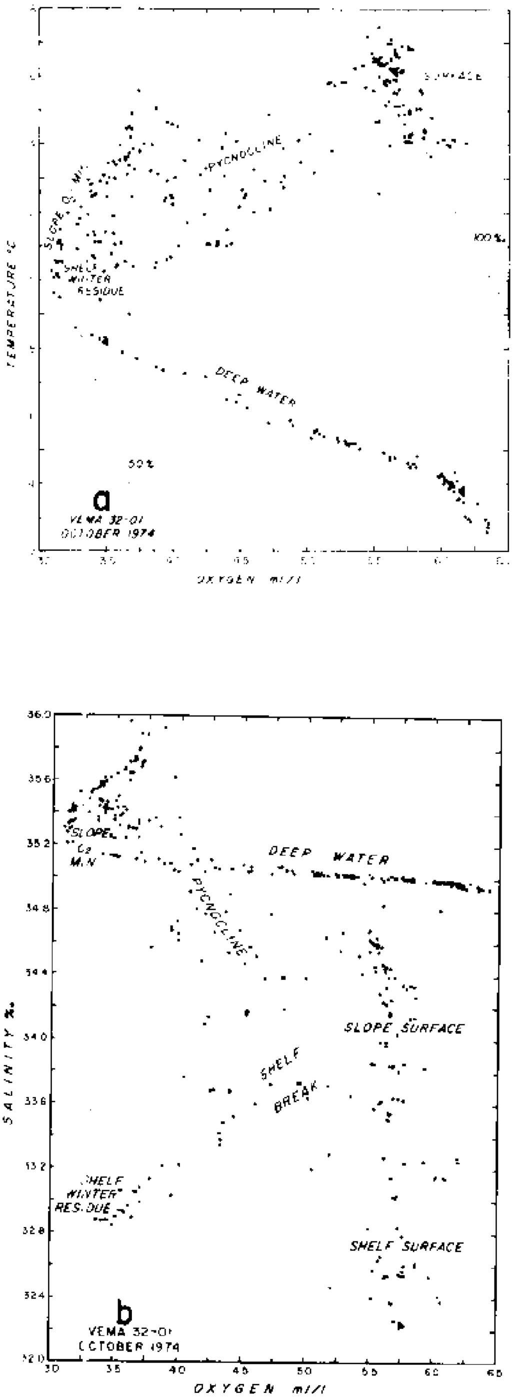

Fig. 5. a. Group temperature-oxygen scatter from water bottle-reversing thermometer data. The 50% and 100% oxygen saturation curves are calculated using the equation given in UNESCO (1974) tables. b. Group salinity-oxygen scatter from water bottle data.

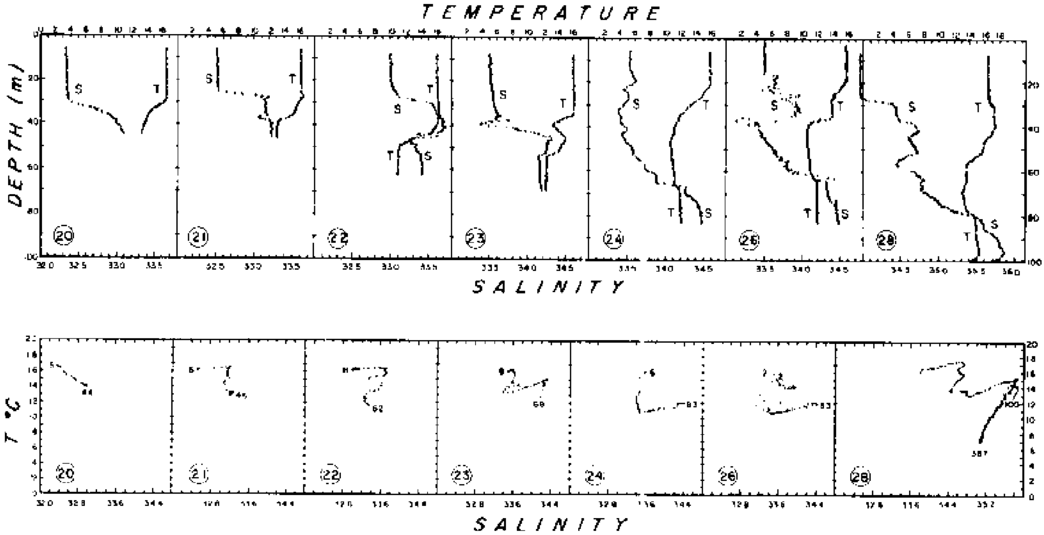

Fig. 6. Profile I. Same as Fig. 3.

Layer 1 (*surface*)—an upper isohaline-isothermal layer with a salinity maximum of variable thickness in a lower isothermal stratum (often a slight temperature rise accompanies S-max). The family of STD traces and their T-S curves indicate that above the shelf pycnocline temperatures show little structure, other than a slight increase associated with the salinity increase observed within the lower part of layer 1. The S-max is most strongly developed over the middle shelf (Fig. 3). Over the outer shelf and shelf break vertical salinity gradient is weak. The group T-S diagram (Fig. 2) shows the cumulative effect of layer 1 is a relatively dense scatter of points span-

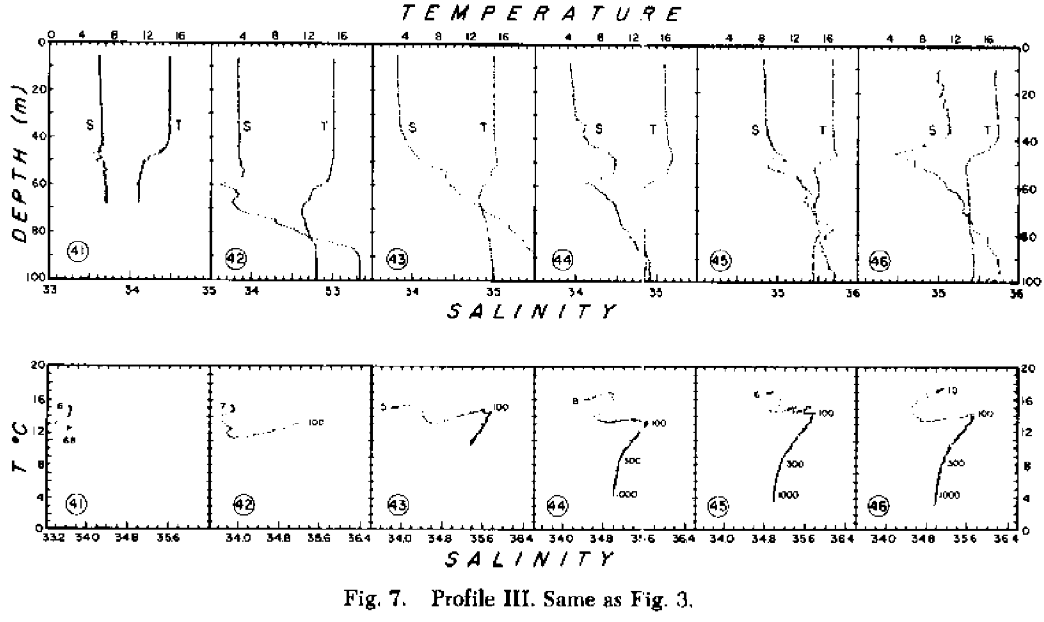

Fig. 7. Profile III. Same as Fig. 3.

ning a curve from 14°C, 32.4‰ to 17.5°C, 35.2‰. Stations 19–24 are exceptions which were obtained at the beginning of the cruise and occupy the southern part of the data set. However, the *T-S* expression of layer 1 is maintained by these stations.

At salinity greater than 34.3‰ (near the shelf break), the progression of curves for layer 1 indicates increased temperature rise between stations. This warming trend may be a result of decreasing autumnal cooling with distance from the coast.

Extrapolation seaward of the *Vema* data of the layer 1 *T-S* trend shows it would enter the domain of North Atlantic central waters. The slope layer above the pycnocline is most likely a mixture of the layer 1 stratum over the continental shelf with surface water of the North Atlantic central water mass (*see* Ford et al. 1952). The central water mass, found east and south of the Gulf Stream, probably enters the slope water region by Gulf Stream shedding of warm anticyclonic eddies.

The character of layer 1 *T-S* distribution suggests that water within the shelf S-max at a particular station is similar to the water near the surface over the outer shelf and shelf break. This indicates that water composing the S-max stratum contains outer shelf or possibly shelf break near-surface water that has spread inshore just above the pycnocline to become mixed with shallower, lower salinity water en route. The surface water of layer 1 may represent seaward spreading of river components which mix with deeper saline waters en route.

Layer 1 may therefore possess a cross-shelf circulation pattern that bears some resemblance to an estuarine pattern and extends to the salinity front at the shelf break (Fig. 8). Naturally, shoreward salt flux across the shelf break is necessary to balance the introduction of river water into the surface layer. The nature of this transfer across the shelf break is not clear, though a diffusive flux across the shelf break salinity front may be responsible.

Surface water sinking over the outer shelf will increase in salinity by lateral diffusion and then spread just above the pycnocline with a coastward component. In the steady state situation the balance of lateral salt diffusion with vertical advection of salt would be

$$K_x \frac{\partial^2 S}{\partial X^2} = w \frac{\partial S}{\partial Z}.$$

Using the layer 1 salinity data for station pairs 56-54 and 54-53, we determine a value of 4.4×10^8 for the ratio K_x/w. This is not unreasonable for typical vertical velocities of 10^{-2} to 10^{-4} cm/s.

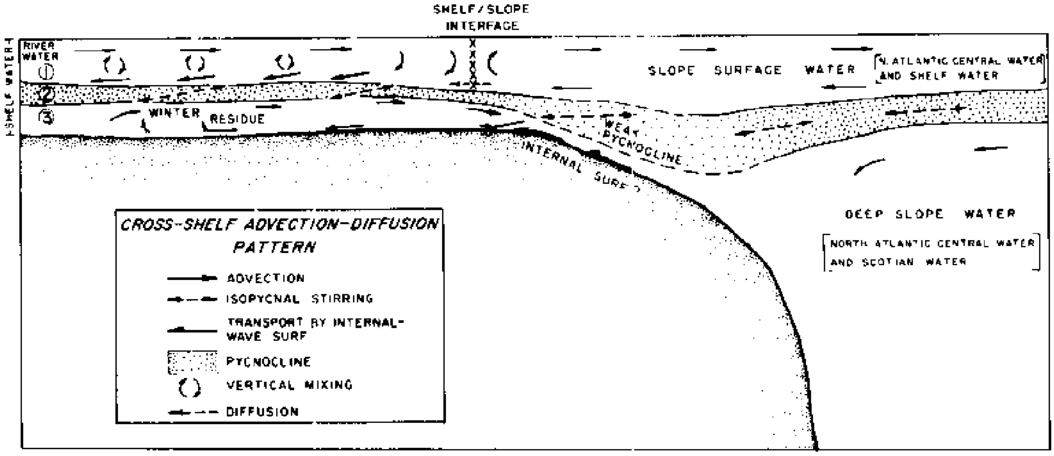

Fig. 8. Schematic of cross-shelf advection-diffusion pattern that may be responsible for the observed thermohaline stratification noted in October 1974. Isopycnal stirring refers to spreading of characteristics on isopycnal surfaces.

Layer 2 (*pycnocline*)—decreasing salinity within the thermocline; occasionally this T-S relation is repeated yielding a picture of interleaving stratification. The shelf water thermocline is coincident with the pycnocline. Within the thermocline, salinity decreases with depth (except for the inner shelf stations) but frequent reversals in salinity gradient are apparent. They are associated with variations in the temperature gradient, so that layers of weakened temperature gradient within the thermocline are accompanied by a rise in salinity. The most significant salinity decrease is marked by relatively intense thermocline layers. The cold, low salinity strata within the shelf pycnocline layer may be components of the middle shelf segment of the winter residue shelf water (of layer 3). Elements of this water within the shelf pycnocline could result from isopycnal spreading (within the 25.0–25.3 σ_T interval), which can transport the winter water seaward and into the pycnocline. This would be a consequence of the isopycnal-pycnocline relation noted above, i.e. isopycnal surfaces tilt downward toward the shore relative to the pycnocline. Isopycnal eddies would be effective agents of isotropic stirring (rather than mixing which implies entropy changes: Eckart 1948; McLellan 1957), whenever significant gradients of temperature and salinity occur on a density surface.

Other examples of isopycnal spreading into layer 2 may be the penetration of the layer 1 S-max stratum along the 24.8 σ_T surface into the upper part of layer 2 (Fig. 4; station 61, Fig. 3), and the seaward extension, as a temperature minimum, of the outer shelf winter residue water into the slope pycnocline (Fig. 4; station 67, Fig. 3) along the 26.0 σ_T surface.

Many of the continental slope stations show an intense salinity minimum in the lower part of the thermocline (stations 46 and 69 are particularly good examples, Figs. 3, 7). The thermocline over the slope is embedded within the upper part of the salinity controlled slope pycnocline. The thin, low salinity stratum is shallower than the continental slope temperature minimum and may have a different origin than winter water.

Layer 3 (*subpycnocline, shelf only*)—a salinity minimum at the base of the thermocline with salinity increase to the shelf floor. Over the continental slope, shelf layer 3 characteristics are embedded within the pycnocline, below which salinity increases to the S-max of the deep water division. The relatively cold water below the shelf pycnocline extends to the outer shelf (to the 80-m isobath) with the temperature minimum at the bottom. Seaward of the 80-m isobath the cold layer is displaced from the bottom by a warmer more saline layer. The cold lower layer may be the winter homogeneous condition. It stands out in sharp contrast to the overlying waters, separated by the pycnocline that effectively protects the lower layers from vertical mixing, though it may be vulnerable to isopycnal stirring.

The seaward extension of the T-min is characterized by dissociated sections in blobs. Cresswell (1967) noted the possibility of calving of the cold shelf water into the slope water. On the basis of BT and hydrographic station data, Wright (1976) found that the interface of shelf to slope waters (which he took as the 10°C isotherm) usually intersects the bottom within 10 mi of the 100-m isobath and that detached parcels of coastal water occur in the slope water at all times of the year, with the maximum in late summer.

Numerous *Vema* stations over the slope show no T-min (stations 35–38, 44, 47, 48, 72–74); all but one (44) occur in or very close to the Hudson Canyon. This suggests that the flow over the canyon is predominantly landward. This is further supported by the temperature and salinity distribution at 10 m (Fig. 9) and the dynamic topography of the sea surface relative to the 40-db level (Fig. 10), which shows landward penetration of the warmer, saltier slope waters over the canyon. In addition, a 6-week current meter record (September–October 1974) (Amos in prep.) indicates a mean up-canyon flow.

The source of the relatively warm saline bottom waters found over the outer continental shelf appears to be an intrusion of

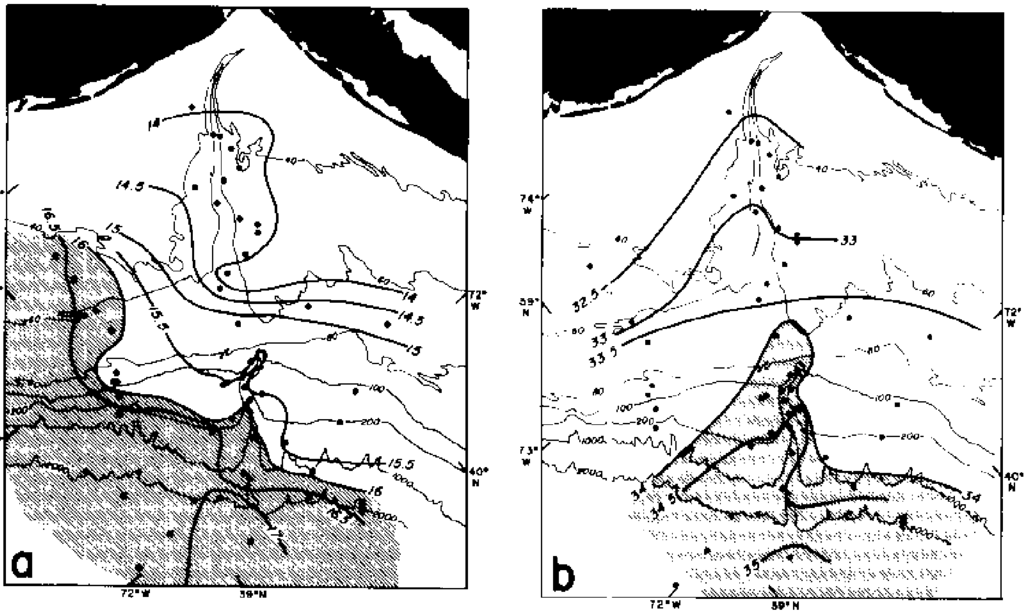

Fig. 9. Temperature (a) and salinity (b) distributions in the 10-m depth surface.

slope water. A particularly fine example of this process is shown by station 50 (Figs. 2 inset, 3). Below 54 m salinity and temperature increase with depth, with accelerated increase in the lower few meters (salinity increase by 0.75‰ in 5 m). This sudden change of gradient may be a product of a recent intrusion, confined to the shelf floor, which has not yet mixed vertically into the shelf water column. A sporadic intrusion of saline water along the outer shelf floor is suggested by action of an internal surf (perhaps a product of the open sea internal tide), though a less dramatic process may be responsible for most of the slope water intrusion.

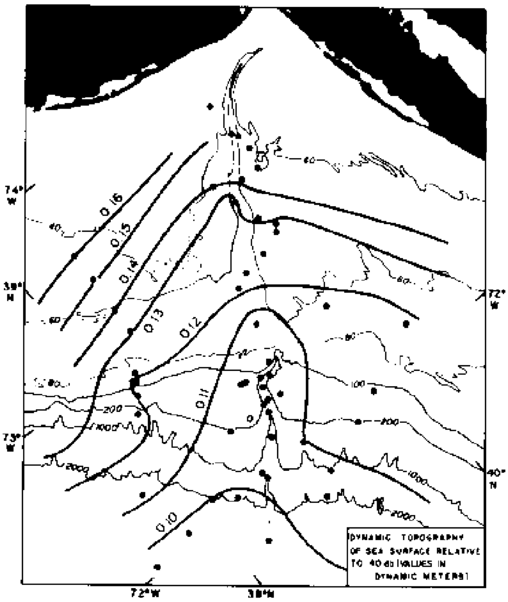

Fig. 10. Dynamic topography of the sea surface relative to the 40-db isobaric surface.

The *T*-*S* characteristics of layer 3 over the shelf and within the halocline over the slope (Fig. 2) appear as a simple mixing curve between the lower salinity shelf water and the top of the "undisturbed" open ocean *T*-*S* curve (appears as a salinity maximum in Fig. 4) in association with large-scale open ocean features.

Intrusion of warm saline slope water onto the shelf would displace or dilute the winter residue water or both. The residue water would also be warmed by vertical mixing and isopycnal stirring. Inspection of the monthly bottom temperature charts prepared by Walford and Wicklund (1968) shows the New York Bight cold shelf water and decrease in areal extent from April (when it first appears as a thermal feature detached from the bottom temperatures along the coast) to November when it disappears, probably because of autumnal overturning.

Mixing of the residue water with slope water would decrease the oxygen concentration of the shelf bottom water, since the slope water components, derived from the slope water pycnocline and possibly subpycnocline waters (by internal surf process: Fig. 5a, b), are lower in oxygen than the expected initial oxygen concentration levels of the residue water [oxygen saturation for shelf bottom waters for March (5°C water) is 7.3 ml/liter]. Therefore the low October 1974 oxygen levels of the residue water may not be solely a product of in situ consumption. The *T-S* relations suggest that the October 1974 residue water would be about 50% diluted with slope water (assuming 5°C, 32‰ initial winter water conditions mixing with 14°C, 35‰ slope water), in which case an oxygen level not lower than 6 ml/liter (assuming 7.3 ml/liter initial winter water condition with a 4 ml/liter oxygen concentration for the slope pycnocline component) would be found on the shelf if no in situ oxygen consumption were occurring. Since the winter water in October 1974 is 3.4 ml/liter, in situ consumption must be a significant factor. If the oxygen level of layer 3 water is decreased to 6 ml/liter by mixing, then an in situ consumption rate of 2.6 ml/liter in a period of 6 months must occur.

Deep stratification

Below S-max, all stations have similar *T-S* curves (Fig. 2). The *T-S* curve is not linear below S-max. Salinity drops off more rapidly from S-max to the 8°C isothermal surface, below which the gradient decreases. A small reversal of the salinity gradient occurs within the 1,000–2,000-m interval, followed by decreased salinity as the sea floor is approached.

At least four subdivisions seem to compose the deep water column below S-max. These are, as labeled in Fig. 2:

A. North Atlantic central water component (or deep slope water)—rapid decrease in salinity and temperature between S-max and the 8°–9°C isotherm within the stratum from 100–300 m.

B. Irminger Atlantic water—somewhat decreased gradients between the 8°–4°C isothermal surface near 1,000 m.

C. Mixing curve between Labrador Sea water (S-min) and Mediterranean overflow (S-max)—reversed salinity gradient between 1,000–2,000 m, which induces a secondary salinity minimum near 1,000 m, a secondary S-max near 2,000 m.

D. North Atlantic deep water (NADW)—the abyssal layer below 2,000 m.

The characteristics of the water column below S-max represent influences remote from the New York Bight. The warm, salty upper layer must be derived in part from waters carried northward by the Gulf Stream, i.e. North Atlantic central water. It may not be pure North Atlantic central water, in that it falls at the low salinity limits of the gross *T-S* scatter of the central water mass (Sverdrup et al. 1942: figure 183). It is more likely, as suggested by McLellan (1957), that the upper layer results from introduction of central water into the slope region by action of the warm anticyclonic Gulf Stream eddies, which have frequently been observed north and west of the warm-wall of the Gulf Stream (Saunders 1971; Gotthardt and Potocsky 1974). The warm eddies migrate toward the southwest embedded in slope waters derived from the Nova Scotian margins. McLellan (1957) attributed the Scotian slope water below S-max (deep slope water) to a mixture of central water and Labrador Sea near-surface water (*see also* Fisher 1972, 1973). Scotian slope water may then be enriched by mixing with the warm eddies in the central water component as it flows southward along the continental margins.

Anticyclonic eddies have been observed to coalesce with the Gulf Stream north of Cape Hatteras (Gotthardt and Potocsky 1974), but some mixing of the warm water with surrounding slope waters must occur. One set of observations (9–14 August 1974, at 38°N, 73°W: Corton 1974) shows the warm water of an anticyclonic eddy covered by low salinity slope surface water. We suggest that overriding of the slope surface water of the eddies and the resultant mixture of central water in the eddies with Scotian waters gives rise to the deep slope water layer (S-max to the 8–9°C level).

On occasion an unmixed anticyclonic eddy is observed directly off the New York Bight (Bowman and Duedall 1975). This was not the case at the time of the *Vema* cruise of October 1974 (Gulf Stream report for October 1974). If an anticyclonic eddy had been observed, the 15°C isotherm would be depressed below the 200-m level.

The second layer of the deep water column is directly influenced by water mass advection from the northern North Atlantic. The *T-S* relation of water within the stratum between 4°–8°C isotherms is similar to Irminger Atlantic water (Smith et al. 1937; Lazier 1973) and so may be derived directly from the northern fringes of North Atlantic central water, south of Iceland, without significant alteration.

The *T-S* scatter near 8°–9°C (near 300 m) falls into two groups: one has lower salinity (35.10–35.15‰) and a more pronounced gradient change in the *T-S* curve, and the other has a slightly higher salinity (35.20–35.25‰) and a weaker gradient change. This second group has higher salinity at all temperature levels below 11°C, though the maximum separation is at the 8°–9°C level. Inspection of individual stations for salinity at the 8.5°C level shows that stations 74, 48, 47, and 72 comprise the higher salinity group two. All are located in the upper part of the Hudson Canyon. Station 37, in the same position as station 48, is closer to the higher salinity end of group one. It is possible to explain the special nature of the upper canyon station by limited inflow of the low salinity component at temperatures below 11°C (derived from northern waters), but a more likely explanation is enhanced vertical mixing between the deep slope water and Irminger water within the canyon due to the added boundary induced turbulence by the canyon walls and possibly a longer residence time of canyon water.

The S-min near 1,000-m (34.979‰, 4.14°C at 1,150 m at station 46) is noted throughout the western North Atlantic and is attributed to the influence of Labrador Sea water (Amos et al. 1971; Worthington and Metcalf 1961; Lazier 1973), which at the source is characterized as 3.4°C, 34.9‰.

The secondary S-max near 2,000 m (34.985‰, 3.69°C near 1,750 m at station 46) is North Atlantic deep water—more likely the upper component derived from the Mediterranean overflow in view of the high temperature (3.55° potential temperature). Worthington and Wright (1970) showed that the high salinity on the 3.6°C potential temperature surface spreads westward from the Mediterranean. Therefore the fringes of the upper NADW apparently reach the continental margins off the New York Bight.

The water column below the secondary S-max decreases in salinity and temperature: the bottom water at station 46 is 2.80°C (potential temperature), 34.950‰ at 2,454 m. This is slightly warmer and saltier than the northern component of NADW, so it is probably above the main core of the Greenland-Norwegian Sea overflow water. Reid and Lynn (1971) indicated that the core of the northern components of NADW is near 4,000 m at 40°N: therefore the main axis of the deep western boundary current is seaward of station 46 and the *Vema* data set.

Conclusions

The pattern of the advection-diffusion cross-shelf component suggested in the study (Fig. 8) is one of estuarine circulation in the layers above and below the pycnocline. Therefore the low salinity surface water and low salinity subpycnocline water are replaced by landward transport of more saline waters along the top of the pycnocline and sea floor, respectively. Contact across the pycnocline may be established by isopycnal stirring.

The major advection of water is not cross-shelf but rather along the shelf, generally directed toward the southwest in the New York Bight (Bumpus 1973). The smaller cross-shelf component would be superimposed on this pattern. The Hudson Canyon apparently disturbs the usual slope oceanography in that landward transfer of oceanic water occurs over the canyon. This is evident from the sea surface dynamic topography (Fig. 10). In addition, vertical mixing may be enhanced within the confines of the canyon.

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Distribution of hydrographic properties in the New York Bight apex[1,2]

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Abstract

Seasonal hydrographic cycles are interpreted through a series of isometric block diagrams and planimetric projections illustrating the distributions of temperature, salinity, and density in the bight apex. Seasonal cycles are typical of those found in coastal seas near river mouths in drowned river valleys in temperate latitudes. The prevailing southwest coastal drift, the right-angle bend of the coastline, seasonal and short term wind patterns, and the presence of the Hudson Shelf Valley, all influence the nearshore circulation.

The New York Bight apex, bounded by New Jersey and the south shore of Long Island, 73°30′W long to the east and 40°15′N lat to the south (Fig. 1), is an area with complex hydrographic properties. The combined outflows of the Hudson, Raritan, and other rivers flow into the northwest corner of the apex, where the coastlines form a right angle. The depth of the water within the apex is usually less than 30 m, except inside the Hudson Shelf Valley, an extension of the Hudson estuary channel, where the bottom drops sharply to over 60 m near the southern boundary of the apex. The apex has an active circulation as indicated by the salinity distribution, which is nearly as high as that found in adjacent coastal areas, except during periods of high river runoff.

Hydrographic properties of the New York Bight apex exhibit clear seasonal cycles of temperature, salinity, and density. This paper interprets these seasonal patterns. Isometric block diagrams, viewing the apex from space looking to the northwest and with bottom water properties plotted on planimetric projections, are used to show temperature, salinity, and density distributions spaced over the five hydrographic seasons of the year.

A comprehensive review of the hydrographic properties of the greater New York Bight is presented elsewhere (Bowman and Wunderlich 1976).

Previous studies

Ketchum et al. (1951) made the first major investigation of the seasonal hydrographic distributions of the apex. By volumetric analysis the residence time of freshwater was found to be about 6–10 days in spite of almost ninefold variations in river flow. The apex was shown to have a relaxation time of about 2 days after the disruption of the basic circulation by a major late winter storm.

Studies by the National Fisheries Service (1972; Charnell and Hansen 1974; Fig. 1) confirmed that waters of the apex are stratified for about 9 months a year; the waters are dominated by river runoff in spring (Fig. 2), followed by formation of a strong thermocline during summer. The preferred path of the Hudson plume is generally parallel to the New Jersey shoreline. By contrast, east of the axis of the Hudson Shelf Valley, the apex water is predominantly shelf-oceanic.

Further sampling was continued by the Marine EcoSystems Analysis Program (MESA) of NOAA (Hazelworth et al. 1974, 1975*a,b*) which also supplemented these hydrographic stations (Fig. 1) with a number of current meter stations. Evidence was found of a slow (4–10 cm s^{-1}) nontidal anticyclonic (clockwise) circulation during late summer and early fall 1973.

The residual surface and bottom drift along the continental shelf of the Middle Atlantic Bight, including the New York

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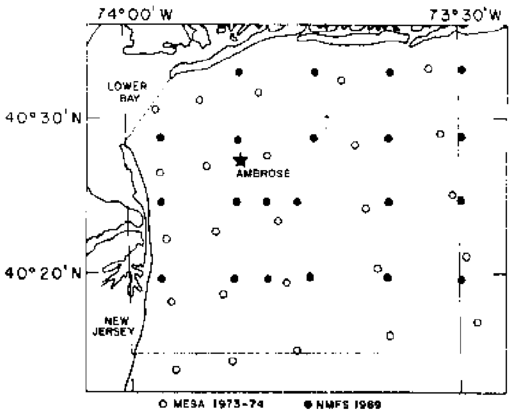

Fig. 1. Location map of the New York Bight apex and hydrographic sampling stations. The inner border represents the limits of the isometric block diagrams and bottom maps (*see Figs. 4–8*).

Bight, has been studied by Bumpus (1965, 1973) and Bumpus and Lauzier (1965). The coastal current is predominantly southwest and southward along the continental shelf. Reversals can be expected between April and September during years when river runoff is abnormally low (Bumpus 1969).

Charnell and Hansen (1974) and Hardy and Baylor (1975) studied the residual surface drift in the apex and found correlations between high and low frequencies of drift card strandings and prevailing northwestward to northeastward summer winds and southward to southeastward winter winds, respectively.

Residual bottom drift along the Hudson Shelf Valley in the apex is generally toward the mouth of the Hudson estuary (Bumpus 1965; Charnell and Hansen 1974; Hardy et al. in press). No evidence of an anticyclonic gyre in the apex was found by Hardy et al. (in press); rather they found a divergence on either side of the valley with bottom drift east of the valley northwest to northeast toward Long Island and west of the valley southwest to northwest toward New Jersey. It was not possible to correlate seabed drifter returns with upwelling (Mar. Sci. Res. Center 1973) or with other specific wind events.

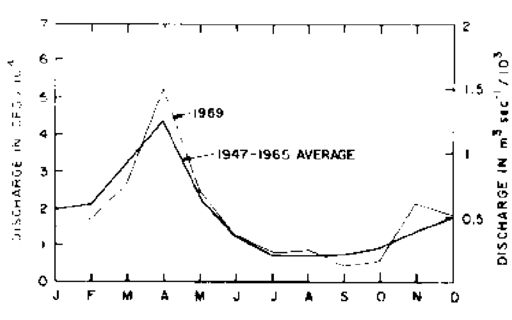

Fig. 2. Estimated mean monthly discharge of the Hudson River at Poughkeepsie for the water years 1947–1965 and the calendar year 1969 (after Giese and Barr 1967).

Kao (1975) investigated the nontidal circulation in the lower bay across the Sandy Hook–Rockaway transect and confirmed the presence of a strong estuarine circulation (Fig. 3). For the period 2–7 June 1952, he found an upstream flux at depth of 1,450 $m^3 s^{-1}$ and a seaward surface flow of 3,370 $m^3 s^{-1}$.

The hydrographic cycle

Winter conditions (*January, February, March*)—River runoff into the apex is low in winter (Fig. 2). Strong vertical mixing causes an almost unstratified water column over most of the region. Temperatures drop to their annual minimum, often less than 2°C in mid-January, and are coldest nearest the coast and inside the apex. These winter minima persist into late February or early

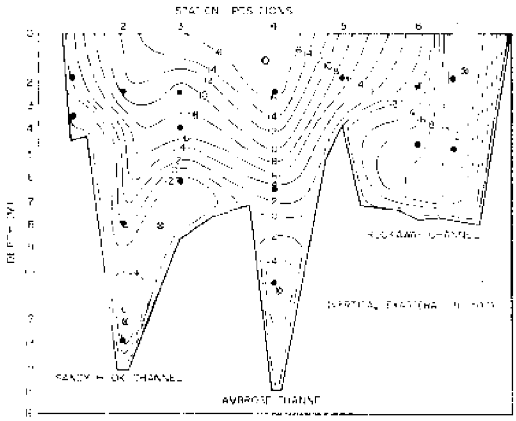

Fig. 3. Averaged nontidal velocities normal to the Sandy Hook–Rockaway sections (*see Fig. 1*) for 2–7 June 1952, looking upstream (ebb velocities positive). Closed circles represent positions of the current meters (after Kao 1975).

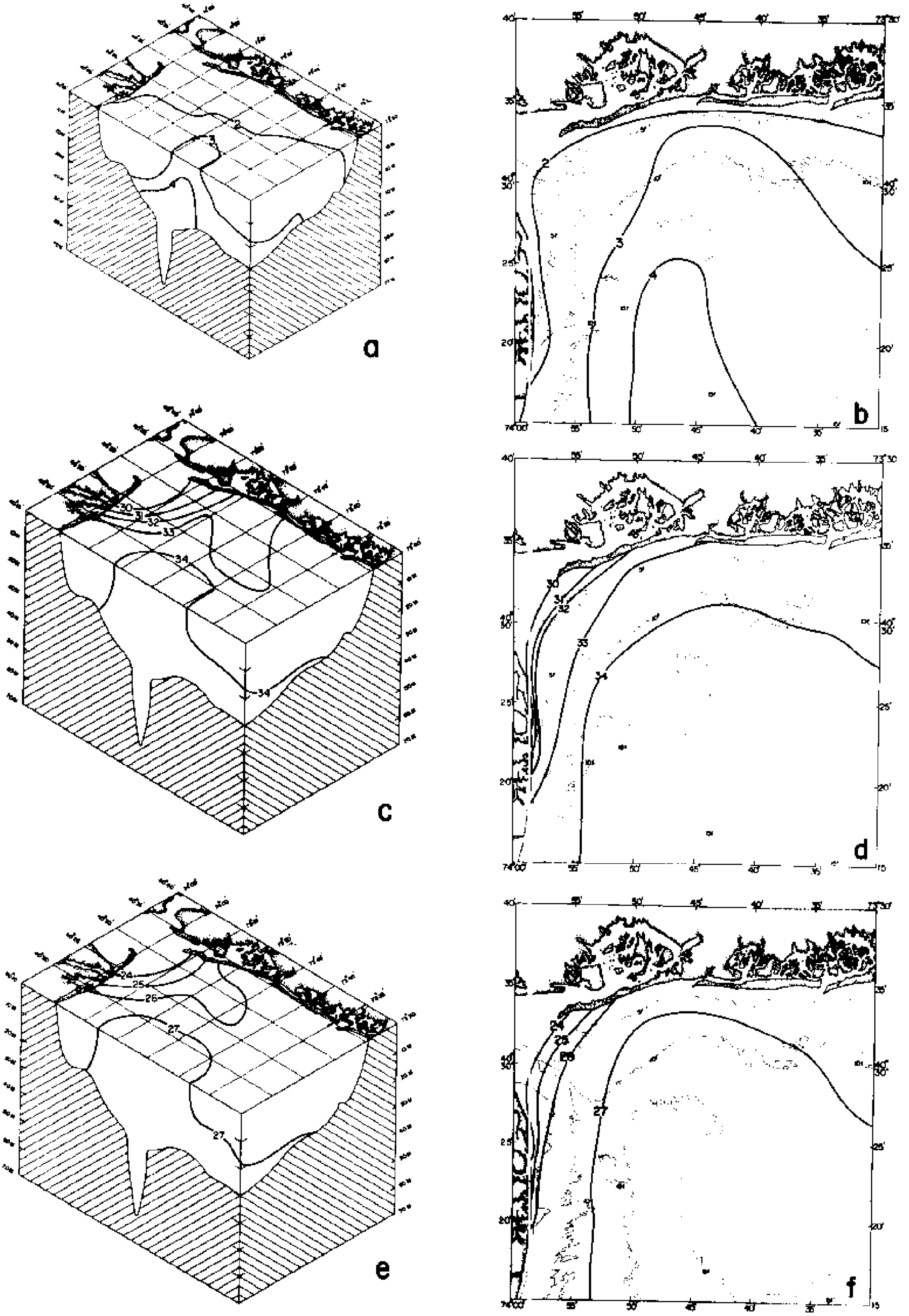

Fig. 4. **Hydrography**, 6–7 February 1969 (cruise 1: Natl. Mar. Fish. Serv. 1972). a,b—Temperature, °C; c,d—salinity, ‰; e,f—density, σ.

March. Successive isotherms at given depths roughly parallel the coastline.

Although the water column is nearly homogeneous, the apex is rarely completely isothermal. Bottom temperatures in November through February tend to be slightly higher than surface temperatures as vertical mixing is unable to keep pace with rapid cooling at the air–sea interface. An intrusion of warm bottom water up the Hudson Shelf Valley is suggested by the presence of 4°C water (Fig. 4a, b).

In January, salinities increase with distance from the mouth of the Hudson estuary from ≃30‰ to their annual maxima of greater than 34‰ in the outer apex (Fig. 4c). Near-bottom isohalines parallel the coastline and are greater than 34‰ over most of the region (Fig. 4d). Salinities begin to slowly decrease in March as river discharge increases; values of 29–32‰ occur over most of the apex.

In winter, density closely follows the distribution of salinity; in January and February density ranges from $\sigma_T \simeq 24$ near the mouth of the Hudson, to its annual maximum of $\sigma_T \simeq 27$ near the outer limit of the apex (Fig. 4e). The density field in the eastern regions remains almost unstratified through March, but the Hudson plume leads to weak vertical gradients in the western apex. Contours of σ_T closely follow bottom salinity isopleths, generally paralleling the coastline (Fig. 4f).

Spring progression (April, May)—Spring months are a transition period; the distribution of apex properties changes from being runoff-dominated to being heat-dominated. Vernal warming in April causes surface temperatures of ≃7–8°C (Fig. 5a); bottom temperature usually remains <4°C except near the coast (Fig. 5b; dashed lines shown on this and later figures represent extra contours added to incease resolution). Strong thermal stratification begins to appear in May as the seasonal shelf thermocline develops. Surface temperatures rapidly rise to ~9–11°C over most of the apex; bottom waters below 30 m remain <4°C.

Water in the apex is strongly influenced by the Hudson River spring runoff, resulting in large horizontal and vertical salinity and density gradients. Half the annual Hudson River runoff occurs during March, April, and May (Fig. 2); this leads to a well developed plume that spreads over most of the apex (Fig. 5c). The position of the plume is apparently sensitive to wind stress and reversals in the residual drift on the shelf; its axis is variable and can swing from a southerly set along the New Jersey shoreline to an easterly set south of Long Island. At other times the plume splits and has southerly and easterly components (Redfield and Walford 1951). The temperature and salinity stratification causes the density field to rotate from a horizontal to a vertical structure during spring months, with surface isopycnals closely following the isohalines (Fig. 5e).

During spring, bottom salinities and density generally follow the bottom contours; a very strong density gradient appears in the mouth of the Hudson estuary as the runoff peaks in April (Fig. 5f). This indicates the presence of a strong estuarine circulation in the lower estuary.

Summer conditions (June, July, August)—The thermocline appearing in May intensifies during June when surface temperatures reach 17°C at the outer edges of the apex. Bottom water temperatures remain relatively unchanged from May and are typically <6°C in the shelf valley. Since the summer thermocline intersects with the bottom, very strong bottom temperature gradients occur over much of the apex, with isotherms closely paralleling the depth contours.

Water temperatures continue to rise through June and July. Surface temperatures reach their annual maximum values, ≃26°C, in August and remain at these peak values through the month (Fig. 6a). During July and August (Fig. 6b), bottom temperatures show a steady but distinct rise to ~10°C in the shelf valley. These bottom isotherms parallel the bathymetry at all depths.

Summer salinities over the surface of the apex range from ≃25–27‰ near the apex mouth to ~30–31‰ at the southeast corner (Fig. 6c). The Hudson plume is ill-defined

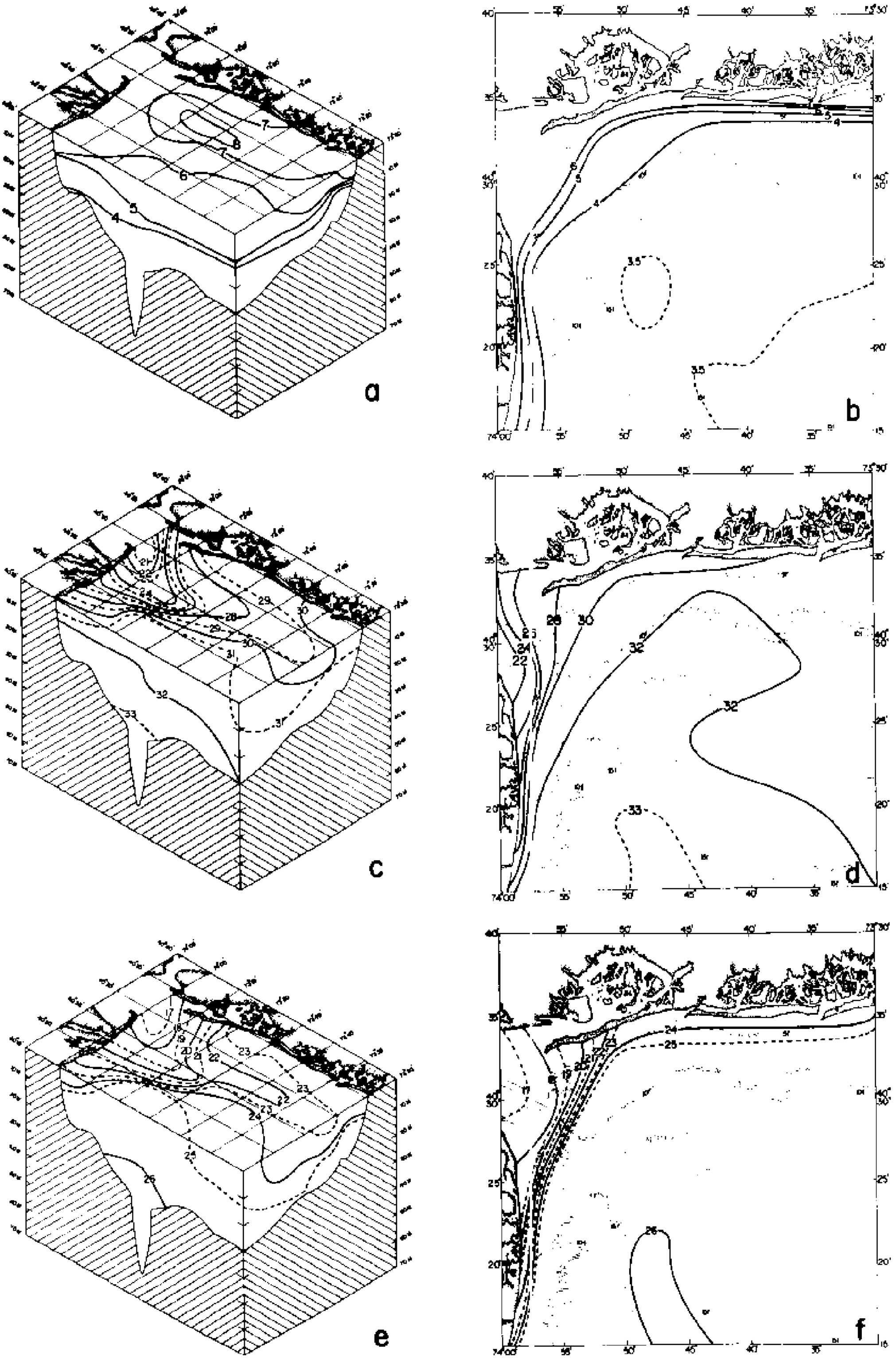

Fig. 5. Hydrography, 8–10 April 1969 (cruise 5: Natl. Mar. Fish. Serv. 1972). a,b—Temperature, °C; c,d—salinity, ‰; e,f—density, σ_T.

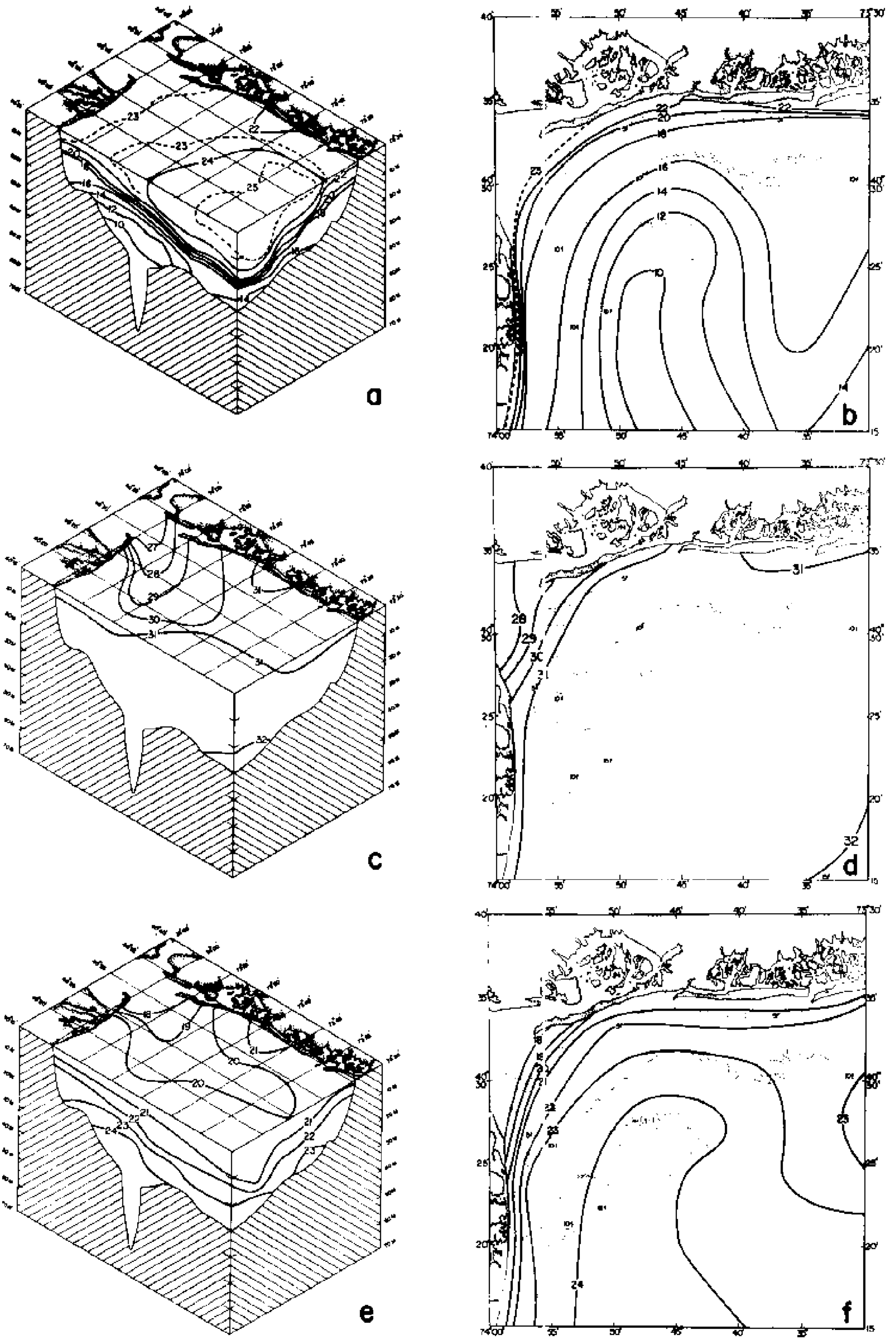

Fig. 6. Hydrography, 18–19 August 1969 (cruise 9: Natl. Mar. Fish. Serv. 1972). a,b—Temperature, °C; c,d—salinity, ‰; e,f—density, σ_t.

and has been observed to break into several isolated pools. Little vertical salinity gradient exists, but bottom salinities increase seaward, with isohalines paralleling the bottom topography near the coastline (Fig. 6d).

The transition to vertical stratification is usually complete by the end of June. From that time on, the density field is dominated by the summer thermocline except near the New Jersey coast where the Hudson plume influence is felt. Surface densities throughout the apex drop steadily to their annual minima of σ_T ≃20 in August (Fig. 6e). Vertical gradients in σ_T are at their maximum for the year over most of the apex and are typically ≃0.06–0.10 m^{-1}. Strong horizontal gradients exist along the bottom and σ_T increases rapidly from ~18 in the Hudson estuary mouth to greater than 24 inside the shelf valley (Fig. 6f).

Fall progression (*September, October*) —Surface cooling during early autumn begins to break down the summer thermocline at a rate determined by local wind strengths and sea surface roughness (Chase 1959). By the end of October, surface temperatures have dropped to ≃16–18°C over most of the apex (Fig. 7a). As fall progresses, heat loss and vertical overturning lowers the average temperature of the water column while increasing the temperature of the bottom water to ≃12°C inside the shelf valley (Fig. 7b). However, nearshore bottom water temperatures decline during September and October as erosion of the summer thermocline continues.

Vertical mixing during fall is often stronger than the buoyancy arising from the Hudson River runoff, except very near the apex mouth. This mixing reduces the vertical salinity gradients at a rate that varies considerably from year to year and leads to a steady increase in surface salinity, often as large as 2‰ between July and October.

Erosion of the summer thermocline during September and October is reflected in the density field; surface densities (Fig. 7c) begin to increase to σ_T ~22 due to a decrease in temperature and an increase in salinity, with bottom values inside the shelf valley reaching ≃25 in October (Fig. 7f).

Early winter conditions (*November, December*)—Vertical mixing down to about 30 m is usually complete by early or mid-November with water temperatures of 12–14°C (Fig. 8a). Bottom temperatures attain their annual maxima in this period, and the overall temperature structure is uncomplicated. Vertical mixing of the water column continues through December and temperatures rapidly decline throughout the apex to within about 2° of their annual minima.

Surface salinity increases through November and December (Fig. 8c) until the winter maximum is attained in January. This general increase in salinity of surface shelf water dissipates any isolated low salinity patches formed during summer. There is a corresponding increase in subsurface salinity, with values exceeding 33‰ in the shelf valley (Fig. 8d).

Density increases as temperatures decline and salinities increase. At the late January extrema, density values σ_T ~24 are found over most of the apex (Fig. 8e). The bottom structure is uncomplicated, and only small vertical gradients are found throughout the water column (Fig. 8f).

Discussion

Many aspects of the hydrography of the New York Bight apex are similar to those near the mouths of other large rivers in drowned river valleys, e.g. the Delaware and Chesapeake Bays (Joseph et al. 1960; Harrison et al. 1967; Walford and Wicklund 1968; Boicourt 1973; Boicourt and Hacker 1976). However, the right-angle bend of the coastline, the presence of the deep Hudson Shelf Valley, and seasonal and short term wind patterns exert important influences on the nearshore circulation.

The distribution of temperature closely follows that of other nearshore regions of the Middle Atlantic Bight, with a large range (≃25°C) between summer and winter surface temperatures. Strong bottom gradients persist through the summer over most of the apex as a consequence of the intersection of the seasonal thermocline with the bottom. The annual maxima in bottom temperatures are delayed until No-

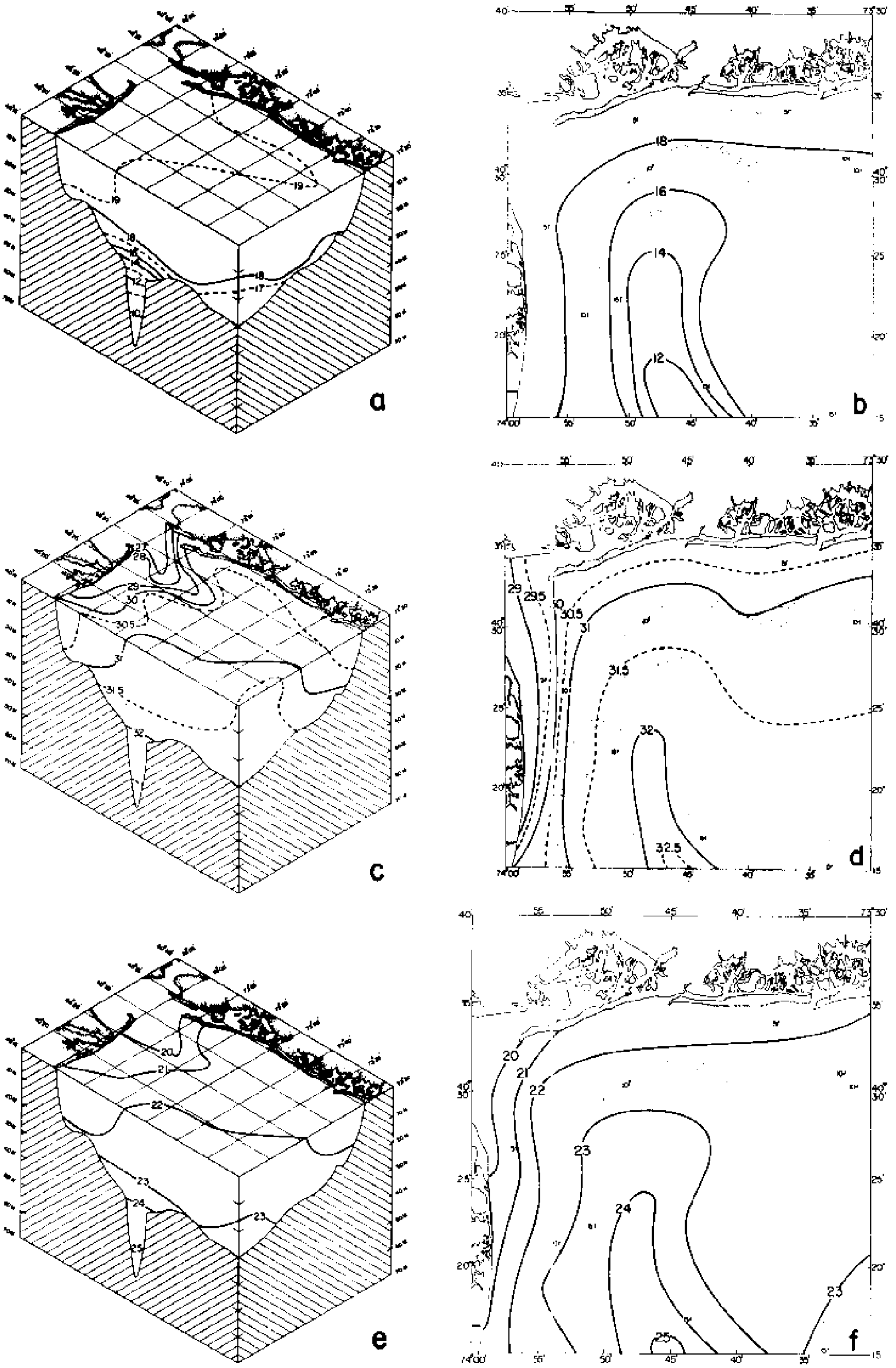

Fig. 7. Hydrography, 1–4 October 1973 (cruise 3: Hazelworth et al. 1974). a,b—Temperature, °C; c,d—salinity, ‰; e,f—density, σ_t.

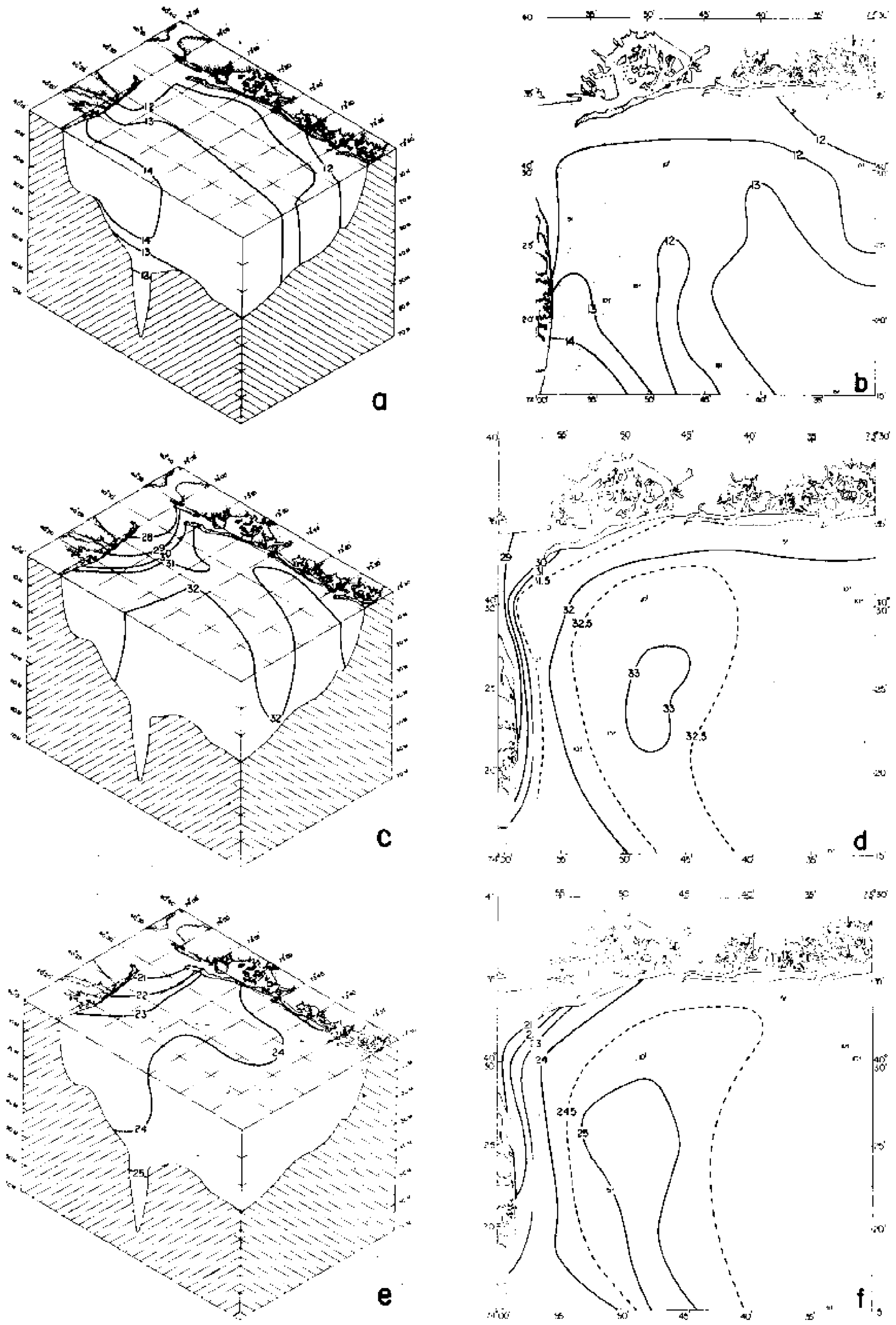

Fig. 8. Hydrography, 5–9 November 1973 (cruise 4: Hazelworth et al. 1974). a,b—Temperature, °C; c,d—salinity, ‰; e,f—density, σ_t.

vember; temperatures then drop rapidly to their annual minima in late winter.

The salinity field closely follows the annual cycle of runoff from the Hudson River; there is almost no lag between runoff and surface salinity at the Ambrose Light Tower (Chase 1969). Near the southeast corner of the apex a 2-month delay is evident, with salinities peaking in January and dropping to a seasonal minimum in June; this is more comparable to coastal conditions away from the direct influence of river outflow (Ketchum and Keen 1955; Howe 1962; Ketchum and Corwin 1964).

Density distribution closely follows the salinity field throughout the water column in winter and surface salinity throughout the entire year. Development of the seasonal thermocline during the warmer months converts the isopycnal surfaces from vertical to horizontal; vertical stability correspondingly increases during summer to its yearly maximum in August.

Previous studies (Charnell and Hansen 1974; Charnell and Mayer 1975; Hardy et al. in press; Hardy and Baylor 1975) on circulation in the apex have led to differing conclusions regarding residual drifts in the region, especially those at depth. Clarification of water movement through the apex under a variety of seasonal and meteorological conditions is critical to our understanding of dispersal of injected wastes. Especially important is a knowledge of the circulation in and beneath the summer thermocline.

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Section 3 Geological processes

Morphologic evolution and coastal sand transport, New York–New Jersey shelf[1]

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Abstract

The surface of the New York–New Jersey shelf has been extensively modified by landward passage of nearshore sedimentary environments during the postglacial rise of sea level. The retreat of estuary mouths across the shelf surface has resulted in shelf valley complexes. Constituent elements include shelf valleys largely molded by estuary mouth scour, shoal retreat massifs left by the retreat of estuary mouth shoals, and midshelf or shelf-edge deltas.

The erosional retreat of the straight coast between estuary mouths has left a discontinuous sheet of clean sand 0–10 m thick. During the retreat process, a sequence of oblique-trending, shoreface-connected sand ridges formed at the foot of the shoreface. As a consequence, the surficial sand sheet of the shelf floor bears a ridge and swale topography of sand ridges up to 10 m high and 2–4 m apart.

The mechanics of sedimentation in these two nearshore environments (estuary mouth and interestuarine coast) are now being investigated for purposes of environmental management as well as for further understanding of shelf history. In late fall and winter 1974, current meters were deployed on the Long Island coast and a radioisotope tracer dispersal pattern was traced over an 11-week period. Eastward or westward pulses of water were generated during this period of successive weather systems. Flows in excess of the computed threshold velocity of substrate materials were sustained for hours or days and were separated by days and weeks of subthreshold velocities, and the sand tracer pattern expanded accordingly. A single intense westward flow transported more sand than all the other events combined. The storm was anomalous with respect to the short term observation period, but it may in fact have been representative of the type of peak flow event that shapes the inner shelf surface.

Systematic observations of sedimentation in New York Harbor mouth have not yet been initiated. However, reconnaissance data reveal a complex pattern of ebb- and flood-dominated zones that control the pattern of sand storage.

We review in this paper our knowledge of the surface of the continental shelf off New York and New Jersey by considering two distinct topics: the geological history of this surface and the nature of sand transport across it. Our knowledge of the New York–New Jersey shelf surface is primarily the result of a decade of work by K. O. Emery and his colleagues at the Woods Hole Oceanographic Institution. A summary of this information and much more has recently been provided by Emery and Uchupi (1972). As the work of the Woods Hole group drew to a close, we attempted to consider in greater detail some aspects of the morphologic evolution of the Middle At-

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[1] Contribution of the New York Bight Project of the NOAA Marine EcoSystems Analysis (MESA) Program.

lantic Bight surface (Swift et al. 1972, 1974; Swift 1973; Swift and Sears 1974; Stubblefield et al. 1974). A summary of this work constitutes the first section of this paper.

As participants in NOAA's MESA (Marine EcoSystems Analysis) program, we have been asked not only to evaluate the geological history of the New York Bight, but also to provide quantitative estimates of sediment transport that will be of direct use to environmental managers. It turns out that these two goals are closely related. Our surveys of the shelf surface have led us to infer that it has been shaped by the landward retreat of two basic sedimentary regimes during the Holocene transgression: tide-dominated sedimentation at estuary mouths, and the sand transport pattern of the adjacent shoreface and adjacent inner shelf. Environmental engineers and managers must deal with these same regimes.

To satisfy their needs, we have initiated real-time studies of fluid motion and substrate response. State-of-the-art techniques for such studies are inadequate and progress has been slow. We report in the second portion of this paper fragments of our studies of sand transport to encourage colleagues engaged in similar studies. Our own initial experiments have raised more questions than they have answered.

Evolution of the continental shelf surface

Evolution of shelf valley complexes—The New York Bight is a pentagonal sector of the North American Atlantic shelf, extending 800 km from Cape May, New Jersey, to Montauk Point, Long Island. Off New York, the shelf is 180 km wide (Fig. 1).

The sandy shelf floor is divided into compartments by shelf valley complexes extending from the shoreline toward shelf edge canyons (Fig. 1). The most obvious elements of these complexes are the shelf valleys themselves which may appear as narrow, well defined channels (Delaware Shelf Valley; Hudson Shelf Valley) or as broad, shallow depressions which barely perturb the isobaths defining the shelf surface (Block Shelf Valley, Long Island Shelf Valley, North New Jersey Shelf Valley, Great Egg Shelf Valley). Shelf valley complexes generally contain other morphologic elements. The north rims of the shelf valleys

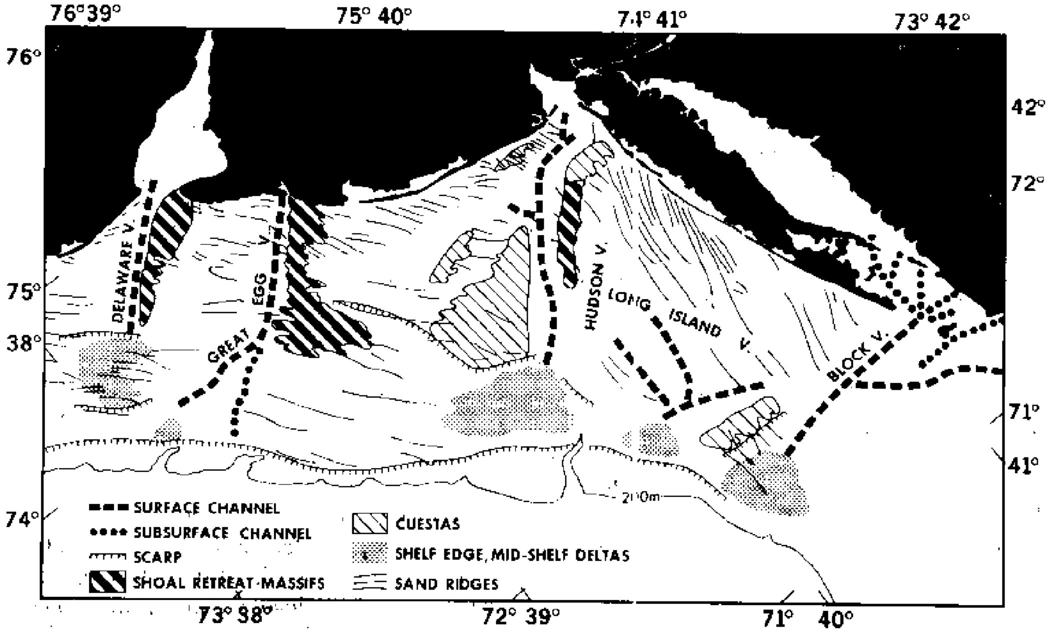


Fig. 1. Morphologic framework of the New York–New Jersey shelf. (Modified from Swift et al. 1972.)

tend to be elevated like levees above the adjacent shelf. Seaward ends of shelf valleys often terminate in delta terraces. Shelf valley complexes tend to be broken into segments by coast-parallel scarps, which may have been formed by temporary stillstands of the returning Holocene sea.

The origin of the shelf valley complexes is best inferred from the configuration of the Delaware Shelf Valley (Fig. 2), which can be traced without interruption into its modern estuary. The Delaware estuary mouth has a sill of sand nourished by littoral drift from the New Jersey coast (Swift 1973). The sill is stabilized by an interdigitating system of ebb- and flood-dominated channels, whose discharge inequalities are a consequence of the phase lag of the tidal wave in its passage across the sill (Swift and Ludwick in press). The Delaware Shelf Valley may be traced directly into the flood channel of the main ebb channel–flood channel couplet. Its leveelike north rim may be traced directly into the complex of smaller ebb channels, flood channels, and sand banks on the north side of the estuary mouth. This shoal area serves as the depositional center for the littoral sand discharge of the New Jersey coast.

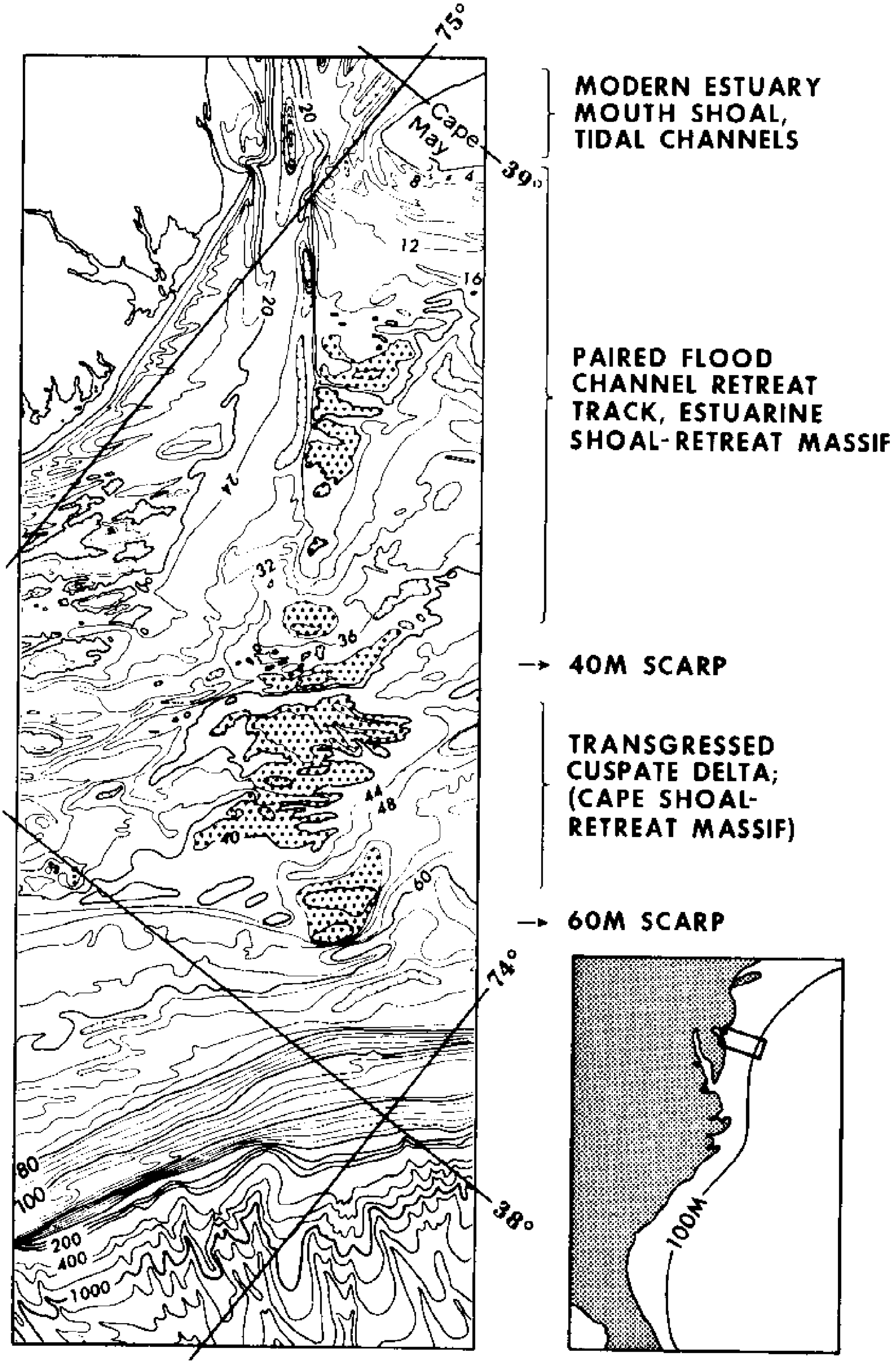
The shelf valley complex, then, is not a drowned river valley, but is rather the track left by the retreat of the Delaware estuary mouth across the shelf during the Holocene sea-level rise. The shelf valley is the retreat path of a flood channel. The north flank levee is the retreat path of the estuary mouth shoal or is a shoal retreat massif—massif in the sense of a compound topographic high consisting of smaller scale highs (Swift 1973). The surface channel does not directly overlie the buried river-cut channel but is offset to the south (Sheridan et al. 1974). As the estuary retreated up the river valley, it not only tended to fill the river valley but in the final, estuary-mouth stage decoupled from it altogether by migrating to the south.

The largely constructional nature of the Delaware Shelf Valley complex is also characteristic of the Great Egg Shelf Valley complex (Fig. 1), although the associated massif has been heavily dissected by the posttransgressional regime of southerly storm flows. To the north, however, the Hudson and Block Shelf Valleys occur on a terrain of innately greater relief. There are cuestalike highs, and the estuarine deposits only partly fill the shelf valleys. The deeply incised nature of the Hudson Shelf Valley may reflect the era when it received Great Lakes meltwater (Veatch and Smith 1939).

Evolution of interfluves—Plateaulike interfluves between the shelf valleys have likewise been intensively modified by passage of the shoreline. Interfluve surfaces range from exceedingly flat plains (slopes of 1:2,000) to irregularly undulating sand ridge topography (Fig. 3). Sand ridges exhibit up to 10 m of relief, are spaced 2 to 4 km apart, and their crestlines may be traced for tens of kilometers. Side slopes are generally less than a degree. Crestlines are not quite parallel to the regional trend of the isobaths but tend to converge to the southwest with the shoreline (Fig. 1). Ridges attain their maximum development on the northeast sides of shoal retreat massifs.

The ridges are molded into a surficial sheet of relatively homogeneous, well sorted sand, 0–10 m thick. In trough axes the sheet thins to a basal shelly, gravelly sand several decimeters or less thick, and a more heterogeneous older substrate is locally exposed (Donahue et al. 1966; Stubblefield et al. 1974). This is commonly a muddy sand or mud deposited behind the retreating Holocene barrier system (Stahl et al. 1974; Sheridan et al. 1974), but it is locally absent due to erosion or nondeposition, so that the Holocene sands rest directly on older Pleistocene sands.

To understand the genesis of this postglacial stratigraphy, it is necessary to consider the dynamics of a transgressing shoreline. We are indebted in this regard to Bruun (1962) and Fischer (1961) who appear to have independently appreciated the role of the landward translation of the wave- and current-maintained coastal profile in generating transgressive stratigraphy. In the New York Bight, as along most low, unconsolidated coasts, the coastal profile consists of a steeply sloping nearshore sector (the shoreface) and a gently dipping in-


75°
Cape May
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20
8
4
12
16
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24
32
36
44
48
40
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74°
80
100
200
400
1000
38°
MODERN ESTUARY MOUTH SHOAL, TIDAL CHANNELS
PAIRED FLOOD CHANNEL RETREAT TRACK, ESTUARINE SHOAL-RETREAT MASSIF
40M SCARP
TRANSGRESSED CUSPATE DELTA; (CAPE SHOAL-RETREAT MASSIF)
60M SCARP
100M

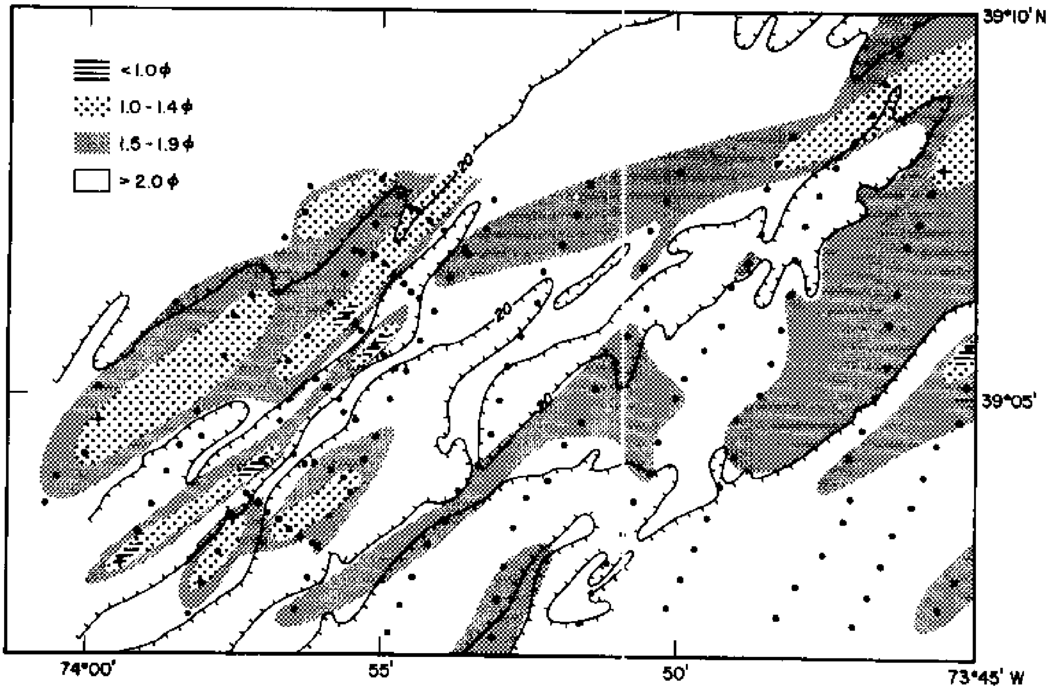

Fig. 3. Simplified bathymetry and distribution of grain sizes on a portion of the central New Jersey shelf. Medium to fine sand occurs on ridge crests. Fine to very fine sand occurs on ridge flanks and in troughs. Locally, erosion in troughs has exposed a thin lag of coarse, shelly, pebbly sand over lagoonal clay. (Reprinted from Stubblefield et al. 1974 by permission of the *Journal of Sedimentary Petrology*.)

ner shelf floor. The break in slope, which may be well defined or gently rounded, generally occurs at depths of 12 to 18 m, some few kilometers from the shoreline.

Bruun (1962) pointed out that if this profile is in fact an equilibrium response of the seafloor to the coastal hydraulic climate, then a rise in sea level must result in a landward and upward translation of the profile (Fig. 4A). Such a translation necessitates erosion of the shoreface. Much of the resulting debris will presumably be entrained in the littoral current and move downcoast, but during periods of onshore storm winds, the littoral drift may leak seaward, due to an offshore component of bottom flow, to be deposited beneath the rising seaward limb of the equilibrium profile on the adjacent inner shelf floor.

Evidence for such seaward bottom transport is varied. Murray (1975) described periods of offshore bottom flow on the gulf coast, when winds are onshore and the water column is not stratified. Sonu and Van Beek (1971) noted that sand loss from North Carolina beaches correlates poorly with periods of high waves but correlates well with high waves generated by onshore northeast winds. On the Long Island inner shelf, we used sidescan sonar to map inner

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Fig. 2. Delaware Shelf Valley complex. Southward littoral drift along the New Jersey coast is injected into the reversing tidal stream of Delaware Bay mouth. The resulting sand shoal is stabilized as a system of interdigitating ebb- and flood-dominated channels. The shelf valley complex seaward of the bay mouth was formed by the retreat of the coastal sedimentary regime through Holocene time. Retreat of the main flood channel has excavated the Delaware Shelf Valley; retreat of the bay mouth shoal has left a levee-like high on the shelf valley's north flank. (Reprinted from Swift 1973 by permission of the *Geological Society of America Bulletin*.)

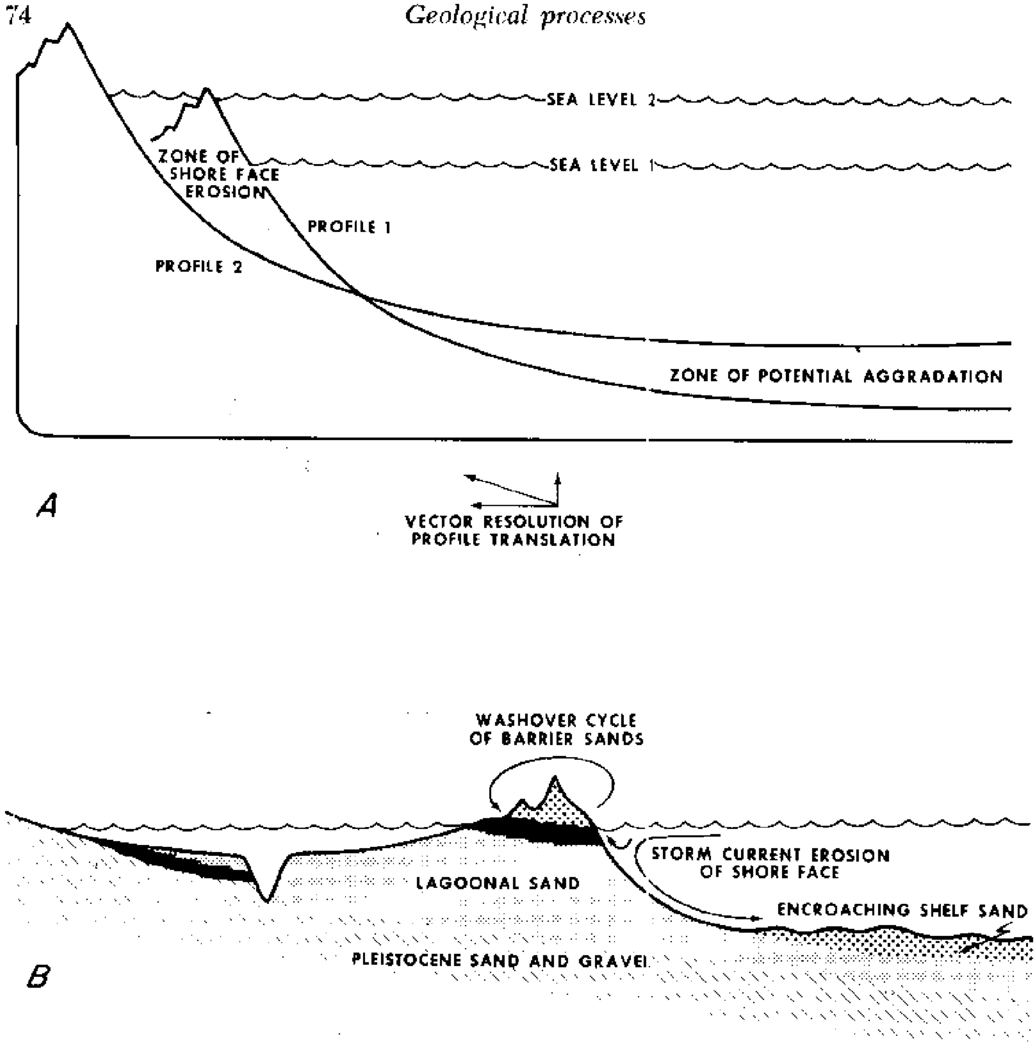

Fig. 4. Models for a coast undergoing erosional shoreface retreat during a rise in sea level. A—Rise in sea level results in landward and upward translation of coastal profile (Bruun 1962). B—Translation is accomplished. Wind and storm washover transport on the barrier surface and erosion of the shoreface and seaward transport of the resulting debris (Fischer 1961).

shelf current lineations that form an eastward-opening angle with the beach (Fig. 5). A poorly defined asymmetry is apparent: the western sides of the lineations are gradational, whereas the eastern sides are sharply defined. The origin of this pattern is not clear. The dark bands are strips of coarse or gravelly sand that may either be troughs between low amplitude, current-transverse sand waves or troughs between current-parallel sand ribbons. However, considering the angle that the lineations make with the beach, sand ribbons seem unlikely for reasons of flow continuity. If sand waves, the lineations are responses to strong bottom flows trending westerly and offshore.

Fischer (1961), Stahl et al. (1974), and Sanders and Kumar (1975) described the stratigraphic consequences of erosional shoreface retreat, based on their observations of the New Jersey and New York coasts. The barrier superstructure will retreat over the lagoonal deposits by a cyclic process of storm washover, burial, and re-emergence at the shoreface (Fig. 4B). Lower shoreface sands will tend to be transported seaward to accumulate over the

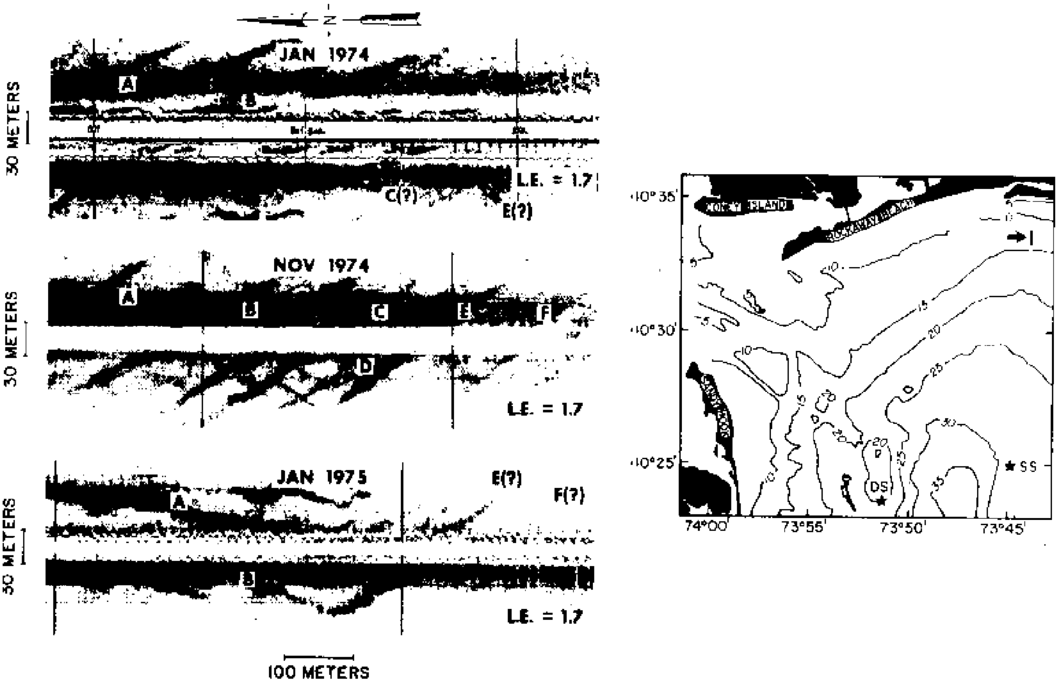

Fig. 5. Sidescan sonar records of current lineations on the Long Island coast, collected at three different periods. Positioning by Raydist. Current lineation pattern (bands A–F) expands to south during observation period. Apparent change in orientation in last panel is due to ship maneuvering. (From Stubblefield et al. in prep.)

eroded surface of the lagoonal deposits as the leading edge of a marine sand sheet. Bruun's hypothesis is compatible with the stratigraphic evidence and with our limited knowledge of coastal hydraulics. A more rigorous test requires bathymetric time series to document changes in the coastal profile. Limited data of this sort are becoming available. Harris (1954) undertook a study of the Long Branch, New Jersey, dredge spoil dumpsite to determine if dumping was nourishing the beach (Fig. 6). In fact, during a 4-year period, the shoreface underwent between 5 and 26 cm of erosion, while an irregular pattern of deposition prevailed on the inner shelf floor. A somewhat longer time series has been prepared by Kim and Gardner (Woodward-Clyde Assoc.) during study of proposed sewage outfall routes for the Ocean County, New Jersey, sewerage authority (Fig. 7). Two out of three profiles taken indicate 1.5–2.0 m of erosion over 20 years. The third profile is immediately south of a shoreface-connected sand ridge; here comparable aggradation has occurred as a consequence of southward ridge migration.

Growth of ridges—Erosional shoreface retreat on the Atlantic cannot be adequately described by a two-dimensional model such as Fig. 4 because the shoreface appears to be the formative zone for sand ridge topography as well as for the sand sheet into which it is impressed. Clusters of shoreface-sand ridges occur on the New Jersey coast between Brigantine and Barnegat Inlets, on the north New Jersey coast between Manasquan and Sea Bright, and on the Long Island coast from Long Beach to the shoreface of eastern Fire Island.

The shoreface-connected ridges are named for their oblique, fingerlike extensions of the shoreface, causing seaward deflections of isobaths as shoal as 5 m. The ridges tend to be asymmetric in cross-section, with steep seaward flanks, although this relationship may be reversed at the base of the ridge where it joins the shoreface. Seaward flanks tend to be notably

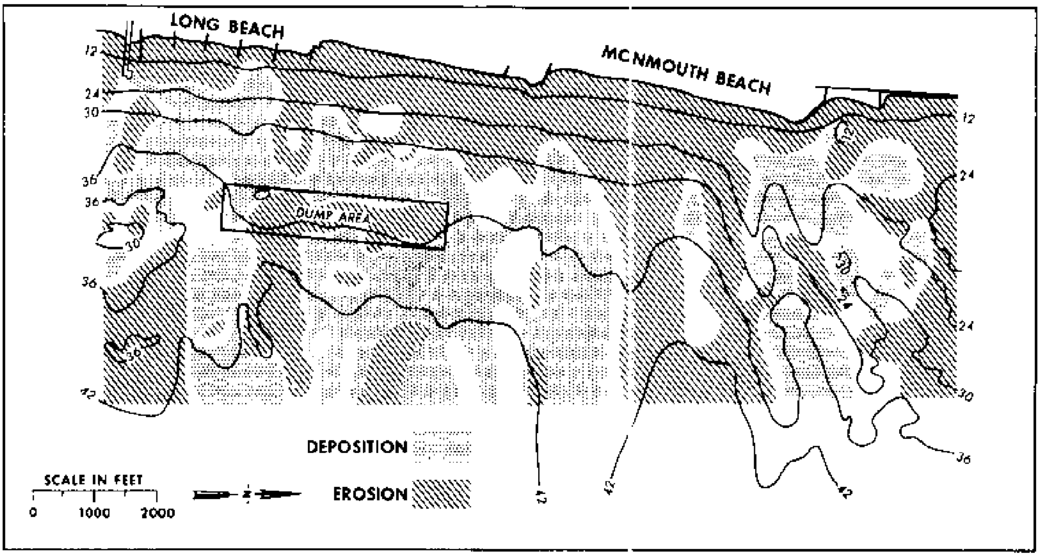

Fig. 6. Erosion and deposition near Long Branch, New Jersey, dredge spoil dumpsite during a 4-yr period. Recorded changes are 0.4–1.4 ft. Shoreface has undergone erosion; adjacent seafloor primarily has undergone aggradation. (From Harris 1954.)

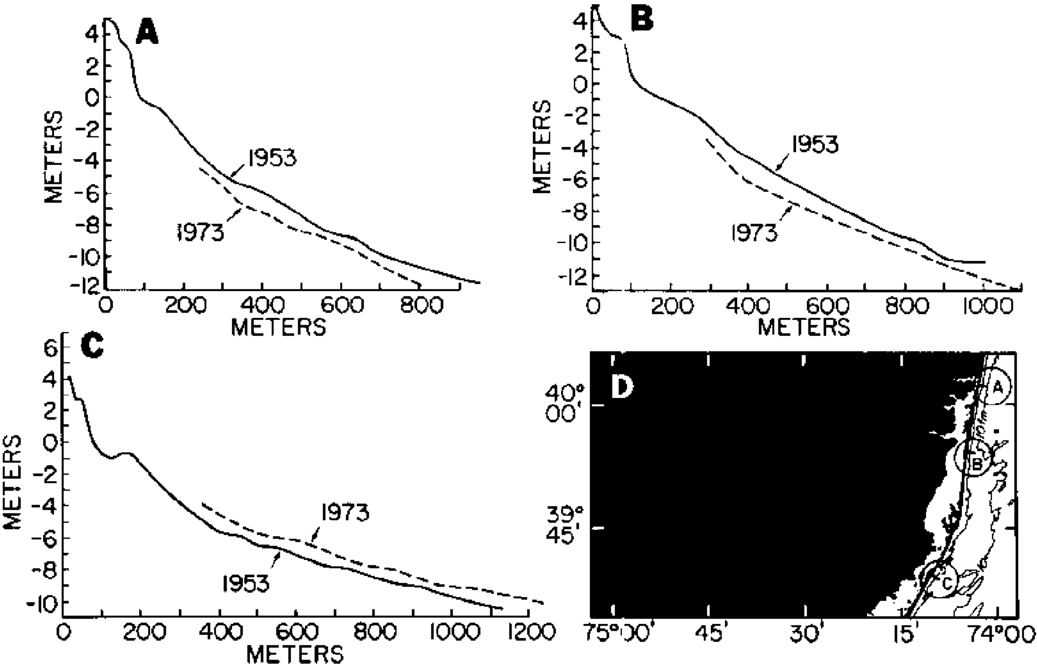

Fig. 7. Profiles of proposed sewage outfall sites on the New Jersey coast. Sites A and B have eroded over a 20-yr period. Site C, immediately downcoast of a shoreface-connected sand ridge, has aggraded. (Reprinted from Kim and Gardner 1974 with permission of Woodward-Clyde Assoc.)

finer than landward flanks. Off Brigantine Inlet and off the New Jersey coast, shoreface-connected ridges are associated with free-standing inner shelf ridges that can be traced seaward for tens of kilometers in apparent genetic sequence. The ridges form on the shoreface in response to south-trending coastal storm currents (Duane et al. 1972) and become detached from the shoreface as it retreats. They tend to migrate downcoast (to the south or west) and offshore, extending their crestlines so as to maintain contact with the shoreline (Fig. 8). Eventually, however, contact is broken, and they are stranded on the deepening shelf floor. Downcoast ridge migration is part of a general pattern of southwesterly sand transport on the Atlantic shelf. In the offshore ridge topography, this pattern is indicated by the tendency of both ridge crests and trough talwegs to rise toward the southwest. Locally, it is indicated by patterns of erosion and deposition near wrecks (Fig. 9).

Sand transport on the inner shelf

The preceding description of the morphologic evolution of the New York shelf surface is based primarily on the interpretation of bathymetric maps, aided by local substrate inventories in which the bottom is examined by grab sampling, photography, Vibracoring, and seismic profiling. The conclusions are qualitative but nonetheless valid. However, fuller understanding of the behavior of the shelf surface requires a different approach.

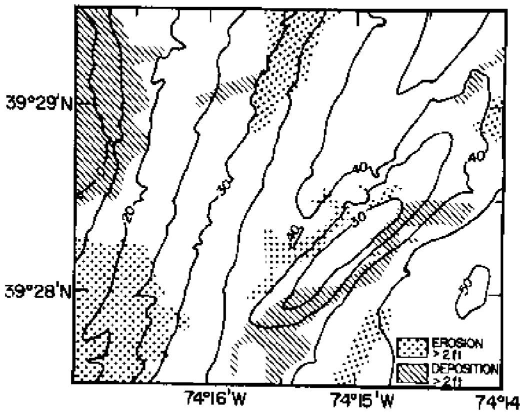

Fig. 8. Patterns of erosion and deposition on Beach Haven Ridge, New Jersey, between 1935 and 1954, superimposed on 1954 bathymetry. Pattern of north flank erosion and south flank deposition indicates downcoast migration of ridge. (From DeAlteris et al. in press.)

We must directly measure fluid and sediment transports involved in the two basic mechanisms that have shaped the shelf surface: tidal flow and sand storage at estuary mouths, and erosional retreat of the shoreface between estuary mouths. Environmental managers who must make decisions about dredged channels, sewage outfalls, sewage and dredge spoil dumpsites, deepwater tanker terminals, and offshore power plants need to understand these processes before they can evaluate the stability of the inner shelf surface.

The nature of coastal sand transport during storms is the first major problem we will consider. Fluid motions in the surf zone have been studied for decades, and the role of longshore currents driven by shoaling and breaking waves has been described (e.g. Bowen 1969). In the New York area,

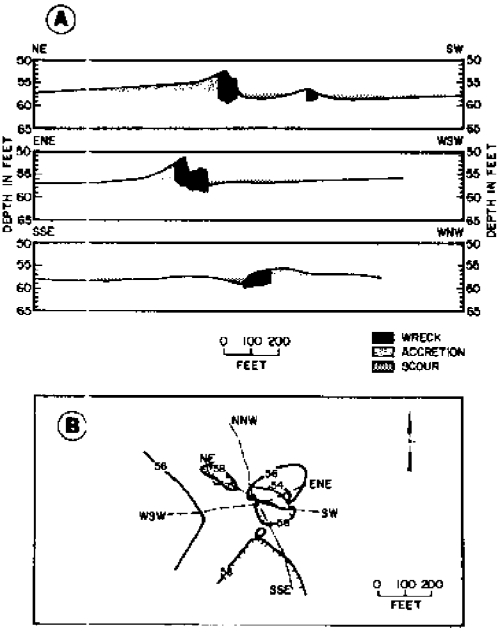

Fig. 9. Accretion and scour by a wreck near Beach Haven Ridge, New Jersey. (From DeAlteris et al. in press.)

massive discharges of sand in the surf zones of the Long Island and New Jersey coasts move toward the New York harbor mouth; these discharges have built Sandy Hook and Rockaway spits within subhistoric to historic times. However, we know almost nothing about fluid motions over the adjacent inner shelf, although the geologic data presented above show that currents seaward of the surf play a major role in the coastal sand budget. We must specifically ask what time and space scales of inner shelf flows are intense enough to entrain sand? Is their velocity field so structured that there are periods of significant offshore bottom flow and sand transport?

Equally important is the problem of the inner shelf sand ridges, which seem to occur wherever a sewage outfall or power plant is to be located. If we wish to predict the probable behavior of these features through the design life of the structure, we must understand their genesis and how they are maintained by flow. It is a truism of loose boundary hydraulics that sheared boundary flows are innately unstable, and that these instabilities tend to interact with the substrate to generate sand ripples, sand ribbons, sand waves, and sand dunes. The circumstantial evidence that inner shelf sand ridges are similarly responses to flow is strong. How are they formed and maintained?

As a first attempt to investigate these questions, Lavelle et al. (in press) placed 40 Aandaraa current meters at 19 stations over the Tobay Beach sand ridges of the Long Island inner shelf (Figs. 10 and 11). The meters were in place for 6 weeks during late November and December 1974; a single meter recorded for an additional 5 weeks. All meters averaged speed over 10 min and took an instantaneous direction reading during each sampling period.

During the observation period, a series of moderate storms induced easterly and westerly flows parallel to the coast. A final storm on 1–4 December was very intense, causing more beach erosion than any storm since the Ash Wednesday storm of 1962 (C. Galvin personal communication).

In Fig. 12, vector averages for all near-bottom, middepth, and near-surface meters are presented for periods of both westward and eastward flows. A wind-controlled pattern of coastal flow emerges. There is a top to bottom speed shear as well as a directional shear. Prevailing fall and winter winds blow out of the northwest, across the east–west Long Island shoreline; the result is a tendency toward coastal upwelling. Surface flows have an offshore component for both eastward and westward flow directions. The response is less symmetrical at depth: westward bottom flows parallel the isobaths, whereas eastward bottom flows have an onshore component. Net water transport during the observation period was eastward.

During the early December storm there was a small offshore component to the water flow near the bottom. Figure 13 shows the winds during the storm and the associated current velocities from a near-bottom current meter, which have been filtered with a 40-h and a 3-h low-pass filter. The 40-h low-pass filtered record, which is a segment

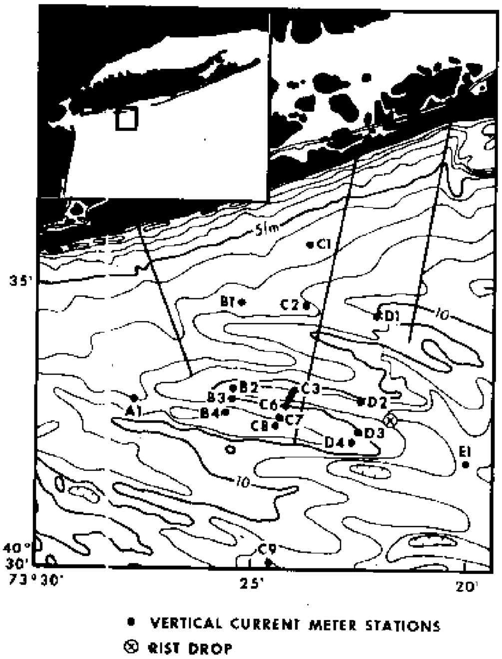

Fig. 10 Bathymetry, current meter stations, and tracer release point (RIST drop) for the Long Island nearshore (LINS) experiment. (From Lavelle et al. in press.)

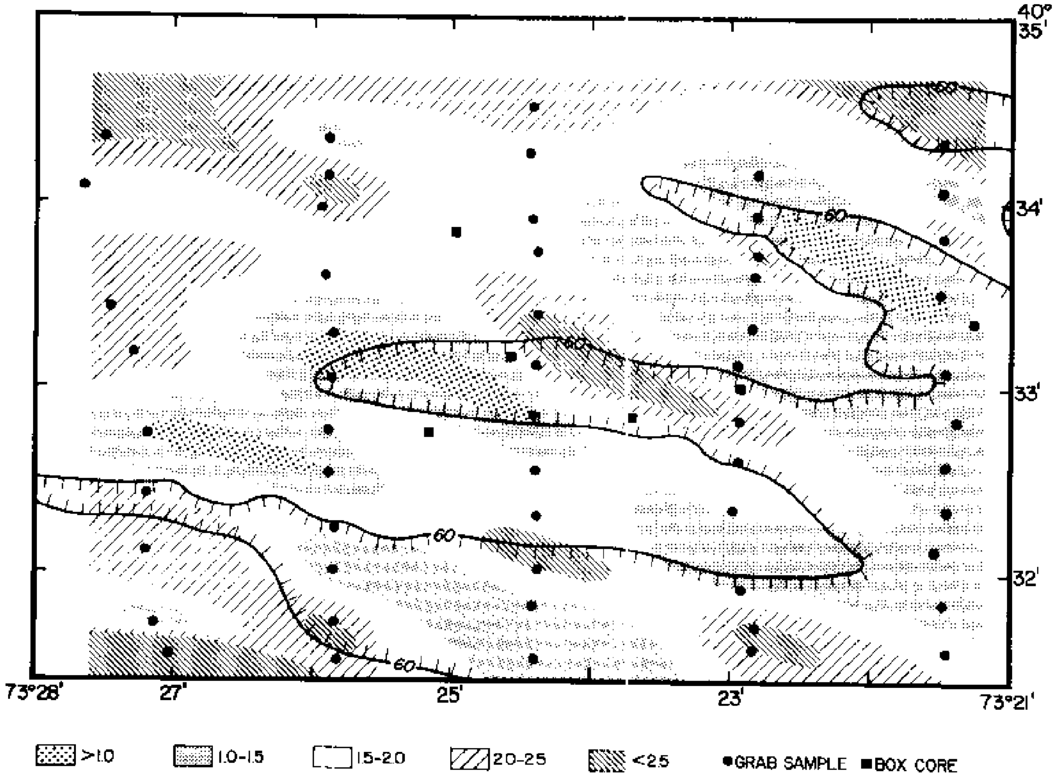

Fig. 11. Distribution of grain sizes over the Tobay Beach ridges, LINS area. Size classes in phi units.

taken from Fig. 12, obscures the brief time-scale flow associated with the storm. The 3-h low-pass record, which is only slightly smoothed and still contains the tidal signal, shows a period of offshore flow more clearly. These results must be viewed cautiously. The Aandaraa current meters which were used have large direction and speed errors when used in shallow water with surface wave amplitudes as large as were present during the event described here.

During the November–December experiment on the Long Island inner shelf, estimates of sand transport were made from calculations from current meter records (Lavelle et al. in press) and also from radioisotope tracer dispersal patterns (Lavelle et al. unpublished data). To generate the patterns, about 500 cm^3 of indigenous fine to very fine sand was surface-coated with 10 Ci of ^{103}Ru (half-life, 39.6 d). On 12 November, equal portions of tagged sand were released in water soluble bags at three points at the east end of the main trough (Fig. 11). The injection points formed an equilateral triangle with sides roughly 100 m long. The developing dispersal pattern of labeled sand was surveyed at intervals by scintillation detectors mounted in a cylinder towed across the bottom. Navigation was by a Raydist system with 10-m resolution. Four postinjection surveys were made during the 11-week tracer experiment. Dispersal patterns mapped 2 and 8 weeks after injection are shown in Fig. 14. After 2 weeks (25 November) roughly ellipsoidal smears trended east from each of the three injection points (Fig. 14A). Each smear could be traced for about 200 m before the signal was lost in the background radiation. After 8 weeks (10 January) the three eastward smears had been replaced by a single, more extensive pattern extending 700 m to the west (Fig. 14B). Partially processed data from an intermediate survey (17–19 December) indicate that the reversal in fact had

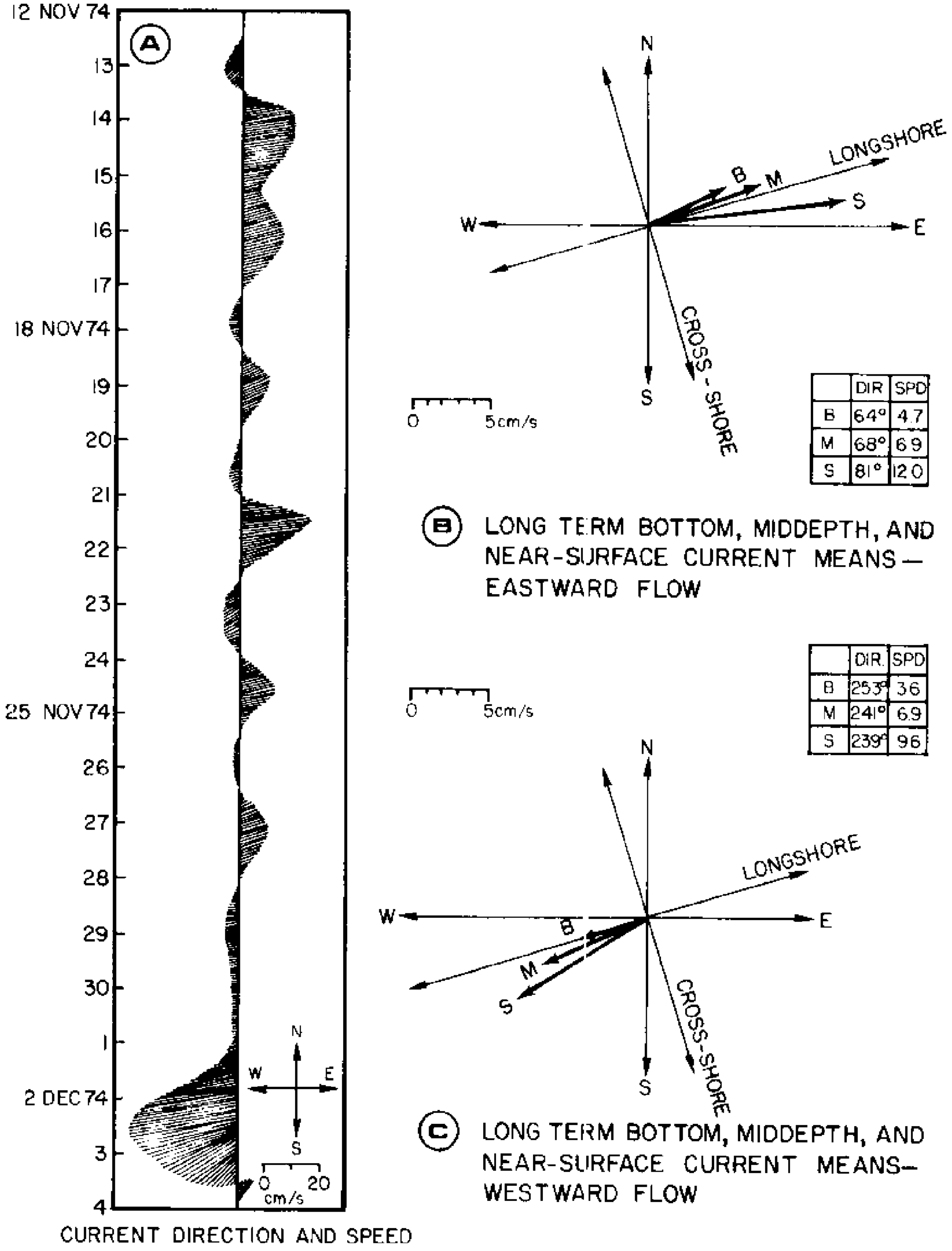

Fig. 12. Summary of flow data for the LINS experiment. A—Vector time series of representative near-bottom flow. Data have been subjected to a 40-h low-pass filter. B, C—Long term velocity averages of eastward and westward flow for meters grouped by depth in water column. Bottom, middepth, and near-surface groupings are labeled B, M, and S. (From Lavelle et al. in press.)

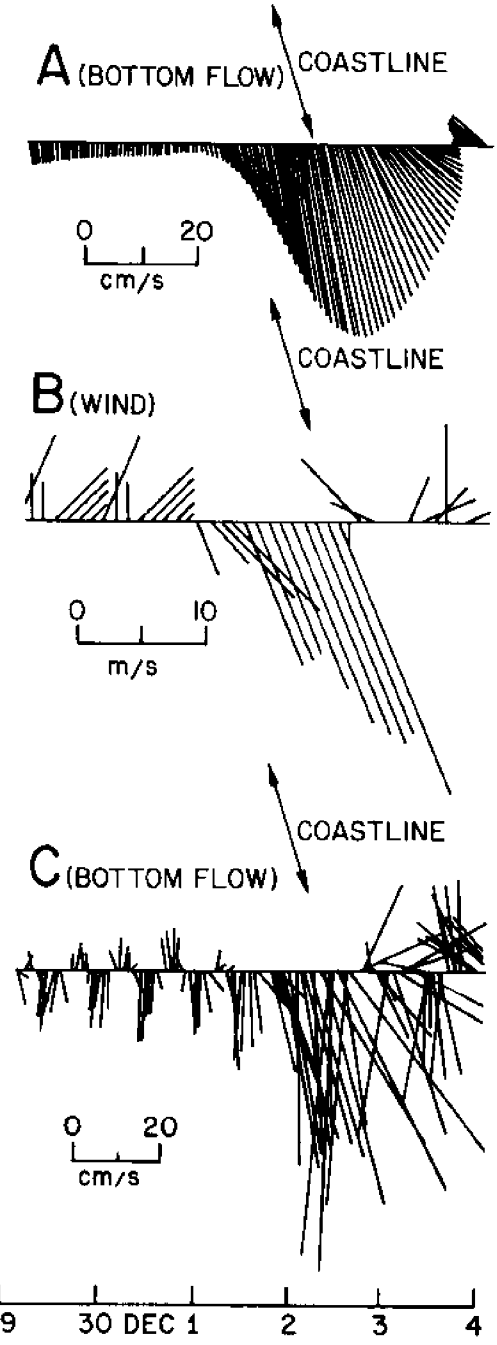

Fig. 13. Vector time series for bottom current and wind velocities during the 1–4 December storm. A—40-h low-pass filtered record (Lanczos filter with response. −6 db at 36 h and −20 db at 40 h). B—Wind record from Ambrose Tower. C—3-h low-pass filtered record (Lanczos filter with response. −6 db at 2.5 h and −20 db at 3 h).

occurred before this and that it initially had been at least 1,200 m long.

The temporal pattern of sediment transport over 60 days may be inferred from Fig. 14C. Current speed, measured 1.5 m from the bed, is plotted against time. The horizontal line at 18 cm/s is an estimated threshold for the fine to very fine sand (mean diameter, 3.0 ϕ) found at the site. It is based on the work of Shields and subsequent workers (Graf 1971: p. 90) and on a choice of 3.0×10^{-3} for the drag coefficient (Sternberg 1972). This choice of threshold velocity was supported by empirical evidence obtained during the course of the experiment (Lavelle et al. in press). Estimates have been prepared for the relative role each transport event played in the overall transport record, based on the concept of frictional energy expenditure proportional to the transport volume (Bagnold 1963). For each event where velocities exceeding threshold were recorded, a transport volume was calculated:

$$Q_i = \alpha \int_{t_i} (|u| - |u_{th}|)^3 \, dt,$$

where $|u|$ is measured current speed, $|u_{th}|$ is threshold speed, α is a constant of proportionality, and t_i is the duration of the transport event (Lavelle et al. in press).

Expression of sand transport as a power of the difference of measured and threshold velocity is supported by Kennedy's (1969) analysis of stream transport data. Without assigning a value to α, we can calculate the rate of transport of one flow event relative to the next or in relation to the sand discharge that occurred over the entire duration of the current meter record. The second of these options has been used in Fig. 14C, where relative sand transport as percent of total transport has been represented as solid bars superimposed on the current meter record. Bar height is a measure of volume percent of transport; bar width is a measure of duration of the transport event. Despite the exceedence of the sediment transport threshold at many points in the record, only the solid bars centered on 2 and 16–17 December are visible in the figure. Thus sand transport during observation consisted of periods of quiescence separated by brief, intense transport events. Furthermore, since discharge is calculated as a power function of excess velocity, intense storms are far more efficient transporters of sand than mild ones. Although the transport index calculated for the 1–4 December storm may be biased by the choice of threshold speed as well as by the functional dependence on velocity, it seems probable that any reasonable parameterization would lead to the same general conclusion: the storm event of 1–4 December moved more sand at 20-m water depth than the combination of all other transport events.

Attempts have also been made to calculate sediment transport indices over longer periods of time in the New York Bight apex. The following computation is based on 30–80-day Aandaraa current meter records (Fig. 15). Data in each current meter record consist of an average speed, u, and an instantaneous direction, θ, taken for each 10-min sampling interval. For each interval in which an assigned threshold speed, u_{th}, is exceeded, a sediment transport index, Q, has been computed, as follows:

$$Q = (|u| - |u_{th}|)^3, \; (|u| - |u_{th}|) > 0.$$

For each current meter, the set of vectors of flow direction, θ $(0° \leq \theta \leq 359°)$, and of sediment transport index, Q, is sorted into 10-degree classes. The results are plotted as

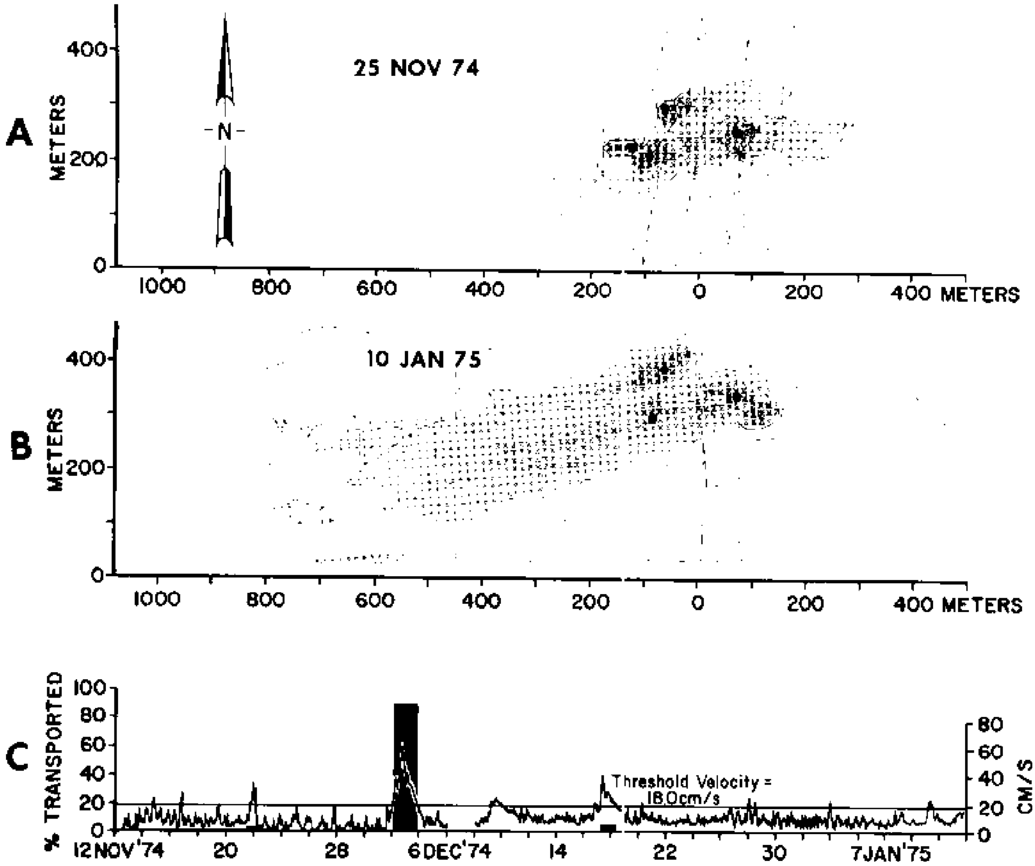

Fig. 14. Sand transport data. A, B—Dispersion patterns measured 13 and 59 days after injection of tagged sand. Point sources are represented by dots. Broken line is the survey trackline. Dots, coarse dots, and Xs indicate increasing intensity of radiation. C—Near-bottom current speed record over the duration of the experiment and calculated sediment transport information. (From Lavelle et al. in press.)

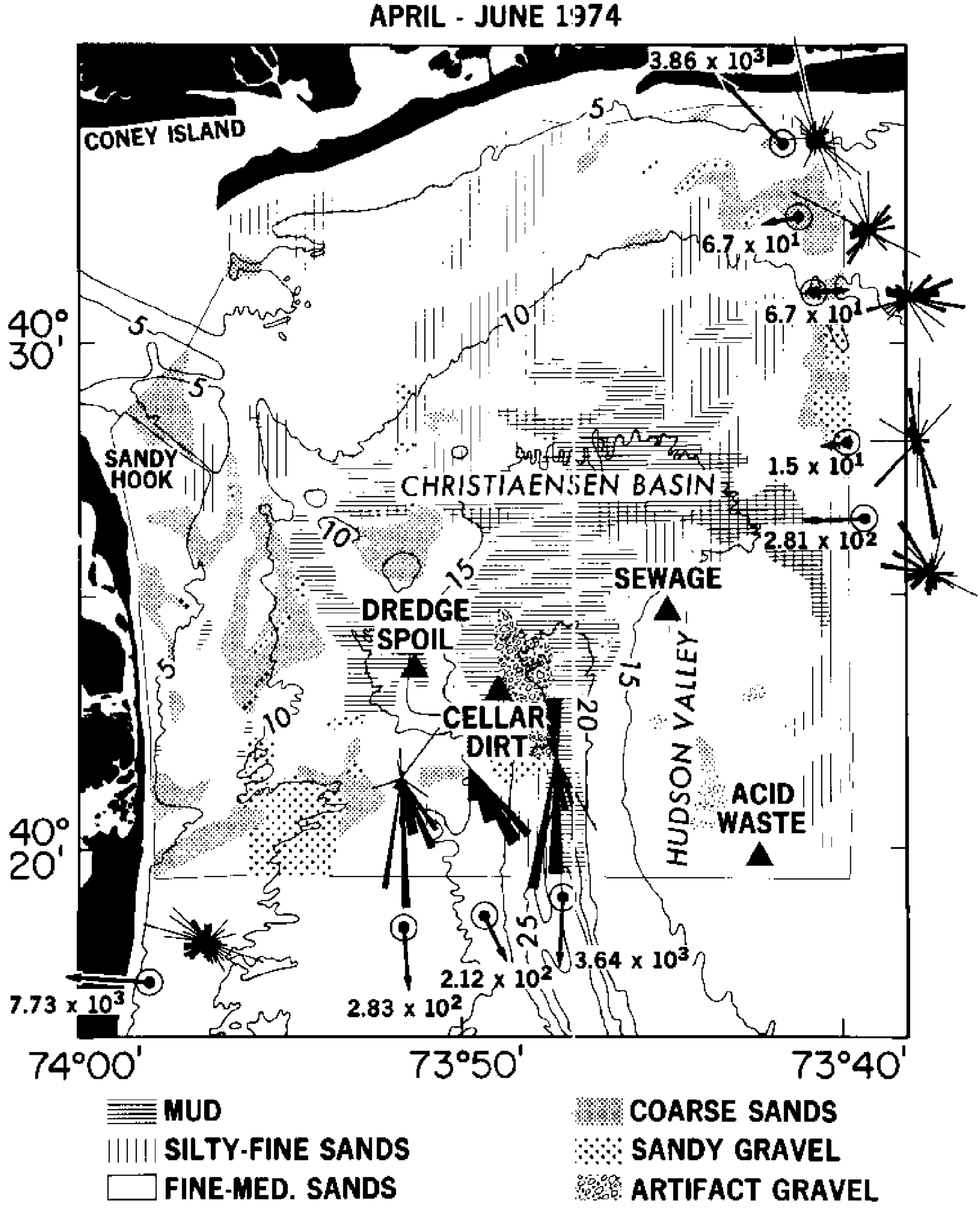

Fig. 15. Bathymetry, bottom sediment character, and calculated patterns of sediment transport for April–June 1974 in the New York Bight apex (*see text*). Depth in fathoms.

rose diagrams in Fig. 15. The length of each radial bar is proportional to the mean sediment transport index, while the width is proportional to the duration of flow above threshold, hence the bars may overlap.

For each current meter station in Fig. 15, the normalized resultant of all sediment transport vectors is indicated by a single arrow. The resulting magnitude has been divided by the total number of days that the current meter was in operation (T_p) to derive a daily average, $\overline{\vec{Q}}_D$:

$$\overline{\vec{Q}}_D = \frac{1}{T_p} \int_0^T (|u| - |u_{th}|)^3 \mathbf{u} \, dt,$$

where $\mathbf{u}$ is a unit vector with direction θ, and T is the total number of days. The integrand is zero when the velocity is less than threshold.

Figure 15 suggests that during April–June 1974, sand transport was westward off the Long Island shore and southward off the New Jersey shore. Nearshore stations reveal a strong onshore component of the sand transport index, perhaps because of wind-induced upwelling or because of the landward directed asymmetry of bottom wave surge, or both. The magnitude of the sand transport index generally decreases seaward but is anomalously large within the Hudson Shelf Valley. The easterly transport revealed by a single station off the Long Island coast is probably due to instrument problems.

Some unsolved problems

The inner shelf sand budget—Our studies of sand transport on the New York inner shelf have resolved some questions but raised others. It is clear that sand transport occurs seaward of the surf zone. Transport is episodic in nature. Sand is entrained and transported by brief, intense, wind-driven coast-parallel flows lasting for hours or days and separated by days or weeks of quiescence. Our measurements suggest that inner shelf bottom flows are more likely to transport shelf sands shoreward than seaward. This appears to be due to intermittent coastal upwelling induced by northwesterly winds and perhaps also to the landward-oriented asymmetry of near-bottom wave surge. Baylor (1973) has also noted this pattern of wind-induced coastal upwelling off Long Island, and R. Scarlet (EG&G, Waltham, Mass., unpublished) reported a similar regime of coastal upwelling for the Beach Haven Ridge site (Fig. 16).

However, our observations indicate that the 1–4 December storm was the only event that caused massive sand transport. It stands out within our two periods of current monitoring not only in its duration, intensity, and westward direction of net transport but in the offshore component of bottom flow. We must consider the hypothesis that the 1–4 December storm, anomalous within the context of our short term winter observation period, is in fact the kind of peak flow event that shapes the inner shelf surface and controls its sand budget. We have noted that Atlantic shelf stratigraphy is best explained by erosional shoreface retreat and seaward transport of the eroded material. We have described the southwest migration of shoreface-connected ridges off New Jersey and have cited evidence for the net southwest transport of sand (Fig. 8). We note that the Tobay Beach ridges (Fig. 11) are, like other Atlantic Bight ridge fields, asymmetrical in both grain-size distribution and morphology; the seaward-facing southwest slopes are steeper and finer grained, implying that westward flows scour the upcurrent flanks and deposit fine sand on the seaward-facing downcurrent flanks.

Recent studies by physical oceanographers also suggest that southwestward currents generated by "northeaster" storms have the greatest potential for shaping the shelf surface. Beardsley and Butman (1974) have described a scale-matching phenomenon, in which the Middle Atlantic Bight tends to interact with "northeasters" of the appropriate size and trajectory so that intensive southwestward flows result (Fig. 17). Their observations indicate that if low pressure cells cross the bight on a trajectory such that the isobars of atmospheric pressure cross the isobaths of the shelf surface at a high angle, then oscillations of the water column may result, but there is little net displacement of water. However, when the trajectory and scale of the storm are such that for a period it rests in the Middle Atlantic Bight so that the isobars parallel the isobaths, then strong sustained coupling of wind and water flow results. The winds blow along the isobars, down the arc of the Middle Atlantic Bight. Landward Ekman transport of surface water causes 40 to 60 cm of coastal setup and results in a south-

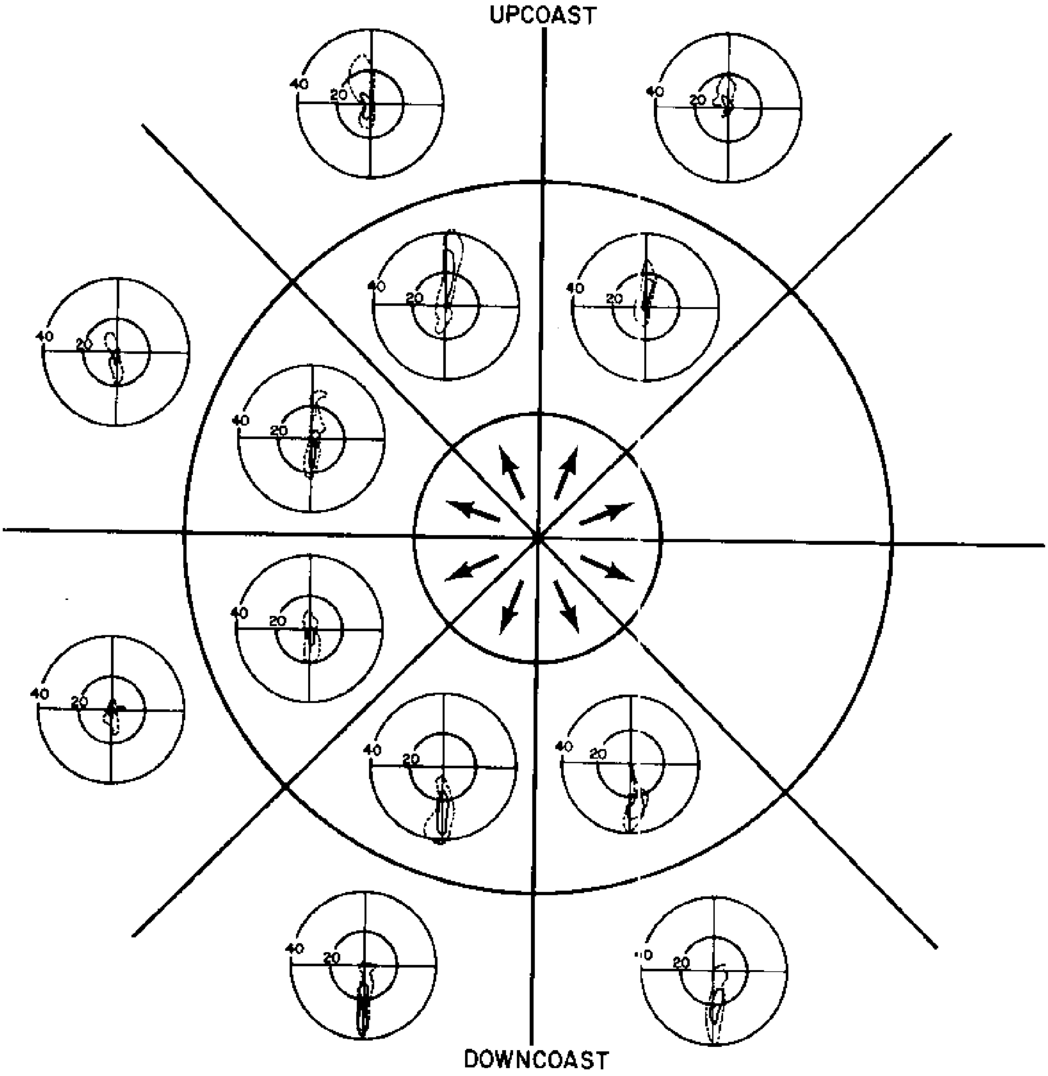

Fig. 16. Polar histograms of hourly averaged, de-tided summer currents in cm/s in the vicinity of Beach Haven Ridge, New Jersey. Only flows associated with winds over 5 m/s are shown. Prevailing wind is indicated by location of histogram on page with wind direction shown in center. Inner ring of histograms is for near-surface measurements, outer ring is for near-bottom measurements. Directions of winds and currents are indicated with top of page representing upcoast motion (036° true). Histograms are omitted if fewer than 35 h of data were found for specified wind condition. Solid contours enclose 50% and dashed contours enclose 90% of data. (Adapted from EG&G Environ. Consult. 1975.)

ward geostrophic transport of the shelf water column that is coherent and slablike. Boicourt and Hacker (1976) described a similar period of southward storm flow on the Virginia coast with sustained middepth velocities of 30–50 cm/s. Both sets of investigators noted a marked asymmetry in the hydraulic climate, whereby southwest storm flows tend to be noticeably more intense than northeast flows.

It is clear from the preceding discussion that the role of storm-driven currents in mediating the coastal sand budget requires additional study. We need to know more about the frequency of southwestward storm flows and their velocity structure. We

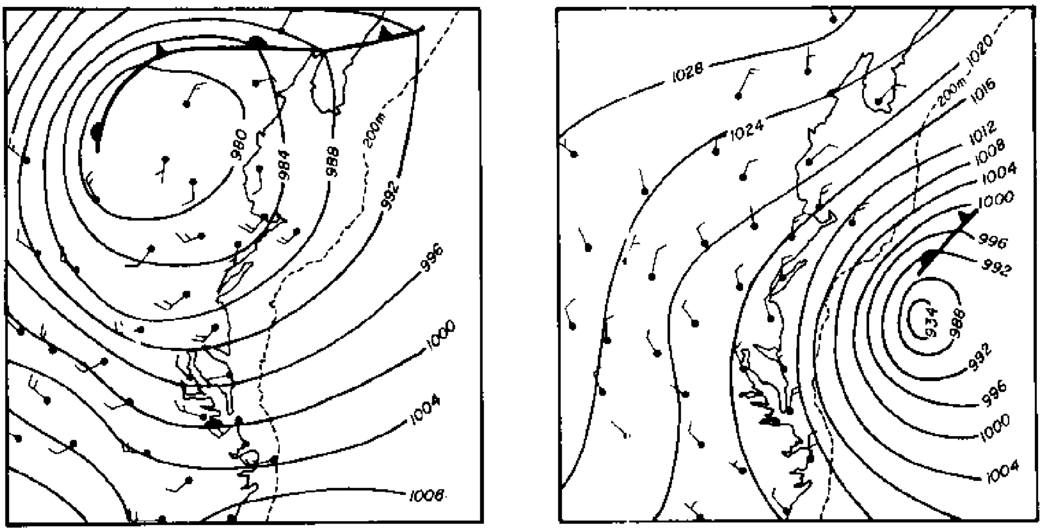

Fig. 17. Surface weather maps for 18 and 22 March 1974 Only the second storm produced sustained coupling between wind and water flow. (From Beardsley and Butman 1974.)

must also learn to design experiments that will resolve perturbations of flow that build and maintain ridge systems.

Sand transport and storage at New York Harbor mouth—Major sections of the New York–New Jersey shelf have been shaped by the tidal regimes associated with estuary mouths during the postglacial rise of sea level. Sand budgets of estuary mouths are also of great interest to environmental managers; the Atlantic coast estuaries are the approaches to the major coast ports and require repeated costly dredging. At present, the only estuary mouth subjected to systematic study is that of Chesapeake Bay (Ludwick 1972, 1974, in press). However, reconnaissance data are available for the Hudson estuary mouth, which suggest directions for further study.

New York Harbor mouth is clearly a sink for the littoral drift of the Long Island and New Jersey coasts. Within the past century, much of the deposition has occurred on the ends of Rockaway and Sandy Hook spits; these features have grown rapidly, nearly closing off the harbor mouth within historic times (Shepard and Wanless 1971). However, it appears that much sand has bypassed the spits; a complex system of sand banks separated by interdigitating ebb and flood channels lies between them (Fig. 18). A profile of velocity residual to the semidiurnal tidal cycle gives some indication of the flow structure responsible for

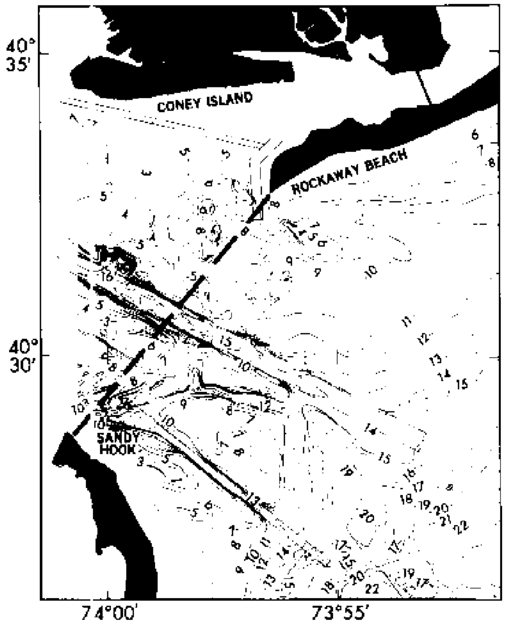

Fig. 18. Bathymetry of the New York Harbor mouth, from a 1973 NOAA/AOML survey. Depth in meters. Dashed line indicates profile of Fig. 19.

bank-channel topography (Fig. 19). The characteristic estuarine two-layer flow is present as indicated schematically in Fig. 19B. The less saline upper water has a residual seaward flow, and the more saline lower water has a residual landward flow. As a consequence of the Coriolis effect, the interface is tilted so that the east side of the harbor mouth is flood dominated while the upper level of the west side is ebb dominated. The distribution of isovels in Fig. 19A suggests that this basic pattern has been modified by the frictional retardation of the tidal wave in the shallow estuary and the resulting phase lag (Swift and Ludwick in press). Because of retardation, there is a brief period during the tidal cycle when the estuary tide is still ebbing through the central channel while the shelf tide has already turned and is flooding on either side of the ebb tidal jet. This flow pattern, integrated over the tidal cycle, results in greater ebb than flood discharge in the central channel (ebb dominance) and greater flood than ebb discharge in the marginal zones (flood dominance; Fig. 19C). It is probably because of this lag-induced flow interpenetration that the Sandy Hook Channel is not completely ebb dominated as required by the two-layer,

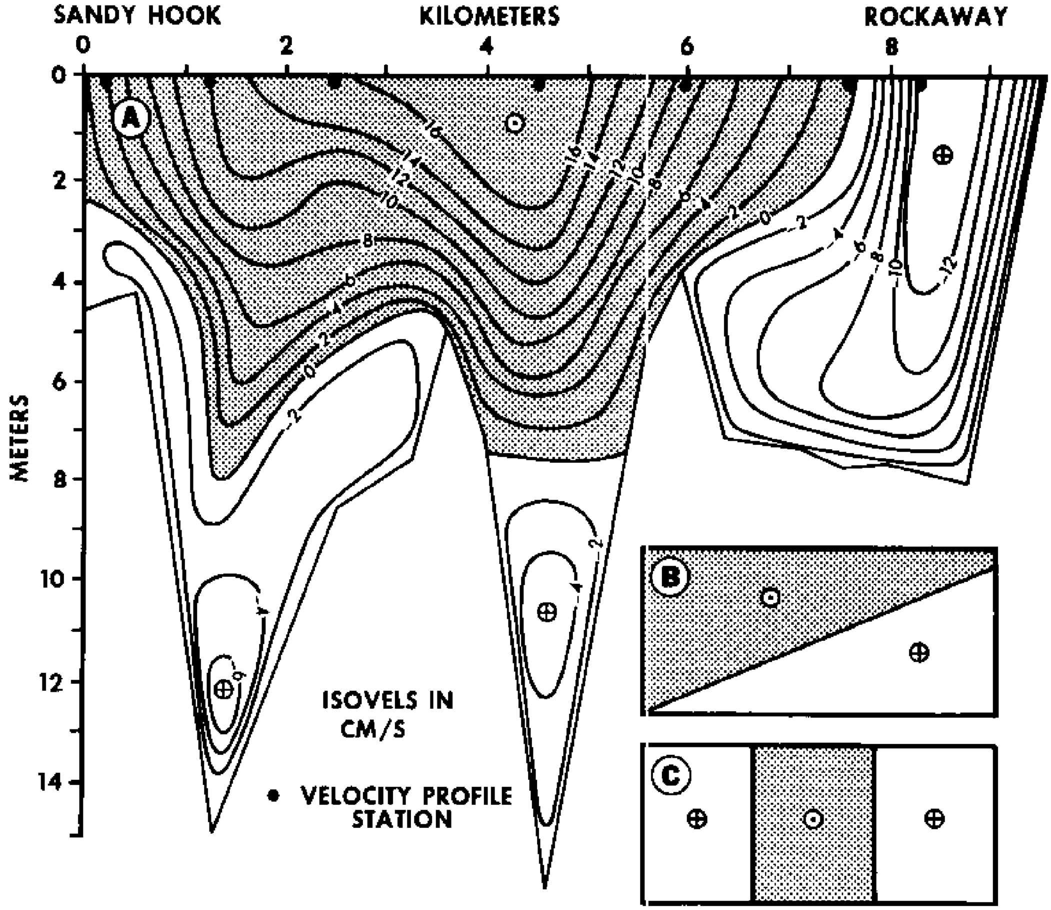

Fig. 19. A—Profile across the Hudson estuary mouth (mouth of New York Harbor), contoured for velocity residual to the semidiurnal cycle. Pattern is interpreted as a resultant response to component patterns shown in B and C. B—Schematic diagram of two-layered, density-driven estuary flow. C—Schematic diagram of pattern resulting from phase lag of the tidal wave. (Modified from data of Kao 1975; reprinted from Duedall et al. in press by permission of *Estuarine and Coastal Marine Science.*)

estuarine component of flow but is flood dominated near the channel floor (Fig. 19A). The two sand ridges that separate the three channels are presumably built by this pattern of flow dominance. Residual flow on the opposite sides of a given sand ridge will have the opposite sense; each ridge is therefore a sand circulation cell or closed loop in the sand transport pattern.

Here perhaps are the ultimate sinks in the littoral sand transport pattern of the New York Bight. Efficient maintenance of the dredged shipping channels demands verification of this inferred pattern of flow dominance and careful analysis of the resulting sand budget.

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Surficial sediments of the NOAA-MESA study areas in the New York Bight

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Abstract

In the New York Bight apex, extensive sedimentological studies and a 1973 bathymetric survey reveal that the only significant change in bottom topography since 1936 occurred at the dredge spoil dumpsite where the dumping of 98×10^6 m^3 of dredged material has caused up to 10 m of shoaling. The center of the Christiaensen Basin, a natural collecting area for fine-grained sediment, is no doubt contaminated with sludge but shows no apparent sediment buildup during the intervening 37 years. The apex outside of the Christiaensen Basin is floored primarily by sand ranging from silty fine to coarse, with small areas of sandy gravel, artifact (anthropogenic) gravel, and mud. Nearshore mud patches appear to be covered at times with sand and occasionally scoured out. Sidescan sonar records show linear bedforms, indicative of sand movement, over most of the apex area.

Two midshelf areas have been proposed as interim alternative dumping areas. The northern area is in a tributary valley of the ancestral Long Island river system. Fine sands cover the northeast part and medium sands predominate to the west and south. Bottom photographs show a smooth, slightly undulatory, mounded or rippled sea floor.

In the southern alternative dumping area coarse sand and gravel deposits lie on the crest and east flank of the Hudson divide, while medium and fine sand occurs in the ridge and swale topography to the west. These distributions suggest fine sediment is winnowed from the crest and east flank of the divide and deposited to the west. Veatch and Smith Trough contains a veneer of shelly, pebble sand with large, angular clay pebbles and occasional oyster shells derived from exposed early Holocene lagoonal clay. These studies suggest that if sewage sludge were dumped, widespread dispersion, mostly to the southwest, could be expected, with winter resuspension and transport of fine material on the bottom. Possible permanent buildup on the bottom could be expected if dredged material were dumped.

The nature of bottom sediments and sediment particles suspended in the water column becomes of interest to environmental managers when man's activities in the ocean disturb the sea floor or the near-bottom water column. In addition to the immediate results, one must also consider the effect on long term natural phenomena. How are these processes affected by what man has done, or perhaps more importantly, how do natural processes modify what man has done to disturb the natural environment?

Here we report work done at the Atlantic Oceanographic and Meteorological Laboratory as part of the NOAA-MESA New York Bight Project.

Hydrographic surveys of the New York Bight were initiated in 1936 by the Coast and Geodetic Survey (now the National Ocean Survey) in nearshore areas and have been repeated periodically. Bathymetric maps at 1-fathom (Stearns and Garrison 1967) and 4-m intervals on the shelf and 200-m intervals on the continental slope (Fig. 1; Uchupi 1970) were made from 1936 survey data. A new survey of the bight was made in 1975; results should be available in 1977.

Surficial morphology of the New York Bight, and sediment distribution across this surface, may be explained by sea level fluctuations caused by continental glaciation during the past several million years. The last glacial stage ended 15,000 years ago (Milliman and Emery 1968) when the eastern North American ice sheet extended as far as Long Island and northern New Jersey. During maximum glacial advance sea level was lowered about 160 m (Veatch and Smith 1939) so that the shoreline of the bight was in the vicinity of Hudson Canyon (*see Fig. 1*). Since the ice melted, the shore-

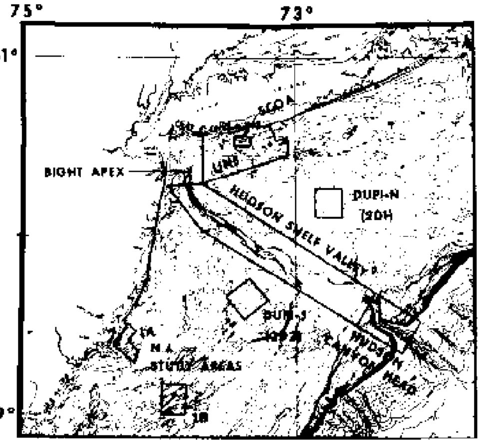

Fig. 1. Index to detailed study areas and topographic features in the New York Bight. (Bathymetry from Uchupi 1970.) Contour intervals 4 and 200 m. 1A—New Jersey nearshore ridge and swale study area and the Atlantic generating station site; 1B—New Jersey central shelf ridge and swale study area; LINS—Long Island nearshore study area; SCOA—Suffolk County outfall area; 2D1, 2D2—proposed interim alternative dumpsites.

line advanced to its present position; many features of the shelf are the result of several sea level fluctuations. Morphologic features are discussed in our companion paper in this volume (Swift et al. 1976) and elsewhere (e.g. McKinney and Friedman 1970; McKinney et al. 1974; Stubblefield et al. 1974, 1975; Knott and Hoskins 1968; Duane et al. 1972; Williams 1976).

Surficial sediments

A comprehensive sampling program for the outer shelf was conducted by the Woods Hole Oceanographic Institution and the U.S. Geological Survey, who sampled on an 18-km spacing. The Corps of Engineers Coastal Engineering Research Center has collected about 4,200 km of geophysical data and over 300 cores as a part of its studies on the inner shelf of the bight (Duane 1969; Williams and Duane 1974; Williams 1976). MESA work had been conducted primarily in New Jersey nearshore and central shelf areas, the bight apex, the nearshore of Long Island eastward to Fire Island, two central shelf alternative dumping areas, and the Hudson Shelf Valley (Fig. 1). Emphasis here is on the bight apex and the central shelf alternative dumping areas.

Source and age of sediments—Sediments covering the floor of the bight were mostly deposited during lowered sea level and were reworked during the landward-seaward migrations of the shoreline. As transgression progressed, fluvial and older sediments were covered by estuarine and lagoonal sediments behind barrier islands or directly reworked by littoral processes associated with the advancing shoreline. During a transgression, bottom currents of the inner shelf interact with the shelf floor to form a concave surface whose profile resembles an exponential curve, with the steep limb comprising the shoreface (Swift et al. 1972). With a loose, sandy substrate, the inner shelf shoreface tends to extend itself laterally across the mouths of bays, closing them, except for inlets, by the deposition of sand in the form of spits and barrier islands. Estuaries and lagoons behind these spits and islands then trap suspended fine sediment (mud), while the barrier islands are nourished by littoral drift from eroding headlands and by sand moving landward from the shelf.

As sea level rose during the Holocene transgression, the inner shelf profile moved shoreward by means of shoreface erosion. Some eroded sand was swept onto the barrier islands by storm overwash and buried, only to be re-exposed again at the eroding shoreface. Most material from shoreface erosion, has, however, been washed downcoast and seaward to form a discontinuous sand blanket 0 to 10 m thick (Stahl et al. 1974). Thus, the New York Bight shelf floor is dominantly sand-sized sediment (Schlee 1973). Fine-grained sediments are generally absent, having been transported either into the Hudson-Raritan estuary, behind barrier islands, or off the shelf edge. Locally, underlying strata of transgressed lagoonal and estuarine semiconsolidated mud deposits or resistant coastal plain strata are exposed on the sea floor (Swift et al. 1972; Stahl et al. 1974; Sheridan et al. 1974).

Sediment types—Sediment types have been mapped in the New York Bight primarily by dominant grain size (Fig. 2).

Generally, the shelf is covered by sand-sized sediment with isolated gravel patches (Schlee 1973, 1975; Williams and Duane 1974; Williams 1976). In deeper water, generally seaward of the 60-m isobath, in the Hudson Shelf Valley, and in lagoons and estuaries where wave action is less pronounced, silt is the dominant sediment (Freeland and Swift in press). In the Long Island nearshore zone west of Fire Island, small mud patches, some of which are seasonal, are of considerable environmental concern owing to contamination of the fines by pollutants.

Suspended sediments—Meade (1972*a*, *b*) noted the following: Pleistocene glaciations and sea level fluctuations drastically altered the composition and distribution of sediments on continental margins; it is not always immediately evident whether present shelf deposits reflect modern or Pleistocene conditions. Fine sediment transport studies are hindered by the fact that deposited sediments may reflect processes acting over thousands of years, whereas our studies of suspended sediment are commonly limited to a few days or months of observations. Natural processes may be impossible to separate from the changes produced by human activities, particularly in estuaries (and at the present dumpsites).

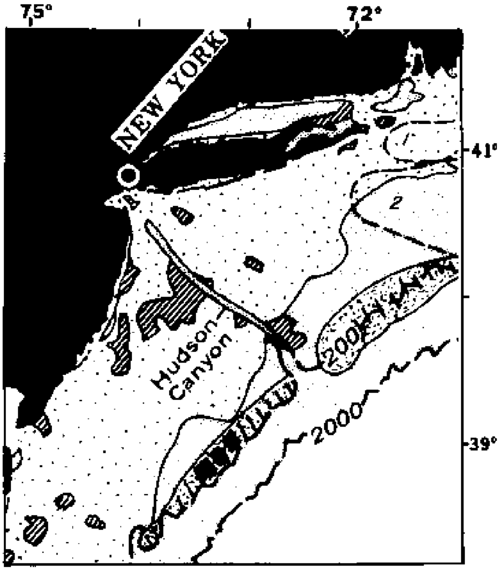

Fig. 2. Sediment types in the bight area (depth in meters). Hatching—gravel, sandy gravel, and gravelly sand; speckling—sand; stippling—silty sand, sandy silt, and clayey silt; dappling—glauconitic sand, silty sand, and sandy silt. ▪▪▪—Pyrite-filled foraminiferal tests. 1—Zone of rounded quartz grains; 2—zone of limonitic pellets. (From Uchupi 1963.)

Table 1. Source of suspended solids in the New York Bight.*

	$\times 10^{9}$ tonnes/yr
Direct bight (68%)	
Dredged (54%)	4.73
Sludge (2.1%)	0.18
Cellar dirt (6.8%)	0.60
Total barged (62.9%)	5.51
Atmospheric (5%)	0.45
Coastal zone (32%)	
(98% of coastal zone input is through the Rockaway–Sandy Hook transect)	
Municipal wastewater (4%)	0.35
Industrial wastewater (0.2%)	0.02
Gauged runoff (16%)	1.4
Urban runoff (12%)	1.1
Total coastal zone	2.87
Total input	8.83

* From Mueller et al. 1976.

Fine sediment sources to estuaries and the shelf—Fine sediment discharged into the bight is shown in Table 1 (Mueller et al. 1976). Fluvial sediment is comprised of roughly 85% inorganic and 15% combustible organic material (Table 2). The fine inorganic fraction is mostly illite, chlorite, feldspar, and hornblende from the Hudson River (Hathaway 1972).

Shelf erosion and coast-parallel transport appear to be significant but unmeasured sources of suspended material and were probably major sources during the Holocene transgression. Hathaway (1972) showed that fine sediments near the mouths

Table 2. Composition of suspended matter.

Rivers	80–90% minerals	10–20% combustible organics
Estuaries	60–80% minerals + biogenic shells	20–40% combustible organics
Shelf	10–70% minerals + biogenic shells	30–90% combustible organics

of coastal plain estuaries differ significantly from the composition of riverborne sediments. It is probable that much estuary-mouth sediment is being eroded from shelf deposits and returned to and trapped in estuaries (Meade 1969). The fact that the sediments from modern rivers have not obscured this conclusion implies that either the modern sediment is bypassing the lower portions of the estuary, or it is trapped almost completely near the river mouths. Along the east coast, the heads of the Chesapeake and Delaware estuaries are far upstream from the estuary mouth, therefore, most river sediment is deposited far inland from the sea. Although saline tidal water is present in the Hudson River up to Albany, fine fluvial sediment is carried by low-salinity surface water to Upper and Lower New York Bays where some fines settle out (Folger 1972*b*) and the remainder is carried with estuarine sediment into the bight apex and mixed with recirculated shelf sediment. In the northeast United States, most of the fluvial suspended sediment is effectively trapped in estuaries and coastal wetlands (Milliman 1972).

At the present, the annual suspended sediment discharge of Atlantic coastal rivers is about equal to the annual deposition on marsh surfaces (Meade 1972*a*). However, much of the deposited material re-enters the shelf water column after the shoreline has passed over the marsh, through the process of shoreface erosion (Fischer 1961).

Particles derived from biologic processes are also a significant component of suspended matter in estuaries and on the shelf (Table 2), ranging from 20–90% in surface waters (Manheim et al. 1970). However, concentrations of combustible biogenic matter decrease rapidly with depth, and little of this material is preserved in sediment deposits (Folger 1972*a*; Gross 1972).

Atmospheric fallout over the New York Bight is small relative to other sediment sources (Table 1), but it may be a significant transport path for specific pollutants (e.g. lead from vehicular exhaust emissions).

Highest concentrations of organic and inorganic suspended materials in the water occur within 10 km of the coastline and decrease nearly exponentially seaward (Manheim et al. 1970). Mineral grains larger than 4 μm (silt-size) comprise 10–25% of nearshore suspended sediment and only 2–5% of offshore samples; the remainder is organic matter. The zone of strong terrigenous influence is restricted to nearshore waters and, specifically, to the inner shelf zone of turbid water drifting away from the estuary mouth. The coarser grains in this zone are effectively trapped in the "estuarine" circulation (which serves to reinforce the surface concentrations) and are transferred from one estuary to the next along the path of the longshore current.

Studies of other areas (Postma 1967) suggest that volumes of suspended sediment transported on the many feedback loops in the bight are probably orders of magnitude greater than both the net volume from the Hudson River that is transported across the shelf and the much larger amounts introduced by dumping.

Although the factors which influence suspended sediment dispersal can be readily defined, many large gaps in our knowledge must be closed before quantitative sediment transport budgets can be constructed on a regional scale. The most important of these are: shelf circulation patterns and mechanisms, particularly during storms; hydraulic properties of suspended sediments, particularly resuspension and settling properties; and the influence of flocculation and biologic aggregation on settling.

Detailed studies in the New York Bight apex

A 1973 bathymetric map (Fig. 3) of the bight apex was made as the result of a NOAA-Corps of Engineers survey. The principal topographic features are the northern end of the Hudson Shelf Valley, Cholera Bank, and the Christiaensen Basin, an amphitheaterlike feature terminating the Hudson Shelf Valley (Veatch and Smith 1939). Dumpsites for dredge spoils (the mud dump), cellar dirt, sewage sludge, and acid wastes are shown. Knolls immediately northwest of Ambrose Light and north and northwest of the dredge spoil dumpsite

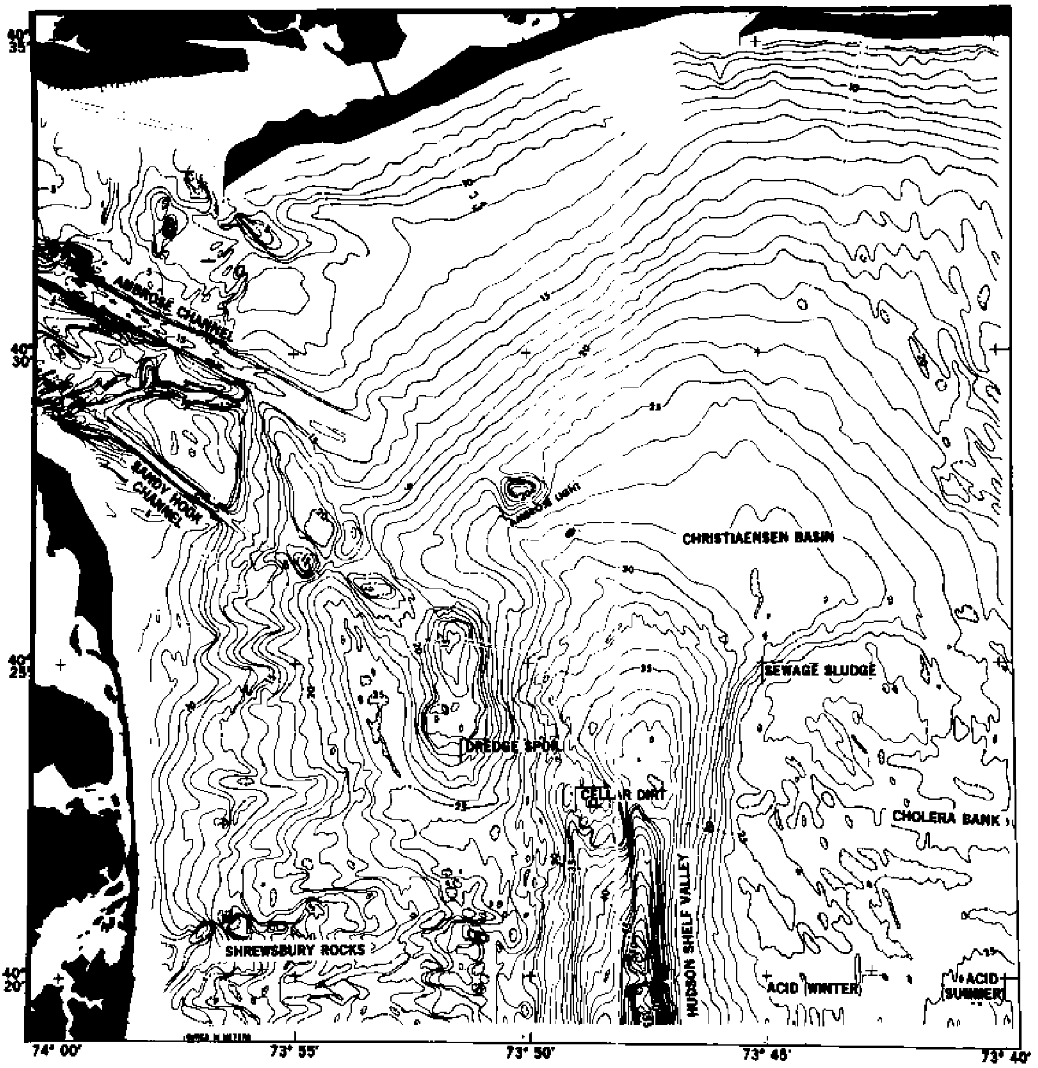

Fig. 3. Bathymetric map of the New York Bight apex. Contour interval, 1 m. Data (in meters) from 1973 NOAA-Corps of Engineers survey.

were formed from early 20th century dumping of assorted building excavation material and sand and gravel from the dredging of Ambrose and Sandy Hook Channels (Williams 1975).

Comparison of the 1973 bathymetric survey results with data from the 1936 survey reveals that only the anthropogenic areas have changed significantly. Figure 4 shows the 1973 and 1936 bathymetry of the dredge spoil site, as well as the net change between the two surveys. The 50-ft knoll on the 1936 map (relatively unchanged in 1973) is itself the result of earlier dumping (Williams 1975). The amount of anthropogenic material accumulated during these years (1936–1973) has been calculated to be about 124×10^6 m^3. This compares with about 142×10^6 m^3 dumped. The difference easily can be accounted for by settling alone.

Surficial sediments have been mapped by analyzing over 700 bottom grab samples collected at 1-km spacing (Fig. 5). The topographically low Hudson Shelf Valley and the Christiaensen Basin are floored

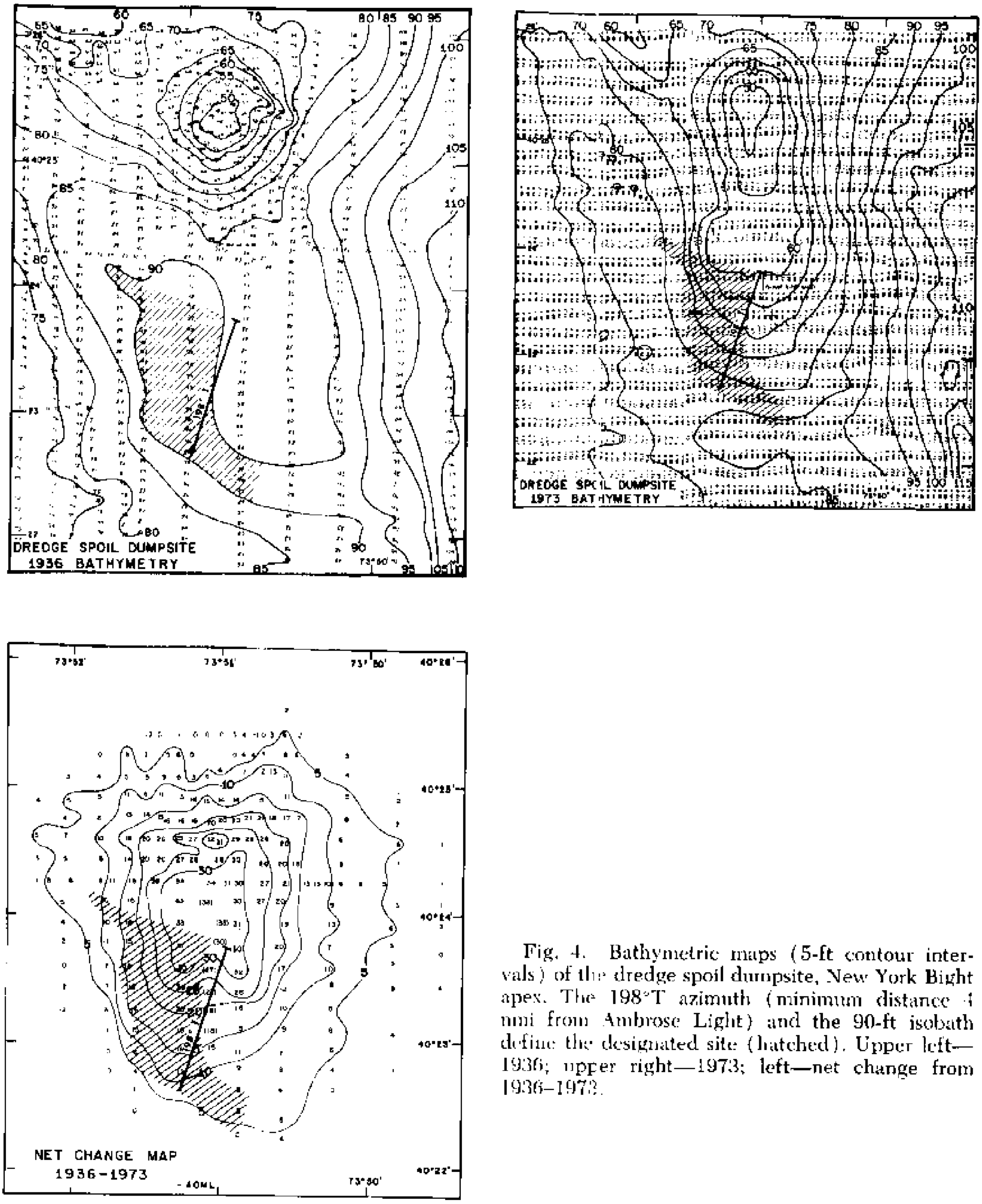

Fig. 4. Bathymetric maps (5-ft contour intervals) of the dredge spoil dumpsite, New York Bight apex. The 198°T azimuth (minimum distance 4 nmi from Ambrose Light) and the 90-ft isobath define the designated site (hatched). Upper left—1936; upper right—1973; left—net change from 1936–1973.

with fine-grained sediment, whereas the rest of the area contains assorted sizes of sand and both anthropogenic (artifact) and natural gravel deposits. Artifact gravels consist of recognizable construction rubble —brick, schist, concrete, etc.

Geophysical data taken during the 1973 survey consisted of 3.5-kHz shallow-penetration seismic reflection records and side-scan sonar records with 150-m range on each side of 610-m-spaced tracklines. Although data interpretation is incomplete, bottom roughness patterns and trends of linear bedforms (sand ribbons and degraded sand waves) have been mapped from side-scan data (Fig. 6). These bedforms appear as alternating light and dark bands corresponding to fine- and coarse-

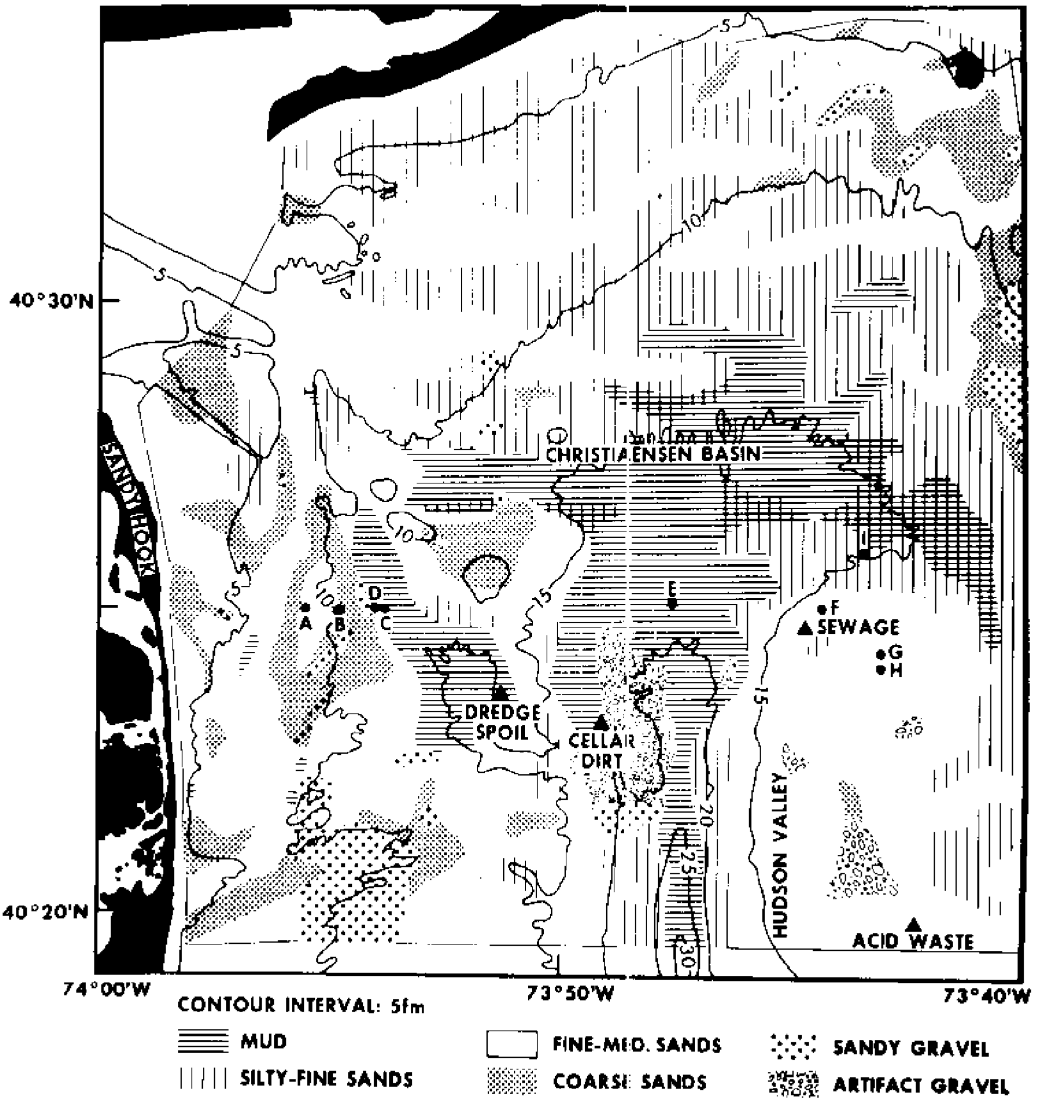

Fig. 5. Distribution of surficial sediment based on visual sample examination. Bathymetry from 1936 data.

grained sediment or as isolated dark bands. Streaky, patchy, and rough textures are associated with the dredge spoil and cellar dirt dumpsites and may be related to individual dumps.

Preliminary analysis of seismic data shows filling of the Hudson Shelf Valley from Cholera Bank.

Suspended sediment studies are particularly important in the bight apex because of the large amounts of fine particles dispersed in the water by waste disposal operations. These particles are in addition to the fine sediments discharged from the Hudson River, other river mouths, and tidal inlets connected to coastal wetlands. Fine-grained sediment is also eroded from the sea floor during storms. Of immediate concern is sewage sludge which contains bacterial, viral, and heavy metal contaminants that adhere to fine sediment particles in the water column. The suspended fraction of dredge spoils is also probably similarly contaminated. All of these fines are largely re-

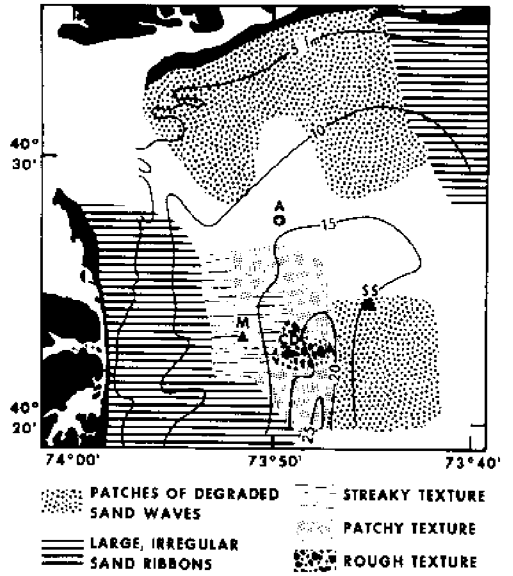

Fig. 6. Distribution of bottom roughness patterns from sidescan sonographs. Blank area NW and SE of Ambrose Light (A) shows no bedforms. M—Dredge spoil dumpsite; CD—cellar dirt site; SS—sewage sludge site.

tained in the nearshore water column as a consequence of the bight circulation pattern.

Suspended sediment studies were initiated in the bight apex during 1973 when sample stations were occupied to collect chemical and physical oceanographic data. Water samples were collected, filtered, and examined from the surface, 10-m depth, and the bottom at 25 stations. Preliminary results for data taken in fall 1973 (Drake 1974; Figs. 7–10) indicate the existence of a fair-weather, clockwise current-circulation gyre, driven in part by the southwest drift of offshore shelf water. This has been verified by current meter studies in the

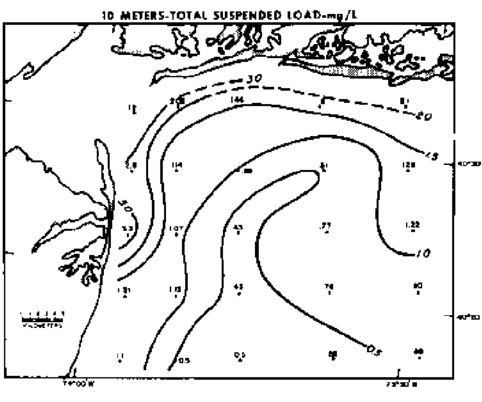

Fig. 7. Total suspended sediment load in waters at 10-m depth, late November 1973. (From Drake 1974.)

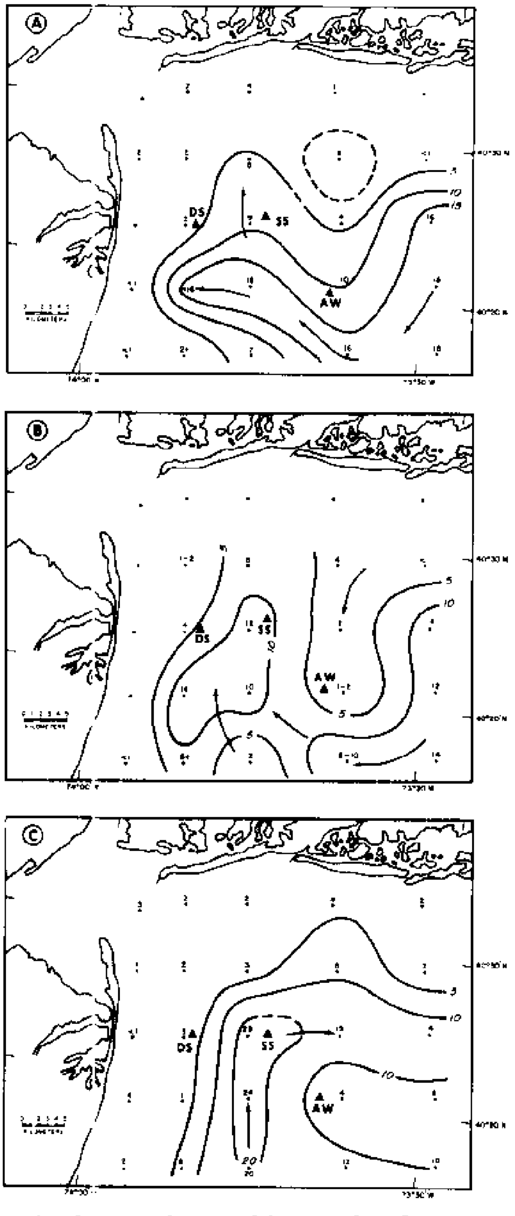

Fig. 8. Distribution of ferric hydroxide particles in the water column in late November 1973 (grains × 10^3/liter). A—Surface; B—midwater; C—bottom water. (From Drake 1974.)

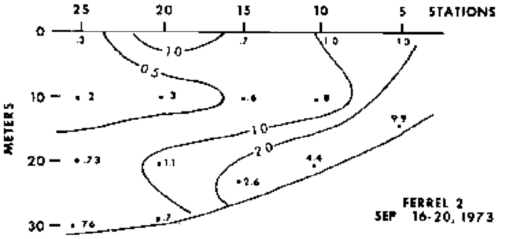

Fig. 9. Vertical distribution of total suspended load (in mg/liter) seaward of Long Beach, Long Island. (From D. E. Drake unpublished.)

apex (Charnell and Hansen 1974). Part of the total suspended load in the bight apex is easily identifiable, red-orange ferric hydroxide particles. These particles are formed by precipitation of iron in seawater as the result of acid waste dumping. They constitute an excellent tracer of suspended sediment circulation. The vertical distribution of suspended sediment shows high values (1.0 mg/liter) near the surface, and 2.0 mg/liter in the near-bottom "nepheloid" layer, typical of shelf areas (Fig. 9). It is expected that this layer will transport much of the suspended particulate matter and its associated contaminants.

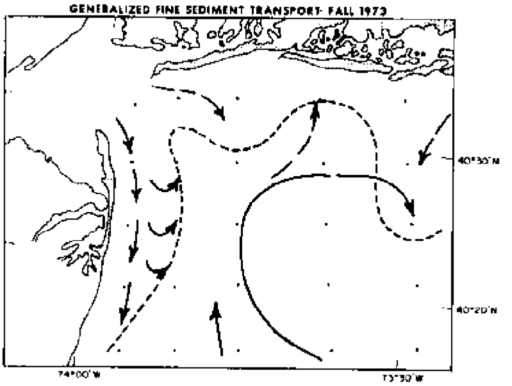

Fig. 10. Fine sediment transport system as inferred from distribution of suspended sediments during fall 1973. Dashed line is mean position of boundary between more turbid coastal water and less turbid offshore water. Clockwise gyre is apparently driven by southwesterly drift of offshore shelf water, and, on the bottom, by influx of saline water into New York Harbor. Regional currents which appear to be persistent are indicated by solid arrows. (From Drake 1974.)

Preliminary results show there is a concentration of fine-grained sediment in enclosed lows in the Hudson Valley axis, sandy mud in the remainder of the valley axis, and coarser sediment up the flanks of the valley and onto the shelf.

Alternative dumping area studies

Two midshelf areas have been designated as possible interim alternative dumping areas for sewage sludge and dredge spoils from the New York metropolitan area (*see Fig. 1*). The northern area is to be a minimum of 46 km from the Long Island shoreline, 18 km from the axis of the Hudson Shelf Valley, and 120 km from the entrance to New York Harbor. The southern area is seaward of the 36-m isobath and the same distances from the Hudson Valley axis and the New York Harbor entrance as the northern area (areas 2D1 and 2D2 on Fig. 1). Each area is 18.5×18.5 km.

Northern area—In the northern area (Fig. 1, 2D1), the sampling grid was placed seaward of the center of the location-criteria triangle to investigate, in part, a shallow tributary valley of the ancestral Long Island drainage system. The surficial sediments consist of sand with some areas of over 5% gravel (Fig. 11). Fine sands lie in the northeastern part of the area, medium sands cover the western and southern parts, with a gravel deposit (≤39% gravel) at one station associated with an area of coarser medium sand in the southern part of the area. Only two stations contained >5% mud. Bottom photographs indicate that the area is characterized by a smooth, slightly undulatory, mounded or rippled bottom. Sidescan sonar records reveal elongate dark areas which may be erosional windows in the Holocene sand sheet that expose the basal Holocene pebbly sand or may be areas of abundant large shell fragments. Grab samples were spaced too far apart to be definitive. Bottom photo and submersible-observation data support the existence of windrows of shell fragments.

Southern area—The southern study area in Fig. 1 (2D2) is centered over the broad, flat high of the Hudson divide (Fig. 12). To

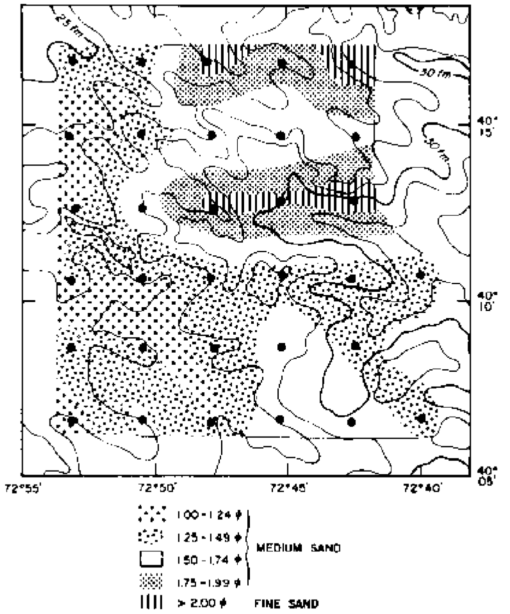

Fig. 11. Northern proposed interim alternative dumping area (2D1 on Fig. 1). Grain-size distribution of sand-sized fraction. Large dots—sample stations. (Bathymetry from Stearns and Garrison 1967; 1-fm contour intervals.)

the northeast the bottom grades gently into the Hudson Shelf Valley, while the western section is characterized by northeast-trending ridge and swale topography. Geophysical data, sediment samples, and two dives in submersibles showed that grain-size patterns appear to be related to bottom topography; coarser sand and gravel deposits lie on the crest and east flank of the Hudson divide, while medium- and fine-grained sand occur in the ridge and swale topography (Fig. 13). These distributions suggest that fine sediment is winnowed from the crest and east flank of the divide and deposited to the west. Observations from a submersible in Veatch and Smith Trough reveal a veneer of shelly, pebbly sand with large, angular clay pebbles and occasional oyster shells derived from the underlying early Holocene lagoonal clay. Seismic data also reveal that the reflector associated with this surface outcrops on the ridge flank. It appears that storm-generated currents from the northeast have winnowed the east flank of the Hudson divide and formed or maintained the ridge and swale topography on the west side of the divide.

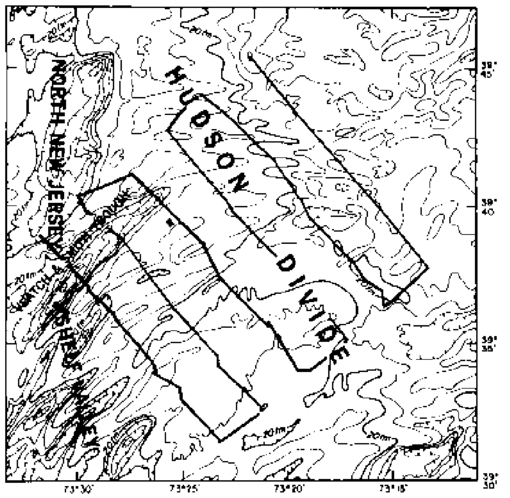

Fig. 12. Southern proposed interim alternative dumping area (2D2 on Fig. 1). (Bathymetry from Stearns and Garrison 1967, 1-fm contour intervals.) Solid lines—geophysical tracklines; bars—sites of dives by submersibles.

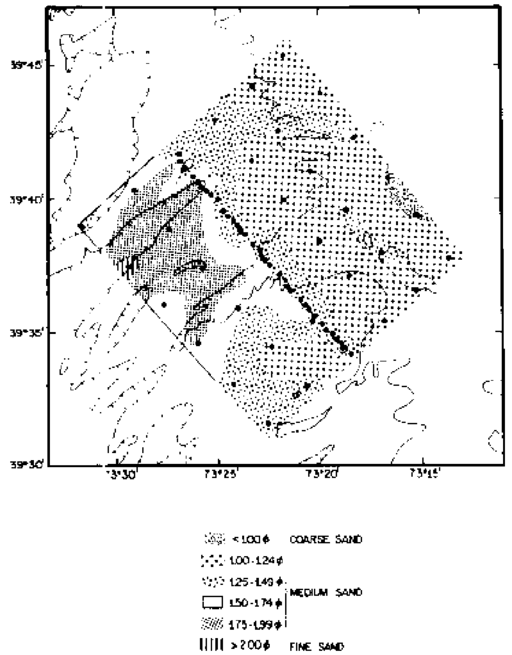

Fig. 13 Southern proposed interim alternative dumping area, (2D2 on Fig. 1). Grain-size distribution of sand-sized fraction. Large dots—sample stations. (Only the 20-fm isobath is shown.)

Suspended sediment—As previously mentioned, most fluvial suspended sediment is effectively trapped in estuaries and coastal wetlands. Consequently, the terrigenous fraction of the suspended matter decreases rapidly seaward. Suspended solids throughout the water column in the alternative dumping areas were predominately plankton and their noncombustible remains. Total suspended matter concentration in surface water is from 100–500 μg/liter, comprised of 5% or less terrigenous matter, 80% combustible matter, and 15% siliceous and calcareous noncombustible planktonic remains (D. E. Drake personal communication). Subsurface water–suspended matter concentration is similar or somewhat less, except in the nepheloid layer 5–10 m above bottom. There, suspended matter concentrations are 500–2,000 μg/liter, consisting of 30–60% combustible matter and 50–80% noncombustible matter which includes 10–20% terrigenous matter. Textural properties of sediment deposits in the alternative dumping areas show that very little sediment finer than 62 microns is present.

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Spatial and temporal variation in sedimentary grain-size facies and sediment heavy metal ratios in the New York Bight apex

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Abstract

In northwest New York Bight a pronounced spatial and temporal variation occurs in sediment grain-size, heavy metal ratios, and concentrations. Substrate mobility is most pronounced near Long Island and in northern Christiaensen Basin. Near Long Island mud patches are most extensive during late spring through summer and may coalesce across intervening sand-wave crests obliterating the shore-zone sand-wave topography. Mud patches are either restricted to sand-wave trough axes or are absent or covered during early fall through early spring, suggesting that muddy sediments are either flushed from the nearshore sand-wave system by bottom currents or undergo in situ microbial degradation.

The northwest periphery of the Christiaensen Basin mud-sandy mud facies apparently oscillates in position. Mud patches in the peripheral zone expanded shoreward from 6.9 km to within 5.0 km of the Long Island shoreline between winter and summer 1974 and contracted seaward to within 9.3 km of the shoreline by early spring 1975; isolated mud patches remained within 6.5 km of the shoreline. Shoreward expansion and seaward contraction of the mud facies of the northwest Christiaensen Basin occurred during the same months as did nearshore mud patch growth and size reduction or disappearance; probably both instances were the result of the same processes.

Individual mud patches near Long Island have not existed for a long time. Mud patches off Atlantic Beach and Lido Beach, Long Island, began to develop as early as summer 1972; others developed later.

The Cr:Zn ratio varies in mud deposits at the sludge disposal area, northern Christiaensen Basin, and Hempstead Bay and in mud patches near Long Island. Maximum Cr:Zn values occur during late spring through early fall, minimum values during early winter.

Zinc was released to bight waters preferentially relative to Cr for which carbon-rich muds apparently served as a sink during spring–summer 1974 expansion of northwest Christiaensen Basin mud facies and growth of mud patches near Long Island. During later contractions of the northwest Christiaensen Basin mud facies and attendant decrease in size or disappearance of nearshore mud patches from fall 1974 through early spring 1975, Zn and Cr both were released at nearly equal rates or Zn was preferentially retained in the sediment. The Cr:Zn annual maximum may be explained by preferential zinc desorption from sludge or mud solids by cation exchange, or preferential adsorption of organic chromium chelates from solution by organic mud substrates, or zinc uptake by phytoplankton or bacteria. The Cr:Zn annual minimum may be explained by dissolution of Mn and Fe hydroxides releasing adsorbed Zn and Cr or oxidation of the organosulfur compounds of the two metals at nearly equal rates.

During ocean disposal only a small fraction of the sludge particles penetrate the pycnocline to accumulate in the disposal area; most are rapidly dispersed in the water (Clark et al. 1971; Callaway et al. 1976). During transport, sludge and natural organic particulates should tend to settle in bathymetric depressions usually characterized by reduced wave and current action, such as the Christiaensen Basin (headward portion of the Hudson Shelf Valley) and the area east of Cholera Bank (Fig. 1).

Sludge particles dispersed in the water during ocean disposal are eventually incorporated in sediment deposits. High carbohydrate content (Walter 1961), mostly cellulose and hemicellulose (Hunter and Heukelekian 1965), characterizes sewage sludge, much of the organic matter in muds of the Christiaensen Basin, and most other sediment substrates of the bight apex, suggesting that most of the organic matter in these sediments comes from sewage (Hatcher and Keister 1976). Fecal steroids, coprastanol, and 24 β-ethyl coprastanol in Christiaensen Basin muds (Hatcher et al. 1976) have been used to show that sewage sludge is a source of organic matter. After

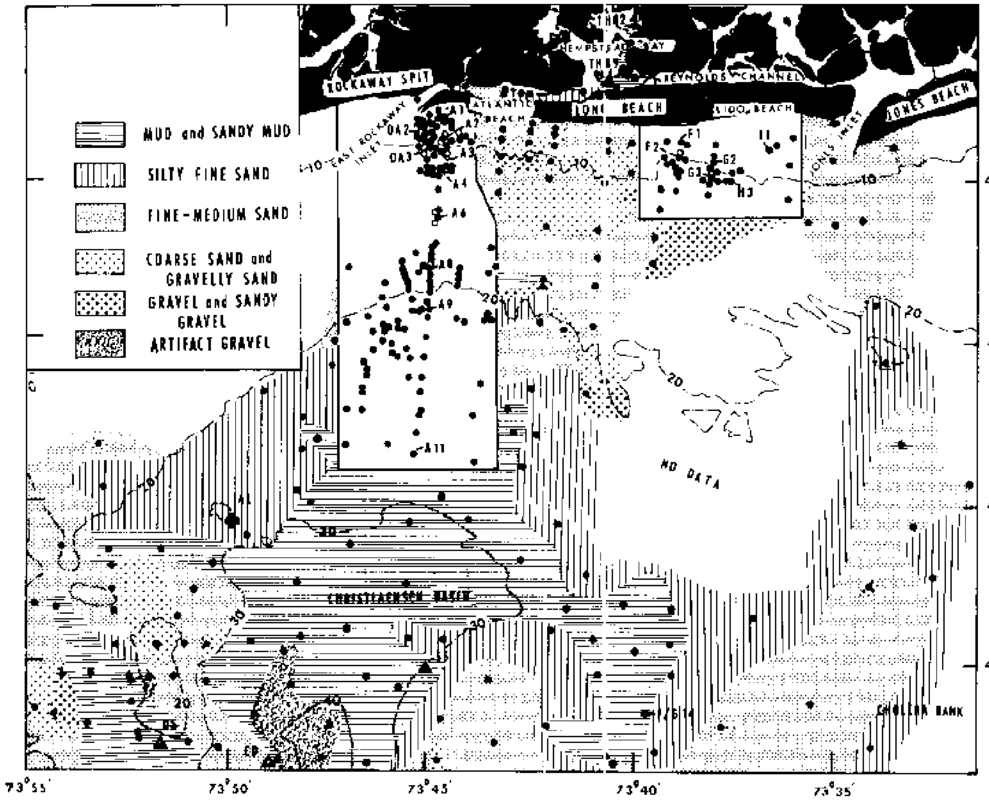

Fig. 1. Sampling locations and April–August 1974 grain-size facies of the northwest New York Bight apex (except for areas of most pronounced temporal substrate mobility south of Atlantic Beach and Lido Beach). Depth in meters. AL—Ambrose Light. Disposal sites for wastes from New York metropolitan area; SS—sewage sludge (northwest corner of disposal permit area); DS—dredge spoil; CD—cellar dirt. (Letter-number sampling stations are discussed in text.)

deposition, fine solids may be readily resuspended and redistributed by wave action and currents caused by storms (Harris 1974; NOAA 1975). Although NOAA (1975) maintains that the 50-year practice of dumping sewage sludge has produced a rather stable distribution pattern of muds in the bight and that this distribution pattern is not likely to change much with time, it is now apparent that whereas no significant transport of dredge spoils has occurred (Williams and Duane 1974), a portion of the sewage sludge solids has nonetheless moved to the north of the present sewage sludge disposal area, and, in particular, into the Christiaensen Basin and nearshore Long Beach area (Pararas-Carayannis 1975; Harris 1974; Maher 1976).

Black silty mud deposits of unknown origin, sometimes resembling sewage sludge, have been detected near Long Island beaches (Harris 1974; Charnell 1975). These muds have high metal concentrations (Harris 1974). They also have high total carbohydrate to total organic carbon ratios (TCH:TOC), suggesting that their organic matter came from sewage although not necessarily sewage sludge (Hatcher and Keister 1976). NOAA (1975) viewed these deposits to be mostly of natural origin with small admixtures of material derived from sewage and to have existed for a long time. Udell et al. (1974) compared organic carbon content and heavy metal concentrations in inshore shelf sands and gravels with those in sediments taken from East Rockaway and Jones Inlets and from behind the Long Beach barrier island in Reynolds Channel and concluded that nearshore Long Island sediments south of Long Beach are influenced by tidal flushing of Hempstead Bay. No nearshore mud patches were sampled during the Udell et al. study. Harris (1974) sampled nearshore Long Island mud patches as well as other nearshore sediments and concluded that, based on selected heavy metal ratios, these mud patches contained sludge solids derived from the Christiaensen Basin and sewage sludge disposal area. His conclusions were based principally on use of the ratio of chromium to zinc (Cr:Zn) in nearshore mud patch sediments taken during fall 1973 and winter 1973–1974. These views are now subsequently to be modified.

My purpose here is to—

1. Evaluate stability of distributions of organic-rich muds in the bight apex by examining changes in distributions of organic-rich muds along the northwest edge of Christiaensen Basin and near Long Island.
2. Determine age of individual nearshore Long Island mud patches by direct observation of substrate change to mud and by examination of the relative abundance of mud-dwelling benthic species and of the age classes of *Nucula proxima* in these mud patches.
3. Identify sources of organic-rich muds in nearshore Long Island mud patches by comparing concentrations of Cr and Zn in mud deposits near Long Island with those of muds from Christiaensen Basin (Zn-rich) and Hempstead Bay (Cr-rich).
4. Determine the distributions of sewage sludge solids between bight sediment and overlying waters by examining and comparing spatial and temporal variations in the Cr:Zn ratio in muds of the sewage sludge disposal area, northern Christiaensen Basin, Hempstead Bay, and near Long Island.
5. Demonstrate utility of the Cr:Zn ratio as a tracer for the movement of organic mud particulates from potential sources to nearshore Long Island mud patches.

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Methods

Stations (309) studied in the northwest apex of the bight (Fig. 1) encompassed the sewage sludge, cellar dirt, and dredge spoil dump sites and other areas including Cholera Bank, Christiaensen Basin, and the nearshore Long Island shelf between East Rockaway and Jones Inlets. Sediments were sampled in August, September, October, and December 1973, January, February, April, May, June, July, August, October, and December 1974, and April 1975. Each station was sampled from 1 to 12 times. Stations from Hempstead Bay (16) behind the Long Beach barrier island, from the nearshore Long Island shelf up to 23 km south of Fire Island Lighthouse (12), and stations (17) up to 115 km down the Hudson Shelf Valley were sampled for comparative purposes.

Sediment samples were obtained with Petersen, Ponar, Shipek, or Smith-McIntyre grab samplers. Replicate bottom grabs were taken at most stations having a muddy substrate. Aliquot samples of the total sediment thickness penetrated by the grab (maximum depth of 15 cm) were taken and stored in 1:1 nitric acid-washed polypropylene jars, and oven-dried at 60°C. Dried sediment samples were subsampled for heavy metals, organic carbon, and size analysis. After removal of sediment samples for the above analyses, the remaining sediment was washed through stainless steel sieves. Organisms retained on a 6.35-mm-mesh sieve and all materials on the 1-mm-mesh sieve were preserved in neutralized Formalin and stained with Rose Bengal.

Particle-size distributions (63 μm to 4 mm) were measured using 8-inch-diameter sieves at 0.5-phi intervals and an additional sieve at 53 μm (4.25 ϕ). Most sediments contained too little fine sediment (finer than 4.0 ϕ or 63 μm) to warrant pipette analysis. Coarse shell material was removed before sieving.

When silt and clay were more than a few percent of the sample, the silt-clay fraction was separated from the remaining sands and gravel by wet-sieving; coarser particles were analyzed by dry-sieve analysis. The silt-clay abundance was estimated from the difference in oven-dry weight between the initial sample and the material retained on the 63-μm sieve (Holme and McIntyre 1971). Silt-clay fractions were analyzed by pipette analysis (Folk 1974; Galehouse 1971). Graphical methods (Folk and Ward 1957; Friedman 1967) were used for textural analysis and textural parameters were calculated (Krumbein 1938).

Formalin-fixed collections were sorted (Day et al. 1971) and individual organisms were handpicked, identified, and counted. Shell length of *N. proxima* was measured with a calibrated stereomicroscope.

Age classes of *N. proxima* were determined by size frequency analysis of shell length. Attempts were also made to correlate growth interruption lines on the shell with the size frequency data. Difficulties are often encountered with Mollusca (Haskin 1954; Wilbur and Owen 1964) because seasonal rings often are indistinguishable from disturbance or reproductive rings.

Subsamples of muddy sediments were wet-sieved through a 2-mm (-1.0 ϕ) sieve then ground, dried, and weighed (0.2–0.5 g). Readily oxidizable organic carbon was determined by the Walkley and Black (1934) method, adapted and modified from Jackson (1958) by Gaudette and Flight (1974). The results have a precision of ±0.25%. The modified Walkley-Black titration method correlates well (0.989) with the LECO combustion method (Lab. Equip. Corp. 1959).

Dried, ground subsamples for heavy metals analysis (25 g) were extracted in 1:1 nitric acid-washed polycarbamate evaporating dishes with 20 ml of aqua regia. Solutions were evaporated to near dryness under IR lamps. To these were then added 20 ml of boiling concentrated nitric acid and the solutions were evaporated to half volume under IR lamps. Samples were filtered through Whatman No. 40 filter paper. Sediment in the filter funnels was doubly washed with 75 ml of 1:1 nitric acid followed by double washes of 25 ml of distilled-deionized water. Filtered extracts, including washings, were combined and diluted to constant volume (250 ml), 34% in concentrated nitric acid with distilled-deionized water, and represent a 1:10 dilution of the dry weight sample (25 g). Filtered samples were analyzed in a Jarrell Ash model 82-520 atomic absorption/atomic emission spectrophotometer. Zinc was analyzed in an air-acetylene flame (213.9 nm) with a spectral bandpass (SBP) of 0.24 nm and chromium in a nitrous oxide-acetylene flame (357.9 nm) with a SBP of 0.16 nm. Nonatomic absorption was determined using a hydrogen continuum hollow cathode lamp for background correction. Standards and blanks were prepared for all determinations and corrections applied. Analyses were made directly of 1:10 dilutions of the original dry weight sample without further dilution by varying the sensitivity of the atomic absorption analysis and comparing absorbance or peak height, utilizing recorder readout, with metal standards of identical nitric acid concentration.

No attempt was made to separate adsorbed, reducible, or oxidizable fractions of Zn and Cr. Analytical results thus include the sum of these three fractions but exclude metals bound in silicate lattices. When analytical sensitivity and replicate sample variability are considered, my reported results have a precision of 0.1 (<10 mg/kg), 3.0 (10–25 mg/kg), 7.0 (25–99 mg/kg), and 9.0 mg/kg (>100 mg/kg) for zinc and 0.1 (<10 mg/kg), 1.0 (10–25 mg/kg), 3.0 (25–99 mg/kg), and 4.0 mg/kg (>100 mg/kg) for chromium.

Results

Sedimentary grain-size facies—Within the northwest apex of the bight, the most common sediments are silty fine sand and slightly gravelly fine to medium sand. Mud and sandy mud characterize the central Christiaensen Basin and are intercalated with silty fine to medium sand along its northern periphery. Although some sandy mud is present within the EPA-designated sewage sludge disposal area, most of this 22.5-km^2 permit area is relatively free of mud and is instead floored by silty fine to medium sand. Mud and sandy mud also characterize the dredge spoil disposal area (Fig. 1). Near Long Island south of Long Beach Island there is a complete spectrum of sedimentary grain-size facies, ranging from sandy pebble or granule gravels and gravelly coarse sand to silty fine sand, sandy mud, and mud (Figs. 1 and 2).

Substrate mobility—Sand waves are well developed near Long Island during winter and early spring (Fig. 2A). The wavelengths are 3–10 m, with amplitudes of about 1 m; individual crests are 30–60 m long. Troughs open to the east-southeast. Sediment grain-size changes rapidly between adjacent sand-wave crests, especially south of Atlantic Beach (Figs. 1 and 2A). Wave crests consist of fine to medium sand; intervening troughs are floored by either very fine sand, muddy silt, or sandy mud. Medium to coarse sandy-pebble gravel or granular-pebble gravelly coarse sand is exposed on the south side of some troughs (Fig. 2A). Muddy sediments if present at this time usually occur on the north side of troughs and may be intimately mixed with

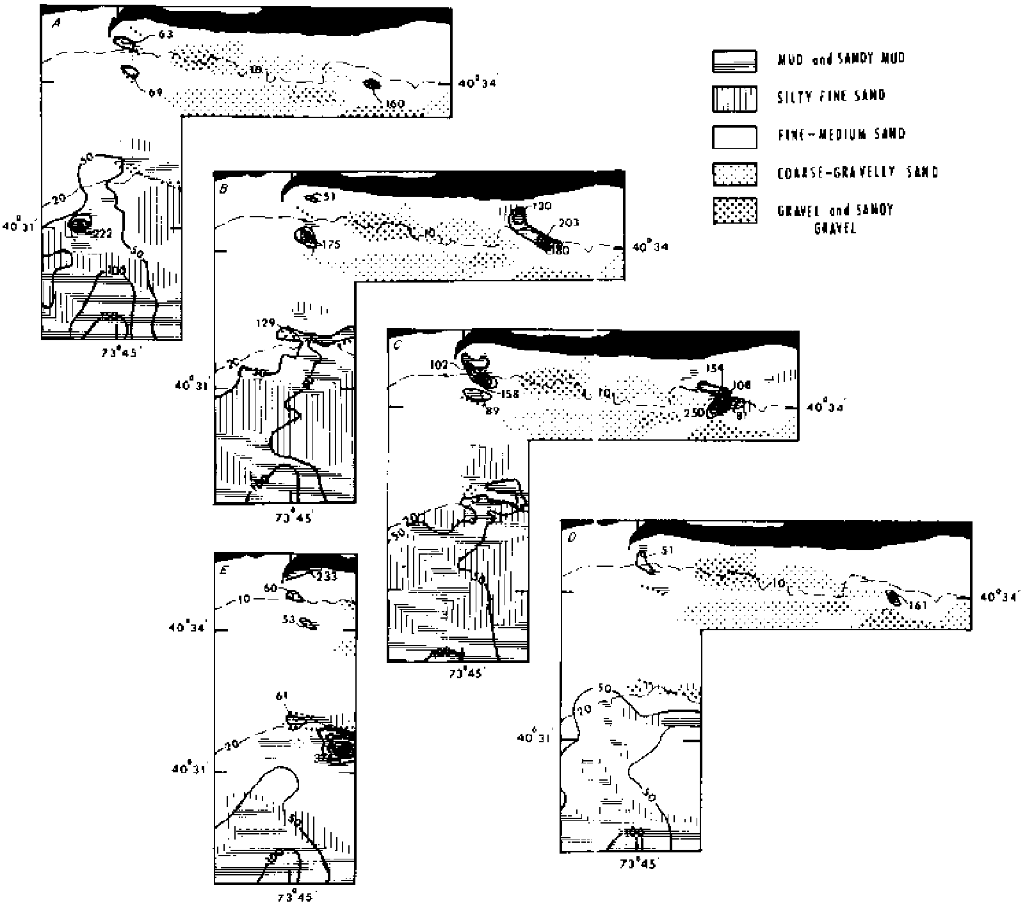


Fig. 2. Grain-size facies and Zn levels (mg/kg, dry wt) at successive times in areas of most prevalent temporal substrate mobility: A—December 1973–February 1974; B—April–May 1974; C—June–August 1974; D—October–December 1974; E—April 1975. Depth in meters. Zn-isopleth contour interval—50 mg/kg. Numbers indicate maximum Zn concentrations >50 mg/kg in nearshore Long Island mud patches and along the periphery of the Christiaensen Basin mud-sandy mud facies. High levels of Zn (>200 mg/kg) on the beach in April 1975 (E) reflect strandline fecal pellet deposits.

coarser sands and gravels. Many troughs were free of muddy sediments during winter and early spring.

Degradation of sand waves occurred during late spring and summer. Troughs containing muddy sediment during winter and early spring 1974 were within 5.8 and 3.0 km of Atlantic Beach and Lido Beach shorelines. Sand waves were degraded by filling the troughs with muddy sediment. Adjacent troughs within 2 and 3 km of Atlantic Beach and Lido Beach were filled with muddy silt or sandy mud, so that muds coalesced over the crests and obliterated the sand waves (Fig. 2A–C). Sand waves began to reappear in early fall 1974 as muddy silt and sandy mud were flushed from the area of nearshore sand waves or decomposed by microbial action (Fig. 2D).

Substrate mobility along the northern periphery of the Christiaensen Basin is characterized by expansion and contraction of the fine-grained, poorly sorted deposits common to the basin (Figs. 1 and 2). Mud patches of the north-central and northwest peripheral zone extended in winter 1973–1974 to within 10.2 and 6.9 km of Atlantic Beach (Fig. 2A) and expanded northward to within 5 km of the Long Island shoreline by summer 1974 (Fig. 2C). At that time mud patches of the northwest Christiaensen Basin coalesced with the nearshore shelf mud patches near A8 (Fig. 1) which had been accreting mostly southward since late spring 1974 (Fig. 2B–C).

The outermost nearshore mud patches observed are apparently not controlled by sand-wave troughs but by a seaward-facing scarp having 2 m of relief near the 20-m isobath (Figs. 1 and 2). By summer 1974 (Fig. 2C), muddy medium silt and sandy mud had been deposited over most of the scarp face up onto the 18-m platform to the north and had accreted 0.5 km southward from the scarp face (Fig. 2B–C).

Eastward as well as northward expansion of the northwest peripheral zone of the Christiaensen Basin mud facies, beginning in spring 1974, occurred at 7 km south of Atlantic Beach (A9) (Figs. 1 and 2B). By winter 1974–1975, the northern peripheral zone had retreated to 6.7 km (Fig. 2D) and by early spring 1975 (Fig. 2E) to within 9.3 km of the Long Island shoreline. Isolated mud patches remained within 6.5 km of the shoreline. As a result of the retreat of the Christiaensen Basin muds, the outermost nearshore mud patches near the 20-m isobath were isolated from the muds of northern Christiaensen Basin (Fig. 2E). Beginning in winter 1974–1975, fine to medium sand replaced silty very fine to fine sand shoreward of the Christiaensen Basin muds (Fig. 2C–E). Inshore of the 13-m isobath at the eastern and western ends of the Long Beach barrier the nearshore mud patches off Lido Beach had mostly disappeared (Fig. 2D) whereas off Atlantic Beach, all but two (A3, A4) were in part covered over by about 6–7 cm of rippled fine sand (Fig. 2E).

Duration of nearshore mud patches—Thirteen benthic species characteristic of the *Nucula/Nephthys/Yoldia* infaunal mud community described by Sanders (1956, 1958, 1960) in Long Island Sound and Buzzards Bay occur in about half the nearshore Long Island mud patches sampled, irrespective of the available substrate, viz very fine sand, muddy silt, or sandy mud. Franz (1976) noted the extension of the *Nucula/Nephthys/Yoldia* infaunal mud community into very fine sand sediment in northeastern Long Island Sound and Fishers Island Sound.

Species of the *Nucula/Nephthys/Yoldia* community from nearshore Long Island mud patches, retained on a 1-mm screen, were bivalves (*N. proxima, Pitar morrhuanna*, and *Yoldia limatula*), gastropods (*Cylichnella oryza* and *Tornatina canaliculata*), polychaetes (*Lumbrineris fragilis, Lumbrineris tenuis, Glycera dibranchiata, Glycera capitata, Nephthys picta*, and *Nephthys incisa*), amphipods (*Ampelisca abdita*), isopods (*Edotea triloba*), and cumaceans (*Diastylis sculpta*). The bivalve *Tellina agilis* was restricted to the 2.0–2.5-mm size range for individuals collected from nearshore mud patches as compared to a spectrum of sizes from 2.0–8.9 mm long for this species in immediately adjacent sands, where it was 7.5 times as abundant.

Analysis of *Nucula* shell size and growth annuli in winter 1973–1974 and summer-fall 1974 collections revealed several well defined age classes from 25% of the mud patches sampled; all other mud patches where *Nucula* was present contained *Nucula* of zero-age class. Growth patterns of *Nucula* showed sharply delimited daily growth bands with 30-day monthly clustering which is probably tidally controlled. Age classes of *N. proxima* and the presence of most of the above species of the *Nucula/Nephthys/Yoldia* infaunal mud community and only dead valves of juveniles of the bivalve *Spisula solidissima* suggest the continued presence of fine-grained sediments since summer 1972 at one mud patch and since summer 1973 at two mud patches within 2.8 km of the Long Island shoreline.

At ten mud patches, *Nucula* was either absent or of zero-age class. *Nucula* age-class data for these mud patches, the presence of juvenile *Spisula* of zero-age class, of *Tellina* from 2–9 mm long, and no evidence of long term colonization by mud dwellers indicate that these sites had not been affected by muddy sediments long enough for any faunal changes to have occurred; they probably developed beginning in winter or early spring 1974. As nearshore mud patches were seen both to develop and to be covered over by sand during the study period, age classes of *N. proxima* in nearshore Long Island mud patches indicate the time that an individual mud patch last began to develop (Figs. 1 and 2).

Organic carbon—Waste deposits in the sewage sludge disposal area and muds in the Christiaensen Basin contain 0.1–4.5% (dry wt) total organic carbon (TOC). Deposits near the dredge spoil disposal area contain 2–9% TOC. Mud patches in the nearshore Long Island zone contain 0.4–3% TOC off Atlantic Beach and 3–7% TOC off Lido Beach. TOC concentrations in Christiaensen Basin and Atlantic Beach mud patches varied seasonally with maximum concentrations in late spring and summer. Mud patches off Lido Beach and Hempstead Bay muds showed comparable TOC maxima in late spring through early fall.

Chromium and zinc—"Normal" Cr and Zn concentrations in muddy sediment uncontaminated by wastes were calculated based on samples of muddy medium silts from the Hudson Shelf Valley, 115 km from Ambrose Light, presumably seaward of any influence of the dump sites in the bight apex. Normal Cr and Zn concentrations in fine to coarse shelf sands were estimated using sands collected up to 23 km south of Fire Island Lighthouse (Table 1).

Because of temporal variability in the concentrations of Zn and Cr in muddy sediments of the bight apex (Table 2), the following comparisons are made for June–September 1974—the period of maximum mud dispersal (Figs. 1 and 2). By comparison, waste deposits in the dredge spoil disposal area had maximum Zn concentrations that were 8.2 times greater, whereas those in the sewage sludge disposal area were equal to or 1.4 times greater than the mean zinc concentration in Hudson Valley normal muddy sediments (Tables 1 and 2). Maximum Zn concentrations in the mud-sandy mud facies of the southern and northern Christiaensen Basin, in Atlantic Beach and Lido Beach mud patches, and in Hempstead Bay muds were 8.1, 2–5, 4, 6.4, and 5.3 times greater than the mean Zn concentration of normal muddy sediments (Tables 1 and 2).

Similarly, Cr was enriched in muddy sediments of the bight apex relative to Hudson Shelf Valley normal muds. Enrichments range from 14×, 7.5×, 6.6×, and 7.8× in muds of the southern and northern Christiaensen Basin, dredge spoil disposal area, and Lido Beach mud patches to 4× and

Table 1. Concentration of Cr and Zn in sediment unaffected by dumping (mg/kg in dry sediment) during June–August 1974.

	Chromium			Zinc		
	Min	Max	Mean	Min	Max	Mean
Shelf sands off Fire Island						
Fine	0.5	9.2	4.2	2.0	9.5	6.1
Medium	4.3	9.6	7.4	6.0	10.0	7.7
Coarse	1.9	7.0	4.5	1.0	5.0	3.0
Muddy silt of Hudson Shelf Valley			25			39

Table 2. Temporal variation by area in the concentration of Cr and Zn and the magnitude of the Cr:Zn ratio in organic-rich muds taken in various areas of the northwest New York Bight apex, nearshore Long Island, and Hempstead Bay.

		Cr	Zn	Cr:Zn
September 1973				
South and central Christiaensen Basin (10 samples)	Min	55	110	0.26
	Max	343	317	2.21
	Mean	137	182	0.95
North Christiaensen Basin (8 samples)	Min	41	41	0.32
	Max	188	201	1.45
	Mean	74	124	0.73
Sewage sludge disposal area (3 samples)	Min	18	44	0.50
	Max	25	56	0.45
	Mean	20	48	0.42
Atlantic Beach nearshore Long Island mud patches (1 sample)		12	54	0.22
December 1973–February 1974				
North Christiaensen Basin (16 samples)	Min	11	41	0.24
	Max	58	222	0.41
	Mean	31	94	0.33
Atlantic Beach nearshore Long Island mud patches (6 samples)	Min	6	44	0.14
	Max	26	72	0.37
	Mean	18	56	0.31
Lido Beach nearshore Long Island mud patches (3 samples)	Min	43	95	0.45
	Max	131	247	0.63
	Mean	91	134	0.54
April–May 1974				
North Christiaensen Basin (3 samples)	Min	45	59	0.49
	Max	84	126	0.76
	Mean	62	100	0.64
Sewage sludge disposal area (3 samples)	Min	13	33	0.29
	Max	28	67	0.46
	Mean	19	48	0.39
Dredge spoil disposal area (3 samples)	Min	70	114	1.08
	Max	134	239	1.56
	Mean	101	178	1.26
Atlantic Beach nearshore Long Island mud patches (9 samples)	Min	27	28	0.42
	Max	148	175	0.85
	Mean	138	69	0.58
Lido Beach nearshore Long Island mud patches (4 samples)	Min	73	83	0.56
	Max	107	192	0.90
	Mean	90	130	0.73
Hempstead Bay Long Island (7 samples)	Min	20	39	0.26
	Max	151	125	1.31
	Mean	86	90	0.78
June–August 1974				
South Christiaensen Basin (1 sample)		16	62	0.26
North Christiaensen Basin (3 samples)	Min	22	43	0.51
	Max	38	63	0.60
	Mean	31	56	0.54
Sewage sludge disposal area (2 samples)	Min	15	33	0.46
	Max	18	39	0.46
	Mean	17	36	0.46
Dredge spoil disposal area (6 samples)	Min	15	51	0.29
	Max	166	320	0.56
	Mean	76	150	0.46
Atlantic Beach nearshore Long Island mud patches (10 samples)	Min	7	15	0.18
	Max	98	158	0.64
	Mean	34	65	0.47
Lido Beach nearshore Long Island mud patches (9 samples)	Min	16	19	0.28
	Max	195	250	0.79
	Mean	57	88	0.52
Hempstead Bay Long Island (2 samples)	Min	15	24	0.34
	Max	70	208	0.60
	Mean	43	116	0.47
October 1974				
Atlantic Beach nearshore Long Island mud patches (8 samples)	Min	12	23	0.34
	Max	27	49	0.92
	Mean	20	36	0.56
Lido Beach nearshore Long Island mud patches (2 samples)	Min	11	37	0.30
	Max	144	161	0.89
	Mean	78	90	0.60
Hempstead Bay Long Island (1 sample)		169	172	0.98
December 1974				
North Christiaensen Basin (8 samples)	Min	16	39	0.41
	Max	54	110	0.66
	Mean	37	65	0.57
Atlantic Beach nearshore Long Island mud patches (2 samples)	Min	17	57	0.29
	Max	46	79	0.58
	Mean	32	68	0.44
April 1975				
North Christiaensen Basin (1 sample)		88	129	0.68
Atlantic Beach nearshore Long Island mud patches (5 samples)	Min	23	40	0.06
	Max	28	374	0.58
	Mean	25	118	0.40
Atlantic Beach strandline fecal pellet deposits (2 samples)	Min	99	233	0.43
	Max	111	233	0.48
	Mean	105	233	0.45

2.8× in Atlantic Beach mud patches and Hempstead Bay muds (Tables 1 and 2). In contrast, waste sediments in the sewage sludge disposal area during the same time period are depleted in Cr by 0.4–0.7×, relative to the Hudson Shelf Valley normal muds (Tables 1 and 2).

Shelf sands between the Long Beach barrier island and the northern periphery of the Christiaensen Basin mud facies had mean Zn concentrations 6× greater (fine sands) and 4× greater (medium sands and coarse sands) than the mean zinc concentration in comparable grain-size normal shelf sands south of Fire Island. Similarly, mean chromium enrichment in shelf sands north of the sewage sludge disposal area and Christiaensen Basin muds is 4× greater (fine sands), 2× greater (medium sands), and about equivalent (coarse sands) relative to Fire Island shelf normal sands.

Distribution of high Zn and Cr concentrations indicates a much greater distribution of solids of presumed sewage origin to the north of the sewage sludge disposal permit area and the Christiaensen Basin than the distribution of muddy sediments suggests (Harris 1974), as shown in part for

Zn in Fig. 2. The Christiaensen Basin mud facies correlates with minimum isopleths of 150 mg/kg Zn in bottom sediments on the southeast side, 100 on the southwest, west-central, and north-central sides, 50 on the northwest, and 40 on the northeast sides of the basin. Muds of the sewage sludge disposal area correlate with the 30 mg/kg Zn minimum isopleth, and dredge spoils of the "mud" disposal area with the 150 mg/kg Zn isopleth. Nearshore Long Island mud patches correlate in the fall and winter with minimum isopleths of 20–30 mg/kg Zn off Atlantic Beach and 70–80 off Lido Beach (Fig. 2). Spring–summer minimum defining concentrations are higher: 30–50 mg/kg off Atlantic Beach and 70–90 off Lido Beach (Fig. 2). Hempstead Bay muds correlate with a minimum zinc isopleth of 100–150 mg/kg. Accordingly, muds, sandy muds, and muddy silts were found to cover 18 km^2 in the dredge spoil disposal area, 11 km^2 in the sewage sludge disposal permit area, 93 km^2 in the Christiaensen Basin, 1 km^2 in nearshore mud patches off Atlantic Beach, and 2.5 km^2 in mud patches off Lido Beach, or about 126 km^2 (56 nmi^2) of the northwest New York Bight apex during spring–summer 1974—the period of maximum mud dispersal observed. During the same time interval, shelf sands in this area containing at a minimum 5 times the background Zn concentration and 3 times the background Cr concentration of Fire Island normal shelf sands covered an additional 53 km^2 (24 nmi^2).

Spatial and temporal variation in chromium and zinc concentrations and the Cr:Zn ratio—There is considerable spatial and temporal variation in Cr and Zn concentrations and Cr:Zn ratios within muds of the northwest bight apex and Hempstead Bay (Table 2, Figs. 3–6). Concentrations of Cr and Zn are higher in Christiaensen Basin muds than in muds of the sewage sludge disposal area (Fig. 3). This is especially apparent in fall and spring, but less so in summer (Table 2). Zn and Cr concentrations were higher in the northern Christiaensen Basin than in muds near Long Island during fall and winter, but were higher in nearshore mud patches relative to the northern Christiaensen Basin in spring and summer. Zn and Cr both decreased shoreward during fall and winter, separately in both shelf sands and in muddy sediments (Harris 1974) (Figs. 1 and 3–6). The two metals attain maximum concentrations in nearshore mud patches in either late spring or summer, whereas summer minimum concentrations characterize the northern periphery of the Christiaensen Basin and Hempstead Bay muds (Table 2, Figs. 3–6). Mud deposits in the sewage sludge disposal area attain maximum concentrations of both metals in late spring (Fig. 3).

In muds of the sewage sludge disposal area, annual variation in Cr:Zn is bimodal with high mean values during late winter and summer (0.46) and low mean values in early spring and early fall–early winter (0.39) (Fig. 3). In northern Christiaensen Basin muds, the annual variation in Cr:Zn is unimodal with high mean values in summer (0.75) and low mean values in winter (0.49) (Fig. 3). A similar unimodal pattern is apparent for some nearshore mud patches off Atlantic Beach (Figs. 1 and 2), with Cr:Zn values highest in late spring (0.85) for mud patches receiving soft sediments since summer 1973 (Table 3, Fig. 5), and in summer (0.66) or early fall (0.69) for those which last began to develop during winter 1973–1974 (Table 3, Fig. 4). Mud patches which last began to develop in early spring 1974 off Atlantic Beach show a bimodal pattern with Cr:Zn highest in late spring (0.59) and early fall (0.51), with summer minor lows (0.37) (Table 3, Fig. 4). Similarly, a bimodal pattern in Cr:Zn characterizes the mud patch off Atlantic Beach which last began to receive soft sediments in summer 1972, with Cr:Zn highest in early summer (0.59) and early fall (0.95) and lowest in late summer (0.18) (Table 3, Fig. 4). With one exception, all Atlantic Beach nearshore mud patches have their lowest values for Cr:Zn in winter (avg 0.29) irrespective of when they last began to receive soft sediment (Table 3, Figs. 1, 4, and 5).

In contrast, Cr:Zn in nearshore Long Island mud patches which last began to develop during early spring 1974 off Lido Beach (Figs. 1 and 2) is highest in late

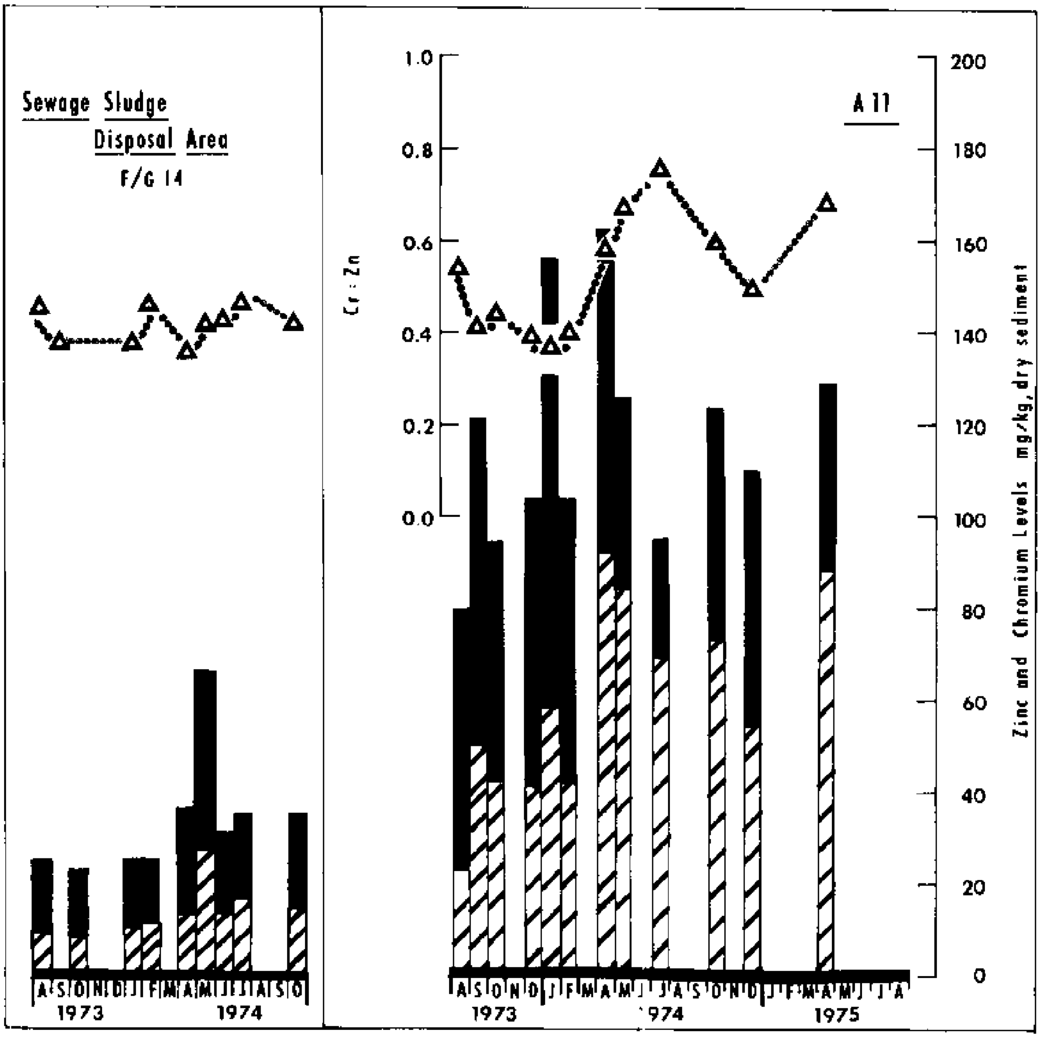


Fig. 3. Temporal variation in the concentrations of Cr and Zn and in the Cr:Zn ratio within organic muds from two potential sources of nearshore Long Island mud patches: F/G 14—sewage sludge disposal permit area; A11—northern Christiaensen Basin.

spring (0.89) and lowest in early fall (0.30) (Table 3, Fig. 4), whereas those which had been receiving soft sediment since summer 1973 have Cr:Zn values that are highest in early fall (0.89) and lowest in winter to early spring (0.63–0.56) (Table 3, Fig. 6). The latter pattern is similar to the annual variation for Cr:Zn in Hempstead Bay muds where Cr:Zn is highest in early fall (0.98) and lowest in early spring (0.61) (Table 3, Fig. 6). The former pattern is associated with Lido Beach nearshore mud patches that were being diluted with sand by early fall (Fig 4). For purposes of comparison, Cr:Zn in Bay Park deposits from Hempstead Bay was 1.31 in early spring 1973 (Fig. 6). Station A9 (Fig. 6) represents the Christiaensen Basin mud facies migrating north and east across former shelf sands (Fig. 2). Once the mud facies developed at A9, Cr:Zn increased to a late spring maximum (0.76) followed by a decrease to a summer minimum (0.42) and secondary late summer maximum (0.50) which was

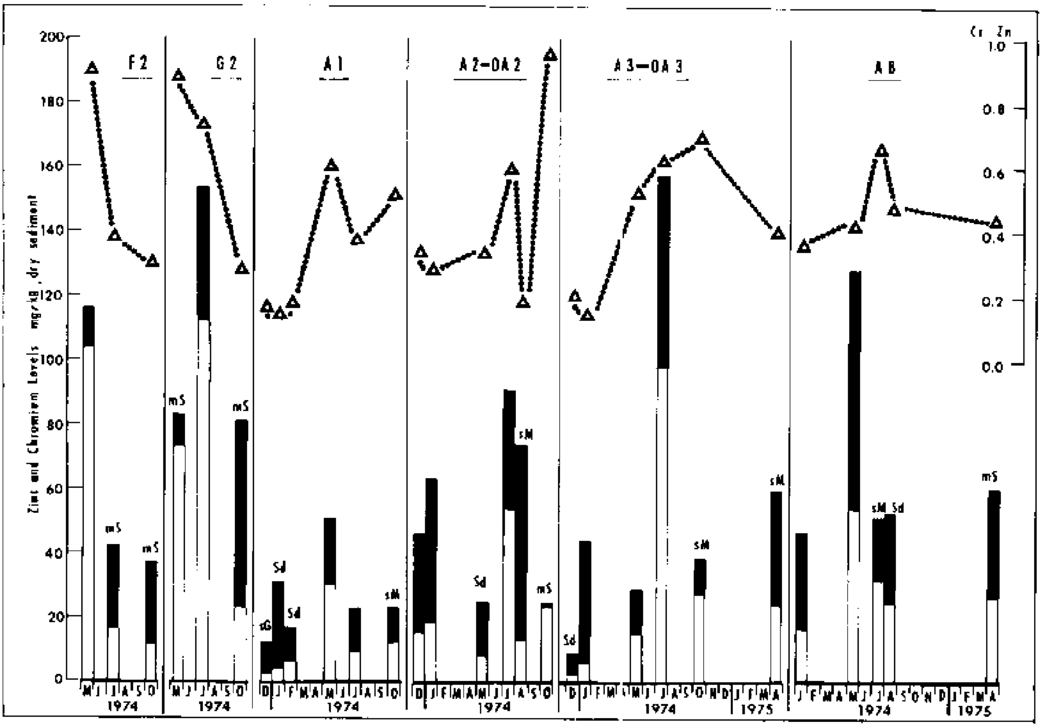


Fig. 4. Temporal variation in the concentrations of Cr and Zn and in the Cr:Zn ratio within nearshore Long Island mud patches: A1, A2-0A2, A3-0A3, A8—south of Atlantic Beach; F2, G2—south of Lido Beach.

maintained (0.51) even after the substrate had returned to sand by winter 1974 (Fig. 6).

Discussion

Distributions of organic-rich muds in the Long Island nearshore area are highly variable and mud accumulation may be cyclic. Nearshore mud patches in fall 1973, winter 1973–1974, or early spring 1974 were initially confined to sand-wave troughs or other topographic depressions. The patches increased in size through late spring, attaining maximum development in summer 1974 by accreting over sand-wave crests and thereby joining formerly isolated deposits. From early fall 1974 through early spring 1975, nearshore mud patches either decreased in size or disappeared, approximating the extent of the previous winter (Figs. 1 and 2).

Distributions of organic-rich muds along the northern edge of the Christiaensen Basin are also variable. The northwest boundary of the mud-sandy mud facies in the Christiaensen Basin appears to move relative to the Long Island shoreline. The peripheral zone of mud patch occurrence expanded shoreward from 6.9 km offshore in winter 1973–1974 to as close to Long Island as 5.0 km by summer 1974 and subsequently contracted seaward to 6.7 km and 9.3 km offshore by winter 1974–1975 and spring 1975. In contrast, only minor oscillations were observed for the north-central and northeast periphery which essentially maintained position at 10.2 km south of Long Island (Figs. 1 and 2).

Shoreward expansion and seaward contraction of the mud facies of the northwest Christiaensen Basin occurred during the same months as did nearshore Long Island mud patch growth and size reduction or disappearance. Presumably, observed changes in the mud-sandy mud boundary along the northwest periphery of the Christiaensen

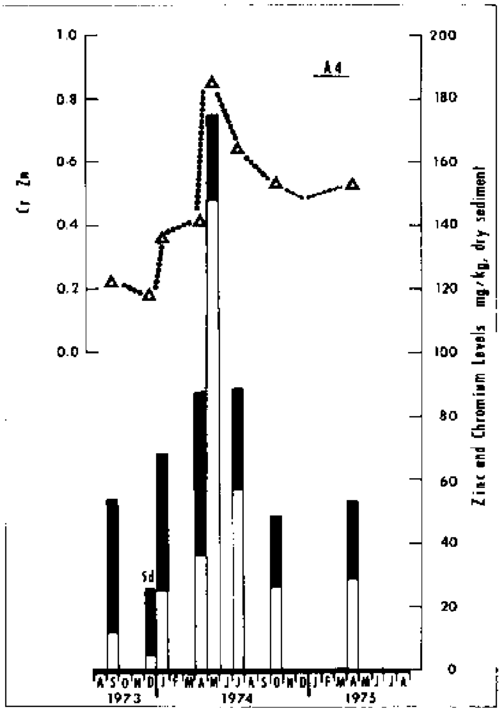


Fig. 5. Temporal variation in the concentrations of Cr and Zn and in the Cr:Zn ratio at A4 within a persistent nearshore Long Island mud patch 2.04 km (1.1 nmi) south of Atlantic Beach.

Basin resulted from the same processes affecting nearshore Long Island mud patches.

On disposal during the stratified conditions of summer, sludge solids retained in the surface layer are dispersed laterally. During nonstratified conditions of winter, sludge solids are rapidly dispersed throughout the water column (Callaway et al. 1976).

Currents associated with storms can resuspend large quantities of fine sediment in the bight apex (Drake 1974). Once resuspended, these sediments can be transported by currents. After discharge of sewage sludge, sludge solids can be transported by currents. Seabed and surface drifters and direct current measurements indicate that conditions are optimum during the summer for fine suspended materials to be transported north from the sewage sludge disposal area or Christiaensen Basin toward Long Island or from the sewage sludge disposal area into the Christiaensen Basin in the surface layer (Bumpus 1974; Charnell and Hansen 1974), intermediate depth, and near-bottom boundary layers (Charnell and Mayer 1976; Patchen et al. 1975). Summer conditions also favor deposition of these materials as a result of reduced wave action and storm flow which may explain the observed northward expansion of the northwest peripheral Christiaensen Basin mud facies and growth of nearshore Long Island mud patches. Conditions are optimum during winter–spring for fine sediments resuspended as a result of storms to be transported toward Long Island in the near-bottom turbid layer (Drake 1974). During periods of peak flow, mud (Drake 1974) and sand (Swift et al. 1976) are re-entrained in the water. Fine particles, unlike sands, can travel for days—perhaps weeks—after a storm, borne by slow fair-weather drift because of their smaller size and lesser density. Increased wave action, with the advent of winter–spring "northeasters," may resuspend offshore muds and transport them toward Long Island. There they could settle out forming nearshore mud patches during the quieter summer months. Wave action could also rework or flush out these mud patches and degrade nearshore sand waves during winter peak-storm flow conditions. The observed reduction or disappearance of nearshore mud patches from fall 1974 into early spring 1975 may also be related to microbial degradation of organic matter or sand-wave degradation with development of an incipent sand cover over formerly exposed mud patches. Several mud patches persisted into spring 1975 with no apparent sand cover, although as in fall 1973 and winter 1973–1974, nearshore muds were once again restricted to sand-wave trough axes.

It is not known whether the observed pattern of nearshore mud patch accumulation-diminution and of Christiaensen Basin mud facies expansion-contraction is repetitive from year to year. The spring 1974 pattern of northwest Christiaensen Basin mud-facies northward expansion and of nearshore Long Island mud patch development or growth had not developed by spring 1975. Thus, if the suggested pattern is repet-

Table 3. Net 1974 annual change in the concentration of Cr and Zn and change in the concentrations of these two heavy metals as it relates to growth and decline of the Cr:Zn ratio in nearshore Long Island mud patches and in potential source areas (TH89, A11, F/G 14). Figures in parentheses, where applicable, extend this history backward to September 1973 or forward to April 1975. (ΔCr and ΔZn expressed in mg/kg.)

Soft sed. last began accumulating	Cr:Zn		Date Cr:Zn		Mo to Cr:Zn				ΔCr	ΔZn	1974 annual	
	Max	Min	Max	Min	Max	Min	ΔCr	ΔZn	mo	mo	ΔCr	ΔZn
A1	0.60		5/74									
Late spring '74		0.37		7/74		2	−21	−28	−11	−14		
	0.51		10/74		3		3	0	1	0	−18	−28
A2-0A2		0.28		1/74								
Summer '72	0.59		7/74		6		36	28	6	5		
		0.18		8/74		1	−69	−17	−69	−17		
	0.95		10/74		2		10	−49	5	−25	−23	−38
		(0.88)		(4/75)		(6)	(−2)	(−1)	(0.3)	(−0.2)		
A3-0A3		0.13		1/74								
Winter '73–'74					6		92	114	15	19		
	0.69		10/74		3		−71	−119	−24	−40	21	−5
		(0.40)		(4/75)		(6)	(−3)	(20)	(−0.5)	(3)		
A4		(0.22)		(9/73)								
Summer '73		0.36		1/74	(4)		(13)	(15)	(3)	(4)		
	0.85		5/74		4		123	106	31	27		
		0.53		10/74		5	−122	−126	−24	−25	1	−20
		(0.53)		(4/75)		(6)	(3)	(4)	(0.5)	(0.7)		
A6	0.55		5/74									
Late spring '74		0.31		7/74		2	14	7	7	4	14	7
A8		0.36		1/74								
Winter '73–'74					4		34	82	9	21		
	0.66		7/74		2		−20	−77	−10	−39		
		0.47		8/74		1	−7	1	−7	1	−8	−17
		(0.44)		(4/75)		(8)	(2)	(8)	(0.3)	(1)		
A9		0.17		1/74								
Winter '73–'74					1		8	−12	8	−12		
					2		−2	10	−1	5		
	0.76		5/74		1		28	−3	28	−3		
		0.42		6/74		1	−24	−8	−24	−8		
					1		−10	−25	−10	−25		
	0.51		8/74		1		13	21	13	21	13	−17
F2	0.90		5/74									
Late spring '74						2	−88	−74	−44	−37		
		0.30		10/74		3	−5	−5	−2	−2	−93	−79
G2	0.88		5/74									
Late spring '74						2	39	71	20	36		
		0.28		10/74		3	−89	−73	−30	−24	−50	−2
G3		0.63		1/74								
Summer '73					6		95	90	16	15		
	0.89		10/74		3		−51	−89	−17	−30	44	1

Table 3. Continued

Soft sed. last began accumulating	Cr:Zn		Date Cr:Zn		Mo to Cr:Zn		ΔCr	ΔZn	ΔCr / mo	ΔZn / mo	1974 annual	
	Max	Min	Max	Min	Max	Min					ΔCr	ΔZn
H3												
Summer '74	0.51		7/74									
I1												
Summer '74	0.37		7/74									
TH89 (**Bay Park, Nassau Co. Sewage Treatment Plant outfall area, Reynolds Channel, Hempstead Bay**)												
		0.61		4/74								
					3		−25	−72	−8	−24		
	0.98		10/74		3		123	124	41	41	98	52
A11 (**Northern Christiaensen Basin**)												
		0.37		1/74								
					1		−16	−52	−16	−52		
					2		44	30	22	15		
	0.75		7/74		3		−17	−39	−6	−13		
						3	4	25	1	8		
		0.49		12/74		2	−19	−11	−10	−6	−4	−47
	(0.68)		(4/75)		(4)		(34)	(19)	(9)	(5)		
F/G 14 (Sewage sludge disposal permit area)												
		0.39		1/74								
	0.46		2/74		1		1	0	1	0		
		0.38		4/74		2	2	11	1	6		
					1		14	30	14	30		
					1		−14	−35	−14	−35		
	0.46		7/74		1		2	4	2	4		
		0.42		10/74		3	−2	0	−1	0	3	10

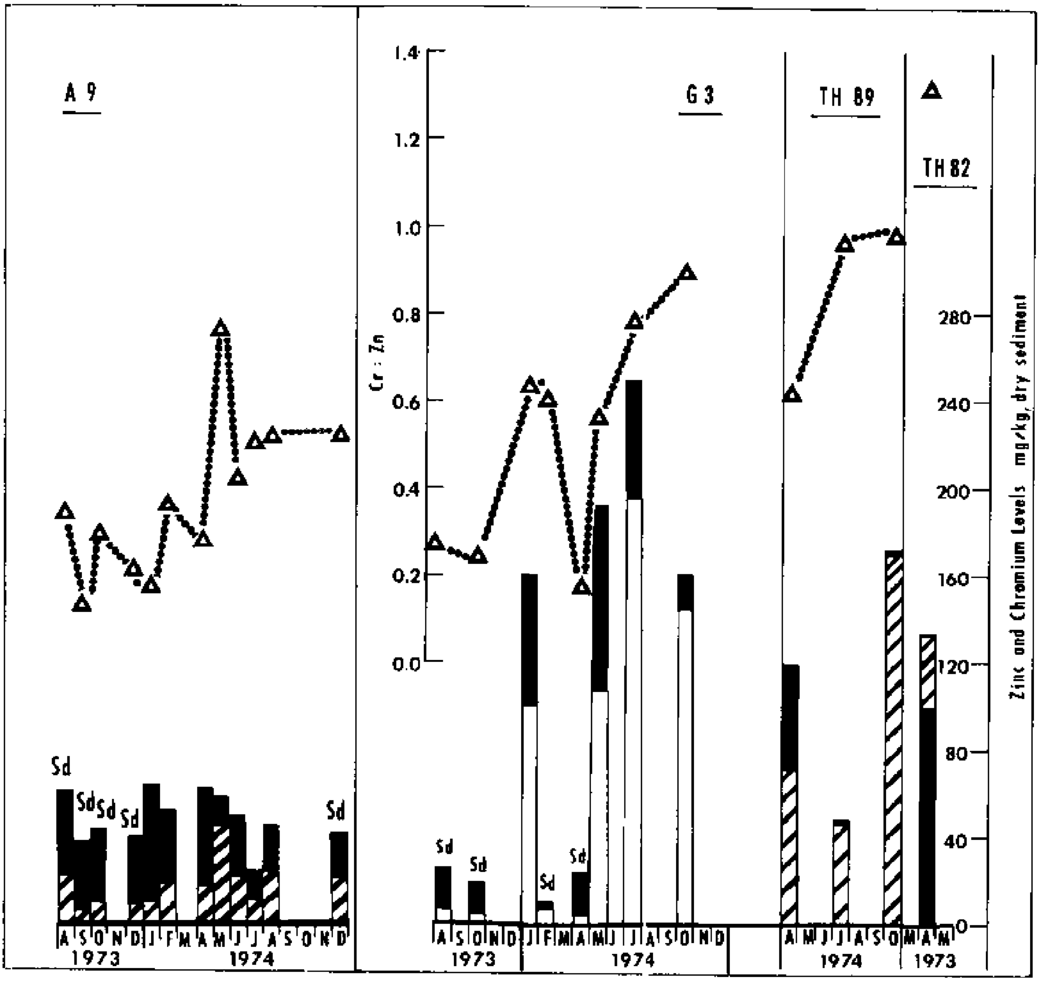


Fig. 6. Temporal variation in the concentrations of Cr and Zn and in the Cr:Zn ratio at G3 within a relatively persistent nearshore Long Island mud patch 3.01 km (1.63 nmi) south of Lido Beach and within organic muds from two potential sources of nearshore Long Island mud patches: A9—northern Christiaensen Basin; TH89, TH82—Hempstead Bay.

itive, it does not occur at exactly the same month in successive years or may only occur during those years when the bottom layer is thickest, as it was during summer 1974, compared to summer 1973 (Hazelworth et al. 1974, 1975*a*, *b*).

A thick bottom layer would inhibit vertical exchange between the upper layers and the bottom layer, thereby allowing a vertically uniform distribution for turbidity in the bottom waters much greater than the maximum turbidity developed in the near-bottom turbid layer when the bottom layers are thinner (Hazelworth et al. 1974, 1975*a*, *b*). This would allow a greater volume of resuspended fine particulate material and sewage sludge solids to reach the nearshore Long Island zone of mud patch development where under the quiet conditions of summer they could settle out to form mud patches in depressions characterized by reduced wave and current action.

Individual Long Island nearshore mud patches have not existed for more than a few years. Age classes of *N. proxima* show that most mud patches within 2.8 km of Long Island last began to develop during winter 1973–1974 or spring 1974. Some last began to develop in summer 1973 or summer 1974. One off Atlantic Beach has been present since summer 1972. Mud patches may be either completely or partially covered by sand, especially during winter months, so age classes of *Nucula* in the mud patches record the elapsed time since mud last began to accumulate at a particular site and the approximate time before mud deposition when the site last was covered by sand.

Chromium and zinc concentrations in nearshore Long Island mud patches suggest that Christiaensen Basin and Hempstead Bay may be sources of organic matter. Zinc is highly concentrated in sewage sludge of the New York metropolitan area (avg annual concn, 175 mg/kg in 1973 and 115 in 1974: U.S. EPA 1975). Zinc is the predominant heavy metal in 74% (1973) to 78% (1974) of the sewage sludge dumped into the bight apex. In contrast, chromium is the principal heavy metal, or equals zinc concentrations, in only 9% (1973) to 13% (1974) of the sewage sludge from New York and New Jersey municipalities (U.S. EPA 1975). Chromium and Zn are of nearly equal concentration (1973: 23 vs. 25 mg/kg) in Nassau County (Bay Park Sewage Treatment Plant) sewage sludge.

In the sewage sludge disposal area the average temporal maximum concentration of Zn is 34 mg/kg, about 2.5× that of Cr. Zn concentrations are higher (100–400 mg/kg) in the Christiaensen Basin, where Zn is enriched relative to Cr by 2–5×. The results reported here are supported by others (e.g. Carmody et al. 1973; Gross 1970, 1972; Gross et al. 1971; Greig et al. 1974; NMFS 1972; U.S. EPA 1974). The average temporal mean concentration of Cr in Hempstead Bay muds (99 mg/kg) is about 4× that in Bay Park (Nassau Co.) sewage sludge and 2.5× that in the 1973–1974 average (39 mg/kg) for Cr in New York metropolitan area sewage sludge dumped in the bight (U.S. EPA 1975).

Zinc and chromium decrease shoreward during fall and winter from the Christiaensen Basin, separately in both shelf sands and in muddy sediments (Harris 1974). Maximum concentrations of the two metals occur in nearshore mud patches in either late spring or summer, but continue to decrease shoreward across inner shelf sands from the mud-sandy mud facies of the Christiaensen Basin. Thus, during all seasons within inner shelf sands and during fall and winter within shelf sands or muddy sediments, chromium and zinc are sensitive indicators for bottom dispersal of sludge particulates northward toward Long Island from the Christiaensen Basin. Similar use was made of zinc and silver as tracers for dispersal of Philadelphia metropolitan area sludge off the mouth of Delaware Bay (Rutherford and Church 1975).

Sewage sludge solids could have moved to within 0.5 km of Long Island (Harris 1974). This is most apparent for mud patches off Atlantic Beach. Mud patches off Lido Beach have Cr concentrations that are much higher or equivalent to Zn concentrations, approaching values common to Hempstead Bay muds. Thus, many Lido Beach mud patches could be derived from Hempstead Bay.

Buildup in Zn and Cr concentrations in late spring and summer in nearshore Long Island mud patches relative to the Christiaensen Basin and muds of the sewage sludge disposal area cannot be explained using metal concentrations alone, rather it requires use of metal ratios as discussed below.

During spring and summer 1974, Zn was released to waters of the bight apex and perhaps to organisms, whereas organic-rich sediments served as a semipermanent sink for Cr. From fall 1974 through early spring 1975, Zn and Cr were released to bight apex waters from Christiaensen Basin muds and nearshore Long Island mud patches at nearly equal rates although Zn was retained in some muds preferentially relative to Cr.

Muds sampled in the apex had a maximum Cr:Zn ratio between late spring and early fall. The data (Table 3) reveal that the late spring-early fall Cr:Zn maximum is attained via several paths, involving: buildup of Cr at a rate 1.2× that of Zn; or combinations involving buildup of Cr at rates equivalent to that of Zn up to 1.5× the rate of Zn buildup coupled with a decline in Zn at about 2–3× the rate of Cr decline; decline in Zn at 2–2.5× up to 4× the rate of Cr decline coupled with a buildup in Zn at rates equivalent to that of Cr up to 2× the rate of Cr buildup; and buildup in Cr with a concurrent decrease in Zn coupled with a buildup in Zn and concurrent decline in Cr. Thus, during attainment of the late spring-early fall Cr:Zn maximum, muds of the northwest apex either gained chromium (2–34 mg/kg) concurrent with losing zinc (1–61 mg/kg) or gained Cr (14–123 mg/kg) relative to zinc (5–106 mg/kg) as did Hempstead Bay muds (Table 3).

For muds showing two Cr:Zn maxima, net Cr gains and Zn losses were also similar. The winter maximum in Cr:Zn at the sludge disposal area resulted from retention of earlier Zn concentrations and buildup in Cr of 1 mg/kg (Table 3). In the expansion of Christiaensen Basin mud facies toward Long Island (A9) in winter 1973–1974, a secondary maximum in Cr:Zn was attained in summer 1974 by a combination of decline in Zn at a rate 2.5× that of Cr decline coupled with a buildup in Zn at a rate about 1.5× that of Cr buildup for a net gain in Cr of 3 mg/kg and a net loss in Zn of 4 mg/kg (Table 3). At A2 the second maximum of the year in Cr:Zn was the major one and was a result of the loss of 49 mg/kg Zn concurrent with a Cr gain of 10 mg/kg (Table 3). Station A1 muds, which started out their latest mud patch history at the late spring maximum in Cr:Zn, exhibit a secondary maximum in early fall which was achieved by retention of summer Zn concentrations concurrent with a buildup of 3 mg/kg in Cr (Table 3).

Decline from the late spring–summer maximum in Cr:Zn also occurs via several paths (Table 3) involving: decline in Cr and Zn at nearly equal rates; decline in Cr at 3–4× the rate of Zn decline; decline in Cr with concurrent retention of summer Zn concentrations; and combination of a buildup in Zn at from 2–8× the rate of Cr buildup coupled with a decline in Cr at a rate nearly equal to that of Zn up to 3× the rate of zinc decline.

Thus, during attainment of the summer, early fall, or winter Cr:Zn minimum, muds of the northwest apex lost more Cr (24–93 mg/kg) than Zn (2–79 mg/kg), lost Cr (2 mg/kg) with no change in Zn concentration, lost Cr (7–15 mg/kg) concurrent with a gain in Zn (1–14 mg/kg), or lost more Zn (28–126 mg/kg) than Cr (18–122 mg/kg) (Table 5).

The spring 1974 minimum in the sewage sludge disposal permit area resulted from a simple buildup in Zn at 6× the rate of Cr buildup, corresponding to a gain of 2 mg/kg in Cr as compared to a gain of 11 in Zn (Table 3). Apparent minima in Cr:Zn, achieved in spring 1975 result from a decline in Cr at 1.5× the rate of Zn decline or from a decline in Cr (3 mg/kg) and concurrent buildup in Zn (20 mg/kg) (Table 3). Where the trend toward the Cr:Zn minima extends from 1974 into 1975, spring 1975 minimal values for Cr:Zn are attained through buildup in Zn at rates 1.4–3.3× that of Cr buildup (Table 3). Lastly, the beginning of growth to a possible summer Cr:Zn maximum at A11 appears to have already started by spring 1975 and was attained through a buildup in Cr at nearly 2× the rate of Zn buildup (Table 3).

Segar and Cantillo (1976) found that dissolved Zn had elevated concentrations at or near the sewage sludge dump site. They concluded that Zn in particular appeared to be released to solution from sewage sludge or dredge spoil. Release of metals may occur during the disposal operation, while the sewage sludge solids are suspended in the water column, after accumulation on the bottom in the disposal area, Christiaensen Basin, or nearshore Long Island zone of mud patch development, or during resuspension and subsequent transport and deposition.

Apparently considerable amounts of heavy metals are released to the water during the initial disposal of sewage sludge. The increased salinity resulting from mixing with seawater favors desorption of heavy metals from sludge solids (Bachman 1963; O'Connor 1968; Burrows and Hulbert 1975). Organic matter in sewage sludge could provide cation exchange sites (Rashid 1969). Rohatgi and Chen (1975) found that 10–20% more Zn than Cr is released from suspended sewage sludge solids on mixing with seawater. This first stage release of heavy metals to solution may also be attributable to the oxidation of organic particles or solution of metal sulfides (Morel 1975). Rohatgi and Chen reported that only 2% of the Cr as compared to 18–39% of the Zn was released to seawater from suspended particulates in digested sludge.

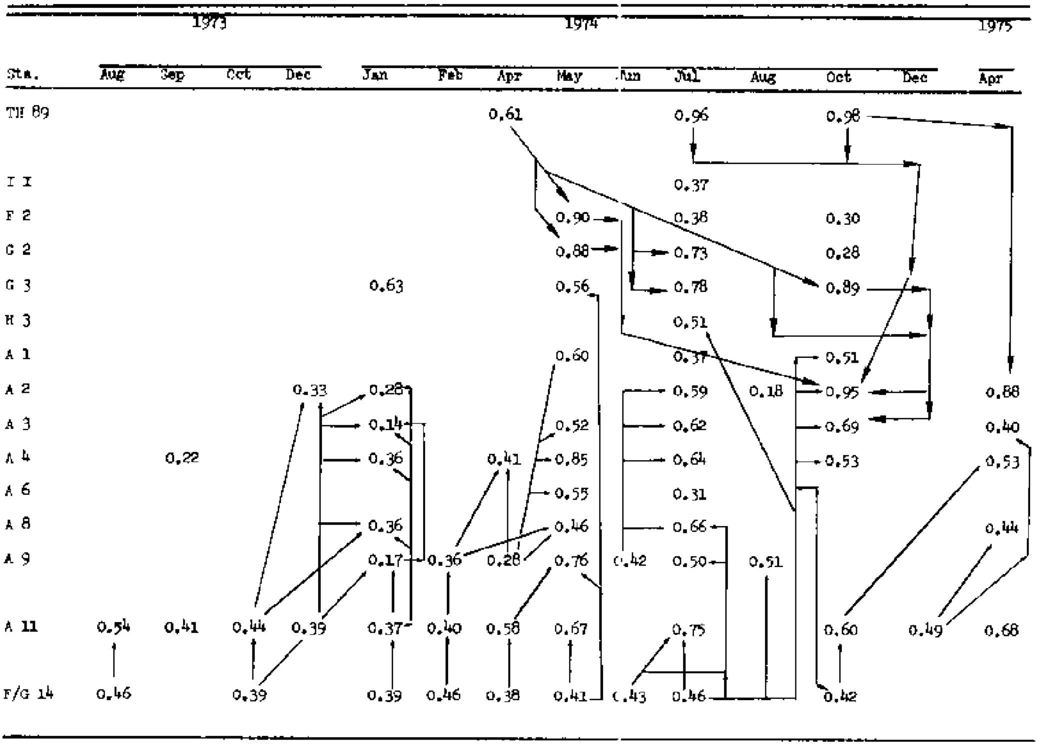
Release of metals to seawater during successive stages might continue to occur as a result of further oxidation of organic matter or of further desorption. All of these processes favor preferential loss of Zn relative to Cr as indicated by the increase in the Cr:Zn ratio of bottom muds toward the observed spring, summer, or early fall maximum values for this ratio. Conversely, metals which are in solution or which come into solution during dumping can be immobilized through sorption to suspended particulate matter or to bottom muds. That this process may also be responsible for the observed increase in Cr:Zn to maximum values (as suggested in Table 3) is supported by the data of Mellor and Maley (1948) who studied the order of stability of natural metal complexes in seawater. Ringbom (1963) and Bradford (1972) found naturally occurring organic chelates to be less selective for Zn than for Cr. Thus, when organic chelates of Cr and Zn are adsorbed from solution by suspended sludge solids, natural fine particulates, or bottom muds, Zn will be sequestered less than Cr (Bradford 1972). Since the Cr:Zn ratio begins to increase in muds at about the time of the initial spring phytoplankton bloom, it may also be related to metabolic uptake of organically chelated Zn by phytoplankton (Provasoli 1963; Davies 1970; Hellebust 1970) or marine bacteria (Budd and Spencer 1968).

The observed summer minimum in Cr:Zn found for muds having a bimodal maxima for Cr:Zn in late spring and late summer–early fall, as well as decline to winter Cr:Zn minimal values for both bimodal and unimodal Cr:Zn maxima muds, may also be related to biological activity. Chromium does not enter into the metabolic activity of marine phytoplankton and bacteria. On the death of the phytoplankton, following the spring or fall bloom, Zn might be concentrated in the bottom muds through accumulation of dead phytoplankton cells or under conditions of bacterial growth. Low dissolved oxygen concentrations characterize much of the Christiaensen Basin (Hazelworth et al. 1974, 1975*b*) and nearshore Long Island (Long Beach) inner shelf (U.S. EPA 1974, 1975) bottom waters in summer. The decrease in redox potential associated with reduced concentration of dissolved oxygen can cause Mn and Fe to change valence and dissolve from the hydrous oxide phase into the overlying water with release of adsorbed Zn and Cr and other heavy metals (Mortimer 1970). Such release would occur at nearly equal rates and could explain the summer minimum observed for Cr:Zn in some mud patches that had attained maximum values of Cr:Zn in spring. Decline toward the winter annual Cr:Zn minimum may be related to an increase in redox potential with increased dissolved oxygen available in these areas beginning in the fall which might be a factor in the oxidative decomposition of the organosulfur compounds of Zn and Cr and other heavy metals at nearly equal rates and the subsequent dissolution of these heavy metals in the aqueous phase. Enhanced resuspension of bottom muds and reworking and partial flushing out of nearshore mud patches in winter and spring would favor the loss of Zn over that of Cr by desorption through cation exchange or continued oxidation of organic matter to restart the cycle.

The observed growth and decline in the Cr:Zn ratio might also be related to change in the Zn and Cr concentration of the nonsettleable fraction of sludge solids disposed of at the sewage sludge disposal area during the study period. However, to achieve the magnitude of the spring–summer–early fall maximum observed for Cr:Zn in bottom muds of the bight apex, it would then be necessary for the Cr:Zn ratio in nonsettleable sludge solids during spring and summer 1974 to be 2–3× the 1974 annual average (U.S. EPA 1975). Similarly, to achieve the magnitude of the winter minimum observed for Cr:Zn in bottom muds, it would be necessary for Cr:Zn in nonsettleable sludge solids during fall and winter 1974 to be nearly equivalent to the 1974 annual average of 0.30 (U.S. EPA 1975). For some nearshore mud patches, the late spring–summer–early fall Cr:Zn maximum could result by contributions from the Cr-rich muds of Hempstead Bay. It is doubtful that this transport process would affect the observed increase in Cr:Zn of Christiaensen Basin muds. Similarly, the shoreward transport of resuspended Christiaensen Basin muds during winter to settle out in the restricted mud patches of nearshore Long Island could account for the minimum values of Cr:Zn characteristic of the nearshore mud patches at this time.

Cr:Zn ratios in muds may be useful tracers for the movement of organic mud particulates from potential sources to accumulation sites of mud patches on nearshore Long Island. A model in which initially suspended (at the disposal area) or resuspended sludge or sludge muds (Christiaensen Basin or Hempstead Bay) are subjected during transport to the chemical interactions with seawater described above suggests that the Cr:Zn ratio in organic-rich mud particulate material settling to the bottom increases between late winter and early fall and decreases between late fall and early winter. Examination of Table 4 with the proposed model suggests that sewage sludge solids from the disposal area may be transported both west into the northern

Table 4. Flow chart showing possible paths and times of transport of sewage sludge or organic mud suspended particulates from potential source areas into the mud patch zone of nearshore Long Island, as based on seasonal change in Cr:Zn in bottom sediments. Small arrows represent paths of transport from the sewage sludge disposal permit area (F/G 14) or northern Christiaensen Basin (A11); large arrows represent paths of transport from Hempstead Bay into the mud patch zone of nearshore Long Island and from Lido Beach to Atlantic Beach mud patches. Numbers tabulated for sampling stations listed (*see Fig. 1*) represent magnitude of Cr:Zn ratio in bottom sediments at the stations between August 1973 and April 1975.



	1973				1974									1975
Sta.	Aug	Sep	Oct	Dec	Jan	Feb	Apr	May	Jun	Jul	Aug	Oct	Dec	Apr
TH 89							0.61			0.96		0.98		
I I										0.37				
F 2								0.90		0.38		0.30		
G 2								0.88		0.73		0.28		
G 3					0.63			0.56		0.78		0.89		
H 3										0.51				
A 1								0.60		0.37		0.51		
A 2				0.33	0.28					0.59	0.18	0.95		0.88
A 3					0.14			0.52		0.62		0.69		0.40
A 4		0.22			0.36		0.41	0.85		0.64		0.53		0.53
A 6								0.55		0.31				
A 8					0.36			0.46		0.66				0.44
A 9					0.17	0.36	0.28	0.76	0.42	0.50	0.51			
A 11	0.54	0.41	0.44	0.39	0.37	0.40	0.58	0.67		0.75		0.60	0.49	0.68
F/G 14	0.46		0.39		0.39	0.46	0.38	0.41	0.43	0.46		0.42		

Christiaensen Basin, from the northern Christiaensen Basin shoreward, or directly shoreward from the sewage sludge disposal area to accumulate in mud patches off Atlantic Beach throughout the year. Travel times would range from less than 1 month to 6 months. Introduction of suspended sludge solids from the sewage sludge disposal area or northern Christiaensen Basin into Lido Beach mud patches could, with this model, only occur during late spring and summer. Suspended sludge mud solids derived from Hempstead Bay could, with the model, be introduced to Lido Beach mud patches during spring through early fall via Jones Inlet over travel times up to 6 months. With the model, Hempstead Bay may also introduce suspended sludge mud solids via either the East Rockaway or Jones Inlets to some Atlantic Beach mud patches (A2 or A3), with travel times of less than 1 month and up to 6 months. If not related to tidal transport via Hempstead Bay inlet systems directly, the anomalously high Cr:Zn ratios in these two Atlantic Beach mud patches could be explained by westward longshore transport of high Cr:Zn mud patch particulates from Lido Beach and their accumulation at the designated times (Table 4) after 1–6 months of travel during spring through early fall.

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Suspended particulate concentrations and compositions in the New York Bight[1]

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Abstract

Suspended particulate concentrations in the New York Bight during 1973–1975 decrease seaward across the shelf, decreasing more rapidly in surface than in bottom waters. Resuspension of fine-grained sediments causes local high concentrations of suspended particles in near-bottom waters. Horizontal displacement of these high concentration plumes from their sources suggests short residence times for suspended particles. Vertical mixing of resuspended particles is limited by the thermocline. Along the upper continental slope over a depth range of ~ 1,000 m, there is a minimum in near-bottom suspended particulate concentrations suggesting horizontal mixing with waters from the open ocean. Anomalous concentrations of trace metals of anthropogenic origin associated with organic particles, Fe-Ti (oxide) coatings and discrete Ti (oxide) particles are potential tracers of particle dispersion paths and transport processes. Different types of organic particles exhibit different interparticle and trace element associations and appear to have different geographic distributions. Some of this variability may be seasonal. Surface water suspended matter has a higher proportion of biogenic (inorganic skeletal as well as organic) particles than near-bottom suspended matter which is dominantly nonbiogenic (principally aluminosilicate). Skeletal debris is primarily siliceous in shelf waters, becoming more carbonate-rich seaward of the shelf break. Aluminosilicate suspended particles in shelf waters are predominately K-rich whereas Mg-Ca-K-Fe-rich aluminosilicates dominate beyond the shelf break.

Suspended particulate matter plays a major role in the cycles of many elements in the oceans and is responsible for most of the elemental concentration gradients observed. This is no less true in coastal waters which are the zone of interaction between both dissolved and solid runoff from the continents and the open ocean. In the New York Bight, suspended particles and chemicals of anthropogenic origin are added both to the runoff waters and directly to bight waters by solid and liquid waste dumping. These particles, of both natural and anthropogenic origins, constitute an important vehicle for the transport of pollutants to and removal from the waters of the continental shelf.

Knowledge of the concentrations and composition of suspended particles is basic to a quantitative understanding of their role in geochemical cycles and of the processes of their dispersal in the bight. Several publications present data on concentrations of suspended particles in waters of various locations on the east coast continental shelf although these consist of less dense sampling over a larger area of the shelf or equally dense sampling of a different shelf area (Manheim et al. 1970; Drake 1974; Meade et al. 1975 and references therein). Few publications, however, present data on the composition of suspended particles and most of those which do report some type of bulk analysis of the material (e.g. Manheim et al. 1970; Meade et al. 1975). Analyses of characteristics of the sample treated as a whole, or in bulk, have an advantage in case of quantifying the data; both X-ray fluorescence and X-ray diffraction analysis on our samples of suspended particles from the New York Bight are in progress. Particle characterizations reported here are based on morphology and chemical composition of discrete particles using combined scanning electron microscopy (SEM) and energy-dispersive X-ray fluorescence (EDXRF). Although it is more difficult to quantify results from this technique, the advantage is that one can directly observe interparticle associations and associations between particle types and trace elements. We present data on suspended particulate concentra-

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tions in the water column at two seasons and compare these to sediment characteristics. We also report preliminary results on particle types and compositions using SEM-EDXRF.

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Sampling and analytical methods

Samples for this study were taken primarily during two L-DGO cruises (Fig. 1). The first, in October 1974 on RV *Vema*, was designed to study standard hydrographic and geochemical parameters during the initial breakup of the seasonal thermocline. Gordon et al. (1976) reported that the seasonal thermocline was still well developed. Three sampling transects crossed the continental shelf and upper continental slope normal to the coastline; one along the Hudson Shelf Channel–Hudson Canyon and one each to the northeast and southwest. Three hundred water samples were filtered during this cruise. The second cruise (July–August 1975) on RV *Conrad* reoccupied many stations along the three *Vema* transects, extended them, and filled in with a number of stations between the transects. This cruise measured hydrographic and geochemical parameters during the buildup of the summer thermally stratified regime; 440 samples of suspended particles were taken. Each cruise lasted just over 2 weeks. In addition, samples were taken during short (1–3 day) cruises in the bight during 1973 and 1974.

Water samples for suspended particles were taken in standard PVC 30-liter Niskin bottles the internal springs of which were Teflon-coated to preclude contamination. A pinger was used to position the bottommost bottle close to, but not touching the bottom, and a double-ping mechanism signalled closure of the bottom bottle to yield height of that sample, and hence the rest of the bottles on the cast, above the sediment-water interface. On deck, water was filtered directly from the Niskin bottles through in-line filter holders into evacuated bottles. For samples at or beyond the shelf break where suspended particulate concentrations were expected to be low, the 20 liters of water filtered was the same as that on which shipboard analysis of radon was performed, the results of which are reported elsewhere (Biscaye and Mathieu in prep.). In shallower, more turbid water, a separate aliquot of between 1 and 10 liters of water was filtered.

Preweighed Nuclepore filters (47-mm diameter, 0.4-μm pore size) were used. Handling of filters in the laboratory and aboard ship was done in a laminar flow hood to minimize contamination. Washing to remove salt was done with filtered, double-distilled water in 12 aliquots of about 5 ml each. A blank filter was treated identically to the other filters at each station with the exception of exposure to seawater.

Analytical error for gravimetric analysis of suspended particulate concentrations is difficult to estimate because of the wide range of volumes of water passed through the filter and weight of suspended material

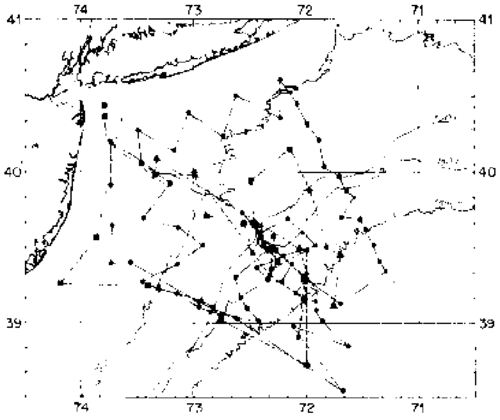

Fig. 1. Cruise tracks and ship stations for October 1974 cruise (*Vema* 32-01 ●-------●) and July 1975 cruise (*Conrad* 19-01 ●———●). Additional points represent samples taken on short cruises during 1973 and 1974. ■—stations where filtered samples of suspended particles were examined by SEM-EDXRF throughout the water column; ▲—stations where only near-bottom filters were examined.

left on the filter. From estimated errors in volume measurement, in weighing (using a Mettler model H 64 semimicro-balance) and from replicate analyses, we estimate our error in concentration (μg/liter) to be ±5% for most samples.

Sample preparations for analysis by SEM-EDXRF were also done in a laminar flow hood to preclude contamination by dust. Small portions were cut from dried and weighed filters and mounted on aluminum stubs with double-stick tape. Edges of the filter and the tape were bonded to the aluminum stub with high purity silver glue to ensure good electrical conductivity between the filter and the sample stub. Mounted filters were vapor-coated with palladium and carbon to supress charge buildup on the sample and to increase electron emission to obtain a better SEM image (Hearle et al. 1972). Palladium was chosen because its position in the X-ray spectrum interferes only minimally with other elements of interest and the thickness of this coating was minimized to enhance the emission of characteristic X-rays from the sample.

About 100 randomly chosen particles were examined on each filter. The composition of each particle was determined by EDXRF, a microanalytical technique in which the SEM electron beam is focused on an individual particle and the elemental composition (for elements of atomic number $Z \geq 9$) is determined from characteristic X-rays visually displayed as peaks on the energy spectrum in the X-ray analyzer.

Particles have been identified and classified, based on morphology and chemical composition, as shown in Table 1. The X-ray spectrum is generally the primary means of characterization since many particle types can exhibit similar morphologies (*see Figs. 6–14*). Biogenic opaline silica (identified by a pure Si peak) and biogenic calcium carbonate (identified by a pure Ca peak or Ca with some Mg) generally showed morphologic features suggestive of biogenic origin.

Although the most common elements in organic matter, i.e. carbon, oxygen, and nitrogen have atomic numbers (Z) <9 and thus cannot be detected by our EDXRF equipment, phosphorus ($Z = 15$) is an almost universal constituent of organic particles. Consequently, P was used as our primary indicator of organic particles along with, generally, some combination of S and/or Cl, or by the complete absence of a characteristic X-ray spectrum. Frequently organic particles exhibit smooth, gelatinous, or symmetric morphological features indicative of biologic origin. These "smooth" organic particles were generally characterized by strong P peaks, weak Si and Al peaks, and an absence of Fe, Mn, and other trace metals. Other organic particles have rough, aggregate-looking shapes with no symmetry and often contain strong Si, Al, Ca, and Fe peaks and sometimes K, Mg, Mn, and other trace metals. These particles appear to constitute a complete mixing series between pure organic and pure clay (aluminosilicate) end members. The distinction between "organic" and "clay" in this series is somewhat arbitrary and we have used relative intensities of the P, Si, and Al lines in making the classification.

Aluminosilicates were identified using Al:Si ratios and cation composition. Obvious cleavage sometimes aided in distinguishing between micas, feldspars, and heavy minerals such as amphiboles, pyroxenes, and sphene. Clay minerals generally could not be identified by morphology and were classified on the basis of EDXRF spectrum. Frequently clay minerals were observed as large aggregates (≈5–30 μm).

Quartz could generally be distinguished from opaline skeletal debris by its morphology (smooth shape, sharp edges, and more dense appearance) and by a much higher Si X-ray count rate. Dolomite, sphene, barite, Fe-oxides, and discrete Ti-oxide and fly ash particles were also observed.

Distributions of suspended particulate concentrations and compositions

Vertical profiles of up to eight samples cover the entire water column over much of the study area. Beyond the continental shelf break, sampling was restricted to the lower portion of the water column since excess radon, which was also part of the analytical program, is generally not mixed upward

Table 1. Classification of major particle types on New York Bight filters by SEM-EDXRF.

Particulate type	Characteristic X-ray lines	Example Fig. No.	Morphology
Biogenic			
Organic			
1. Smooth	None; spectrum obtained is pure background (C, N, O not detectable) or P with any combination of S and/or Cl.	7	Generally particles show some regular, smoothed or symmetric morphologic features indicative of biogenic origin. Otherwise a gelatinous appearance.
2. Rough	May show the same spectrum as smooth but may also contain relatively high concentrations of Al, Si, and important trace elements.	8	Generally rough aggregate particles sometimes associated with clay. Appear to be organic detritus.
Opaline silica	Si alone (*see quartz*)	6	Often shows regular, symmetrical features suggestive of biogenic origin, but not necessarily so, especially in small fragments. Diatoms most common.
Calcium carbonate	Ca alone or Ca with Mg		Often shows regular, symmetrical features suggestive of biogenic origin, but not necessarily so, especially in small fragments. Form most often recognized is coccolith. Some particles with definite subhedral to euhedral crystal habit observed; not known if these are biogenic.
Nonbiogenic			
Aluminosilicate	All particles of this type contain Al and Si (these not repeated below). Elements in parentheses are occasionally present.		
1.	K	9	Sometimes these have blocky or platy morphology and peak ratios permit specific identification of feldspars, micas, and heavy minerals.
2.	K-Mg-Fe(Ti)		
3.	Mg-Fe		
4.	Ca-Fe-(Mg)	11	
5.	Na-(Ca)		
6.	Pure Al-Si		Generally aggregate morphology—sometimes massive and platy. In the presence of P, S, or Cl these are difficult to classify as clay or rough organic and the decision usually lies with the relative intensities of the P, Al, and Si peaks but may be influenced by morphology.
7.	K-Fe		
8.	K-Ca-Fe (S, Cl, P)		
9.	Mg-Ca-K-Fe (S, Cl, P)	12	
10.	Other combinations of above		
Quartz	Si alone		Much more "dense" appearance than opaline silica and Si X-ray count rate is higher.
Other			
1. Dolomite	Mg=Ca	10	Euhedral shape.
2. Heavy minerals			
Barite	Ba=S(±Sr)		Subrounded but can be more irregular.
Sphene	Ca-Ti-Si		Platy or wedge shape.
3. Iron (oxides)	Fe, Ti (may also include Al, Si, and trace elements)		Variable but often in submicron spheres or sphere clusters. Also exists as coatings on suspended matter of the bight apex.
4. Titanium (oxides)		13	
5. Fly ash	Al/Si = 7/9 — 9/8, trace K, Ca, Fe	14	Spheres—1–40 μm in diameter.

more than 100–200 m above the bottom. During the July 1975 cruise at many stations an extra bottle was taken in the surface water for suspended particles only. These profiles exhibit a great deal of variability in concentration vertically, from station to station and, in some instances where stations have been reoccupied, in time as well. Here we will present only the more important aspects of the data since the study is still in progress.

Concentration data (Figs. 2–4) have the following limitations. First, because of the great variability in water depth and because sampling was done with respect to the sediment–water interface, except for surface concentrations, the maps show concentrations at a given height or integrated through a height interval above bottom. Data from different topographic settings are thus shown together, e.g. flat shelf and the Hudson Shelf Channel. Second, presentation of data from a 2-week cruise together on a map presumes synopticity of the data which is unlikely. Time scales of at least local variation are shorter than that, but certainly gross regional features remain sufficiently constant that we feel it profitable to present the data in this manner.

Concentrations of suspended particles—Figure 2 shows concentrations of suspended particles 10 m above bottom (mab) in October 1974 and July 1975. To smooth out small-scale vertical variations in concentration, we integrated the concentrations vertically up to a constant height of 30 mab to

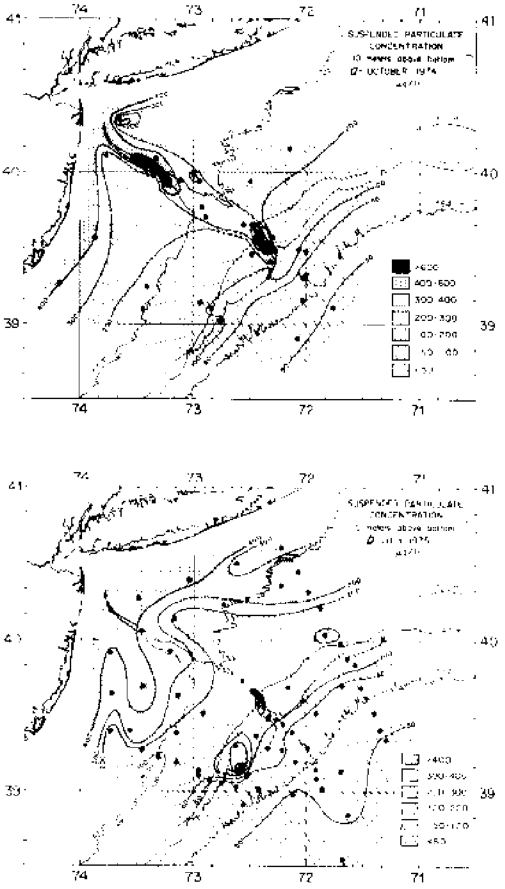

Fig. 2. Concentration of suspended particles (µg/liter) 10 m above bottom (mab) during (a) October 1974 and (b) July 1975.

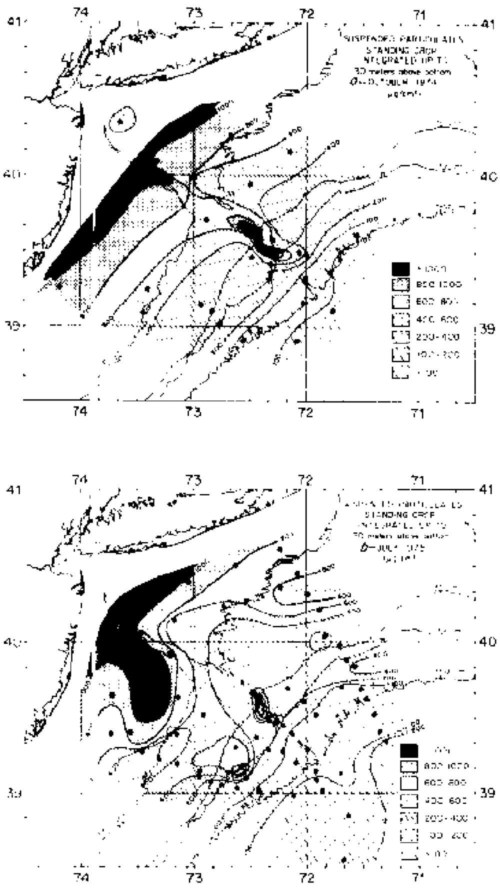

Fig. 3. Standing crop of suspended particles (µg/cm^2) in the bottom 30 m of the water column during (a) October 1974 and (b) July 1975.

yield a standing crop expressed in micrograms per square centimeter (Fig. 3). Despite differences in areal coverage of the two cruises, four general features are apparent.

First is a decrease in concentrations away from the continent. This first-order feature has previously been noted by Manheim et al. (1970) in surface water and Meade et al. (1975) in surface and near-bottom water. Given isopleths of concentration in near-bottom water are displaced seaward (Fig. 2b) compared with surface water (Fig. 4). Concentration levels reported here are comparable to those in similar positions on the shelf reported by the Manheim and Meade groups.

Second, local near-bottom, high concentration anomalies with horizontal scales ranging from perhaps several kilometers to several tens of kilometers are superimposed on the seaward concentration gradient. Comparison of these areas of high concentration or standing crop with the nature of the bottom sediment indicates that they are due to resuspension from areas of fine-grained sediments. Figure 5 shows the distribution of sediment texture in terms of the silt and clay content (finer than 63 μm).

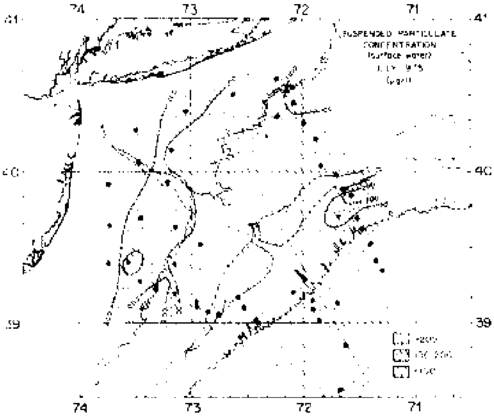

Fig. 4. Concentration of suspended particles (μg/liter) in surface water during July 1975. Note that both the 200- and 100-μg/liter isopleths are nearer the coastline than at 10 mab (Fig. 2b). Note also that local high concentration anomalies at 10 mab (Fig. 2b) such as in the Hudson Shelf Channel and Hudson Canyon do not extend to the surface.

These data are based on sieve analyses done on Shipek grab samples taken during both the October and July cruises and for the more closely spaced sample traverses across the Hudson Shelf Channel during May 1975.

The principal area of fine-grained sediment and source of high suspended particulate concentrations in near-bottom shelf waters in the immediate study area is the Hudson Shelf Channel. A plume of turbid water (400 to >600 μg/liter) was apparently closely associated with fine-grained deposits in the shelf channel during October 1974 but displaced to the southwest during July 1975 (Figs. 2 and 3). Assuming synopticity of the data over the 3 days it actually took to make the stations around the shelf channel, we can apparently trace the advection of resuspended particles for tens of kilometers downstream from their source.

An additional source of suspended particles was noted in the July transect across the shelf southeast of Long Island. Several samples with concentrations >300 μg/liter (at 5, 20, and 30 mab as well as at 10) were separated from the Long Island shore by less turbid water. We believe the source of these high concentrations of particles to be a large area of fine-grained sediment south of eastern Long Island and Block Island (Schlee 1975).

Meade et al. (1975) noted a correlation between high concentrations of near-bottom suspended particles and fine-grained deposits in a more widely spaced sample grid over much of the Middle Atlantic Bight. The data presented here confirm that relationship and suggest rough limits for the horizontal scale over which the relationship exists. Our data thus suggest relatively short resident times of a major portion by weight of the suspended particles in near-bottom waters. If, for example, one uses 5 cm/s as the net southwest drift of shelf water, one arrives at a figure on the order of 1–2 weeks for displacement of the shelf-channel plume of suspended particles during July 1975.

The upper Hudson Canyon is another area of high suspended particulate concentrations in which both concentrations at a given height above bottom and standing crops are as much as a factor of five greater than in the near-bottom waters on the upper continental slope on either side of the canyon. This highly turbid water apparently comes from resuspension of fine-grained sediments. The canyon is a more important source of fine-grained sediment than the Hudson Shelf Channel (Fig. 5). We note a possible displacement up-canyon of the most turbid water with respect to the finest bottom sediment, e.g. the 80% contour (% <63 μm) in Fig. 5 does not extend as far up into the head of the canyon as does the zone of high turbidity indicated by the closed contours in Figs. 2 and 3. Results from a current meter (4 mab) and a long term nephelometer (9 mab) on the same mooring above the axis of the canyon at 802-m depth are to be reported elsewhere (Amos et al. in prep.). These observations (overlapping our October 1974 cruise) indicated current reversals from up- to down-canyon with large variations in velocity and water turbidity. The up- and down-canyon power spectrum of current velocity showed peaks at the inertial and the diurnal and semidiurnal tidal frequencies, but with a net up-canyon velocity of almost 3 cm/s. Power spectrum analysis of the nephelometer record also contained significant tidal-frequency peaks in light scattering. It thus appears that advection of water up- and down-canyon is responsible for the resuspension of fine-grained sediments resulting in the observed high concentrations of suspended particles in the upper Hudson Canyon.

The third general feature is a minimum in the near-bottom suspended particulate concentrations and standing crops, approximately parallel to the shelf break somewhere between the 1,000- and 2,000-m isobaths (Figs. 2 and 3). This is a reversal of the general decrease in bottom suspended particulate concentrations away from the continent. It is also anomalous with respect to the correspondence between high suspensoid concentrations in near-bottom water and underlying fine-grained sediments noted above, i.e. the sediments underlying this minimum zone are very fine-grained. In the region of the Hudson Canyon the anomaly is most striking in that, in terms of percent fines (Fig. 5), the canyon sediment below about 1,000 m is much finer than that in the upper canyon where the water turbidity is much higher. The first two general features cited above (general decrease of bottom suspended particulate concentrations with distance away from the continent and local response of suspended particulate concentrations to bottom sediment texture) thus appear to be interrupted over a substantial zone of the upper continental slope.

A possible origin of this phenomenon is suggested by the excess radon data (Biscaye and Mathieu in prep.). The depth interval in which the minimum in near-bottom suspended particles occurs corresponds to the interval in which excess radon concentrations also exhibit a minimum. Expected fluxes of excess radon from the fine-grained sediments in this zone should be high, making the deficiency of excess radon in the overlying water also anomalous. We conclude that, over the depth zone of the excess-radon minimum, a substantial proportion of the near-bottom water has not been in contact with the sediment–water in-

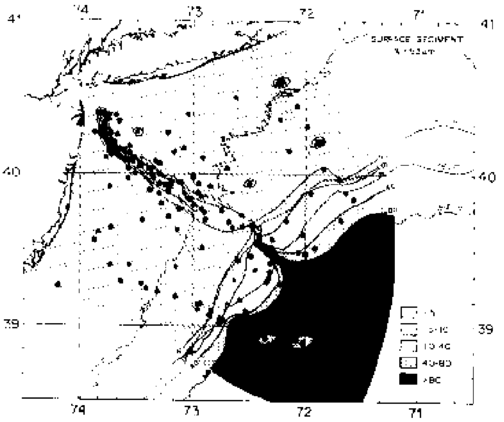

Fig. 5. Distribution of percent fines (< 63 μm) in surface bottom sediments. With the exception of the Hudson Shelf Channel, shelf sediments in the study area contain little fine-grained material. To the east of the study area a large area of the shelf is covered with fine-grained sediment as shown by Schlee (1975); beyond the shelf break, sediments of the upper continental slope and Hudson Canyon floor become increasingly fine-grained with depth.

terface for 1–2 weeks. Because isopleths of temperature, salinity, and oxygen are essentially horizontal in this depth range, the anomalous deficiency in excess radon and suspended particles must represent lateral mesoscale mixing with open ocean waters. Interior ocean water in this depth interval would have relatively low concentrations of suspended particles and be devoid of excess radon.

These anomalies are persistent (we have observed them in April and October 1974, July 1975, and January 1976) and are potentially important to understanding the role of canyons in the exchange of water between the upper continental slope and continental shelf. Control of the depth interval of this phenomenon along the upper slope is spotty, but there are indications from the radon distributions that this water sometimes flows up to the head of the upper canyon. Here, presumably because of turbulent mixing within the confined canyon walls, suspended particulate concentrations increase, but radon concentrations remain low. Given estimates of the flux of radon from the canyon sediment, this phenomenon may allow us to place limits on the residence time of water in the canyon.

Fourthly, where local fine-grained sediments produce high concentrations of suspended particles in the overlying water, the vertical distribution of particles is limited by water stratification. The strong seasonal thermocline at about 30–40-m depth in the shelf waters during the October 1974 observations (Gordon et al. 1976) coincided with the upper extent of high suspensoid concentrations, indicating that the thermocline acts as a barrier to vertical mixing of particles. This was also true during July 1975 when the thermocline was slightly shallower. This is seen in individual vertical profiles of concentration (not shown here), but also in the fact that the map of surface concentrations (Fig. 4) shows none of the local resuspension features of Fig. 2b and in the seaward displacement of given isopleths on Fig. 4 (cf. Fig. 2b).

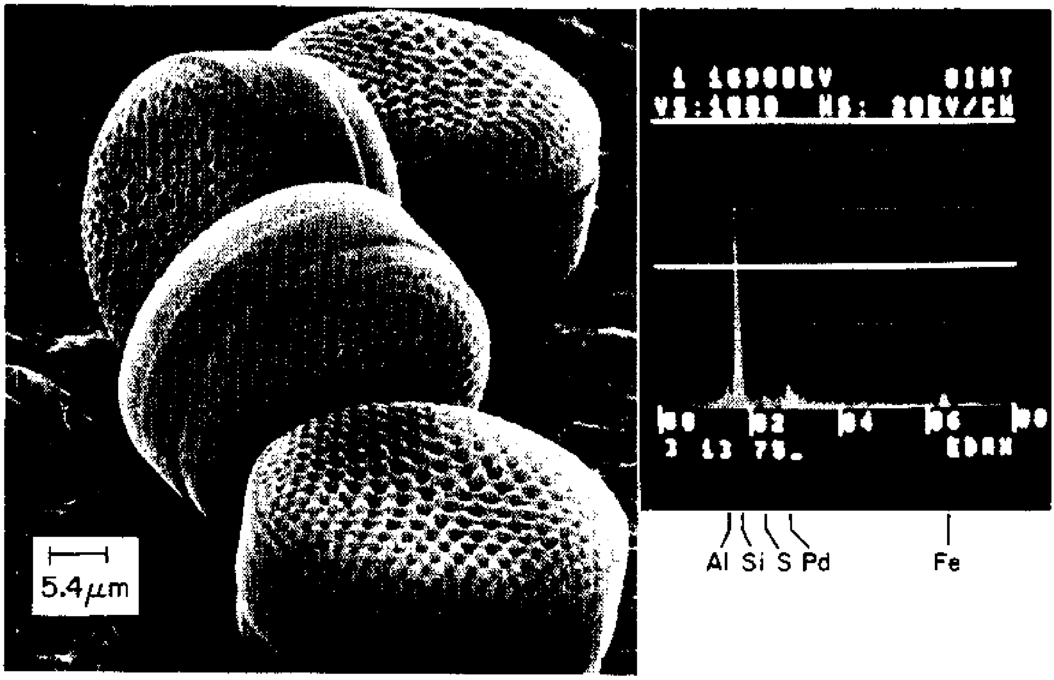

Fig. 6. Siliceous centric diatoms from the bight apex. Note the Fe peak in the X-ray spectrum. Fe and frequently Ti coat much of the suspended matter in the apex. Palladium on this and all the X-ray spectra shown (Figs. 6–14) is from the metallic vapor coating used to prepare the sample for SEM.

Particle compositions—We grouped suspended particles in the New York Bight into two major categories: biogenic and nonbiogenic (Table 1). Of the biogenic fraction, skeletal debris is easiest to identify and is predominately siliceous, consisting primarily of diatoms (Fig. 6) with some silicoflagellates and a few radiolarian fragments. Calcareous skeletal debris (primarily coccolithophorids) became increasingly more abundant in waters near and beyond the shelf break.

Nonskeletal biogenic debris (termed organic) is a much more complicated subdivision and includes a wide range of morphologies and elemental compositions. Organic matter can originate from in situ biogenic productivity in surface waters and consists of soft-bodied plants and animals and their waste and decay products. Organic particles are also introduced by estuarine runoff or dumping of sewage sludges and dredged wastes. There appear to be regional differences in types of organic matter.

Both "smooth" (Fig. 7) and "rough" (Fig. 8) organic matter (Table 1) comprise a considerable portion of the suspended material in the bight apex and along the New Jersey coast. These nearshore organic particles are characterized by strong P peaks relative to Si and Al, indicating little aggregation with or incorporation of aluminosilicate material. Organic particles in the apex contain significant concentrations of Fe, Mn, Ti, Cu, Sn, Cr, Zn, Pb, Ni, and As. Although organic particles containing Fe and Mn are ubiquitous in the New York Bight, they were observed most frequently in New Jersey nearshore waters. Both the high concentration of organic particles and the high concentrations of Fe and Mn associated with these organics may reflect input of nutrients and metals from the Hudson River estuary or via ocean dumping. Organic particles containing detectable quantities of Cu, Zn, Cr, Ni, Pb, As, and Sn, however, appear to be most abundant in the bight apex (Fig. 8). Occasionally similar rare-metal-enriched organic particles were observed off the New Jersey coast and in upper and intermediate outer shelf waters. The most likely source for these trace-metal-bearing organic particles is sewage sludge and dredge spoils dumped in the apex. Gross (1970) showed that sewage sludge contains 45% aluminosilicates, 55% organic

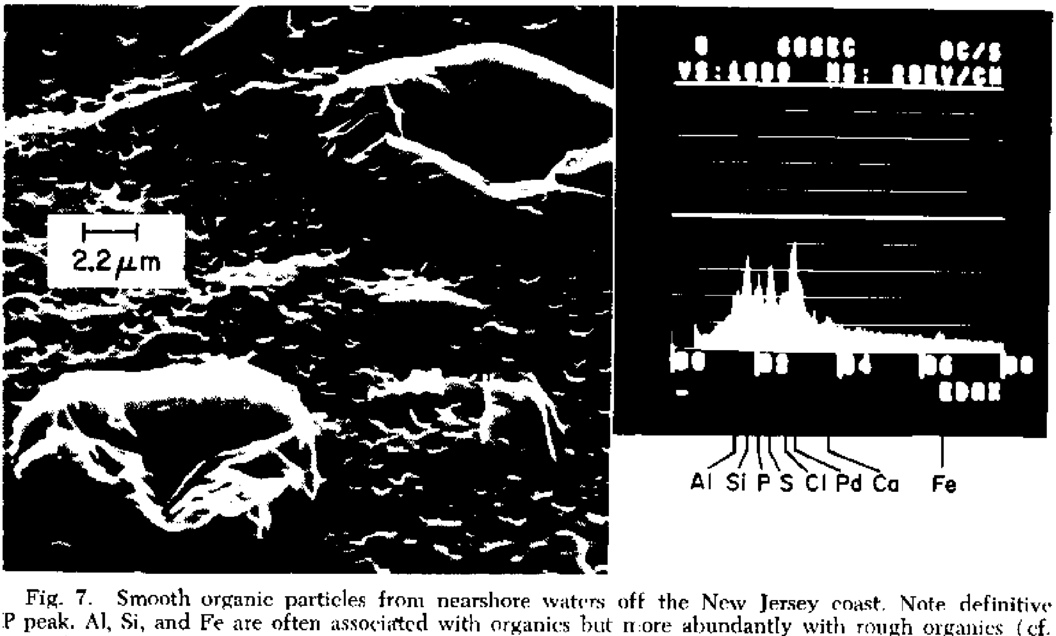

Fig. 7. Smooth organic particles from nearshore waters off the New Jersey coast. Note definitive P peak. Al, Si, and Fe are often associated with organics but more abundantly with rough organics (cf. Fig. 8).

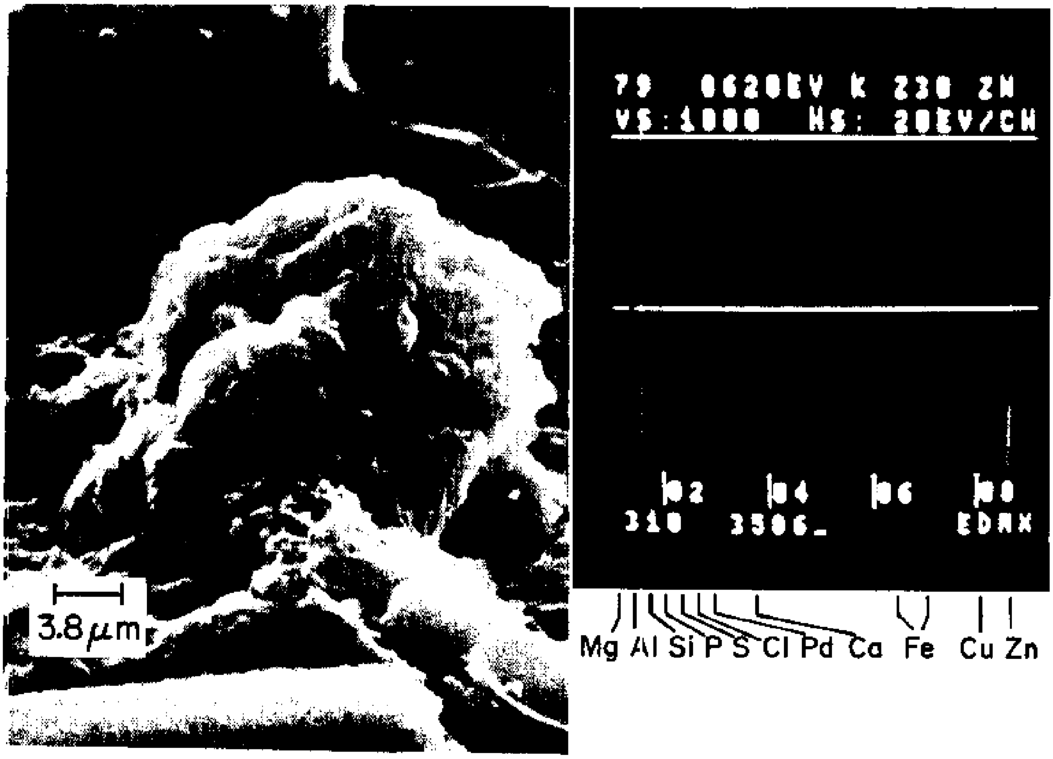

Fig. 8. Rough organic particles from the waters of the bight apex, near the disposal sites for sewage sludge and dredge spoils. Note definitive P peak and high concentrations of Fe, Cu, and Zn.

matter and is enriched relative to soils and sedimentary rocks in metals such as Cr, Cu, Pb, and Sn.

Whenever detectable concentrations of trace metals other than Fe, Mn, and Ti were observed, they were always associated with particles having a pronounced P peak. In addition, Mn appears to be associated most frequently with organic particles whereas Fe was associated with skeletal debris and mineral detritus as well as organic matter. Apparently organic particles play an important role in metal cycles in the bight and may serve as tracers of particle transport processes.

Organic particles in surface and intermediate waters over the outer shelf and upper continental slope may contain Fe or Mn but are relatively depleted in other trace metals. Although smooth organic particles showing strong P peaks completely dominate these waters by number, we question whether they would be as important on a weight basis. Large masses of rough organic-clay aggregates, showing relatively small P peaks are more common in the outer shelf waters, especially near bottom.

Organic matter and skeletal debris comprise a considerable portion of suspended matter in surface and intermediate waters of the bight, but there is a marked decrease relative to the nonbiogenic fraction in near-bottom waters. Meade et al. (1975) reported that the combustible fraction (probably largely organic) in the near-surface waters averaged around 80% whereas near bottom it averaged about 40%.

The nonbiogenic fraction consisted of clay aggregates and individual mineral grains. The clay mineral composition of the suspended matter over the Middle Atlantic Bight is more than 50% illite with less chlorite and still less montmorillonite, mixed layer clays, and kaolinite (Meade et al. 1975). Identifiable mineral grains consisted primarily of quartz, K-feldspar, plagioclase,

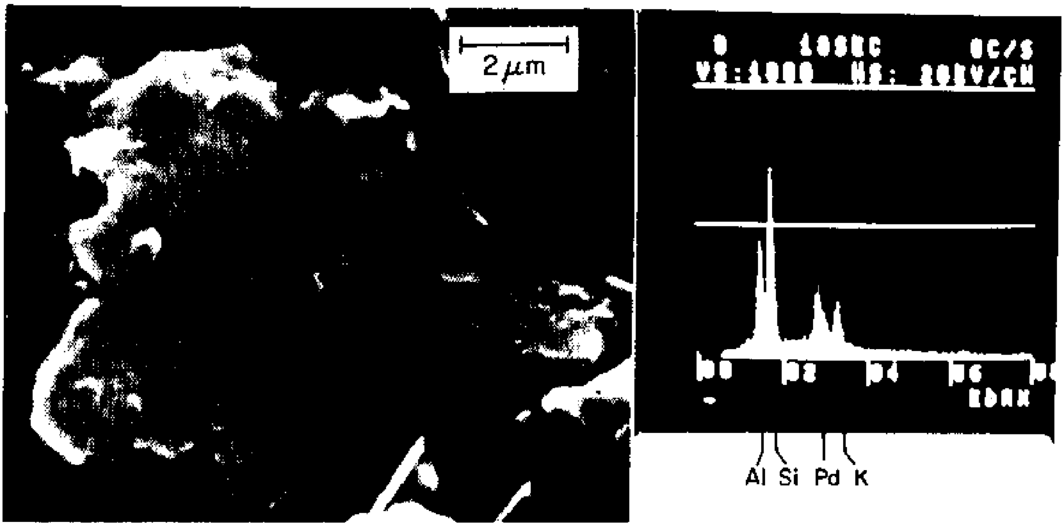

Fig. 9. Muscovite flake from near-bottom waters of the outer shelf.

and muscovite (Fig. 9) with minor amounts of biotite, chlorite, dolomite (Fig. 10), and heavy minerals such as amphiboles, pyroxenes (Fig. 11), sphene, and barite. Preliminary analyses indicate: 1) individual mineral grains, as opposed to aggregates, are relatively more numerous in near-bottom waters overlying coarser-grained bottom sediment areas of the outer shelf; 2) mica flakes, predominantly muscovite with some biotite and Mg-Fe phyllosilicates (chlorite), appear to be more abundant in outer shelf and slope waters relative to more subspherical quartz and feldspar grains; 3) the predominant aluminosilicates are K-rich in nearshore waters and in waters above the thermocline of the outer shelf; 4) by contrast, Mg-Ca-K-Fe aluminosilicates over-

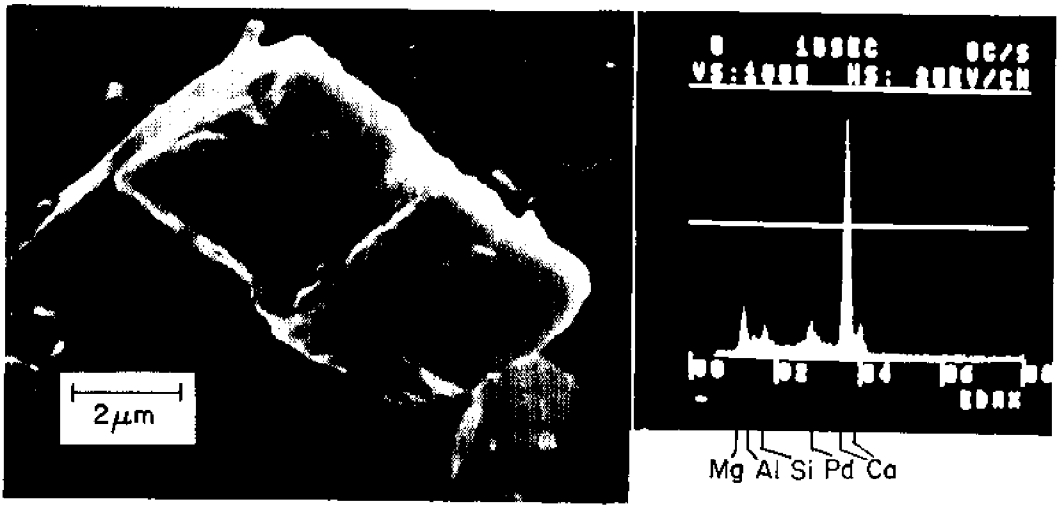

Fig. 10. Euhedral dolomite rhombs from near-bottom waters of the upper continental slope. Concentrations of Mg and Ca are equal but the Mg peak is relatively smaller than the Ca peak due to lower EDXRF detection efficiency for low energy X-rays because of self-absorption in the sample and screening by the detector window. The small Al and Si peaks may reflect a clay coating, but they also occur in the background.

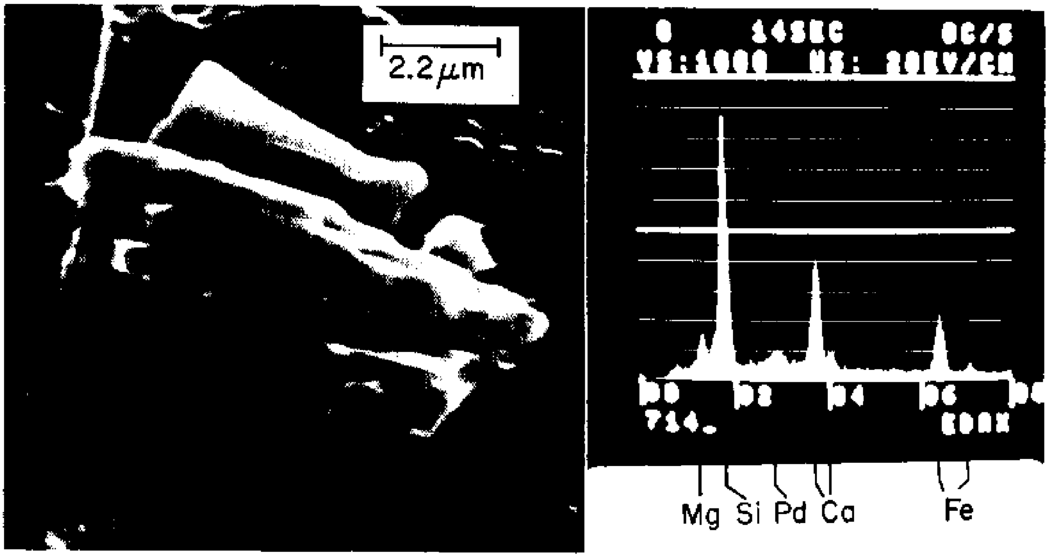

Fig. 11. Pyroxene grain from near-bottom waters of the upper continental slope. Note cleavage which sometimes aids in identification of specific minerals, e.g. pyroxenes, micas, feldspars, dolomite, etc.

whelmingly dominate the suspended particles in near-bottom waters along the upper continental slope and in the Hudson Canyon and are most often seen as large aggregates (5–30-μm diameter; Fig. 12). Occasionally these aggregates contain small

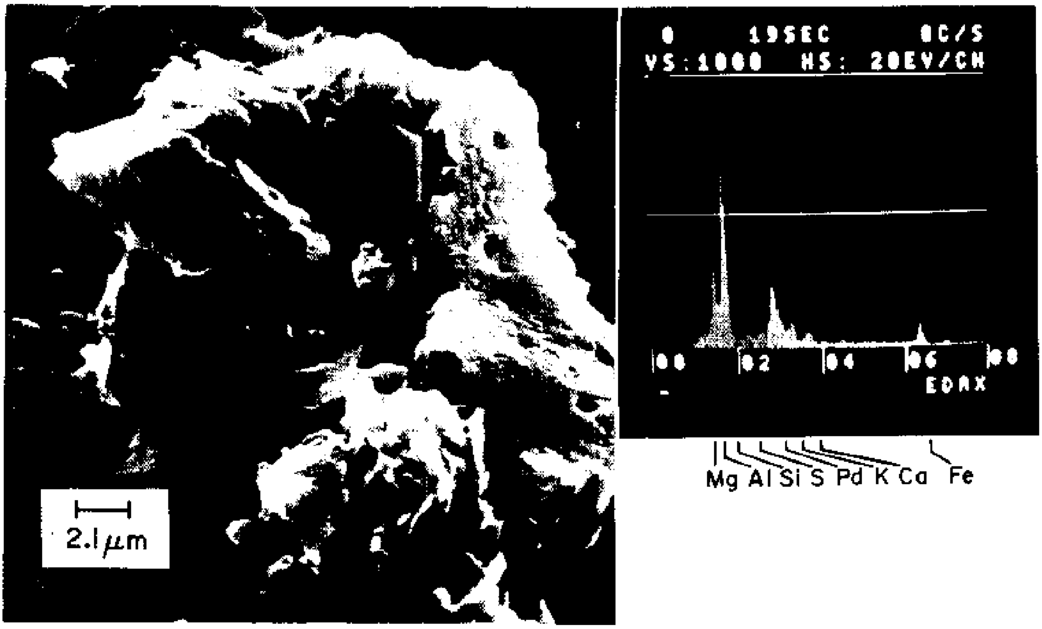

Fig. 12. Mg-Ca-K-Fe-aluminosilicate from near-bottom waters in the Hudson Canyon. Note incorporation of individual mineral grains (nicely cleaved albite grain in upper left-hand corner) into aggregates, of which Mg-Ca-K-Fe-aluminosilicate is the matrix material. These aggregates sometimes exhibit small P peaks indicating presence of organic material.

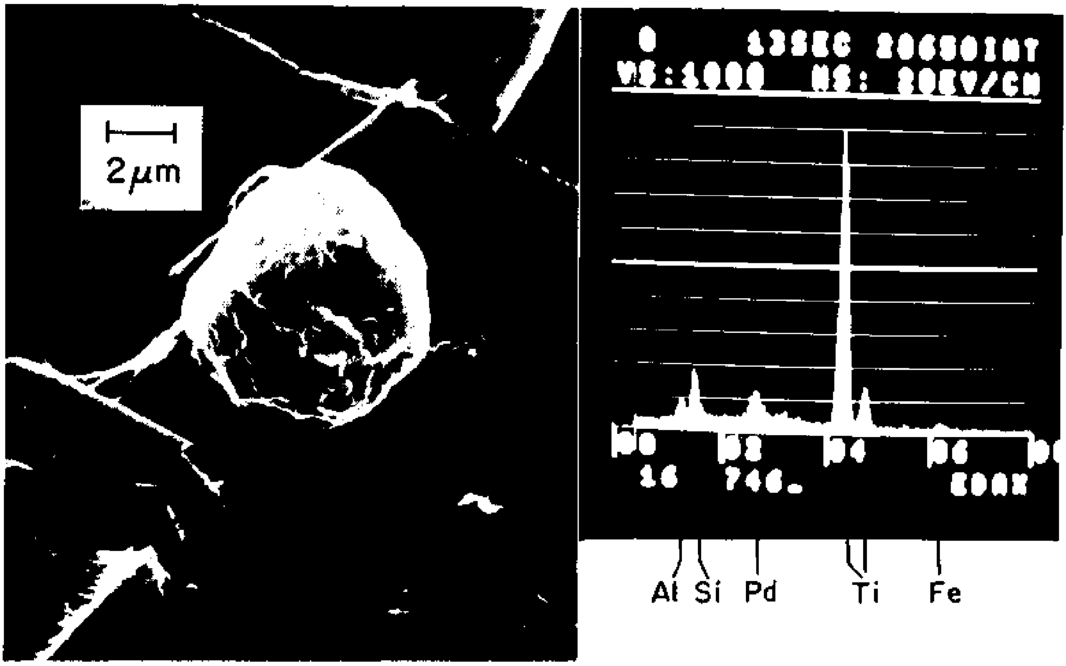

Fig. 13. Discrete Ti (oxide) particle from waters of the bight apex probably derived from acid Fe-Ti waste dumping. Small Al and Si peaks may reflect a clay coating.

amounts of P, S, and/or Cl, suggesting organic matter as an aggregating agent.

Near the disposal sites for sewage sludge, dredge spoils, and acid Fe-Ti wastes, iron and frequently titanium coatings (probably oxides) cover much of the suspended matter, including plankton (*see Fig. 6*). Although precipitation of dissolved Fe from the Hudson River estuary may provide a natural source for the Fe coating, the Ti is

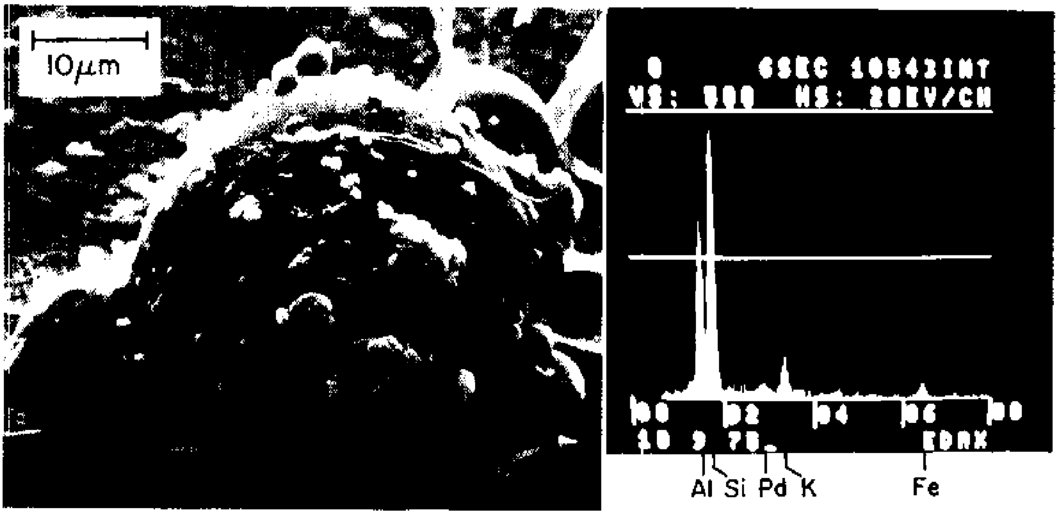

Fig. 14. Fly ash spherules from nearshore waters off the New Jersey coast. Note melted drop morphology. Small spherules attached to large spherule also have the same composition. Al:Si ratio shown here ($\simeq$ 0.7) is the lowest we have observed on spherules and can range up to 1.2.

almost certainly anthropogenic and is derived from acid Fe-Ti wastes. Vaccaro et al. (1972) estimated that ≃50 million tons of acid Fe-Ti wastes have been dumped in the coastal waters off New York since 1948.

Discrete Ti (oxide) particles (Fig. 13), a few microns in diameter, have also been observed at several depths in the bight apex. Similar Ti (oxide) particles have been observed in near-bottom waters of the Hudson Channel, in surface waters along the New Jersey coast and in near-surface waters of the outer shelf. The abundance of Fe-Ti coated and Ti (oxide) particles decreases away from the bight apex, indicating their potential as tracers of particle dispersal.

Fly ash spherules, reported by Manheim et al. (1970), ranging from 1–40 μm in diameter, have been observed at several different depths in the bight apex, along the New Jersey coast, and in waters above the Hudson Shelf Channel. The spherules (Fig. 14) are characterized by a melted drop morphology and by high Al:Si ratios (~0.7–1.2 cf. ≃0.3 for most aluminosilicates) and generally contain some K, Ca, and Fe. Fly ash spherules may come from several sources such as ship traffic, airborne soot from the metropolitan area, and from ocean dumping (coal ash).

Conclusions

Suspended particulate concentrations decrease seaward in both the near-bottom and surface waters but more rapidly in the latter. Patches of anomalously high particulate concentrations in near-bottom waters represent local resuspension of fine-grained sediments. The scale of displacement of such patches from their sources suggests brief residence times for most of the particles. A zone exists near the bottom along the upper continental slope from about the 1,000–2,000-m isobath in which suspended particulate (and excess radon) concentrations are anomalously low. This apparently represents a zone of rapid horizontal mixing with open oceanic water. A seasonal thermocline on the shelf limits vertical mixing of particles resuspended from the bottom. In the bight apex, Fe and Ti coatings cover much of the suspended matter, including plankton. Ti (oxide) also occurs as discrete particles. The number of Fe-Ti coated and Ti (oxide) particles decreases away from the apex, indicating that they may be useful tracers for particle dispersal paths in the bight. Detectable concentrations of trace metals (e.g. Cr, Cu, Zn, Ni, Pb, As, and Sn) are nearly always associated with organic particles (i.e. particles exhibiting a pronounced P peak). In nearshore waters and in waters above the thermocline of the outer shelf, K-rich aluminosilicates are dominant, but in the lower waters of the outer shelf, Hudson Canyon, and upper continental slope, Mg-Ca-K-Fe-aluminosilicates predominate.

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Developmental tests on the use of fluorescent tracers and backwash sediment-load samplers to measure the beach drift component of littoral transport at Sandy Hook, New Jersey

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Abstract

Simple, flat-bottomed, troughlike devices made of sheet metal were used to sample bedload and suspended load of individual backwashes in time-integrated studies of beach drift rate on the foreshore at Sandy Hook, New Jersey. In the most complete experiment first arrival of fluorescent tracer particles, in the medium to coarse sand range, showed a negative linear relationship between particle diameter and beach drift velocity. Peak-of-distribution arrivals, based on smoothed recovery distributions obtained by weight ratio conversion procedures, confirm the inverse sediment size-beach drift velocity relationship.

Increase in mean backwash sediment mass is roughly correlated with increase in the product of breaker height squared and breaker period squared. However, both this relationship and the anomalous variability and apparent periodicity in sediment mass entrained by individual backwashes will require further investigation.

Sandy Hook is a long barrier spit forming a northerly projection from the north end of the New Jersey coastal plain. Wave erosion has smoothed the embayed coast just south of Sandy Hook and thus provided littoral materials for its growth. At present about 376,000 m^3 per year of sediment moves by littoral transport toward the north end of Sandy Hook. This is the greatest volume of coastal sediment movement in the New York Bight. A discussion of topographic evolution and beach processes at Sandy Hook and other areas of the bight is given by Yasso and Hartman (1975).

In spite of the large volume of littoral drift many of the ocean-facing beaches of Sandy Hook are undergoing severe erosion. A survey of wave climate and beach conditions along the entire Sandy Hook shoreline is given by Nordstrom et al. (in press).

How much of the sediment removed from Sandy Hook beaches and moving past these beaches from the south travels on the foreshore? How much travels in and seaward of the breaker zone? What are the specific mechanisms by which beach sediment is entrained and transported in the foreshore and offshore zones? What is the relation between wave parameters and the rate or volume of sediment transport? Such deceptively simple questions cannot be answered yet, except in general terms, for natural beaches.

Dynamics of the foreshore zone

The swash-backwash mechanism is somewhat akin to the flow of blood through the human body. Pumped from the heart the blood travels through the body delivering to the cells oxygen, water, and other life support materials. Waste materials are carried back to the kidneys and lungs so that bodily equilibrium can be maintained.

Breaking waves are analogous to the heart action. On breaking, the wave sends a surge of sediment-laden and highly oxygenated water up the foreshore slope toward the berm. In addition, foreshore sediment is entrained by this turbulent swash. Some of

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[1] C. Ware III and G. Walcott assisted me on the first two experiments reported. Students in my coastal oceanography class assisted in test Y-8-F. Field expenses for the first two experiments were met by U.S. Navy, Office of Naval Research, Geography Branch, project NR 388-057 with the Geology Department of Columbia University. A. N. Strahler was project director during the early phase of the study. R. Stoll and C. J. Galvin, Jr., gave helpful advice on aspects of data analysis. S. Jeffress Williams and N. Psuty reviewed the final manuscript. M. G. Cross encouraged me to prepare this report. M. Lorenz typed the manuscript and provided secretarial support services. J. Pepenella prepared the figures. M. Yasso selected the material and sewed the backwash sample bags. Shop personnel at Lamont-Doherty Geological Observatory allowed me to use their facilities for fabricating backwash samplers.

the swash water percolates into the foreshore sediment to add to or replace groundwater leaving the foreshore at a lower level. After coming to a short-lived rest position high on the foreshore the swash water gathers momentum in a backwash that rushes downward toward the breaker zone carrying with it a sediment load picked up from the foreshore. This sediment, analogous to waste products carried by the blood, represents a size fraction or volume in excess of profile equilibrium requirements.

Krumbein (1963, p. 3) gives an insightful description of the swash-backwash zone:

> The uprush-backwash zone, essentially coincident with the foreshore, lies between the berm and the plunge point where the waves break. This zone is subjected to a succession of highly turbulent up-rushing tongues of water that come momentarily to rest on the foreshore, and then return downslope as backwash less turbulent in its flow. The effect of this continuous to-and-fro motion is to sort out and arrange the foreshore material selectively according to its particle size, shape, and density; and to produce the geometrical form of the foreshore as expressed by slope, width, and height of berm.

If the backwash is in phase with the breakers then its water and sediment load are smoothly incorporated into the next breaker. If out of phase the backwash partially or completely cancels the incoming swash. As the tide rises or falls the effect of breakers, swash, and backwash is felt higher or lower on the foreshore.

A conceptual model

The complex web of relationships between foreshore wave energy, tide level, foreshore material, and foreshore profile geometry can be analyzed in a conceptual model. Dynamic elements of the foreshore model (Fig. 1) include the wave energy (wave height and period) which generates the swash and gravitational force which causes the backwash. Changes in tidal height allow migration of the swash-backwash zone to higher or lower levels of the foreshore. Each swash carries sediment thrown into turbulent suspension at the breaker line. Foreshore sediment is also disturbed and entrained by the swash motion. In addition, tidal migration of the breaker zone landward or seaward causes a much deeper disturbance of foreshore sediment. These combined disturbances are shown by the thick dashed line of the erosional and accretional profiles in Fig. 1. The dashes are thickest at the breaker line and shrink to zero at the upper limit of the swash.

A particular combination of wave height and period may cause foreshore erosion as

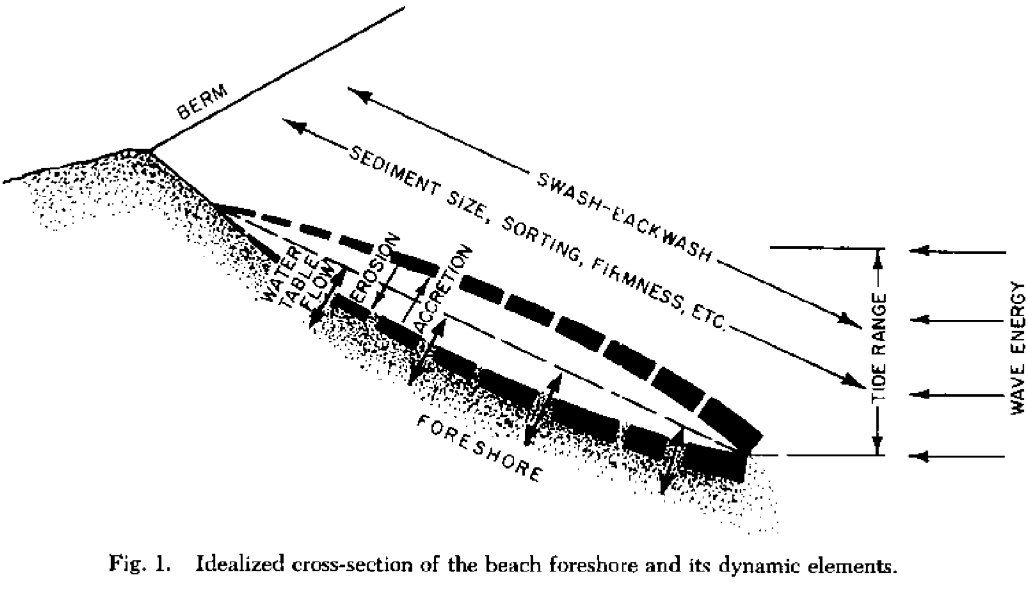

Fig. 1. Idealized cross-section of the beach foreshore and its dynamic elements.

represented by the up-concave curve in Fig. 1. Another combination of wave parameters may cause foreshore accretion as represented by the up-convex curve. Saturation of foreshore sediment by swash water and its flow into or out of the sediment at different topographic levels is thought to be a major control on changes in foreshore profile geometry.

A more complex conceptual model arises if we consider areal characteristics of wave action such as refraction and shore currents or areal changes in shore geometry or materials. Krumbein (1968) suggested three possible paths for greater understanding of conceptual models like the one just discussed. One leads to a deterministic model based on physical principles. A second leads to a statistical predictor model based on processes and responses in the system being analyzed. The third leads to a stochastic process model based on transition probabilities.

Field experiments discussed here represent an initial attempt to understand some of the physical principles pertaining to foreshore sediment movement. The eventual goal is a sharply defined process-response model for the foreshore.

Beach drift studies

Lacking other types of tracer materials Evans (1939) used spheres of wood, rubber, or coal to determine the relation between particle size and beach drift velocity. Beach drift is a foreshore sediment transport process caused by the swash and backwash of waves breaking at an angle to the shoreline. In almost all cases the larger sphere ran ahead of the smaller one having the same composition. Evans ascribed this phenomenon primarily to the smaller spheres being stranded above the swash line more often than the larger spheres.

Apparently, results of similar experiments with large, easily traced objects have engendered the notion that beach drift mechanisms generally cause larger sizes of sediment to travel faster than smaller sizes. This is contrary to logic which suggests that smaller, more easily entrained sand-size sediment particles will also remain suspended for a longer time in the turbulence of swash and backwash motions. Thus, they should be transported along the foreshore more rapidly than coarser sediment.

A field experiment, designated Y-3-F, was conducted at Kingmill Beach, Sandy Hook, N.J., to investigate the relationship between sediment size and beach drift velocity (Yasso 1965). Four size classes of fluorescent tracer sand were used. Only the two smaller tracer sizes were recovered in sufficient quantity for data analysis. However, both first arrival and converted peak-of-distribution arrival for the two sizes of tracer confirmed the expected inverse size-velocity relationship for particle transport.

Because of the difficulty in channel sampling procedures, limited quantity of tracers, and limited sampling time of test Y-3-F, I felt that a further series of experiments might yield better procedures and more complete data by which to evaluate the size-velocity relationship in beach drift transport. Therefore, four additional fluorescent tracer experiments have been performed at Sandy Hook using a backwash sampling technique. One of these, Y-5-F, did not yield data relevant to this discussion. Another set that used coring tubes to instantly sample areal properties of the foreshore during tracer experiments will be described elsewhere.

Backwash sampling

Because only one or two assistants were available for use in tracer experiments, I needed a one-person sampling technique for the time integration study of tracer motion. I could then use field assistants for time-keeping, sample-labeling, photography, and other activities.

Sampling sediment from either the swash or backwash flow suggested itself as a suitable method of obtaining beach drift data. Such sampling may also give an index of material and profile geometry changes on the foreshore. If a foreshore is in a net accretionary phase (prograding) each swash should carry more sediment than a backwash sampled at the same foreshore level. If the foreshore is in a net erosion phase (retrograding) then backwash sediment volume should exceed that of the swash at a given level. Under equilibrium conditions both swash and backwash sediment volume should be equal. Field investigation of this swash-backwash sediment volume concept may reveal that it applies only over a single tidal cycle rather than to an individual swash-backwash couple.

Should the swash-backwash sediment volume concept not prove out one might still be able to determine if either the swash- or backwash-sediment volume is a better index to beach response under a given set of wave conditions. However, for the beach-drift experiments described here, sampling both swash and backwash was not possible. Backwash sampling seemed an easier procedure because the investigator could choose the backwash to be sampled based on the observed swash limit. That is, backwash samples might be from only larger swashes, average swashes, or other swash variations could be chosen.

A method of sampling the moving backwash would avoid the obvious problems associated with foreshore samples taken by conventional means. However, both the suspended and bedload of the backwash should be sampled over the entire time each individual backwash is flowing.

Suction devices for sampling were considered and rejected, because a suction device to obtain a representative sample of sediment in such a complex flow field was beyond practicality for these studies.

A simple, flat-bottomed, sheet metal trough tapering from an opening of 21 cm at one end to 10.6 cm at the other end was made. It was 33 cm long. In use a 57-cm-long by 42-cm-wide sample bag made of finely woven cotton cloth (mesh opening about 0.065 mm) was slipped over the sampler from its back end. One hand held the lower part of the bag and underside of the sampler. The other hand held the upper part of the bag in open position directly above the front of the sampler.

An instant before a backwash flow began the sampler was placed as nearly parallel to the foreshore sediment surface as possible with its front end in contact with the sediment. Suspended and bedload flowed into the sample bag with the backwash water. Occasionally the sampler had to be turned slightly to keep it facing directly into the backwash. At the end of backwash flow the sampler was lifted up to avoid the next swash and to allow slow drainage of water from the bag. Sample bags were tied, labeled, and returned to the laboratory for analysis. This procedure was followed during test Y-4-F.

It was difficult to keep the front end of the sampler exactly at the sediment surface during the violence of backwash flow, so the sampler was modified for later tests. The trough opening was reversed so the 10.6-cm end faced the backwash. A flat, sheet metal baseplate 24.1 cm wide and 26.7 cm long was spot-welded by appropriate supports to the trough. The trough had its front end in contact with the baseplate and 10.6 cm from the front of the plate. The trough angled slightly upward from the baseplate toward its rear to allow sample bag and sampler support as described above. This sampler was used successfully for the subsequent tests.

Procedures and beach conditions: Test Y-4-F—Sediment removed from the foreshore zone at Sandy Hook was dried and sieved on screens of the Tyler fourth-root-of-two series in which the width ratio of mesh openings between successive screens is 1.189. Particles in the four size classes $1.651>d>1.397$, $1.168>d>0.991$, $0.840>d>0.701$, and $0.589>d>0.495$ mm were selected for marking as tracer particles. Size designation indicates that particles in each class passed through the larger mesh opening but were retained on the smaller mesh opening given as the first and second number respectively in each pair listed. Particles were coated with a mixture of Switzer Day-Glo acrylic lacquer and American Cynamid beetle resin (Yasso 1966). Each size class of particles was coated with a different color to allow rapid analysis of samples under longwave UV light. No attempt was made to have the proportion of each tracer size reproduce the size distribution of natural particles on the beach.

About 30 min before high tide 1.7 kg of tracer was released on the foreshore mid-way between the breaker line and swash limit. During the test, wave conditions were as follows:

Mean breaker period (T_b) = 3.4 s.

Mean breaker height (H_b) = 0.42 m.

Maximum breaker height (H_b max) = 0.60 m.

Nearshore wave crest angle to shoreline = ca. 5°.

Included swash-backwash angle = 34.0°.

Foreshore slope angle = 7.3°.

Nearshore wave crest angle with the shoreline was measured by Brunton compass. Included swash-backwash angle was measured by a protractor, attached by a pivot nail through its center to a wooden support driven into the midswash area of the foreshore. Protractor zero-line was aligned with an incoming swash and held in position. Then a thin string attached to the pivot nail was aligned with the backwash flow. The included angle between protractor zero-line and pivot string averaged 34.0° for test Y-4-F.

Backwash sediment was sampled at roughly 3-min intervals 4.6 m seaward from the upper limit of mean swashes. Sediment mass, measured in grams of dry weight, is plotted against sample time in Fig. 2. These samples were taken with the 21-cm end of the original design of backwash sampler. Sediment mass of the backwash averaged 1,417.2 g with a high degree of variability in sediment load between backwashes

Sieving revealed that geometric mean diameter of backwash sediment has a fairly consistent inverse relationship to backwash sediment weight. The dashed line in Fig. 2 shows geometric mean diameter plotted against sample time. Anomalies in the inverse relation between sediment mass and mean diameter might be explained if a complete time history of swashes were available. The overall mean of sample geometric mean diameters (0.69 mm) is slightly greater than geometric mean diameter (0.54 mm) of a 21-cm-deep core of foreshore sediment taken at midtide level during the low tide preceding the experiment.

Samples were dried and examined for tracer content under longwave UV illumination. Fluorescent particles in each size class were easily distinguished by their unique colors. Recovery of tracer particles in each sample is shown in Fig. 3. Sampling did not begin early enough to intercept the first arrival of the smallest tracer. However, first arrivals for the three larger sizes of tracer

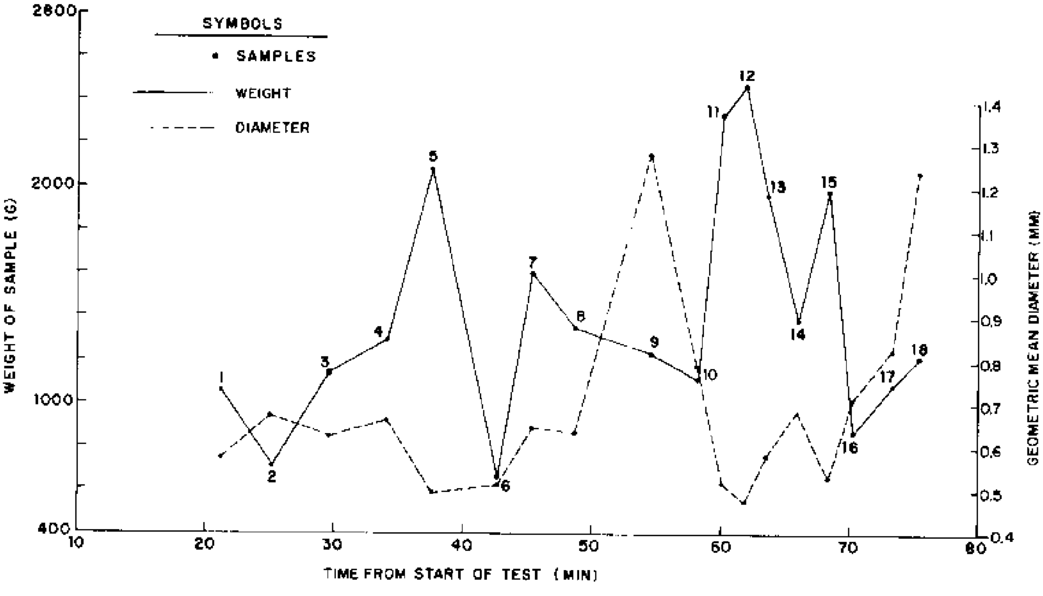

Fig. 2. Distribution of weight and geometric mean diameter for backwash samples, test Y-4-F.

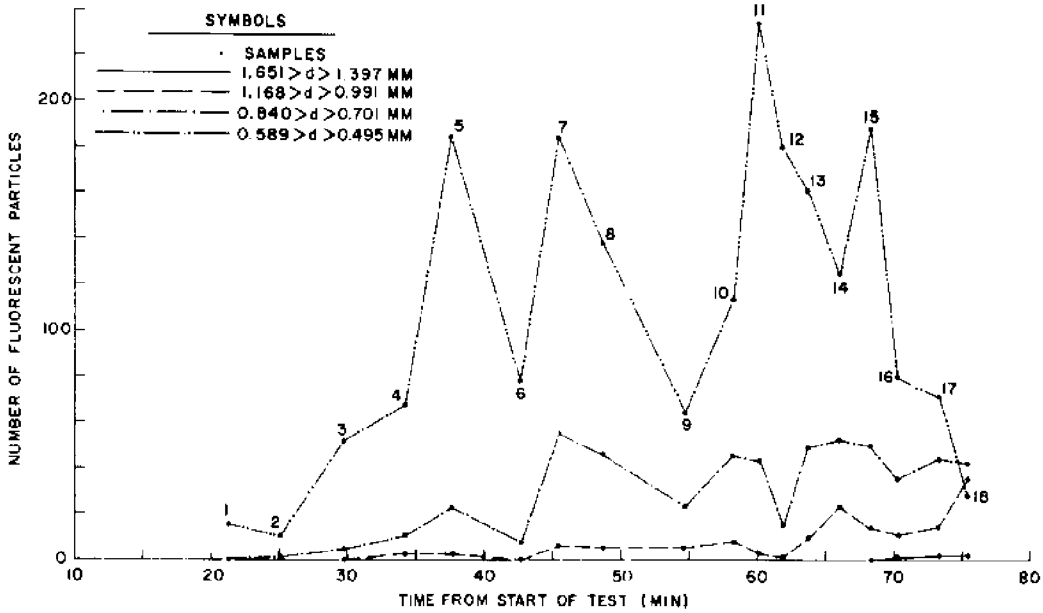

Fig. 3. Recovery distribution of fluorescent tracer particles, test Y-4-F.

show the inverse size-velocity relationship predicted from the results of test Y-3-F.

It is not known exactly which instant the first particle(s) in each tracer size arrived at the 30.5-m sampling line. However, it is reasonable to assume such particles arrived sometime between the time of the sample in which they appear and the time of the previous sample. A mean of the two sample times for first arrival in each class size was used to arrive at the maximum transport velocities listed in Table 1.

When plotted arithmetically, an apparently linear relation between first arrival velocity and particle diameter is revealed (Fig. 4). The equation for this relationship is

$$V_{FA} = -1.94\,D + 3.68,$$

where V_{FA} is the first arrival velocity in cm/s and D is the diameter of the class midvalue of particles in mm. Naturally, different wave conditions are expected to cause at least different negative slopes for lines representing the relation between maximum transport velocity and tracer particle diameter. Large variation in tracer particle recovery from sample to sample makes it difficult to estimate time of passage of the peak of the tracer distribution. Both theoretically and from cursory examination of the curve, there is obviously a discrete peak hidden somewhere in the recovery distribution for at least the smaller tracer size (Fig. 3).

Earlier I suggested (Yasso 1965) that variation in sample size and variation of the

Table 1. First arrival and peak-of-distribution arrival for tracer particles.

Sediment class size (mm)	Class midvalue (mm)	Transport velocity (cm/s)	
		First arrival	Peak arrival
	Y-3-F		
0.701 > *d* > 0.589	0.645	2.79	1.26
0.840 > *d* > 0.701	0.770	1.97	1.16
	Y-4-F		
0.589 > *d* > 0.495	0.542	>2.40	0.86
0.840 > *d* > 0.701	0.770	2.19	—
1.168 > *d* > 0.991	1.080	1.59	—
1.651 > *d* > 1.397	1.524	0.73	—
	Y-8-F		
0.840 > *d* > 0.701	0.770	>7.04	—
1.981 > *d* > 1.651	1.816	>7.04	—
4.760 > *d* > 3.327	4.044	4.87	—

Table 2. Sieve analyses and weight-ratio conversions for backwash samples collected during test Y-4-F.

Sample No.	Total wt (g)	Wt % of fraction	Total wt of fraction (g)	Fluor. particles		Wt ratio ($\times 10^6$)	Wt % of fraction	Total wt of fraction (g)	Fluor. particles		Wt ratio ($\times 10^5$)
				No.	Wt ($g \times 10^3$)				No.	Wt ($g \times 10^3$)	
		—0.589 + 0.495-mm fraction					*—0.840 + 0.701-mm fraction*				
1	1,055.2	28.696	302.805	15	5.245	1.73	9.904	104.508	0	—	—
2	706.8	21.304	150.579	10	3.496	2.32	10.122	71.540	1	0.820	1.14
3	1,140.6	24.258	276.689	51	17.832	6.44	10.165	115.939	4	3.279	8.64
4	1,292.4	20.976	271.092	67	23.426	8.64	8.855	114.446	10	8.200	7.16
5	2,081.4	28.596	595.190	184	64.336	10.81	4.688	97.574	22	18.033	18.48
6	660.2	31.035	204.892	78	27.273	13.31	5.236	34.570	7	5.738	16.60
7	1,600.3	25.929	414.944	184	64.336	15.50	10.834	173.384	55	45.082	26.00
8	1,345.9	22.874	307.856	138	48.252	15.67	11.758	158.254	46	37.705	23.82
9	1,226.4	9.997	122.598	64	22.378	18.25	7.123	87.353	23	18.852	21.58
10	1,107.2	19.420	215.024	113	39.510	18.37	13.665	151.299	45	36.885	24.38
11	2,335.1	29.644	692.227	234	81.818	11.82	5.836	136.269	43	35.246	25.86
12	2,470.9	29.255	722.872	180	62.937	8.71	2.350	58.064	15	12.295	21.18
13	1,963.0	25.160	493.891	161	56.294	11.40	7.741	151.952	49	40.164	26.43
14	1,381.0	23.175	320.052	125	43.706	13.66	10.134	139.948	52	42.623	30.45
15	1,984.7	26.477	525.493	188	65.734	12.51	6.889	136.728	50	40.984	29.98
16	868.4	23.022	199.919	80	27.972	13.99	12.856	111.637	35	28.688	25.70
17	1,082.8	19.025	206.000	71	24.825	12.05	10.235	110.826	44	36.066	32.54
18	1,209.0	6.842	82.719	28	9.790	11.84	8.618	104.194	42	34.426	33.04
		—1.168 + 0.991-mm fraction					*-1.651 + 1.397-mm fraction*				
1		—	—	0	—	—	—	—	—	—	—
2		—	—	0	—	—	—	—	—	—	—
3		—	—	0	—	—	—	—	—	—	—
4		4.731	61.143	2	4.908	8.03	—	—	—	—	—
5		1.158	24.101	2	4.908	20.36	—	—	—	—	—
6		—	—	0	—	—	—	—	—	—	—
7		4.910	78.575	6	14.724	18.74	—	—	—	—	—
8		5.455	73.418	5	12.270	16.71	—	—	—	—	—
9		7.631	93.591	5	12.270	13.11	—	—	—	—	—
10		8.000	88.574	8	19.632	22.16	—	—	—	—	—
11		1.386	32.370	3	7.362	22.74	—	—	—	—	—
12		0.391	9.654	1	2.454	25.41	—	—	—	—	—
13		3.289	64.565	9	22.086	34.20	—	—	—	—	—
14		5.616	77.564	23	56.442	72.76	—	—	—	—	—
15		2.201	43.680	14	34.356	78.76	—	—	—	—	—
16		6.555	56.923	11	26.994	47.42	3.006	26.109	1	6.369	24.39
17		6.660	72.114	14	34.356	47.64	4.885	52.898	2	12.738	24.08
18		11.201	135.416	35	85.890	63.43	10.080	121.870	2	12.738	10.45

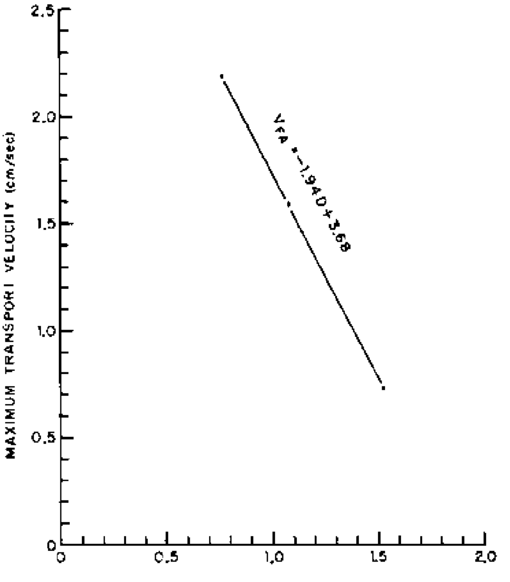

Fig. 4. Relationship between first arrival velocity and particle diameter, test Y-4-F.

weight percent of material in the size classes corresponding to tracer sizes were the major causes of irregularity in the tracer recovery distributions. The more complete data of test Y-4-F allow that hypothesis to be examined. As an example, Table 2 gives weight percent, fraction weight, tracer particle recovery, tracer weight, and weight ratio for each size class of tracers used in the experiment. Size classes are listed as −0.589 + 0.495 mm, for example, which is the same as the designation 0.589>*d*>0.495 mm used elsewhere in this report. Weight ratio, listed in the last column of each set of data, is obtained by dividing total weight of tracer particles in a size class by total weight of all particles in that size class. Complete weight-ratio data for the two smaller sizes of tracer are plotted in Fig. 5. To avoid confusion, the few data points for the largest tracer size are not plotted and only the higher portion of the weight-ratio curve for the second largest tracer size is shown.

Weight-ratio conversion of the smallest tracer particle recovery distribution does reveal a distinct peak in sample 11 which was taken about 60 min from the start of the test. The shape of the weight-ratio distribution curve is of interest. Its distinct peak and sharp drop to sample 12 is similar to shapes of weight-ratio distribution curves found in test Y-3-F. Perhaps the lack of expected symmetry around the peak portion of the curves reflects some physical process of particle transport on the foreshore about which nothing is now known.

Figure 2 shows the efficacy of using weight-ratio data for correcting recovery distribution. That figure shows that sample 11 had neither the largest volume nor the smallest mean diameter of the backwash samples. But, it did have the largest number of tracer particles in the 0.589>*d*>0.495-mm size class of any backwash sample.

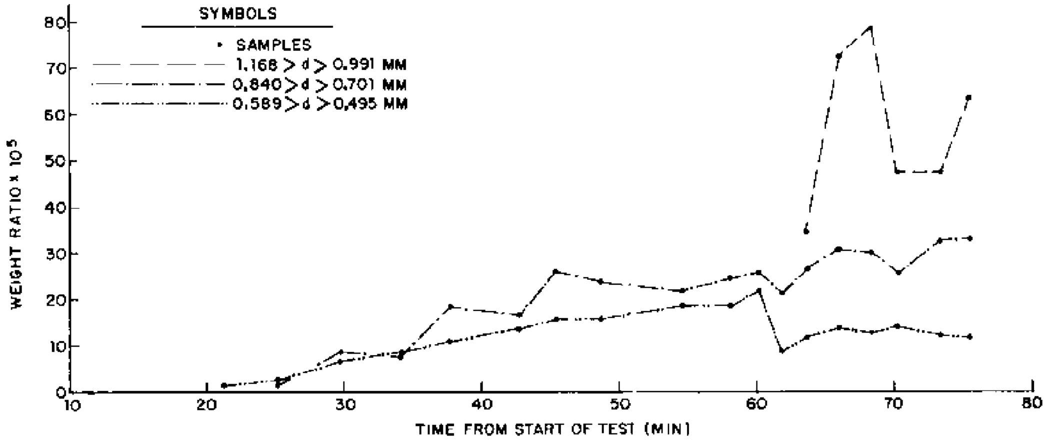

Fig. 5. Weight-ratio conversion of tracer recovery distributions, test Y-4-F.

The correspondence of tracer recovery distribution peak and weight-ratio peak for sample 11 (in the $0.589>d>0.495$-mm size class) suggests that sampling procedures that give uniform sample size will essentially obviate the necessity for laborious sieving or alternate means of determining sediment size. Plastic coring tubes have been used to obtain fairly uniform sample volumes from the foreshore in-between swashes. Results of these recent tests will be reported when data analysis is completed.

A distinct peak is lacking in the weight-ratio plot for the $0.840>d>0.701$-mm tracer size. A peak subsequent to time of the last sample is suggested by the upward trend of the curve. Of interest is elimination in the weight-ratio curve of the spurious peak in the recovery distribution shown by sample 7 in Fig. 3.

Residual variation in the weight-ratio curves for the three largest tracer sizes may result from failure to use a suitably large number of tracer particles. About 700,000 tracer particles of the $0.589>d>0.495$-mm size, but only half as many in the $0.840>d>0.701$-mm size, were used in the test. Proportionally fewer particles in the larger size classes of tracer were used. This is a problem related to acquiring and tagging significantly large quantities of the larger sediment sizes. Numerically, only 0.28% of the smallest tracer was recovered in the relatively large volume samples taken in test Y-4-F. Much larger volumes of tracer may be needed for stability in recovery distributions based on smaller volume samplers such as coring tubes.

Peak-of-distribution transport velocity—As described earlier even though the peak of the weight-ratio distribution occurs at sample 11, velocity of transport of the peak of the $0.589>d>0.495$-mm tracer size is based on mean elapsed time of samples 10 and 11. Whereas peak of the recovery distributions for the earlier test (Y-3-F) could not be confirmed at that time, meaningful peaks in the weight-ratio distribution were demonstrated. Thus the peak-of-distribution velocities for that test are reported with the Y-4-F data in Table 1. However, no additional peak-of-distribution data were yielded by tests Y-5-F, Y-6-F, or Y-8-F. It is premature to speculate on the relation of these data to controlling variables.

Description and results of later tests—For test Y-6-F large volumes of tracer in the $0.840>d>0.701$-mm size class were deposited in succession at the midswash line during midtide conditions at Spermaceti Beach. A differently colored mass of these particles was used at each of four stations spaced at increments of 30.5 m updrift from the sampling line. Tracers at the farthest (121.9 m) station were deposited 0.5 h before those at the next closest (91.4 m) station and so on. This was done to intercept all four recovery distribution peaks over a reasonable length of sampling time.

One goal of the test was to see how the tracer dilution factor varied with distance: how much is tracer recovery dependent on transport distance. A second goal was to learn how the shape of the entire recovery distribution is affected by transport distance. It was assumed that recovery distributions would flatten out for longer transport distances. A third goal was to obtain baseline data to determine if the 4,500 bathers on the beach, beginning immediately updrift from the farthest tracer emplacement, affected sediment transport rate.

Sampling was to have begun 0.5 h after the tracers were deposited at the 30.5-m station: this corresponded to 1.5 h before low tide. Unfortunately, soon after the last mass of tracers was deposited waves were effectively damped out by a submarine bar about 100 m offshore.

Visual examination of foreshore sediment at the sampling line showed that tracers were not present at their anticipated arrival time. Swash and backwash were virtually absent from the foreshore until about 1.5 h past low tide. Tracer particles were seen to be moving past the sampling line 3.5 h past low tide. Sampling at 3-min intervals began 19 min later. The backwash sampler with baseplate was used for sampling.

Sample analysis yielded the data for each tracer color-listed in Table 3. No first arrival or peak-of-distribution velocities can properly be elicited from these data. Fluo-

Table 3. Sieve analyses and weight-ratio conversions for backwash samples collected during test Y-6-F.

		−0.840 + 0.701-mm fraction		Fluor. particles			Fluor. particles		
Sample No.	Total wt (g)	Wt %	Total wt (g)	No.	Wt ($g \times 10^3$)	Wt ratio ($\times 10^5$)	No.	Wt ($g \times 10^3$)	Wt ratio ($\times 10^5$)
				Orange (emplaced at 30.48 m)			*Green (emplaced at 60.96 m)*		
1	1,846.5	10.037	185.333	0	—	—	14	11.475	6.192
2	1,842.1	7.431	136.886	3	2.459	1.796	12	9.836	7.186
3	2,318.2	7.004	162.367	1	0.820	0.505	13	10.655	6.522
4	1,774.3	8.209	145.652	4	3.279	2.251	15	12.295	8.441
5	2,708.9	6.202	168.006	6	4.918	2.927	19	15.573	9.270
6	1,105.2	5.853	64.687	0	—	—	2	1.639	2.534
7	2,499.1	7.663	191.506	5	4.098	2.140	14	11.475	5.992
8	2,791.1	4.369	121.943	0	—	—	3	2.459	2.016
9	1,329.3	8.287	110.159	3	2.459	2.232	7	5.737	5.208
10	783.2	7.296	57.142	0	—	—	0	—	—
11	2,164.3	7.703	166.716	1	0.820	0.492	17	13.934	8.358
12	2,690.1	7.178	193.095	3	2.459	1.273	8	6.557	3.396
13	2,698.2	7.346	198.210	2	1.639	0.827	2	1.639	0.827
14	3,517.4	3.851	135.455	1	0.820	0.605	6	4.918	3.631
15	2,351.8	4.410	103.714	2	1.639	1.581	3	2.459	2.371
16	2,447.3	5.443	133.206	2	1.639	1.231	3	2.459	1.846
17	2,231.2	7.320	163.324	1	0.820	0.502	5	4.098	2.509
18	2,038.4	8.734	178.034	2	1.639	0.921	8	6.557	3.683
19	1,847.1	8.467	156.394	2	1.639	1.048	5	4.098	2.620
20	1,801.7	5.979	107.724	3	2.459	2.283	10	8.196	7.609
21	2,907.0	6.405	186.193	1	0.820	0.440	2	1.639	0.880
22	1,830.2	7.729	141.456	2	1.639	1.116	5	4.098	2.897
23	2,576.3	9.023	232.460	6	4.918	2.116	6	4.918	2.116
24	2,897.2	7.557	218.941	4	3.279	1.498	9	7.377	3.369
25	2,693.0	9.334	251.365	5	4.098	1.630	7	5.737	2.283
26	1,823.8	10.493	191.371	6	1.918	2.570	4	3.278	1.713
27	1,439.2	10.895	156.801	4	3.279	2.091	2	1.639	1.045
				Red (emplaced at 91.44 m)			*Yellow (emplaced at 121.92 m)*		
1				2	1.639	0.884	1	0.819	0.442
2				1	0.819	0.599	0	—	—
3				6	4.918	3.029	0	—	—
4				8	6.557	4.502	1	0.819	0.563
5				13	10.655	6.342	0	—	—
6				6	4.918	7.603	1	0.819	1.267
7				8	6.557	3.424	1	0.819	0.428
8				6	4.918	4.033	1	0.819	0.672
9				7	5.737	5.208	2	1.639	1.488
10				4	3.278	5.738	0	—	—
11				12	9.836	5.900	0	—	—
12				12	9.836	5.094	1	0.819	0.424
13				18	14.754	7.444	2	1.639	0.827
14				10	8.196	6.051	4	3.278	2.420
15				8	6.557	6.322	4	3.278	3.161
16				12	9.836	7.384	1	0.819	0.615
17				14	11.475	7.026	6	4.918	3.011
18				25	20.491	11.510	5	4.098	2.302
19				7	5.737	3.669	5	4.098	2.620
20				10	8.196	7.609	5	4.098	3.804
21				14	11.475	6.163	9	7.377	3.962
22				5	4.098	2.897	6	4.918	3.477
23				8	6.557	2.821	11	9.016	3.879
24				15	12.295	5.616	11	9.016	4.118
25				8	6.557	2.609	10	8.196	3.261
26				8	6.557	3.426	6	4.918	2.570
27				4	3.278	2.091	10	8.196	5.227

rescent tracer recovery is especially sparse considering that 1.22×10^6 particles were used in each of the three farthest tracer deposits. About 610,000 particles were deposited initially at the 30.5-m station.

Weight of backwash samples recovered in test Y-6-F is plotted against sample time in Fig. 6. Four geometric mean diameters show the expected inverse relation between sample size and mean diameter for samples taken 4.6 m from the mean upper swash line.

During the next experiment at Kingmill Beach samples were collected at the mean backwash limit over a roughly 2-min interval 11.2 m downdrift from the midswash point of sample introduction. Even though sampling began 2.2 min after tracer introduction, the first arrival of the two smallest tracer sizes (medium and coarse sand) preceded the time of first sample. However, first arrival of the largest tracer size (fine gravel) was found in the second sample taken 4.1 min after tracer introduction.

Because test Y-8-F was primarily a coring-tube study of areal distribution on the foreshore of tracer over time, the effort at backwash sampling was terminated before peak-of-distributions were intercepted. Appropriate first arrival velocities are listed in Table 1. Most importantly fine gravel and coarse sand particles appear to travel according to the inverse size-velocity relation found for finer sand sizes.

Significance of backwash sediment mass —Mass distribution of all backwash sediment samples is plotted against sample time intervals in Fig. 6. The data for test Y-4-F have been halved to make them equivalent to the data obtained by the smaller trough opening of the improved backwash sampler used in tests Y-6-F and Y-8-F. Maximum breaker height and mean breaker period are also given to the right of each data plot. Plots shown in Fig. 6 suggested a possible direct correlation between mean sample weight and the product of the square of wave height and square of wave period (H^2T^2 is proportional to wave energy).

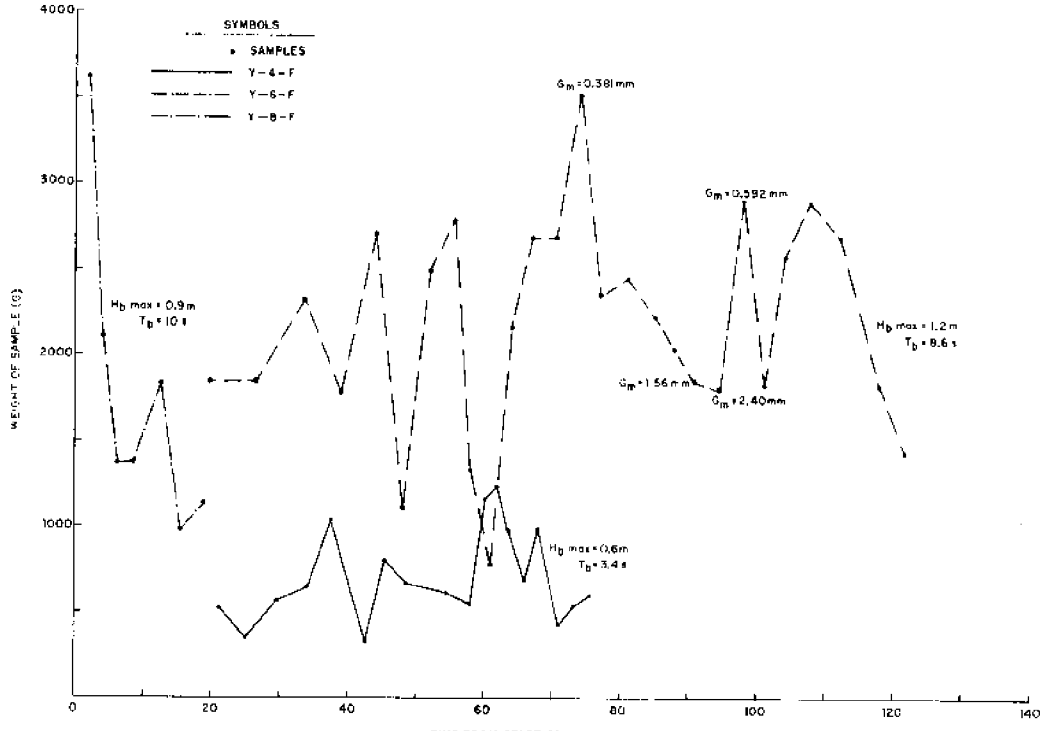

Fig. 6. Distribution of sample weights.

An arithmetic plot (not shown) of mean sample weight versus product of squared maximum breaker height and squared mean breaker period suggests a positive linear relation between them. However, this is a highly tentative inference. The true relationship between mean backwash sediment mass and wave characteristics expressed in this manner or by using significant waves or other parameters may be elicited from a new test program now in progress.

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Section 4

Waste sources and effects

Sources of urban wastes[1]

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Abstract

The coastal ocean has long been used by cities along the Middle Atlantic Bight—particularly in the New York–New Jersey metropolitan region—to dispose of municipal, industrial, and dredged wastes; the volumes increased by more than 5% per year in the early 1970s. Sediment eroded from agricultural land and from construction sites must be dredged from navigation channels after deposition by rivers. Solids from sewage (treated and untreated) mix with the riverborne sediments so that large volumes of dredged materials must be handled as wastes. Riverborne sediment load and littoral drift must be dredged from navigation channels; that material is now dumped at sea. Industrial wastes, such as from titanium dioxide production and coal ash, have been dumped at sea. Others, such as steel-making slag, have been used for landfill. Construction and demolition debris have been dumped at sea when no landfill sites were available. Refuse, garbage, and incinerator ash are commonly disposed of in coastal wetlands.

Alternative disposal strategies and sites will be needed to supplement present regulations to reduce the urban wastes now dumped at sea.

Urban areas are prolific producers of a wide variety of wastes (Table 1) because of their dense populations, the many industries located in and around them, and local dislocations of sediment transport caused by harbor facilities. The materials involved—both solids and liquids—are considered wastes because they have no commercial value in their present form or location. This review deals primarily with waste solids; liquid wastes, particularly nutrients and their effects on urban waterways, were discussed in earlier symposia (Nat. Acad. Sci. 1969; Likens 1972).

Waste solids are usually mixed with other materials such as river sediment and so require special handling because some fraction of the mix imparts undesirable physical, chemical, or bacterial attributes to the whole. For example, sewage-derived solids and petroleum wastes mix with riverborne sediment so that all dredge spoils from navigation channels must be handled as wastes.

Coastal urban areas of the United States have limited opportunity to use the traditional landfill for waste disposal. Sites for landfills are scarce in coastal urban areas and it is difficult to obtain consent of the various surrounding political jurisdictions for waste disposal operations.

Without adequate land disposal sites, coastal cities have turned to the coastal ocean as a convenient, relatively inexpensive site for disposal of waste. Ocean waste disposal has been practiced in the New York region since the late nineteenth century.

Despite long term use of coastal ocean areas for disposal of urban waste, ocean disposal has received much less scientific attention than air pollution or disposal of

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[1] Chesapeake Bay Institute Contribution 227.

Table 1. Common urban wastes: sources, composition, and usual disposal areas (after Gross 1972).

Wastes	Sources	Major constituents	Minor constituents	Disposal area
Municipal	Domestic, industrial	Paper and wood (50%) Food wastes (12%)	Glass, stones (10%) Metals (8%)	Landfill, incineration
Dredge spoil	Harbor and channel construction and maintenance	Sand, shell, gravel River sediment	Sewage solids Industrial wastes	Ocean
Rubble	Construction and demolition	Stone, concrete, steel		Landfill, ocean
Sewage solids	Municipal sewage systems and treatment plants	Organic matter (50%) Aluminosilicates (50%)	Industrial wastes	Ocean, harbor
Coal ash	Coal combustion, primarily power generation	Quartz, mullite		Ocean
Fermentation byproducts	Pharmaceutical industry	Organic matter	?	Ocean
Acids	Metal processing	?	?	Ocean
Alkalis	Petrochemical industry	?	?	Ocean
Steel-making slag	Steel production	FeO (25%) CaO (40%) SiO_2 (17%)	MgO (9%) MnO (4%) Al_2O_3 (1%) TiO_2, S, P_2O_5 Na_2O, K_2O Cr_2O_3, V_2O_5 Firebrick rubble	Landfill, harbor fill

garbage and rubbish. For example, Bower et al. (1968), in their study of waste management in the New York–New Jersey metropolitan region, did not discuss ocean disposal. The North Sea receives an estimated 7 million tonnes of dredged materials and 3 million tonnes of industrial wastes (McCave 1973), but data on amounts and composition are extremely scarce. (A tonne is 2 205 pounds.)

Waste disposal in U.S. coastal ocean waters has long been subject to judicial and legislative control. For instance, New York City dumped its garbage at sea for decades until the Supreme Court in 1933 (New Jersey vs. City of New York, 290 U.S. 237, 1933) upheld a ruling that the city cease such disposal. In 1973 the Marine Protection, Research, and Sanctuaries Act of 1972 (33 U.S.C. 1401) established a permit system for ocean dumping based on environmental effects.

Because of this permit procedure, information is now available on the volume and composition of wastes dumped at sea from urban areas. By combining data on ocean dumping with published reports of other urban waste sources, we can more accurately assess the amount and type of urban wastes that reach the ocean, whether carried by rivers or dumped from barges and dredges.

In 1974 the Middle Atlantic Bight (Cape Cod to Cape Hatteras) received the greatest variety of wastes—industrial, municipal, and dredged—of any coastal ocean area in the United States (EPA 1975). Of all EPA-issued permits for ocean waste disposal, 75% were for the middle Atlantic region. In addition to the nearly 10 million m^3 of dredged wastes, the region's waters received 6.1 million m^3 of sewage sludges—the only U.S. coastal ocean area receiving such wastes. And 4.4 million m^3 of industrial wastes, primarily TiO_2 processing wastes, were also dumped in these waters (Table 2). Not only were the amounts large but they were increasing at annual rates of 4.5% to 93.1% (Table 3).

Data discussed here come primarily from the New York–New Jersey metropolitan region—one of the world's largest urban areas. Some waste sources, such as riverborne wastewater, sewage solids, and demolition debris, are common to all cities in developed countries; others, particularly in-

Table 2. Wastes dumped on middle Atlantic continental shelf, 1974 (from EPA 1975: Regions II and III).

Type of waste	Volume (10^6m^3)	Volume (%)	Solids (10^6 tonnes)	Solids (%)
Dredge spoils	9.9	47.0	5.0	73.6
Sewage sludges	6.1	29.0	0.3	4.4
Rubble	0.6	3.0	1.2	17.6
TiO_2 wastes	3.1	14.8	0.3	4.4
Other chemical	1.3	6.2	?	—

Table 3. Waste dumping in the Atlantic Ocean in millions of tonnes, wet weight (from EPA 1975).

	1973	1974	Change (%)
Industrial waste	4.0	4.8	+19.3
Sewage sludges (wet weight)	5.4	5.7	+4.5
Construction and demolition debris	1.2	2.2	+93.1
	10.6	12.7	+19.8

dustrial wastes, vary widely with local conditions.

Riverborne solids and dredge spoils

Most coastal cities in the New York–New Jersey metropolitan region are located on estuaries. Along the U.S. Atlantic coast, many cities are close to heads of navigation channels where water and sediment from rivers enter the marine environment. In such locations riverborne sediment is deposited where downstream movement of water along the bottom is matched by upstream movement from estuarine circulation (Pritchard 1955).

Navigation channels are often depositional sites for much of the riverborne sediment load, necessitating frequent dredging to maintain channel depths. The problem of sedimentation and dredging in the lower Hudson River was reviewed by Panuzio (1965). Between 1930 and 1970 an estimated 2.2×10^6 tonnes of wastes were dredged each year from New York Harbor. This compares to 2.4×10^6 tonnes carried each year by the region's rivers and by littoral drift (Gross 1974).

Because of the shortage of on-land disposal sites for the dredged deposits and because of the high carbon and bacterial counts from mixing with sewage and other waterborne wastes, dredged wastes from urban areas are often dumped at sea (Gross 1972). In the New York–New Jersey metropolitan region, this was accepted practice even before 1890. In the mid-1960s dredged wastes deposited at sea amounted to 5.7×10^6 m^3/yr or an estimated 3.5×10^6 tonnes/yr (Gross 1972). Because of the amount of dredged wastes coming from urban regions, the sources and probable future supply of riverborne solids to urbanized coastal areas are worth examining.

Agriculture and mining in upriver areas are long-standing sediment contributors (Kenahan 1971). In urban areas the principal sediment source is erosion of sites left bare during construction (Wolman 1967; Wolman and Schick 1967). As Table 4 indicates, erosion of construction sites yields 10 to 100 times more sediment per unit area than mining or agriculture. The sediment yield to rivers from agriculture has decreased markedly from its peak in 1900 to 1920 because the amount of land farmed has declined and because conservation practices have improved. For example, sediment yield from an area in the southern Piedmont (South Carolina, Alabama, Georgia) decreased from about 200 tonnes/km^2 per year in 1920–1934 to about 30 tonnes/km^2 per year in 1967–1972 (Meade and Trimble 1974).

Despite decreases in sediment yield to rivers soils eroded from farm land in past decades doubtlessly reach urban estuaries, especially during floods. Only about 5% of the soil eroded from upland slopes since European settlement has reached the ocean (Trimble 1975). The remainder is deposited temporarily in stream channels and banks (Wolman 1967; Wolman and Schick 1967) or impounded by dams (Meade and Trimble 1974). These deposits are scoured during high river flow and carried downstream; some of this sediment load is deposited in urban estuaries. For example, the Susquehanna River transported an estimated 30×10^6 tonnes of sediment during floods from Tropical Storm Agnes in 1972 (Schubel

Table 4. Sediment yield in Maryland (from Wolman and Schick 1967).

	Sediment yield (tonnes/km^2 per year)	
Areas or activities (No. of observations)	Range	Median value
Wooded areas (20)	5.3–73	50
Rural areas (8)	82–320	180
Strip mine (1)	1,040	1,000
Urban and development (4)	37.5–820	600
Housing and industrial construction (6)	2,000–28,000	10,000

1974); the river normally transports 0.5 to 1.0×10^6 tonnes per year (Schubel 1968, 1972). Studies of sediment deposits from hydraulic mining during the California Gold Rush showed that these deposits were mobilized over decades, not affecting the San Francisco Bay region until nearly a century later (Gilbert 1917; Smith 1965).

Erosion accompanying conversion of crop- or woodland to housing developments is another major sediment source in urban areas (Wolman 1967). The sediment eroded from a small construction site can equal 28,000 tonnes/km^2 per year (Wolman and Schick 1967).

Deposition from such large sediment yields causes extensive alteration of stream channels. For the Baltimore–Washington metropolitan area, Wolman and Schick (1967) estimated that 630–1,600 tonnes of sediment are mobilized per 1,000 increase in population. Assuming comparable erosion in the New York–New Jersey metropolitan region, we can estimate that as much as 2×10^5 tonnes of sediment were mobilized by the addition of 200,000 persons between 1971 and 1972 as approximately 75 km^2 of agricultural land was converted to suburbs (Tri-State Regional Planning Comm. 1973). Eventually, streams in urban areas may be channelized and covered over to become sewers. When that happens, the sediment load is mixed with sewage and becomes part of the region's sewage discharge.

Alternative strategies for dealing with sediments and reducing dredging requirements are discussed later.

Sewage and sewage solids

Because of potential public health hazards, sewage and sewage solids are among the most troublesome wastes originating in cities.

Storm runoff and used water, with their waste loads, are discharged into the sewers in areas of high population density and directly into waterways in areas less densely populated. About 82% of the population in the New York–New Jersey metropolitan region is served by sewers in areas (Fig. 1) covering about, 3,350 km^2 (Tri-State Regional Planning Comm. 1973). Per capita sewage discharge in 1972 was 660 liters/day—an increase from 545 liters/day in 1963. Most (85%) of the wastewater discharge comes from industrial activities (Table 5), especially cooling waters from steam-electric generating plants.

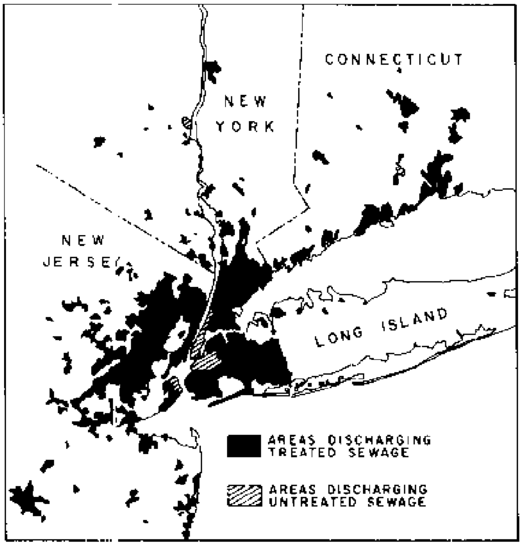

Fig. 1. Sewered areas in the New York–New Jersey metropolitan region (modified after Tri-State Planning Comm. 1969).

In the mid-1960s per capita domestic wastewater discharges in the New York–New Jersey metropolitan region were 230 liters/day in unsewered areas and 280

Table 5 Wastewater discharges from sewered areas and industries in the New York–New Jersey metropolitan region, mid-1960s.

	Flow		
	(m^3/s)	(mgd)	(%)
Municipal sewage (levels of treatment)*			
None	14.0	(320)	2.2
Primary	28.5	(650)	4.6
Secondary	51.8	(1,180)	8.3
	94.3	(2,150)	15.1
Industrial			
Manufacturing*	50.4	(1,150)	8.1
Electrical generation	480	(11,000)	76.8
Approximate totals	625	(14,300)	100

* Data from Tri-State Regional Planning Commission (1969).

liters/day in sewered areas (Tri-State Regional Planning Comm. 1969: p. 33). Interestingly, the inhabitants of Rome in 97 A.D. each used about 140 liters/day (Herschel 1913).

About 85% of the waterborne wastes collected by the sewage systems receive various levels of treatment (Table 5). Pending completion of new sewage treatment plants, much of the sewage from Manhattan and some other areas in New York City was discharged untreated to the Hudson River or New York Harbor. Sewage consists of two major components: organic matter, including bacteria, and inorganic matter, such as soil particles (Table 6). Solids are separated from liquids and both are treated in sewage treatment plants. The levels of sewage treatment commonly used are:

Primary treatment—screens eliminate large objects and small particles settle out, removing about 25–35% of the biochemical oxygen demand (BOD).

Secondary treatment—oxidation of organic matter, removing about 75–90% of the BOD; this is the most common level of treatment in the New York–New Jersey metropolitan region.

Tertiary treatment—supplemental filtration and some chemical treatment removes 98% or more of the BOD and may remove nutrients (phosphates, nitrates), depending on the process; this level of treatment is not much used in the New York–New Jersey metropolitan region.

Of the liquid effluents discharged by the region's sewers, about 85% flows into New York Harbor and the New York Bight apex, within a 30-km radius of The Battery (Fig. 2). On an equivalent population basis, this equals the discharge of untreated sewage from [illegible]1 million persons (about 3 million equivalent from industrial discharges); the equivalent of 7.5 million enters Upper New York Bay and the Hudson River (Tri-State Regional Planning Comm. 1973) through the discharge of sewage treatment plants or sewage overflows and bypasses which handle extralarge flows, usually storm runoff.

Table 6. Composition of rainfall, sewage, and stormwater runoff (in mg/liter) (after Weibel 1969).

	Rainfall	Untreated domestic sewage	Stormwater runoff
Suspended solids	13	200	227
Chemical oxygen demand	16	350	111
Biochemical oxygen demand	—	200	17
Total nitrogen	1.3	40	3.1
Inorganic nitrogen	0.7	30	1
Total phosphate (as P)	0.08	10	0.4

Solids remaining in the treated effluents settle out in the waterways, especially in the deepened navigation channels, and must eventually be removed by dredging. Gross (1974) estimated the contribution of sewage-derived solids to New York Harbor at 8.3×10^{1} tonnes per year. This does not include the organic matter produced in the harbor and offshore waters because of nutrients discharged from sewers and sewage treatment plants.

Solids, called sludges, removed during sewage treatment must also be disposed of. Since land disposal sites are unavailable, these sludges have been barged to sea and dumped in the New York Bight since 1924. Ocean disposal is especially attractive for coastal urban areas because of the convenience, low cost, and relative freedom from complaints.

Sludges are semiliquid slurries with total solid concentrations exceeding 2,500 mg/liter (Burd 1968). Besides sewage sludges, there are industrial sludges produced by SO_2-scrubbing of combustion stack gases and from industrial processes such as steel production (Environ. Sci. Technol. 1975). Volumes of all sludges produced in urban areas are expected to increase rapidly, due to advanced wastewater treatment and air pollution control measures.

Sewage sludge disposal is expensive, typically representing 25–50% of the total capital and operating costs of a wastewater plant (Burd 1968). Sludge disposal is often the most troublesome part of wastewater treatment.

Sludges from New York City wastewater treatment plants average about 50% organic matter; the remainder is composed primarily of aluminosilicates resembling soils

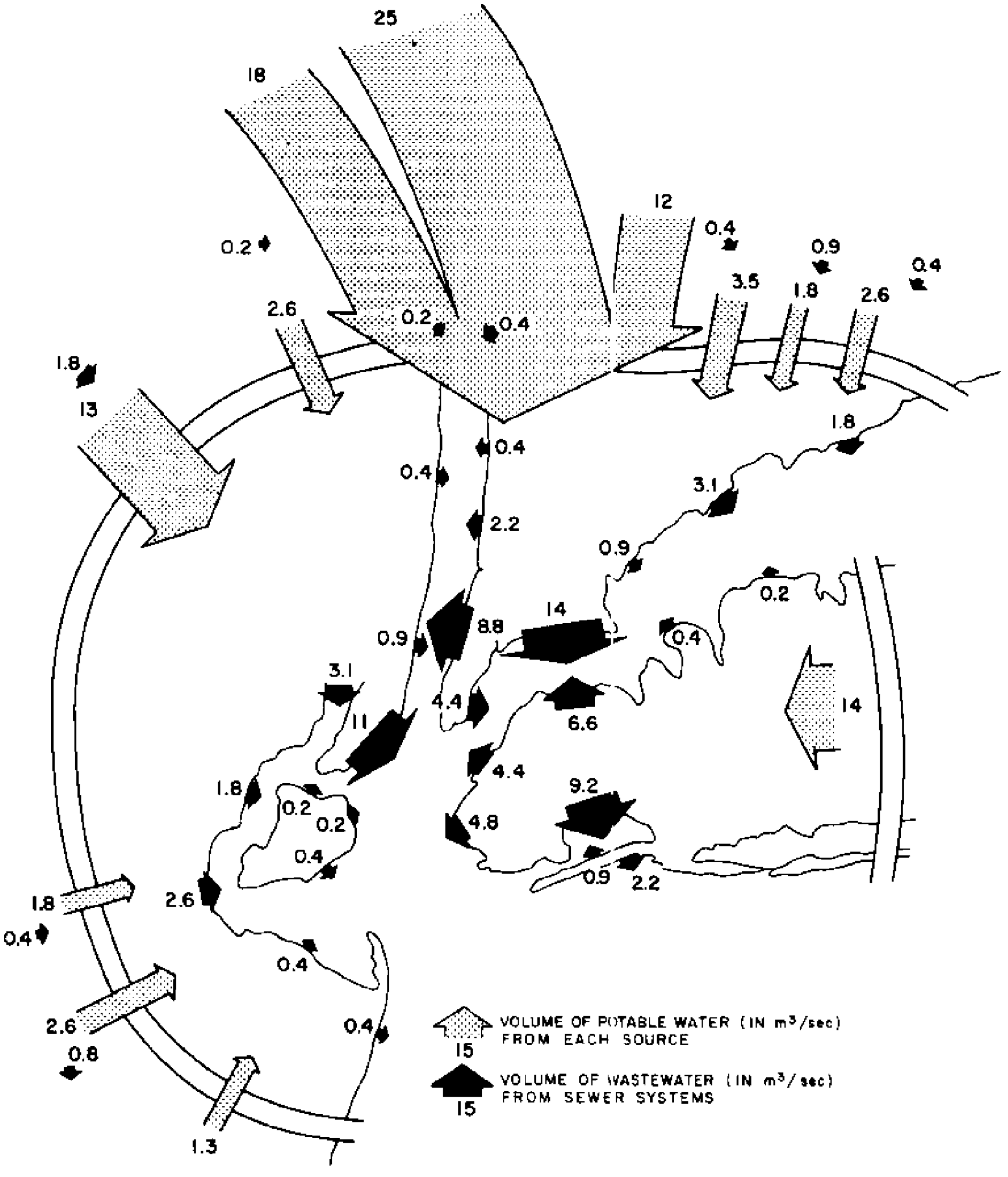

Fig. 2. Schematic representation of the flows of waters and wastewaters in the New York–New Jersey metropolitan region (after Tri-State Planning Comm. 1969).

(Gross 1972). A large fraction of the inorganic component presumably was washed off streets; an unknown fraction comes from storm runoff and from atmospheric fallout (Mueller et al. 1976). Sludges are greatly enriched in certain metals with respect to normal soils, particularly Ag, Cr, Cu, Pb, Sn, and Zn (Gross 1972; Carmody et al. 1973).

Bacterial concentrations are understandably high in sludges. Fecal coliforms have been used to map sludge distributions on the ocean floor near disposal areas (Udell et al. 1974; Verber 1976).

Dried sewage sludges have been successfully used as fertilizers or soil conditioners in small rural towns and have been sold by a few large treatment plants (Burd 1968). Urban areas generally have difficulty marketing sludges because of the large quantities involved, the distant location of markets, odor problems when the sludges are wet, high labor costs for handling, high fuel costs for drying, and possible disease transmission (Burd 1968). Such uses are attractive, however, because they return some nutrients to the soil.

Construction and demolition debris

Cities are constantly rebuilding and generate a great deal of demolition and construction debris in the process. These wastes formerly accumulated in the city itself. The ancient city of Troy, for example, is marked by a mound of debris nearly 15 m high, built up during its occupancy between 3000 and 1300 B.C. (Gunnerson 1973); this amounts to a buildup of 50 to 140 cm/century. Since 100 A.D., London has built up at a rate of 34 cm/century.

The New York–New Jersey metropolitan region has also accumulated debris since its founding in 1625. The most striking evidence of the large volumes involved are the changes caused by the dumping of debris along the shores of the region (Fig. 3).

Most debris that went into the ocean was placed in the cellar dirt disposal site (Gross 1972). From 1964 to 1968, this amounted to 0.54×10^6 m^3/yr or an estimated 0.59 million tonnes per year. The amount varied markedly with construction and with available shoreline disposal sites. For example, large amounts of waste materials were used to construct railroad facilities in Upper New York Bay during the early part of the twentieth century; debris from subway construction was used to fill around Governor's Island between 1900 and 1910 and to build port facilities in Newark Bay during the 1950s and 1960s. Gunnerson (1973) calculated that if the construction and demoli-

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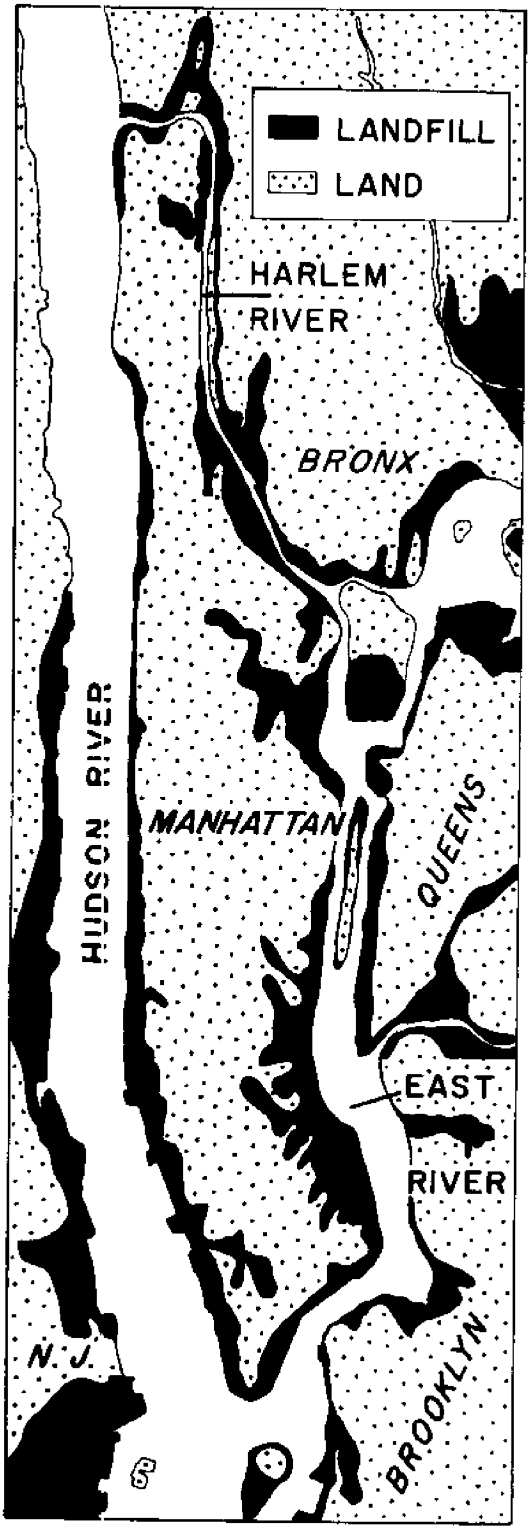

Fig. 3. Landfill areas on Manhattan and in the lower Hudson River area (Bower et al. 1968).

tion debris barged to sea had remained on Manhattan, it would have caused a buildup on the island of about 110 cm/100 yr. He estimated that an additional 30 cm/100 yr would come from ash remaining after incineration of rubbish (27 cm/100 yr) and from solids in sewage sludges (3 cm/100 yr). The total waste deposition would amount to 140 cm/100 yr. His estimate apparently ignores the large volume of coal ash produced between 1850 and 1940 when coal was the primary domestic and industrial fuel (Borchert 1967).

Industrial waste solids

Industries in coastal areas discharge large quantities of wastes and wastewater. Many factories discharge waterborne wastes to municipal sewers and waste solids to municipal landfills. Several industrial operations produce such large volumes of liquid and solid wastes that they operate separate disposal systems, which discharge into estuarine or coastal waters. Among the wastes involved are steel-making slags, coal ash from electric power generation, and waste acids from TiO_2 production.

Steel mills are located on Chesapeake Bay and Delaware Bay because of the relative ease and low costs of transporting the coal, iron ore, and limestone used in steel production. In the various processes, large volumes of waste solids are generated, which, because of their high density, are traditionally disposed of on the plant site to minimize hauling costs.

A large steel mill manufacturing about 6×10^6 tonnes of steel per year produces about 2×10^6 tonnes of slag. Typically 40–80% of the slag produced is recycled at the plant or sold for other uses; the remainder (1.5–6×10^5 m^3/yr) goes into landfill or coastal or estuarine waters. In waters 5 m deep, this requires an area of about 9 ha per year.

Coal was a major fuel in coastal cities between 1850 and 1960 (Borchert 1967). The ash content of typical U.S. coals is between 5 and 15% (Abernathy et al. 1969); that means large volumes of ash and cinder have been produced. An unknown but probably large fraction has been dumped at sea. Of the estimated 3.0×10^8 tons of fly ash produced in the United States between 1945 and 1970, only about 3% was used for any purpose (Capp and Spencer 1970). If coal replaces oil as the major fuel for steam-electric power generation, disposal of coal ash will become an important problem.

Coal ash has been dumped in the New York Bight at the site commonly used for dredge spoil materials (Gross 1972). The amount averaged about 0.1×10^6 tonnes per year between 1960 and 1968, ranging from 0.046 in 1965 to 0.18 in 1963. Because of the substantial reduction in coal use for fuel in the New York metropolitan region and the increased use of ash (Environ. Sci. Technol. 1970), little coal ash was dumped in the bight in the early 1970s. Because of its high sulfur content and air pollution problems, coal is used now primarily for steam-electric power generation where SO_2 in stack gases can be removed by scrubbers.

Nonetheless, large volumes of ash have been dumped in the past and may be locally abundant. A sample of coal ash in the New York area (Table 7) had median grain diameter of about 20 μ and a grain density of 2.2 g/cm^3; typical settling velocities were around 0.015 cm/s (Gross 1972). Chemically, coal ash resembles shales and sandstone (Bowen 1966) except for slight enrichment in Ti, K, Co, and Pb. Coal ash consists of a mixture of quartz and hollow spheres of mullite, a mineral formed during combustion and not found in normal sediment deposits.

Another industrial waste produced in large quantities is from the Ti-pigment process (Peschiera and Freiherr 1968); these wastes are residues of extracted titanium ores suspended in dilute iron-rich sulfuric acid. This material is dumped at sea from special barges at a separate acid wastes disposal site (Redfield and Walford 1951). These wastes have a solid content of 10% by weight of the waste liquid, based on the analysis of a single sample. Except for unusually high concentrations of Fe and Ti (Fig. 4), Ti-pigment wastes resemble shales. No adverse effects on water quality or marine organisms were found to result from disposal of Ti-pigment wastes in the

Table 7. Physical and chemical properties of waste solids, New York metropolitan region (after Gross 1972). All values on a dry-weight basis unless otherwise indicated.

	LOI* (%)	Carbon concentrations (%)			Grain density (g/cm³)	Solid content (g/liter)
		Total	Carbonate	Oxidizable		
Sewage sludges (6)	50.6	29.5	1.64	22.7	1.81	56.2
Duck sludge, Long Island	11.8	6.1	0.31	2.9	2.64	nd†
Fermentation wastes	50.7	21.9	2.51	16.7	1.49	31.9
Coal ash	nd	6.0	0	3.6	2.2	700.0
Sewage sludge ash	nd	1.03	nd	nd	2.71	na‡
Waste alkali	nd	2.6§	nd	1.0§	nd	120.0
Harbor sediments						
Sands	1	0.3	0.1	0.2	2.65	750.0
Silts	11	5.5	0.3	5.2	2.53	500.0

* Loss on ignition.
† Not determined.
‡ Not applicable.
§ Measured on liquid samples.

New York Bight (Redfield and Walford 1951).

Solid wastes

Solid wastes—garbage (food wastes), refuse, and other household wastes—are produced in large volumes. In 1972, the New York–New Jersey metropolitan region produced 16×10^6 tonnes of refuse. About 40% of that was incinerated; the remainder plus incinerator ash went to landfills (Tri-State Regional Planning Comm. 1973). Until 1934, New York City disposed of garbage and street sweepings near the mouth of New York Harbor. Some abandoned disposal sites still have widespread deposits of tin cans and other human artifacts.

New York City transports about half its daily refuse production to Fresh Kills, a 1,200 ha land reclamation site in a marshy area on Staten Island, in use since 1948. Disposal sites in use in 1975 include a site in the Bronx (Pelham Bay) and one in Brooklyn (Fountain Ave.). Sites previously used for solid waste include Flushing Meadow Park (site of the 1964 World's Fair), Orchard Beach on Long Island Sound, and Great Kills Park on Staten Island. Some 11% of the city area has come from using wetlands for landfill disposal operations (TowLine 1971).

Some refuse and incinerator ash falls off barges in transit and refuse accumulates around transfer stations where collecting trucks unload. Only a minor fraction of the refuse produced in the New York region entered the area's waterways in the mid-1970s.

Discussion

Coastal urban areas are indeed major sources and depositional areas for waste solids. As we have seen, data on their physical and chemical characteristics are incomplete. Wherever a large volume of waste must be handled in an urban area, it is a troublesome and expensive activity. It is worthwhile to examine alternate strategies for coping with such disposal problems. Four basic approaches will be discussed for specific wastes:

1. Controlling at source
2. Trapping in route
3. Recycling
4. Rehabilitation of polluted areas

Municipal wastes—refuse, debris, and sewage—seem to provide especially intractable problems. These wastes, produced in large quantities, often cannot be transported for disposal outside the municipality where they were collected. Studies of alternatives for ocean disposal of sludges, begun in the New York–New Jersey metropolitan region in 1974, are expected to be available in mid-1976 (Interstate Sanitation Comm. unpublished data).

Two areas seem to have been neglected in the New York–New Jersey metropolitan region: control of erosion from construction

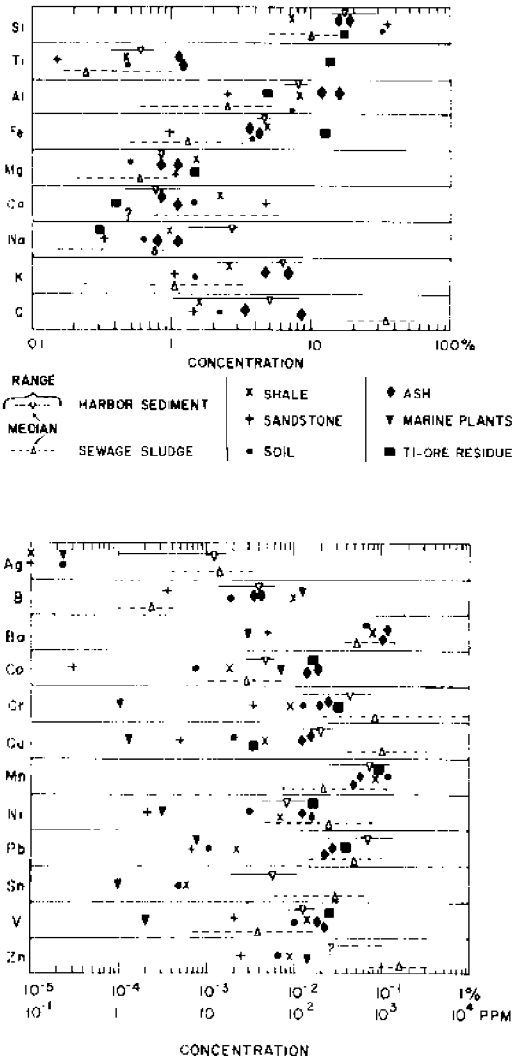

Fig. 4. Chemical composition of waste solids, sediments, and soils. Upper—major element concentrations; lower—minor elements. Concentrations are expressed on a dry-weight basis (after Gross 1972).

sites and reduction of dredging by trapping sediments before they can be deposited in navigation channels. After a heavy rain, it is common to see sediment eroded from bare construction sites, but efforts to control such erosion are rarely in evidence. Rapidly developing municipalities could gain much from the study of sediment control measures in use by local governments in the Washington–Baltimore metropolitan region where erosion is especially severe.

Dredging is a major waste-producing activity in the bight region. In the mid-1970s all these sediments were disposed at sea regardless of their physical characteristics. An effort should be made to classify these wastes and to use them when possible. For instance about half the annual dredging in the region is done in the New York Harbor entrance and involves mostly the removal of relatively coarse-grained sands and some gravels. Perhaps some of this material could be stockpiled for use in construction. These clean wastes could also be used to cover abandoned disposal areas, in keeping with environmental rehabilitation efforts.

Another possibility is to construct sediment traps on river systems to trap river-borne sediment loads before they reach the urban areas. This would require a more detailed understanding of river sediment transport than we now have and more study of the effects of floods on rivers and estuarine systems.

Construction rubble and debris might be used in beneficial ways in the region. In the past, large volumes of wastes have gone into landfills for instance, the Governor's Island enlargement in 1900–1910 and a Hudson River waterfront area recently filled by rock excavated during construction of the World Trade Center. Ocean disposal is necessitated by lack of suitable disposal sites on land at the time the wastes are produced. It may be possible to use such materials beneficially and thereby reduce the amount to be dumped at sea. One way is to designate areas for rehabilitation where the rubble and debris could be used to bury old waste deposits. This would provide a hard irregular substrate that might attract fish and lobster. Such wastes could also be used to build artificial reefs that would attract pelagic fish for sport fishing. This use would require careful planning to optimize use of the site for fishing (Stone 1972).

Although no deleterious effects of waste acid disposal in the New York Bight have been documented (Redfield and Walford 1951), alternative disposal strategies should be developed in the event that ocean disposal is eliminated for political or other reasons. One way is to develop different

TiO_2 manufacturing processes where the waste acid production is eliminated. The net effect of such a disposal strategy may be, however, to substitute a solid disposal problem for a liquid disposal problem.

Another approach is to combine waste disposal operations, such as combining the disposal of waste acids and municipal sewage treatment plant sludges in the same disposal area or even from the same barges. Flocculation of the iron-rich materials and their subsequent removal from the water column might facilitate the removal of organic matter put in the water column by the disposal of sewage sludges. Certainly more attention should be given to developing techniques to reduce or eliminate the environmental impact of waste disposal through combinations of waste disposal operations.

Successful management of urban waste disposal problems requires an understanding of waste sources and of their transport and deposition in estuarine and coastal waters. It also requires an imaginative search for alternative strategies for waste management.

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Contaminants entering the New York Bight: Sources, mass loads, significance[1]

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Abstract

Major contaminant inputs to the New York Bight originate from the New York metropolitan area and the Hudson River drainage basin, principally from wastewater, runoff, and barged discharges.

Major sources of the microbial load are unchlorinated municipal wastewater discharges and urban runoff. Sewage sludge barge dumps constitute an insignificant microbial load on bight waters.

The seasonal variability of municipal wastewater and gauged runoff and annual variability of barge discharges range from a maximum of 1.5–2.0 to a minimum of 0.5–0.7 times the average mass loads. Between 1960 and 1975 the sludge volume as well as fraction digested shows a definite increasing trend. Few data are available on atmospheric and urban runoff inputs.

The mass loads reaching the bight are highly related. Poor quality of the dredged materials is caused by contaminants settling from poorly treated wastewater and urban runoff. Increasing levels of wastewater treatment produce more municipal and industrial sludge for disposal. For conservative substances, such as heavy metals, various control measures may redistribute the load among the sources but cause no net decrease in the total.

The New York Bight presently receives a major portion of the wastes generated in the Hudson River, southern Long Island, and eastern New Jersey drainage basins, including the heavily populated New York metropolitan area (Fig. 1).

Although much information on the contaminants discharged into these coastal waters existed, it was not compiled or properly evaluated. Here we summarize data on contaminant discharges entering the New York Bight. Twenty-two parameters were chosen to represent water quality. The information includes location, frequency, magnitude, and type of contaminant. Data gaps were identified to help guide future monitoring and surveillance programs. Assessing the significance of the information collected was a prime objective of the project. Detailed information for each specific area, methods used, as well as raw data were reported by Mueller et al. (1975).

Geographical zones

To obtain estimates of the mass loads into the bight, we grouped contaminant inputs into four geographical zones (Fig. 1). One zone is the bight itself, which receives discharges from barge dumping and from atmospheric fallout. The three coastal zones discharge wastewater and runoff to receiving waters that ultimately flow into the bight. Through the Sandy Hook–Rockaway Point transect drains a substantial portion of the runoff from upper New York State and the New York City metropolitan area. The New Jersey and southern Long Island zones are less densely populated than the metropolitan area and have significantly smaller drainage areas than the transect zone.

The population and drainage area of each zone are given in Table 1.

Sources

Four contaminant inputs were identified: 1—barge dumps, 2—atmospheric fallout, 3—runoff, and 4—wastewater. Wastes from the first two sources discharge directly to the bight; wastes from the last two originate in the coastal zones. Only part of the contaminants from the last two sources reaches the bight because of physical, chemical, and biological processes that remove or retain the wastes in the inland waterways and

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[1] This project was supported by NOAA grant 04-4-002-35 for the Marine EcoSystems Analysis (MESA) New York Bight Project.

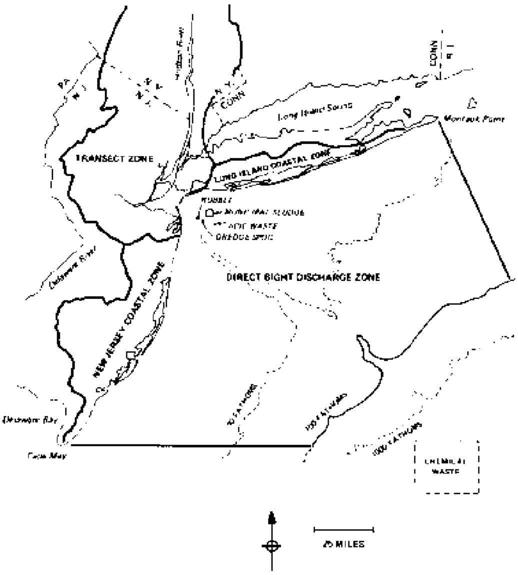


Fig. 1. Geographical zones.

coastal estuaries. Detailed procedures for estimating these waste discharges are discussed elsewhere (Mueller et al. 1975).

Distribution of total inputs

The major inputs come from direct bight and transect zone discharges (Table 2). Groundwater mass loads (Table 3) are negligible for all parameters; direct industrial discharges are relatively small.

To illustrate the contribution of each zone and each source to the total mass load entering the bight, we prepared pie charts of flow (Fig. 2) and six parameters of water quality by location and source (Figs. 3–6). Parameters included suspended solids, because of their effect on light penetration and sediment deposition, organic carbon as a typical oxygen-demanding substance, nitrogen and phosphorus as typical nutrients for biological growth, lead as typical of industrial metal, and fecal coliforms as microbial indicators of contamination.

Cadmium differs from lead in having a heavier barge load contribution, and mercury differs by having a significantly greater wastewater contribution. The latter estimate may be higher than actual because of possible mercury contamination during sample preservation of the New York City wastewater samples. Most of the flow input is from rainfall; gauged runoff from the transect zone contributes most of the remainder

Table 1. Coastal zone characteristics.*

Zone	Population 1970	Drainage area (mi²)	Population density cap/mi² Zone avg	Population density cap/mi² Range by county
Transect	15,500,000	16,300	950	3–67,000
New Jersey coastal	875,000	2,000	440	220– 1,700
Long Island coastal	1,440,000	670	2,150	1,210– 4,500
Total	17,815,000	18,970		

* Mueller et al. 1975.

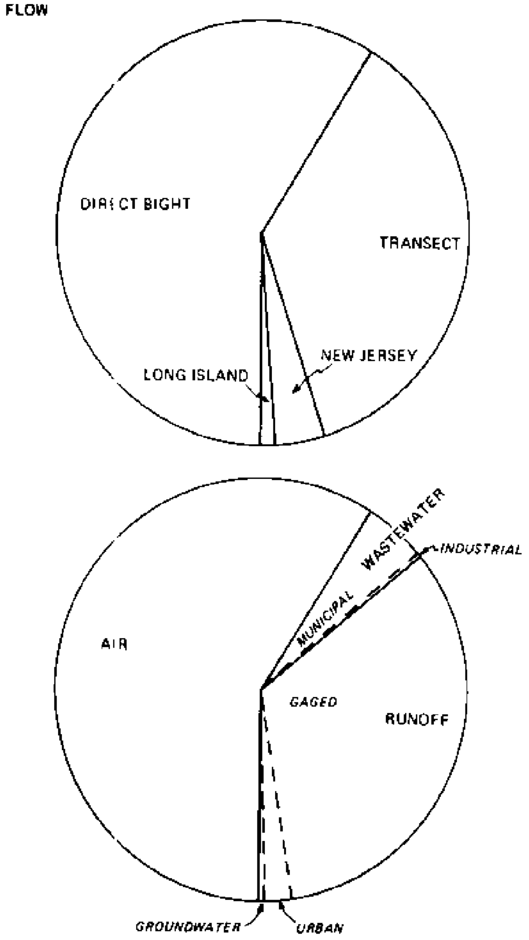


Fig. 2. Distribution of flow inputs by location and source

Table 2. Total mass loads into the bight.

	Mass load (tonnes/d)	Percentage contribution by location			
			Coastal zone		
		Direct bight	Transect	New Jersey	Long Island
Flow, cfs (m^3/s)	82,700 (2,340)	59	36	4	1
SS	24,000	68	31	0.[illegible]	0.1
Alk	5,100	1	96	2	1
BOD_5	2,100	30	67	3	0.5
COD	10,000	42	48	9	0.5
TOC	2,600	37	58	4	0.6
MBAS	59	—	95	4	0.5
O & G	870	38	53	9	0.6
NH_3-N	210	28	67	3	2
Org-N	190	27	68	3	2
TKN	400	27	68	3	2
NO_2+NO_3-N	120	33	55	10	2
Total-N	520	29	65	4	2
Ortho-P	51	1	91	3	5
Total-P	138	51	45	2	2
Cd	2.4	84	15	0.5	0.07
Cr	5.0	51	44	0.6	4
Cu	13.8	54	45	0.9	0.2
Fe	230	82	16	2	0.2
Hg	0.30	9	85	6	0.6
Pb	12.7	53	46	0.5	0.2
Zn	33	47	52	1	0.3
F. coli* winter	5.6×10^7	<0.01	100	0.2	<0.001
summer	4.9×10^7	<0.01	100	0.2	<0.001
T. coli* winter	21×10^7	<0.01	100	0.1	<0.001
summer	11×10^7	<0.01	100	0.2	<0.001

* Coliform load [=] 10^{10} org/d.

(Fig. 2). Only about 5% of the flow to the bight comes from the Long Island and New Jersey coastal zones.

The contaminant contribution from the New Jersey and Long Island coastal zones (Fig. 3) is small, generally less than 6%. Most of the suspended solids load is from the direct bight discharges. The majority of the organic carbon and nitrogen loads and nearly all the fecal coliform loads come from the transect zone.

Over 50% of the suspended solids input comes from dredge spoils; stream and urban runoff are next in importance (Fig. 4). Organic carbon comes from all sources, but wastewater and runoff dominate. Barge dumps, mostly of dredge spoils, contribute a quarter of the organic carbon load.

The distribution of the nitrogen and phosphorus nutrient loads differs significantly (Fig. 5). Wastewater is a major contributor of both nitrogen and phosphorus, but barged waste disposal contributes 50% of the phosphorus and only about 15% of the nitrogen; dredge spoils are again the major portion of the barge dumps. Atmospheric fallout is significant for nitrogen but negligible for phosphorus; gauged runoff nutrient loads are at least twice as high as urban runoff.

Lead comes from many sources, but dredge spoils contribute the largest fraction (Fig. 6). Urban runoff contributes a significant portion because of lead emissions from vehicles. The fecal coliform load (Fig. 6) comes primarily from unchlorinated raw sewage discharges (New York City) and from the combined sewer overflows (New York City and New Jersey).

Variability and reliability

To estimate the variability of wastewater mass loads, we calculated seasonal scale factors for 1972 influent and effluent loads for two New York City municipal treatment plants. Figure 7 presents the results

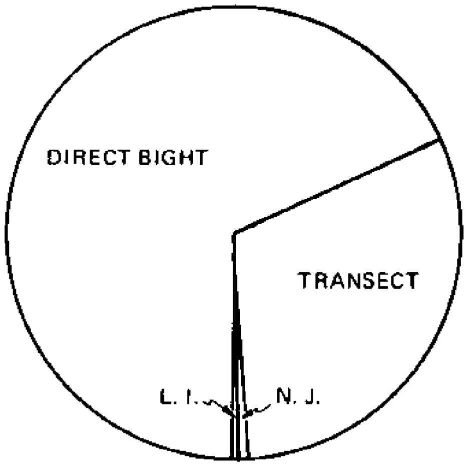
Table 3. Percent of loads by source.

	Direct bight		Coastal zone				
			Wastewater		Runoff		
	Barge	Atmospheric	Municipal	Industrial	Gauged	Urban	Groundwater
Flow	0.02	59	5	0.4	33	2	0.4
SS	63	5	4	0.2	16	12	0
Alk	1	0	35	0.3	59	5	0.03
BOD_5	21	9	48	2	11	9	0.01
COD	32	10	35	1	13	9	0.01
TOC	25	12	29	1	18	15	0.02
MBAS	—	—	86	—	5	9	0.05
O & G	38	—	22	0.7	16	23	—
NH_3-N	24	4	55	3	10	4	0.04
Org-N	19	9	45	2	21	5	0.02
TKN	21	6	51	2	15	5	0.02
NO_2+NO_3-N	0.07	33	6	0.3	60	0.6	0.7
Total-N	16	13	40	2	25	4	0.2
Ortho-P	—	1	72	—	18	9	0
Total-P	50	0.7	35	1	9	4	0
Cd	82	2	5	0.6	5	5	0.001
Cr	50	1	22	0.8	10	16	0
Cu	51	3	11	9	10	16	0.006
Fe	79	3	5	0.5	6	6	0.01
Hg	9	—	71	2	13	5	—
Pb	44	9	19	3	6	19	0.004
Zn	29	18	8	2	21	22	0.009
F. coli winter	<0.01	0	87	0.2	0.01	13	0
summer	<0.01	0	85	0.2	0.01	15	0
T. coli winter	<0.01	0	91	0.1	0.05	9	0
summer	<0.01	0	84	0.2	0.1	16	0

from the Jamaica plant, which is a 93 MGD (4.1 m^3/s) step aeration-activated sludge plant obtaining an average of 89% BOD_5 removal. The plant has insignificant salt water infiltration and some combined sewer systems with stormwater bypass through regulators. The BOD_5 and suspended solids seasonal scale factors are based on daily composite samples and therefore should yield a reliable estimate of seasonal variability. Although seasonal variation of rainfall was significant during 1972, an extremely wet year, the influent sewage flow at Jamaica was relatively constant. The plant is at design capacity under dry weather flows, and storm flow is diverted through regulators directly to receiving waters. Significant seasonal variability of both effluent BOD_5 and suspended solids occurs, apparently not correlated to influent load but to temperature. This is due mainly to better settling properties at high temperatures and low water viscosity and to greater biological activity producing somewhat better removal of soluble BOD.

Both Raritan River and Hudson River gauged runoff data were analyzed for seasonal variability. In the Raritan River (Fig. 8), nitrate-nitrogen, orthophosphate, BOD_5, and alkalinity mass loads are directly correlated with flow and inversely with temperature, indicating that urban and rural runoff from this area may be a major source of contaminants. Increased degradation at warm temperatures may be significant. The coliform (geometric mean) seasonal loads, however, vary inversely with flow and directly with temperature. This may be due to either the bactericidal effects of cold water or the stimulatory effects of warm water in promoting growth.

The marked variability in annual volumes of barge discharges (Fig. 9) from 1960 to 1974 is affected by federal regulations, varying degrees of sewage treatment, dredging schedules, and the amount of construction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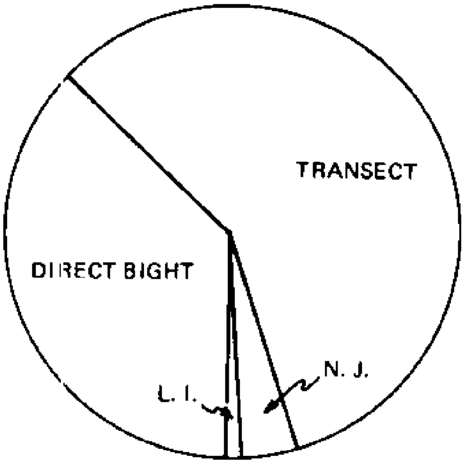
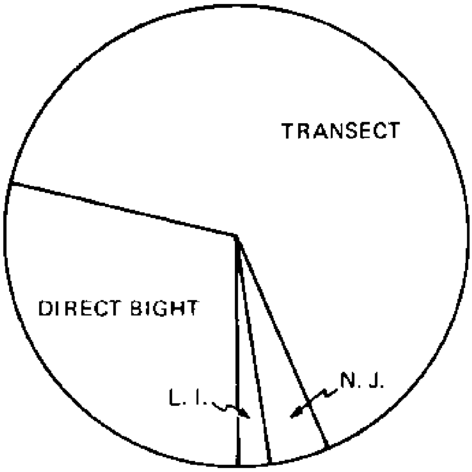
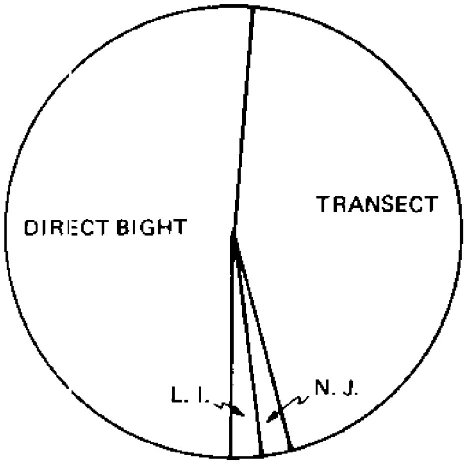

ORGANIC CARBON


NITROGEN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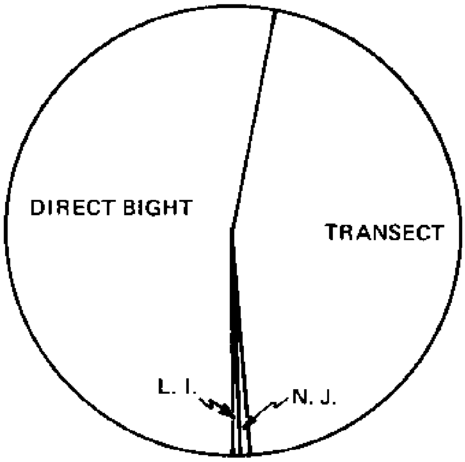
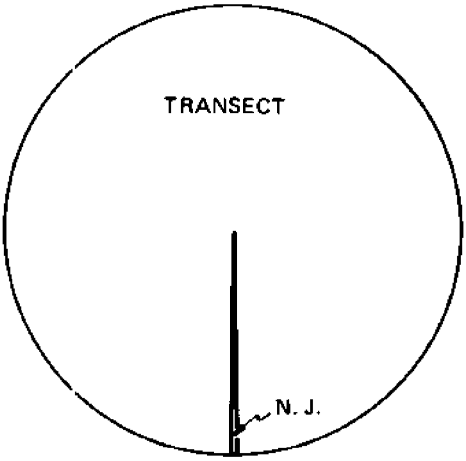

LEAD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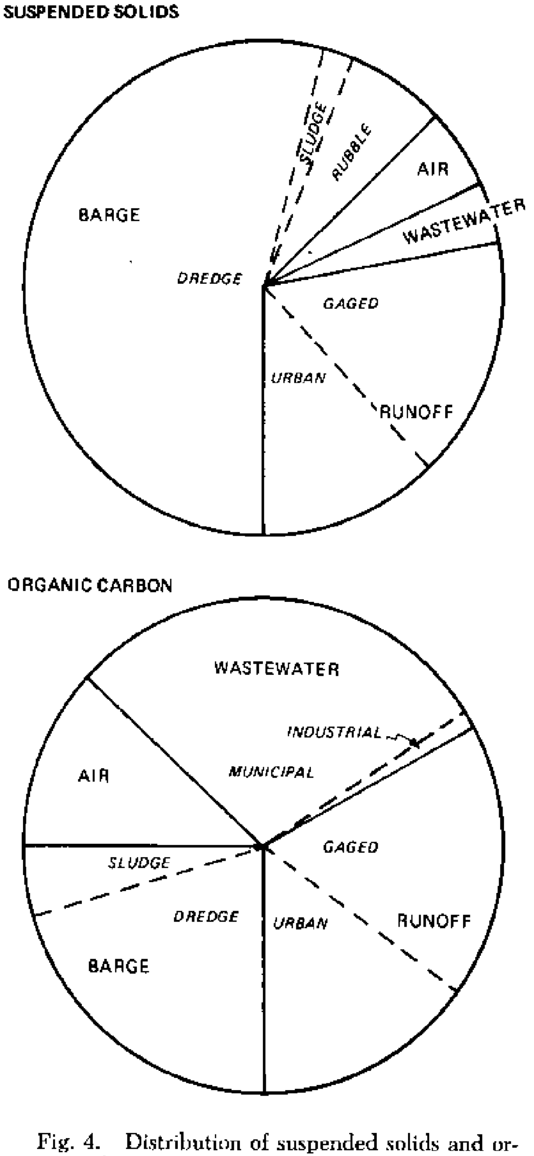

Fig. 4. Distribution of suspended solids and organic carbon loads.

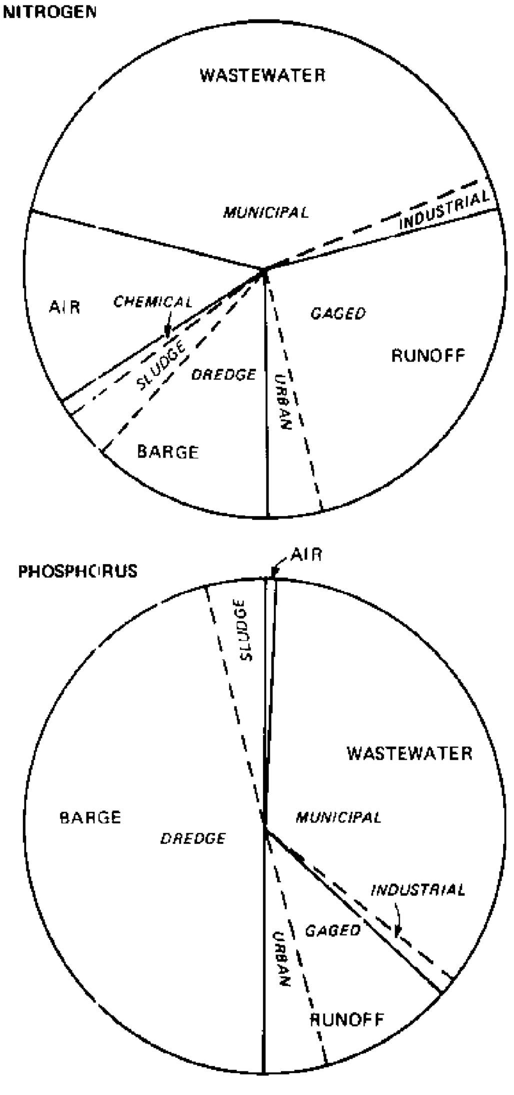

Fig. 5. Distribution of nitrogen and phosphorus loads.

activity. The increasing fraction of digested sludge dumped into the bight is shown in Fig. 10.

The variability of the mass loads discharged into the bight is typically large (Table 4). The variability in the annual barge dump mass loads is due to the variation in the volume dumped over the past few years when a significant temporal trend was obvious and on the total 15-year period of record when no temporal trend was present. The yearly quantity of dredge spoils dumped into the bight, the most significant barge input, has the highest variability. Sewage sludge, although less variable, shows a definite increasing trend with time.

The seasonal mass loads for both wastewater and gauged runoff generally have minimum values (0.5 to 0.7 times the average) in summer due to low flow or high

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Fig. 3. Distribution of loads by location.

temperature. An exception is the Raritan River data where the stream coliform values peak in the summer because of high temperatures. The maximum seasonal load for both sources was about 1.5 times the average. No estimate of the variability of the urban and groundwater runoffs and atmo-

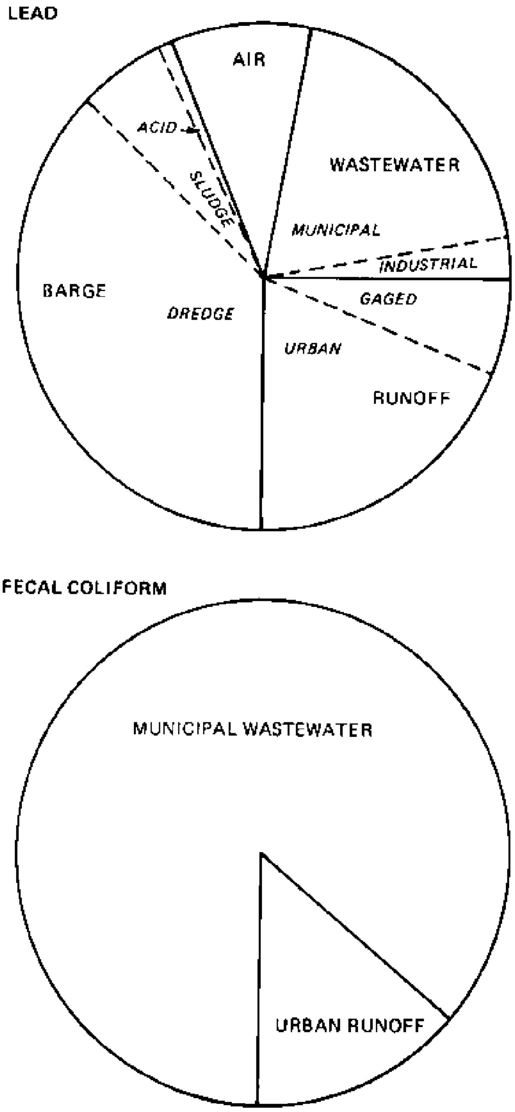

Fig. 6. Distribution of lead and fecal coliform loads.

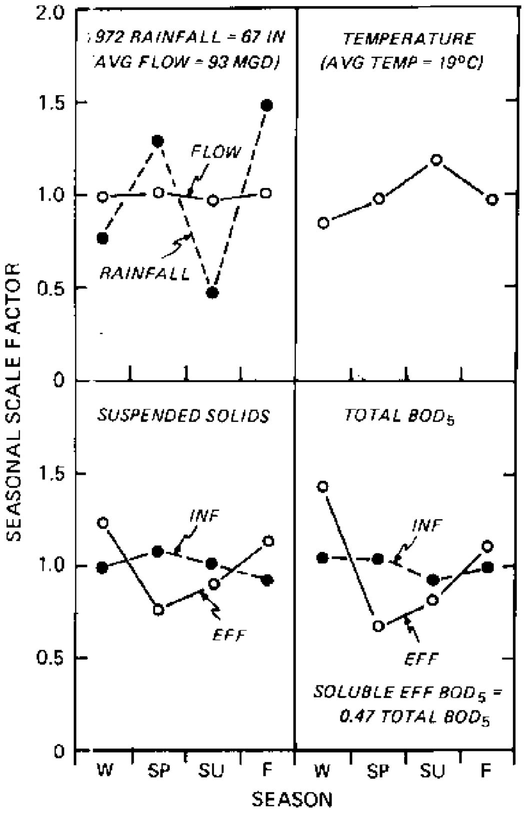

Fig. 7. Seasonal variability of Jamaica Treatment Plant BOD$_5$ and suspended solids, 1972 mass loads.

Table 4. Estimate of variability of mass loads.

Source	Type	Years considered	Variability Max/Avg	Min/Avg	Max/Min
Barge dumps					
Dredge spoils	Annual	1960–1974	2.0	0.7	2.5
Sewage sludge	Annual	1970–1974	1.2	0.8	1.5
Acid wastes	Annual	1965–1974	1.1	0.8	1.4
Chemical wastes	Annual	1972–1974	1.3	0.8	1.6
Rubble	Annual	1960–1974	1.4	0.4	3.3
Municipal wastewater	Seasonal	1972	1.4	0.7	2.0
Gauged runoff	Seasonal	1969–1972	1.5	0.5	3.0

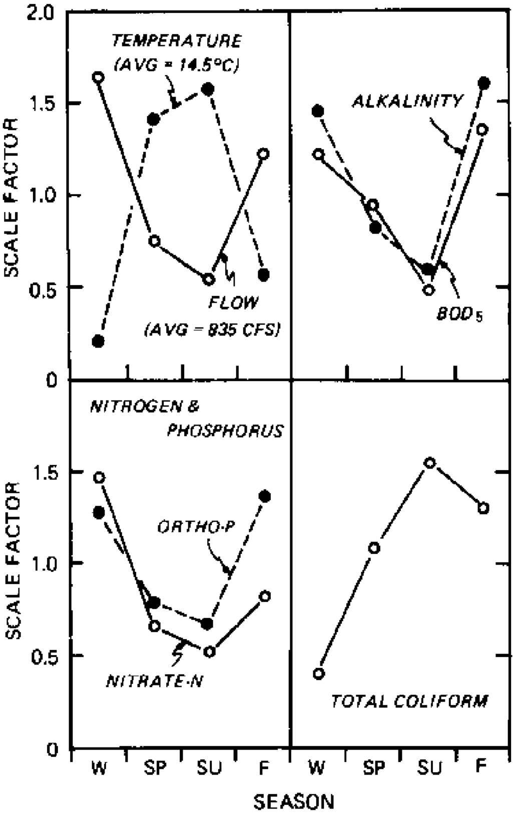

Fig. 8. Seasonal variability for the Raritan River, November 1968–September 1972.

spheric inputs could be made for the New York Bight.

The most reliable estimates of the average mass loads from each source are the wastewater and gauged runoff mass loads. The least reliable values for these sources are the coliform and detergent loads for wastewater and oil and grease loads for gauged runoff.

The barge dumps and industrial waste loads are considered next in reliability, since they were based mainly on a Corps of Engineers survey and EPA permit files. The least reliable data are the atmospheric inputs, urban runoff, and groundwater outflow; these are difficult to measure and data for the bight area are scarce.

Significance

To estimate the significance of the contaminant sources in the New York Bight,

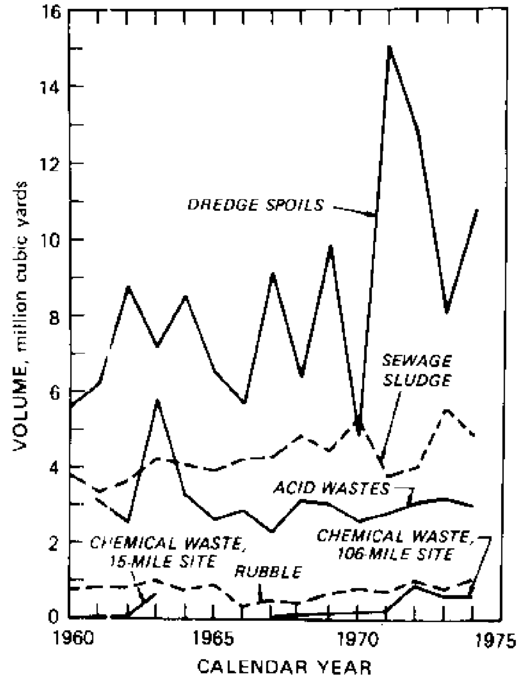

Fig. 9. Historical volumes of barge discharges.

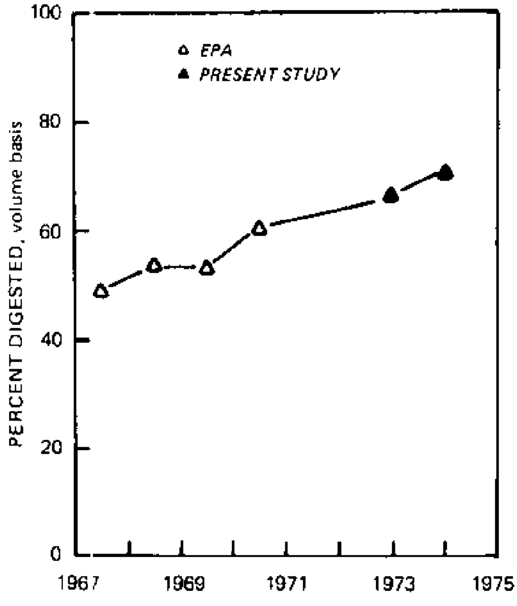

Fig. 10. Digested fraction of sewage sludge dumped into the bight. (EPA data from EPA 1973.)

relationships among them must be considered. Load distributions discussed earlier are based on 1970–1974 conditions. One obvious trend is the upgrading of wastewater treatment facilities and reduction of raw sewage discharges. Thus the wastewater mass load will be generally decreased. Increased treatment should improve the quality of New York Harbor waters and sediments and reduce dredge spoil mass loads. But a significant increase in sewage sludge mass loads will also increase conservative substances such as heavy metals. Evaluation of alternative management policies for the bight inputs must consider the effects of specific control measures with respect to all sources.

Another consideration is the distribution of the various waste loads between overlying waters and the bottom. The fraction of the load that ultimately reaches the bight from coastal zones may be significantly modified due to bacterial decay, sedimentation, leaching, and biological growth cycles.

The fraction of the barge dump mass loads reaching the overlying waters is unknown. For the dredge spoils, comprising the major barge dump load, this fraction may be small due to the density of the material. Swanson (unpublished) recently reported that significant bottom deposits have formed in the dredge spoil area. No long term leaching studies of these bottom deposits have been conducted.

To roughly assess the relative significance of each of the parameters considered in this study, we estimated the background loads entering the bight area from the oceanic boundaries. For the boundary inputs we assumed that a net ocean current of 6 to 7 cm/s existed parallel to the bight coastline, entering at the northeast boundary and exiting at the southern boundary (R. Charnell personal communication). No net flow was assumed to occur across the southeast boundary, the 100-fathom bottom contour. The cross-section areas of the northeast boundary (Montauk Point to the 100-fathom depth) and the southern boundary (Cape May to the 100-fathom depth) were estimated at 2.8 and 2.5 nmi^2 (9.6 and 8.6 km^2), respectively. An average cross-section area of 2.63 nmi^2 (9.02 km^2) with an average net velocity of 6.5 cm/s results in a net flow across the boundaries of 2.07×10^7 cfs (5.9×10^5 m^3/s).

Table 5. Comparison of total mass loads to background loads.

	Typical seawater composition* (mg/liter)	Background mass load (tonnes/d)	Total bight† mass loads : background loads (%)
Flow, cfs (m^3/s)	2.07×10^7	(5.9×10^5)	0.40
SS	3	150,000	16
Alk	130‡	6,600,000	0.077
TOC	1‡	51,000	5.1
Org-N	0.2‡	10,000	1.9
Total-N	0.5	25,000	2.1
Total-P	0.07	3,600	3.8
Cd	0.0001	5.1	47
Cr	0.00005	2.5	200
Cu	0.003	150	9.2
Fe	0.01	510	45
Hg	0.00003	1.5	20
Pb	0.00003	1.5	850
Zn	0.01	510	6.5

* Goldberg 1963.

† Total bight mass loads from Table 2.

‡ Riley 1965. Alkalinity based on 32‰ salinity; TOC and Org-N both soluble mean annual values in the North Sea for 1958–1959.

Using a typical seawater composition and the above flow, we computed the background mass loads given in Table 5. A comparison of these loads to those measured in this study indicates that the most significant manmade inputs are metals: 200% and 850% greater than the background inputs for chromium and lead. Even though the accuracy of the net flow across the boundaries is unknown, the ratios in Table 5 indicate the relative importance of the various parameters.

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Trace metals in the New York Bight

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Abstract

Large quantities of trace metals are introduced to the New York Bight apex from many sources. Distributions of dissolved Mn, Fe, Cd, Cu, and Zn are extremely nonuniform in the waters of the apex due to the many sources and complex reactions taking place. Estuarine discharge and dredge spoil dumping are major sources, while sewage sludge and acid waste dumping are minor sources for most elements studied. Much of the dissolved Cu and Fe occurs in a chemical form that is not extractable by chelation/solvent extraction, even after acidification. The quantity of this metal fraction increases with distance from the Hudson-Raritan estuary. Loss from solution of some elements, notably Mn, occurs when estuarine water mixes with oceanic water. Metals, particularly Zn, are released to solution during ocean dumping of sewage sludge and other materials. Concentrations of dissolved metals in the apex are higher than on the open shelf and higher in summer than in spring and fall. This suggests that the apex flushes slower in summer, as inputs do not vary significantly with season.

Budget calculations show that contaminant metals, exemplified by Cu and Zn, do not accumulate in the apex but are rapidly removed either to the estuaries or the surrounding shelf waters. Mean residence times of contaminant metals in the apex waters are less than 6 months, perhaps considerably less.

Many activities in the New York metropolitan region produce significant quantities of waste products containing trace metals at concentrations above those normally found in materials introduced to the environment naturally. Atmospheric dust and rainfall, freshwater runoff from land, urban and industrial liquid effluents, and solid wastes dumped into the ocean all provide unnaturally high levels of trace metals to the ocean in the New York Bight. These inputs are concentrated in the apex region of the bight (Fig. 1) where most of the river input takes place, where almost all of the solid waste dumping is concentrated (Mueller et al. 1975), and where atmospheric dust loads apparently are highest (Duce et al. 1976).

Perhaps the first realization of the magnitude of problems associated with metal discharges to the ocean came with the diagnosis of organic mercury as the cause of Minimata disease (Irukayama et al. 1961). The first studies of trace metal contamination of the New York Bight were reported in 1970 (Gross 1970*a*,*b*). Much remains to be learned about the transport, reactions, and ultimate fate of metals in the bight. Here we describe the results of some of the early stages of an investigation of trace metal cycles in the apex.

Methods

The availability of data concerning the concentrations of trace metals in New York Bight sediments and some organisms (Gross et al. 1971; NMFS 1972; Carmody et al. 1973) caused us to direct our initial sampling and analysis toward trace metals in dissolved and suspended particulate states.

Distributions of trace metals and the variability of these distributions with season were investigated between April 1974 and March 1975. On each cruise, continuous vertical profiles of salinity, temperature, and depth were obtained with an Inter Ocean model 513-10 CSTD. The CSTD was interfaced with a General Oceanics model 1015 rosette multibottle array equipped with

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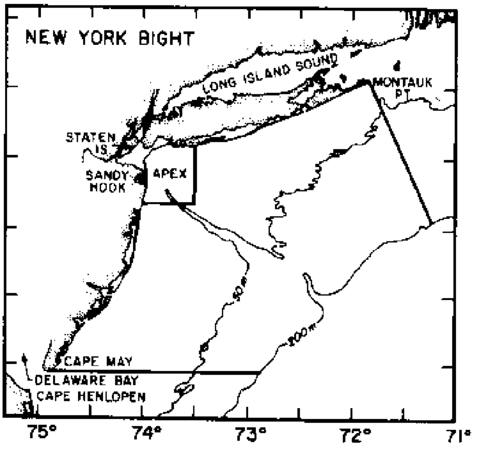

Fig. 1. New York Bight, showing apex area.

General Oceanics model 1070 10-liter top-drop Niskin bottles. Water samples were collected at the surface (1 m), at 10-m intervals, and at about 2 m above the sediment–water interface. Each water sample has been analyzed for salinity, dissolved oxygen, pH, nitrate, nitrite, phosphate, silicate, total suspended load, particle size distribution, suspended total organic carbon, suspended total carbohydrates and total proteins, and the dissolved trace metals Fe, Mn, Cd, Cu, and Zn (Cantillo et al. 1976). Some suspended sediment samples are currently being analyzed for mineralogy and inorganic composition (Betzer personal communication).

Seven cruises were conducted at 36-day intervals between April and November 1974 and occupied 25 or 26 stations in the bight apex (Fig. 2). A single additional cruise in February–March 1975 occupied a series of 64 stations (Fig. 3).

Samples for trace metal analyses were drawn from the Niskin sampler and filtered through a 0.4-μ Nuclepore membrane filter, with great care being taken to minimize atmospheric contamination. Filtered samples were collected in precleaned 1-liter linear polyethylene bottles and acidified immediately with 1 ml/liter of silica redistilled concentrated nitric acid. Samples from the first cruise (16–20 April 1974) were not filtered.

Analyses for total Fe, Mn, Cu, Cd, and Zn were performed with a Perkin Elmer 503 atomic absorption spectrophotometer by direct injection of the sample into a Perkin Elmer HGA 2100 heated graphite atomizer (Segar and Cantillo 1976). Zn was determined only on cruises after July 1974. The

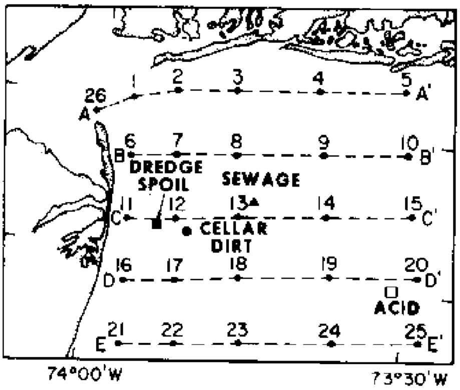

Fig. 2. Station locations in the apex. Designated dumpsite locations and transect designation (A′–A′, B′–B′, etc.) for which vertical sections are plotted in Figs. 4–7 and 12–28.

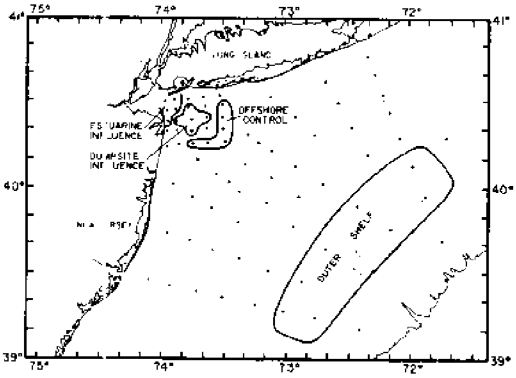

Fig. 3. Extended sampling area occupied 26 February–3 March 1975, showing station locations and regimes: estuarine and dumpsite influenced, offshore control, and outer shelf for which average concentrations are calculated and shown in Figs. 30–34.

analytical precision determined from multiple aliquots from the same sample storage bottle was better than ±10% for concentrations of metals in excess of 10 times the instrumental detection limit (ca. 0.3 ppb for Fe, Mn, and Cu, and 0.01 ppb for Cd and Zn). Generally only single samples were taken from each Niskin sampler. Random duplicates were obtained on several cruises and analyzed. Overall precision of the sampling and analysis was better than ±15% for concentrations in excess of ten times the instrumental detection limit (Segar and Cantillo 1975).

Results and discussion

April 1974 data (unfiltered samples)—Analyses of samples from the first cruise (16–20 April 1974) were performed without filtering. Samples were acidified and allowed to stand for several weeks and the subsample for analysis was taken carefully from the bottle without shaking. Our analytical technique does not discriminate between chemical or physical forms of the element (Segar and Cantillo 1975), so these analyses include all of the metal whether dissolved or remaining in suspension. The

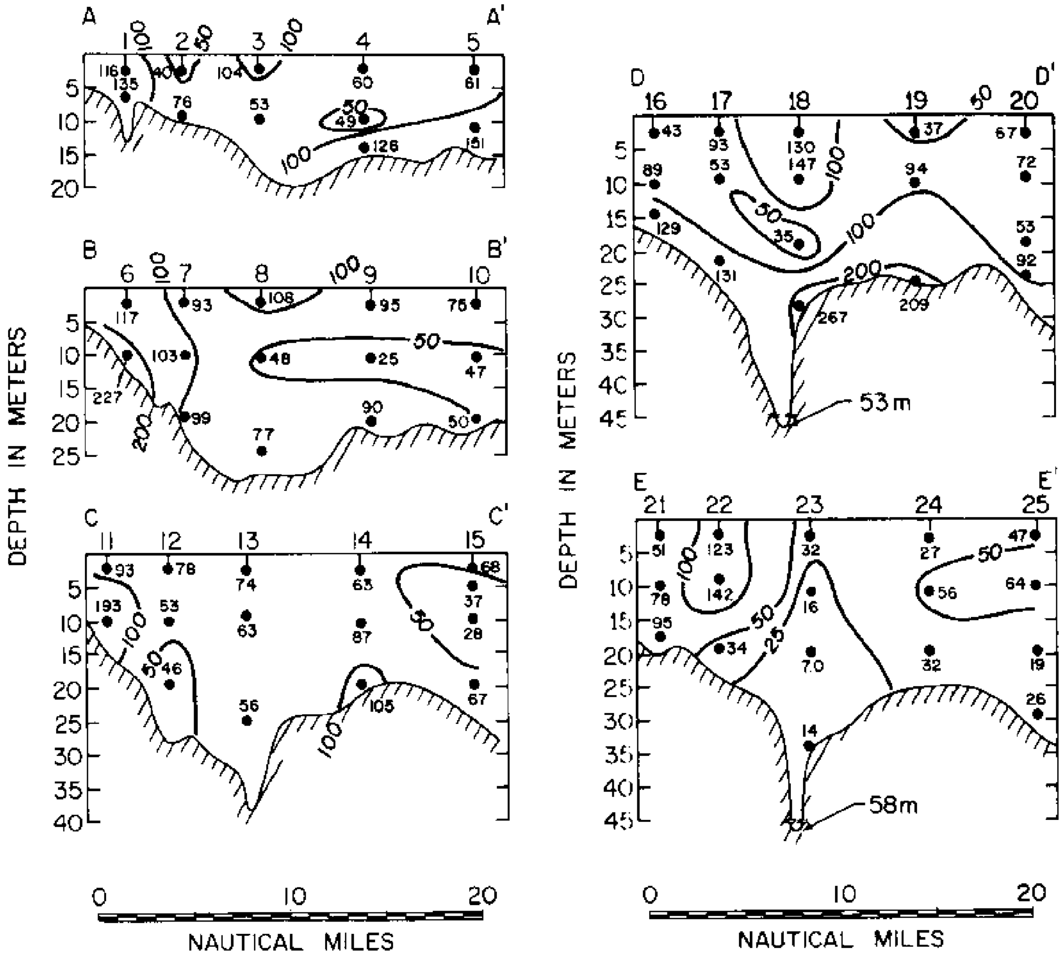

Fig. 4. Total dissolved and particulate iron (μg/liter) in the apex, 16–20 April 1974.

analysis includes the dissolved and weak acid-soluble metal, but excludes metal associated with coarse mineral fractions. The distributions of total particulate plus dissolved Fe, Mn, Cu, and Cd for the April 1974 cruise are shown in Figs. 4–7. There is considerable variation in metal concentration between stations and with depth. Although our sampling density is inadequate to describe this variability, certain features are clear. The lower salinity surface water discharging from Lower New York Bay (Fig. 8) contains high concentrations of both Mn and Fe, but Cd and Cu concentrations do not seem to be strongly influenced by the estuarine discharge. Fe, but not Mn, is also found in high concentrations in water near the sediment–water interface over much of the apex. The absence of this layer in the rest of the apex may be a sampling artifact since samples often could not be taken near the sediment–water interface,

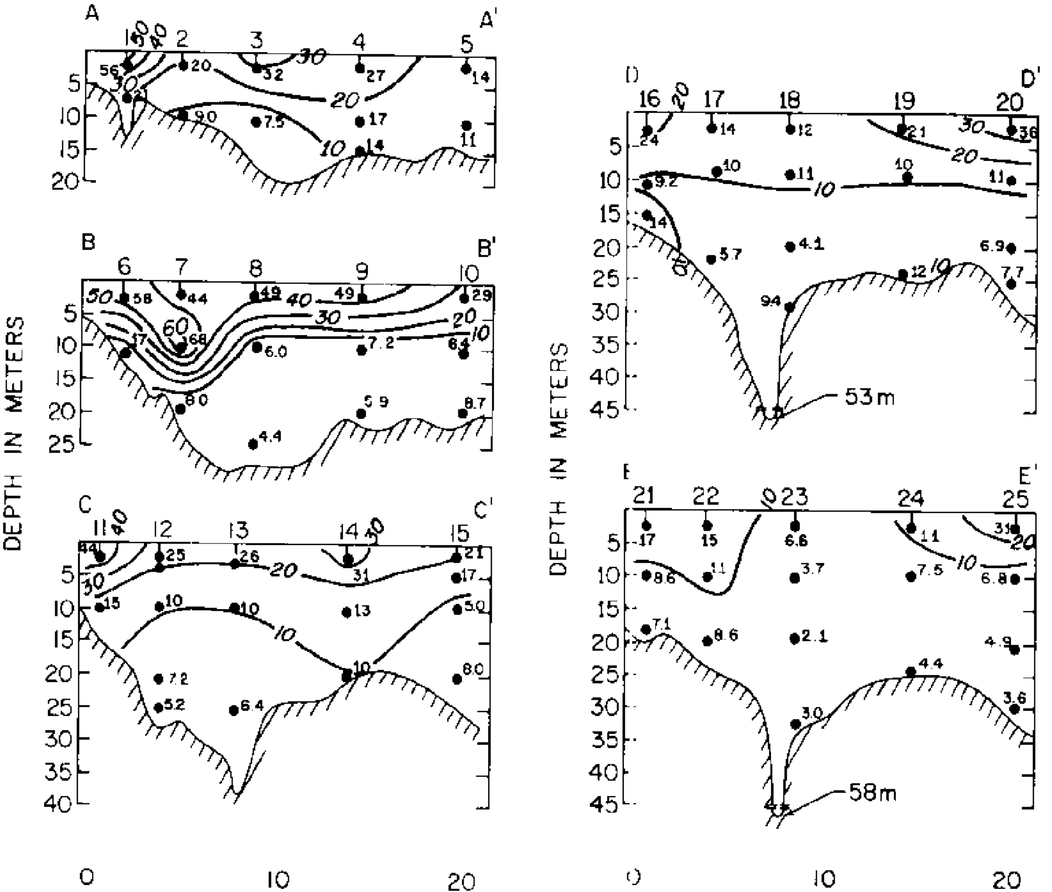

Fig. 5. Total dissolved and particulate manganese (μg/liter) in the apex, 16–20 April 1974.

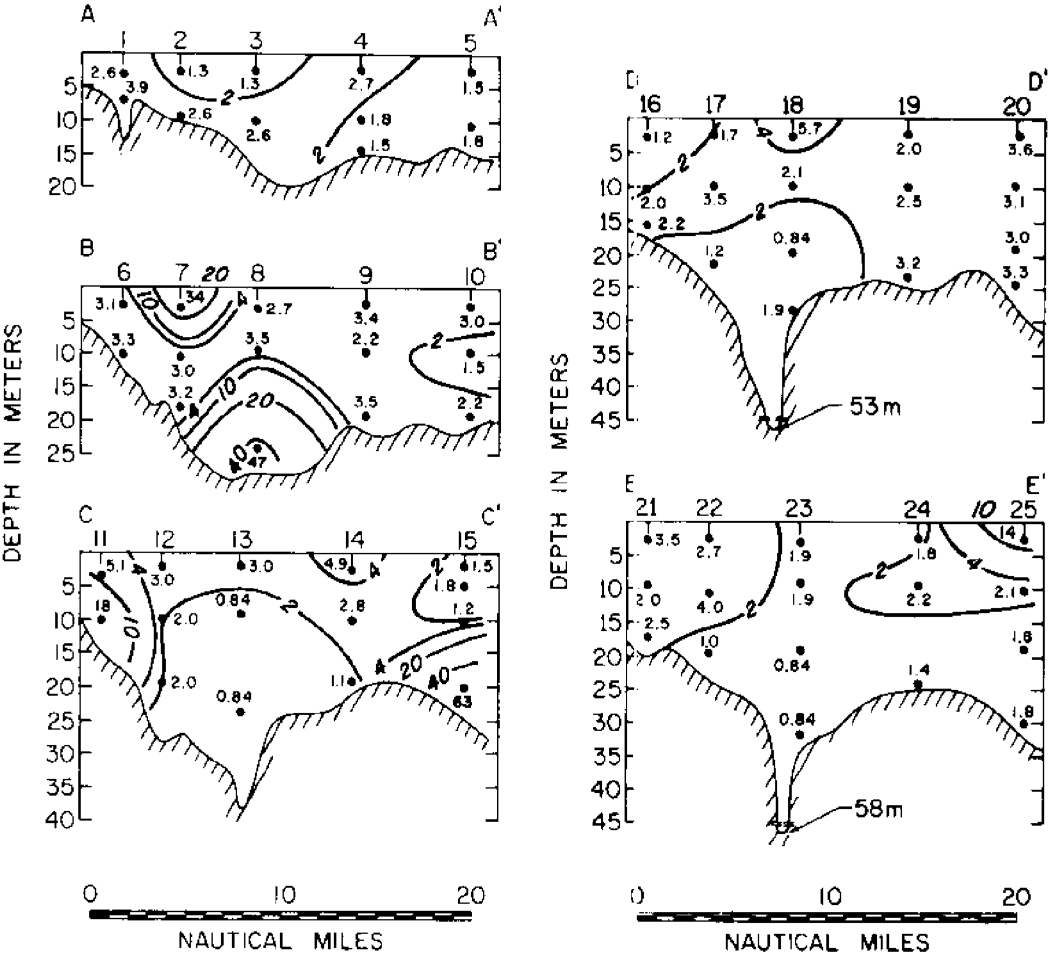

Fig. 6. Total dissolved and particulate copper (μg/liter) in the apex, 16–20 April 1974.

particularly during rough weather. This layer of water with high Fe concentrations may be the nepheloid layer containing high concentrations of fine particles. Such particles, if representative of resuspended bottom sediments, would contain considerably higher concentrations of Fe than of Mn, Cd, and Cu, and most of this Fe should be soluble in weak acid (Gross et al. 1971).

Concentrations of Mn and Fe plotted against salinity (Figs. 9 and 10) suggest that total Fe concentrations are useful in tracing water mass mixing. Iron added to the New York Bight in low salinity river discharge is mixed conservatively with the high salinity-low Fe concentration water of the open ocean. However, at high salinities iron is also introduced to the water column from the sediments. Hence, the complex distribution of points in Fig. 9.

The plot of Mn against salinity does not show any Mn input at high salinities and, because of the curvature, suggests that Mn injected by the low salinity discharge is not conservative but is lost from the water column during mixing with the high salinity-low Mn water. In Fig. 11, Fe concentrations are plotted against those of Mn. The distribution suggests a three end-member mixing diagram with almost all of the points falling on two straight lines representing two distinct Mn:Fe ratios. Two water masses, low Mn-high Fe and high Mn-high Fe, are mixing simultaneously with low Mn-low Fe water. Points falling on the high Mn-Fe ratio line are almost all surface or near-surface samples, while those falling on the low Mn-Fe ratio line are deeper samples. The implication is that little mixing takes place between the lower and upper layers when the water column is stratified. In addition, the limited data available here suggest that total Fe:Mn ratios or Fe:Mn ratios in suspended particulates may serve as a tracer for suspended sediment mixing.

Cu and Cd concentrations in the apex during April (Figs. 6 and 7) seem to be almost uniform except in a few isolated areas. These high values could either represent poor data or reflect the inadequacy of the sampling density in describing the small-scale variations in concentration. The latter seems more probable (*see below*).

May-November 1974 data (filtered samples)—Six apex cruises followed the April cruise (6-9 May, 10-13 June, 16-19 July, 21-24 August, 29 September-2 October, 4-7 November). The quantity of low salinity

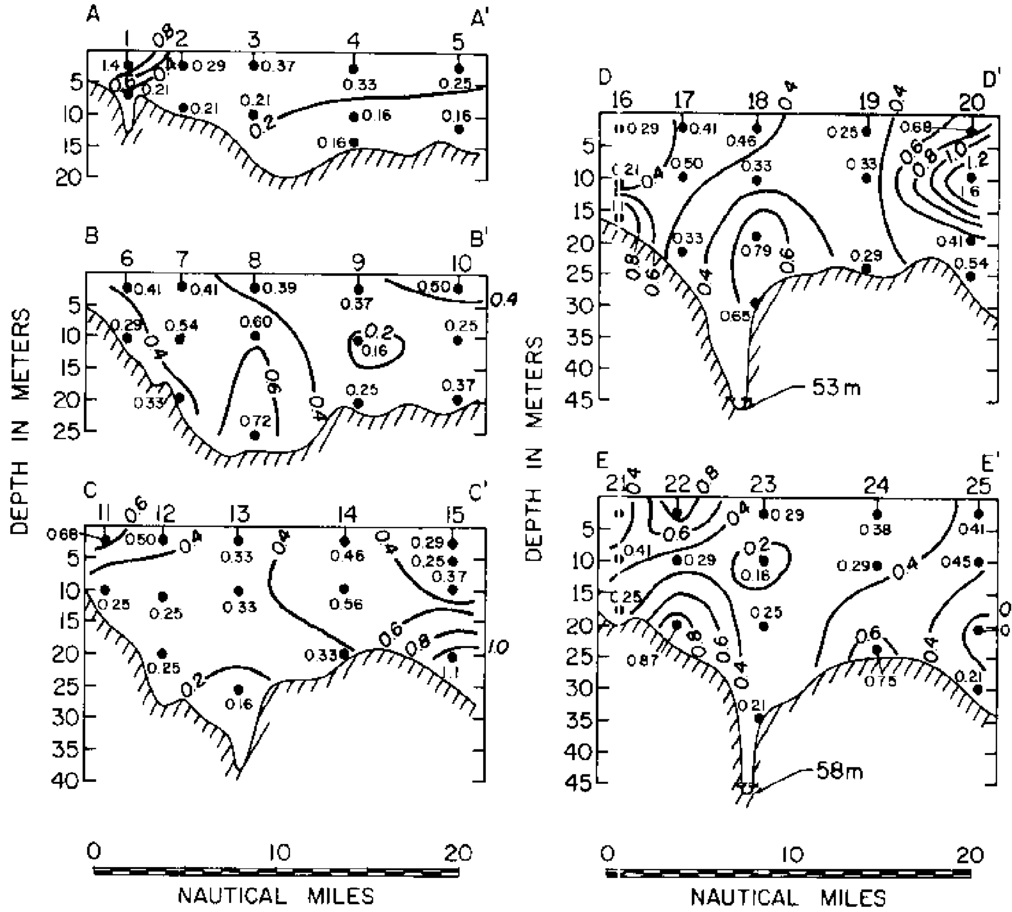

Fig. 7. Total dissolved and particulate cadmium (μg/liter) in the apex, 16–20 April 1974.

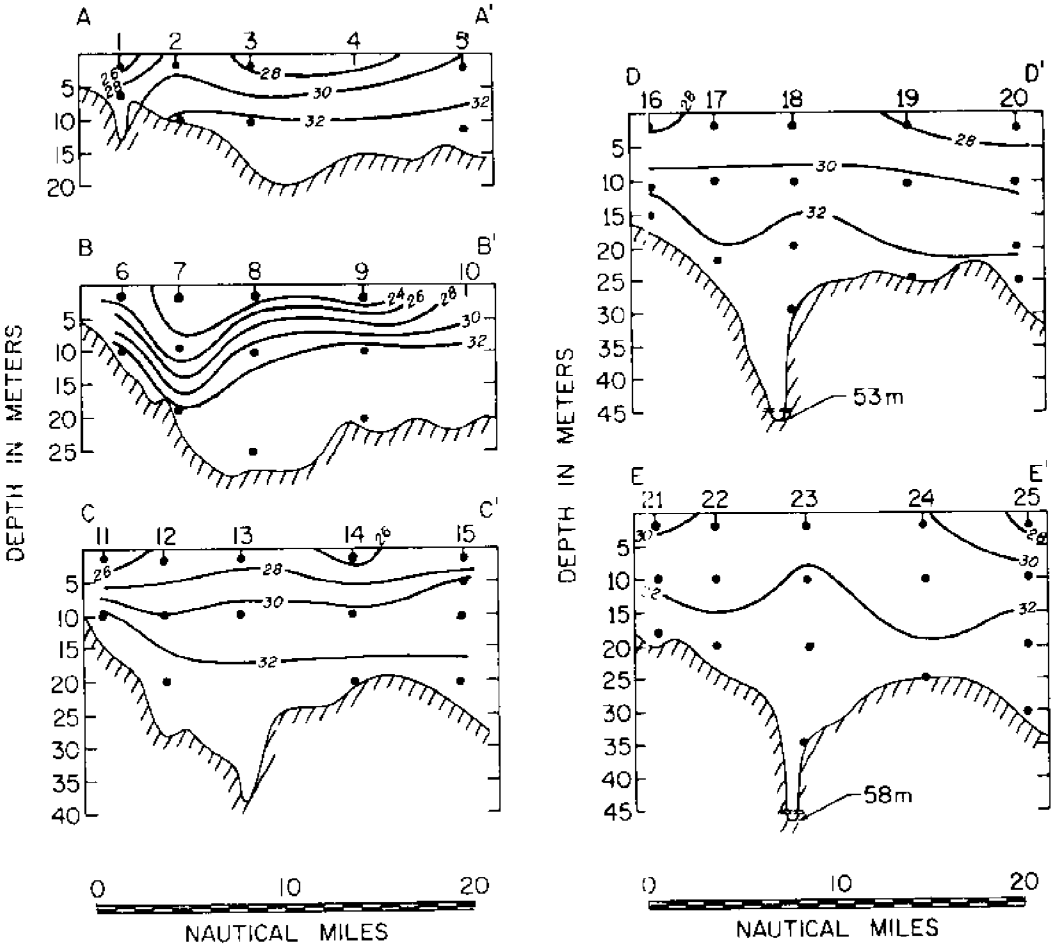

Fig. 8. Salinity (‰) in the apex, 16–20 April 1974.

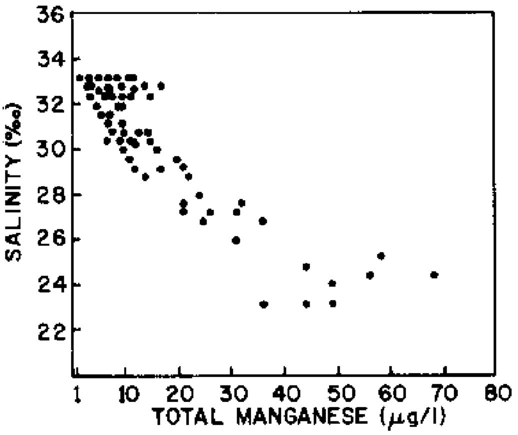

Fig. 9. Total dissolved and particulate manganese in the apex as a function of salinity, 16–20 April 1974.

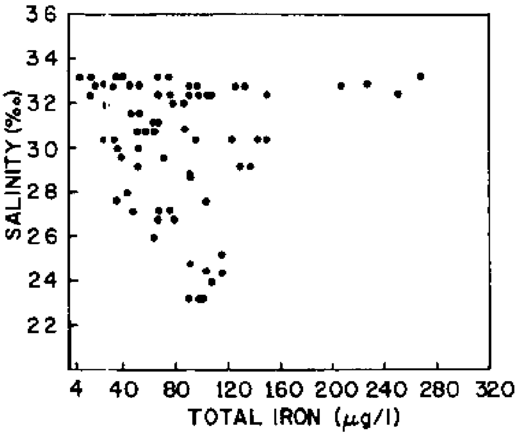

Fig. 10. Total dissolved and particulate iron in the apex as a function of salinity, 16–20 April 1974.

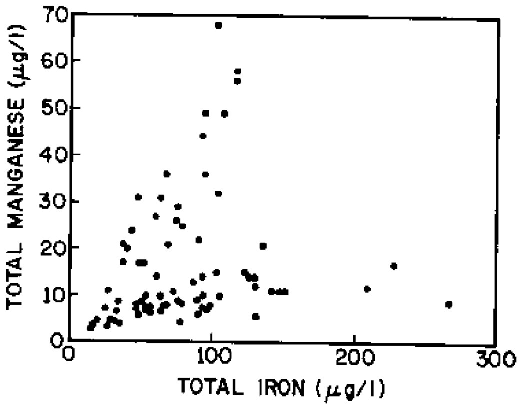

Fig. 11. Total dissolved and particulate iron in the apex as a function of total dissolved and particulate manganese, 16–20 April 1974.

water discharging from Lower New York Bay was less and average salinities throughout the apex were always several parts per thousand higher during this period than during April (Hazelworth et al. 1975*a*,*b*). All samples collected on these later cruises were filtered through 0.4-μ Nuclepore membrane filters and acidified immediately after they were brought aboard ship. Analyses thus determine the total elemental concentration in solution. Station 26 was occupied only on the cruises in September and November, and Zn was determined only on samples from these and the August cruise.

Distributions of total dissolved iron in June, July, August, and September are

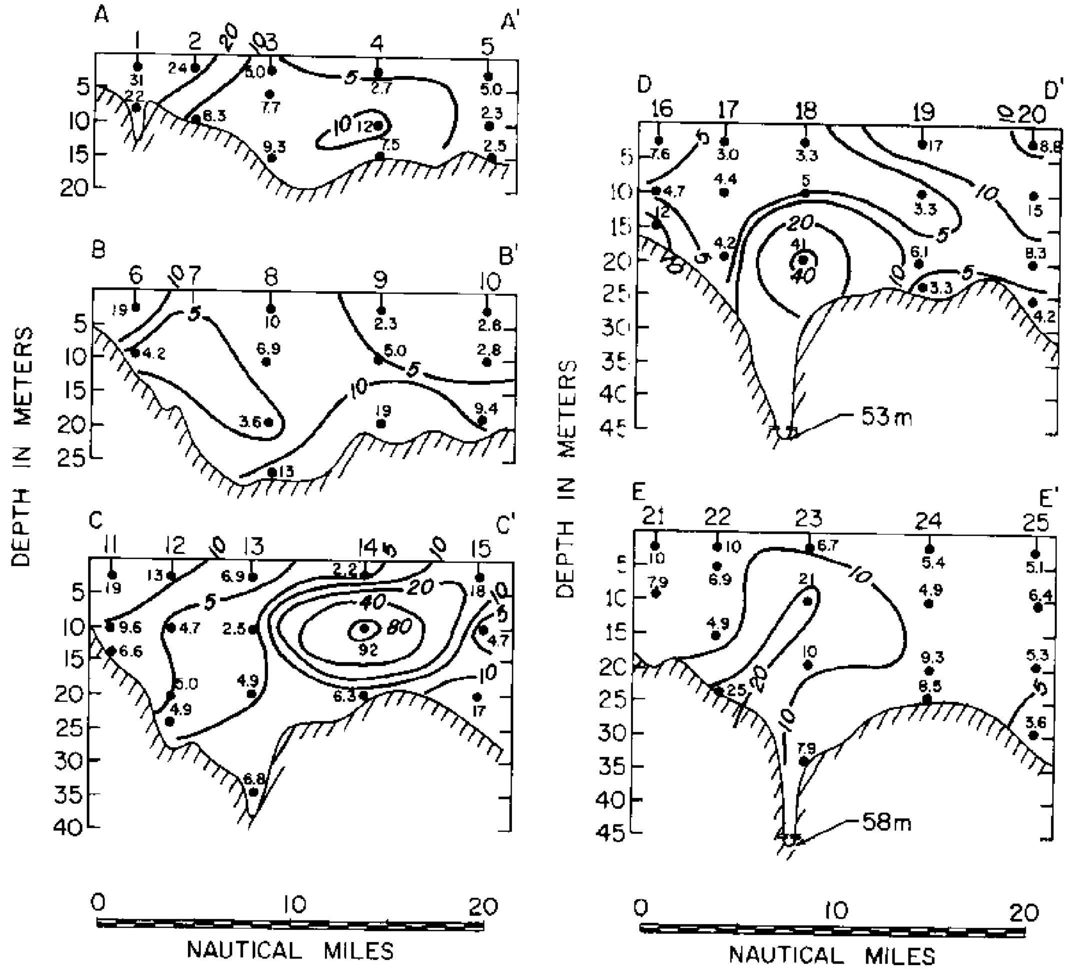

Fig. 12. Total dissolved iron (μg/liter) in the apex, 10–13 June 1974.

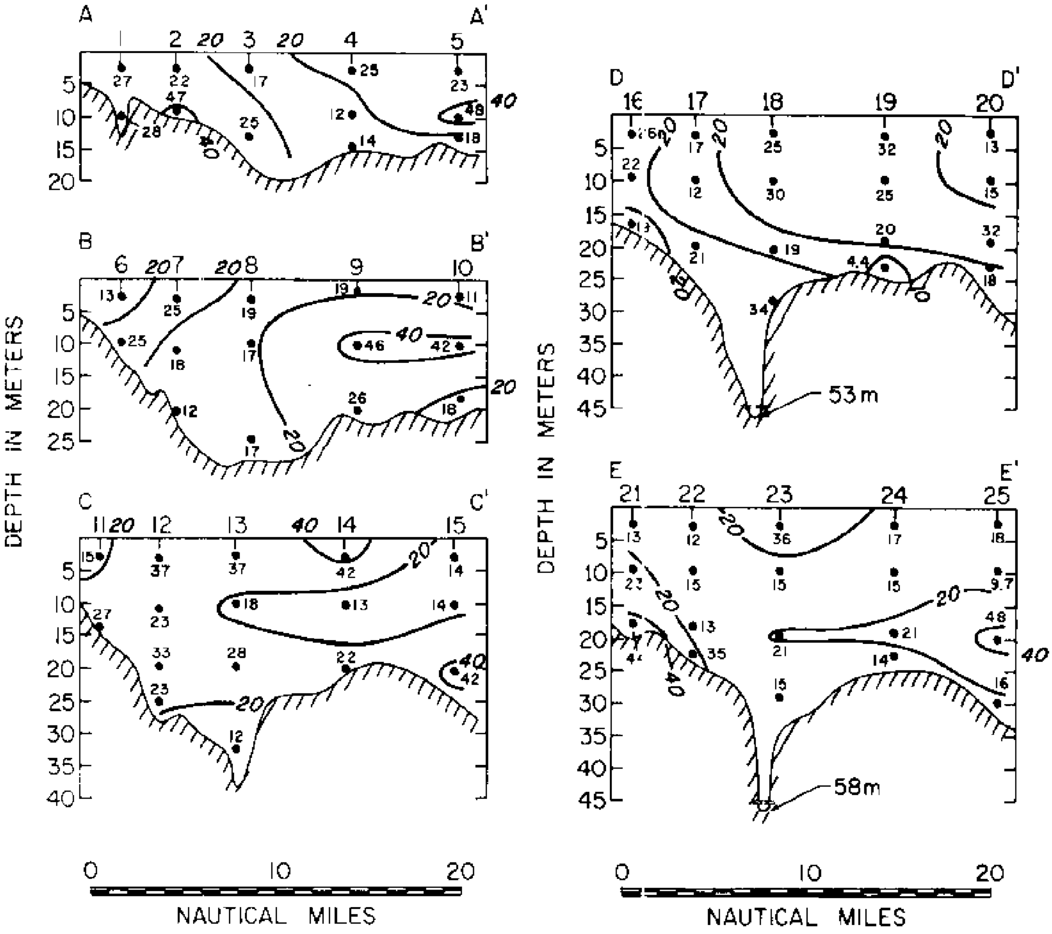

Fig. 13. Total dissolved iron (μg/liter) in the apex, 16–19 July 1974.

shown in Figs. 12–15. The June distribution suggests significant Fe inputs from the estuary; during other months no such input is apparent. Distributions are generally nonuniform with higher concentration regions of water often occurring either to the north or south of the acid waste dumpsites (*see Fig. 2*). Uniform concentrations tend to occur at a particular sampling depth over large distances, despite relatively large changes in concentration with depth. This is particularly apparent in July (Fig. 13) and suggests that little vertical mixing of dissolved species takes place, at least across the thermocline. High dissolved Fe concentrations were not generally found close to the sediment–water interface. This contrasts with the April observations of total acid-soluble iron discussed above and supports our hypothesis that the excess iron near the bottom is particulate and occurs in a nepheloid layer.

Total dissolved Mn distributions for the June, July, August, and September cruises are shown in Figs. 16–19. The Mn distributions are more consistent than those of Fe. High Mn concentrations are always found near the sediment–water interface. Somewhat higher concentrations are also generally observed in the surface water east of the sewage sludge dumpsite and north of the acid waste dumpsite. The excess Mn in these surface waters may come from the acid waste, which contains appreciable quantities of Mn (Gross 1970*b*) and probably disperses northward with the surface drift (Charnell et al. 1975).

High dissolved Mn concentrations in the bottom waters of the bight are probably caused by loss of reduced Mn from anoxic sediments. The Mn will be oxidized and reprecipitated, but the oxidation process is slow (Morgan 1971), particularly when dissolved oxygen concentrations are low (Segar and Berberian 1976).

The Cu concentrations in June, July, and September (Figs. 20–22) are rather uniform, ranging between about 2 and 4 ppb over most of the apex. However, concentrations in the estuarine outflow and apparently in cells of surface water north of the acid waste dumpsite are usually somewhat higher than concentrations elsewhere in the apex. The region north of the acid waste dumpsite is where high Mn concentrations are also found. Cells of high Cu

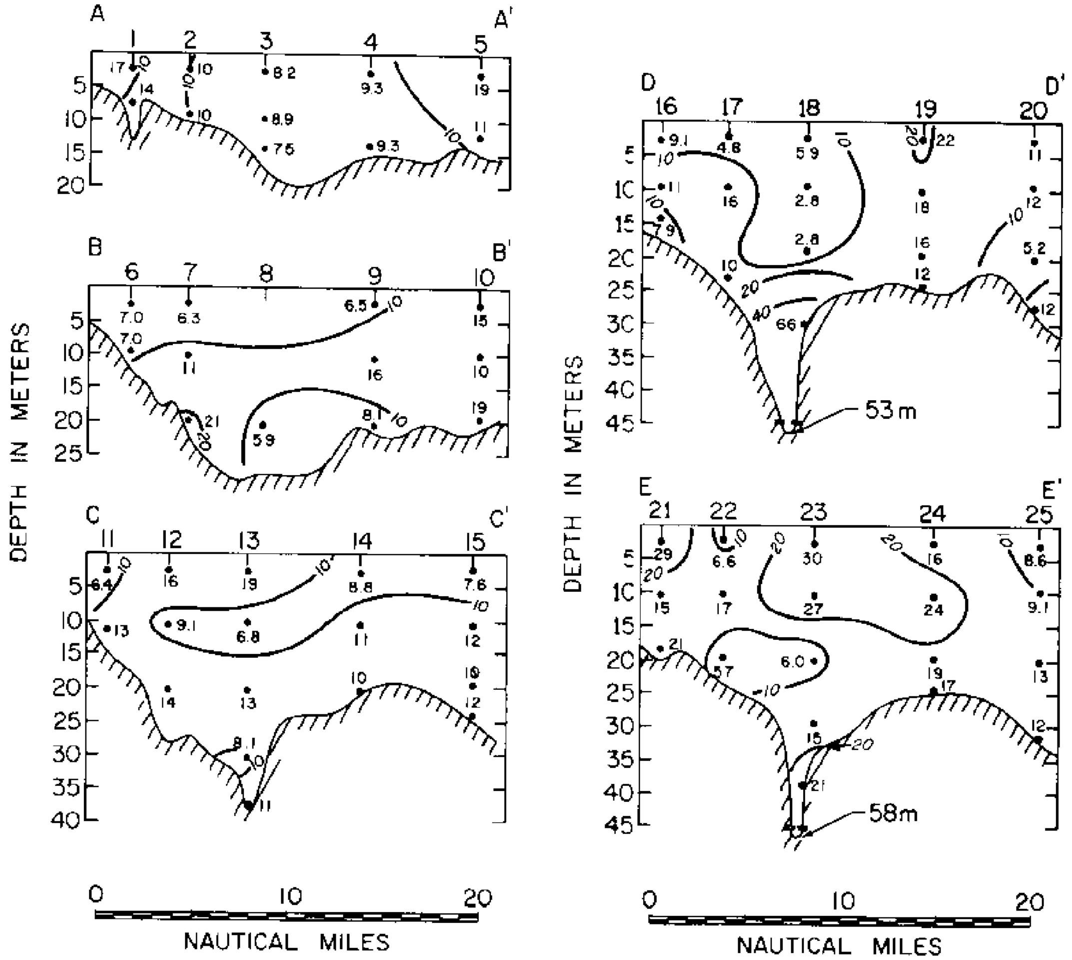

Fig. 14. Total dissolved iron (μg/liter) in the apex, 21–24 August 1974.

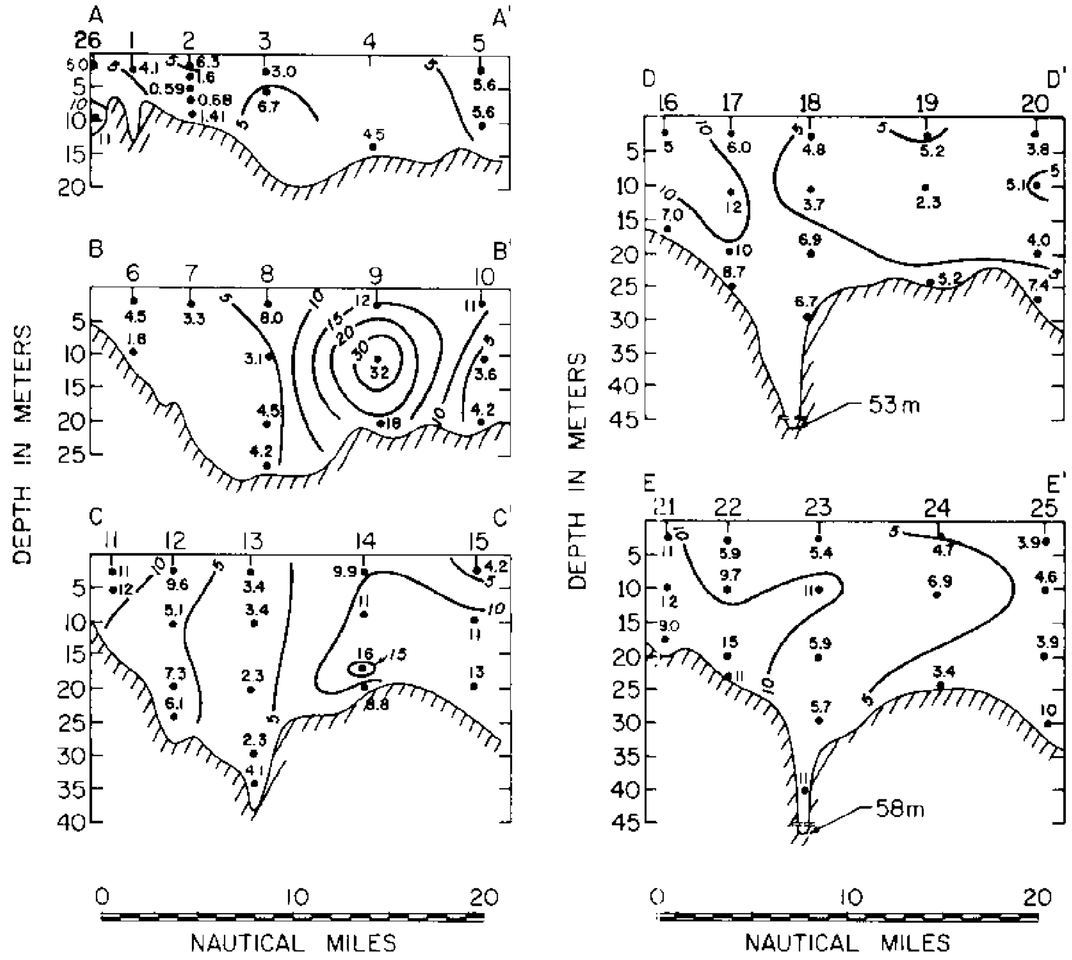

Fig. 15. Total dissolved iron (μg/liter) in the apex, 29 September–2 October 1974.

concentration are found in deeper waters but generally not extending to the deepest sample. They appear to be associated with the regions where high Mn and low oxygen concentrations are observed. Perhaps, they are generated by biological decomposition and associated Cu release or by oxidation of sulfides in resuspended bottom sediments. If the latter were the case, then the deepest samples would show lower concentrations, due either to scavenging by nepheloid layer particles or to sulfide reprecipitation.

Cd concentrations in June, July, and September (Figs. 23–25) do not vary greatly throughout the apex, although the estuarine outflow and near-bottom samples tend to have higher concentrations than waters elsewhere in the apex. Areas of high Cd concentration water were observed, but no obvious pattern can be delineated.

Zn concentrations in August, September, and November (Figs. 26–28) are high compared to reported values for other coastal areas (Morris 1971; Windom et al. 1971; Windom and Smith 1972; Butterworth et al. 1972; Abdullah et al. 1972; Steele et al. 1973; Preston 1973). However, the lowest values found in the outer stations in September

(Fig. 27) are within the range reported for other regions.

Estuarine waters entering the apex have Zn concentrations above background there. Concentrations near the sediment–water interface are often high, and discrete cells of Zn-rich water are observed.

The most striking features of the Zn distributions are the high surface concentrations at stations 23 and 24 in August (Fig. 26) and the high concentrations at stations 12 and 13 in August (Fig. 26) and station 13 in September (Fig. 27). Station 13 is located almost at the sewage sludge dumpsite, and high Zn concentrations there are probably caused by release of Zn to solution during sewage sludge disposal. The high concentrations at stations 23 and 24 in August are possibly due to releases of Zn from acid wastes.

The highest zinc concentrations observed were below levels known to be acutely toxic to marine organisms. But the concentrations are high enough and may be persistent enough to cause concern about potential chronic toxic effects (Connor 1972; Brown and Ahsanullah 1971).

It is relevant at this point to consider the validity of trace metal data showing cells of water with anomalous concentration.

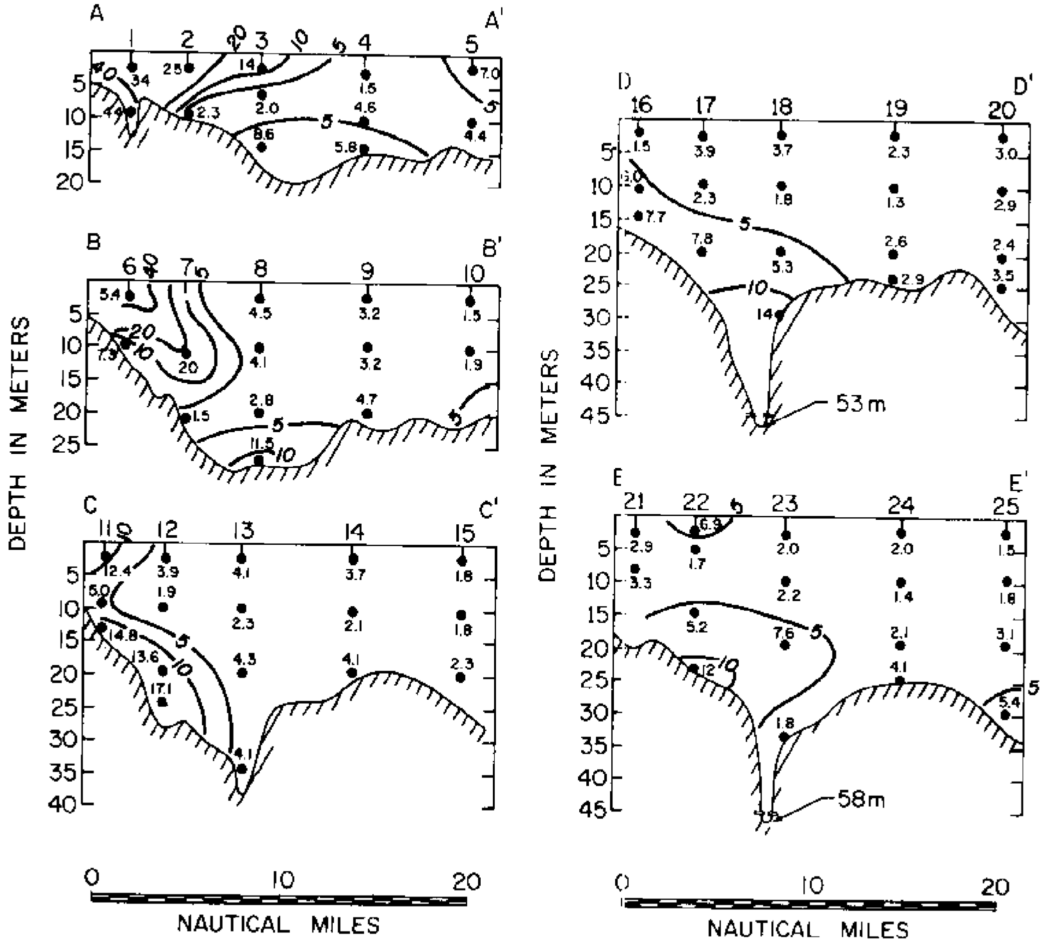

Fig. 16. Total dissolved manganese (μg/liter) in the apex, 10–13 June 1974.

Often the anomalous water is characterized by only a single data point. In such instances it is possible that analytical error is responsible for the anomaly. However, we believe that our anomalous values are genuine, with perhaps one or two exceptions. We have sampled the same stations twice, several days apart, usually obtaining good agreement between successive samplings, even when observed concentrations were anomalous compared to other areas. In addition, the vertical and horizontal integrity and the repetition of the high Zn concentration at station 13, a phenomenon clearly explicable by known chemical processes, convince us that such features elsewhere are significant although less easily explained.

Samples from the August cruise were analyzed for V and Ni after extraction with ammonium pyrollidine dithiocarbamate into methyl isobutyl ketone. Vanadium concentrations were generally below the detection limit (ca. 0.2 µg/liter), while Ni concentrations ranged from about 0.5 to 7 µg/liter. Both metals occurred in higher concentrations in the estuarine influenced region (V ca. 3 µg/liter) and were uniformly low

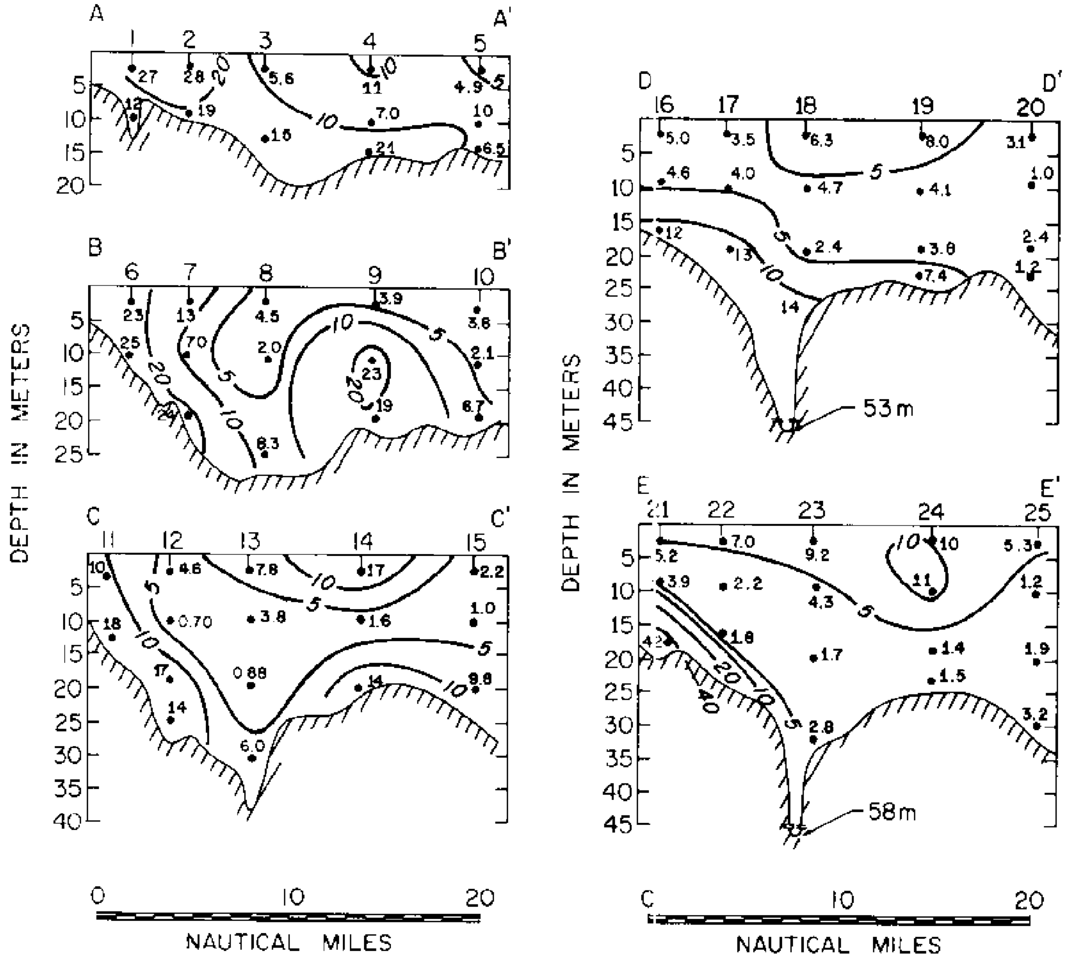

Fig. 17. Total dissolved manganese (µg/liter) in the apex, 16–19 July 1974.

throughout the rest of the apex. In the outer areas of the apex, concentrations of Ni in surface waters (1–2 μg/liter) were generally somewhat higher than in deeper water (0.5–1 μg/liter).

Seasonal variations—To simplify the data so that seasonal trends and geographical variations might be more easily understood, average metal concentrations for several regimes were calculated. The average concentration, including all depths sampled, was calculated for each cruise for four areas (Fig. 29): estuarine influenced (stations 1, 2, 6, 7, and 11), dumpsite influenced (stations 8, 12, 13, 14, and 18), offshore control (stations 10, 15, 20, 23, 24, and 25), and, for our single expanded grid cruise (26 February–3 March 1975), outer shelf (Fig. 3). These averaged data are plotted for Fe, Mn, Cu, Cd, and Zn in Figs. 30–34. Several observations can be made.

1. Outer shelf stations had lower average concentrations of each metal except Cd than the apex in winter. In winter the outer shelf stations had lower average Mn concentrations than the average concentrations in the apex at any time of year.

2. Average surface layer Mn concentrations in the estuarine influenced regions

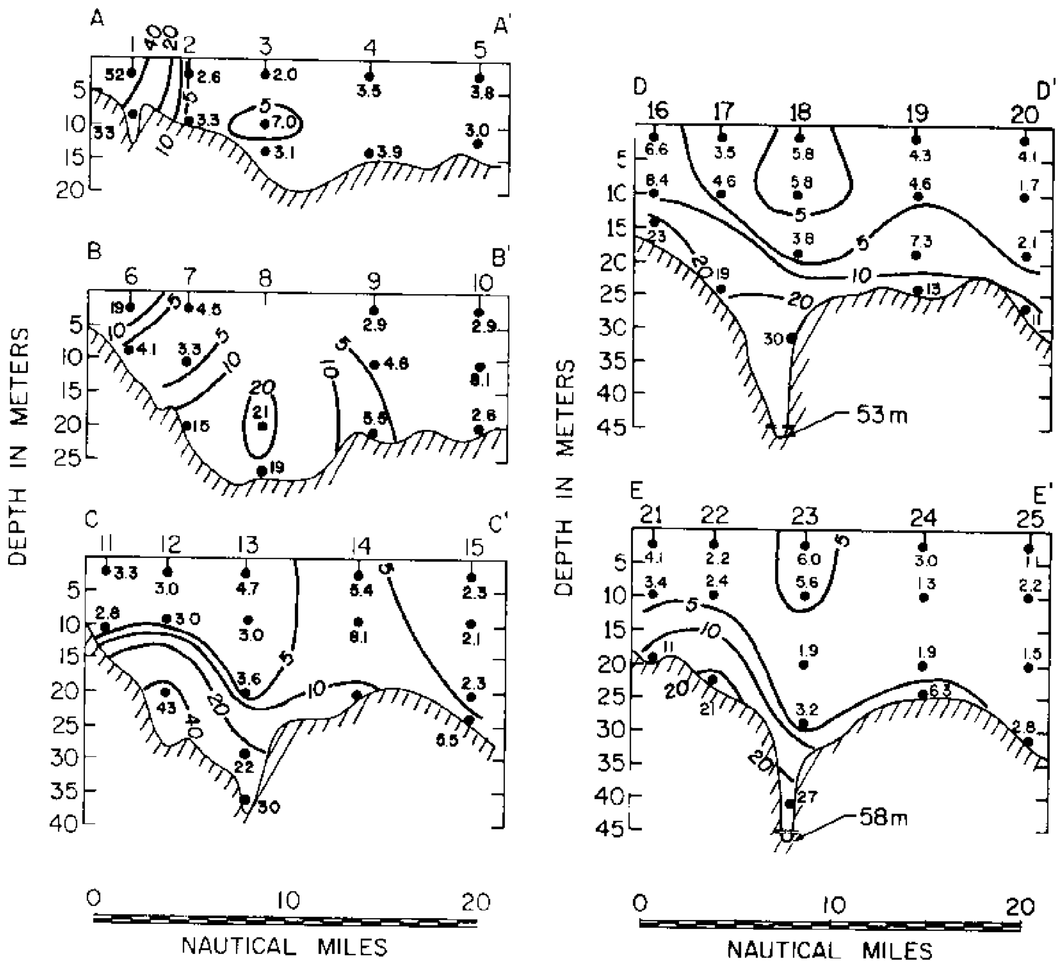

Fig. 18. Total dissolved manganese (μg/liter) in the apex, 21–24 August 1974.

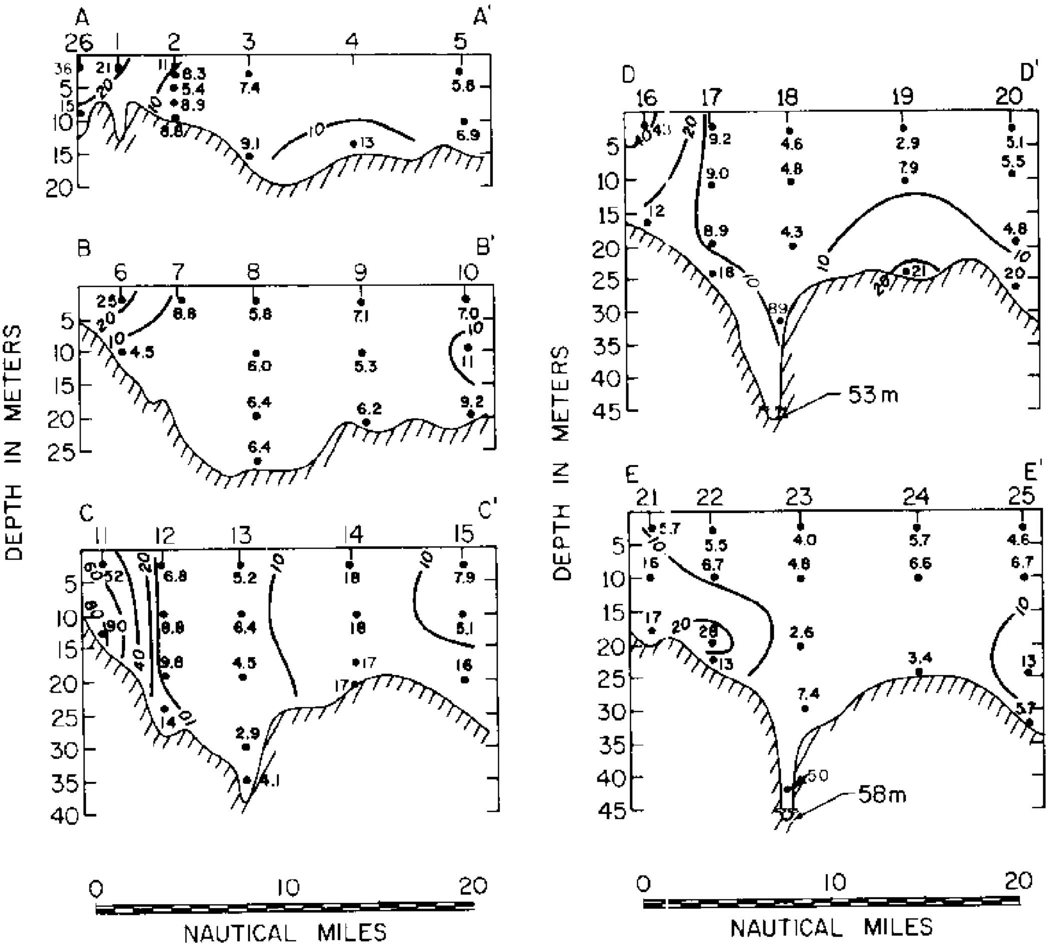

Fig. 19. Total dissolved manganese (μg/liter) in the apex 29 September–2 October 1974.

were always higher than the average concentrations offshore where Mn was rather uniform. This suggests that the estuary is a major source of dissolved manganese. Mn added from dumping must be precipitated or remain in the suspended phase.

3. The estuary seems to be a significant source of high Cd and Zn concentrations only at certain times.

4. In no instance is the average concentration of a metal in the dumpsite region significantly higher than in the rest of the apex, although the concentration of Zn in the immediate vicinity of the sewage sludge dumpsite is often anomalously high (Figs. 26–28).

5. Surface and lower layer (below 10-m depth) average metal concentrations are not significantly different except for Mn. Average lower layer Mn concentrations in the dumpsite region are significantly higher than surface concentrations. The differential is caused by high near-bottom Mn concentrations in this region where oxygen

concentrations are low and reduced Mn is probably diffusing from the sediments.

6. Concentrations of each metal in the apex were lower during spring and fall than during summer or midwinter. High summer concentrations are probably caused by restricted circulation and the consequent longer residence time and higher equilibrium concentration of contaminants during this season. High concentrations in winter, particularly in the deeper water, are probably caused by release of dissolved trace metals from resuspended sediments, as the winter cruise took place in the days immediately following a storm.

7. Average Zn and Cd concentrations in the bight are higher than reported in other coastal waters. Cu, Fe, and Mn concentrations are high but within the range reported for contaminated coastal waters (Morris 1971; Windom et al. 1971; Slowey and Hood 1971; Butterworth et al. 1972; Abdullah et al. 1972; Sundaraj and Krishnamurthy 1972; Steele et al. 1973; Preston 1973; Sankaranarayanan and Reddy 1973). High concentrations of the various metals observed in the bight may be due either to analytical methodology differences (our method determines essentially total dis-

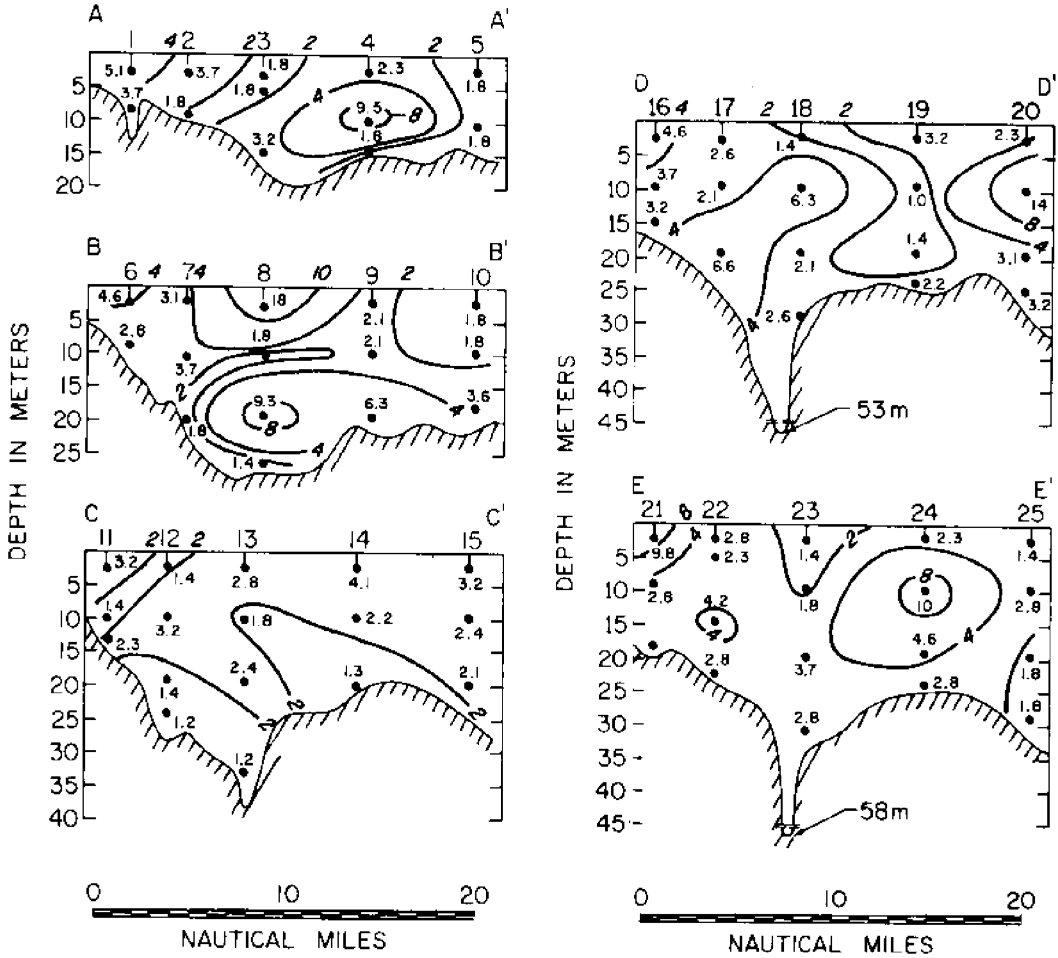

Fig. 20. Total dissolved copper (µg/liter) in the apex, 10–13 June 1974.

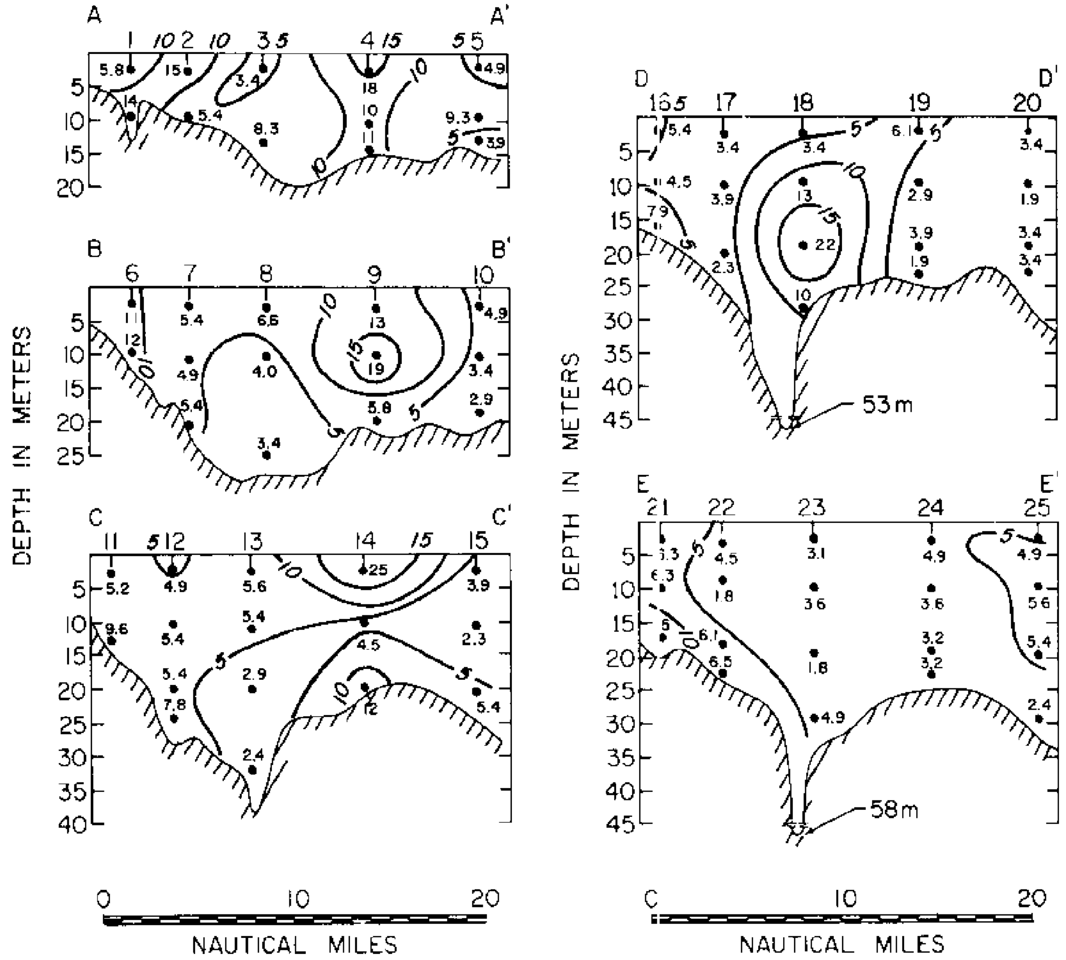

Fig. 21. Total dissolved copper (μg/liter) in the apex, 16–19 July 1974.

solved metal) or to the impact of man's contamination or to both.

Some considerations on metal speciation—A new analytical technique was used which determines the total metal concentration in the sample without regard to chemical form (Segar and Cantillo 1975). The results obtained were compared with analyses performed by separating metals from the salt matrix by chelation with ammonium pyrollidine dithiocarbamate (APDC) and extraction with methyl isobutyl ketone (MIBK). The extractive analyses were carried out by standard additions using a modification of the method of Kremling and Petersen (1974). Excellent agreement was found between methods for each metal examined if analysis was carried out on water samples from the estuarine discharge region. However for Fe and Cu and possibly Cd, Zn, and Mn in samples from outside the estuarine influence, concentrations found by the total analysis method were much higher than those obtained by the extraction method. The extraction of standard additions was quantitative from all samples regardless of origin.

Total Cu and extractable Cu are plotted against one another for samples from the August apex cruise in Fig. 35. Points lying between the two straight lines represent agreement between the two values and therefore 100% extractability within the estimated errors of the two analyses. Samples with quantitatively extractable Cu were predominantly taken at the surface in the area of estuarine influence, whereas points lying progressively to the right of this line (i.e. <100% extractability) derive from deeper samples or samples from progressively farther offshore. Salinity variations were relatively small in the apex during this cruise, the lowest salinity sample being about 28‰. No obvious relationship existed between salinity and either total copper or extractable copper concentrations.

Apparently in the open shelf waters of the bight apex metals occur in a form that is not present in the estuarine discharge and is not extracted by APDC/MIBK even when the sample has been acidified to pH 1 and stored for several weeks before analysis. The inextractable metal may be the fraction released by oxidation of seawater samples (Corcoran and Alexander 1964; Hood 1967; Williams 1969; Slowey and Hood 1971), and it may be organically associated. However, if it is organically associated metal, it is surprising that it does not seem to be present in the estuarine water which has a high dissolved organic carbon concentration (Alexander and Alexander in press).

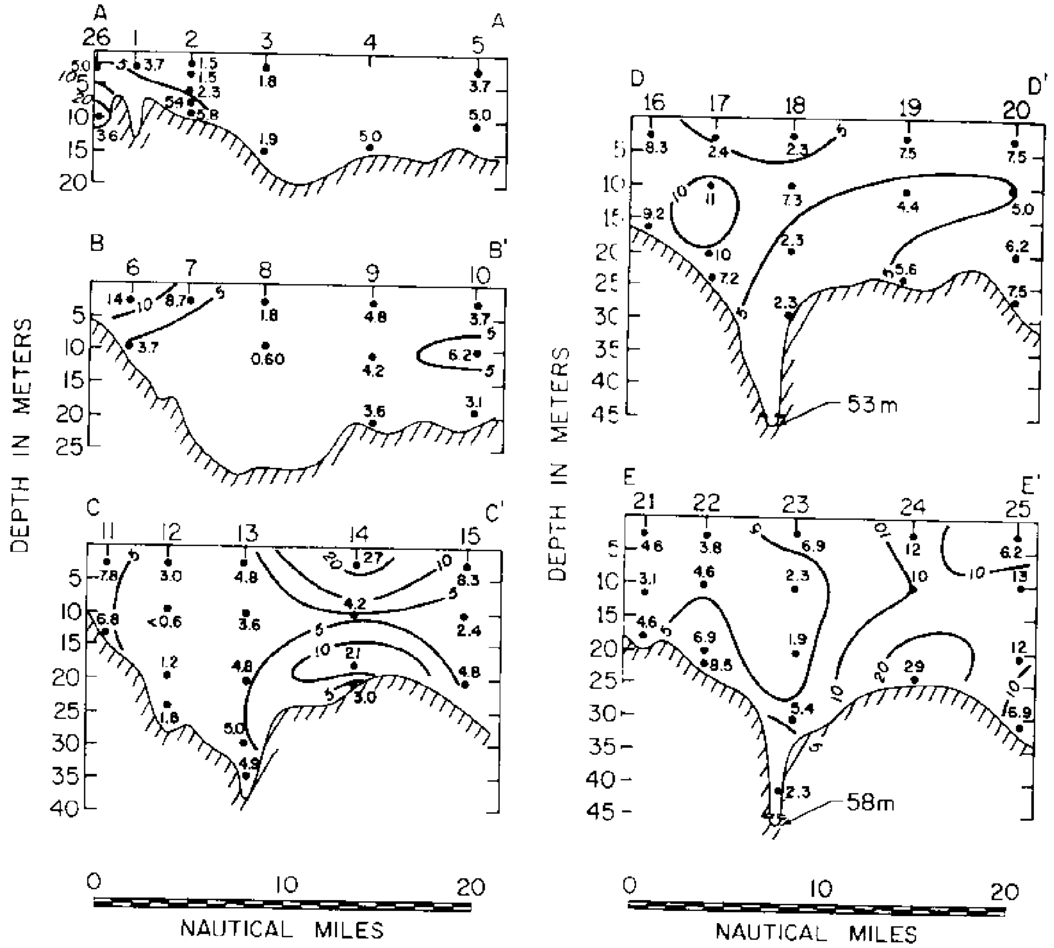

Fig. 22. Total dissolved copper (μg/liter) in the apex, 29 September–2 October 1974.

Metal budgets—Budgets for trace metals in the bight can be made only imprecisely because existing data are inadequate. We lack adequate estimates for most metal inputs and for the distribution of most trace metals in the New York Bight ecosystem. At this time, reasonable budget calculations can be made for Cu and Zn.

The total quantity of Cu and Zn in the apex, a region defined by our 26 station grid (Fig. 2), with an area of 2,400 km^2 has been estimated from data discussed above. The quantity of these metals in the upper 1 cm of the sediments of the apex has been calculated from the data of Carmody et al. (1973) (Table 1). The amount of each metal in the water column is comparable to the amount in the upper 1 cm of sediment. Only the upper few centimeters of sediment are able to exchange elements with the water column and the biota (Duursma and Gross 1971). As the deposition rates in the bight are generally low (G. Freeland per-

Table 1. Total metal in the apex (tonnes).

	Cu	Zn
Average water column*	212	1,500
Sediments (upper 1 cm)	410	855

* Dissolved metal only.

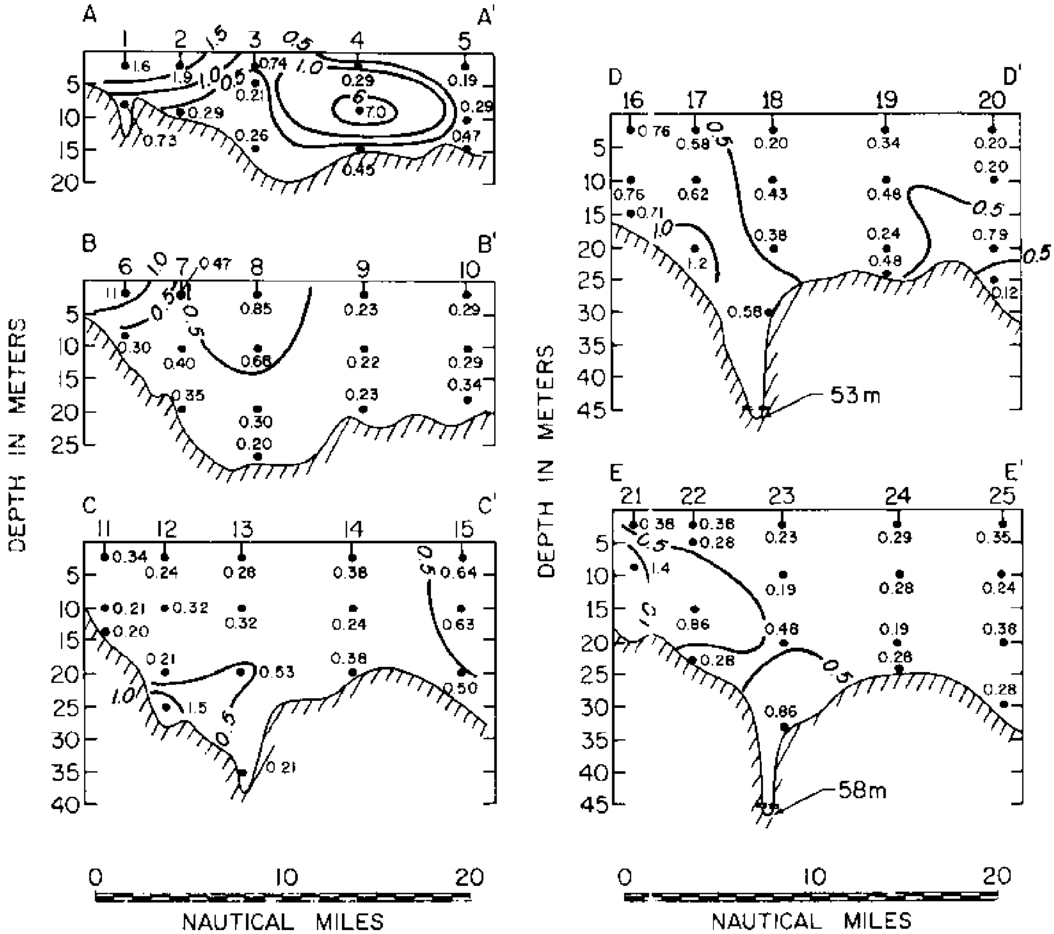

Fig. 23. Total dissolved cadmium (μg/liter) in the apex, 10–13 June 1974.

sonal communication), and as some equilibrium exists between sediments and water, this implies that an appreciable fraction of Cu or Zn (and probably other elements) added as dissolved contaminants to the bight apex must stay in solution until either taken up by organisms or physically transported out of the apex.

The total quantity of metal in the water column, as calculated in Table 1, does not include metal incorporated in suspended particles. Some idea of the relative magnitudes of suspended and dissolved metals can be obtained by comparing the concentrations in samples collected in the apex in April, which were not filtered, and those collected later in the year, which were. This comparison is made by averaging all trace metal analyses for the April cruise and averaging all trace metal analyses for the six subsequent apex cruises (Table 2). However, an accurate estimate of the total particulate metal cannot be made from Table 2 because the April hydrographic conditions and, presumably, metal distributions were different from those in the other sampling periods. Also, the April analyses do not include any metal that might be present in the acid-insoluble settleable fraction of the suspended particulates. Nevertheless, from

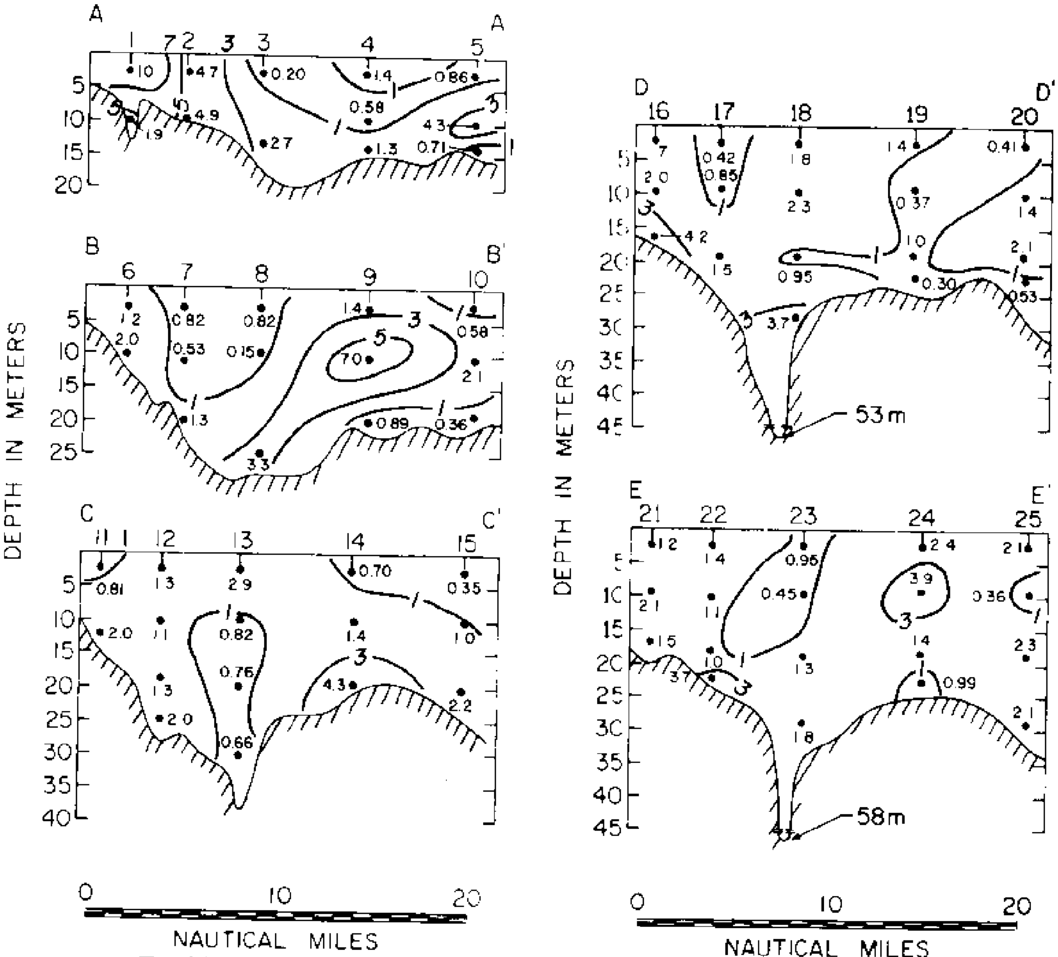

Fig. 24. Total dissolved cadmium (µg/liter) in the apex, 16–19 July 1974.

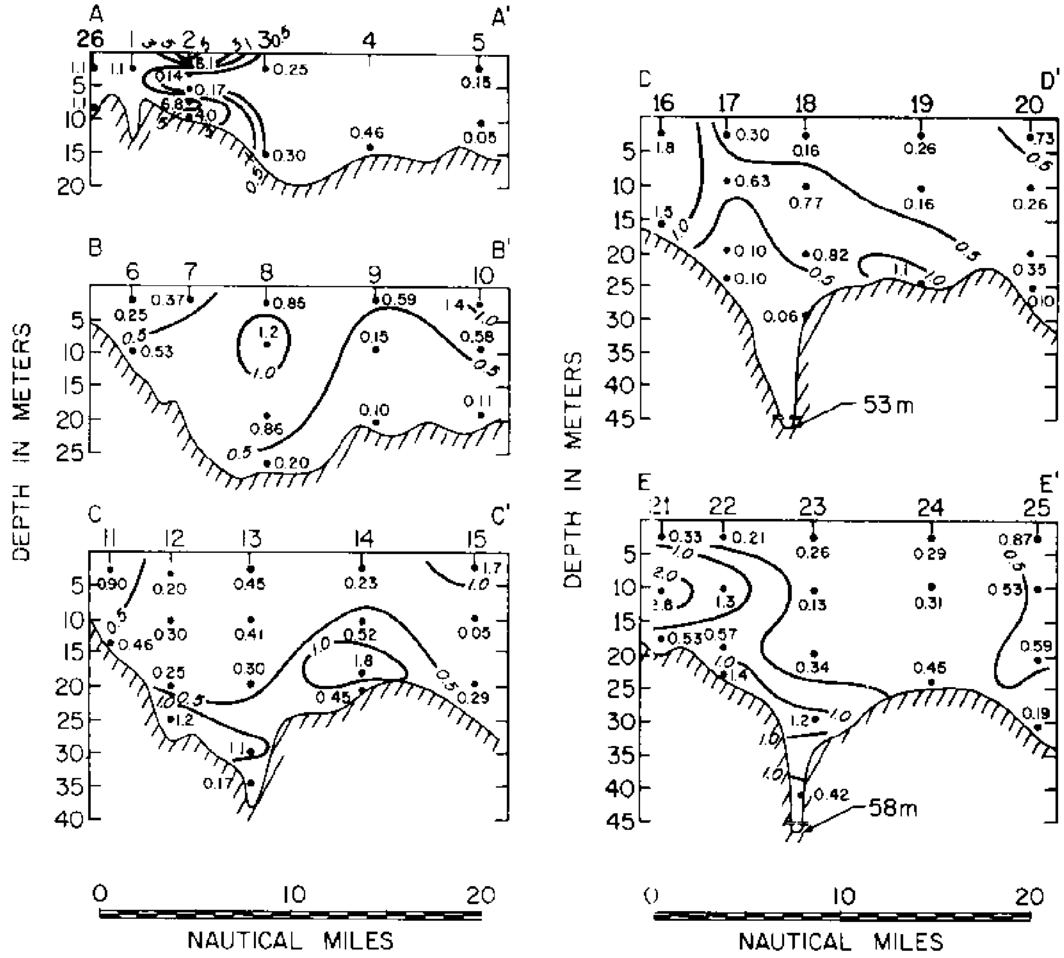

Fig. 25. Total dissolved cadmium (μg/liter) in the apex, 29 September–2 October 1974.

Table 2, and from total suspended load data and preliminary analyses of concentrations in suspended matter (Betzer personal communication), it seems that while Fe and Mn may be present in significant quantities in the suspended phase, Cu, Cd, and Zn will be found predominantly in the dissolved phase. Therefore, suspended Cu, Cd, and Zn are present in only small quantities in the bight apex, and the values in Table 1 underestimate the total metal contents of the water column by at most a few percent.

Table 2. Average metal concentration for all stations (ppb).

	Apr*	May–Nov†
Fe	82	12
Mn	17	8.0
Cu	4.2	4.5
Cd	0.42	0.82
Zn	—	32

* Includes dissolved, plus acid-soluble particulate metal.
† Dissolved metal only.

Cu and Zn inputs to the apex from various sources are estimated in Table 3. It is apparent from these estimates that both acid waste dumping and atmospheric input of Cu are negligible, and of Zn are small,

Table 3. Inputs of metals to the apex (tonnes per day).

	Cu	Zn
Sewage sludge*	0.7	1.8
Dredge spoils*	6.3	7.3
Acid wastes*	0.05	0.2
Atmospheric input†	0.02	0.4
River runoff		
Mueller et al. (1975)	6.2	17
River flow rate × concn	0.48	3

* From Mueller et al. (1975).

† From Mueller et al. (1975), scaled to 2,400 km^2, assuming uniform fallout.

compared to other sources. Dredge spoils appear to be the dominant source for each metal, although river inputs may be large—perhaps comparable to dredge spoil dumping. Even if the low estimates are correct, input of metals from sewage sludge is no greater than that from the estuarine discharge. Evidently caution must be exercised in using concentrations or ratios of trace metal concentrations to identify sewage sludge in apex sediments (Harris 1974).

Differences between the two estimated values of input from the rivers (Table 3) are large. The estimates of Mueller et al. (1975) are almost certainly too high. Their estimates are made by summing the discharges of wastewater and industrial effluents to the lower parts of the Hudson-Raritan-Passaic estuary and New York Bay,

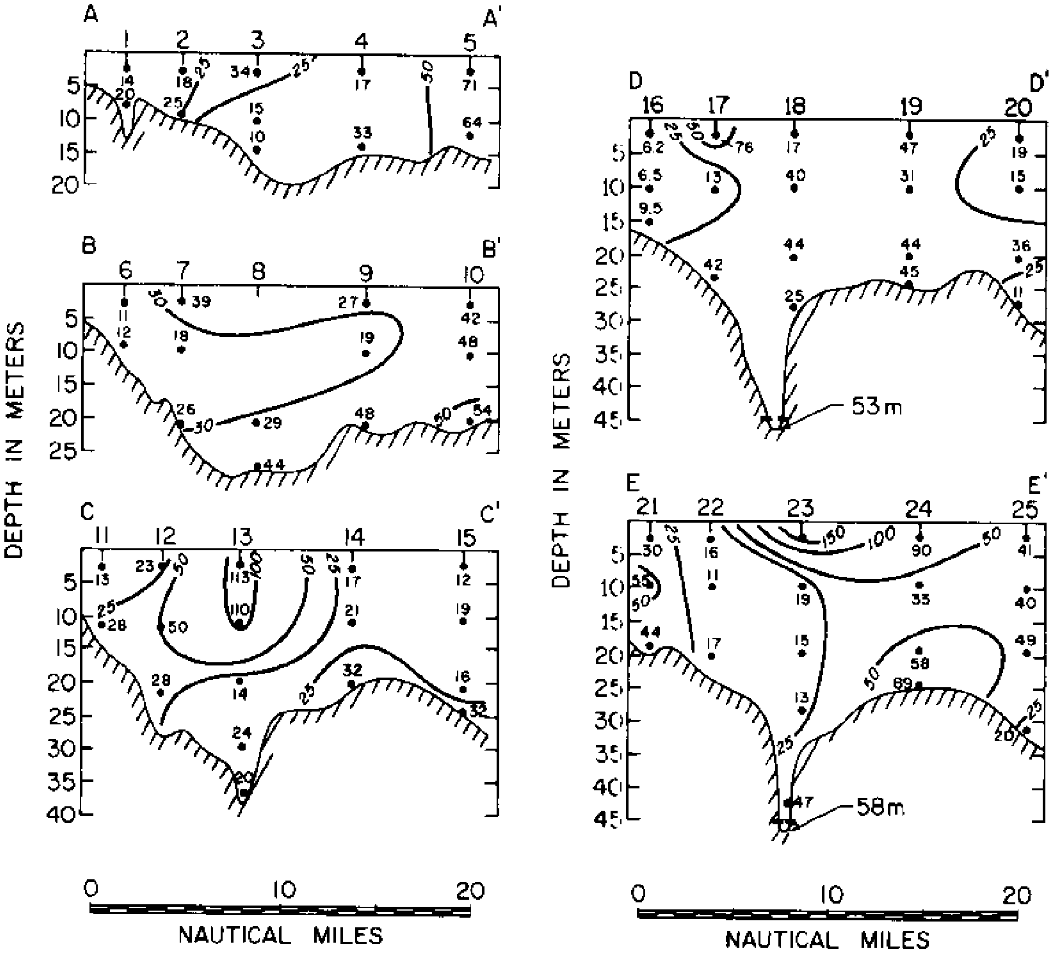

Fig. 26. Total dissolved zinc (μg/liter) in the apex, 21–24 August 1974.

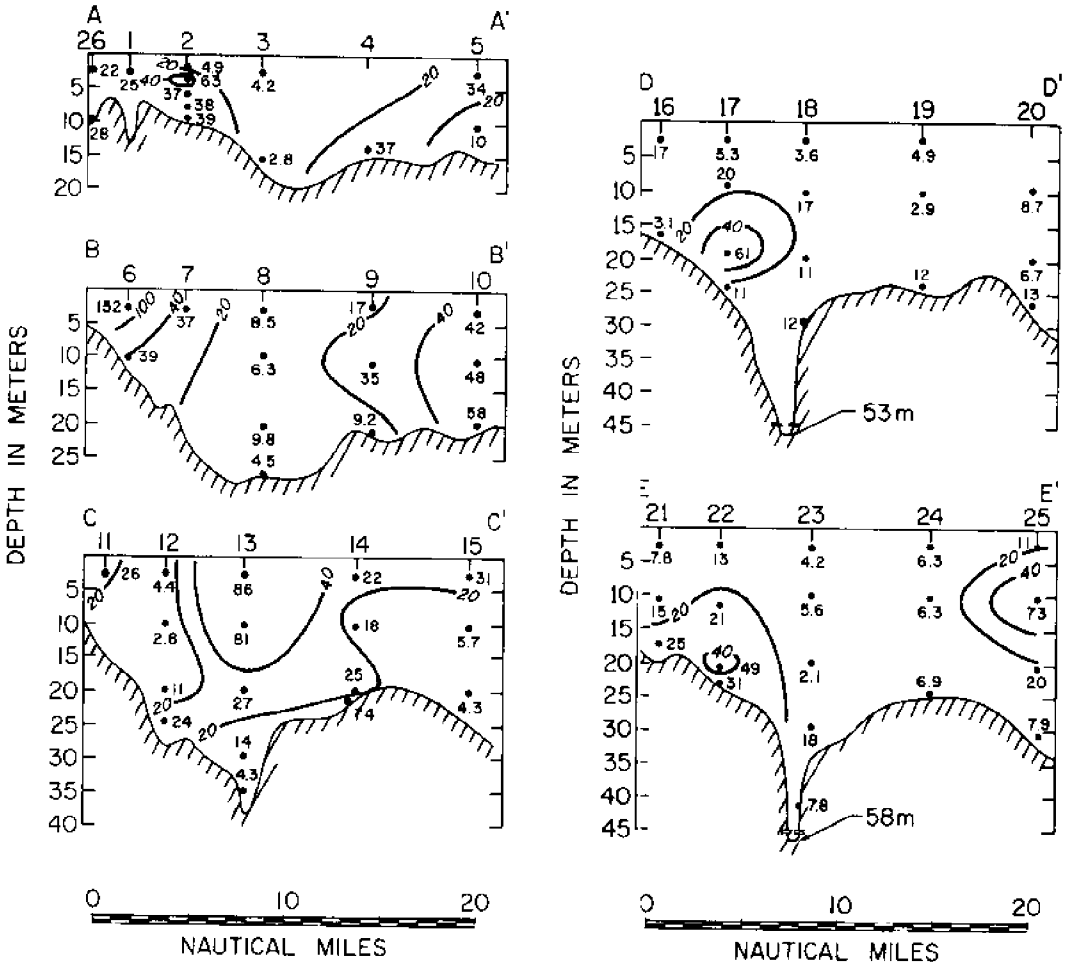

Fig. 27. Total dissolved zinc (μg/liter) in the apex. 29 September–2 October 1974.

together with loads carried by these rivers from their upper reaches. However, a large fraction of the metal contaminants added to the estuary will not be transported to the bight but are probably deposited in the estuary. The deposited material is transported to the bight as dredge spoils. If the estuary has no net accumulation or depletion of contaminants, then the input calculated by Mueller et al. (1975) should be equal to the sum of the actual estuarine transport and the dredge spoil removal. If the estuarine discharge calculated from the mean river flow rates and metal concentrations are added to the dredge spoil input, the sum does approximate the Mueller et al. (1975) estimate of input to the rivers.

The river input of metals, which is calculated from the mean flow rate multiplied by the average metal concentration in water at the mouth of Lower New York Bay (Table 3), is only a crude approximation. Tidal exchange, neglected here, may create a significant net transport of metals either into or out of Lower New York Bay. Better estimates cannot be made from existing data. As long as metal concentrations in low salinity discharges are consistently equal to

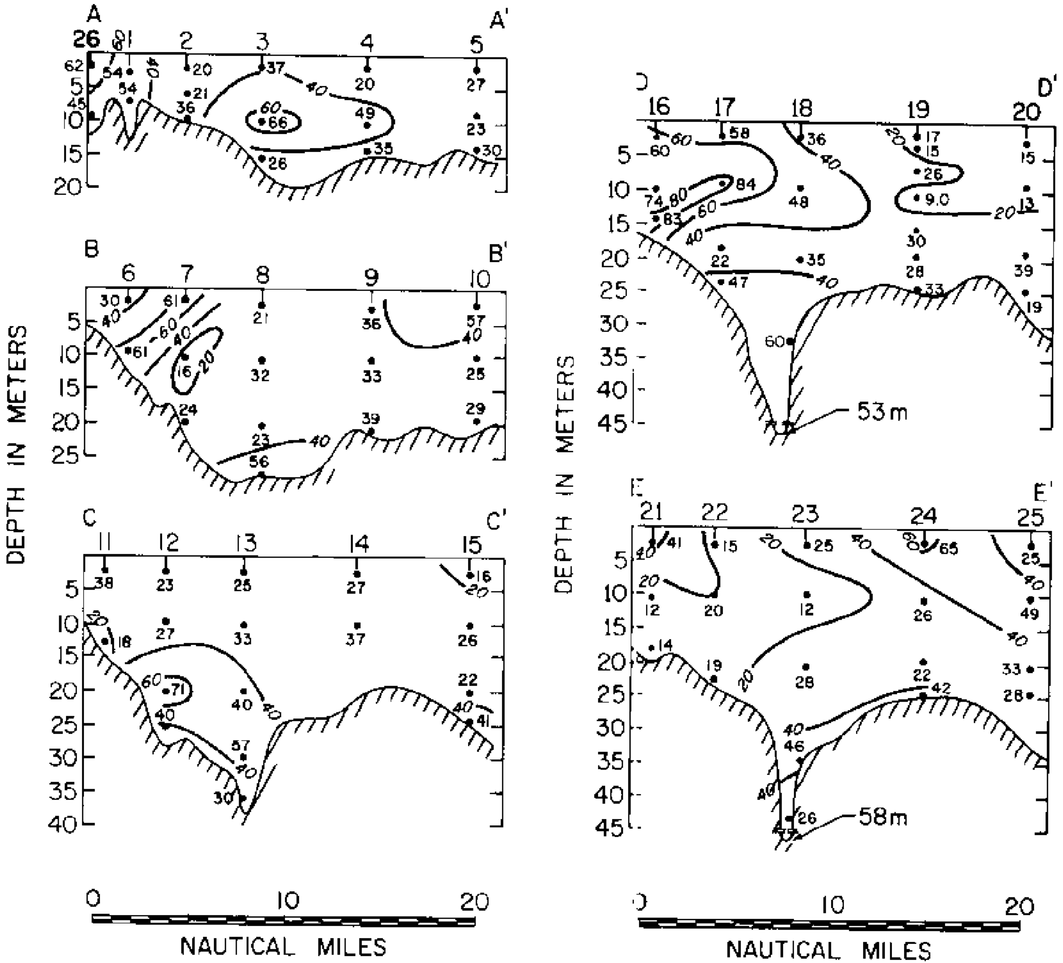

Fig. 28. Total dissolved zinc (μg/liter) in the apex, 4–7 November 1974.

or higher than that in the high salinity return flow to the estuary, input estimates calculated from mean river flow will be minimum values. Limited data for Cu and Zn concentrations in the estuarine discharge (Alexander and Alexander in press) suggest that this condition is fulfilled.

Input rates from the combined dredge spoil and sewage sludge dumping (Table 3) are sufficient to provide the metals contained in the upper 1 cm of sediments in the apex (Table 1) in as little as 94 days for Zn and 59 days for Cu. Net sediment accumulation rates in the New York Bight apex are considerably smaller than this. In fact most of the apex has been eroded slightly between 1936 and 1973 (G. Freeland personal communication). Net accumulation occurred only at the dredge spoil site, $\simeq 1\times 10^8$ m^3 in 37 years (G. Freeland personal communication), and is sufficient to account for only about half of the volume of dredge spoils dumped during this period ($\sim 2\times 10^8$ m^3). These data indicate that much of the solid material dumped into the apex is rapidly dispersed and transported

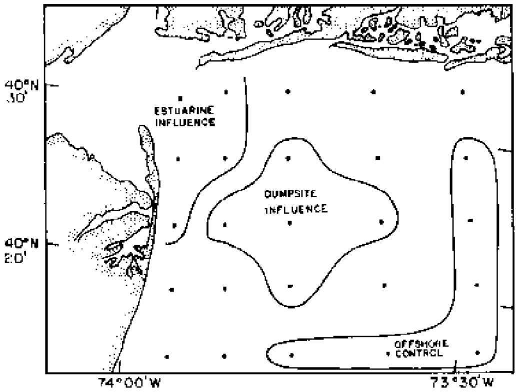

Fig. 29. Stations designated estuarine influenced, dumpsite influenced, and offshore control.

Table 4. Calculated mean residence time (days) of Cu and Zn in the apex.

	No net sediment accumulation		Accumulation of 50% of dredge spoils in sediments	
	Max*	Min†	Max*	Min†
Cu	28	16	48	21
Zn	118	56	165	65

* River input calculated from average concentration × average flow rate.

† River input from Mueller et al. (1975).

from it, presumably seaward and possibly back to the estuaries.

Some idea of the rate of removal of contaminant metals from the apex can be obtained by calculating maximum mean residence times for Cu and Zn in the apex from the input data in Table 3 and the total metal content of the apex from Table 1. Estimates of maximum residence times (Table 4) have been calculated by assuming that there is no input of metals from the seawater entering the apex and that, first, there is no net sedimentation in the apex and, second, half of the dredge spoils accumulate in the sediments (G. Freeland

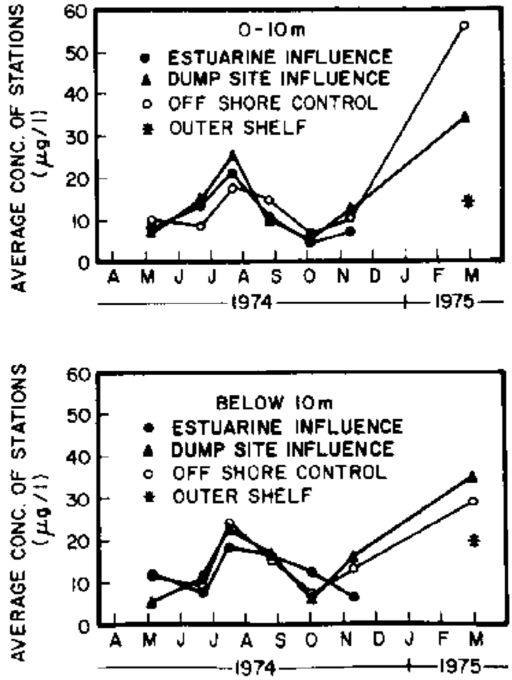

Fig. 30. Seasonal variation of average total dissolved iron in the apex, above and below 10-m depth for estuarine and dumpsite influenced, and offshore control regions, and winter average for outer shelf regime. (*See Figs. 3 and 29.*)

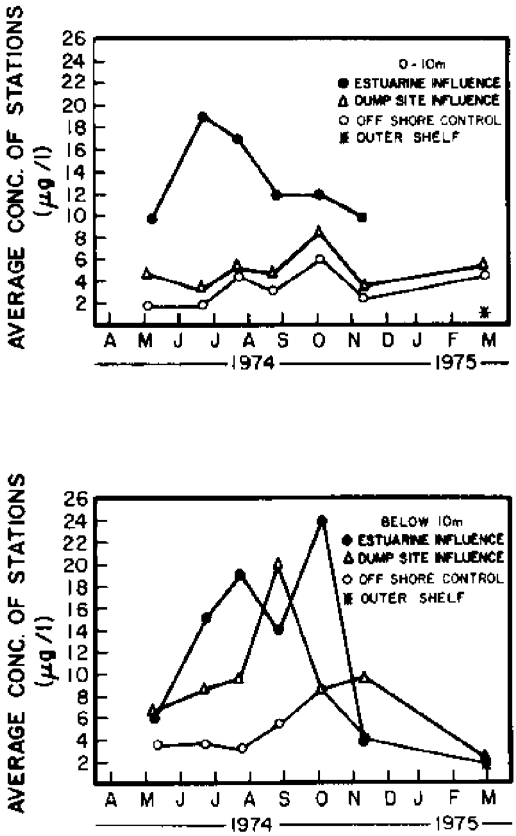

Fig. 31. Same as Fig. 30 for manganese.

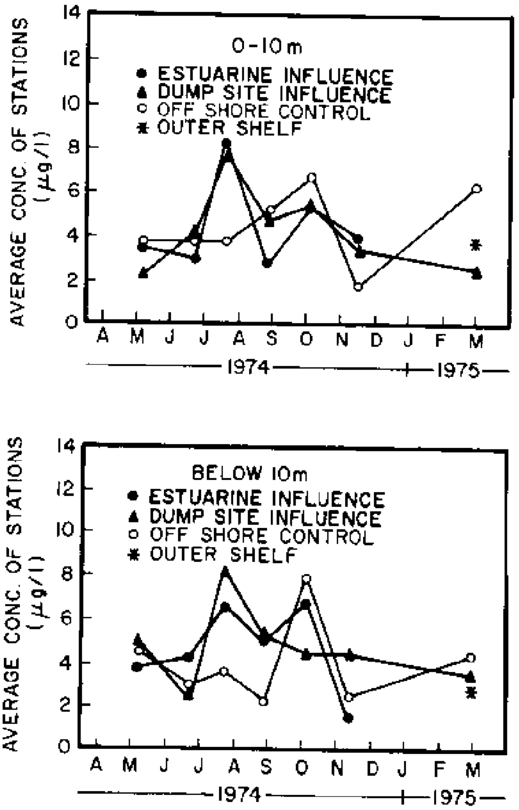

Fig. 32. Same as Fig. 30 for copper.

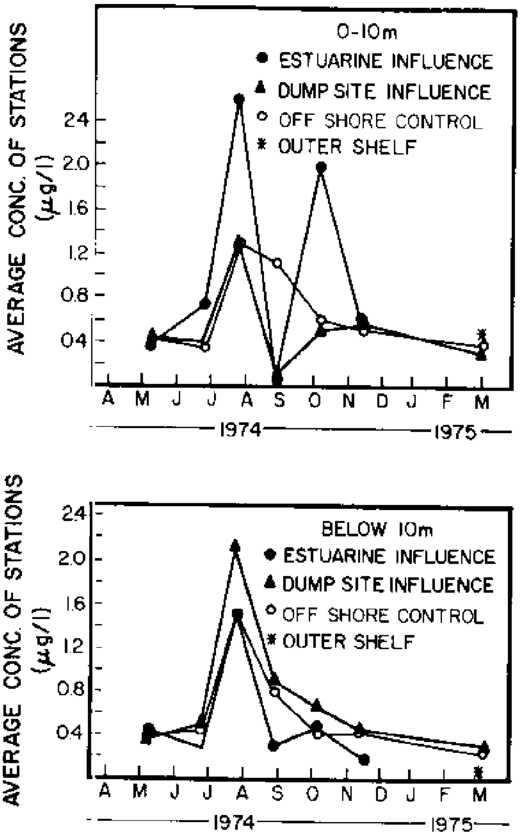

Fig. 33. Same as Fig. 30 for cadmium.

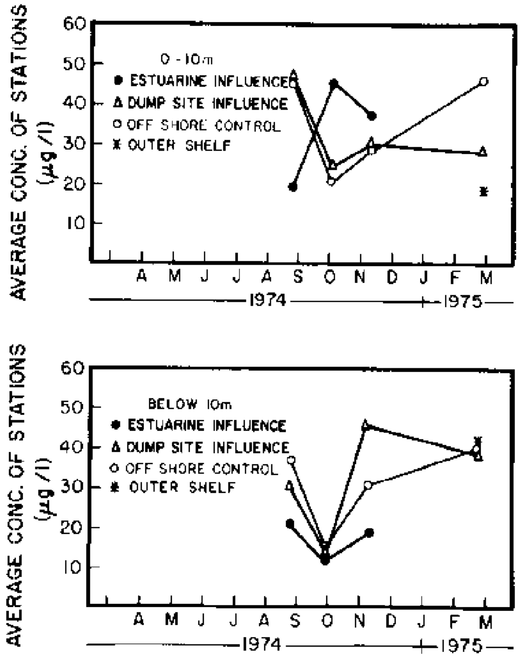

Fig. 34. Same as Fig. 30 for zinc.

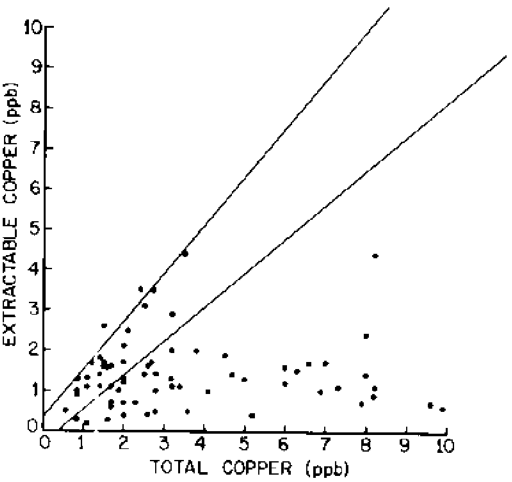

Fig. 35 Dissolved extractable (APDC/MIBK) copper and total dissolved copper in the apex, 21–24 August 1974. Solid lines represent approximate error limits. Samples with values lying between solid lines have ~100% extractable copper. Samples to right of lines have <100% extractable copper.

personal communication) and there is no net sedimentation otherwise. In each case the extreme values of the estuarine input rate from Table 3 have been used to establish separate estimates. The residence times estimated by assuming zero inputs of metals from seawater entering the apex are maximum values as seawater flushing the apex does contain metals. Input rate estimates of metals from the ocean are very imprecise due to poor data concerning the flushing rate of the apex and the concentrations of metals in seawater outside it. Inputs from the ocean probably exceed those from other sources by as much as a factor of three and the true mean residence times of Cu and Zn in the apex, will, therefore, be as much as a factor of four smaller than estimated in Table 4. In any event the true mean residence times of Cu and Zn cannot exceed 6 months and are probably 10-50 days.

Flushing time of apex waters has been estimated from salt balance studies to be about 1 week (Ketchum et al. 1951). Therefore, it appears that contaminants as represented by Cu and Zn are removed from the apex only a little slower than the water column is flushed, and resuspension of sedimentary contaminants and transport of particulate material out of the apex must constitute a very efficient process.

ADDENDUM (4 October 1976): Since the preparation of this paper there have been three reports (Contract Rep. D-75-4, D-76-1, and D-76-7 of the Dredged Material Res. Program, U.S. Army Waterways Exp. Sta., Vicksburg, Miss.) that in laboratory experiments Mn and Cd but not Fe, Cu, and Zn are released to solution during dredged material dispersal and sedimentation in seawater. Re-examination of our data shows that high concentrations of Mn (Figs. 16–19) and Cd (Figs. 23–25), but not Fe, Cu, or Zn, were generally observed in near-bottom water at the station (No. 12) nearest to the designated dredge spoil dumpsite. It is likely that the high concentrations of Mn and Cd at this and adjacent stations are caused by the dumping of dredged material.

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Preliminary analysis of the dispersion of sewage sludge discharged from vessels to New York Bight waters

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Abstract

New York City sewage treatment plant wastes discharged to the New York Bight apex average 2.6% solids with an average solids density of 1.50 g cm^{-3}. Bulk waste density is about 1.009 g cm^{-3}, whereas the density of surface seawater in the sludge dumping area ranges from 1.019–1.025. Solids concentration of the wastes in the sludge vessels ranged from 5 to 50 g $liter^{-1}$. Correlation of extinction coefficient from a 10-cm light-path beam-transmissometer with total suspended matter (TSM) allowed continuous profiling of TSM. STD and beam transmittance profiles were made either by towing the instrument through a sludge patch or wake or by making vertical profiles. Dilution from a near-instantaneous release was on the order of 1,000 within 10 min of release. Dilution in the wake of a release ranged from 500–1000.

The time for TSM to reach background or equilibrium values (0.5–2 mg $liter^{-1}$) depends on initial concentration. Equilibrium time was approached exponentially for well mixed conditions in about 5.5 h. Pycnocline formation in the upper 8 m caused a similar approach to equilibrium time; below that depth TSM increased slightly with time.

Settling velocities for the larger flocculated sludge particles averaged about 0.5–1 cm s^{-1}. Values of 0.01–0.3 cm s^{-1} were obtained from plots of the center of mass of the waste field. The remainder of the dispersing sludge field had velocities of 10^{-3} cm s^{-1} and less.

TSM from New York Harbor can reach the permit area, but oceanographic conditions in the apex usually prevent this.

Relocating the permit area to other deeper areas would cause the affected bottom area to increase in proportion to the increased depth, but concentrations of settled-out material would be inversely proportional, if the oceanographic environment was similar.

The volume of sludge processed at municipal sewage treatment plants is steadily growing due to increased population, increased treatment requirements imposed by legislation, and increased plant efficiency and advanced waste treatment processes. This growing volume has caused disposal problems. Ocean dumping is the solution used in the New York–New Jersey metropolitan area. A summary of the magnitude of the problem, the legislation involved and related EPA intramural and extramural research has been reported elsewhere (EPA 1975) as well as in this volume. An excellent review of the technical aspects of ocean dumping was presented by Kullenberg (1974).

Our report focuses on one aspect of ocean disposal: the dispersion of sewage treatment plant wastes discharged into the New York Bight apex. We present the broad features of our field experiments and relate them to existing and potential disposal practices.

We thank J. Pearce for storage and dock facilities; E. Hansen and E. O. Wagner for making possible the collection of sludge samples and scheduling city vessels; the vessel masters for their cooperation; R. Dewling and others for logistic support; D. Denbo for computer programing; P. Anderson, D. Baumgartner, and P. Larsen for reviewing the manuscript; and M. G. Gross for editorial remarks.

Methods and materials

Sewage sludge discharge—New York City operates four self-propelled vessels which discharge once or twice daily, 6 days a week, excluding holidays, year round, weather permitting. These vessels, plus one commercial vessel (*Susan B. Frank*), were used in the study. Barges out of the metropolitan area also discharge in the area, but their discharge was not studied.

Normally sludge vessels begin dumping as soon as they are inside the dump area;

they make a U-turn while completing their dump inside the area en route to the harbor. The dumping permit generally requires discharge over a distance of at least 5 nmi. There are two classes of vessels (Table 1) operating, characterized by the volume of sludge handled. Initially, the sludge is discharged about 4.5 m below the surface; as it empties, the vessel rises about 2 m. Vessels are double-hulled and exit ports are flush with the inner and outer hulls. Some solids can accumulate in the lower part of each tank during loading and in transit to the dump area.

Physical oceanography—Of principal concern for dispersion of suspended matter from sewage vessels is the structure of the pycnocline and current. Charnell et al. (1975) showed a clockwise gyre in the apex centered in the vicinity of the sludge dump site. This would indicate an area of maximum accumulation because of sluggish net circulation. Nearshore currents flow eastward along Long Island; there is a tendency for southerly transport along the New Jersey coast facilitated by freshwater runoff from New York Harbor. These features are also found in a numerical circulation model of the bight (Laevastu et al. 1974).

Representative currents in the apex near the sludge dumping grounds are about 5 cm s^{-1} to the northeast and swing to the east along the coast of Long Island. Haight (1942) showed 5–10 cm s^{-1} easterly surface currents at the Ambrose Channel lightships. These stations are inshore of the dumping ground and probably under the influence of harbor outflow.

The degree of stratification determines the penetration depth for suspended matter. Heavier solid particles will settle out rapidly and, depending on their density and size, become permanently incorporated in sediments or undergo scour and periodic resuspension.

Table 1. Vessel characteristics.

Vessel	Capacity (m^3)	Normal load (m^3)	No. of ports	Port diam (cm)
Bowery Bay	1,840	1,700	12	35.6
Owl's Head	1,870	1,700	12	35.6
North River	3,030	2,775	16	43.2
Newtown Creek	3,115	2,775	16	43.2

An approximate value of the depth, *Dm*, of wind mixing in the apex is $Dm \simeq 10\ W$, where W is wind speed in meters per second. Thus, little wind is required to deepen the wind-mixed layer to the shallow depths of the dump site ($\simeq 22$ m). For this reason, one rarely sees a single shallow, well developed wind-mixed layer in the apex; rather, there is usually either no pycnocline or a gradual change of temperature, salinity, and density downward to the depth of the first maximum density gradient.

Field experiments—We used New York City sludge vessels to study conditions of normal vessel procedure and some special operations. Stationary and underway experiments were performed. Quasi-instantaneous point source discharges were monitored with the discharge occurring from one tank or from all tanks simultaneously (Table 2). Some experiments were initiated on the scene owing to vagaries in the weather or modified because of instrumentation difficulties. All experiments except those in October 1975 were carried out by EPA personnel aboard the RV *Atlantic Twin*. The October experiments were performed from the RV *Sub Sig*. The field and laboratory procedures and data are detailed in volumes 1–3 of an unpublished data report (*Dispersion of sewage sludge discharged from sludge disposal vessels into New York Bight*).

An InterOcean STD fitted with a 10-cm baffled light transmissometer was used to monitor in situ concentrations of TSM. Calibrations were made using sludge collected from various sewage treatment plants by placing the instrument in a 50-gallon salt water solution, adding sludge in increasing concentrations, and carrying out the resulting regression as TSM in mg $liter^{-1}$ versus extinction coefficient in m^{-1}. Field calibrations were made by obtaining steady transmission readings and drawing off samples for gravimetric analysis.

In the stationary phase the STD was raised and lowered in the control or sludge field. A garden hose secured to the STD at the same level as the transmissometer was used to collect water samples. After lowering the instrument package and clearing the hose, a sample was drawn off and placed in one or more Cubitainers for analysis of suspended matter. Samples were either filtered (Millipore HA 0.45 μ) on board within 2 h of collection or preserved in Lugol's solution and air-mailed to Corvallis, Oregon, for analysis. Forty-eight duplicate samples show that on average the preserved samples were 1.8% higher in TSM than the unpreserved. The results were within the experimental error of the gravimetric analyses and we conclude that preservation and later analysis did not affect this particular measurement.

Underway sampling through an instantaneous release patch or in the wake of a continuous release was done by towing the instrument about 2–3 m outboard of the vessel to lessen interference. A depressor was attached to the STD and towing was done either at one depth or by raising and lowering the probe through the waste field to provide a sawtooth sampling profile. A Motorola Mini-Ranger with three transponders was used for continuous positioning (±5 m at the distance to the dumping grounds); data were recorded on tape.

Paired reversing thermometers were used at two depths for STD field calibration; salinities were drawn off at the same depths for salinity calibration. Salinity was determined in the laboratory with a model 6230 Plessey salinometer.

Before an operation, a buoy was set in the center of the permit area, and a Marine Advisers Q-15 or Aanderaa current meter was installed beneath it. Progressive vector diagrams of currents were obtained covering the whole of an individual field operation. Current meter data were also obtained vertically at 2.5-m intervals with a Hydro-Products 460A/465A or Marine Advisers Q-15 current meter fitted with an extensible fin.

Results

Sludge characteristics—Before a vessel departed, a 5-gallon carboy was partially filled with a sample of sludge which was later sent to Corvallis for analysis. Samples were also processed for physical characteristics in the EPA Region 2 laboratory. Aside from an accumulation of gas in the shipped sample, little difference could be detected and we assumed that shipping and cold storage had little physical effect on the sample. Table 2 shows sludge characteristics.

The volume of sludge discharged from a vessel during a particular experiment ranged from 400–2,890 m^3. The smaller value was a single tank discharge. Solids in the wastes ranged from about 0.5–5%; thus the maximum amount of solids in a fully loaded ship would be about 150 tonnes.

Initial average dilutions of 1,000 reduce shipboard solids concentrations of 5–50 g liter^{-1} (Table 2) to 5–50 mg liter^{-1}. Subsequent dilution determined the duration of a given experiment since background levels are approached at times proportional to initial concentration.

Solids density ranged from about 1.12–1.75 g cm^{-3} with a mean value of 1.50. We were unable to detect the initial impact of settling out by the largest particles. Since it took us at least 6 min after discharge started to begin sampling the patch of an instantaneous dump, any larger particles may have already settled out. Average depth in the permit area is ≃22 m; if settling occurred within 5 min, say, this would require (for seawater at 10°C, 33‰) particle sizes of 886 and 493 μ for Stokes' settling, uncorrected for shape or wet density, at particle densities of 1.25 and 1.75 g cm^{-3}. Neither the settling velocity required (7.3 cm s^{-1}) nor the particle sizes indicated are realistic, hence we assume that the larger particles were not detected. I. Duedall (personal communication) found settling rates of about 1 cm s^{-1} by monitoring ammonia concentrations near the bottom of an instantaneous, stationary release.

Bulk density of sludge was always less than seawater density (Table 2) in the disposal area. Bulk densities averaged about 1.0090 g cm^{-3}, ranging from 1.0007–1.0181. This limits initial mixing to the upper levels of the water column. Fibers and other suspended matter will then be transported hor-

Table 2. Summary of field experiments and sludge characteristics.

	Vessel	Plant	Type of dump	Sludge vol (m³)	Dump duration (min)	Solids* concn (g/liter)	Bulk density (g/cm³)	% solids (by wt)	Solids* density (g/cm³)	Seawater density at dumptime (g/cm³) 0 m	5 m	10 m	Remarks
27 Jun 74	*Newtown Creek*	Unknown	Underway, normal speed† and rate	2,800	≃25	Not analyzed				1.0216	1.0220	1.0224	Density gradient 0–6 m (E-2.5E-4)‡ Wind 12 knots
1 Jul 74	*Owl's Head*	Unknown	Underway, normal speed and rate	1,800	~25	Not analyzed				1.0205	1.0223	1.0229	Density gradient to 6 m (E-3, 5E-4) Wind <5 knots
18 Dec 74	*Newtown Creek*	Bowery Bay	Drifting, fast rate	2,800	6	22.2	1.0095	2.2	1.75	1.0252	1.0252	1.0252	Isopycnal Winds to 25 knots
19 Dec 74	*Newtown Creek*	Newtown Creek	Drifting, fast rate	2,800	6	25.4	1.0106	2.5	1.72	1.0237	1.0241	1.0253	Density gradient 3–13 m (E = 1.7 E-4) Winds to 20 knots
20 Dec 74	*Newtown Creek*	Ward's Isl.	Drifting, fast rate	2,800	6	Not analyzed							Density gradient 8–13 m (E = 2.3 E-4)
5 May 75	*Owl's Head*	Owl's Head	Underway, dead slow,† fast rate	1,810	12	50.2	1.0181	4.9	1.56	1.0251	1.0251	1.0251	Water column mixed to 11 m then gradient to bottom (E = 4 E-5) Wind 20 knots
6 May 75	*North River*	Unknown	Stopped—fast rate thru single port	400	6	Not analyzed				1.0252	1.0252	1.0252	Density in water column monotonically decreasing (E = 7 E-6) Winds 15–25 knots
8 May 75	*North River*	Newtown Creek	Stopped—fast rate thru single port	400	5	27.5	1.0095	2.7	1.53	1.0251	1.0251	1.0251	Density in water column monotonically decreasing (E=7 E-6) Winds 15–25 knots
8 May 75	*North River*	Newtown Creek	Stopped—fast rate thru single port	2,490	6	27.5	1.0095	2.7	1.53	1.0251	1.0251	1.0251	Density in water column monotonically decreasing (E = 7 E-6) Winds 15–25 knots
9 May 75	*North River*	Newtown Creek	Underway, slow,† fast rate	2,890	8	22.3	1.0062	2.2	1.38	1.0239	1.0249	1.0252	Density gradient to 7 m (E = 1.5 E-4) Winds 6 knots
15 Jul 75	*Owl's Head*	Coney Isl.	Stopped	1,810	12	44.7	1.0168	4.4	1.60	1.0205	1.0207	1.0221	Density gradient 7–13 m (E = 4.5 E-4) Winds <10 knots

Table 2. Continued

	Vessel	Plant	Type of dump	Sludge vol (m³)	Dump duration (min)	Solids* concn (g/liter)	Bulk density (g/cm³)	% solids (by wt)	Solids* density (g/cm³)	Seawater density at dumptime (g/cm³) 0 m	5 m	10 m	Remarks
16 Jul 75	*Owl's Head*	Owl's Head	Stopped	1,810	16	31.5	1.0125	3.1	1.66	1.0206	1.0211	1.0227	Density gradient 6–11 m (E = 4 E-4) Wind <3 knots
16 Jul 75	*Susan B. Frank*	Unknown	Underway, normal speed and rate	Unknown	≃25	Not analyzed				1.0195	1.0212	1.0224	Density gradient 0–6 m (E = 2.6 E-4) Wind <3 knots
18 Jul 75	*Owl's Head*	Owl's Head	Stopped	1,810	15	33.3	1.0101	3.3	1.43	1.0211	1.0216	1.0234	Density gradient 6–10 m (E = 4 E-4) Winds <7 knots
19 Jul 75	*Newtown Creek*	Newtown Creek	Stopped	2,830	11	27.7	1.0055	2.8	1.25	1.0190	1.0216	1.0234	Double thermocline with maximum gradient 0–8 m (E=5 E-4) Winds 12 knots
14 Oct 75	*Owl's Head*	Owl's Head	Stopped, normal rate	1,810	15	22.9	1.0085	2.3	1.59	1.0216	1.0220	1.0221	Water column well mixed with density gradient 12–18 m (1.6 E-4) Winds <3 knots
15 Oct 75	*Owl's Head*	Owl's Head(?)	Underway, slow speed, normal rate	1,530	15	Not analyzed				1.0210	1.0221	1.0222	Three-layer density structure Winds 3–5 knots
15 Oct 75	*Newtown Creek*	Ward's Isl.	Underway, slow speed, normal rate	2,830	30	6.7	1.0007	0.7	1.12	1.0210	1.0221	1.0222	Three-layer density structure Winds 3–5 knots
15 Oct 75	*North River*	26th Ward	Underway, slow speed, fast rate	2,890	7	Not analyzed				1.021	1.0221	1.0222	Three-layer density structure Winds 3–5 knots
16 Oct 75	*North River*	Newtown Creek	Underway, normal speed fast rate	2,890	10	24.3	1.0070	2.4	1.40	1.0208	1.0214	1.0219	Three-layer density structure Winds 12–22 knots
17 Oct 75	*Newtown Creek*	Hunt's Point	Underway, normal speed	2,890	9	5.2	1.0019	0.52	1.57	1.0216	1.0215	1.0233	Density gradient 6–10 m Wind 10–12 knots

* Computed.

† Normal speed is about 10 knots, slow speed about 5, and dead slow about 2.5.

‡ $\frac{1}{\rho}\frac{\partial\rho}{\partial z} = (\text{E-2, 5E-4}) = 0.01\text{–}0.0005\ \text{m}^{-1}$.

izontally with the surface currents after a period of initial vertical mixing.

Three density structures prevailed: well-mixed conditions where the pycnocline converged with the bottom, single pycnoclines, and double pycnoclines. The relative strength of these is indicated in some degree by the density gradient, expressed as $(1/\rho)(\partial\rho/\partial z)$. As examples, the density gradient ranged from 0 on 18 December 1974 to 0.0004 m^{-1} in the upper pycnocline and 0.0006 m^{-1} in the lower pycnocline on the following day. On 5 May 1975, the gradient was 0.0003 m^{-1} in the upper 4 m to 0.00003 m^{-1} in the 4–8-m layer, decreasing to 0 below 8 m.

Background total suspended matter—Concentrations (>2 mg $liter^{-1}$) of suspended matter are high in New York Harbor waters. Under the right wind, tide, and runoff conditions, this material can move offshore but it is usually confined nearshore and directed south along the New Jersey coast (Fig. 1).

Summer concentration of TSM at the harbor entrance is about 3 mg $liter^{-1}$, decreasing exponentially to 1 mg $liter^{-1}$ about 37 km offshore; there is an indication of a high pocket (>2 mg $liter^{-1}$) to a depth of 10 m in the sewage grounds (station "S") to Ambrose which might be the residue from a sludge vessel. During winter, concentrations decrease from 5 mg $liter^{-1}$ to 1 mg $liter^{-1}$ in 16 km; once again a high is present at station "S". These two transects, then, do not clearly show a harbor influence in the vicinity of the dump site.

Schubel and Okubo's (1972) model for suspended sediment dispersal was used to determine the approximate conditions necessary for such an event. They solved the two-dimensional transport-diffusion model $U\partial S/\partial x = K\partial^2 S/\partial z^2 - W_s\partial S/\partial z$, where S is suspended material concentration, U is a composite velocity including advection and diffusion, K is a vertical eddy diffusion term, W_s is a settling velocity, and x, z are the horizontal and vertical coordinates, with $x=z=0$ being the surface source location at the harbor entrance. The solution is in terms of dimensionless vertical sections of suspended material for a steady state point source at $x=z=0$. The solution is easily programed, and we modified it to incorporate a mean depth of 30 m rather than 50 as used by Schubel and Okubo.

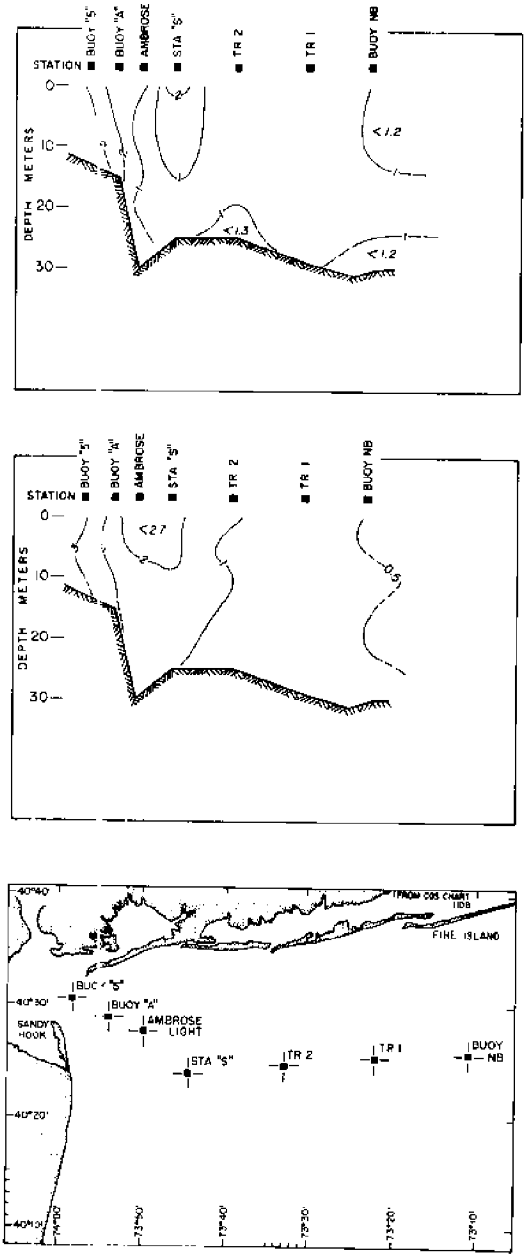

Fig. 1. Winter (26 February, 2 March 1974, upper) and summer (30 June 1974, center) profiles of TSM (mg $liter^{-1}$). Station locations, lower panel.

Figure 2 shows four simulations, all with $W_s = 0.001$ cm s^{-1}. Horizontal velocities of 2 and 5 cm s^{-1} and vertical diffusion coefficients of 1, 5, and 10 cm^2 s^{-1} were used. The parameter L is the distance a particle will travel before hitting bottom at depth $H (L = UH/W_s)$. Since station "S" is only 24 km from the harbor entrance, these solutions show that the harbor-originated particles could easily pass through the dumping area before settling from the water column. The values are those anticipated on a long term average. The suspended solids in the sludge grounds that are of harbor origin are likely to be in extremely low concentrations as shown by our computations; the prevailing currents and dilution processes enroute to the grounds are the causes.

Stationary discharges—Different sludges can have different optical properties (Fig. 3). During the October 1975 cruise the sludges sampled were quite similar. Figure 4 compares the field calibration points with those in the laboratory on the samples of October sludge. Regressions were run on the field samples by (1) using all data points and (2) using all data points and forcing through zero. The second method

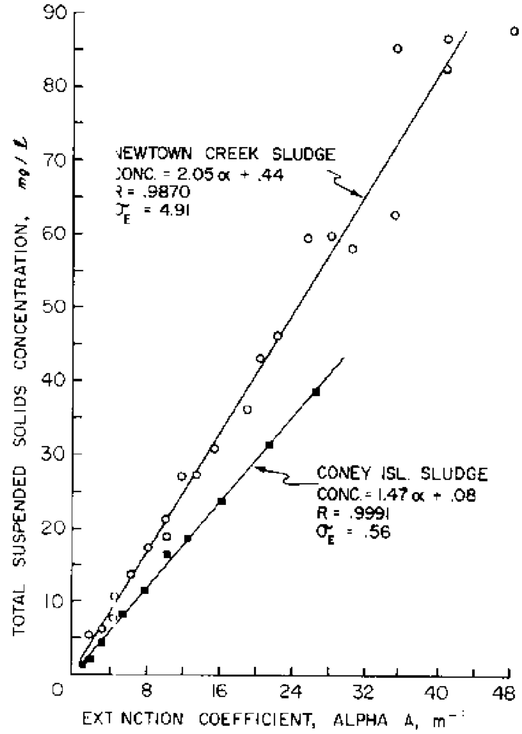

Fig. 3. Laboratory TSM vs. transmissometer extinction coefficient.

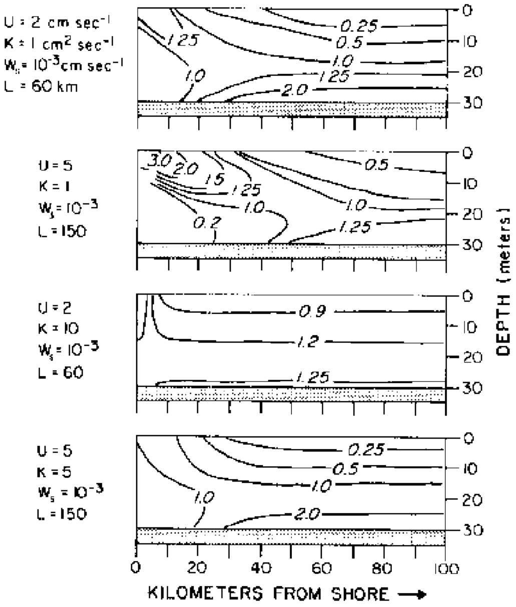

Fig. 2. Dimensionless vertical sections of suspended material. Permit site centered at 24 km.

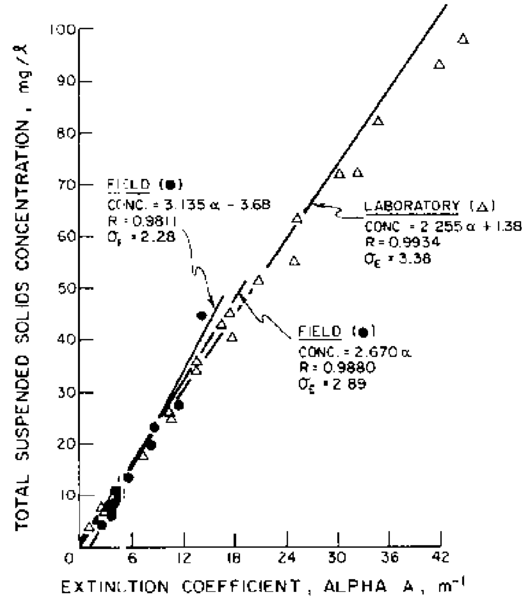

Fig. 4. Field and laboratory correlation between extinction coefficient and TSM 16–17 October 1975.

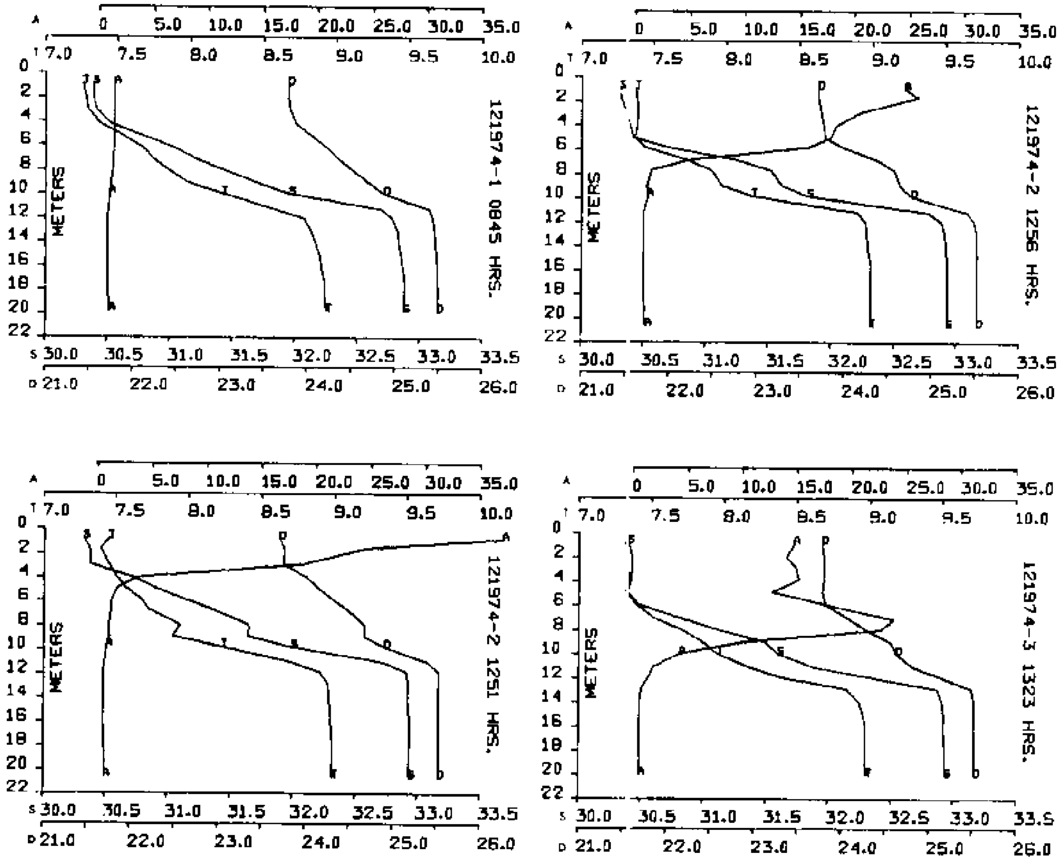

Fig. 5, parts 1–4.

Fig. 5. Salinity, S(‰), temperature, T (°C), sigma-t, D, and extinction coefficient, A (m^{-1}), before and after sludge release, 19 December 1974. (Fig. 5 continued on next page.)

showed a greater correlation (*R*) but larger standard error (σ_e). When the possibilities for field sampling error in a patchy field are considered, the laboratory and field results are in good agreement, hence, continuous profiles of light extinction provide a reasonable estimate of TSM.

Variations between sewage treatment plants result from differences in the sewage received and differing treatment processes. These differences are also shown by the physical properties of the sludge materials (*see Table 2*). Swartz et al. (in press) discussed this variability with respect to the toxicity of wastes from Bay Park to benthic organisms.

Vertical profiles (Fig. 5) of a *Newtown Creek* sludge release (2,800 m^3 in 6 min) on 19 December 1974 show the extinction coefficient, *A*, calculated as $A = (-1/d) \ln (\%T/100)$ where *d* is transmissometer path-length and %*T* is percent transmission. The 0845 hours panel shows the control taken 3.5 h before dumping began. There is some evidence of an internal wave oscillating about 12 m with a 2–3-m amplitude. The density profile shows an initial weak upper pycnocline to about 8 m with a near-linear increase to 12 m and a sharp breakoff to isopycnal condition below 12 m. A gradual indication of a strong double pycnocline reaching maximum development is shown in the 1338 hours panel.

On 18 December there was intense wind mixing and no pycnocline; *Newtown Creek* sludge was discharged in the same volume at the same rate. Thirty minutes after dis-

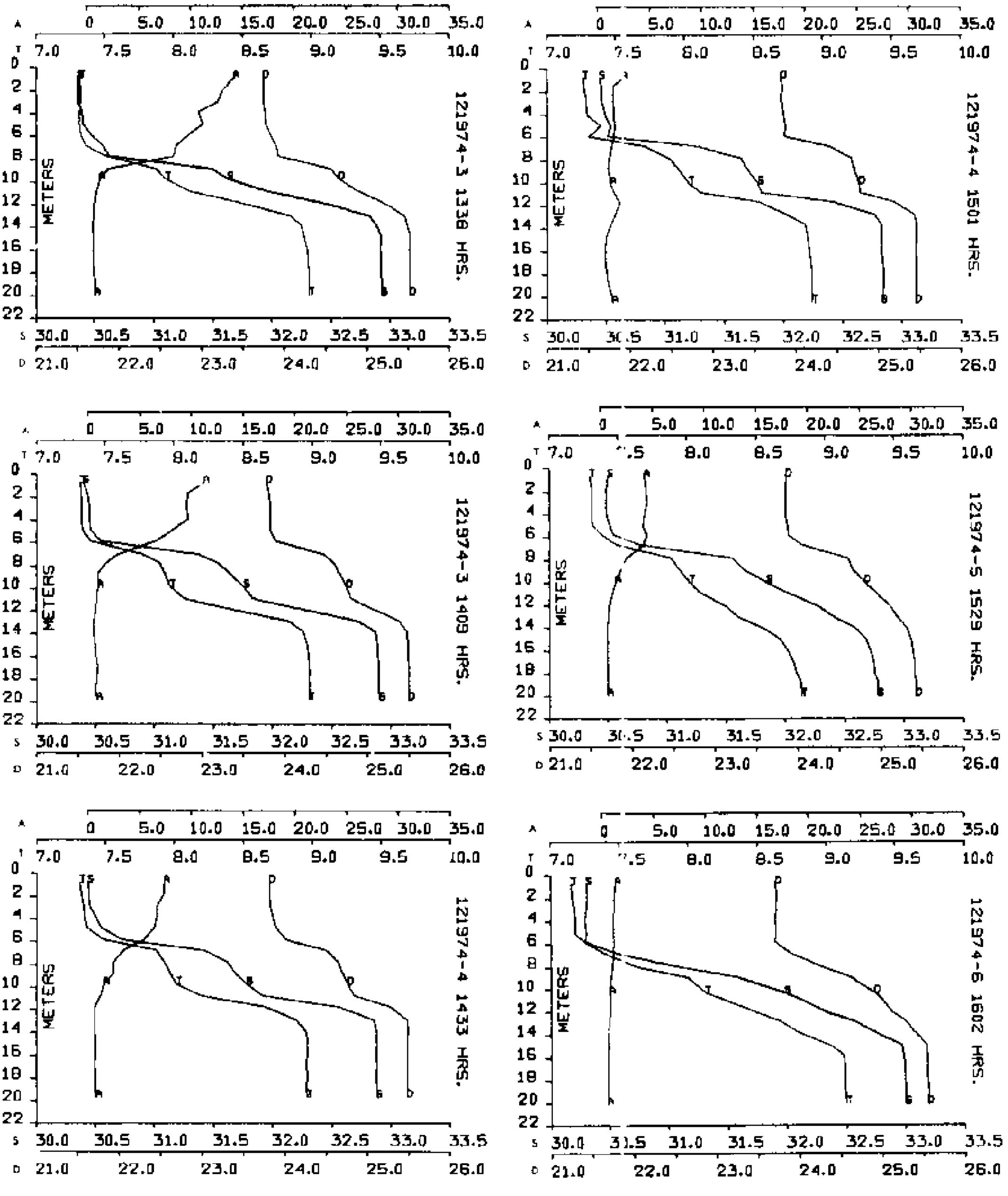

Fig. 5, parts 5–10.

charge the wastes were diffused throughout the water column (Fig. 6).

Figure 7 summarizes extinction coefficient averages over various parts of the water column. On 18 December the entire water column was used. The decrease with time is approximately as $A = Ao$ exp $(-0.00585t)$, where $Ao = 8.6$ m^{-1} is the average coefficient at $t = 0$, the first sampling. Time to background $A(= 1.24$ m$^{-1})$ is, then, about 5.5 h. On 19 December the slopes of the 0–8-m, 0–12-m, and 0-bottom average extinction coefficients parallel each other and decrease approximately as $A =$

$Ao\ \exp(-0.0078t)$. Time to background in the entire water column ($0.9\ \mathrm{m}^{-1}$) was about 5 h. In the upper 8 m the decrease is as $\exp(0.008t)$ with time to background ($1.15\ \mathrm{m}^{-1}$) about 5.5 h. The 9–12-m layer, somewhat below the first pycnocline, shows above background levels at all times except for $t = 0.32$ h. In the 13–19-m layer, beneath the main pycnocline, slightly above background levels are present on occasion indicating some accumulation there. There is a suggestion of an increase in A with time.

Rapid mixing on 18 December indicates a very high vertical diffusion coefficient. Defining an equilibrium time, Te, of about 1 h and assuming $K = h^2/Te$, where h is

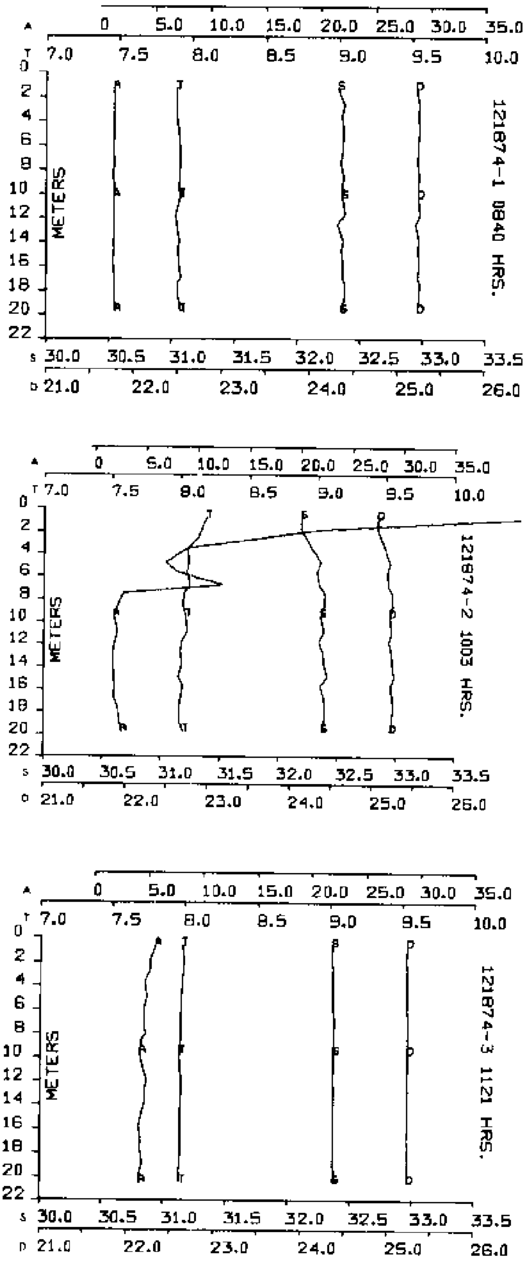

Fig. 6. Same as Fig. 5, 18 December 1974.

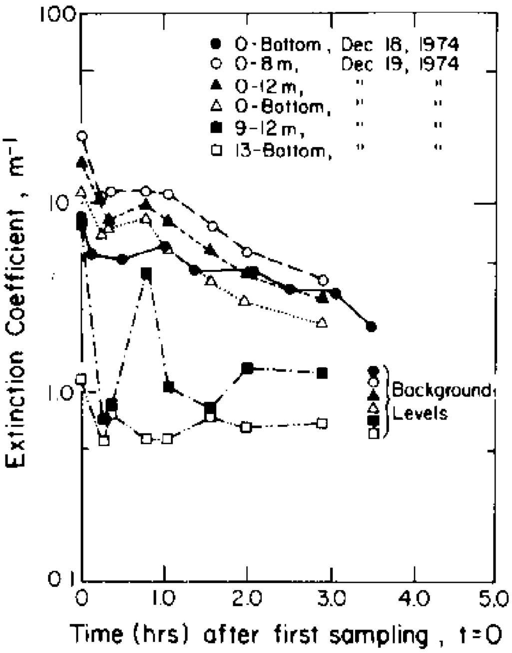

Fig. 7. Variation of extinction coefficient with time, 18–19 December 1974.

water column depth, we find a value of about 10^3 cm^2 s^{-1}. Wind stirring occurred the following day, but it was not as great. Equilibrium time, as indicated by successive similar profiles (but decreasing concentrations), was reached in about 80 min in the upper 10 m, giving $K \simeq 210$ cm^2 s^{-1}. Both of these values are large but not inconsistent with values computed from time-series salinity profiles as

$$K = \left(\frac{1}{\partial s/\partial z}\right)\int_0^z \frac{\partial s}{\partial t}\,dz.$$

On 8 May 1975, a single-port dump (400 m^3) was made. Before discharge there was a gradual increase in density (as sigma-t) from the surface ($D = 25.11$) to 17 m ($D = 25.22$); conditions were near isopycnal below 17 m. The extinction coefficient in the upper layer was uniform, about 0.48 m^{-1}; at 17 m it increased linearly from 0.53 to 0.70 m^{-1} at 22 m then to 0.81 m^{-1} at 23 m. At 9 min after discharge, the plume penetrated to 5 m, to 15 m after 12 min, and to 18 m after 17 min. After 23 min there was no detectable material below about 18 m. Most of the detectable suspended matter remained in the upper 10 m. A sinking velocity of $\simeq 6$ cm s^{-1} is indicated from time 9 to time 12 min, and $\simeq 1$ cm s^{-1} from time 0–9 and time 12–17 min.

Settling velocities determined in this manner are approximate but probably of the correct order of magnitude for larger flocculated particles. By plotting the total distribution with depth, subtracting the background beam attenuation, and determining the center of mass, we obtained velocities of 0.01–0.3 cm s^{-1} for December and May cruises. Figure 8 shows the mass centroids calculated for 18 and 19 December 1974.

Underway discharges—We monitored underway discharges by making tows along the plume at one depth, making continuous sawtooth (with depth) profiles in the wake, or sampling vertically at a fixed location in the wake. The last method was conditioned by weather since the transmissometer could not be towed underway. Data from these tows are still being processed and are not discussed here.

On 5 May 1975, the *Owl's Head* discharged 1,810 m^3 of sludge over a 12-min period maintaining headway at dead slow speed in a 20-knot wind. Waters were well mixed (isopycnal) to 11 m at $D = 25.15$; there was a gentle gradient to $D = 25.28$ below that depth to 22 m. A current meter suspended at 2 m indicated velocities of about 5 cm s^{-1} setting to the south and southwest at the beginning of the experiment. Speeds gradually rose to about 20 cm s^{-1} while shifting to the southwest at the end of the experiment (about 2.5 h later). Initial extinction coefficients were uniform with depth being about 0.35 m^{-1} or a trans-

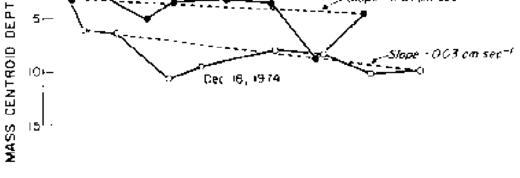

Fig. 8. Position of mass centroid with time, 18–19 December 1974.

mittance of 97%. Eleven minutes after discharge, wastes had penetrated to 6 m indicating a settling velocity of about 0.9 cm s^{-1}. At $t = 45$ min wastes had penetrated to the bottom indicating a settling velocity of at least 0.75 cm s^{-1} since there were no samples taken between $t = 11$ and 45. A transmittance maximum was present between 5 and 9 m at $t = 45$. The remainder of the vertical section showed erratic individual profiles with the penetration depth ranging from 8 m to the bottom. Values of A averaged over the water column ranged from 1.2 to 3.8 m^{-1} with an average of 2.65 against an average control of 0.4.

Figure 9 is a sawtooth horizontal profile through the wake of the plume for a continuous discharge of 2,890 m^3 of *North River* sludge made on 9 May 1975. The length of the field (which runs from right to left) is 515 m at a vessel speed of 0.86 m s^{-1} through the wake; contours are of the extinction coefficient. Several cells are present. The concentrations are computed as TSM $\sim 2A$ and maximum concentration is about 100 mg $liter^{-1}$. The solids concentration in the sludge vessel was 22.3 g $liter^{-1}$, indicating a minimum dilution of about 200. Surface dilutions are greater, in the range of 400–600.

Discussion

Hatcher and Keister (1976) reported that the major source of organic matter in apex bottom sediments is sewage derived, based on total carbohydrate:total organic carbon (TCH:TOC) ratios. They show rather widespread TCH:TOC values of 30–50. The ratios are not centered about the disposal area but, as Hatcher and Keister (1976) showed, there may be an increase of the ratio with time due to TOC loss. Although a sludge release may not be detectable by the methods we used after about 5 h, most of the suspended matter remains in the water and can be widely dispersed.

An approximation of the amount of solids that could accumulate on the bottom can be gained from data in Table 2. With 2.6% solids (by weight), 1.50 g cm^{-3} solids density, and 1.009 g cm^{-3} bulk density, 3,000 m^3 of sludge contains about 52 m^3 of solids. If these wastes settled out evenly in a circle of radius 100 m, a deposit 1.7 mm thick would ultimately form from each release (disregarding bulk expansions caused by bound water, etc.). For a vessel underway at 5.15 m s^{-1} (10 knots) and discharging continuously for 20 min over a beam width of 12 m, about 0.7 mm would accumulate, again assuming an even spread and settling in the prescribed rectangle.

Alternate disposal sites—The question arises as to the physical consequence of moving the disposal site to new areas north and south of the Hudson Canyon. A two-

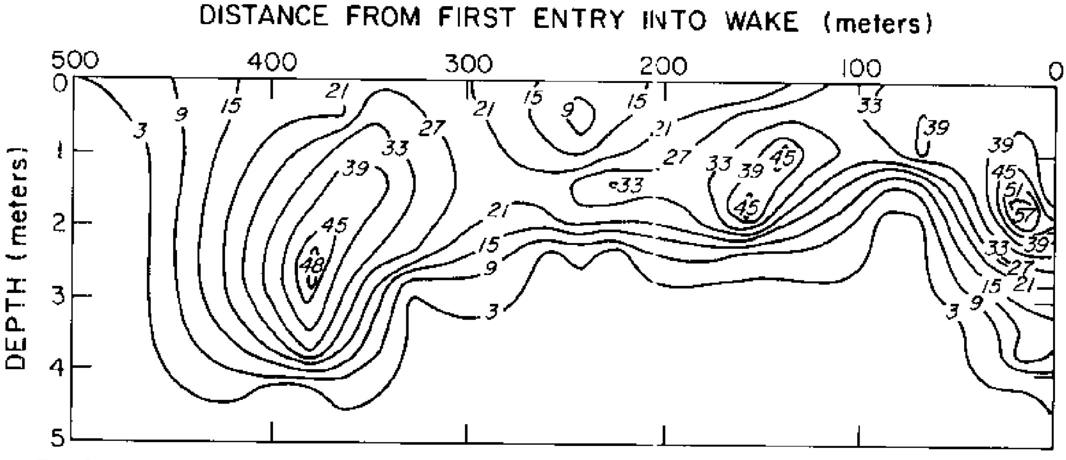

Fig. 9. Extinction coefficient (m^{-1}) vs. depth in wake of sludge vessel (moving from right to left), 9 May 1975. Contour interval 6 m^{-1}. Redrawn from computer plot.

layer model of the bight circulation (Laevastu et al. 1974) suggests that dumping to the south of the Hudson Canyon would result in offshore transport of wastes while onshore transport is indicated for dumping north of the canyon. If we assume a point source, a 10 cm s^{-1} horizontal current, U, and a particle settling velocity, W_s, of 0.1 cm s^{-1}, the size of the area of a newly impacted dumping ground will be proportional to the depth change of the existing ground (depth $= H = 22$ m) to the proposed grounds. Particles will settle out over the length $L = UH/W_s$. The proposed new grounds north and south of the Hudson Canyon have average depths of about 46 and 37 m. For $H = 22$, 46, and 37 m, $L =$ 2.2, 4.6, and 3.7 km. If a circular settling patch is assumed, the north area is 51 km^2 and the south 28 km^2 greater than the area of the existing ground (15 km^2). There will be an accompanying decrease in solids per unit area relative to the existing grounds by a factor of 4.4 and 2.9 for the north and south grounds, if we assume an even distribution within the computed area.

These are, of course, only approximations to be used in assessing the relative effects of relocating the grounds based on depth alone. Increasing or decreasing the settling velocity by an order of magnitude results in an inverse order of magnitude L response.

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Raritan Bay as a source of ammonium and chlorophyll *a* for the New York Bight apex[1,2,3]

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Abstract

Measurements in June 1974 and 1975 in Raritan, Sandy Hook, and Lower New York Bays showed that water nearest Sandy Hook had low salinity and high Chl *a* (10–60 mg/m^3) and NH_4^+ concentrations (10–40 μM). Sandy Hook Bay had the highest Chl *a* values (45–90 mg/m^3) and low NH_4+ concentrations (5–8 μM). High Chl *a* concentrations ($\gtrsim$ 50 mg/m^3) were also found at the center of Raritan Bay accompanied by high NH_4^+ concentrations (60–70 μM). Near Rockaway Inlet, NH_4^+ concentrations were 0–40 μM while Chl *a* values were minimal (3–4 mg/m^3). The Narrows also showed high NH_4+ (15–40 μM) and low Chl *a* concentrations (0–10 mg/m^3).

By comparing the salinity, ammonium, and chlorophyll *a* distributions at the Sandy Hook–Rockaway Point transect with the characteristics of the different sources of water flowing out to the bight, an identification of water masses at the transect is possible.

Of particular importance to the New York Bight are the fluxes of nutrients into the apex between Sandy Hook and Rockaway Point (Fig. 1). O'Connors and Duedall (1975) and Duedall et al. (1976) estimated the quantity of nutrients transported into and out of Lower New York Bay. They did not, however, indicate the relative contribution of the several possible nutrient sources to this flux. High chlorophyll *a* and nutrient concentrations occurred on the south side of the Sandy Hook–Rockaway Point transect. The source of these chlorophyll *a* and nutrient concentrations could not be determined without a more detailed study of the Raritan–Sandy Hook and Lower New York Bay area.

On 5 June 1974, the Sandy Hook–Rockaway Point transect, the Sandy Hook–Staten Island transect, The Narrows, and Rockaway Inlet (Fig. 1) were sampled over one tidal cycle for NH_4^+ and Chl *a* concentrations as well as for temperature and salinity. On 4 June 1975, another cruise obtained NH_4^+ and Chl *a* data throughout the entire Raritan–Sandy Hook Bay system. Here we report the main results of these two investigations.

Methods

The 5 June 1974 cruise used four ships, each sampling an area over a 12-h period. At each area, stations were sampled sequentially at depths of every 3 m, except at station S where surface, bottom, and two middepths were sampled. Samples for salinity, chlorophyll *a*, and ammonium analyses for stations A–H and J–P were obtained with a submersible pump. At stations Q, R, and S, Nansen or Niskin bottles were used. Temperature was measured with a bead thermistor at stations A–H and S, with reversing thermometers at Q and R, and with a thermometer submersed in the pump outflow at stations J–P.

On 4 June 1975, stations 1–16 (Fig. 1) were occupied and sampled at the surface and depths of every 3–4 m. A pumping system again was used to measure temperature and salinity and to obtain samples for ammonium and chlorophyll *a* analyses.

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[2] Contribution 169 of the Marine Sciences Research Center, SUNY, Stony Brook. This paper contains material from a M.S. thesis presented by J.H.P. to State University of New York at Stony Brook, August 1976.

[3] The assistance of the Sandy Hook Marine Laboratory (RV *Rorqual*), the U.S. Army Corps of Engineers (RV *Hatton*), the Environmental Protection Agency (RV *Cleanwater*), and H. Stuebe, Captain of the RV *Onrust*, is gratefully acknowledged. W. Miloski, G. Hulse, S. Oakley, A. Robbins, J. Restivo, and D. Gassaway were also helpful.

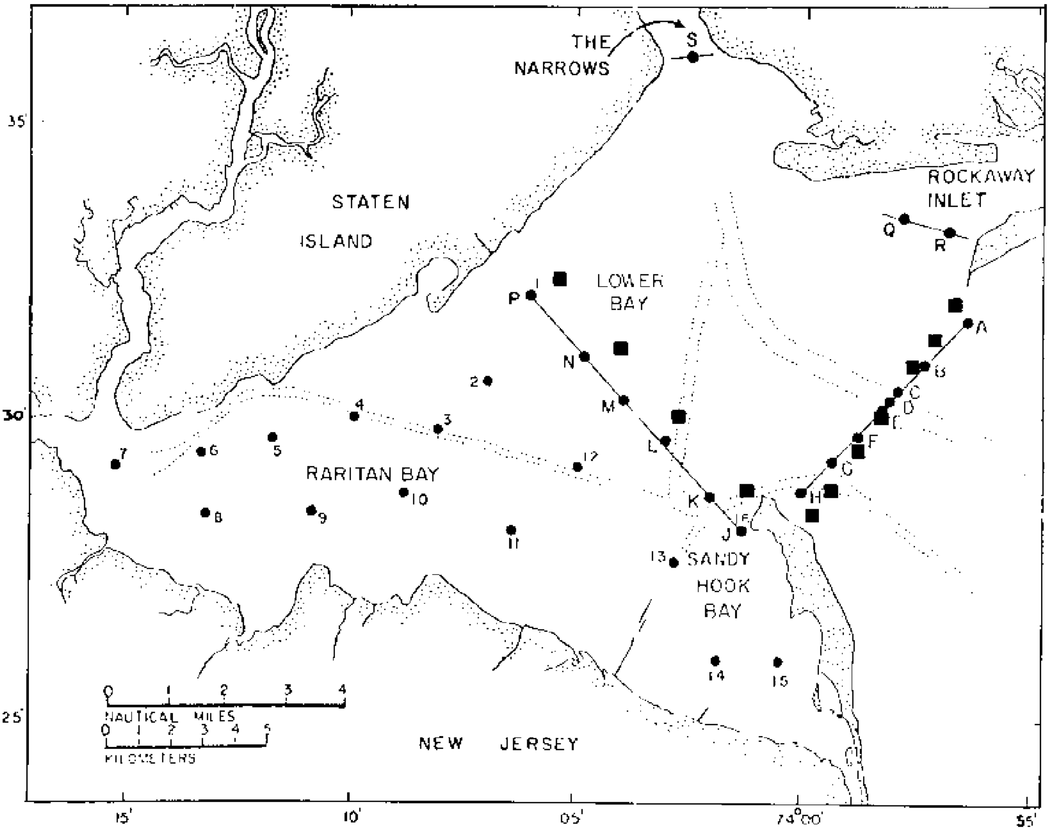

Fig. 1. Station locations for 5 June 1974 and 4 June 1975. Current meter locations—■.

Salinity measurements were made with a Plessey bench salinometer. The concentration of chlorophyll *a* was measured trichromatically using acetone extracts of the pigment (Strickland and Parsons 1972). An ultrasonic probe was used to rupture the cells previously collected on Millipore filters.

Glass fiber filtered (8-μm porosity) water samples for NH_4^+ were collected in 125-ml polyethylene bottles. Immediately upon collection, these samples were stored under ice until they could be analyzed in the laboratory. Concentrations of NH_4^+ from samples collected on 5 June 1974 were determined by the indophenol method (Solórzano 1969) about 20 to 30 h after collection. For the 4 June 1975 study, NH_4^+ was determined using an automated indophenol method.

Results

Vertical sections—Vertical sections for salinity, NH_4^+, and Chl *a* data obtained during the 5 June 1974 cruise at the Sandy Hook–Rockaway Point and Sandy Hook–Staten Island transects (Figs. 2 and 3) depict tidally averaged distributions of water properties. For The Narrows and Rockaway Inlet stations, we present tidally averaged vertical profiles for salinity, NH_4^+, and Chl *a* concentrations (Fig. 4).

At the Sandy Hook–Rockaway Point transect low salinities dominated the south side; higher salinities from inflow of bight water were found near Rockaway Point (Fig. 2). The NH_4^+ concentration distributions were inversely related to the salinity distributions; lower NH_4^+ concentrations were found at station A and higher concen-

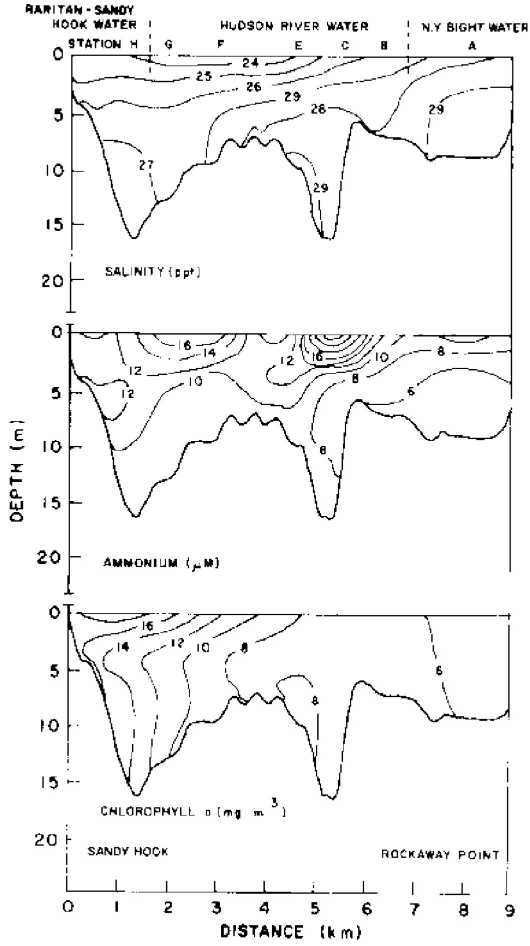

Fig. 2. Vertical sections of tidally averaged salinity, NH_4^+, and Chl *a* concentrations in the Sandy Hook–Rockaway Point transect, 5 June 1974.

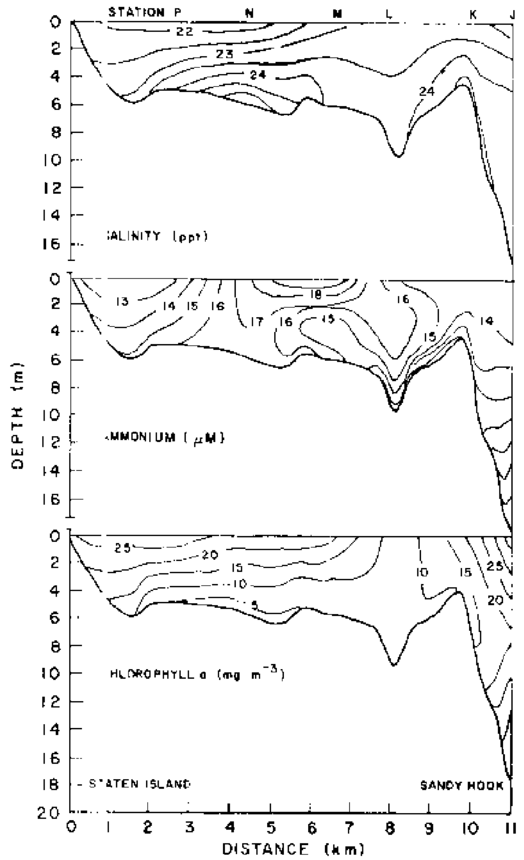

Fig. 3. Vertical sections of tidally averaged salinity, NH_4^+, and Chl *a* concentrations in the Sandy Hook–Staten Island transect, 5 June 1974.

trations at stations C, F, and G. The observed distribution of NH_4^+ results from the flow of bight water into Lower New York Bay near Rockaway Point and the seaward flow of nutrient-rich New York Harbor and Raritan Bay waters in the surface layer near Sandy Hook. The distribution of chlorophyll *a* (Fig. 2) again suggests a flow of bight water, with its low chlorophyll *a* concentrations, through the Rockaway Point side of the transect. The high chlorophyll *a* values found at Sandy Hook indicate that Raritan–Sandy Hook Bay and Lower New York Bay are possible sources of high chlorophyll *a* concentrations for the bight apex.

Kao's (1975) calculations for nontidal current velocities for the Sandy Hook–Rockaway Point transect in 1958 and his section illustrating the net flow of water across the transect (Fig. 5) support the distribution of water properties shown in Fig. 2. For lack of recent current meter data, Figs. 5 and 6 are presented here merely to indicate the general flow characteristics of the area. However, Kao has

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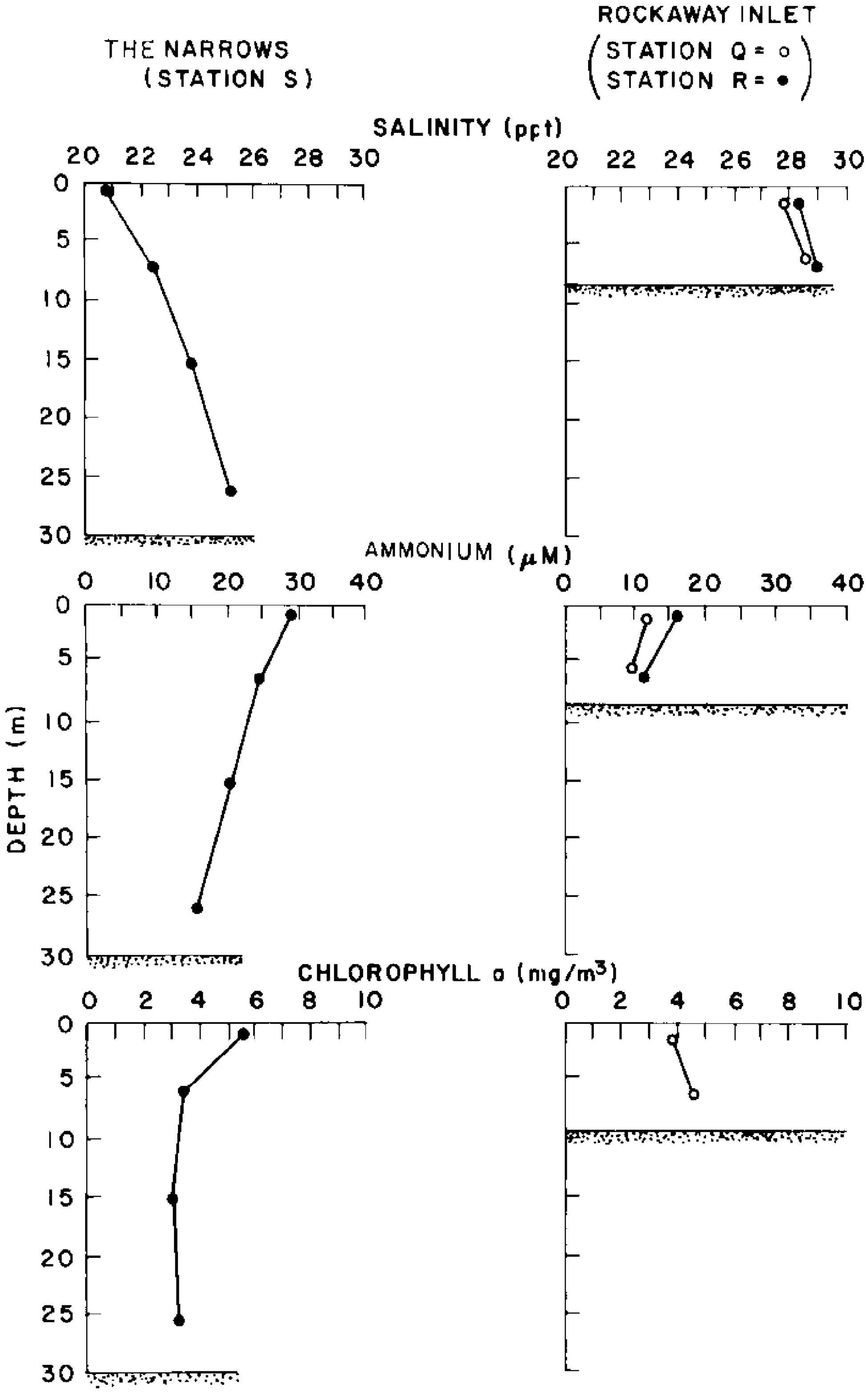
Fig. 4. Vertical profiles of tidally averaged salinity, NH_4^+, and Chl *a* concentrations for The Narrows and Rockaway Inlet, 5 June 1974.


THE NARROWS
(STATION S)
ROCKAWAY INLET
(STATION Q = ○
STATION R = ●)
SALINITY (ppt)
20 22 24 26 28 30
AMMONIUM (μM)
0 10 20 30 40
CHLOROPHYLL a (mg/m³)
0 2 4 6 8 10
DEPTH (m)
0 5 10 15 20 25 30

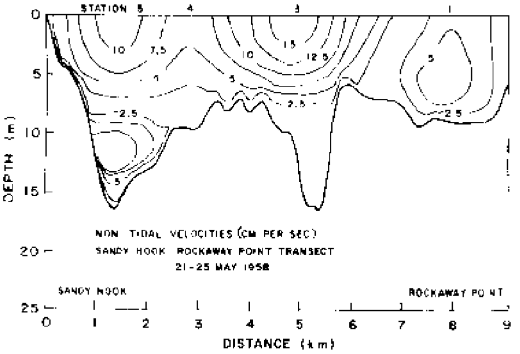


Fig. 5. Nontidal current velocities for the Sandy Hook–Rockaway Point transect. Negative values indicate flow into Lower New York Bay.

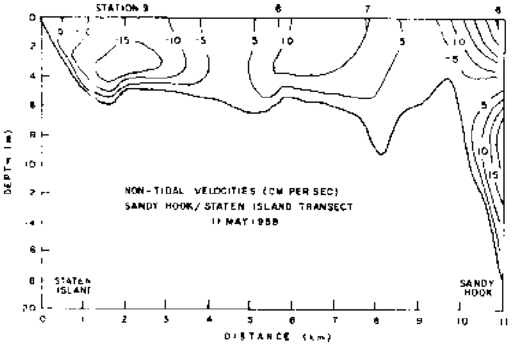


Fig. 6. Nontidal current velocities for the Sandy Hook–Staten Island transect. Negative values indicate flow into Lower New York Bay.

shown that current structures for 1952, 1958, and 1959 are similar and, therefore, it would be reasonable to assume that the 1974 and 1975 nontidal current structures are similar to those shown.

At the Sandy Hook–Staten Island transect, distributions of salinity, NH_4^+, and Chl *a* concentrations (Fig. 3) and a vertical section of nontidal current velocities, calculated from May 1958 C&GS current meter data (Fig. 6), indicate that water leaves the Raritan–Sandy Hook Bay system at both ends of the transect but enters near its middle.

Distribution of water properties in the Raritan–Sandy Hook Bay system—To illustrate the spatial distribution of water properties in the Raritan–Sandy Hook Bay system, surface and bottom contours (Fig. 7) were constructed for salinity, NH_4^+, Chl *a* concentrations, and the freshwater fraction (f). Ketchum (1951) defined f as $(S_s - S_n)/S_n$, where S_s equals the salinity of the source water ($S_s = 30‰$) and S_n equals the observed salinity of a particular water sample. The freshwater distribution shown in Fig. 7 illustrates the passage of fresh Raritan River water into the bay system as well as the freshwater contribution from The Narrows on the ebbtide.

Discussion

The circulation of Raritan Bay is a prime factor in controlling distributions of water properties within the Raritan–Sandy Hook Bay system. This circulation is complicated by freshwater inflow and tidal and nontidal advection. Inflows of freshwater from the Raritan and Hudson Rivers are variable due to tidal and seasonal effects. The net drift southwestward near the middle of Raritan Bay and eastward on the south side suggests the presence of a counterclockwise gyre. The resulting sluggish circulation (Jeffries 1962) allows time for a buildup of NH_4^+ and Chl *a*. This fairly stable pattern is only disrupted by severe storms, after which the earlier circulation is normally reestablished within about 2 days (Ketchum et al. 1951).

Figure 7 shows elevated concentrations of NH_4^+ and Chl *a* near the center of Raritan Bay, corresponding to the area of the counterclockwise gyre. The NH_4^+ and Chl *a* concentrations here are significantly higher than those found in the Sandy Hook–Rockaway Point transect.

Using Ketchum's (1951) modified tidal prism model, we calculated the flushing, or residence, time from the exchange ratio (the proportion of water moved on the ebbtide) for each segment of Raritan Bay (Fig.

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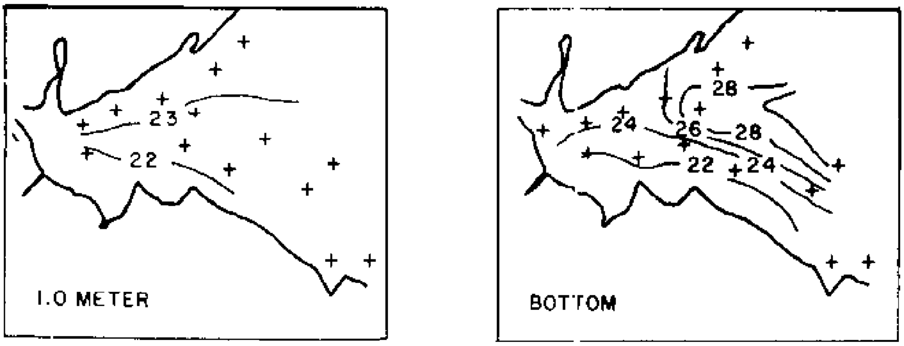
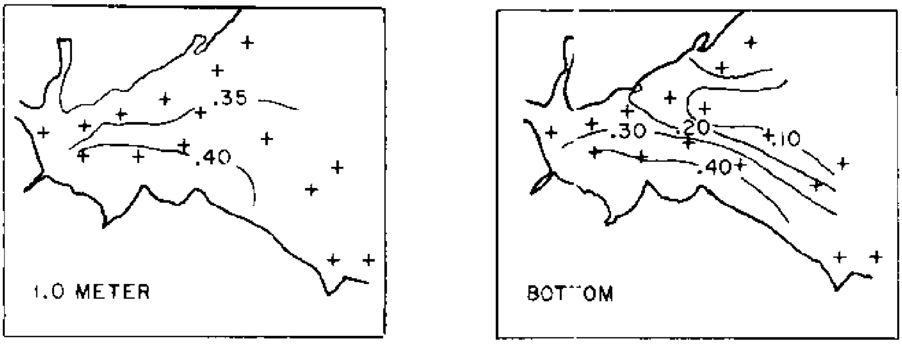
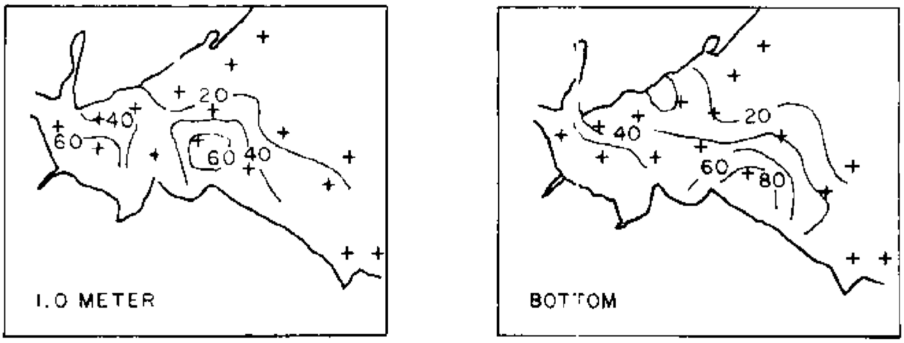
Fig. 7. Horizontal contours for salinity, freshwater fraction, NH_4^+, and Chl *a* concentrations in Raritan–Sandy Hook Bay, 4 June 1975.



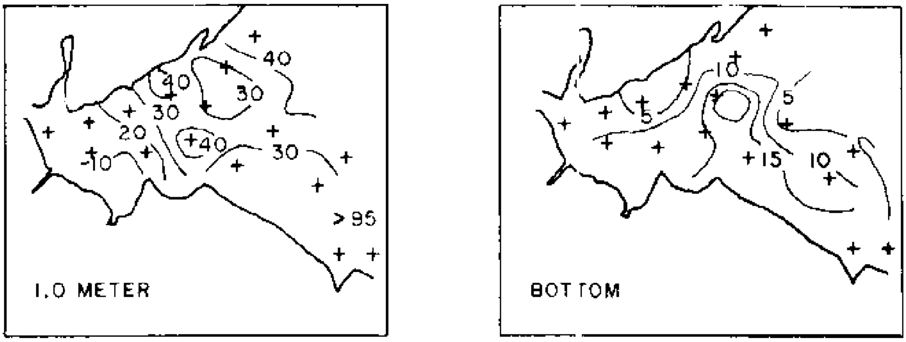
SALINITY (ppt)



FRESH WATER FRACTION



AMMONIUM (μM)



CHLOROPHYLL a (mg m^{-3})

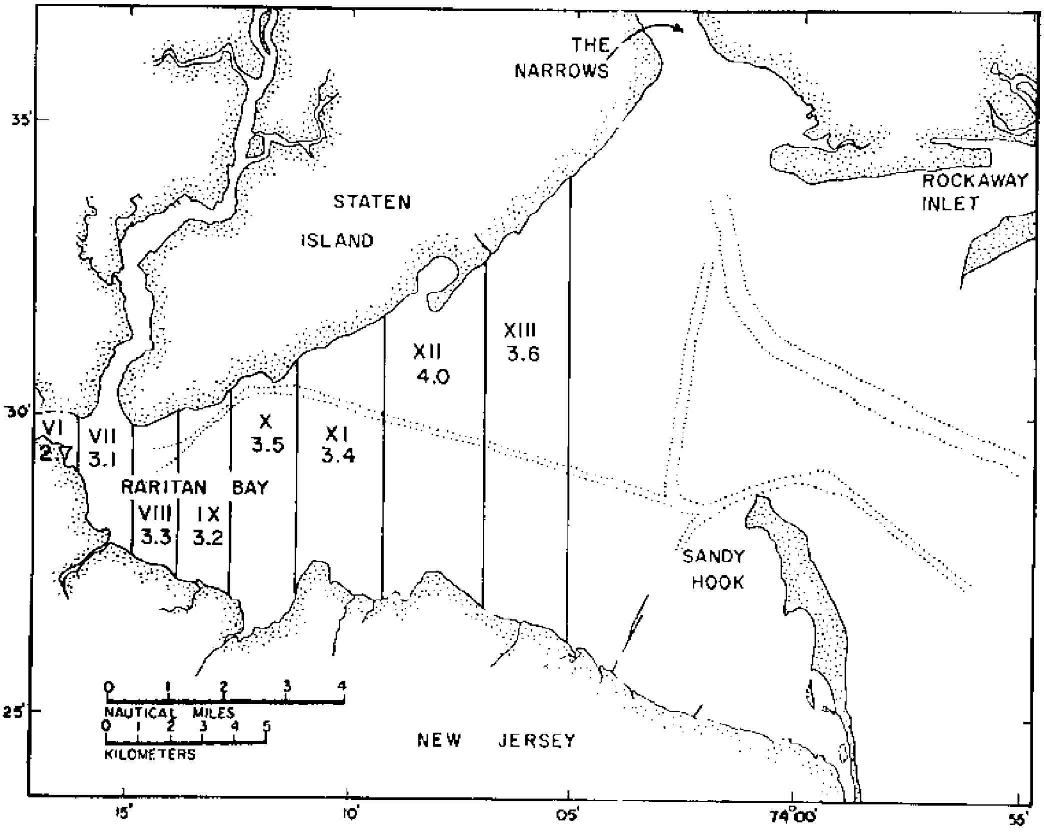


Fig. 8. Segmentation of Raritan Bay (adapted from Ketchum 1951). Number in segment is residence time in tidal cycles.

8). Those segments of Raritan Bay having the longest flushing times correspond well to those regions where NH_4^+ and Chl *a* concentrations are highest. These extended flushing times are evidence of the sluggish circulation mentioned by Jeffries (1962).

A seaward flow of chlorophyll *a* around Sandy Hook is indicated by comparing vertical sections of chlorophyll *a* concentrations (Figs. 2 and 3) and nontidal current velocities (Figs. 5 and 6) for the two transects. The importance of the seaward flow is also evident in the tidal variability of chlorophyll *a* concentrations and salinities at stations J and H (Fig. 9). Maximum concentrations for chlorophyll *a* indicate transport of chlorophyll *a*-rich water around Sandy Hook during periods of strong ebb flow. The corresponding flow of low salinity water at this time supports the hypothesis that the Raritan–Sandy Hook Bay system is a major source of chlorophyll *a* for the bight apex.

It is possible to identify the water masses flowing out through the Sandy Hook–Rockaway Point transect by examining the vertical sections for NH_4^+ and Chl *a* (Fig. 2). High NH_4^+ concentrations present at Ambrose Channel (stations C and E) are accompanied by low Chl *a* values, characteristic of Hudson River water. At station H, high NH_4^+ and Chl *a* concentrations occur, typical of the water conditions in the Raritan–Sandy Hook Bay system. Chlorophyll *a* concentrations of 60 mg/m^3 and greater were observed in Sandy Hook Bay, a strong indication that Sandy Hook Bay may be a major source of chlorophyll *a* to the apex.

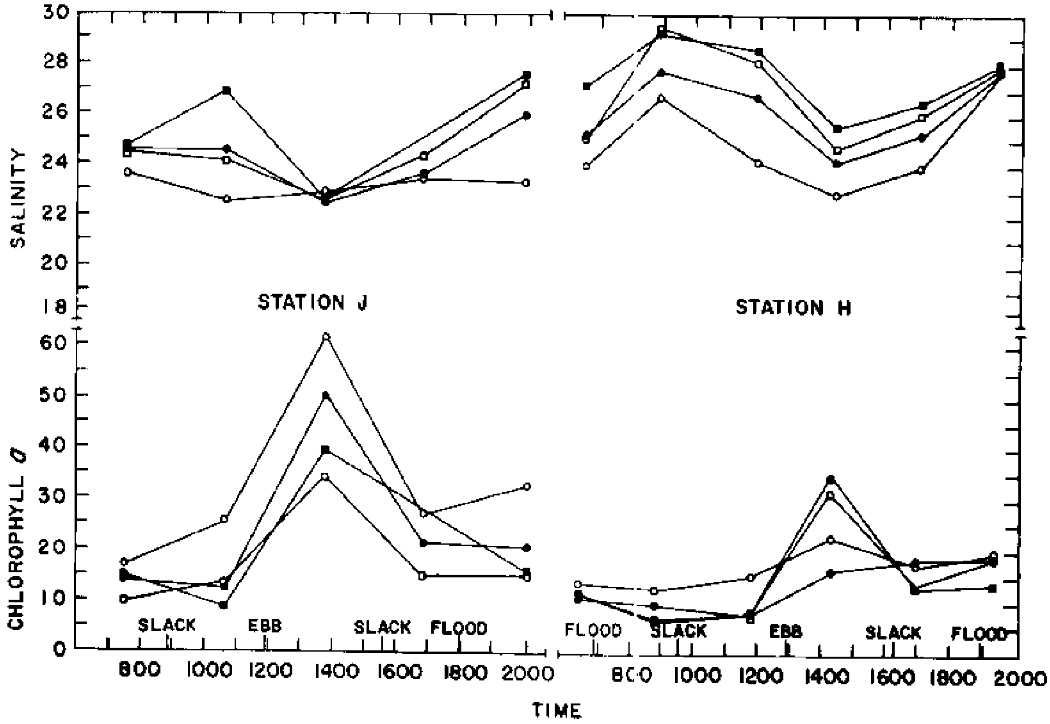


Fig. 9. Tidal variability of salinity (‰) and chlorophyll *a* concentrations (mg/m^3) for stations J and H, 5 June 1974.

Finally, the low $NH_4{}^+$ and Chl *a* concentrations present at Rockaway Point (stations A and B) identify New York Bight water.

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Oxygen depletion in the New York Bight apex: Causes and consequences

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Abstract

Dissolved oxygen concentrations in waters of the New York Bight apex are near saturation except in summer when a stable thermocline exists and concentrations in the lower layer can drop to 10% of saturation.

Mass balances of oxygen and carbon cycles in the apex were examined. Photosynthetically produced carbon accounts for most of the oxygen demand, particularly in summer. Oxygen demand due to sewage sludge and dredge spoils is small compared to that from organic carbon produced in situ. Oxygen demand of particulate and dissolved organics in the estuarine discharge may be as great as the sewage sludge and dredge spoils together.

Midsummer primary productivity in the apex is high due to nutrient inputs, particularly nitrogen. Most nitrogen, supplied to the apex in forms suitable to support photosynthetic production, comes from the discharge of the Hudson-Raritan-Passaic systems. Most of this nitrogen comes from liquid effluents of sewage treatment plants discharged to the rivers.

Ocean dumping in the bight apex does not cause the low oxygen concentrations found in summer. These are caused primarily by nitrogen supply from rivers. Improvement in dissolved oxygen concentrations could be achieved by removing nitrogen from sewage treatment plant effluents.

The impact of man's activities on the New York Bight, particularly the inner bight, has been appreciable (NMFS 1972). One probable cause of the reported degradation of benthic communities in the bight apex is the low oxygen concentration in bottom waters during summer (NMFS 1972). Low oxygen values have been observed in the region of the sewage sludge and dredge spoil dumpsites (Corwin 1970; NMFS 1972; Horne et al. 1971; Ketchum et al. 1951). It has been assumed generally that the oxygen demand of dumped material is responsible for removing the oxygen from the water. Studies of man's impact on the New York Bight supported by NOAA's Marine EcoSystems Analysis (MESA) Program included the question of low oxygen concentrations in the apex of the bight.

Distribution and seasonal variation of dissolved oxygen in the New York Bight

In 1948–1949 the first observations of low oxygen concentrations, about 50% saturation, in the bottom waters of this area were made (Ketchum et al. 1951). In September 1969 Corwin (1970) found oxygen at 26% of saturation while Horne et al. (1971) reported saturation values as low as 10% at the center of the dumping area. In 1969–1970 investigators from the Sandy Hook Laboratory found the lowest oxygen concentrations during summer when a strong thermocline was present. During winter the well mixed water column was essentially saturated with oxygen (NMFS 1972).

Distributions and seasonal cycles of certain dissolved and suspended particulate chemical species in the apex were studied on cruises made between April 1974 and March 1975. On each cruise continuous vertical profiles of salinity, temperature, and depth were obtained at a series of stations with an InterOcean model 513-10 CSTD. A General Oceanics model 1015 rosette multibottle array, equipped with General Oceanics 10-liter model 1070 top-drop Niskin bottles, was interfaced with the CSTD system. Water samples were obtained at the surface (1 m) and at intervals of 10 m to the bottom with a bottom sample about 2 m above the sediment–water interface. Each water sample was analyzed for salinity, dissolved oxygen, pH, nitrate, nitrite, phosphate, silicate, total suspended particulate

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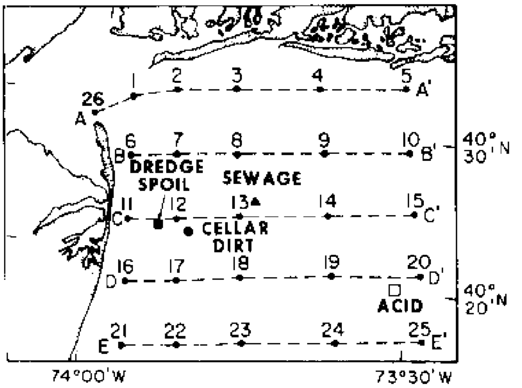

Fig. 1. Station locations, designated dumpsites, and transects (A'–A', B'–B', etc.) for which vertical sections are plotted in Figs. 3–7, 11–15, and 17–18.

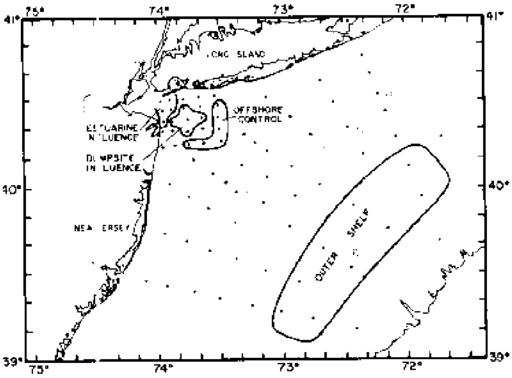

Fig. 2. Extended sampling grid, showing designated regimes for which average values are calculated (*see text*). Station grid sampled 26 February–3 March 1975.

load, particle size distribution, suspended total organic carbon and nitrogen, suspended total carbohydrates and total proteins, and dissolved metals including Fe, Mn, Cd, Cu, and Zn (Cantillo et al. 1976).

Seven cruises were carried out between April and November 1974 at 36-day intervals to coincide with ERTS (Earth Resour. Technol. Satellite) overpasses. Cruise dates were 16–20 April, 6–9 May, 10–13 June, 16–19 July, 21–24 August, 29 September–2 October, and 4–7 November. A grid of 25 stations (26 in September and November) was occupied (Fig. 1). An additional cruise was carried out in the period 26 February–3 March 1975; this cruise occupied a series of 64 stations over a larger area (Fig. 2).

Oxygen analyses were performed by the Winkler titration (Strickland and Parsons 1968). Percentage saturations of oxygen were calculated from these values and the salinity and temperature with the equations of Weiss (1970).

Vertical sections of percentage saturations of oxygen in the apex regions are shown in Figs. 3–7 for cruises in April, June, August, September, and November. In April the water column was almost isothermal but was weakly stratified with lower salinity water at the surface (Hazelworth et al. 1975*a*). The water column was almost saturated or slightly supersaturated with oxygen, although lower saturations occurred consistently in the lower layer (Fig. 3). By June the waters were stratified. Although near-surface waters were saturated or supersaturated with oxygen, the oxygen saturations in subthermocline waters had dropped, falling below 80% near the bottom in extensive areas of the apex (Fig. 4). In August (Fig. 5) with a well stratified water column, the area and depth of low oxygen waters were considerably extended, and saturation values as low as 28% were observed. By late September (Fig. 6) surface water temperatures had fallen, and the stratification was less well defined. The oxygen saturation values, although still lower near the bottom than at the surface, were not as low as they had been during midsummer. During the November cruise and for a number of days immediately before it, the weather was mild, and it was unusually calm and sunny in the New York area. The water column had cooled, and the thermocline had been partly destroyed by mixing. In this unusual situation, oxygen saturation values were variable and generally low (Fig. 7). Low oxygen waters extended to the surface. Rates of re-aeration would have been slow due to the surface film integrity that resulted from the extremely calm seas.

Lowest oxygen concentrations observed in summer were about 2 mg/liter. Investigators from the Sandy Hook Laboratory observed oxygen concentrations as low as 1 mg/liter in waters within 1 m above the sediment–water interface during 26 August–6 September, immediately after our August cruise (Thomas et al. 1976). They also con-

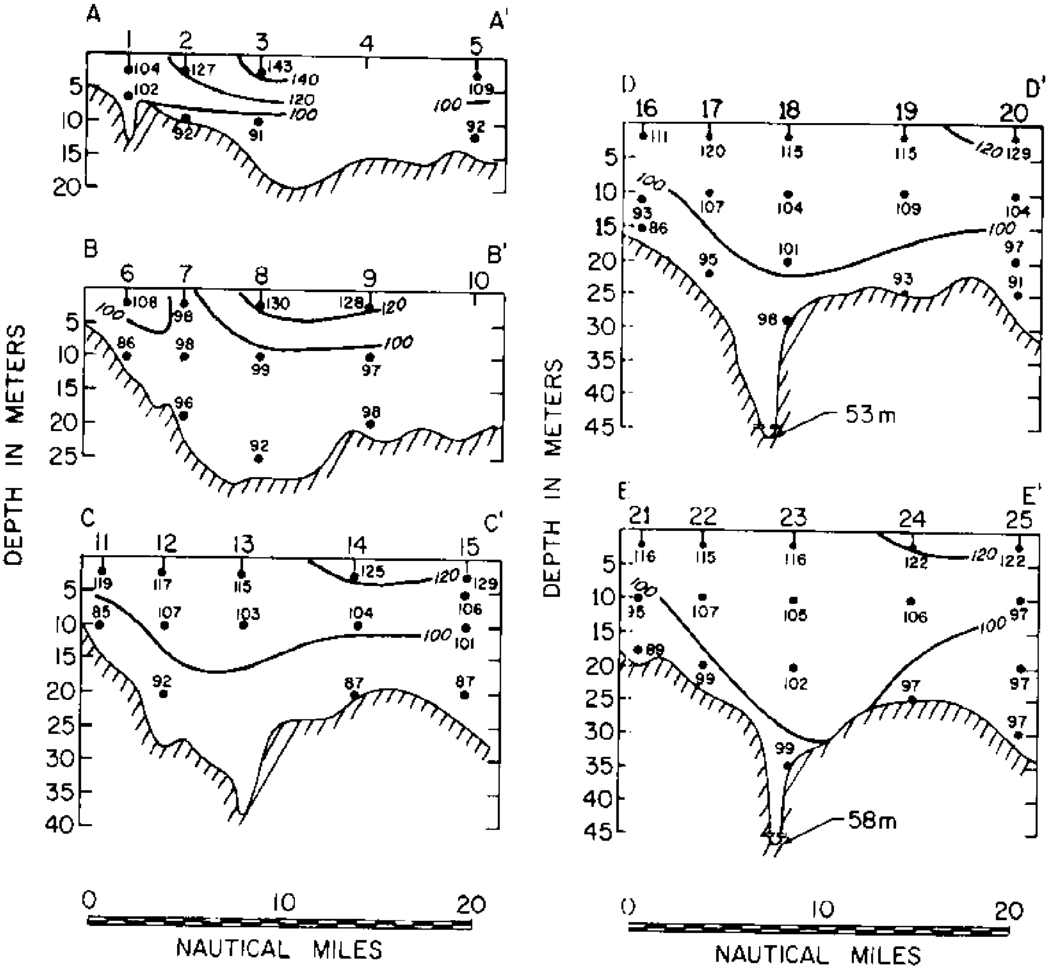

Fig. 3. Percent of saturation of dissolved oxygen in the apex, 16–20 April 1974.

firmed that large areas of the bight apex had bottom water with low oxygen concentrations.

There is considerable evidence that oxygen concentrations below about 4 mg/liter are responsible for changes in species composition and the abundance of marine organisms (Rheinheimer 1972; Vidaver 1972). The effects of low oxygen concentrations in the apex may therefore be responsible, at least in part, for some of the reported degradations in marine biological communities (NMFS 1972; Pararas-Carayannis 1973).

Processes affecting oxygen concentrations

The low oxygen concentrations could be due either to oxygen demand within the apex or to inflows of low oxygen water. Several considerations suggest that although low oxygen waters may enter from outside, this source is quantitatively less important in maintenance of low oxygen concentrations than oxygen demand within the apex itself.

Water entering the apex must come as a low salinity discharge from Lower New York Bay or from the open shelf. Low salinity water from Lower New York Bay was thought to be responsible for some of the low saturation water found in the open waters of the apex (Horne et al. 1971). However, measurements of oxygen concentrations and percent saturation in the discharge area itself do not support this view

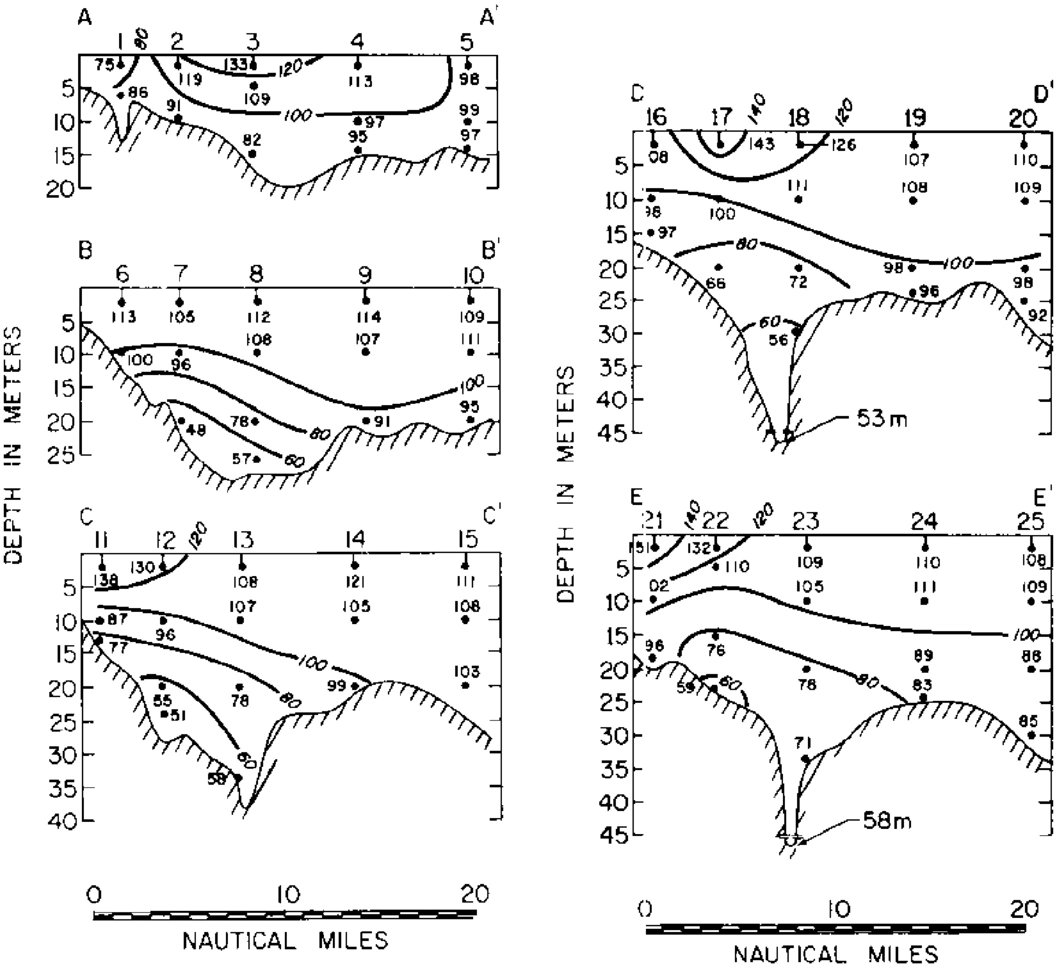

Fig. 4. Percent of saturation of dissolved oxygen in the apex, 10–13 June 1974.

(J. E. Alexander and I. Duedall personal communications; Figs. 3–7: stations 1, 2, 6, and 26). Oxidizable material in the estuarine discharge exerts an oxygen demand and influences dissolved oxygen concentrations. Such oxygen demand takes place within the apex itself and is discussed below.

Water entering the apex from the open shelf does not have anomalously low oxygen concentrations. Water below the thermocline on the Atlantic shelf was essentially saturated or slightly undersaturated with oxygen in September 1969 (Corwin 1970). Also, our data indicate that a substantial fraction of the oxygen must be removed from the water column within the bight apex. The average oxygen concentration and percent saturation for samples below the thermocline can be calculated by averaging all values for samples below 10 m. The thermocline is generally between 10 and 20 m (Hazelworth et al. 1975*a*,*b*). Averages have been calculated for groups of stations representing the dumpsite region (stations 8, 12, 13, 14, and 18—Fig. 8) and the outer edge of the apex through which exchange of water with the open shelf must take place (stations 10, 15, 20, 23, 24, and 25—Fig. 8—referred to as offshore control). Consistently throughout the year oxygen saturations and concentrations within the apex (dumpsite influenced region) are lower than at the ocean boundary (Figs. 9 and 10). Dissolved oxygen in waters at the ocean boundary of the apex is near saturation throughout the year. Therefore, the bulk of the oxygen depletion must take place through oxygen-demanding processes in the apex itself.

Waters with low percent saturation of oxygen are consistently found in the deeper portion of the Hudson Shelf Valley (station 23, Figs. 3–7). If all bottom water in the apex were derived from northward flow in the axis of the shelf valley, then the low oxygen concentrations in the apex might be due to influx of low oxygen water through this section. However, the data from waters outside the apex during June and September 1975 (unpublished data) suggest that this is not the case. Low oxygen waters at station 23 are not found at stations farther south in the shelf valley, except within several meters of the bottom. Current meter data from the Hudson Shelf Valley axis indicate that, during summer, mean flow near the seabed is slow and reversible, although net movement is normally southward (Lavelle et al. 1975). The quantity of oxygen-poor water that could be introduced to the apex by the slow mean flow in the shelf valley axis is insufficient to account for much of the oxygen depletion in the apex, even if this flow were permanently northward.

There are several sources of oxygen demand in the apex: oxidation of sewage sludge, organic material and sulfides in

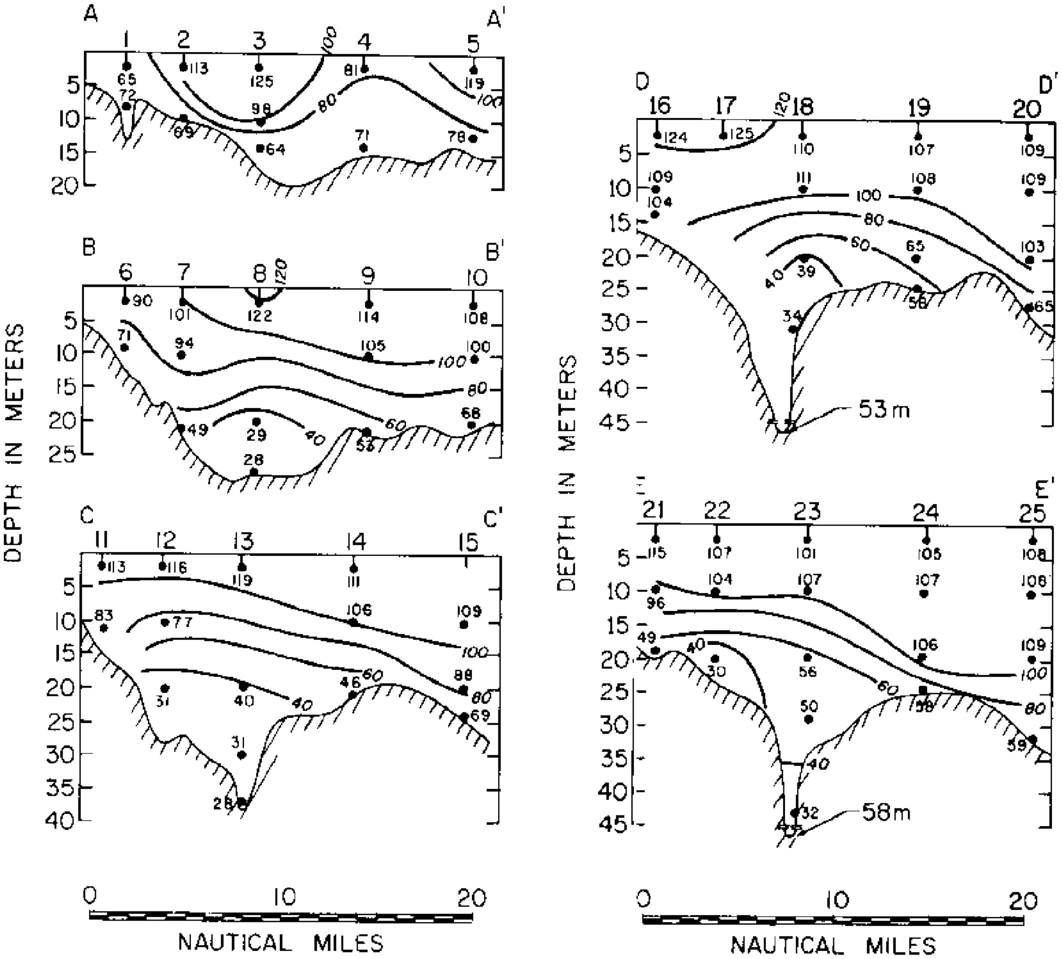

Fig. 5. Percent of saturation of dissolved oxygen in the apex, 21–24 August 1974.

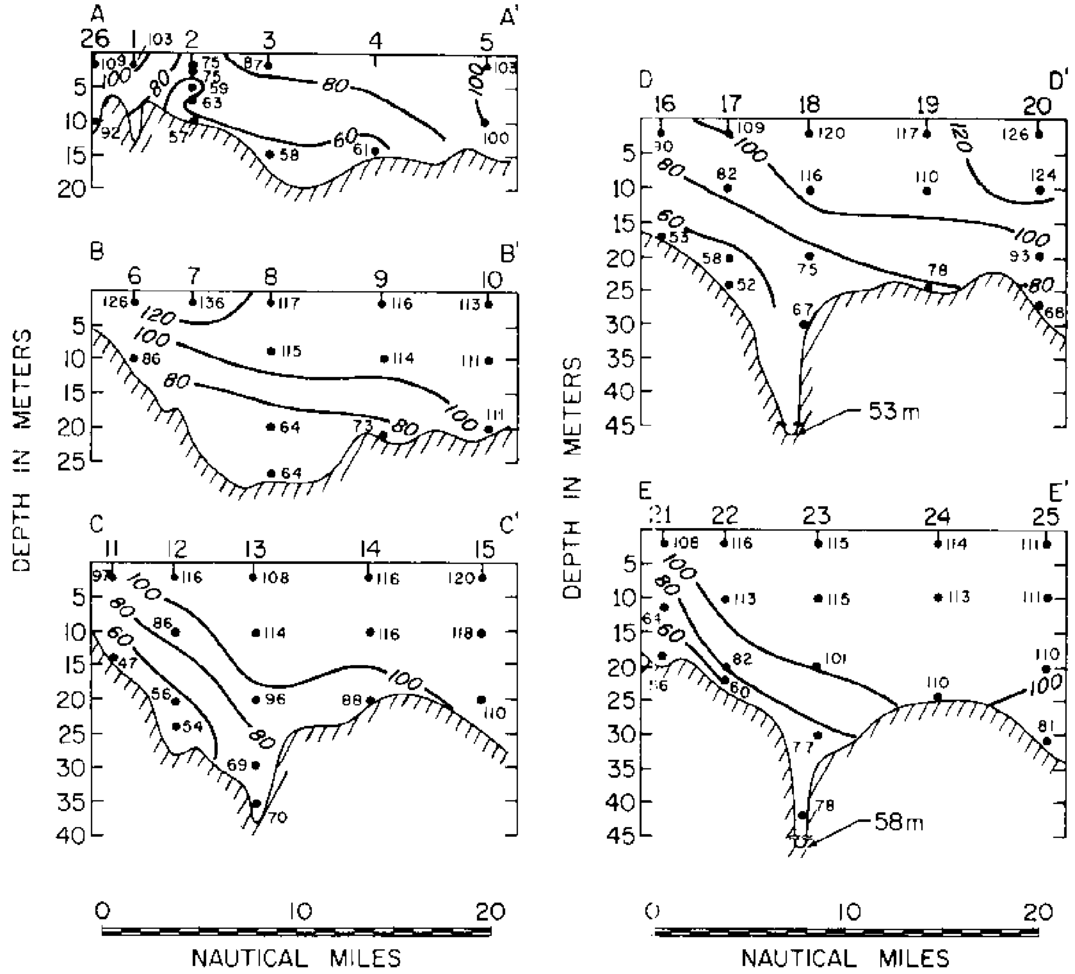

Fig. 6. Percent of saturation of dissolved oxygen in the apex, 29 September–2 October 1974.

dredge spoils, organic matter introduced in the estuarine discharge, acid wastes, and organic matter produced by photosynthesis. The chemical oxygen demand of acid wastes is small (Mueller et al. 1975). Potential oxygen demands of the four significant sources can be estimated. For this purpose, the apex is assumed to be 2,400 km^2 in area, or the region covered by our apex sampling grid (Fig. 1). This area represents quite well the area of the ocean where exchange with the southwest flow of the open shelf is somewhat restricted (Charnell and Hansen 1974). Amounts of total organic carbon input per day from the four sources are estimated in Table 1 and oxygen demands in

Table .. Approximate total organic carbon inputs to the apex (kg oxidizable C per day).

Sewage sludge dumping*	1.1×10^5
Dredge spoil dumping*	5.4×10^5
Primary production†	
Average	2.2×10^6
Summer	6.5×10^6
Estuarine input	
Particulate‡	7.3×10^4
Dissolved§	$\sim 1 \times 10^6$

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* From Mueller et al. (1975).

† Calculated from Malone (1976). Productivity interpolated between stations and integrated over 2,400 km^2 apex.

‡ Average mean flow rate of estuarine discharge of Raritan Bay 85) m^3/s (Mueller et al. 1975). Total particulate organic carbon 0.6 mg/liter (Hatcher unpublished data).

§ Assumes mean discharge of Raritan Bay 850 m^3/s (Mueller et al. 1075) and about 10 mg/liter of dissolved organic carbon (Alexander et al. 1974). Considerable uncertainty exists as to the dissolved organic carbon estimate.

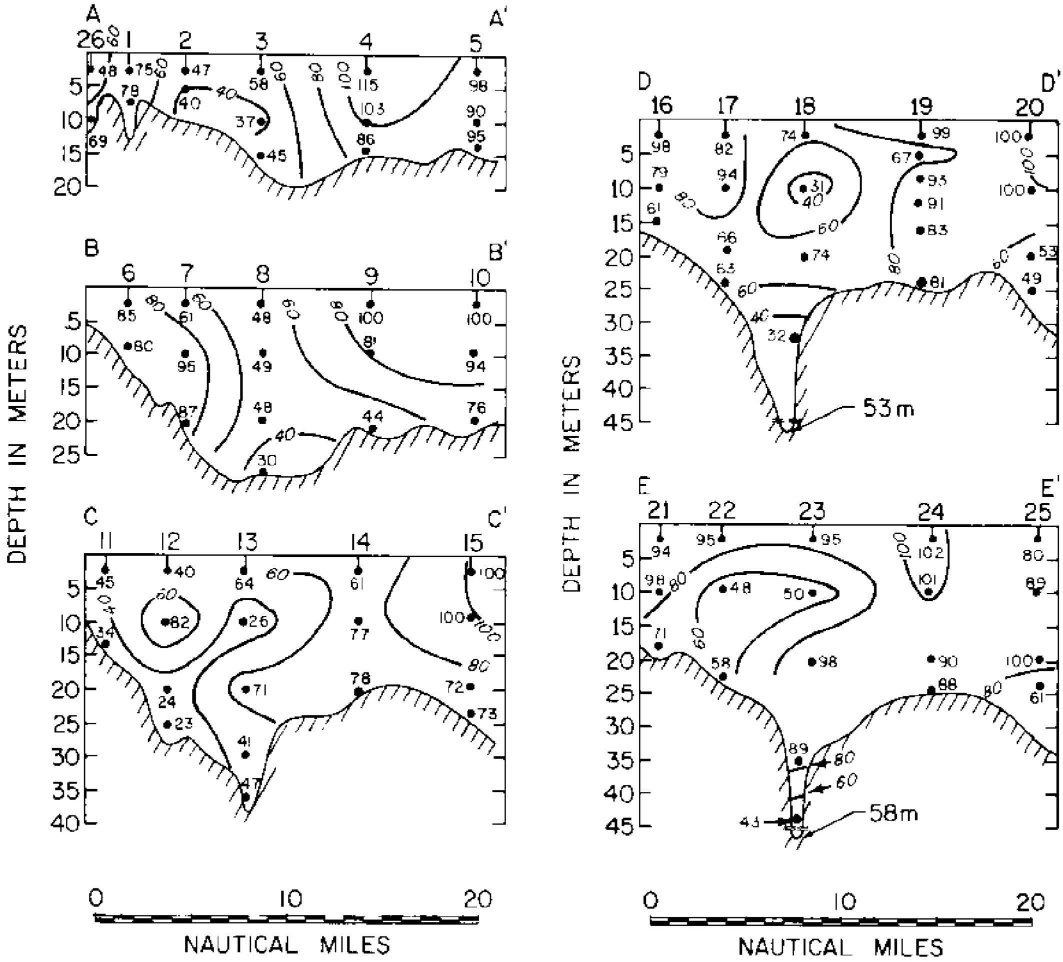

Fig. 7. Percent of saturation of dissolved oxygen in the apex, 4–7 November 1974.

Table 2. The major oxygen demand is from decomposition of photosynthetically produced organic matter, especially in summer. Ocean dumping is quantitatively less important. The river input may be quantitatively important. Analytical uncertainties of dissolved organic carbon concentrations in the river are such that the river input (Tables 1 and 2) is probably overestimated.

Four types of particulate organic matter having similar densities and size ranges are introduced to the apex through surface waters. Therefore, similar fractions of each type of organic matter will be transported vertically downward through the thermocline in summer, except that some dredge spoil and sewage sludge organics may be entrained with the rapidly settling mineral material.

Organic matter from sewage sludge, dredge spoils, and the estuarine outflows is

Table 2. Total potential oxygen demand from major sources in the apex (kg O_2 per day).

Sewage sludge dumping*	1.1×10^6
Dredge spoil dumping*	2.1×10^6
Primary production†	
Average	5.7×10^6
Summer	1.7×10^7
River input†‡	$\simeq5\times10^6$

* From COD data (Mueller et al. 1975).
† From organic carbon (Table 1) and Redfield ratio of carbon to oxygen (Redfield et al. 1963). Assumes complete oxidation.
‡ Considerable uncertainty exists.

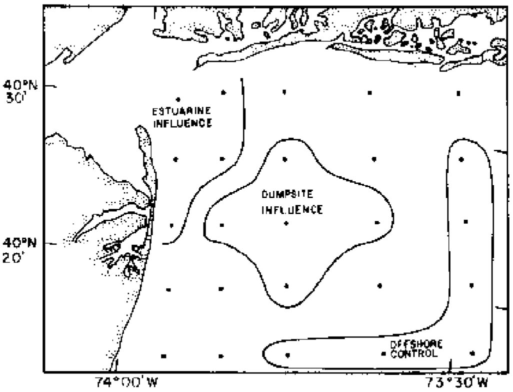

Fig. 8. Stations designated estuarine influenced, dumpsite influenced, and offshore control, for which average concentrations are calculated (*see text*).

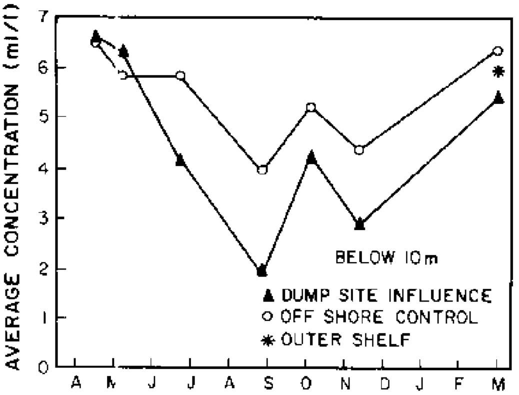

Fig. 10. Seasonal variation of average dissolved oxygen concentration in the water column below 10-m depth in the apex. (*See Figs. 2 and 8.* N.B. 1 ml/liter = 1.43 mg/liter. The latter units are used in the text.)

already partially oxidized. It will, therefore, be more refractory and oxidize slower than the organic matter produced by phytoplankton in the apex.

Mean residence times of low density particles in the apex are estimated to be not more than several weeks (Segar and Cantillo 1976). The proportion of organic matter introduced in dredge spoils or sewage sludge which is oxidized during residence in the apex will be lower than the proportion of photosynthetically produced material oxidized, assuming they have similar mean residence times in the apex. The dominance of the photosynthetic material over other sources in oxygen uptake in the lower layer of the apex is, therefore, probably underestimated in Tables 1 and 2.

If the water were saturated, the total quantity of dissolved oxygen below 10 m in the apex in summer would be about 2×10^8 kg ($\sim$8 mg O_2/liter in 2.6×10^{13} liters of water). Total oxygen demand of sewage sludge, dredge spoils, and summer primary production ($\simeq 2\times10^7$ kg O_2/d; Table 2) is sufficient to deplete the dissolved oxygen in this water layer in about 10 days. Some oxygen demanding material will be transported out of the apex before being oxidized, and, in the surface layer, some oxygen will be replaced from the atmosphere by photosynthesis. Additionally, much organic matter will be oxidized before being transported below the thermocline. Observed oxygen depletion (average of about 50% saturation) could be achieved if half of the total oxygen demand ($\sim 1\times10^7$ kg O_2/d) occurs below the thermocline and if the residence time of water in the apex is about 1 week (Ketchum et al. 1951). As the mean residence time calculated by Ketchum et al. (1951) probably underestimates the residence time of water below the thermocline in summer, the total oxygen demand in the lower layer required to produce the observed depletion is probably $<1\times10^7$ kg O_2/d.

Decomposition of organic matter and utilization of oxygen may take place either

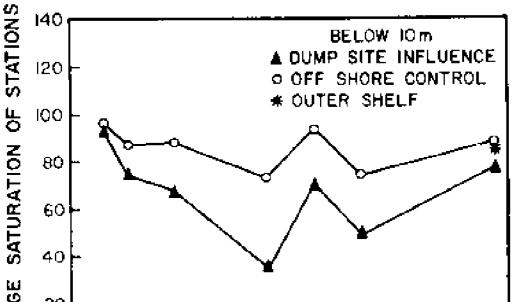

Fig. 9. Seasonal variation of average percent of saturation of dissolved oxygen in the water column below 10-m depth in the apex. (*See Figs. 2 and 8.*)

in the sediments or in the water column. Seabed oxygen consumption in the apex has been determined, and both geographical and temporal variations have been examined (Thomas et al. 1976). Total seabed oxygen demand in the 2,400-km^2 area considered, during summer, estimated from these data is about 1.1×10^6 kg/d. This is small compared to the oxygen demand inputs listed in Table 2. Therefore, most oxygen utilization must occur in the water column, perhaps concentrated in the euphotic zone and near the seabed where suspended sediment concentrations are highest (Drake 1974). Rates of oxidation will be high in the euphotic zone, as much of the easily oxidized organic matter will be oxidized before organic detritus is transported below the thermocline. Rates of oxidation will also be high near the bottom due to the presence of resuspended sediments. Sediments utilize oxygen considerably faster when resuspended (Berg 1970).

Oxygen uptake rates of the waters in the bight apex are not well defined. The only available data appear to be those of Thomas et al. (1976) who measured the oxygen consumption of water samples taken in summer, close to the sediment–water interface at a number of stations, coincident with seabed oxygen consumption measurements. The total oxygen demand of the water column below a depth of 10 m during summer in the apex calculated from these data is

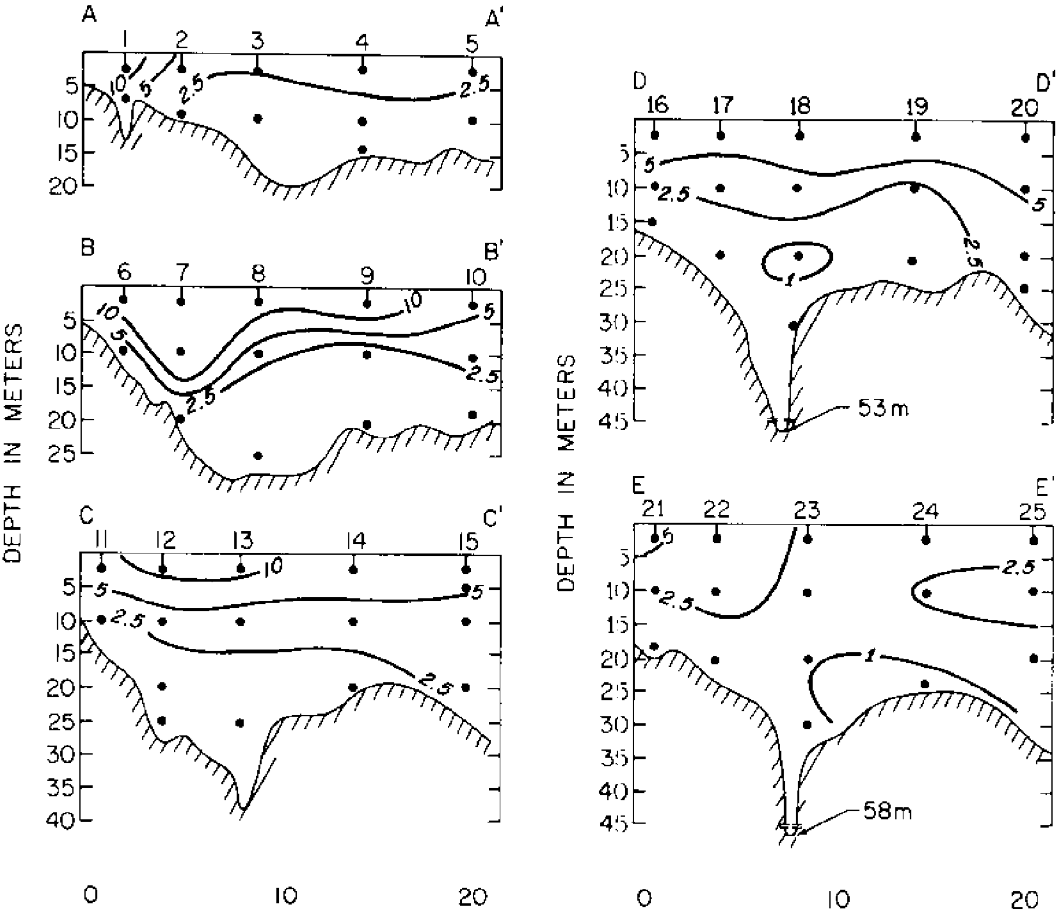

Fig. 11. Dissolved nitrate and nitrite (μg atoms N/liter) in the apex, 16–20 April 1974.

about 10^7 kg O_2/d—about ten times the seabed oxygen consumption—but the estimate is not good because the data are sparse.

Total oxygen consumption in the apex below a depth of 10 m, including seabed and water column consumption, is estimated to be about 10^7 kg O_2/d. This is comparable to the total oxygen demand inputs to the apex listed in Table 2.

Primary production and nutrients

Malone (1976) showed that mean primary production in the apex is 370 g C m^{-2} yr^{-1}; Mandelli et al. (1970) found a mean annual production of 420 g C m^{-2} yr^{-1} within the region extending to 5 mi south of Long Island. Primary productivity in waters outside the apex has been estimated by Ryther and Yentsch (1958) to decrease from 160 to 100 g C m^{-2} yr^{-1} going from 50 to 1,000 m deep. Therefore, the apex productivity is high compared to offshore values. Much of the excess production takes place in midsummer when productivity farther offshore is reduced because of nutrient depletion.

Several factors influence the productivity of oceanic phytoplankton. The availability of light and of the micronutrients nitrogen and phosphorus normally controls the productivity in coastal waters (Harvey 1963). In the New York Bight little is known about the relative importance of these two factors.

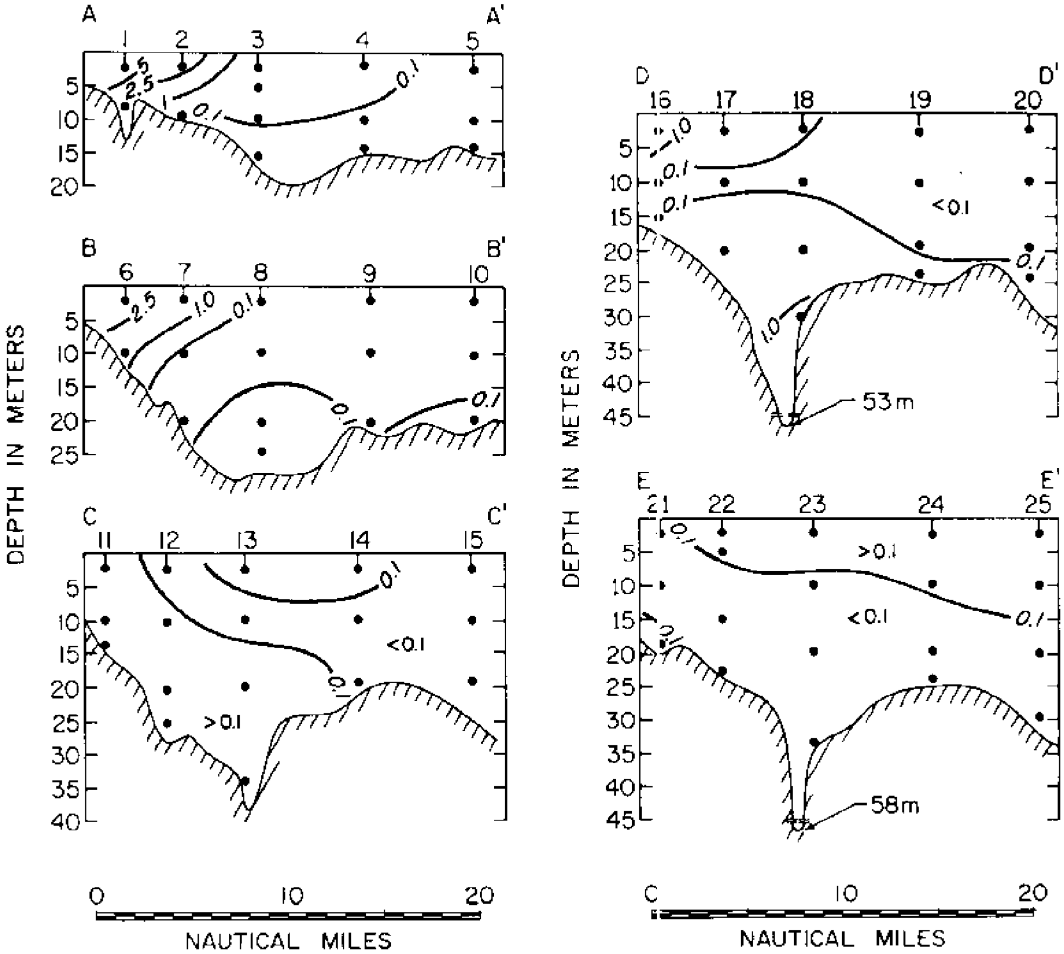

Fig. 12. Dissolved nitrate and nitrite (μg atoms N/liter) in the apex, 10–13 June 1974.

Ryther and Dunstan (1971) concluded that phytoplankton growth in the bight is nitrogen limited. They based this on the distribution of dissolved inorganic nitrogen and phosphorous in September 1969 and on bioassay experiments with *Skeletonema costatum*. From a study of productivity, light intensity, and chlorophyll *a*, Malone (1976) concluded that phytoplankton growth in the apex is light limited. However, he qualified this statement by noting that the low dissolved inorganic nitrogen to phosphorus ratios found in the bight could indicate that nitrogen is a growth limiting factor in summer in the outer portion of the apex.

Dissolved inorganic nitrogen (nitrate and nitrite) distributions in the apex between April and November 1974 are shown in Figs. 11–15. Highest concentrations are always associated with the plume of waters from Lower New York Bay. The plume normally lies along the New Jersey coast. The dissolved inorganic nitrogen concentrations in the surface layer of the center and outer part of the apex drop almost to zero and remain low during summer (Fig. 16). However, the dissolved inorganic nitrogen in the estuarine discharge area (stations 26, 1, 2, 6, 7, and 11: Fig. 1) remains higher throughout the year, as it does in the water below 10-m depth (Fig. 16).

Our dissolved inorganic nitrogen values do not include ammonia which may constitute the major fraction of inorganic nitrogen

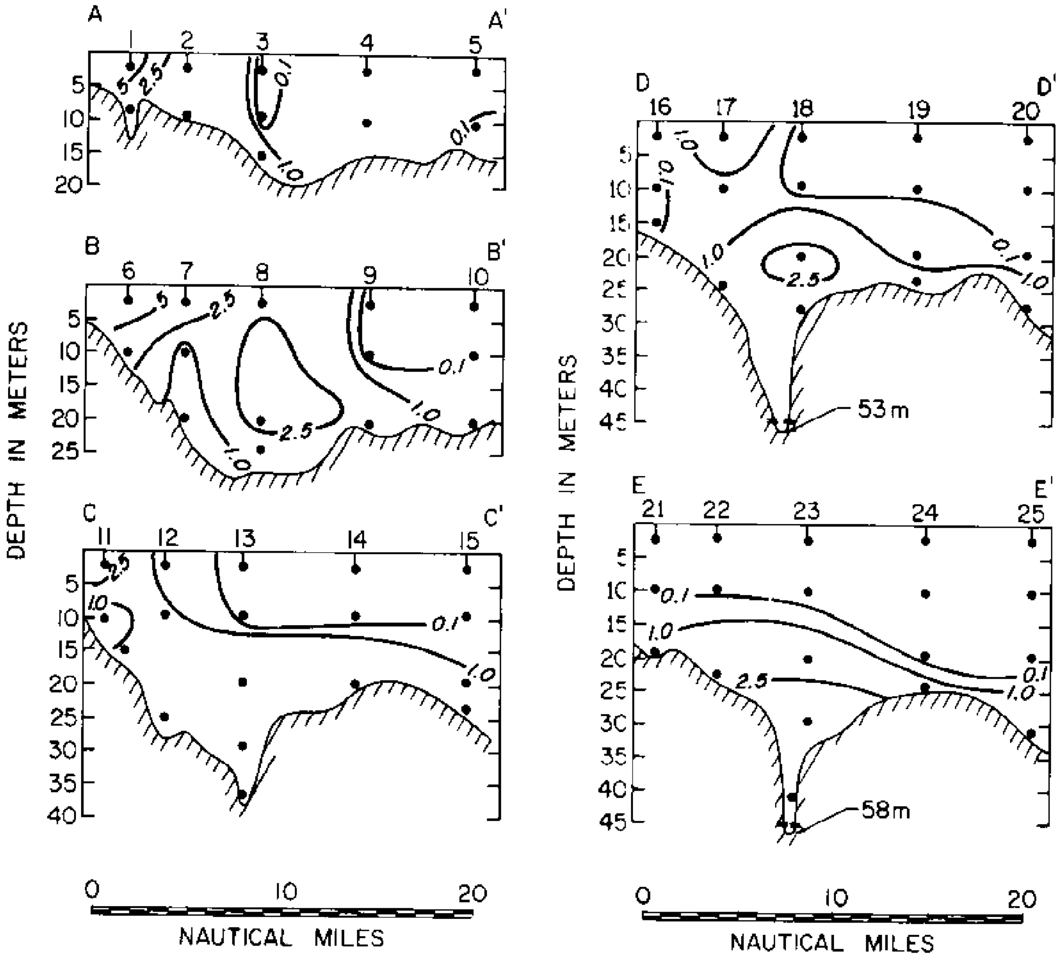

Fig. 13. Dissolved nitrate and nitrite (μg atoms N/liter) in the apex, 21–24 August 1974.

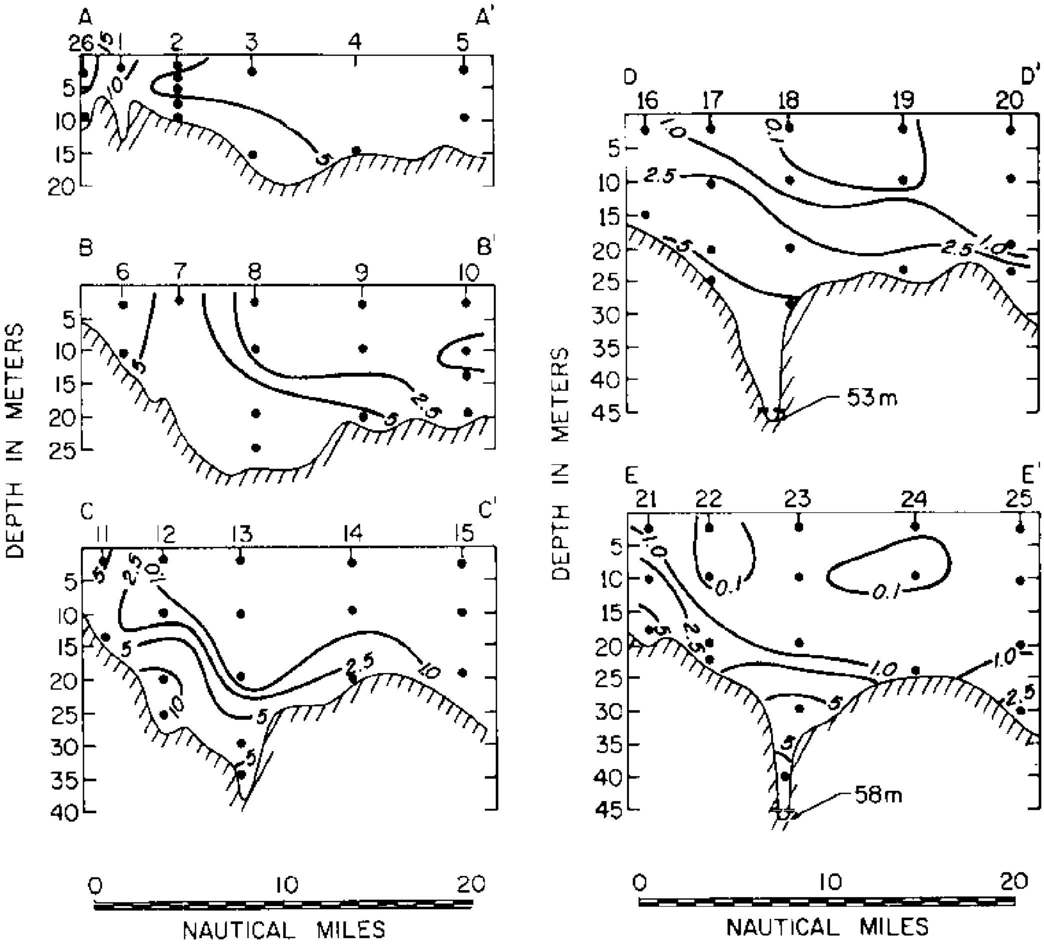

Fig. 14. Dissolved nitrate and nitrite (μg atoms N/liter) in the apex, 29 September–2 October 1974.

in the surface waters in summer when nitrate and nitrite are depleted (Cooper 1933). Ammonia originates primarily from decomposition of organic matter in the surface water, either by bacteria or zooplankton feeding and excretion and is rapidly reassimilated by phytoplankton. Ammonia is added to the apex both from the estuarine discharges and dumping of sewage sludge and dredge spoils (Duedall et al. 1975). However, phytoplankton generally exhibits a marked preference for ammonia-nitrogen over nitrate and nitrite (Harvey 1963; Dugdale and Goering 1967). If the quantity of ammonia supplied to the apex were sufficient to support all of the primary production in the apex, then the nitrate and nitrite would not be depleted.

Malone (1976) reported ammonia concentrations in the bight apex from April to August 1974. His data suggest that ammonia may be the most significant fraction of dissolved inorganic nitrogen in the water column below the thermocline, particularly where low oxygen concentrations occur during midsummer. Ammonia is regenerated from organic matter by bacterial decomposition—the process which is also primarily responsible for the reduction in oxygen concentrations.

Ammonia concentrations observed by Malone (1976) in surface waters of the bight apex during summer were generally high only near the estuarine discharge. In the rest of the apex, concentrations were close to the analytical detection limit.

Dissolved inorganic phosphate distributions in June and August 1974 are shown in Figs. 17 and 18 and the seasonal variation in Fig. 19. Although low phosphate concentrations are observed in summer, they remain above those normally associated with growth limitation (Fuhs et al. 1972). Like dissolved inorganic nitrogen, the highest phosphorus concentrations are found in the discharge from Lower New York Bay, while higher values are almost always found in water below 10 m than are found in the upper 10 m (Fig. 19).

The atom ratio of dissolved inorganic nitrogen to dissolved inorganic phosphorus in open oceans is about 15:1, the same ratio that is generally found in phytoplankton cells (Redfield 1958). However, in areas of restricted circulation and high biological activity, this ratio can change. Usually it becomes lower because nitrogen is removed from the surface layers more rapidly than phosphorus due to its slower regeneration from decomposing organic matter (Ryther and Dunstan 1971). Very low values of this ratio where phosphorus is more abundant than nitrogen are found only in areas where nitrogen is limiting for phytoplankton growth. The variation in the ratio of dissolved inorganic nitrogen to dissolved inor-

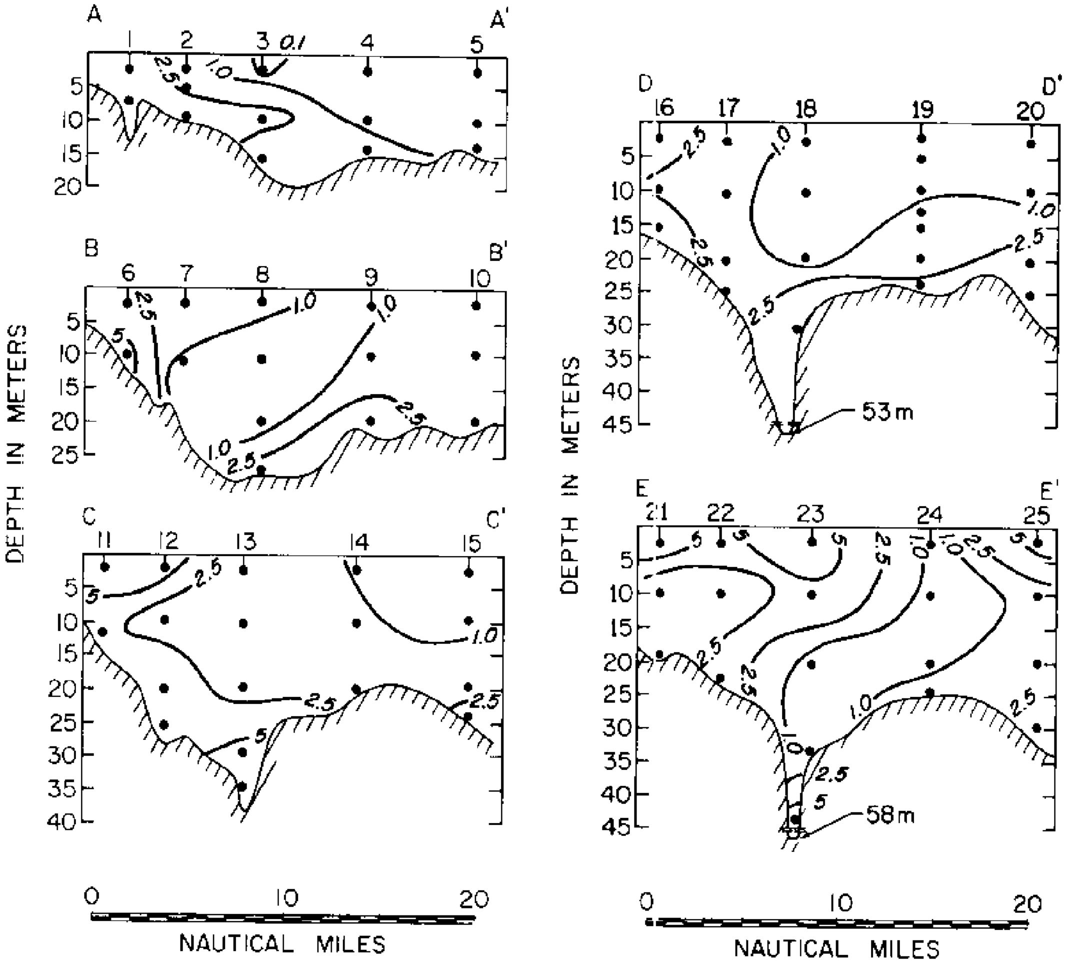

Fig. 15. Dissolved nitrate and nitrite (µg atoms N/liter) in the apex, 4–7 November 1974.

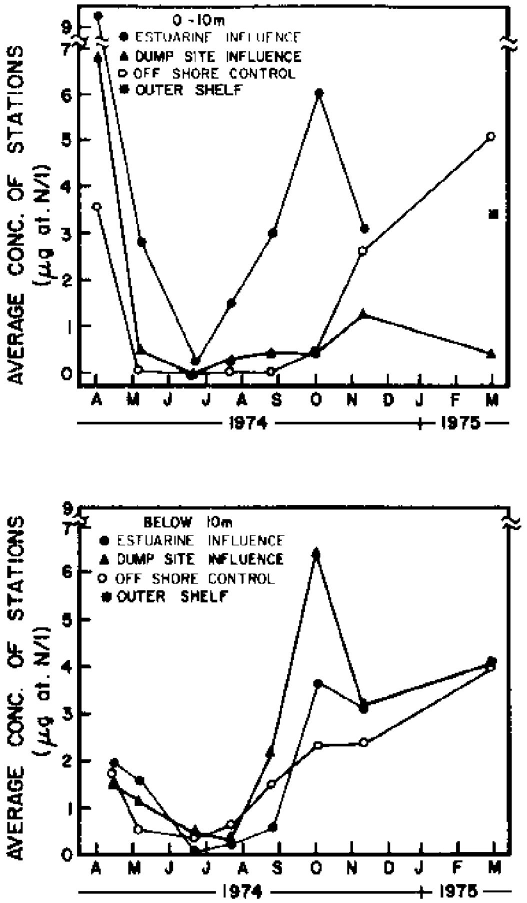

Fig. 16. Seasonal variation of average dissolved nitrate and nitrite, 10 m and above, and below 10-m depth in the apex for estuarine and dumpsite influenced regions, offshore control, and winter average for outer shelf region. (*See Figs. 2 and 8.*)

ganic phosphorus for surface waters in the bight is shown in Fig. 20. The ratio does not include ammonia-nitrogen and would be higher with this included, particularly in summer. However, Malone's (1976) data indicate that the ratio with ammonia included would be no more than two or three times the summer values in Fig. 20. The ratio, even corrected for ammonia, is extremely low throughout summer. Note that the estuarine water consistently has higher N:P ratios than the central apex (dumpsite influenced), and the stations farthest offshore consistently have the lowest N:P ratio.

We interpret the data as follows. During winter, productivity in the apex is light limited. In spring, as light intensity increases, productivity in and near the turbid estuarine plume is light limited, while areas outside sustain phytoplankton blooms, stripping surface waters of nitrogen. When nitrogen becomes limiting in the outer area of the apex, the total production of the apex region is also limited by nitrogen, even though the growth rate in the more turbid inner apex remains light limited. All of the nitrogen supplied to the surface layer of the apex is consumed by phytoplankton growth within the apex during this summer period. Organic matter produced is partially reoxidized in the surface waters, releasing some nitrogen for recycling. The remaining organic matter is transported below the thermocline and oxidized, becoming the primary cause of the low oxygen values observed in this water mass.

It is possible that some intense localized phytoplankton blooms in the New York Bight apex in spring and summer are caused by mixing high-nutrient low-transparency estuarine water with nutrient-depleted clear ocean water. If the edge of the turbid estuarine plume is sharp, as it often appears to be from satellite images, then phytoplankton in a very restricted area at the interface will have an adequate nutrient supply and will not be light limited, due to settling out of some estuarine-derived suspended particles.

The sources of nitrogen input to surface waters of the apex during the summer stratified period include dumped waste materials, estuarine discharge, atmospheric fallout and washout, and mixing across the thermocline. The last of these is likely to be small. Water entering the apex from the outer shelf during midsummer is depleted of dissolved inorganic nitrogen (Figs. 12, 13, and 16) except for near-bottom waters. It is difficult to estimate the amount of nitrogen transported into the apex from the outer shelf because the dynamical information available is limited. However, if the differential between water entering the apex and that leaving were 1 μg atom N/liter, the flushing time were 7 days, and all of the nitrogen thus supplied were brought into the surface layer and made available for photosynthesis, then the rate of input would be about 1×10^5 kg N/d. This estimate is probably too high, as the concentration differential is less than 1 μg atom N/liter in surface waters moving in and out of the apex in midsummer, and inhibition of vertical mixing by the thermocline will prevent much of the nitrogen introduced in the deeper water from being made available for photosynthesis. Therefore, the supply of nitrogen to the apex for photosynthesis from the outer shelf is probably small compared to the other sources whose inputs are estimated in Table 3. In any event, nitrogen from outside the apex would be supplied irrespective of man's presence and, therefore, cannot account for the high productivity of the apex in comparison to other continental shelf regions.

Table 3. Total nitrogen supplied to the apex (kg N per day).

Sewage sludge dumping*	1.7×10^4
Dredge spoil dumping*	6.3×10^4
River input†	1.2×10^5
Atmospheric input*	6.4×10^4‡

* From Mueller et al. (1975); total Kjeldahl nitrogen.
† From Garside et al. (1976); summer value, winter is 1.6×10^5.
‡ Considerable uncertainty.

Table 3 gives estimates of nitrogen inputs to the apex from dumping sewage sludge and dredge spoils, from the atmosphere, and from estuarine input. The river input is the major contributor as suggested by the

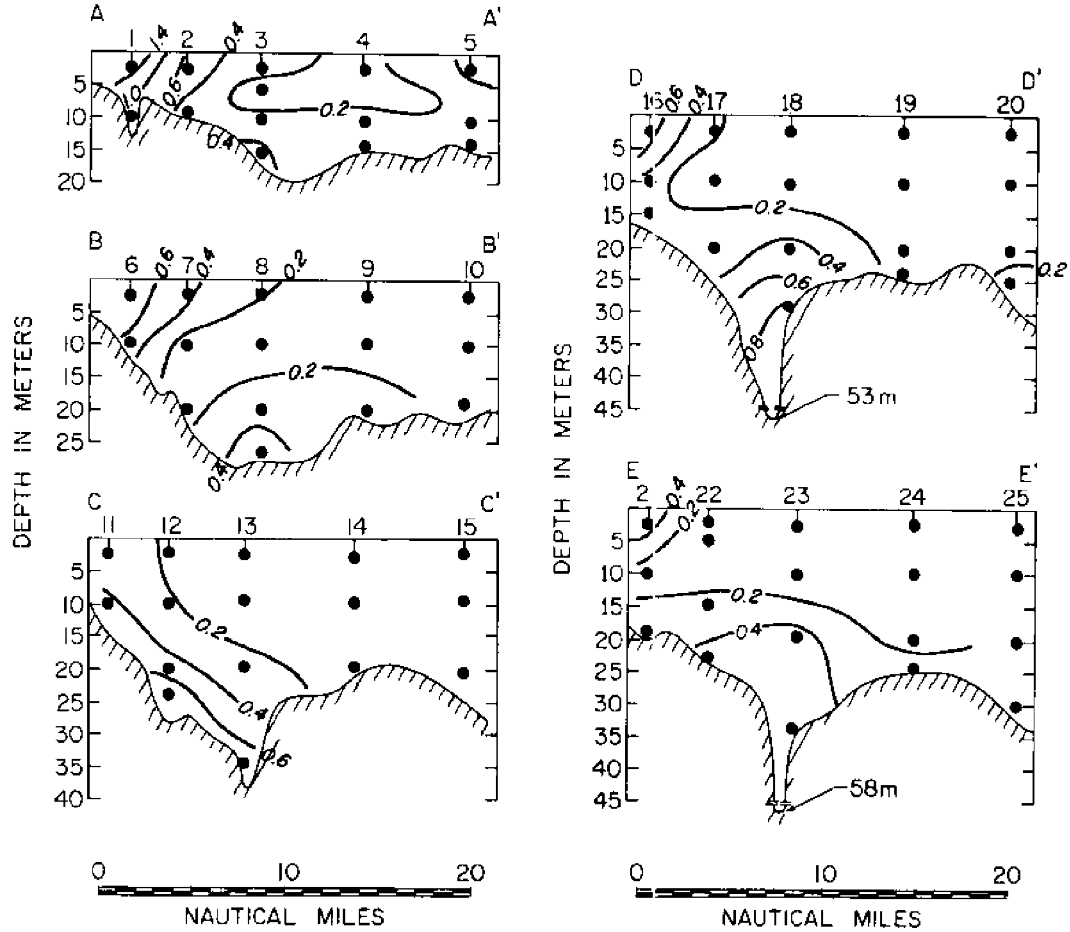

Fig. 17. Dissolved reactive phosphorus (μg atoms P/liter) in the apex, 10–13 June 1974.

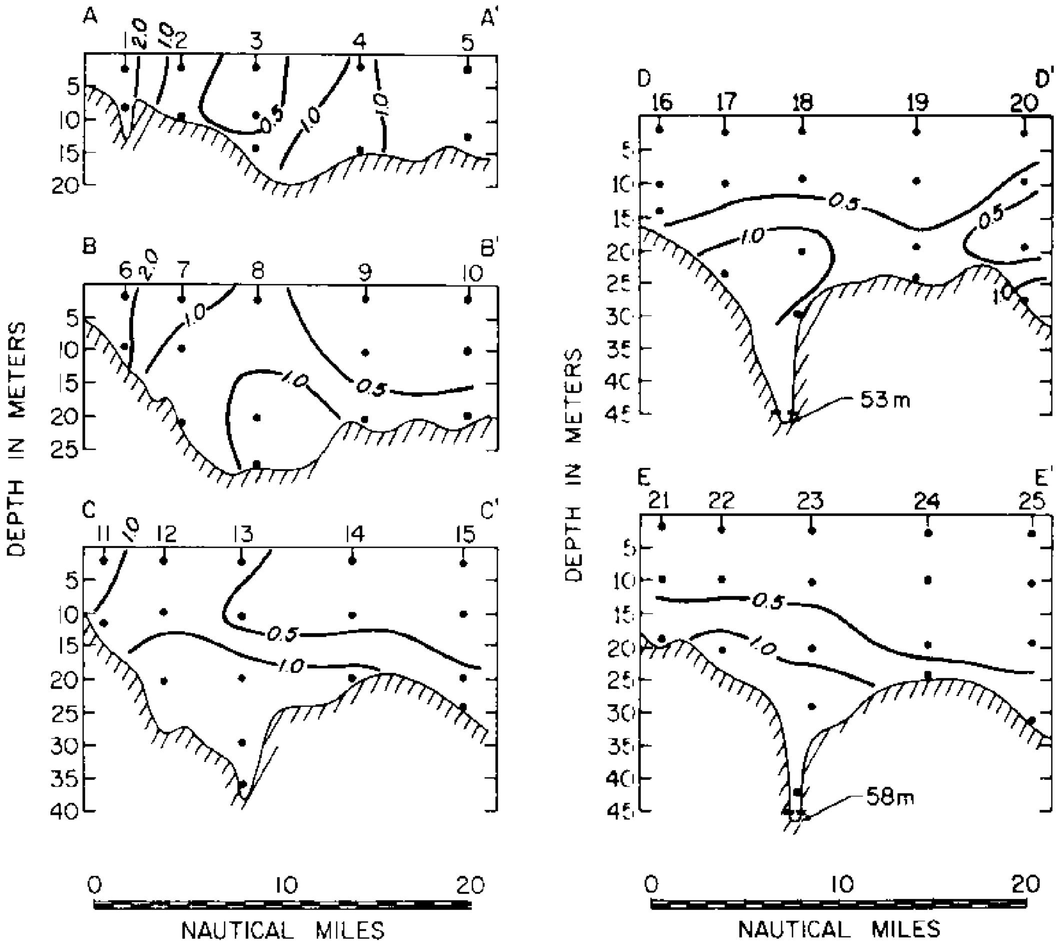

Fig. 18. Dissolved reactive phosphorus (μg atoms P/liter) in the apex, 21–24 August 1974.

nitrogen distributions (Figs. 11–16). In fact, the importance of the river input is probably underestimated in Table 3. The quantities of nitrogen calculated for the dredge spoils and sewage sludge are total Kjeldahl nitrogen values. Much of the nitrogen is organically bound (Mueller et al. 1975) and will not be available for phytoplankton growth before being transported either out of the apex or below the thermocline. Garside et al. (1976) included most but perhaps not all of the inorganic nitrogen which is introduced to the river with sewage or sewage treatment plant effluents in their estimate of estuarine input. Other sources of nitrogen to the rivers, although comparatively small, are not negligible. Mueller et al. (1975) estimated estuarine input of total Kjeldahl nitrogen plus nitrate and nitrite to the apex to be 3.4×10^5 kg N/d.

It is noteworthy that the atmospheric dust and rain input of nitrogen to the bight apex may be significant compared even to the river input and that it consists almost entirely of inorganic nitrogen which is readily available to organisms. However, there is considerable uncertainty about the accuracy of the atmospheric input estimate.

If the only nitrogen supplied to the apex euphotic zone were from the estuarine outflow (1.2×10^5 kg N/d), and no recycling of the nitrogen in the surface waters took place, the C:N ratio of the phytoplankton produced in summer would be 63. Values of

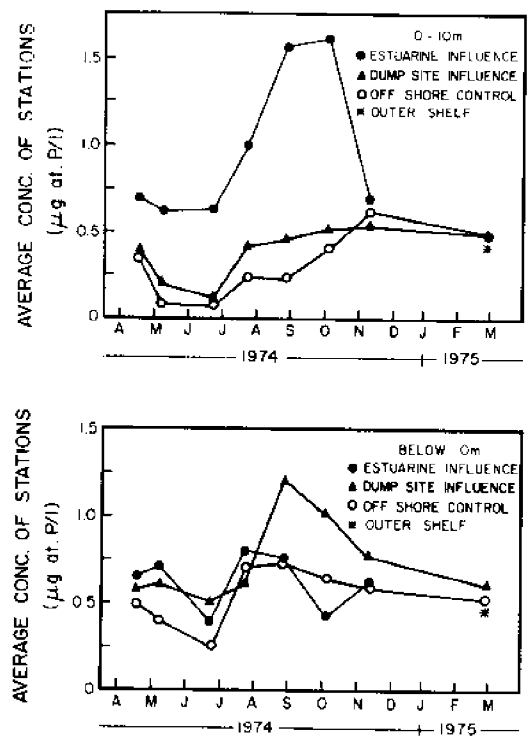

Fig. 19. Same as Fig. 16 for average dissolved reactive phosphorus.

the C:N ratio this high are unknown, but values as high as 20 have been observed in nitrogen limited phytoplankton (Caperon and Meyer 1972), while a ratio of 7 is considered normal (Redfield et al. 1963). The estimate of Garside et al. (1976) for nitrogen input by the estuary is certainly too small, and a substantial proportion of the phytoplankton nitrogen will be regenerated and reutilized within the surface layer before being transported below the thermocline. Therefore, nitrogen from the estuary is enough to sustain much of the phytoplankton production in the apex.

Consequences

The above conclusions have considerable importance for decisions concerning the necessary steps to reduce the deleterious effects of man's activities on the New York Bight ecosystem. The NOAA-MESA Project was designed specifically to aid the transfer of scientific knowledge to the environmental manager's decision-making process. Therefore, it is relevant to review briefly the content of this paper in terms of impending decisions concerning the disposal of sewage sludge generated in the New York–New Jersey metropolitan area.

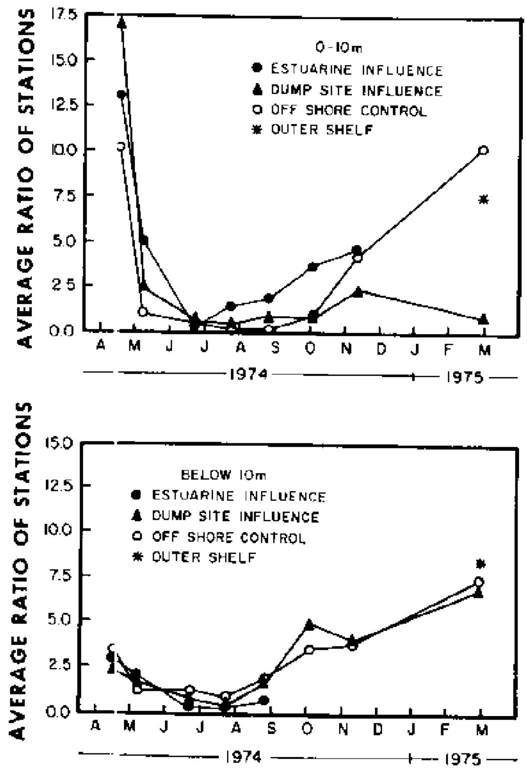

Fig. 20. Same as Fig. 16 for atom ratio of dissolved nitrate and nitrite nitrogen to dissolved reactive phosphorus.

The Environmental Protection Agency has declared its intent to consider halting all dumping at the present sewage sludge dumpsite in 1976 (EPA 1974). The dumpsite would be moved to a position about 65 nmi from the entrance to Lower New York Bay. The extra distance that the sludge would have to be transported translates into a considerable additional expenditure by local authorities.

Conclusions reached in this paper indicate that the removal of the sewage sludge and/or dredge spoil input to the water column in the New York Bight apex would probably have little effect on the low oxygen concentrations observed in the lower water layer in summer. In fact, the decrease in suspended solids and, therefore, turbidity might increase the productivity of the apex during times when light limitation exists. Thus, removing sewage sludge and/or dredge spoil inputs could conceivably extend the period when very low oxygen concentrations occur, even though the directly added oxygen demand would be reduced.

Low oxygen concentrations in the bight apex are, however, not the only cause for concern about ocean dumping in the New York Bight apex. Toxic metals, organic compounds, and pathogenic microorganisms are introduced with the dumped material. Segar and Cantillo (1976) demonstrated that stopping sewage sludge dumping in the apex would have little effect on trace metal concentrations within it, but that stopping dredge spoil dumping might significantly reduce these concentrations. Similarly, less extensive studies by MESA personnel suggest that sewage sludge dumping is probably not the major source of organic contaminants such as pesticides and hydrocarbons in the apex. In addition, no evidence exists that pathogenic microorganisms have been transferred from dumped materials to man by way of the oceans.

We conclude that the removal of the sewage sludge dumpsite alone will have little beneficial effect on the New York Bight apex. Removal of both sewage sludge and dredge spoil dumpsites would have only marginal benefits. However, dumping these materials at offshore alternate sites would have potentially harmful effects on the biota living in a region of ocean not presently as heavily impacted by man as is the apex. In fact, the recommendation of the NOAA-MESA Project is that, unless some as yet unidentified critical impact of sewage sludge in the New York Bight becomes known, sewage sludge should continue to be dumped at the present location until EPA phases out ocean dumping entirely in 1981 (Wallace 1976). At the same time, every effort should be made to expedite a solution to the entire sewage treatment problem.

Low oxygen concentrations are perhaps the most critical environmental problem so far identified in the bight apex. Any significant increase in the nitrogen input to the apex would further reduce the oxygen concentrations. The apex is eutrophic. If nitrogen input, particularly from the rivers, were to increase significantly, then it is likely that anoxia would occur. Anoxia is well known in fjords and lakes where the oxygen demand input to bottom water layers is smaller than in the New York Bight but where the flushing time is longer. If anoxic conditions develop in the bight, they would occur episodically. The balance of oxygen demanding processes against flushing of the apex with more oxygen-rich waters and oxygen resupply from the atmosphere is controlled by weather conditions. Fish kills, sulfide release to the atmosphere, and beach contamination associated with anoxia in fjords might well be in the future for New York unless steps are taken to limit nitrogen loading of the rivers. The principal source of nitrogen to the rivers emptying to the New York Bight is wastewater discharge Therefore, this problem is intimately linked with the sewage sludge disposal problem.

We believe that the possibility that anoxic conditions in New York Bight waters may be caused by nutrient loading, is of sufficient concern that the problem of nutrient removal from wastewater discharges, i.e. better treatment of sewage rather than deeper ocean disposal of sewage sludge, should be examined closely by the New York and New Jersey municipalities and the appropriate agencies.

ADDENDUM (4 October 1976): During May and June 1976 unusual weather conditions obtained in the New York Bight with winds from the south and southwest during a 5-week period. During this time upwelling of cold nutrient-rich water probably occurred, which added to the existing high productivity of *Ceratium tripos*. *Ceratium* had been observed in unusually large numbers along the mid-Atlantic coast during February and succeeding months. At the end of the upwelling period conditions were anoxic and near anoxic in extensive areas of the bight near the New Jersey shore, large quantities of decomposing *C. tripos* were observed in a mat close to the sediments, and fish and shellfish mortalities were recorded. The anoxic conditions have persisted at least through September 1976. An interim report ("Mortalities of fish and shellfish associated with anoxic bottom water in the Middle Atlantic Bight" NOAA, Middle Atlantic Coastal Fisheries Center, Sandy Hook, N.J., September 1976) stated

> A major environmental event that has affected the sport and commercial fisheries along the New Jersey shore occurred in New York Bight this summer [1976]. The immediate impact includes fish and shellfish mortalities and unusual fish distribution patterns and/or concentrations. Long-term effects, including interruption of spawning, mortalities of eggs and larvae, and disruption of food chains may be as significant to the resource as the actual numbers of organisms killed.

This anoxic episode likely would have been geographically smaller and less persistent, and may in fact not have occurred, had the problem of river nitrogen contamination from sewage treatment plant effluents discussed here not existed to aggravate natural conditions.

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Carbohydrates and organic carbon in New York Bight sediments as possible indicators of sewage contamination[1]

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Abstract

Sediments of the New York Bight were analyzed for total organic carbon (TOC) and total carbohydrates (TCH). The TCH:TOC ratio was significantly more elevated than comparable sediments from other areas. High TCH:TOC values (≥ 30) may be attributed to sewage-derived materials which contribute significant quantities of refractory organic matter to the bight annually. Nonanthropogenic sources of organic matter to these sediments have little or no influence on TCH:TOC except near the shelf break. The TCH:TOC ratio may be useful as a qualitative and semiquantitative indicator of sewage-derived organic matter in sediment deposits. The observed TCH:TOC ratios suggest that organic material being deposited in the Christiaensen Basin and mud patches near Long Island is predominantly of sewage origin, and that seaward of the apex, the sedimentary organic matter becomes less influenced by sewage-derived organic matter and oceanic organic matter becomes a more significant fraction.

Increasing alarm has been expressed over the potential threat of sewage-derived chemical and biological contaminants accumulating on the beaches along coasts of the New York Bight. Consequently, a means of identifying and quantifying sewage-derived materials is of great interest.

Visual observations of sediment samples resembling "black mayonnaise" have been used in an attempt to map areas of "sludge" accumulation (Harris 1974). Sewage sludge constituents such as tomato seeds and human artifacts have been used elsewhere as sewage indicators with varying degrees of success (Shelton 1971). In the New York Bight, the distribution of organic matter in sediments was used to delineate the spatial extent of sewage contamination (Pearce 1972). More recently, in connection with the Coastal Water Research Project off southern California, various toxic heavy metals and chlorinated insecticides have been been used as qualitative indicators of sewage contamination (South. Calif. Coastal Water Res. Proj. 1974). These parameters can serve only as qualitative or semiqualitative indicators. A positive quantitative tracer for sewage is lacking.

Sewage is a complex and heterogeneous mixture of organic and inorganic materials. As it is incorporated into marine sediments, the resultant mixture becomes even more complex. Organic matter from sewage and natural sources becomes intermixed with the inorganic components of both sewage and natural materials, so we must first segregate organic from inorganic components of the sediments. The inorganic components are generally predominant in bight sediments as the organic matter content is usually <20% of dry weight (Pearce 1972). The overwhelming presence of natural silt and sand-sized particles may limit the usefulness of inorganic constituents as sewage indicators. Because the organic content of natural sediments is low and that of sewage is so high, admixture of sewage with natural sediments, even in small percentages, should substantially affect the nature of the resultant organic matter. By examining the nature of the organic matter in sediments of the bight we may be able to estimate the relative amount of the organic matter that is derived from sewage.

As part of the Marine EcoSystems Analysis Program, an organic geochemical investigation of New York Bight sediments was

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[1] This study was conducted at the Atlantic Oceanographic and Meteorological Laboratories of NOAA as part of the NOAA–MESA Program, New York Bight Project.

initiated. Samples were collected from the bight apex and Hudson Shelf Valley and from other study areas being investigated as part of the MESA Program (Fig. 1). Our findings indicate that total organic carbon (TOC) and total carbohydrates (TCH), together as a ratio (TCH:TOC), may be used as qualitative and even semiquantitative indicators of sewage-derived materials in sediments.

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Methods

Sediment samples taken with a Shipek sampler in fall 1973 were freeze-dried, and aliquots were analyzed in duplicate for total organic carbon (TOC) by a dry combustion technique (Konrad et al. 1970; Hatcher 1974). Other aliquots were analyzed in duplicate and, in some cases, triplicate for total carbohydrate (TCH) by the phenol-sulfuric acid method (Artem'yev 1970; Gerchakov and Hatcher 1972).

Results and discussion

TOC in sediments of the New York Bight—Gross (1970, 1972) analyzed various sediment samples from the New York Bight and Raritan Bay areas for TOC and noticed an enrichment of organic matter (TOC = 5%) over typical open shelf sands. Both areas are located in regions of relatively low hydraulic energy and therefore are less exposed to severe scour and erosion than open shelf sediments. These areas, therefore, act as sediment traps for muds and other fine-grained particles entrained in the water column, be they of natural or contaminant origin. Since Hunt (1961) and Froelich et al. (1971) have pointed out that TOC is inversely related to the mean size distribution of the sediment particles, we would expect the TOC to be greater in muds than in sands of the bight. It seems possible that the distribution observed by Gross (1970) may be explained, at least partially, by this phenomenon.

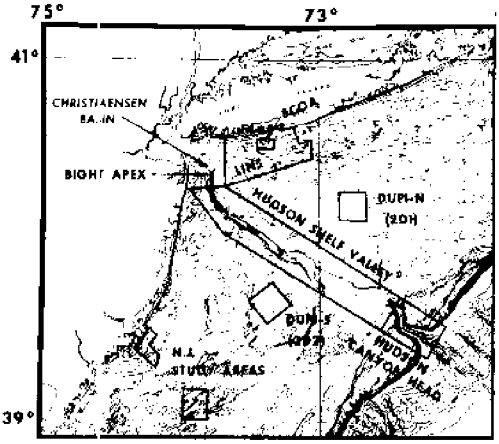

Fig. 1. Index map of the New York Bight showing the MESA geology study areas. SCOA—Suffolk County outfall area; LINS—Long Island nearshore; 2D1 and 2D2—alternate dumpsites.

Pearce (1972) has determined total organic matter in various sediment samples from the bight apex and found the concentrations to be on the order of 10–20% organic matter in the mud and silt deposits of the Christiaensen Basin, which is a low topographic feature in the apex (Fig. 1). The sediments of one area in the Christiaensen Basin had an organic matter concentration >20% dry weight. This area was classified as a dead sea, devoid of the large numbers of benthic organisms typical of other similar muddy environments. The sediments were identified as black, oily muds having a consistency similar to that of mayonnaise.

In our investigation, TOC values were obtained for sediments in the bight apex and adjacent Hudson Shelf Valley (Fig. 2). Values range from less than 0.1% (dry wt) in some sands to about 5% in some silty muds found in the topographic lows. A strong correlation is observed between the bathymetry of the area shown in Fig. 1 and TOC distribution.

Although the sediments in the axis of the shelf valley and the Christiaensen Basin are all similar in grain size, some spatial differences exist in TOC concentrations. The TOC isopleths indicate a down-valley depletion of muds high in organic carbon. It seems that these muds may be originating from the Christiaensen Basin at the head of the shelf valley. This area of the apex has silty muds relatively high in TOC (Gross 1970). However, enrichment in TOC alone cannot be taken as indicative of sewage contamination.

Carbohydrates and carbohydrate:TOC ratios—Analyses of marine sediments for carbohydrates are scarce in the literature, and no previous data are available concerning the carbohydrate content of New York Bight sediments. Our studies were initiated to measure the TCH of the sediments. The distribution as a function of dry sediment weight is shown in Fig. 3: the pattern is strikingly similar to that of TOC with the high concentrations located in the Christiaensen Basin and down the axis of the shelf valley.

To obtain a qualitative source indicator for organic matter in the bight, we report carbohydrate concentration as a percentage of TOC. In doing so, this parameter (TCH: TOC) becomes independent of sediment dry weight or of absolute concentration of organic matter and, as shown later, it more nearly defines the type of organic matter present.

The distribution of TCH:TOC in the sediments of the bight is presented in Fig. 4. Values generally range from 20 to 60 with highs located near the Long Island shore, the head of the shelf valley, and in the shelf valley itself. Since this parameter more nearly typifies the nature of the organic matter, the contours suggest that a source of organic matter exists at the head of the shelf valley and the organic material is being diluted seaward within it.

We propose that the major source of this high carbohydrate organic matter is sewage. The assumptions supporting this proposition are based on literature reports concerning the TCH:TOC ratio of various types of organic matter present in the environment. In the following discussion attention is focused on the TCH:TOC ratio of terrestrial material, phytoplankton, and sewage, since these are the major sources of organic matter in the bight. We attempt to show how the TCH:TOC ratio of each source constituent contributes to the TCH:TOC ratio in sediments of the bight.

Terrestrially derived organic matter is typically low in carbohydrates (Kononova 1966), the TCH:TOC ratio being on the

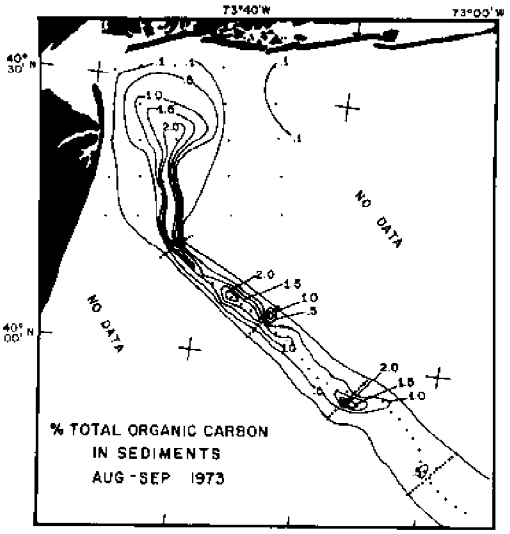

Fig. 2. Distribution of total organic carbon in sediments (dots represent sampling stations).

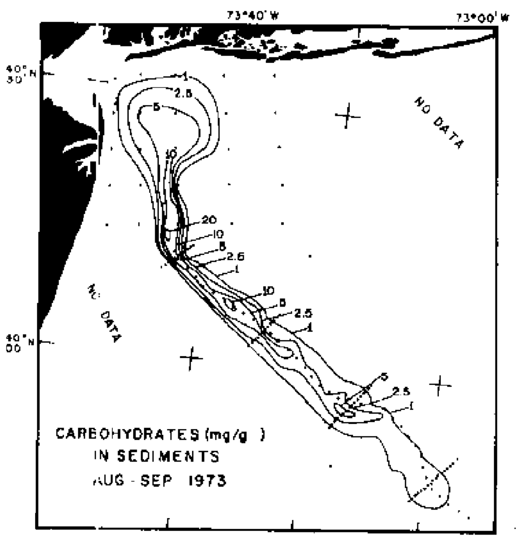

Fig. 3. Distribution of total carbohydrates in sediments.

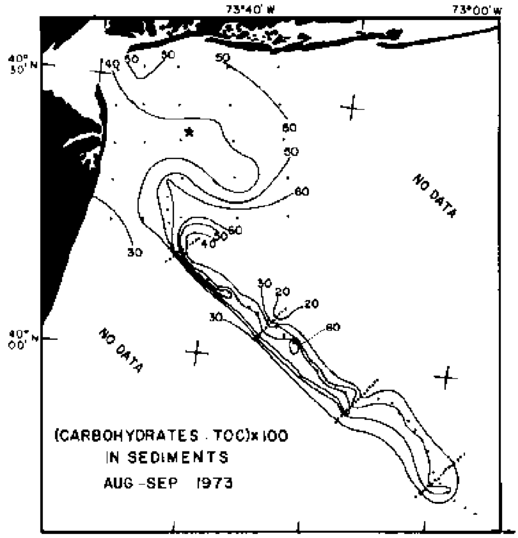

Fig. 4. TCH:TOC ratio in sediments. Star designates sewage sludge dumpsite.

order of 10–20. Waksman and Stevens (1929) showed that typical sawgrass peats had a TCH:TOC ratio of about 5—indication that a significant fraction of the carbohydrates, which are abundant in the living plants, are rapidly degraded bacterially. Waksman and Hutchings (1935) showed that a large fraction of the organic matter in various types of soils could be present as cellulose and hemicelluloses, giving a TCH:TOC ratio of up to 50; however, most of the values ranged from 10–20. Since organic matter undergoes substantial bacterial decomposition in aerobic soils, it is expected that most of the remaining carbohydrates are typically composed of resistant carbohydrates such as cellulose and hemicelluloses, and even this material undergoes bacterial decomposition with time. In fact, only small amounts of cellulose are found in materials older than the Pleistocene (Vallentyne 1963). Soils, therefore, are expected to exhibit a relatively high TCH:TOC ratio in early stages of deposition; however, this ratio decreases to essentially zero after about 10^6 years. This time frame may well apply to the terrestrial material transported to the New York Bight, because, according to Meade (1969), the net transport of fine muds in the Hudson River estuary is landward. The trapping of terrestrial material is rather efficient in this estuary and relatively little is expected to be deposited in the bight. The contribution of terrestrial material to the TCH:TOC in the bight is therefore expected to be negligible.

A major fraction of the particulate matter in the oceans, especially in nearshore areas, is composed of living planktonic plants and animals. These organisms contain substantial amounts of carbohydrates, and the TCH:TOC ratio is often around 30–80 (Strickland 1965). Easily hydrolyzable sugars such as glucose and galactose compose the major fraction of the carbohydrates with more resistant carbohydrates such as cellulose and hemicellulose composing a minute fraction (Parsons et al. 1961).

Handa (1967) observed that, with increasing depth in the oceans, the particulate carbohydrates decrease from 35 μg/liter at the surface to 10 μg/liter at 200 m and that, below this depth, the concentrations do not vary. He pointed out that, within the 200-m surface layer, a major portion of the carbohydrates is mineralized. Only the resistant polysaccharides survive this bacterial decomposition. It is expected that the TCH:TOC ratio would decrease rapidly as phytoplankton cells become part of the detritus and settle to the sediments.

As particulate detritus settles to the sediment, an increase in the decomposition rate takes place, usually within the top few centimeters. The detritus is subjected to bacterial decay, which further decreases the amounts of labile carbohydrates and even resistant carbohydrates. The net result is a decrease in TCH:TOC if the carbohydrates being decomposed are more labile than other fractions of the organic matter (TOC). So, for marine phytoplankton, whose carbohydrates are predominantly labile, a continuously decreasing TCH:TOC will occur as the organisms die and are buried in the sediments. For example, the surface sediments of the Santa Barbara Basin, where organic matter is primarily derived from phytoplankton, have a TCH:TOC ratio of 5 (Degens 1967). Sediments from an experimental Mohole (Rittenberg et al. 1963) had a TCH:TOC of 0.4 at the surface, decreasing to less than 0.1 at a depth of 40 m. In sediments from the eastern continental shelf of the USSR, the TCH:TOC ratio was <2 (Shabarova 1955).

A large fraction of sedimentary detritus in coastal areas is derived from phytoplankton. Malone (in press) has measured the annual productivity in the New York Bight apex as 370 g/m^2 of carbon. This represents the most significant annual input of organic matter to the bight apex when compared to that of the Hudson River (Meade 1969) and of dumped wastes (Gross 1972). However, if phytoplankton was the major source of organic matter to sediments of the bight, the TCH:TOC ratio would be about 10 or less. Sediments of the New York Bight have a larger TCH:TOC ratio, suggesting the presence of a major organic component other than phytoplankton.

Sewage contains a substantial amount of carbohydrates, mostly in the form of cellulose and hemicellulose (Hunter and Heukelekian 1965). Its TCH:TOC ratio is about 30 (Hatcher unpublished data). After sewage is introduced into the marine environment, it undergoes microbial decomposition and a fraction of the organic matter (TOC) is lost. The cellulose and hemicelluloses may not undergo equivalent decomposition in such a relatively short period of time; TCH:TOC may therefore increase as the organic matter undergoes early stages of decomposition. With increased biodegradation, TCH:TOC for sewage-derived organic matter may increase to values ranging from 40 to 60 or greater. These values were estimated based on our observations of the TCH:TOC distribution in the bight. We do not expect a high TCH:TOC to be derived from either terrigenous material or phytoplankton. As stated earlier, the TCH:TOC of sediments derived from each of these sources is observed to decrease with depth in the sedimentary column. Decomposition thereby decreases the TCH:TOC ratio of terrigenous organic matter and phytoplankton-derived organic matter. Our inability to explain high values such as 50 or 60 as being derived from a high TCH:TOC source other than that of sewage leads us to assume that decomposition of sewage increases its TCH:TOC.

Input of sewage to the bight occurs via ocean disposal of sewage sludge, ocean outfalls, and outfalls within the Hudson River. In addition, dredge spoils from the Hudson River estuary, barge-dumped in the bight, are likely to contain large amounts of sewage-derived materials. The total amount of organic matter of sewage origin can thus be calculated, based on the values of Gross (1972), to be roughly half that supplied by primary productivity. However, we believe that, of the organic matter which eventually settles to the sediment, the portion derived from sewage should constitute a major fraction. This is due primarily to the fact that sewage materials are relatively less biodegradable than phytoplankton detritus, and a larger portion of the former is apt to be retained in the surface layers of sediment.

Since the organic matter in sediments of the bight is derived mostly from the two aforementioned sources (phytoplankton and sewage) the resultant TCH:TOC will, in general, be a weighted average of TCH:TOC from each source, the weight factor of each being based on its relative contribution to the sediment organic matter. A high TCH:TOC ratio of 40–50 should indicate that the organic matter contains relatively large amounts of sewage-derived materials. It follows that a TCH:TOC ratio of 20 or less should indicate that the sediment organic matter is composed of phytoplankton organic matter alone or in combination with a smaller amount of sewage. It is obvious, from the previous discussion, that the TCH:TOC ratio is mostly used in a qualitative manner and, at best, a semiquantitative fashion. By semiquantitative, we mean that, in measuring TCH:TOC of a sediment, it is possible to delineate only whether the individual sources contribute a large or small percentage to the organic matter. Before making any more quantitative estimates of source contribution, substantially more information must be obtained on the various sources of organic matter to the bight. First, there must be better estimates of the total inputs of terrigenous matter and phytoplankton to the bight apex. Second, in situ measurements should be made to determine the change in TCH:TOC of various source materials with decomposition. Due to the difficulty of the aforementioned tests, it seems doubtful that all the proper experiments necessary to render the TCH:TOC ratio quantitative will ever be performed. For now, our best hopes are for semiquantitative estimates of source inputs. In the following discussion we address several regions of the bight and attempt to estimate the relative contribution of sewage to the organic matter in sediments based on the TCH:TOC ratio.

Various sedimentary facies were sampled in the bight, and a wide range of sediment types exists. Samples collected near the Long Island shore and in the apex have a high TCH:TOC ratio (40–60) as shown in Fig. 4. The few samples collected in this area do not necessarily give a true indication of the range of values to be expected. Several additional cruises were made to collect more samples from this area. These cruises were code-named substrate monitoring program (SUMP) and substrate inventory sampling (SIS). The SUMP cruise was designed to obtain sediment samples every ¼ mi along two lines: one extending from the sewage sludge dumpsite to within ½ mi of Atlantic Beach, Long Island, and the other extending from the New Jersey coast to Cholera Bank (73°44'W) along 40°25'N lat (Fig. 5). The SIS cruise was designed to obtain sediments on a dense grid along the Long Island shoreline extending from Rockaway Point to Jones Beach (Fig. 6). The TOC and TCH:TOC data for selected samples from both cruises are presented in Table 1. In general, the sands have a TCH:TOC ranging from 50–60 and a TOC <1%. The silty or black muds have a TCH:TOC of 40–50 and a TOC >1%.

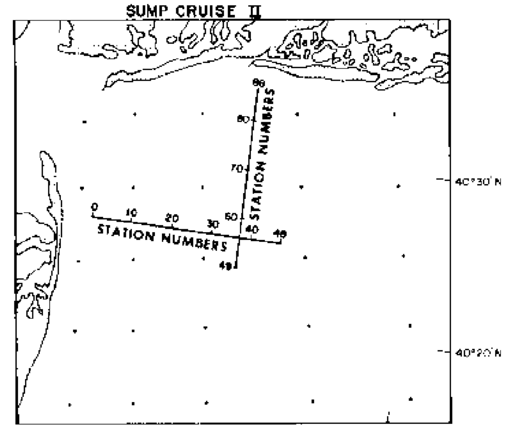

Fig. 5. Tracklines and sample locations for SUMP cruise, January 1974.

The silty muds located in the Christiaensen Basin (SUMP samples 29, 21, 33) and in various sections of the SIS area (SIS samples P-23, P-21, Q-30) were similar to those described (Harris 1974) as having a black "mayonnaisey" consistency which is most likely imparted by the amount of sewage or sewage-derived materials present. The silty muds from both areas have a similar TCH:TOC suggesting that the organic matter is of similar origin. As stated, the high TCH:TOC suggests that the source of this organic matter is predominantly sewage.

The sands located along the Long Island shore typically have a higher TCH:TOC than the silty muds of the same area. Here again, the high TCH:TOC strongly suggests that organic matter is of sewage origin. However, the organic matter (TOC) has most likely undergone more biological decomposition and the ratio has been elevated by this process.

In speculating about the distribution and fate of sewage-derived materials along the Long Island shore and in the apex, we suggest that sewage is being dispersed throughout the apex and that it accumulates mostly in topographic lows. However, some of the sewage particles settle and are trapped within the sands where they undergo more biological decomposition than if they had settled in the mud patches or topographic lows.

We must also point out that while organic matter in the muds may be primarily derived from sewage or sewage sludge, the muds themselves are far from being similar to sewage sludge. Almost an order of magnitude difference exists between the TOC of sewage sludge (≃40%) and that of the silty muds (3–5%). Obviously a substantial amount of decomposition, dissolution, or

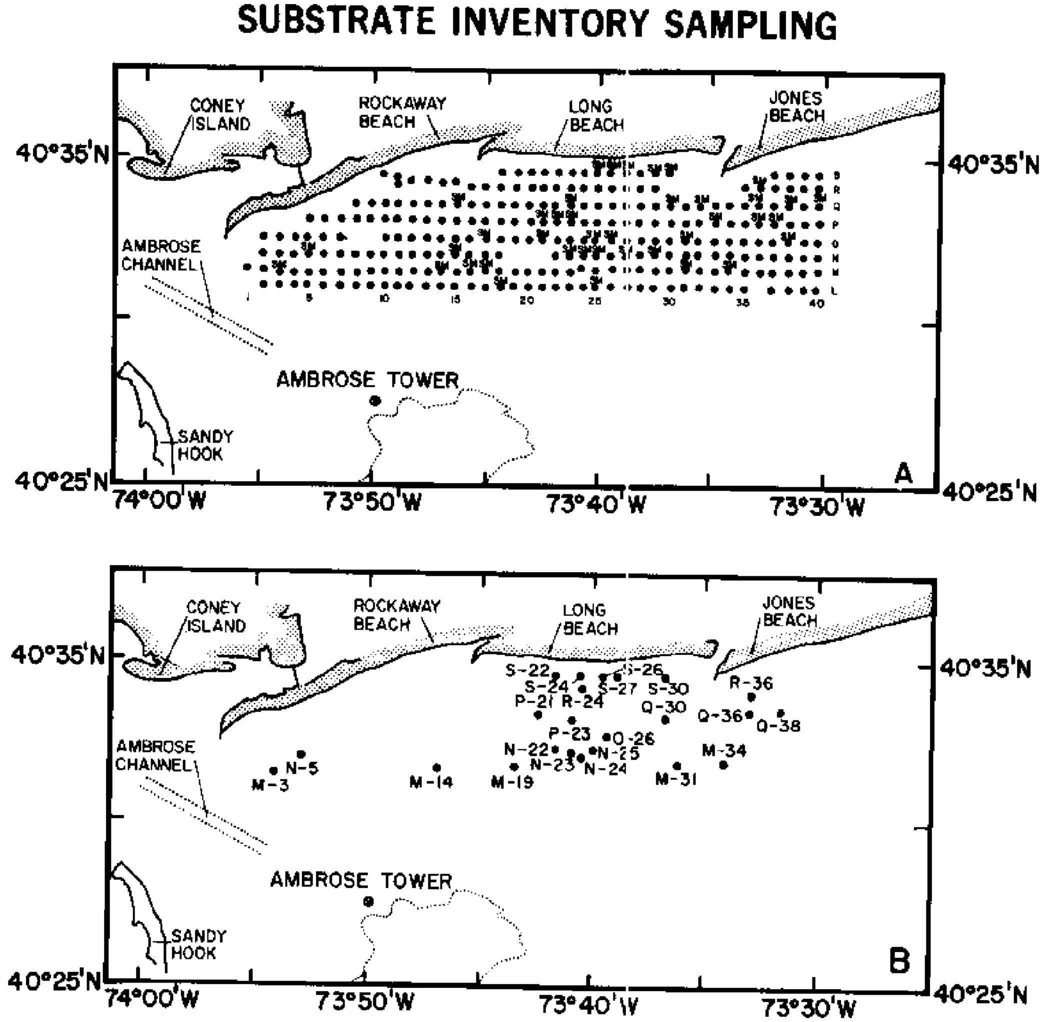

Fig. 6. Sampling locations for SIS cruise (June 1974). A—Proposed grid; B—station locations of samples taken for organic analyses.

dilution of the sewage sludge organic matter must occur before it is incorporated in the sediments. Sewage-derived materials, therefore, comprise only a minor fraction of the dry weight in silty muds. An even smaller amount of sewage-derived materials exists in the sandy sediments.

Sandy sediments near the New Jersey coastline have high TCH:TOC (30–40) values, but not nearly as high as those from the Long Island shoreline. Again, sewage must play a large role in supplying organic matter to the sediments. However, the supply of organic matter may be influenced by the Hudson River, whose plume almost always flows south along the New Jersey coastline. Although the organic matter in the Hudson plume may be primarily of sewage origin, a smaller fraction may consist of terrestrial and phytoplankton organic matter having a lower TCH:TOC ratio. The resultant mixture of sewage and terrestrial material imparts a TCH:TOC of 30–40 to the sediments in contrast to the 50–60 value expected if they were similar to the Long Island sands.

Sands located on the outer continental shelf, adjacent to the Hudson Shelf Valley, have a much lower TCH:TOC ratio, varying from 18–30. We would expect that or-

Table 1. Bulk organic properties of SUMP and SIS sediments.

SUMP			SIS		
Sta.	% TOC	TCH:TOC	Sta	% TOC	TCH:TOC
9	0.46	35	M-3	0.09	52
11	2.99	42	M-14	0.77	53
13	1.40	30	M-19	0.15	57
15	0.64	44	M-31	0.71	57
21	2.45	39	M-34	0.23	43
23	1.96	52	N-5	0.09	53
25	1.64	61	N-22	0.08	54
28	1.26	58	N-23	0.28	52
29	2.85	52	N-24	0.08	44
31	5.08	43	N-25	0.05	50
33	3.85	49	O-26	1.20	40
35	3.04	65	P-21	2.31	47
58	0.35	41	P-23	5.21	46
62	0.41	56	Q-30	1.86	51
66	0.84	52	Q-35	0.18	42
69	0.48	62	Q-38	1.99	41
78	0.05	40			
82	0.17	47			
85	0.12	54			
88	0.10	64			

ganic materials present in the bight apex should contribute little to the organic matter of sediments in this offshore region, due simply to dilution processes as the materials are transported there. Oceanic material, therefore, becomes a more significant contributor to the sedimentary organic matter. The TCH:TOC ratio would, therefore, be expected to be more representative of phytoplankton detritus (ca. 10 or less). The fact that TCH:TOC is higher (18–30) indicates that sewage or terrestrial organic matter is still present.

Within the Hudson Shelf Valley, which extends from just south of the apex area out to the shelf break, the sediments are primarily fine-grained silty muds and TCH:TOC varies from 30–60 (Fig. 4). The TCH:TOC ratio in silty muds at the head of the shelf valley, along the axis, is about 60. As we progress seaward, the TCH:TOC ratios in muds of the axis steadily decrease to values of 30 or less. The fact that TCH:TOC decreases probably indicates that nonsewage-derived components (phytoplankton) constitute a more significant fraction of the organic matter in the muds at greater distances from shore.

The muds located just south of the Christiaensen Basin in the Hudson Shelf Valley have a relatively high TCH:TOC ratio (50–60). The sedimentary environment is similar to that found in the Christiaensen Basin, and the TCH:TOC ratio would be expected to be similar. However, the values are closer to those found in the sands near the Long Island shoreline than to those of muds of the Christiaensen Basin. It seems likely that the organic matter, if derived from the same source as Christiaensen Basin muds, has undergone more decomposition. It is possible that the muds of the Hudson Shelf Valley are derived from muds being deposited in the Christiaensen Basin and that these are being resuspended and redeposited in the shelf valley. This resuspension, transport, and redeposition effectively subject the sewage-derived organic matter to increased decomposition, thereby increasing the TCH:TOC ratio.

As we have pointed out, the TCH:TOC ratio may be used to semiquantitatively measure the relative contribution of sewage to the sediment organic matter of the bight. This measure, however, only relates to that part of the sediment which is organic. The inorganic fraction of the sediment may be derived from different sources. This is especially true for sands where the fine and coarse fractions are transported by different mechanisms and forces. So, in order to estimate the absolute amount of sewage-derived materials in the sediments, we must take into account both the TCH:TOC ratio and the absolute amount of organic matter. We must also attempt to define the amounts of sewage-derived inorganic components, since these may contribute equally as much to the sediment weight. As little is known of inorganic constituents at this time, we must rely on TCH:TOC and TOC to measure semiquantitatively only the contribution of sewage-derived organic matter to the sediment.

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Section 5 The ecosystem and productivity

A conceptual representation of the New York Bight ecosystem[1]

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Abstract

Large scientific enterprises need effective techniques for organizing and relating research findings from different fields. Graphical representations (or models) of the ecosystem can be particularly useful organizational tools. These *conceptual* models help lay the groundwork for detailed mathematical or empirical descriptions of ecological processes; they also demonstrate the scope of the problem being studied. The New York Bight ecosystem model discussed here is based on a descriptive technique developed by H. T. Odum and widely applied by others. The model is constructed from a few basic components which fall into the following categories: energy or mass storage compartments; energy or mass sources; energy or mass flow regulators. These components are pieced together into a comprehensive representation of physical, chemical, and biological processes in the bight. Selected aspects of the representation are examined and, where possible, mass and energy fluxes are estimated.

It can be difficult, in a research effort as large and diverse as the MESA New York Bight Project, to maintain a broad understanding of the system being studied. Individual research grants are designed to sponsor specific studies, usually along traditional disciplinary lines. The results of these diverse research projects must be integrated and compared before the implications of various policy alternatives can be appreciated. In the New York Bight, for example, research efforts are being conducted in physical oceanography, marine geology, water chemistry, and various branches of biology and toxicology. Although these efforts are closely related in concept, they tend to proceed independently in practice.

Here we present a unified description of the New York Bight ecosystem which focuses on the relationships between various physical chemical, and biological processes currently under study. Since our model (or "representation") of the bight is conceptual, it can include processes that are either poorly understood or understood only in a qualitative sense. Much of the value of the model stems, in fact, from its ability to point out where quantitative information is either missing or limited.

A set of graphical symbols developed by Odum (1972) provides a particularly convenient way to describe mass and energy flow in the bight ecosystem. Odum's concepts have been applied in a number of fields, including oceanography (Jannson 1972) and estuarine studies (Kremer and Nixon 1975). His approach is easily ex-

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tended to include circulation and sedimentation processes as well as biological interactions.

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Conceptual representation

The three basic components of Odum's energy-mass flow language are illustrated in Fig. 1 and defined as follows:

> *Energy or mass storage compartments*—Plants, animals, chemical constituents, or any other reservoirs of stored energy or mass which are part of the ecosystem.
>
> *Energy or mass sources*—Solar radiation, wind energy, estuarine inflows, dumped wastes, or any other external inputs of energy or mass.
>
> *Energy or mass flow regulators*—Environmental relationships which control energy and mass flow in the ecosystem; examples include relationships between temperature and nutrient uptake and between turbulence and sediment erosion.

These components can be interconnected to form complete ecosystem models of nearly any desired degree of complexity (Odum 1972). The resulting diagrams indicate the pathways which nutrients, toxicants, or various forms of energy take as they move from one storage compartment to another. Ultimately, we would like to have quantitative estimates of the mass (or energy) fluxes over these pathways, as well as estimates of the concentrations of mass or energy stored in particular compartments. For the moment, however, we will consider only the overall structure of the ecosystem, as illustrated in Fig. 2.

This figure is a highly aggregated, broad-brush picture of the New York Bight ecosystem, constructed almost entirely from the three basic components defined above. The lines connecting storage compartments, sources, and regulators in the diagram represent the dominant pathways taken by energy or mass flowing through the ecosystem. The resulting network can be broken into a physical oceanography section (on the left) and a biological, geological, and chemical oceanography section (on the right).

Physical oceanography—The dominant sources of fluid energy in the bight enter through the atmospheric and oceanic boundaries of the water mass. These include kinetic energy (transmitted by winds and offshore currents), potential energy (generated by hydraulic and atmospheric pressure fields), and thermal energy (from solar radiation and offshore currents). Kinetic and potential energy in the water mass are continually interchanged, a process represented diagrammatically by the "energy exchange loop" in Fig. 2. The small grounding symbols at various points in the figure indicate energy losses to heat.

The energy viewpoint adopted is conceptually convenient because it relates quite naturally to spectral representations of velocity. Temporal and spatial distributions of fluid energy (computed as velocity spectral densities) provide a means for assessing the relative importance of advective

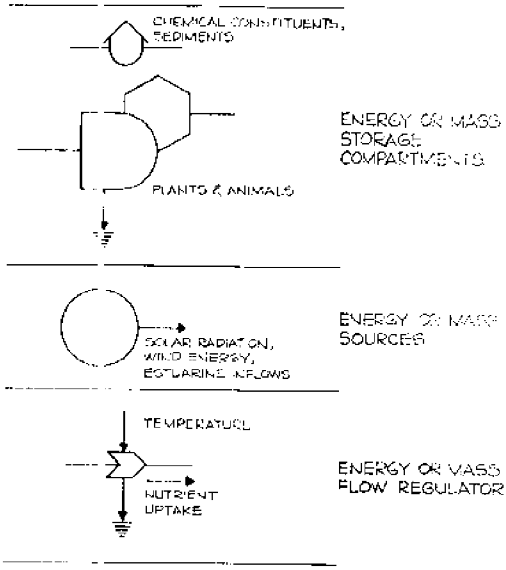

Fig. 1. Basic components of Odum's energy-mass flow language.

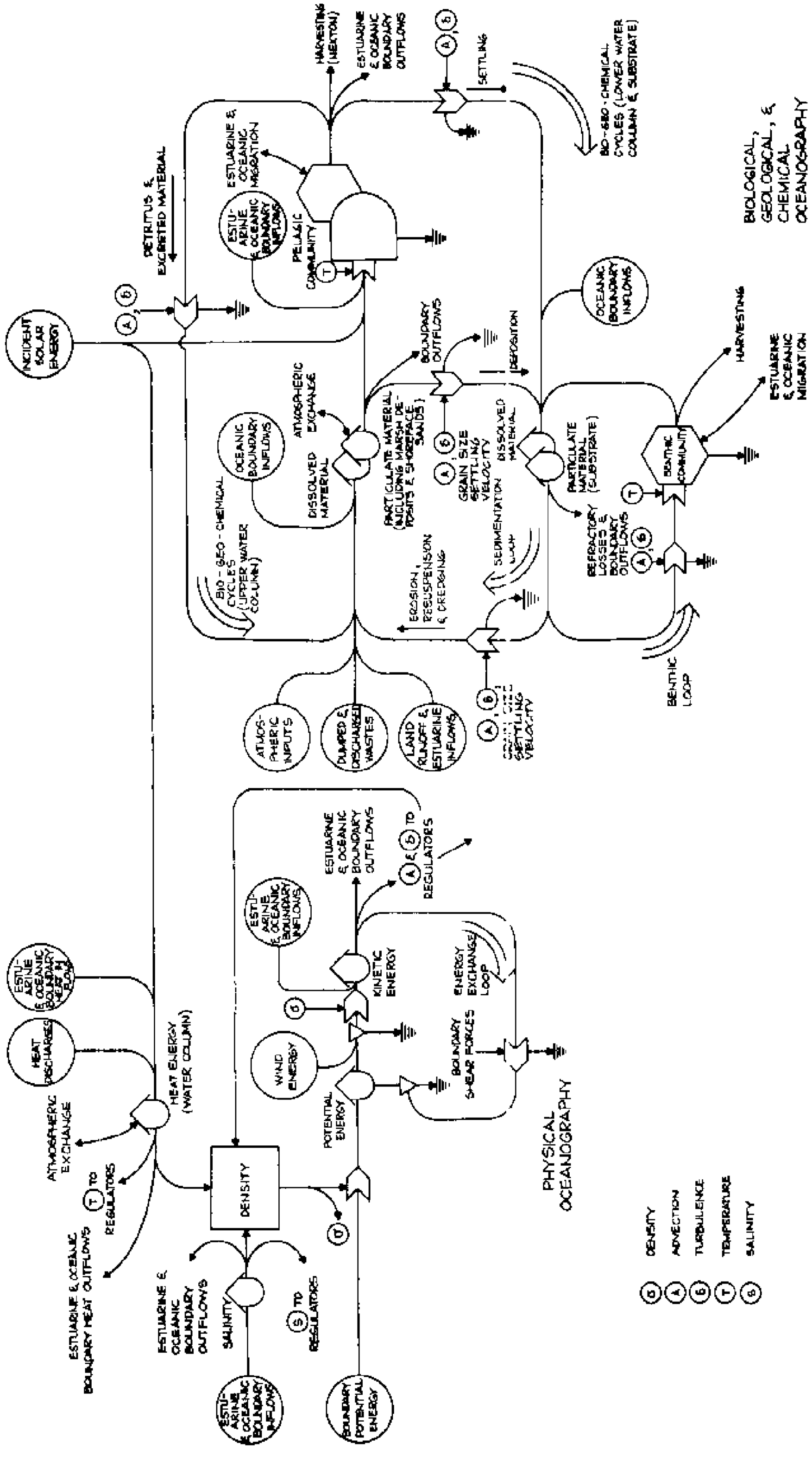

Fig. 2. Generalized diagram of the New York Bight ecosystem.

and dispersive processes. This is why the advection and turbulence variables shown in the representation originate from the kinetic energy compartment.

Biological, chemical, and geological oceanography—The major external sources of mass entering the bight can be grouped into four categories: Dumped and discharged wastes, land runoff and estuarine inflows, atmospheric inputs, and inflows across oceanic boundaries. The relative importance of these sources depends on the constituent being considered (dissolved nutrients, trace metals, inorganic sediment, etc.). Our aggregated representation combines all nonliving matter into two storage compartments—one for the upper water column and one for the lower water column and substrate. Interchange between the upper and lower portions of the water column is regulated primarily by vertical advection and turbulence.

Figure 2 shows that dissolved and particulate materials may pass through either the pelagic or the benthic biological communities. Of course, specific uptake mechanisms and transport paths differ significantly, depending on the constituent being considered. Figure 2 must therefore be disaggregated into a number of constituent-oriented diagrams before it can be applied to specific problems. A comprehensive model would have to include detailed diagrams of at least organic carbon (detritus and dissolved organic carbon), oxygen, nutrients (N, P, and Si), toxicants (trace metals, hydrocarbons, pesticides, etc.), inorganic sediment, and sulfur and iron. The two examples discussed below illustrate how specific constituent models can be developed from the general concepts outlined in Fig. 2. Other examples are presented in McLaughlin et al. (1975).

Quantification of fluxes and input rates

Next we review two relatively simple examples to demonstrate the practical usefulness of conceptual models. The first is concerned with the carbon-oxygen budget of the New York Bight apex. We do not offer any new results but, rather, only wish to show how existing data can be used to estimate fluxes and input rates defined conceptually. The second, which qualitatively outlines the mass budget for a "typical" toxicant, is presented to indicate the versatility of Odum's modeling approach and to show where some serious gaps in water chemistry and biological research still exist.

Apex carbon-oxygen budget—A convenient carbon-oxygen budget can be constructed from the bio-geo-chemical section of Fig. 2. This budget (Fig. 3) includes three major constituents—organic carbon, dissolved oxygen, and oxidizable inorganic compounds (ammonia, nitrite, acid wastes, and other inorganic substances that exert a chemical oxygen demand)—as well as the biological community. Together, these constituents determine the rates of oxygen production and consumption in the water column. A complete representation of the carbon-oxygen budget would include estimates of all input rates, fluxes, and in situ concentrations identified in Fig. 3 (as functions of both time and space). Since this is, at present, an unrealistic expectation, we must settle for rough temporal and spatial averages which can be computed for a few of the better understood processes. The remaining gaps in the budget show where further research efforts are needed. For illustrative purposes, we will consider only mean annual flux and concentration estimates averaged over the entire 1,400 km^2 (35 km × 40 km) of the apex.

Crude input rate estimates can be computed for each of the four exogenous organic carbon sources shown in upper left of Fig. 3. Segar and Berberian (1976) estimate that dumped and discharged wastes (primarily sewage sludge and dredge spoil) together contribute about 0.15×10^6 tonnes/yr of oxidizable carbon (C). Land runoff and estuarine inflows probably fall in the range between 0.03 and 0.12×10^6 tonnes C/yr, if we assume an estuarine BOD concentration of from 5–20 mg/liter and an average estuarine flow of about 20 km^3/yr (Bue 1970). Particulate atmospheric inputs are difficult to estimate, but it is unlikely that total solids exceed 0.25×10^6 tonnes/yr (Gross 1974). Since the organic carbon por-

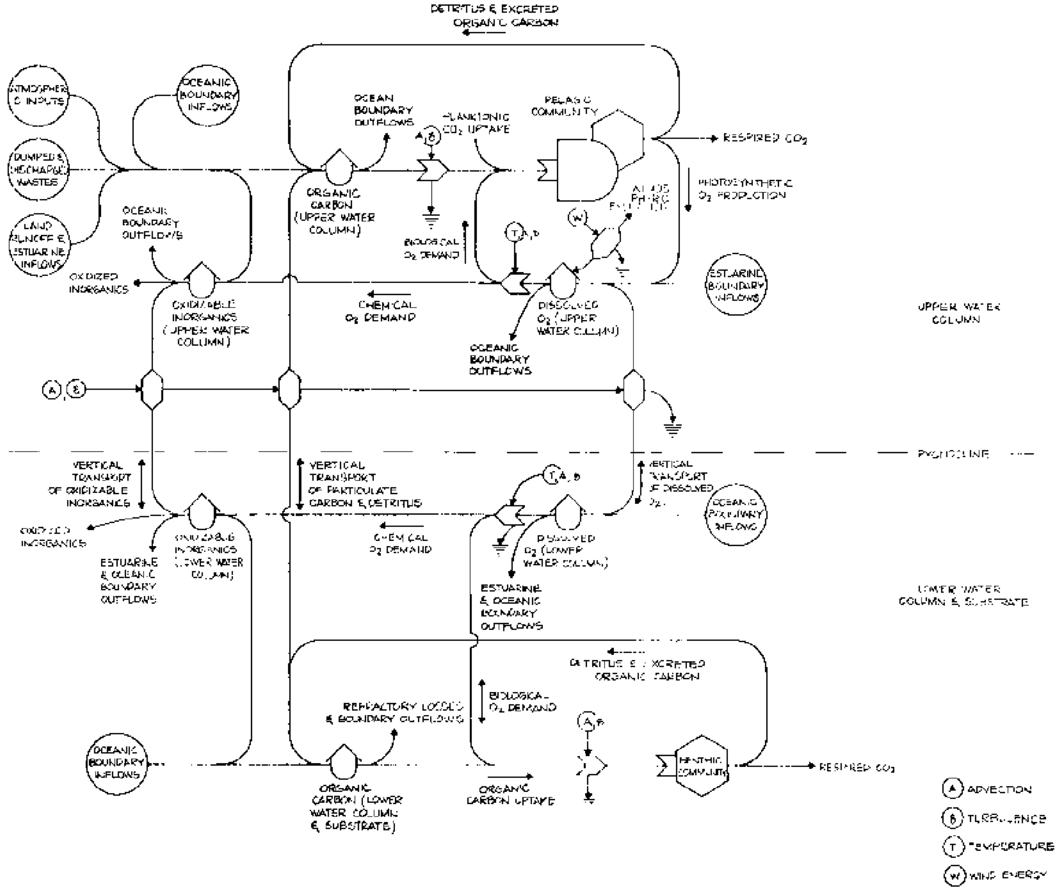

Fig. 3. Carbon-oxygen budget for the New York Bight apex.

tion of atmospheric particulates is probably only a few percent, 0.03 × 10^6 tonnes C/yr is an upper bound on organic carbon inputs from the atmosphere. Unfortunately, little can be said about advective inflows and outflows of organic carbon across the oceanic boundaries of the apex. For now, this portion of the budget must be left undefined.

The most important sources of organic carbon in the apex are primary producers—both photosynthetic plankton and nonphotosynthetic bacteria. Photosynthetic organisms are represented in Fig. 3 by the bullet-shaped symbol in the pelagic community. Hexagons represent nonphotosynthetic organisms in both the pelagic and benthic communities. Average annual photosynthetic production rates for the apex have been computed before (e.g. Ryther and Yentsch 1958; Malone 1976). Most of these estimates are near 0.4 × 10^6 tonnes C/yr (300 g C/m^2/yr in the 1,400-km^2 apex), a value significantly larger than any of the exogenous inputs discussed above.

Organic carbon input to, or produced in, the upper water column is either consumed by the pelagic community or transported downward to the seabed (*see Fig. 3*). Unfortunately, it is difficult to estimate the relative magnitudes of the uptake and vertical transport fluxes since the effects of turbulent mixing and stratification vary so throughout the year. At best, we can predict that the total flux of organic carbon out of the upper water column will equal the sum of all exogenous and in situ sources, about 0.80 × 10^6 tonnes C/yr. This figure neglects the unknown effect of advection across the oceanic boundaries of the apex.

The organic carbon fraction reaching the seabed either accumulates in the bottom sediments or is consumed by the benthic community. A certain portion of the material consumed is, of course, recycled as detritus while the remainder is excreted, mostly as carbon dioxide. Net benthic organic carbon consumption can be estimated from seabed oxygen measurements (Thomas et al. 1976; Segar and Berberian 1976). Available data indicate that average seabed oxygen consumption in the apex is about 0.5×10^3 tonnes/yr, equivalent to 0.17×10^3 tonnes/yr of organic carbon oxidized (using the respiration equation of Redfield et al. 1963). This value is orders of magnitude smaller than the total organic carbon input rate mentioned above, an indication that aerobic benthic consumption plays a relatively small role in the overall carbon-oxygen budget of the apex.

The organic carbon fluxes estimated above are summarized in Table 1. It is apparent that the most important uncertainties in the carbon-oxygen budget are associated with nonphotosynthetic primary production, horizontal and vertical transport, anaerobic benthic consumption, and pelagic uptake. All of these fluxes must be studied more extensively before a complete carbon-oxygen model of the apex can be constructed. Although uncertainties in estuarine inflows and atmospheric inputs certainly exist, they are probably of secondary importance from a mass budget point of view.

Table 1. Summary of organic carbon flux estimates for the New York Bight apex (in 10^6 tonnes C/yr, dry wt).

Exogenous inputs and outputs	
Dumped and discharged wastes	0.15
Land runoff and estuarine inflows	0.05–0.20
Atmospheric inputs	0.03
Oceanic boundary flows	unknown
Total	0.38(?)
In situ sources	
Photosynthetic production	0.40
Nonphotosynthetic production	unknown
Vertical transport	
Net downward	<0.80
Consumption	
Pelagic uptake	<0.80
Aerobic benthic uptake	0.00017
Anaerobic benthic uptake	unknown

Techniques similar to those outlined above can also be used to estimate the inflow and consumption rates of oxygen identified in Fig. 3. The considerable temporal and spatial variability associated with processes such as reaeration and organic carbon oxidation make accurate estimation of average oxygen fluxes difficult; however, even rough figures can be instructive.

Various quasi-empirical equations have been developed for estimating oxygen flux across the air–water interface. Kanwisher's (1963) reaeration formula is a typical example:

$$Q_a = K_r(C^* - C), \qquad (1)$$

where

Q_a = atmospheric exchange rate (mg/m^2/s);

K_r = $D/[a + b(w^{1/2})]$ = reaeration coefficient (liter/m^2/s),

D = molecular diffusivity of oxygen (m^2/s),

w = wind speed (m/s),

a, b = empirical coefficients;

C^* = oxygen saturation concentration at local temperature and pressure (mg/liter);

C = dissolved oxygen concentration averaged over the mixed surface layer (mg/liter).

If a reaeration coefficient of 0.045 liter/m^2/s (Kanwisher 1963; Havens et al. 1973) and an average surface oxygen concentration of 6.5 ml/liter are assumed (Corwin 1970), the resulting atmospheric exchange rate for the apex is about 1.8×10^6 tonnes of oxygen/yr (from the air to the water).

Equation 1 is a precise mathematical description of the atmospheric exchange regulator symbolized in Fig. 3 by a two-way arrow. Similar equations can be used to define the other regulators shown in Fig. 3. Temperature-uptake (T) regulation can, for example, be described with classical exponential relationships having the general form

$$Q_u = Q_{u0} e^{a(T - T_0)}, \qquad (2)$$

where Q_u = oxygen or organic carbon uptake rate at temperature T for a particular species (mg/m/s); T = water temperature (°C); Q_{uo}, a, T_0 = empirical coefficients. Regulation by advection and turbulence (A, δ) is, on the other hand, described by the classical mass transport equation (Neumann and Pierson 1966). Such functional expressions, which are the building blocks of computerized simulation models, should be interpreted pragmatically since they are really only mathematically convenient summaries of limited observations in the field. They do, however, provide a means for developing quantitative estimates of at least some fluxes and input rates in the ecosystem.

The major exogenous sources of oxygen (other than reaeration) shown in Fig. 3 are estuarine inflows and oceanic boundary inflows. Surface estuarine waters are nearly saturated during most of the year, implying an inflow of about 0.15 × 10^6 tonnes/yr of oxygen (assuming an average saturation concentration of about 7.0 ml/liter and an estuarine flow of 20 km^3/yr). A comparable amount of oxygen probably leaves the apex in surface waters flowing across oceanic boundaries. Since bottom waters in some areas of the apex have oxygen levels well below saturation, there may be a net oceanic inflow of oxygen at depth. This flux is, however, difficult to estimate, if it exists at all.

The photosynthetic organic carbon production mentioned earlier is, of course, matched by an equivalent in situ photosynthetic production of oxygen. Using the ratios of Redfield et al. (1963), we can compute an average annual oxygen production of about 1.4 × 10^6 tonnes/yr.

Although nearly all oxygen input to the ecosystem enters through the upper water column, a certain portion moves downward through vertical advection and mixing. Unfortunately, this vertical transport is difficult to estimate, even on an aggregate basis. It is certainly restricted by summer stratification and aided by winter mixing and is probably much less than the sum of all oxygen inflows (about 3.35 × 10^6 tonnes/yr). More precise quantitative statements cannot be made at this time.

Oxygen demands in the water column and on the seabed are exerted both by reduced inorganic compounds ("oxidizable inorganics") and by biological oxygen consumption. The most important oxidizable inorganics are probably ammonia, nitrite, and various sulfide and ferrous by-products of anaerobic decomposition. Strictly speaking, these materials should be treated in separate mass budgets constructed in much the same way as the organic carbon budget. Fluxes into and out of the oxidizable inorganic compartments of Fig. 3 could then be deduced from complete mass balance computations based on the nitrogen, sulfide, and iron cycles in the ecosystem. Details are discussed in McLaughlin et al. (1975).

Chemical oxygen demands in the bight apex cannot be estimated from the limited data now available. Although some rough figures for nitrogenous demands can be deduced from ammonia and nitrite concentrations in the water column, these represent only a fraction of the total inorganic demand. Oxygen demands from anaerobic by-products resuspended from the bottom sediments during periodic winter storms may be much more significant on an annual basis (Stanier et al. 1963), although existing measurements do not provide conclusive evidence.

Oxygen consumption by the pelagic biological community depends, of course, on the amount of organic material oxidized in the water column. Table 1 indicates that the latter figure is less than 0.80 × 10^6 tonnes C/yr, implying an upper limit of 2.7 × 10^6 tonnes/yr for pelagic oxygen consumption (Redfield et al. 1963). A more accurate estimate is, unfortunately, not currently available. If the seabed oxygen demand reported by Thomas et al. (1976) is presumed to be primarily biological consumption, then the reported value of 0.5 × 10^6 tonnes/yr gives a good estimate of average oxygen uptake by the benthic community.

Table 2 summarizes these oxygen flux estimates in a format similar to that used for Table 1. It is evident that several major areas of uncertainty exist in the oxygen budget—particularly the magnitude of oceanic boundary flows, the role of vertical transport and turbulent mixing, and the role of chemical oxygen demands exerted by the periodic resuspension of anaerobic by-products in the bottom sediments. Biological oxygen demands are also uncertain, primarily because of uncertainties in the organic carbon budget. This close connection between the carbon and oxygen budgets is the major reason for combining them into a single conceptual diagram (Fig. 3).

We believe the above analysis illustrates how a conceptual model can be used to guide mass budget computations as well as to identify major research needs. An attempt has been made to estimate the organic carbon and oxygen fluxes defined on Fig. 3 and functional relationships have been proposed for certain important regulators. Extensions of this approach could (when combined with a coordinated field program) yield a complete model of the carbon-oxygen budget in the apex. Such a model could be used to evaluate the impacts of various waste disposal alternatives in the bight and to investigate the fundamental dynamics of the ecosystem.

Toxicant mass budgets—Toxicants and biological contaminants (particularly trace metals, pesticides, and hydrocarbons) pose a more complex modeling problem than the carbon-oxygen budget. Toxicological effects depend on the concentration of the toxicant in an organism's tissue as well as on the ambient concentration in the water column. These effects are apparently regulated by temperature and dissolved oxygen levels, although experimental results are often not definitive (Pickering 1968; Clubb 1974). To complicate matters further, several toxicants acting together may have either synergistic or antagonistic effects, making single-constituent analyses potentially misleading (Gray and Ventilla 1973). The conceptual model of toxicant flow shown in Fig. 4 illustrates some of these factors in an elementary way. Note that storage compartments are defined within the biological communities as well as in the water column and sediments. Regulators dependent on both temperature and oxygen level are also included at various critical points.

Table 2. Summary of dissolved oxygen flux estimates for the New York Bight apex (in 10^6 tonnes/yr).

Exogenous inputs and outputs	
Atmospheric exchange (reaeration)	1.80
Estuarine inflows	0.15
Oceanic boundary flows (net)	unknown
In situ sources	
Photosynthetic production	1.40
Vertical transport	
Net downward	$\leqslant 3.35$
Consumption	
Chemical oxygen demands	unknown
Pelagic consumption	<2.70
Benthic consumption	0.0005

If the ambient and biomass toxicant concentrations indicated in Fig. 4 are to be distinguished from one another, a number of uptake, excretion, respiration, and decomposition (weathering) rates must be defined. Unfortunately, these important rates and the regulatory processes that govern them have rarely been quantified, even for toxicants of great public concern. The experimental and analytical problems involved are considerable, particularly when in situ measurements are desired.

Exogenous toxicant inputs are probably less difficult to evaluate than biological uptake and excretion rates, although many practical problems do exist. Recent studies have begun to define fluxes of certain toxicants moving across the Sandy Hook–Rockaway transect of the apex (Mueller et al. 1976; Alexander et al. 1974) and some preliminary estimates of atmospheric inputs have been made (Gross 1974). Trace metal analyses of dumped and discharged wastes have provided fairly accurate estimates of these particular exogenous inputs (Pararas-Carayannis 1973; Mueller et al. 1976). Oceanic boundary inflows and vertical transport rates in the water column are, however, essentially unknown.

Computational techniques similar to those used in the carbon-oxygen budget could be used for toxicants if sufficient field data were available. Research efforts eventually may provide fairly good estimates of exogenous input rates for certain

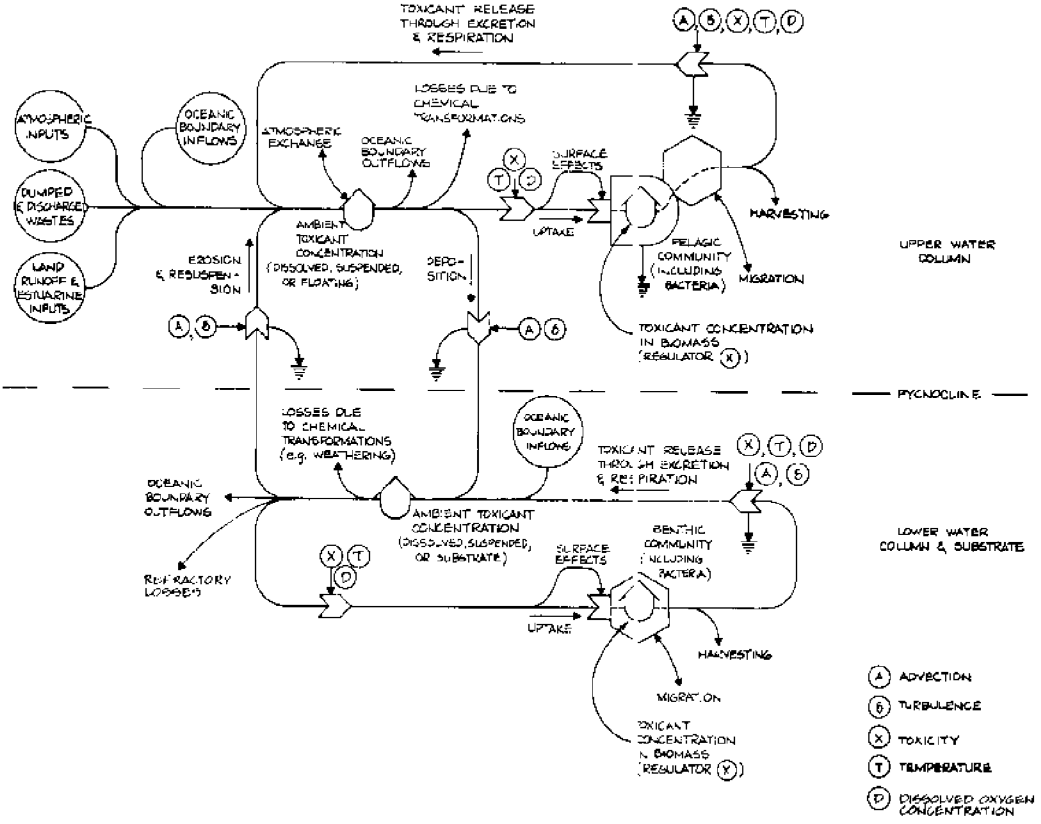

Fig. 4. Elementary mass budget diagram for a typical toxicant.

trace metals and hydrocarbons and for a few selected pesticides. Biological responses to toxicant concentrations in the water column and sediments will, however, be difficult to define, particularly when long term chronic effects are considered. Thus, it appears that quantitative models of toxicant cycling in the ecosystem will not be available for some time, and diagrams such as Fig. 4 must remain qualitative and conceptual.

Prospects for ecological modeling in the New York Bight

The mass budget computations and occasional equations presented above suggest that our conceptual representation could form the basis for one or more mathematical models of the bight ecosystem. Therefore, we conclude our discussion with some brief comments on the short term prospects for predictive ecological modeling in the bight. Three general areas are of particular interest: mass transport models (chemical and geological oceanography), hydrodynamic models (physical oceanography), and ecological-toxicological models (biological oceanography). Each of these is briefly reviewed below.

A survey of available information indicates that rough mass budgets for organic carbon, oxygen, nitrogen, and suspended sediments can be constructed now, at least for the apex region. While there may be some disagreement about the magnitude of specific fluxes or concentrations, data being compiled from the MESA Program will resolve many areas of uncertainty. One of the important advantages of the mass budget approach is that it gives a feeling for which processes are important on a long term or areally averaged basis. This helps guide the modeler in making simplifications or generalizations in his mathematical representation of the ecosystem. Aggregated mass budgets cannot, however, provide the kind of detailed information needed to predict the spatial distribution of materials within the apex or the movement of materials away from the dump sites. Such predictions can only be made with mass transport models based on detailed three-dimensional velocity fields which are not available for the bight.

Although comprehensive three-dimensional hydrodynamic models have been developed (*see* Leendertse et al. 1973), their boundary condition requirements are impractical for open coastal areas such as the New York Bight. The most promising solution to this dilemma may lie in a judicious combination of current meter measurements and mathematical modeling. Hydrodynamic models similar in concept to Leendertse's could, for example, generate consistent velocity fields by interpolating between current meter records obtained from a few carefully selected sampling stations on the shelf. Extensive meteorological information (perhaps obtained from satellite observations) will be needed even in this application, and it is not yet clear whether satisfactory results will be obtained soon.

Once the mass transport-hydrodynamic modeling problem is solved, an even more difficult ecological modeling task must be confronted. This task will be particularly challenging in toxicology, where comparatively little experimental information is available. Perhaps the first and most important need in toxicology is to define which toxicants are potentially hazardous or are otherwise important in the bight. The field of candidate toxicants must be reduced to a manageable size before any meaningful experiments or modeling exercises can be conducted. When this is accomplished, a concerted effort can be made to define the uptake, excretion, and detoxification rates identified earlier in Fig. 4. Quantitative rate estimates will allow modelers to predict toxicant accumulations in the water column, the sediments, and in various organisms for a range of input levels. Of course, the relationship between toxicant concentrations and adverse biological effects (including chronic sublethal effects) will still have to be established. Accurate toxicological models will be particularly difficult to develop because of the important roles of synergism, antagonism, and chemical speciation.

The problems associated with ecological modeling are considerable, but progress is being made. Water quality and thermal transport models have been successfully tested in freshwater and estuarine applications (Chen and Orlob 1972; DiToro et al. 1971). Many primary productivity models have also been developed over the last few years—some have, in fact, recently been used in the New York Bight (Malone 1976).

Clearly, work will have to proceed in parallel in several different fields before a comprehensive mathematical model of the New York Bight can be constructed. This work will be facilitated if modelers and experimentalists learn to cooperate on a regular basis. Mathematical modeling should, after all, be viewed as one more experimental tool to complement rather than supplant fieldwork. When this viewpoint is accepted, specialists in all fields will benefit.

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Phytoplankton productivity in the apex of the New York Bight: Environmental regulation of productivity/chlorophyll *a*[1]

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Abstract

Phytoplankton productivity within a 600-km^2 area of the New York Bight apex ranged from a December minimum of 0.1 g C m^{-2} d^{-1} to a June maximum of 6 g C m^{-2} d^{-1}. Netplankton productivity peaked in February (1.7 g C m^{-2} d^{-1}) and June (2.2 g C m^{-2} d^{-1}); nanoplankton productivity was highest in June (2.1 g C m^{-2} d^{-1}) and July (3.7 g C m^{-2} d^{-1}). Productivity above 2 g C m^{-2} d^{-1} was restricted to within 20 km of the mouth of the Hudson-Raritan estuary. Netplankton and nanoplankton accounted for 41% and 59% of the annual phytoplankton production (370 g C m^{-2}).

The supply of dissolved inorganic nitrogen from the Hudson estuary exceeded phytoplankton demand except during June, July, and August when regeneration within the apex was an important source of nitrogen. No evidence of nitrogen-limited phytoplankton growth was found. Evidence shows that chlorophyll *a* specific phytoplankton productivity was regulated by light and temperature. Ocean dumping had no observable effect on phytoplankton assimilation numbers or on the distribution of environmental factors regulating phytoplankton growth in the apex.

The apex of the New York Bight (Fig. 1) is a transition region between an estuarine and a continental shelf, coastal system. Its waters are a varying mixture of three water masses (Ketchum 1967): estuarine water characterized by low salinity (<30‰), high nutrient concentrations, and high concentrations of suspended detritus (Panuzio 1966; Busby and Darmer 1970; Garside et al. 1976); surface coastal water of moderate salinity (31‰) and low nutrient concentration; oceanic water of high salinity (34‰) and moderate nutrient concentrations. Low salinity estuarine water is usually concentrated in a narrow band along the New Jersey coast when the water column is well mixed (late fall–early spring) and more uniformly distributed over the surface when the water column is thermally stratified during summer (Ketchum et al. 1951).

Large quantities of dissolved and suspended waste materials from the New York-New Jersey metropolitan area are transported into the apex via estuarine runoff and ocean dumping. The net volume transport of estuarine water into the apex is relatively constant over most of the year and averages about 2 × 10^8 m^3 d^{-1} (Abood 1974; O'Connors and Duedall 1975). About 1.2 × 10^4 m^3 d^{-1} of sewage sludge and 2.5 × 10^4 m^3 d^{-1} of dredge spoils are dumped at two locations (Fig. 1) 20 km from the mouth of the estuary (U.S. EPA 1974). Estuarine circulation and sewage sludge dumping are major sources of allochthonous dissolved inorganic nutrients. The high nutrient concentrations typical of estuarine water are due primarily to discharge of sewage wastes into the lower Hudson estuary (Garside et al. 1976). About 3 × 10^6 m^3 d^{-1} of domestic wastes, or 6.1 × 10^7 kg N yr^{-1} and 0.6 × 10^7 kg P yr^{-1}, are discharged into the lower Hudson estuary (Tofflemire and Hetling 1971; Howells et al. 1970). O'Connors and Duedall (1975) and Garside et al. (1976) estimated that the net transport of dissolved inorganic nitrogen (DIN) into the apex from the estuary approximates 5.5 × 10^7 kg N yr^{-1} (90% of the total input to the lower estuary due to sewage discharge). The disposal of sewage sludge 20 km from the mouth of the estuary (Fig. 1) results in an NH_3-N input of about 2.6 × 10^6 kg yr^{-1}, 5% of the net DIN flux from the estuary (O'Connors and Duedall 1975). This is a highly localized input relative to estuarine

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[1] This research is part of the Marine EcoSystems Analysis (MESA) Project, NOAA contract 03-4-043-310.

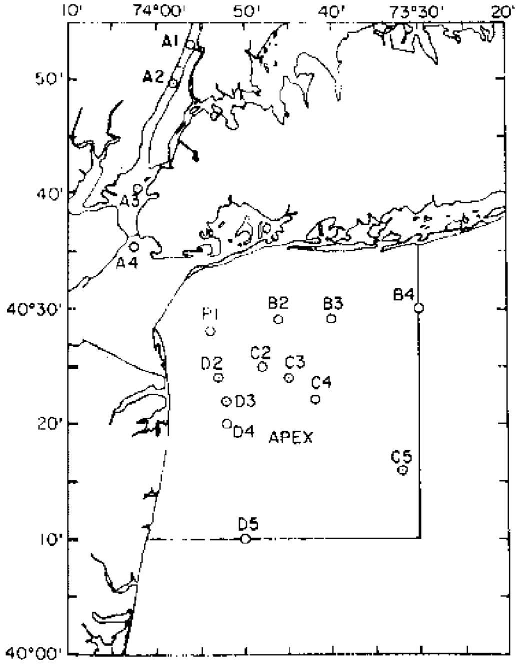

Fig. 1. Station locations in the lower Hudson estuary (A1–A4) and apex of the New York Bight: C2, C3, and C4 are in the sewage sludge dump; D2, D3, and D4 are in the dredge spoil dump.

discharge and can give rise to transient anomalies of high NH_3-N concentration (O'Connors and Duedall 1975).

Materials transported into the apex via ocean dumping and estuarine circulation can affect phytoplankton productivity by nutrient enrichment (stimulating growth), by increasing the attenuation rate of downwelling light (limiting growth), or by exposing phytoplankton cells to toxic materials (inhibiting growth). Using natural phytoplankton populations as bioassay organisms in batch culture experiments, Young and Barber (1973) were unable to demonstrate inhibition of exponential growth by waters from the dump sites relative to water collected outside the dump sites, although the lag phase of the growth response was longer than the controls. Ketchum (1967) speculated that phytoplankton growth might be light limited due to the high turbidity of estuarine water. Based on nutrient enrichment-bioassay experiments with *Skeletonema costatum* and on the distributions of DIN and PO_4-P, Ryther and Dunstan (1971) concluded that phytoplankton growth in the apex is nitrogen limited.

Studies of phytoplankton productivity in the apex have been limited to the coast of Long Island (Mandelli et al. 1970). Mean primary productivity at the surface decreased seaward from 0.35 g C m^{-3} d^{-1} in estuarine embayments to 0.16 g m^{-3} d^{-1} offshore 8 km with peaks during summer and fall. Photic zone productivity in coastal waters (within 8 km of the coast) varied from 0.28 to 2.90 g C m^{-2} d^{-1} giving an annual mean of about 1.14 (420 g C m^{-2} yr^{-1}). Variations in the assimilation number (light-saturated photosynthesis per unit Chl *a*) were related to temperature and species composition; dinoflagellates had higher assimilation numbers than did diatoms, especially at high temperatures.

Outside the apex, primary productivity in the bight has been estimated by Ryther and Yentsch (1958). Annual production decreased from 160 to 100 g C m^{-2} as water column depth increased from less than 50 m near the New Jersey and Long Island coasts to greater than 1,000 m near the shelf break. Inshore, productivity exceeded 0.5 g C m^{-2} d^{-1} from December through April; offshore values above 0.5 occurred only from March through April. The higher levels of annual production inshore presumably reflect proximity to land, the limits imposed by water column depth on vertical mixing, and more rapid nutrient recycling in shallow waters.

The environmental factors potentially regulating phytoplankton productivity fall into two categories: those directly affecting productivity per unit biomass (growth rate) and those directly affecting biomass. In shallow, open systems like the apex, the distribution of phytoplankton biomass as it is produced is affected primarily by advection, turbulence, and grazing. Environmental factors that could be involved in the regulation of phytoplankton growth rates include temperature, nutrient supply rates, the availability of light energy, and the presence of toxins. If materials transported into the apex by estuarine circulation or ocean dumping affect phytoplankton productivity, the spatial and temporal distributions of phytoplankton growth rates will reflect these effects. This paper describes monthly variations in phytoplankton productivity within the apex area subject to estuarine discharge and ocean dumping. The environmental factors responsible for observed variations in chlorophyll *a* specific primary productivity are identified, and the effects of ocean dumping on phytoplankton growth are evaluated within this context.

I thank C. Garside for performing the nutrient analyses, S. Patterson and M. Chervin for their shipboard and laboratory assistance, and the Institute of Oceanography of the City University of New York for providing ship-time on the RV *Commonwealth.*

Methods

Environmental factors—Salinity was measured with an induction salinometer and temperature with protected reversing thermometers. Dissolved inorganic nutrient analyses were performed using a Technicon AAII three-channel autoanalyzer. Standard manifolds were used for nitrate, nitrite, ammonia, phosphate, and silicate determinations (Strickland and Parsons 1972). Each method was calibrated and checked for linear response using a series of replicate standards before samples were analyzed, and replicate standards were run every 30 samples during the analyses. Precision was ±0.25 μM or better for all methods.

Continuous recordings of incident solar radiation were obtained with an Eppley pyranometer. Daily integrals of photosynthetically active radiation in gcal cm^{-2} d^{-1} were estimated by measuring the area under the curve with a planimeter and applying a correction factor of 0.5 during cloudless periods (Jerlov 1968). Light percent depths were estimated from Secchi disk readings, and mean photic zone light energy was calculated from the expression:

$$I' = I_0 (1 - e^{-k d'}) / k d',$$

where I' = mean photic zone light energy (gcal cm^{-2} d^{-1}), I_0 = photosynthetically active radiation incident at the sea surface (gcal cm^{-2} d^{-1}), k = mean extinction coefficient (1.7/Secchi disk reading), and d' = 1% light depth. The 1% light depth was used in this expression rather than mixed-layer depth because of limited water column depths in the study area, and discrete sample depths were selected on the basis of light attenuation rather than water column stratification.

Measurements of suspended particulate matter (microseston) were made to evaluate the factors responsible for light attenuation in the water column. Suspended organic and inorganic matter were measured gravimetrically and the concentration of particulate organic carbon was analyzed by the wet oxidation technique (Strickland and Parsons 1972).

Phytoplankton standing crop and productivity—Nanoplankton and netplankton chlorophyll *a* concentrations were measured by fluorometry (Strickland and Parsons 1972) following serial fractionation through a 22-μm-mesh Nitex net-disk and a Gelman type A glass fiber filter. Total chlorophyll *a* concentration was estimated from the sum of the nanoplankton and netplankton fractions. Comparison of calculated concentrations with chlorophyll *a* measurements of unfractionated samples gave a mean error of 10.5 ± 3.2% (95% confidence limits). Based on the Wilcoxon signed rank test of differences between the sum of fractionated samples and unfractionated samples, fractionation did not have a significant ($P < 0.01$) effect on estimates of total chlorophyll *a*.

The photosynthetic production of particulate organic carbon by nanoplankton and netplankton fractions was estimated from ^{14}C-uptake measurements (Malone 1971). Light-saturated photosynthetic rates (P_{max} = photosynthetic capacity) were derived from photosynthesis-light experiments using fluorescent light incubators. Surface water samples (1–2 m below the surface) were incubated for 2 h at surface water temperature under light intensities of 0.10, 0.06, 0.03, 0.015, and 0.005 gcal cm^{-2} min^{-1}. In situ primary productivity was estimated using sunlight incubators in combination with neutral density filters that simulated the percent light depths from which the samples were collected (100, 60, 30, 15, and 5%). Samples were incubated for 24 h at surface water temperature. Following both artificial light and sunlight incubations, the samples were fractionated by serial filtration through a 22-μm-mesh Nitex net-disk and an HA Millipore filter. The filter disks were washed with particle-free seawater, placed over fuming HCl for 30 s, dried, and their activity measured using a liquid scintillation counter. Mean coefficients of variation between duplicate light bottles were 16 ± 5% and 8 ± 4% (95% confidence limits) for the netplankton and nanoplankton fractions.

Primary productivity at stations in the apex other than C3 and C5 (where measurements were made) was calculated from the photic zone productivity/chlorophyll *a* ratios observed at these two stations. Annual production (g C m^{-2} yr^{-1}) was calculated by weighting daily productivity at each station according to the area each was assumed to represent. Boundaries were located half the distance between stations and 4 km from the New Jersey and Long Island coasts within the defined area of the apex. Productivity ratios of netplankton:nanoplankton were calculated for stations P1, D3, and D5 from the netplankton:nanoplankton ratios of photosynthetic capacity (R_{pc}) which were significantly correlated ($r = 0.99$; $P < 0.01$) with the corresponding primary productivity ratios (R_{pp}) at A3, C3, and C5 ($R_{pp} = 0.016 + 1.06\ R_{pc}$). Thus, netplankton:nanoplankton ratios of photosynthetic capacity were representative of the corresponding ratios of productivity in situ, and the different incubation times did not selectively affect either size fraction.

Sampling program—Stations were occupied along three transects radiating from the mouth of the Hudson-Raritan estuary (Fig. 1). An additional transect was made down the axis of the Hudson estuary from Spuyten Duyvil (mile point 15) to Lower Bay. This paper focuses on transects C and D (Fig. 1). Stations C2, C3, and C4 are in the sewage sludge dumping area; D2, D3, and D4 are in the dredge spoil dumping area. Stations P1, C5, and D5 are not directly affected by ocean dumping. Station P1 is most strongly influenced by estuarine runoff and tidal currents; D5 and C5 are affected least by estuarine runoff. The water columns of all stations in the apex are of about equal depth (20 to 30 m). The area covered by these transects is about 600 km^2. Stations were occupied at approximately monthly intervals from September 1973 through August 1974. Samples were collected from three to six depths depending on water depth and the rate of light attenuation as estimated from Secchi disk measurements.

Of the biological variables measured, only chlorophyll *a* was measured at all stations and depths sampled. Surface photosynthetic capacity was measured at stations A3, P1, B4, C3, C5, D3, and D5 and photic zone primary productivity at A3, C3, and C5. The concentrations of dissolved and particulate matter were measured at all depths only at stations where photosynthetic capacity was measured. Otherwise, measurements were made on surface and near-bottom samples only. All sampling was conducted during the day between 2 h after sunrise and 2 h before sunset over a 3-day period in an effort to minimize the effect of diel periodicity.

Results

Temperature and salinity—The geographic range of surface temperature was <2°C throughout the year, while the range of surface salinity (excluding P1) varied from 0.6‰ in November to 4.4‰ in July (Fig. 2). Surface temperature varied from a seasonal minimum of 3°C in February to a maximum of 24°C in July. Vertical temperature profiles showed little stratification from October to mid-April and marked stratification from May through August (Fig. 2A). Salinity stratification was observed during all months except October, November, December, and mid-April, and stratification was best developed from May through July (Fig. 2B). Water column stability, as indicated by vertical gradients of σ_T, was most pronounced from May through August (maximum stability in June and July).

Surface salinity was highest and least variable at stations C5 (mean = 31.2‰, range = 2.7‰) and D5 (mean = 30.2‰, range = 2.1‰). Except for January, high salinities were typical of the survey area from August to mid-April when mean surface salinity (excluding P1) ranged from 30.3‰ in September to 32.1‰ in November. Low salinity water (<30‰) was confined to the C transect in January (28.2–29.0‰) and the D transect in May (27.6–28.1‰). Low salinity estuarine water was most widespread during June and July when mean surface salinity was lowest. Rapid decreases in salinity (December–January and April) and the sustained presence (April–August) of low salinity water in the surface layer were due to both increased river runoff (December–January and April) and water column stratification (May–August). Thus, the development of a seasonal thermocline tends to confine nutrient-rich estuarine water to the surface layer during a period when light-driven nutrient demand is greatest.

Dissolved inorganic nutrients—Mean mixed layer levels of dissolved SiO_4-Si and inorganic nitrogen ($DIN = NO_3$-N + NO_2-N + NH_3-N) ranged from 0.5 to 3.6 μM Si and 1 to 14 μM N (Fig. 3A, B), exclusive of P1 near the mouth of the estuary where concentrations exceeded 1 μM Si and 5 μM N throughout the year. Dissolved PO_4-P varied between the relatively narrow limits of 0.5 and 1.5 μM P. High concentrations of SiO_4-Si (>1 μM) and DIN (> 5 μM) characterized the apex from August through January and December through April, respectively. Both nutrients declined sharply from January to February and again from April to May. DIN continued to slowly decrease during the summer to a minimum in August while SiO_4-Si showed a sharp increase.

Fluctuations in atomic N:P ratios paralleled temporal variations in DIN (Fig. 3C) in that the highest ratios coincided with high DIN and vice versa. N:P ratios were generally (80% of observations) less than 10, especially during July, August, and

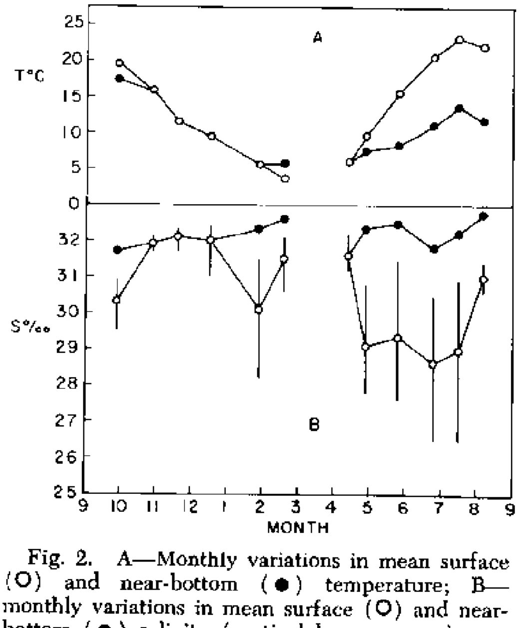

Fig. 2. A—Monthly variations in mean surface (O) and near-bottom (●) temperature; B—monthly variations in mean surface (O) and near-bottom (●) salinity (vertical bars = range).

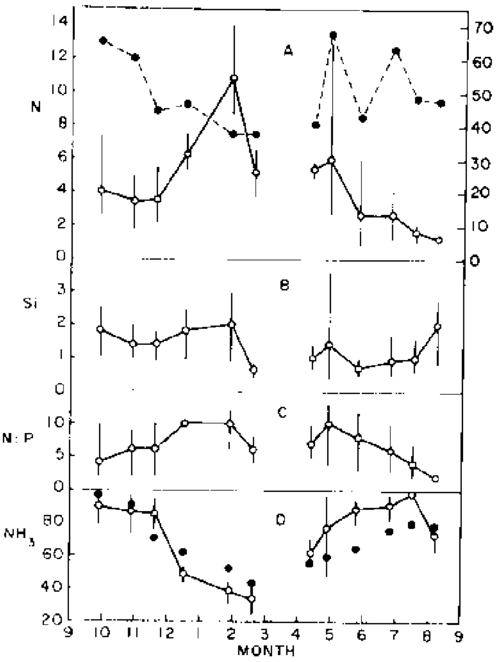

Fig. 3. A—Monthly variations in mean mixed-layer DIN (μM NO_3-N + NO_2-N + NH_3-N) in the apex (O, left axis) and at A3 in the lower Hudson estuary (●, right axis); B—variations in mean mixed-layer silicate (μM SiO_4-Si); C—variations in DIN : P in the surface layer; D—proportion of NH_3-N in the DIN pool of the mixed layer of the apex (O) and the lower estuary (●) (vertical bars = range)

September when ratios below 5 characterized the apex. Ratios above 10 were observed most frequently in January (N:P = 12–18) along the C transect and in late April (N:P = 11–13) along the D transect, immediately following sharp declines in surface salinity. Peaks in DIN also occurred at these times due to influxes of NO_3-N.

Concentrations of DIN above 5 μM (mean mixed layer) were most often a consequence of influxes of NO_3-N associated with low salinity estuarine water. The concentration of NO_3-N exceeded that of NH_3-N 78% of the time when DIN was greater than 5 μM N, but NH_3-N usually (71%) accounted for more than 80% of the DIN pool when DIN was less than 5 μM N. Variations in DIN were not statistically correlated with salinity due to variations in the DIN content of estuarine water and the rates of nitrogen regeneration and phytoplankton uptake within the apex. However, maximum concentrations of DIN were invariably associated with minimum salinities (excluding P1) and low DIN concentrations with the higher salinities typical of the outer stations (C5 and D5). Anomalies in the distributions of surface nutrients were not associated with the dumping sites (centered around C3 and D3), but with low salinity estuarine water.

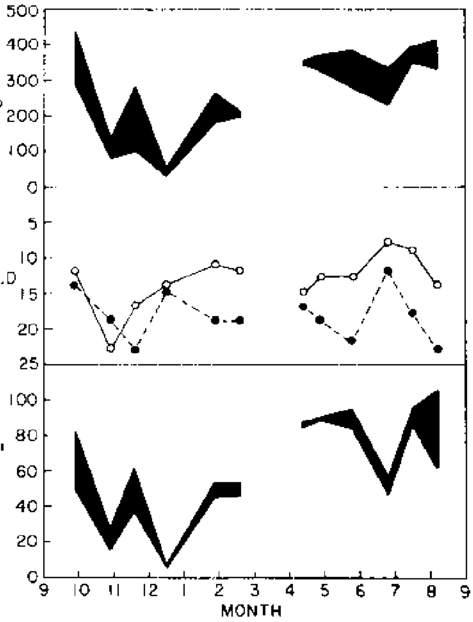

Fig. 4. Monthly variations in incident photosynthetically active light energy (I_0 = gcal cm^{-2} d^{-1}), mean 1% light depth (m) and mean photic zone light energy (I' = gcal cm^{-2} d^{-1}), showing the range of light intensities encountered during each cruise (○ = inner stations, ● = outer stations).

Solar radiation and light extinction—Mean photic zone light energy varied from 6 gcal cm^{-2} d^{-1} in December to 105 in August (Fig. 4). Extinction coefficients for downwelling radiation varied from 0.2 to 1.7 m^{-1}, with low coefficients characteristic of the apex in November and high coefficients characteristic in June. Photic zone depths exceeded 10 m except at the inner stations during June and July and were typically deeper at the outer stations (Fig. 4). Light extinction was significantly correlated ($P<0.01$) with surface microseston concentrations (Fig. 5). Mean water column concentrations of microseston in the apex ranged from 4–12 mg liter^{-1}, of which 7–49% was oxidizable organic matter. If we assume a C:Chl *a* ratio of 35 (Eppley 1972), phytoplankton averaged 2.8 ± 1.5% (1 SD) of the microseston except during February and June when it averaged 5.8 ± 2.1% (1 SD). If we assume the extinction coefficient of 1 mg Chl *a* is 0.0138 m^{-1} (Lorenzen 1972), chlorophyll *a* rarely accounted for more than 20% of the light attenuation (except at D3 in October and February and C5 in January). Thus, light attenuation due to absorption and scatter-

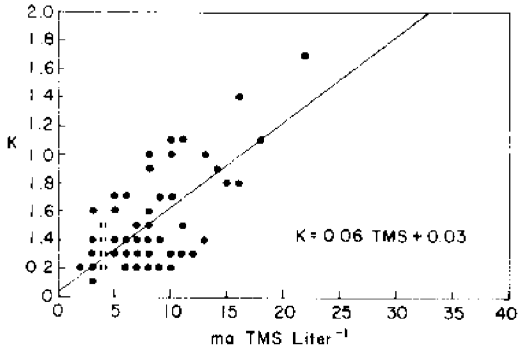

Fig. 5. Extinction coefficient (Secchi disk) as a function of surface microseston concentration (r = 0.72, P <0.01).

ing by suspended microseston was primarily a function of the concentration of nonphotosynthetic particles.

Productivity and chlorophyll—Phytoplankton productivity and chlorophyll *a* generally decreased with distance from the mouth of the estuary, a trend most pronounced during blooms. Baseline levels of mean photic zone (MPZ) chorophyll *a* and photic zone productivity were about 2 μg Chl *a* liter^{-1} and 0.5 g C m^{-2} d^{-1} at the inner stations (P1, C2, C3, C4, D2, D3, and D4) and about 1 μg Chl *a* liter^{-1} and 0.1 g C m^{-2} d^{-1} at the outer stations (C5 and D5) (Fig. 6, 7). Concentrations of chlorophyll *a* above 5 μg liter^{-1} and productivity above 2 g C m^{-2} d^{-1} were restricted to the inner stations and occurred most frequently in February (Chl *a* only) and June when blooms were observed near the mouth of the estuary.

During the February bloom, peak concentrations of chlorophyll *a* occurred at C2 (MPZ = 10.0 μg liter^{-1}) and along the New Jersey coast (6.0–7.1 μg liter^{-1}); the June bloom was localized at D2 (MPZ = 17.8 μg liter^{-1}) and the inner stations of the C transect (10.6–13.5 μg liter^{-1}). Horizontal distributions were least variable (C = 28–40%) from September through April and most variable (C = 71–81%) from June through August when the decreasing seaward gradient of chlorophyll *a* was most pronounced. Productivity followed similar trends except that the February peak was not as pronounced. Productivity above 2 g C m^{-2} d^{-1} was observed most frequently in June (2.2–6.4 g C m^{-2} d^{-1}) and July (2.3–3.3 g C m^{-2} d^{-1}), in contrast to February when productivity at the inner stations ranged from 1.2–2.1 g C m^{-2} d^{-1}.

The highest levels of productivity (>2 g C m^{-2} d^{-1}) were observed when maximum chlorophyll *a* concentrations were near the surface (upper 6 m) at the inner stations in June and July. At these times vertical distributions of chlorophyll *a* were well stratified (C >40%), mixed-layer depths were less than 10 m deep, and low salinity estuarine water was confined to the upper photic zone by a strong thermocline (Fig. 8). High concentrations of chlorophyll *a* were vertically more dispersed during the February bloom, probably a consequence of vertical mixing. With few exceptions, low productivity (<1 g C m^{-2} d^{-1}) character-

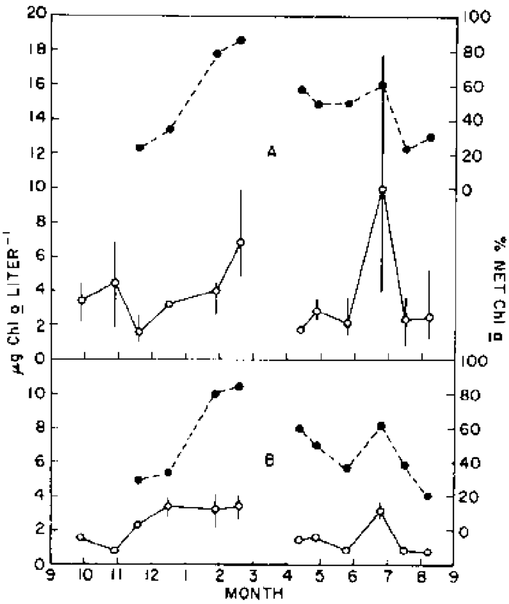

Fig. 6. Monthly variations in mean photic zone concentration of chlorophyll *a* (○) and the proportion of netplankton chlorophyll *a* (●) at the inner (A) and outer (B) stations (vertical bars = range).

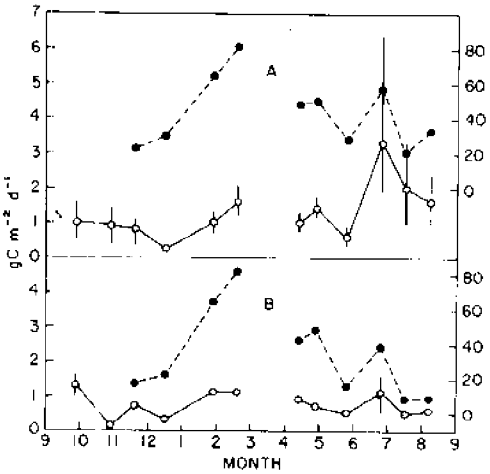

Fig. 7. Monthly variations in photic zone primary productivity (○) in g C m^{-2} d^{-1} and the proportion of netplankton productivity (●) at the inner (A) and outer (B) stations (vertical bars = range).

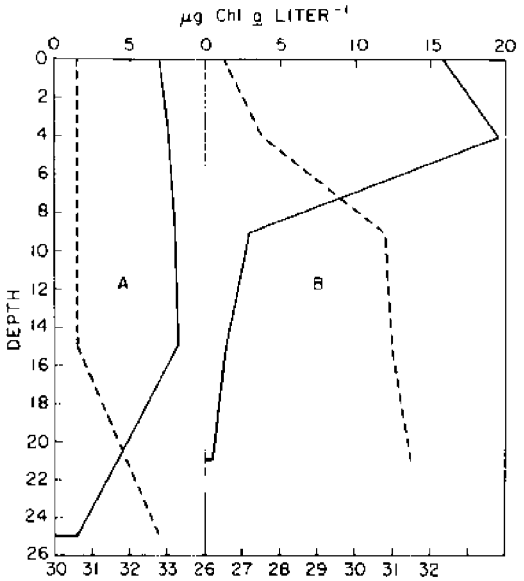

Fig. 8. Vertical distributions of chlorophyll *a* (——) and salinity (- - -). A—Station C2, February; B—station D2, June.

ized the apex when vertical profiles of chlorophyll *a* were uniform (C <20%) and the water column was well mixed.

Peaks in MPZ chlorophyll *a* greater than 4 µg liter^{-1} were dominated by netplankton (Fig. 6) except at P1 in July when nanoplankton accounted for 83% of MPZ chlorophyll *a*. However, netplankton productivity consistently exceeded nanoplankton productivity only during January and February (*see Fig. 6*). Netplankton accounted for 79–87% of the productivity in February (0.8–1.7 g C m^{-2} d^{-1}) and for 38–70% in June (0.8–2.2 g C m^{-2} d^{-1}). Nanoplankton productivity peaked during June and July, reaching 0.5–0.6 g C m^{-2} d^{-1} at the outer stations and 1.0–3.7 at the inner stations. On the basis of maximum chlorophyll *a* concentrations in the water column, netplankton biomass peaked at all stations in February (4.4–9.7 µg liter^{-1}) and along the New Jersey coast (P1, C3, D3, and D5) in June (3.7–7.9 µg liter^{-1}). Nanoplankton chlorophyll *a* peaked at the outer stations in December (2.0–3.1 µg liter^{-1}) and at the inner stations in June and July (2.4–10.5 µg liter^{-1}). The productivity and chlorophyll *a* content of both fractions decreased with distance from the mouth of the estuary, although the netplankton tended to decrease more rapidly. In general, netplankton dominated the winter bloom while nanoplankton dominated during the summer when productivity was highest.

Microscopic examination of surface samples from C3 shows that the February peak in netplankton chlorophyll *a* was dominated in numerical abundance by *S. costatum* (66% of total cells) and *Asterionella japonica* (18%) (Table 1). The summer phytoplankton was dominated by *Nannochloris atomus*, a nanoplankter that reached a maximum cell density in June of 8.2 × 10^{8} cells liter^{-1}. The high cell density of *N. atomus* at this time tends to obscure the fact that netplankton cell density (dominated by *A. japonica*, *S. costatum*, *Phaeodactylum tricornutum*, *Leptocylindrus danicus*, and *Cerataulina bergonii*) also reached its maximum of 4.1 × 10^{6} cells liter^{-1} at this time.

Temporal variations in primary productivity per unit photic zone chlorophyll *a* (*P*/*B*) were similar at C3 and C5 except during June, July, and August when *P*/*B* at C5 was less than that at C3 (Fig. 9). *P*/*B* was less than 40 g C g Chl a^{-1} d^{-1} ex-

Table 1. Nanoplankton and netplankton cell densities (×10^{6} cells liter^{-1}) and dominant species (smallest number of species whose cumulative abundance exceeded 75% of total cells) in surface samples from station C3.

Date	Net	Nano	Total	Dominant organisms*
9 Sep	1.8	1.4	3.2	Ld, O
9 Oct	0.7	0.4	1.1	Sc, Rf
6 Nov	0.1	0.6	0.7	O
5 Dec	0.4	0.3	0.7	Sc, T, Tn
9 Jan	1.5	0.2	1.7	Sc, Aj, T
6 Feb	3.0	0.4	3.4	Sc, Aj
4 Apr	0.3	0.2	0.5	Rd, Ns, Sc
9 Apr	0.4	0.2	0.6	Ld, Sc, Rd
8 May	0.1	0.1	0.2	Na, Cl
8 Jun	4.1	823	827.1	Na
5 Jul	0.4	420	420.4	Na
2 Aug	0.7	83	83.7	Na

* Ld = *Leptocylindrus danicus*, O = nanoplankton, Sc = *Skeletonema costatum*, Rf = *Rhizosolenia faeroense*, T = *Thalassiosira* sp., Tn = *Thalassionema nitzschioides*, Aj = *Asterionella japonica*, Rd = *Rhizosolenia delicatula*, Ns = *Nitzschia seriata*, Na = *Nannochloris atomus*, Cl = *Ceratium longipes*.

cept during these summer months when it ranged from 50–65. Assimilation numbers followed a similar temporal trend but were relatively uniform over the entire apex despite the frequent presence of nutrient gradients (Fig. 9). The Wilcoxon signed rank test of differences between stations showed no significant difference between stations P1 and C3, P1 and D3, C3 and C5, and D3 and D5.

Mean assimilation numbers of netplankton were significantly ($P < 0.05$) less than those of nanoplankton except during February, April, and June when the 95% confidence limits overlapped (Fig. 9). Netplankton assimilation numbers were highest (mean >8 g C g Chl a^{-1} h^{-1}) in February, June, and July; nanoplankton assimilation numbers were highest in June, July, and August (mean >15 g C g Chl a^{-1} h^{-1}).

Discussion and conclusions

Nutrient supply—Phytoplankton productivity integrated over a 600-km² area of the apex ranged from a winter minimum of 1.7×10^5 kg C d^{-1} to a summer maximum of 10.1×10^5 kg C d^{-1}. Annual phytoplankton productivity was about 2.2×10^8 kg C or about 370 g C m^{-2}, equivalent to values quoted by Ryther (1969) for upwelling systems and Riley (1956) for Long Island Sound. On the basis of ambient DIN concentrations in the mixed layer and an assumed atomic C:N ratio of 5 (Eppley et al. 1971, 1973), complete DIN depletion would occur in from less than 1 day during summer to 10 days during winter (<4 days throughout the year except October–December). These levels of phytoplankton productivity require a continuous supply of DIN to the photic zone, especially during summer.

The annual production of 2.2×10^8 kg C requires a DIN supply of about 5.2×10^7 kg (C:N = 5), which is equivalent to the annual flux of DIN from the estuary (O'Connors and Duedall 1975; Garside et al. 1976). On a monthly basis, the supply of estuarine DIN exceeded phytoplankton demand throughout the year except during June, July, and August when estuarine discharge accounted for 50 to 70% of the nitrogen demand. Since the DIN pool was never exhausted and ocean dumping cannot account for more than 10% of the DIN

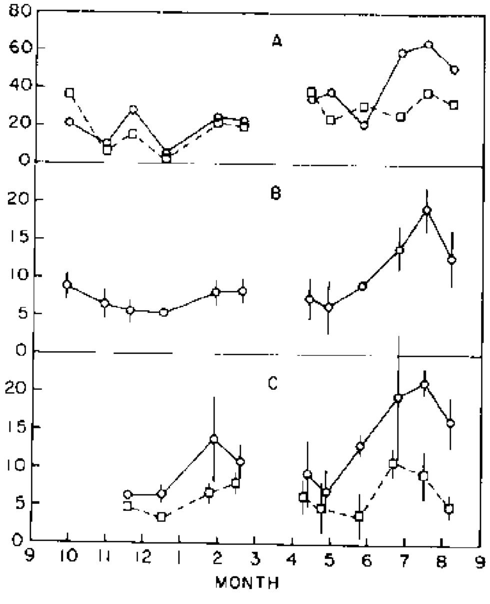

Fig. 9. Monthly variations in (A) photic zone productivity per unit chlorophyll a (P/B = g C g Chl a^{-1} d^{-1}; ○—C3, □—C5); (B) mean assimilation number (P_{max}/Chl a = g C g Chl a^{-1} h^{-1}; vertical bars = 95% confidence limits); and (C) nanoplankton (○—95% confidence limits) and netplankton (□—95% confidence limits) assimilation numbers.

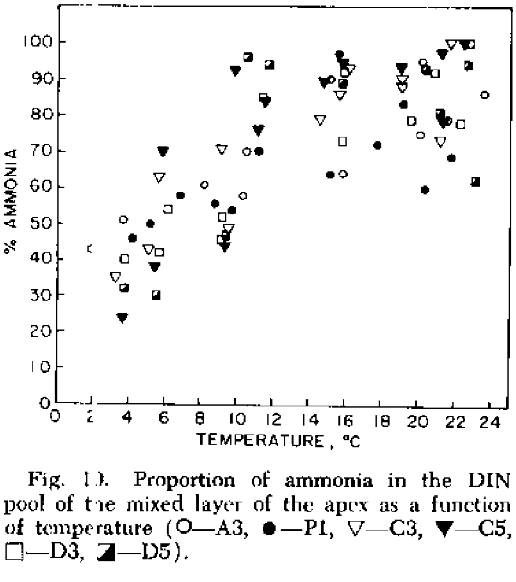

Fig. 10. Proportion of ammonia in the DIN pool of the mixed layer of the apex as a function of temperature (○—A3, ●—P1, ▽—C3, ▼—C5, □—D3, ◪—D5).

requirement, autochthonous ammonia regeneration probably contributed the difference between the high summer demand and DIN flux from the estuary. This is supported by variations in the proportion of NH_3-N in the DIN pool of the mixed layer in the apex (Fig. 3). The proportion of NH_3-N increased with temperature up to about 12°C (November–April) and fluctuated around 85% at temperatures above 14°C (Fig. 10). During May, June, and July when the water column was well stratified and surface temperature was high and increasing, the proportion of NH_3-N in the surface pool was higher by 10–25% than the proportions observed in estuarine or bottom water. Since NH_3-N is selectively assimilated by phytoplankton (MacIsaac and Dugdale 1969; Eppley et al. 1969), it is likely that NH_3-N regeneration rates in the upper water column of the apex were high during the warm summer months. Thus, although estuarine discharge was a major source of both nitrate and ammonia and accounted for the major influxes of DIN, regeneration in the upper water column was probably required for the continued presence of DIN in the surface layer during prolonged and intense summer stratification, especially when NH_3-N accounted for 100% of the DIN present (Fig. 3A, D).

The low and decreasing N:P ratios and DIN concentrations of the mixed layer (Fig. 3A, C) during the summer period of water column stratification support Ryther and Dunstan's (1971) conclusion that phytoplankton productivity in the apex is not phosphorus—but nitrogen—limited. However, if Caperon and Meyer's (1972) chemostat observations are applicable to natural environments, phytoplankton populations do not become nitrogen limited as long as DIN concentrations remain above 0.24 μM. Since DIN concentrations were never reduced to less than 1 μM and phytoplankton assimilation numbers (Fig. 9) were highest during summer when DIN concentrations were lowest (Fig. 3A), it is unlikely that phytoplankton growth was nutrient limited. However, the possibility does exist that the wide divergence between assimilation numbers of nanoplankton (>15) and netplankton (<10) during July and August was due to low concentrations of DIN in the mixed layer (Parsons and Takahashi 1973).

Environmental regulation—Phytoplankton productivity is a function of phytoplankton biomass and the specific growth rate of that biomass. In this study, chlorophyll *a* is assumed to be an index of biomass and productivity per unit chlorophyll *a* (P/B = g C g Chl a^{-1} d^{-1}) an index of growth rate. The assimilation number (g C g Chl a^{-1} h^{-1} at light saturation) can be considered an index of the maximum potential growth rate for a given temperature, nutrient regime, and species. Eppley (1972) reviewed some errors associated with these assumptions, which are due to environmentally induced variations in the C:Chl *a* ratio.

Variations in photic zone P/B (including a station in the lower Hudson estuary) were directly related to variations in MPZ radiation (Fig. 11). The regression, $P/B = 0.43$ $(I') + 0.97$, is highly significant ($r = 0.93$, $P < 0.01$) and is good evidence that P/B was light limited. Since light absorption and scattering were primarily due to suspended detritus, changes in the detritus load of

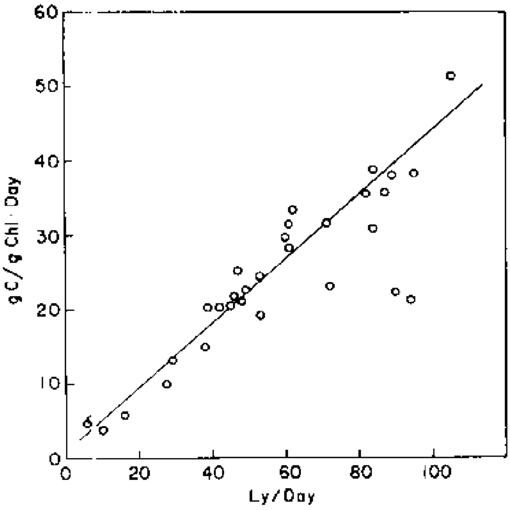

Fig. 11. Phytoplankton P/B (g C g Chl a^{-1} d^{-1}) as a function of light energy in the mean photic zone (ly d^{-1} = gcal cm^{-2} d^{-1}) at A3, C3, and C5; $P/B = 0.43$ $(I') + 0.97$ ($r = 0.93$, $P < 0.01$) excluding two points (P/B = 20, 22; I' = 90, 95).

estuarine water discharged or the volume transport of this water into the apex will affect phytoplankton P/B there. In contrast, an increase in DIN supply to the apex will not affect P/B within the limited area of the apex but will increase the area over which P/B is high.

Although temporal variations in assimilation numbers were not related to concurrent variations in nutrient concentrations, the seasonal cycle of ambient water temperature appeared to be important. Assimilation numbers were significantly correlated with temperature during periods of decreasing (August–December) and increasing (April–July) temperature (Fig. 12). The two regressions (with 95% confidence limits for the slopes) of assimilation number ($P_{\max}$/Chl a) on temperature were:

$$\log_{10}(P_{\max}/\mathrm{Chl}\ a) = 0.031 \pm 0.006\,(T) + 0.38; \quad (1)$$

$$\log_{10}(P_{\max}/\mathrm{Chl}\ a) = 0.045 \pm 0.007\,(T) + 0.26. \quad (2)$$

Thus, except for January and February when high assimilation numbers were observed (Fig. 12B), temperature set an upper limit on the maximum potential rate of photosynthesis per unit chlorophyll a; that limit was variable depending on whether temperature was decreasing ($Q_{10} = 2.04$) or increasing ($Q_{10} = 2.82$).

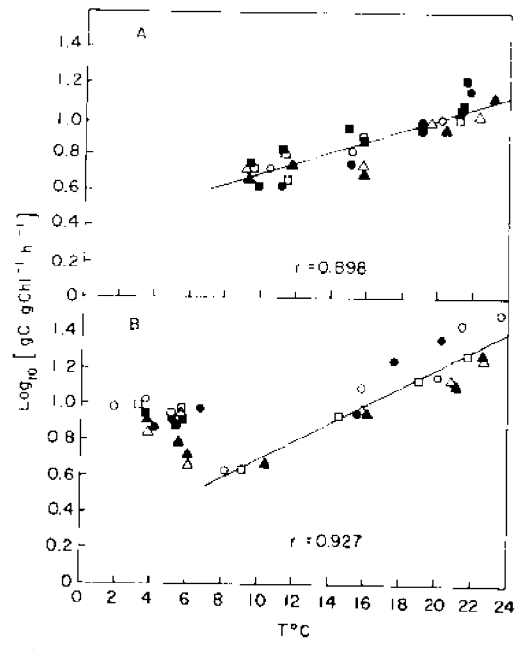

Fig. 12. Log_{10} transformed assimilation numbers as a function of temperature. A—August through December; B—January through July. Assimilation numbers at C5 conformed to the A regression except during January and February.

The high assimilation numbers observed in January and February may reflect temperature acclimation (Eppley 1972), a phenomenon reported for *S. costatum* (Jørgensen 1968). The Q_{10} at C5 was 2.04 throughout the year and the higher Q_{10} observed during the period of increasing temperature corresponds to when the proportion of estuarine water in the photic zone and nanoplankton cell densities (dominated by the chlorophyte *N. atomus*) were greatest. This adds a new dimension to the problem and suggests that the two $P_{\max}$/Chl a, T relationships observed may reflect changes in taxonomic composition (Mandelli et al. 1970).

Takahashi et al. (1973) synthesized a mathematical model that calculates photosynthesis/chlorophyll a (P/Chl a) at any given point in time and space from the corresponding light intensity (I) and temperature (T). The model is based on Steele's (1962) expression of P/Chl a,I and on a linear relationship between $P_{\max}$/Chl a and temperature. The $P_{\max}$/Chl a,T relationship observed in the apex of the New York Bight was exponential rather than linear. Consequently, the following modification of Takahashi's equation 7 was used to calculate P/Chl a from observed values of light and temperature in the lower estuary and apex:

$$P/\mathrm{Chl}\ a = c\,I \exp[1 - a\,I \times 10^{-(d \cdot T + c)}], \quad (3)$$

where P/Chl a = g C Chl $a \cdot$ h, I = gcal cm^{-2} min^{-1}, T = °C, a = constant derived by the least squares method from $\ln(P/I \cdot e),I$, and c,d = constants derived by the least squares method from $\log_{10}(P_{\max}/\mathrm{Chl}\ a),T$. Two sets of constants for the ($P_{\max}$/Chl a),T relationship were derived corresponding to the period of decreasing temperature and station C5 ($c = 0.383$, $d = 0.031$) and to the period of increasing tem-

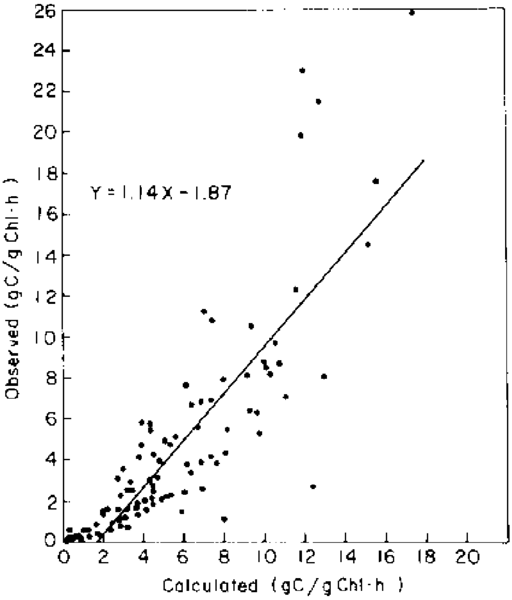

Fig. 13. Regression of P/B (g C g Chl a^{-1} h^{-1}) measured at each light percent depth (simulated in situ) and predicted P/B using the model of Takahashi et al. (1973).

perature less station C5 ($c = 0.262$, $d = 0.045$). The constant a was derived as described by Takahashi et al. (1973) based on the P,I relationship reported by Ryther (1956) for several diatom and dinoflagellate species ($a = 25.40$).

The correlation between calculated and observed estimates of in situ P/Chl a was highly significant ($r = 0.821$, $P < 0.01$), but calculated rates tended to underestimate observed rates, especially at low and high P/Chl a (Fig. 13). Takahashi et al. encountered the same problem and suggested that underestimates might be due to overemphasis of the role of photoinhibition in the model. However, this cannot explain underestimates of low P/Chl a, which could reflect adaptation to low light intensities during periods when the photic zone was stratified.

In spite of these discrepancies, estimates of P/Chl a based on observed levels of light and temperature in the apex agree well with simulated in situ measurements. While the distribution of chlorophyll a, and to a lesser extent productivity, was roughly correlated with the distribution of low salinity estuarine water in the apex (suggesting a causal relationship between nutrient supply and phytoplankton biomass), there can be little doubt that P/B was regulated by light and temperature during this study.

The distribution of DIN in the mixed layer appeared to be related primarily to the interaction between estuarine discharge, regeneration, phytoplankton uptake, vertical stability, and circulation, rather than to dumping. The temporal covariance between water column stability and phytoplankton P/B, combined with the spatial correlation between salinity, DIN, and chlorophyll a, suggests that estuarine discharge is the major source of nutrients (especially silicate and nitrate) assimilated by phytoplankton in the apex. The short nitrogen depletion times (<4 days) observed during most of the year suggest that nitrogen supply and phytoplankton uptake rates are closely coupled, especially during the summer when productivity was high and depletion times were less than 1 day. During this period, ammonia regeneration in the water column of the apex prevented depletion of DIN in the photic zone. Although nutrient gradients were often pronounced, phytoplankton assimilation numbers showed no significant variability between stations, and it is concluded that ocean dumping had no observable effect on phytoplankton growth rates within the dumping areas relative to stations outside the dumping areas. This is consistent with the distributions of the environmental variables shown to regulate P/B in the apex.

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Transient forcings of the lower trophic levels during the spring bloom within the New York Bight[1]

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A time-series study of the spring bloom in the New York Bight was conducted across the continental shelf, south of Long Island, from 26 March to 9 April 1975. A buoy was installed about 95 km offshore at the shelf break in a depth of 90 m, and a 2-week mesoscale and a 30-h diel time series were conducted at this location (39°54'N, 72°4'W) to monitor effects of storm and shelf-break mixing. Two underway line maps of nutrients and chlorophyll fluorescence at 3 m and a series of three offshore transects, of varying station spacing from 5 to 30 km apart, were taken along the main buoyline with additional transects east and west of this zonal study to monitor the time rates of change of system properties across the shelf. Additional stations were occupied along the axis of the Hudson Canyon and longshore between Moriches Bay, Long Island, and Martha's Vineyard to assess both the impact of the offshore canyon and the inshore boundary condition for nutrient cycling on the shelf. A 48-h diel time series of the midshelf ($\simeq$70 m) productivity was also taken in the region of maximum chlorophyll biomass, about 25 km inshore of the shelf-break time series and about 70 km seaward of Shinnecock Inlet. Rainwater samples were collected from the ship's mast for nutrient analyses during one of the three storms encountered.

Measurements of temperature, salinity, irradiance, nutrients, chlorophyll, phytoplankton, particles, particulate nitrogen and carbon, zooplankton, primary production, respiration, and nitrate uptake over a 2-week period suggested maximum productivity and minimum grazing stress at midshelf. The mean spring phytoplankton biomass of 6.2 mg Chl *a* m^{-3} for the 187 midshelf measurements and the observed PN:Chl *a* ratio of 0.6 µg-atom : 1 µg for this area suggests a particulate nitrogen standing stock of 3.7 µg-atoms $liter^{-1}$ (the mean of the 34 PN measurements in this region was 2.4 µg-atoms $liter^{-1}$), in contrast to a total dissolved inorganic nitrogen stock of 4–5 µg-atoms N $liter^{-1}$ during March–April 1975 and about 6–8 µg-atoms N $liter^{-1}$ during January 1958, 1962, and 1975. Moreover, the mean chlorophyll concentration of 79 observations along the same cross-shelf transect at the end of January 1975 was 1.25 µg Chl *a* $liter^{-1}$, or 0.75 µg-atom PN $liter^{-1}$ wtih a PN:Chl *a* conversion of 0.6:1; we could conclude that the seasonal flux of nitrogen from the dissolved to particulate pools within the bight might be accounted for in such a budget.

However, the mean productivity of 2.97 g C m^{-2} d^{-1} for seven simulated in situ stations and that of 2.91 g C m^{-2} d^{-1} calculated from the 21 potential productivity stations in this area, together with the mean atomic PC:PN ratio of 7.44:1 imply a total daily nitrogen demand of 1.1 µg-atoms N $liter^{-1}$ d^{-1} over the upper 30 m. This agrees fairly well with the NR estimate of just a nitrate utilization rate of $\simeq$ 0.5 µg-atom NO_3 $liter^{-1}$ d^{-1} (over a 10-h day). Steady state assumptions of a ΔN transfer of 2–3 µg-atoms N $liter^{-1}$ from the water column to phytoplankton between January and March–April 1975 could thus really occur in as brief a time as 2–3 days with the above rates. The above dissolved and particulate nitrogen standing stocks could not have been maintained over almost all of the 2-week period of our study without other nitrogen inputs. Furthermore, seasonal storm mixing and possible breaking of internal waves appear to be mechanisms for replenishing nutrients within the spring continental shelf ecosystem; nutrient recycling through herbivore excretory products also appeared to be an important nutrient source, whereas rainfall and river discharge did not appear to add significant nitrogen at the time of the spring bloom.

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[1] This work was performed under the auspices of the U.S. Energy Research and Development Administration.

The cause of the apparent decline of the spring bloom at the end of our cruise is unknown, because (1) the dissolved inorganic nitrogen was still over 4 µg-atoms N liter^{-1} as a result of the above additional inputs; (2) the N:P ratio was as low as 2:1 inshore, presumably reflecting eutrophication; and (3) the range in NO_3 : SiO_4 ratio was from 1:1 to 1.5:1 across the shelf—similar to upwelled source water off Peru and Baja California—implying that silicic acid may not have been limiting at this time of year in the New York Bight ecosystem as well. Grazing stress may not have terminated the bloom either, for zooplankton biomass was minimum at midshelf, and our calculations suggest that only 5% of the algal standing crop might have been consumed each day. Moreover, the *fo/fa* index was highest (>1.8) at midshelf, suggesting little detrital chlorophyll, while ammonia concentrations were minimum, and the particle counts suggested large size, but minimum total particles in this area of the shelf as well. Finally, the well mixed water column apparently resuspended the phytoplankton in this area, and it is not clear that a sinking loss to the sediments can be invoked instead as a major cause of the decline of the spring bloom.

It is also interesting that the mean PN : Chl *a* value (µg-atom liter^{-1} : µg liter^{-1}) was 3.8:1 at the inshore stations, 0.6:1 at the midshelf stations, and 1.4:1 at the offshore stations, implying a smaller detrital or microzooplankton contribution to the particulate nitrogen in the midshelf region. Analysis of the PN:Chl *a* ratios for phytoplankton cultures further suggests a mean value of 0.8:1 for actively growing cells, while values <1 have been observed in eutrophic inshore areas of the northwest African upwelling ecosystem in contrast to values >1 for the more oligotrophic offshore areas associated with high detrital content. Finally, the PC:PN ratio was a mean of 7.44:1, suggesting that diatoms were the dominant phytoplankton.

Enumeration of the phytoplankton species confirmed that *Thalassiosira nordenskioldi* was the most abundant organism during the 1975 spring bloom; *Thalassiothrix fraunenfeldii* and *Chaetoceros* spp. were subdominants. At most of the inshore stations, about 60% of the phytoplankton were diatoms, while diatoms constituted over 95% of the phytoplankton in the midshelf and shelf-break areas. The total number of particles (1–100-µ size fraction) showed a seaward decline with distance offshore, suggesting that those of lithogenic origin are restricted to the inshore region, whereas those of biogenic origin (>10 µ) are correlated with the distribution of phytoplankton cells except for the inshore area of high detrital content.

The number of cells per chain of *Thalassiosira* increased from less than 4 per chain nearshore to 10 per chain at midshelf and then declined to less than 5 cells per chain offshore. In a previous seasonal study of the grazing stress on *Skeletonema costatum* in Narragansett Bay, chain lengths of 3 cells per chain were observed under high grazing stress and >7 per chain under low grazing stress. These observations corroborate the inverse relationship of the >10-µ particles with chlorophyll at midshelf and suggest that increased grazing at the boundaries of the shelf might have been responsible for the larger number of short chains in these regions. Furthermore, tintinnids consisted of over 5% of the total particles as far as 40 km off the coast, with a maximum of 35% at 20 km; these protozoans are herbivores and might have been responsible for the meager phytoplankton biomass of the inshore region. We are now continuing research to further examine the fate of the spring bloom within the Middle Atlantic Bight. Because of time constraints in preparation of these proceedings, we were unable to provide a full paper, but more detailed analysis and presentation of the data are underway.

Microbiological studies of the Atlantic continental shelf[1]

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Abstract

Microbial degradation of petroleum, polychlorinated biphenyls, and similar substances was investigated in the laboratory and at stations in the Atlantic Ocean near offshore drilling sites. Microorganisms were analyzed as to type, including total viable, aerobic, heterotrophic and petroleum-degrading bacteria, yeasts and fungi, actinomycetes, and chitin-hydrolyzing bacteria.

Mixed hydrocarbon substrate generally yielded higher counts than media made up with South Louisiana crude oil. The inorganic nutrient (PO_4, NO_3, NO_2, NH_4) content of the seawater medium influenced the amount of petroleum hydrocarbons degraded by microorganisms present in sediment and in the water column. Sediment bacteria from the deep ocean environment (3,500 m) were able to degrade crude oil; microorganisms present in surface seawater samples collected at stations along the southeast Atlantic coast degraded crude oil, but the microbial potential for degradation of oil was limited.

Polychlorinated biphenyl (PCB) and PCB-degrading bacteria were recovered from surface water and sediment samples collected along the southeast Atlantic coast from Miami to Cape Hatteras. Oil and PCB-degrading bacteria are potential indices for oil and PCB contamination.

Microorganisms are just beginning to be appreciated as a key component of the health of an ecosystem (Colwell 1976). The composition and metabolic capabilities of microbial populations, i.e. protozoa, filamentous fungi, yeasts, and bacteria, can provide a "fine-tuning" indication of environmental effects at an early stage of alteration.

A program in marine microbiology at the University of Maryland, underway since 1964, has permitted the accumulation of extensive data on microbiological parameters for water and sediment samples collected in the southeast Atlantic Ocean from Miami, Florida, to Cape Hatteras, off the coast of North Carolina. Studies of petroleum-degrading, pesticide-degrading, heavy metal-metabolizing, and antibiotic-resistant bacteria, as well as aerobic, heterotrophic bacteria and fungi, have provided enough data to indicate seasonal fluctuations for specific types of bacteria and to identify and classify microbial species involved in selected processes of microbial ecology. Studies of deep-sea bacteria that are resistant to antibiotics of clinical significance show that such bacteria can be isolated from water samples collected in the deep ocean (Sizemore 1975).

Results of several recent cruises in the southeast Atlantic Ocean are presented here to provide information for evaluating the impact on coastal regions of ocean dumping, offshore drilling for petroleum, ocean mining, and related activities.

Materials and methods

Water and sediment samples were collected with a Niskin sampler and a Shipek grab. Four of the most recent cruises provided data on petroleum and polychlorinated biphenyl (PCB)-degrading microorganisms, as well as aerobic, heterotrophic bacteria and fungi. The cruises aboard the RV *Eastward* were carried out in August 1973, November 1974, and March 1975. Details of media, methods of counting microorganisms, culture systems, extraction and analytical procedures, and statistical analyses were reported elsewhere (Schwarz et

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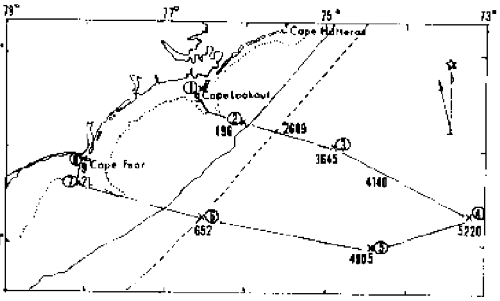

Fig. 1. Sampling area for RV *Eastward* cruise E-10-73, 1–4 August 1973.

al. 1974, 1975; Walker et al. 1976; Colwell et al. 1976; Conrad et al. 1976; Seesman et al. 1976; Sayler et al. 1976).

Results and discussion

Physical and chemical data collected at stations comprising the cruises have been reported (Colwell et al. 1976; Sayler et al. 1976). Station locations are shown in Figs. 1–3. Measurements taken during the cruises included: station depth, sample depth, salinity, temperature, transparency, chlorophyll *a*, pheo-pigment, phosphate, nitrate-nitrogen, nitrite-nitrogen, ammonium-nitrogen, dissolved organic carbon, particulate organic carbon, pH, dissolved oxygen, benzene-extractables in the sediment, ^{14}C productivity, and total PCB.

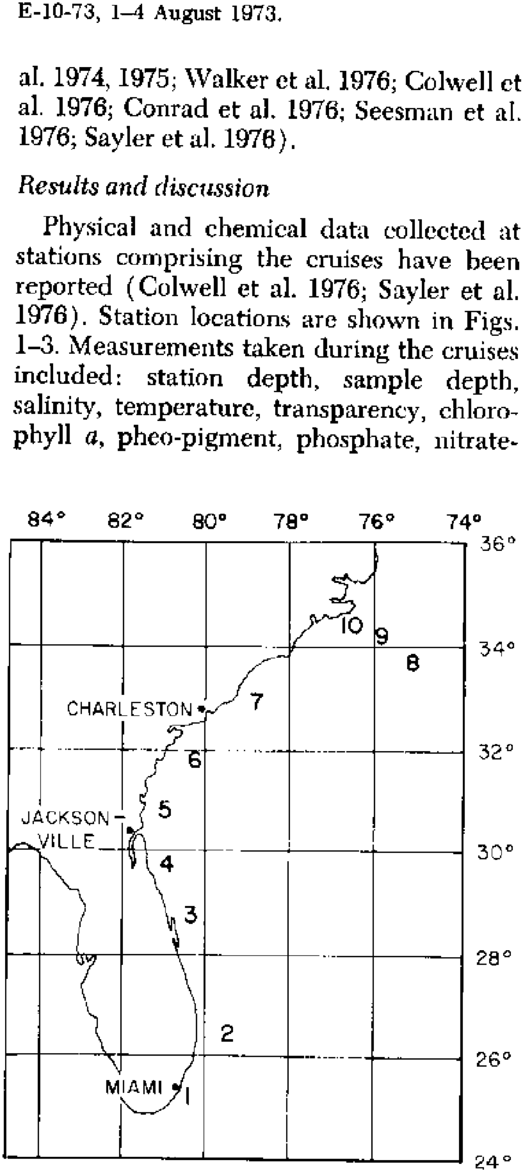

Fig. 2. Stations sampled on RV *Eastward* cruise E-16B-74, 16–21 November 1974.

Fig. 3 Stations sampled on RV *Eastward* cruise E-1E-75, 22–24 March 1975.

In general, the results showed that the highest concentrations of NH_4-N and PO_4-P were found in nearshore waters, with nutrient concentrations and net activity of surface waters lower off the continental shelf at the deepest stations (Walker et al. 1976; Colwell et al. 1976; Sayler et al. 1976). Samples collected at nearshore stations showed the highest productivity indices and the largest microbial populations. Table 1 lists data for aerobic, heterotrophic bacteria counted on a modified seawater yeast extract (MSWYE medium) (Schwarz et al. 1974).

In Table 2, the total viable count data for RV *Eastward* cruise E-16B-74 are given; these were obtained using Marine Agar 2216 (Difco Laboratories). Comparable data for total counts were also obtained in replicate samples using MSWYE medium (Table 3). Microbial counts for open ocean samples were obtained by membrane filtration (Millipore Corp.) because microorganisms occurred in low numbers in these samples (Tables 2–4).

Table 1. Selected data from RV *Eastward* cruise E-10-73.

Sta.	Station depth (m)	Heterotrophic bacteria/ml sediment	Benzene-extractables sediment (µg/g)
1	9	5.7×10^4	81
2	62	1.1×10^3	105
4	5,000	2.1×10^3	131
6	575	2.8×10^3	ND*
8	11	4.6×10^4	55

* Not determined.

Table 2. Microbial populations in samples collected on RV *Eastward* cruise E-16B-74.

		Concentration of each group*				
Sta.	Sample type†	Bacteria TVC‡	1254 metabolizers	1242 metabolizers	Fungi§	Yeasts‖
1	W	5.7	1.3	0.3	0.3	0.08
	S	2,100.0	95.0	130.0	30.0	13.0
3	W	20.0	0.1	1.0	0.09	0.1
	S	120.0	3.5	1.9	0.13	1.3
5	W	16.0	1.0	0.9	0.05	0.05
	S	260.0	45.0	0.8	0.08	0.4
7	W	38.0	1.0	1.2	0.001	0.1
	S	260.0	12.0	1.1	0.05	0.7
8	W	0.07	0.07	0.02	0.002	0.003
	S	3.6	1.2	0.3	0.4	0.05
9	W	0.3	0.3	0.2	0.0003	0.05
	S	40.0	5.4	3.4	0.03	2.1
10	W	7.7	0.7	0.005	0.002	0.003
	S	1,200.0	39.0	19.0	0.4	3.6

* Per milliliter, or gram, × 10^2.

† W—Surface water, collected 10 feet below the hull of the ship; S—sediment.

‡ Total viable count on Marine Agar 2216.

§ Sabouraud dextrose agar.

‖ Littman oxgall medium.

Table 3. Numbers of petroleum-degrading bacteria in seawater samples collected on RV *Eastward* cruise E-16B-74.

Sta.	Sample depth (m)	Petroleum degraders	Bacteria/ml (total count)	Petroleum degraders (%)
1	0	<0.01	1.40	0
	9	0.4	1.70	23
	11	<0.01	30	0
2	0	<0.01	0.20	0
	490	<0.01	0.55	0
	530	<1	<1	0
3	0	0.1	0.10	100
	155	<0.01	0.01	0
	160	<1	<1	0
4	0	<0.01	0.01	0
	160	<0.01	0.08	0
	195	90	90	100
5	0	<0.01	0.01	0
	161	<0.01	0.01	0
	190	<0.01	60	0
6	0	<0.01	0.25	0
	200	0.01	0.02	50
	210	40	40	100
7	0	0.01	0.01	100
	152	<0.01	0.01	0
	180	<0.01	120	0
8	0	<0.01	0.02	0
	3,199	<0.01	<0.01	0
	2,260	<1	<1	0
9	0	<0.01	0.01	0
	50	<0.01	0.01	0
	65	<0.01	20	0
10	0	0.40	0.40	100
	8.5	<0.01	40	0

Table 4. Numbers of petroleum-degrading bacteria in seawater samples collected on RV *Eastward* cruise E-1E-75.

Sta.	Sample depth (m)	Petroleum degraders	Bacteria/ml (total count)	Petroleum degraders (%)
1	3	<0.01	0.03	0
	9	<1	10	0
2	3	<0.001	0.001	0
	99	0.01	0.01	100
	120	<1	<1	0
3	313	<1	<1	0
4	270	20	20	100
5	420	<1	<1	0
6	3	<0.001	<0.001	0
	435	<0.001	<0.001	0
	525	<1	<1	0
7	680	<1	<1	0
8	770	<1	<1	0
9	685	<1	<1	0
10	3	<0.001	<0.001	0
	501	<0.001	<0.001	0
	612	<1	<1	0
11	3	<0.001	<0.001	0
	4,191	<0.001	<0.001	0
	4,420	<1	<1	0

Chitin-hydrolyzing bacteria were isolated on media without salt (chitin-0) and with 3% NaCl added (chitin-3) (Seesman et al. 1976). The chitin-3 medium provided larger total counts and a slightly better recovery of chitin-hydrolyzing bacteria (*see Tables 6 and 7*). Earlier cruises in waters north of Puerto Rico yielded similar results for counts of chitin hydrolyzers.

Actinomycetes were also isolated on the chitin media, with larger numbers of actinomycetes, representing a higher percentage of the total actinomycete population, being isolated on chitin medium without NaCl.

Petroleum-degrading microorganisms were present in the water and sediment samples in larger numbers in nearshore waters and sediment, but the total numbers of petroleum degraders suggested a very sparse distribution (*see Tables 3 and 4*). Comparable samples collected in oil-contaminated harbor areas have been found to contain as many as 680 petroleum degraders per gram of sediment.

Seawater samples collected during RV *Eastward* cruise E-16B-74 (Fig. 2) were inoculated into media containing 0.1% (v/v) South Louisiana crude oil, and degradation was monitored by gas-liquid chromatography (Seesman et al. 1976). Relatively little degradation of the crude oil was observed after an incubation of 7 weeks at 15°C in shaken flask culture. We obtained detectable degradation for seawater samples collected at stations 1–7 and 9, but only samples from stations 3a, 8, and 10 showed significant degradation; the low-boiling n-alkanes were the most readily degraded components and a small increase in the C_{27}–C_{30} n-alkanes was noted.

Examination of a mixed bacterial culture obtained from the sediment–water interface of a core sample collected off the coast of Florida at a depth of 4,940 m showed that the culture could utilize n-hexadecane as the sole carbon source for growth at the in situ temperature (4°C) and pressure (500 atm). However, the rate of utilization under deep ocean conditions was found to be relatively slower than under ambient pressures (1 atm) and low temperature (4°C) (Schwarz et al. 1975).

A comparison was made of seawater, sterile seawater, and salts solution supplemented with nitrate and phosphate as diluent for South Louisiana crude oil and mixed hydrocarbon substrate media. Aliquots of sediment samples collected during RV *Eastward* cruise E-16B-74 were inoculated into the media. Salts solution, supplemented with nitrate and phosphate, produced the highest net log yield of bacteria (Conrad et al. 1976).

The mixed hydrocarbon substrate produced better growth yields than the South Louisiana crude oil and was used to significant extent; growth on the crude oil substrate and degradation were followed with GLC (Walker and Colwell 1974). The salts solution supplemented with nitrate and phosphate, provided optimum conditions for use of both the mixed hydrocarbon substrate and the South Louisiana crude oil. We concluded that the utilization of petroleum hydrocarbons by microorganisms in the sediment at the stations from which samples were collected was stimulated by addition of N and P and that the higher saturated hydrocarbon composition of the mixed hydrocarbon substrate, compared with the South Louisiana crude oil (93% and 58%), preferentially supported growth of the microorganisms in the sediment inocula. The amounts of mixed hydrocarbon substrate and South Louisiana crude oil used were correlated with amount of growth (Conrad et al. 1976). Thus, petroleum hydrocarbons are utilized by sediment bacteria in the marine environment, but the extent of this use is controlled by several factors, including petroleum composition, pressure, and available nutrients.

Also examined during RV *Eastward* cruise E-16B-74 were the presence of PCB and the distribution of PCB-degrading bacteria (Table 5). PCBs were recovered from all the marine water and sediment samples collected during the cruise, but only five of the samples contained concentrations at or above the sensitivity limits of the methods used, i.e. ca. 10 μg/kg sediment and 0.1 μg/1,000 ml seawater (Sayler et al. 1976).

Bacteria capable of growth on media containing Aroclor 12 (a PCB) as the primary carbon source were recovered from all of the water and sediment samples. We confirmed this in subsequent laboratory anal-

Table 5. Numbers of PCB-degrading bacteria in surface water and sediment samples on RV *Eastward* cruise E-16B-74.

Sta.	Sample type*	TVC	PCB metabolizers	Composition (%)	Total PCB (μg/liter)
1	W	5.7×10^{2}	1.3×10^{2}	22.8	0.3
	S	2.1×10^{5}	9.5×10^{3}	4.5	12.0
3	W	2.0×10^{3}	1.0×10^{1}	0.5	<0.1
	S	1.2×10^{4}	3.5×10^{2}	2.9	<10.0
5	W	1.6×10^{3}	1.0×10^{2}	6.2	0.5
	S	2.6×10^{4}	4.5×10^{3}	17.3	<10.0
7	W	3.8×10^{3}	1.0×10^{2}	2.7	0.5
	S	2.6×10^{4}	1.2×10^{3}	4.6	<10.0
8	W	7.2×10^{0}	7.2×10^{0}	100.0	<0.1
	S	3.6×10^{2}	1.2×10^{2}	33.3	<10.0
9	W	3.0×10^{1}	3.0×10^{1}	100.0	<0.1
	S	4.0×10^{3}	5.4×10^{4}	13.5	<10.0
10	W	7.7×10^{2}	7.0×10^{1}	9.0	0.7
	S	1.2×10^{5}	3.9×10^{3}	3.2	<10.0

* W—Water; S—sediment.

yses. The number of PCB-degrading bacteria was higher in the sediment samples examined than in the water, with total numbers of PCB-degrading bacteria highest at the shallow stations with the numbers decreasing with increasing depth and distance from shore (Sayler et al. 1976). Clearly, PCB is ubiquitous in its distribution in the marine environment.

In assessing the effects of pollutants on the marine ecosystem, it is necessary to establish effects on microecology of the system as well as the macrobiological components. Since bacteria play a role in nutrient recycling, selected substrates, such as chitin, cellulose, starch, etc., were examined to determine the microbial populations able to degrade these substrates. Data obtained for the chitin hydrolyzers on two RV *Eastward* cruises are given in Tables 6 and 7. Experiments are now in progress to determine the effects of petroleum on the microflora of seawater and sediment and to compare the potential ecological impact of such pollutants on the microflora of coastal and deep ocean waters.

The results of the studies reported here make it clear that the microbial ecology of the ocean offers a valuable means of measuring the environmental impact of selected

Table 6. Numbers of chitin-hydrolyzing bacteria isolated from sediment samples collected on RV *Eastward* cruise E-16B-74.

	Chitin-hydrolyzing bacteria (cfu)*			
	Chitin-0 medium		Chitin-3 medium	
Sta.	Hydrolyzers	Total	Hydrolyzers	Total
1	$<10^{2}$	1.4×10^{4}	$<10^{2}$	2.0×10^{5}
2	$<10^{2}$	4.0×10^{2}	$<10^{2}$	4.0×10^{2}
3	$<10^{2}$	1.0×10^{2}	$<10^{2}$	2.0×10^{2}
4	$<10^{2}$	3.0×10^{2}	1.0×10^{2}	4.0×10^{2}
5	$<10^{2}$	1.0×10^{2}	$<10^{2}$	5.0×10^{2}
6	$<10^{2}$	1.1×10^{3}	1.0×10^{3}	4.0×10^{4}
7	$<10^{2}$	1.0×10^{2}	$<10^{2}$	7.5×10^{2}
8	$<10^{2}$	8.0×10^{2}	$<10^{2}$	3.0×10^{2}
9	$<10^{2}$	1.0×10^{2}	$<10^{2}$	1.7×10^{3}
10	$<10^{2}$	2.5×10^{3}	$<10^{2}$	2.1×10^{4}

* Colony-forming units.

Table 7. Numbers of chitin-hydrolyzing bacteria isolated from sediment samples collected on RV *Eastward* cruise E-1E-75.

	Chitin-hydrolyzing bacteria					
	Chitin-0 medium			Chitin-3 medium		
Sta.	Hydrolyzers	Total	%	Hydrolyzers	Total	%
1	1.5×10^3	1.5×10^3	100	8.0×10^4	3.9×10^6	2
2	$<10^1$	8.3×10^2	0	2.6×10^3	8.1×10^3	32
3	5.0×10^0	3.4×10^2	1	$<10^1$	9.3×10^2	0
4	5.0×10^2	1.0×10^3	50	1.8×10^2	1.0×10^3	2
5	1.0×10^1	1.6×10^2	6	3.6×10^2	7.0×10^2	51
6	3.0×10^2	2.7×10^3	11	3.5×10^2	5.2×10^3	6
7	5.0×10^1	1.8×10^3	2	2.5×10^2	2.1×10^3	11
8	$<10^1$	1.7×10^3	0	5.0×10^1	1.4×10^3	3
9	2.0×10^1	1.5×10^2	13	1.5×10^3	6.5×10^3	23
10	$<10^1$	3.8×10^3	0	$<10^1$	1.4×10^2	0
11	$<10^1$	1.6×10^2	0	$<10^1$	2.2×10^3	0

pollutants and the capacity of the system to react at the primary level, i.e. via microbial degradation. We propose that "microbial reactive indices" be developed, especially for those areas subject to exploration and development. Such an index could suggest the maximum sustainable microbial population, under prevailing environmental conditions, that is capable of degrading or mineralizing a given substrate—be it crude oil, fuel oil, PCB, or some other substrate expected to have an environmental impact. The microbial response may well prove to be a predictor of the ultimate fate of a given environment.

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Section 6 Fish and fisheries

Effects of coastal pollution on fish and fisheries—with particular reference to the Middle Atlantic Bight[1]

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Abstract

Industrial contamination of coastal waters exerts great local impact on fish and shellfish populations. Known offenders include heavy metals, halogenated hydrocarbons, and petroleum residues. Even in low concentrations, many industrial chemicals have profound effects on some or all life stages of marine animals; effects may be reflected in mortality, increased occurrence of abnormalities, and physiological disturbances, with resultant slow growth or spawning inhibition.

Many toxic chemicals in industrial effluents and agricultural runoff—pesticides, heavy metals, and petroleum products—are also components of sewage treatment plant effluents or ocean outfalls from domestic sewage systems. Additionally, bacteria and viruses force closure of many areas to shellfish harvesting. Still another effect is increased organic loading of coastal waters which should increase productivity, but unmanaged it may contribute to reduced oxygen levels, to blooms of toxic phytoplankton, or of algal species too small to be utilized by shellfish, menhaden, and other filter-feeding animals.

Orderly development of marine aquaculture in coastal waters can be severely impeded by contaminant-related public health problems, both chemical and microbial, and by contaminant-induced problems associated with survival, reproduction, and growth of cultivated animals.

Long term effects on sportfish abundance are difficult to quantify. In some instances party boats have shifted from bays and estuaries to open waters and offshore reefs. Localized effects of pollutants on some species, particularly flatfish, have been observed, but increases in abundance of striped bass, weakfish, and bluefish suggest that conclusions about human effects on population changes must be assessed very carefully.

Available evidence suggests that environmental stress may precipitate diseases in fish and shellfish. Public health aspects are presently confined largely to those microorganisms and toxic substances which do not produce obvious disease in fish and shellfish—but which may be accumulated or transmitted passively to humans from aquatic animal vectors (hepatitis, cholera, typhoid, and mercury poisoning). A few pathogens of fish and shellfish (e.g. *Vibrio parahemolyticus*) may be transmissible to humans, but a clear role for pollutant effects has not been demonstrated.

Evidence exists for *localized* effects on fisheries, but there is as yet little specific evidence of *widespread* damage to major fisheries resource populations resulting from coastal pollu-

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tion. Other factors, such as repeated year-class successes or failures, long term shifts in geographic distribution, and overfishing, may also cause pronounced changes in fisheries.

Field investigations of possible relationships between pollution and localized damage to fish and shellfish populations usually yield evidence that is at best circumstantial. Field investigations combined with results of experimental contaminant studies and information about long term trends favor the conclusion that pollution does locally damage fish and shellfish populations. Continued and increased effort is needed to test and document the reality and extent of the effects.

UNESCO's Intergovernmental Oceanographic Commission defined marine pollution as "Introduction by man, directly or indirectly, of substances or energy into the marine environment (including estuaries) resulting in such deleterious effects as harm to living resources, hazards to human health, hindrance to marine activities including fishing, impairing the quality for use of sea water, and reduction of amenities."

Any discussion of the extent of marine pollution and the urgency of the need for effective controls is colored by the viewpoint of the discussant—whether it be human health, recreational amenities, industrial economics, or natural resources. This paper assesses pollution effects on living resources of coastal waters, with particular reference to the Middle Atlantic Bight.

Pollution of U.S. coastal waters is increasing. It does so as the human population and industries in coastal zones increase, as ocean outfalls proliferate in numbers and capacities, and as ocean dumping continues and expands. Indicators of the problem include high coliform levels in coastal waters, increasing organic content in inshore sediments, increasing reports of coastal algal blooms, and reports of significant levels of chlorinated hydrocarbons, petroleum residues, and heavy metals in ocean waters, sediments, and organisms. The U.S. coastline is characterized by some badly degraded but localized waters which get much of the public attention (the New York Bight is probably the best example) and by much *relatively* clean coastal water. As might be expected, the degree of estuarine and coastal pollution is directly proportional to human population density in adjacent land areas. A similar conclusion has been reached recently by an international working group on pollution in the North Sea—whose findings were that the North Sea as a whole was not seriously polluted, but that areas of the coastline adjacent to high population centers and industrial centers had apparent problems that required further attention. Predictions of population increases in the Atlantic coastal zone of the United States suggest that pollution problems may intensify and spread.

We lack adequate baseline data on previous abundance levels of resource and food chain organisms, and on previous environmental chemical levels. In a few places—the New York Bight and the Southern California Bight, for example—we are at least acquiring adequate present-day baseline data, against which future changes can be measured; but historically, we still must rely on incomplete records to assess man's impact on marine populations and ecosystems. The absence of good baseline information limits conclusions about environmental changes to the circumstantial rather than the proven category.

Several summaries of pollution effects are available. *Marine pollution and sea life* (Ruivo 1972) details worldwide examples of pollutant effects on marine populations and ecosystems. Oil pollution of the oceans has been summarized by Nelson-Smith (1970). Effects of pesticides and related compounds on fish and shellfish have been extensively explored (Butler 1969*a*, *b*; Butler et al. 1972; Duke et al. 1970; Walsh 1972). Cole (1975) and Waldichuk (1974) have attempted more general assessments of pollutant impacts on fish.

Here I consider five aspects of marine pollution—

1. Effects of industrial contamination on fish and shellfish.
2. Effects of domestic sewage contamination on fish and shellfish.

3. Effects of coastal pollution on marine aquaculture.
4. Effects of coastal pollution on recreational fisheries.
5. Pollution-associated diseases of fish and shellfish.

Where clear associations have been established, this will be indicated; where only suspected or poorly documented relationships exist, this will also be indicated. Correlations between fisheries problems and coastal pollution may be categorized as *substantiated*, with good documentation, *suspected*, with weak documentation, and *speculated or inferred*, with little or no documentation.

It should be emphasized that marine pollution problems are largely *coastal* and not *oceanic*, although some high-seas pollution exists (NOAA 1973; Risebrough et al. 1972). Principal problem areas are estuaries, coastal areas adjacent to estuaries, other coastal areas adjacent to municipalities, and to a much lesser extent the continental shelf areas farther removed from the immediate coast. During the past 5 years attempts have been made and some successes achieved in reducing pollution loads in rivers and canals, but often these benefits have been achieved at the expense of bays, estuaries, and coastal waters, with the increased marine sludge dumping, and proliferation, consolidation, and extension of ocean outfalls. Major sources of coastal pollution are shown in Fig. 1.

The technology for solution of coastal pollution problems is available, but awaits suitable application of money and determination. Coastal waters have multiple uses (recreation, fishing, aquaculture, transportation, waste disposal), with priorities for each in a state of dynamic and uneasy equilibrium.

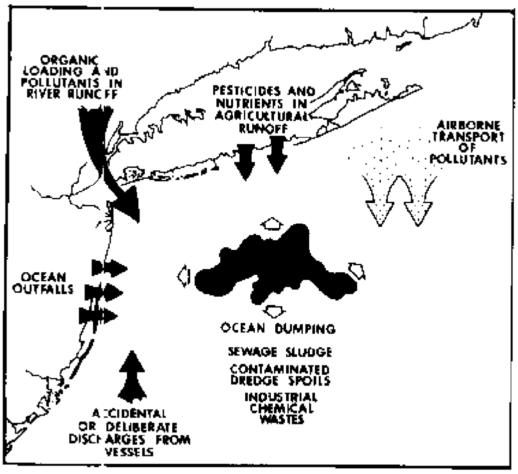

Fig. 1. Principal sources of pollutants in the New York Bight.

Effects of industrial contamination on fish and shellfish

Contamination of coastal waters is clearly deleterious to fish and shellfish. Common industrial contaminants include petroleum residues, halogenated hydrocarbons, heavy metals, radioactive wastes, detergents, agricultural chemicals other than chlorinated hydrocarbons, plastics, acids, and heat.

Prime offenders are heavy metals, halogenated hydrocarbons, and petroleum residues. Experimental evidence indicates that, even at very low concentrations, many industrial chemicals have profound effects on some or all life stages of marine animals, causing mortality, increased prevalence of abnormalities, and physiological disturbances reflected in slow growth or spawning inhibition (Connor 1972; Jeffries 1972; Butler et al. 1972).

In examining and assessing experimental effects of chemical pollutants on living marine organisms, some critical questions must be considered: 1—What is the normal background level (and the observed maximum background level) of the material in relatively uncontaminated waters? 2—What are the observed environmental levels of the material (and the range of these levels) as reported from various polluted marine areas? 3—In what physical forms or compounds does the chemical exist in polluted waters? 4—How does the material enter marine organisms under natural conditions (gills, digestive tract, etc.)? 5—What are the short term (acute) effects? 6—What are the long term (chronic) effects? 7—Are the various life history stages (particularly eggs and larvae) differentially vulnerable to the pollutant? 8—What are the possible synergistic effects of combinations of contaminants?

A serious problem with many experimental studies is that dosage levels often have been far above observed environmental levels. Results of such studies should not be considered as evidence of potential damage to natural populations, since even the more innocuous chemicals may be toxic if sufficiently concentrated. However, experiments with abnormally high concentrations may provide clues about target tissues and the nature of physiological disturbances to be examined with studies using dosages approximating environmental levels. In any event, results must not be extrapolated directly.

Before considering examples of each of the primary classes of offenders, I wish to emphasize some general points.

1. Bioaccumulation—the concentration of pollution chemicals by organisms against a diffusion gradient, and the further concentration of such chemicals at higher trophic levels—is a critical phenomenon, both for the well-being of the organism and for possible effects on humans. Coupled with this is the public health problem of increases in contaminant levels in processed products such as fish meal or fish protein concentrate, where the whole fish may be used instead of fillets.

2. Reworking of sediments, particularly those containing industrial chemicals which may persist for long periods, can be an important source of continuing contamination of water and organisms. Such reworking may result from bottom currents, from dredging and spoil deposition, from activity of benthic animals, and from microbial action.

3. Much of the publicity about heavy metals and chlorinated hydrocarbons in marine organisms has evolved quite naturally from concern about possible toxic concentrations in the flesh of fish and shellfish, and effects on humans. Actually, much of the real impact of these and other contaminants may be on reproduction, survival, and growth of the animal itself. Levels of contaminants harmful to fish and shellfish may be either below or above levels considered harmful to humans.

4. While chemical contaminants may affect any life stage of fish and shellfish, there is much evidence that the earliest stages—eggs, embryos, larvae, and early juveniles—are particularly vulnerable.

Heavy metals—Concentrations and effects of heavy metal contaminants in coastal waters have been subjects of concern and investigation during the past decade, with substantial impetus derived from apprehension over the mercury poisoning incidents in Japan. Experimental evidence increasingly documents harmful effects of some heavy metals on marine animals. Toxicity due to metals varies from species to species but in most instances the nature and extent of long term chronic toxicity is not fully understood. Mercury, lead, and copper are usually considered prime culprits, but recent information suggests that cadmium and other metals may have serious effects.

The toxic effects of mercury for marine species have been summarized by Keckes and Miettinen (1972), and mercury in the marine environment has been discussed in many papers during the past four decades (Lepple 1973). Molluscs have a generally low tolerance to mercury, exhibiting acute toxicity signs at concentrations as low as 0.1 ppm. Fish are also very sensitive to mercury compounds and have remarkable ability to accumulate mercury in their tissues. Gill and skin epithelium is the primary target of mercury ions, and signs of mercury poisoning in fish include body rigidity, widespread fins, eye cataracts, and loss of equilibrium. Toxic effects may be exacerbated by changes in other environmental factors such as temperature and dissolved oxygen. Acute toxicity can occur at 1 ppm, and chronic exposure as low as 0.01 ppm can be lethal.

Cadmium is a normal trace metal in seawater, but in heavily industrialized coastal areas, concentrations may be several times higher than in oceanic waters (Abdullah and Morris 1972; Preston 1973). Cadmium is toxic to marine animals, particularly larvae (Brown and Ahsanullah 1971; Calabrese et al. 1973; Rosenthal et al. 1975).

Eggs and larvae of several species including herring and flounder accumulate cadmium differentially (Westernhagen et al. 1974; Westernhagen and Dethlefsen 1975) and die differentially (Rosenthal et al. 1975). Concentrations of cadmium used in the cited experiments were well above those reported from natural waters.

The cunner *Tautogolabrus adspersus* exhibited deleterious effects after acute exposures to cadmium (Calabrese et al. 1974). Cadmium uptake was rapid, and clearance was slow (Greig et al. 1974); oxygen consumption was reduced in exposures as low as 3 ppm (Thurberg and Dawson 1974); bactericidal activity of phagocytes was reduced at 12 ppm (Robohm and Nitkowski 1974); liver transaminase and reductase activity was depressed (Gould and Karolus 1974); and histopathological changes were noted at higher dosages in kidney, intestine, hemopoietic tissue, epidermis, and gills (Newman and MacLean 1974).

Copper is also a normal trace constituent of seawater that may be toxic at higher than normal levels. Liver disorders in *Fundulus* and flounders were reported (DeCalventi and Nigrelli 1961; Baker 1969) as a result of copper exposures. Degenerative changes in sensory receptor cells (lateral line and olfactory) of *Fundulus* and *Menidia* were reported by Gardner and LaRoche (1973) after exposure to 0.5 ppm of copper. Other pathology included renal lesions and reduced larval survival. These observations point to damage to perceptive structures which undoubtedly influences behavioral responses, but, again, dosage levels were well above those reported from natural waters.

Copper pollution in Raritan Bay, New Jersey, was indicted over a half century ago by Nelson (*cited in* Dewling et al. 1972) as the cause of the virtual extinction of the oyster industry of that bay.

Silver, one of the more toxic heavy metals, is a significant pollutant in waters near some industrialized areas, such as Long Island Sound, where concentrations are 10 times higher than in other western North Atlantic waters (Schultz and Turekian 1965). Oxygen consumption was altered in common species of bivalve molluscs (oyster, hard clam, mussel, soft clam) in chronic exposures to as little as 0.1 ppm silver, and uptake of silver by tissues was rapid (Thurberg et al. 1974). Salinity changes acted synergistically with the contaminant in some instances.

Other metals with demonstrated toxicity to marine organisms include lead, arsenic, chromium, zinc, and nickel (Portmann 1970).

Halogenated hydrocarbons—Pesticides and related synthetic halogenated organic compounds exist in measurable quantities throughout the ocean, and a number of them increase in concentration at higher trophic levels. A large body of evidence—probably the best available concerning contaminant effects on resource species—indicates harmful activity of chlorinated hydrocarbons at very low levels. For example, 50 parts per trillion of DDT will kill newly hatched crab larvae; this is a level 1,000-fold lower than that found in adult crabs in certain areas. Pesticide levels well below those causing mortalities may produce adverse effects on growth and reproduction of marine animals.

Studies of sublethal effects began with Davis (1961) who demonstrated effects on growth of clam and oyster larvae. Subsequent studies (O'Brien 1967; Engel et al. 1972; Roberts 1975) indicate that chlorinated hydrocarbons interfere with enzyme activity, specifically glycolytic and gluconeogenic enzymes, in molluscs and depress oxygen consumption.

Polychlorinated biphenyls (PCBs) from industrial sources are as abundant or more abundant than chlorinated hydrocarbons of agricultural origin in many marine ecosystems. Effects of PCBs on marine organisms are similar to those of organochlorine pesticides. High levels of PCBs in fish were reported by Risebrough et al. (1972). Median concentration on a wet weight basis was 1 ppm for eight fish species sampled in Long Island Sound. Zooplankton from the middle Atlantic continental shelf also contained high PCB concentrations (1.5 ppm, wet wt).

Chlorinated hydrocarbons accumulate in the tissues of adult animals, and several studies have shown that high tissue concentrations in spawning adults caused mortalities in developing eggs and larvae (Burdick et al. 1964; Johnson and Pecor 1969). Reproductive failure of a Texas sea trout population was attributed by Butler et al. (1972) to trophic magnification of DDT residues. The sea trout population inhabited an estuary heavily contaminated by DDT and fed on menhaden which were also heavily contaminated. Sea trout spawning appeared normal, but eggs failed to develop. DDT concentration in the ovaries reached a peak of 8 ppm prior to spawning, compared to less than 0.5 ppm in sea trout from other less contaminated Texas estuaries.

Smith (1973) found evidence that excessive mortalities of larval winter flounders in a Massachusetts estuary could be related to pesticide pollution. Adult females concentrated DDT, DDE, and heptachlor epoxide in their ovaries as spawning approached, and mortality of postyolksac larvae approached 100%.

Petroleum residues—A National Academy of Sciences (1971) study identified petroleum as the major organic pollutant of the marine environment, with estimates of about five million tons introduced annually. Most of this came from air pollution. United States coastal waters are estimated to receive 135,000 tons of petroleum residues annually from sewage treatment plants, with an equivalent amount added by spills, and an additional, much larger, but poorly defined amount by airborne contaminant fallout. Some components of petroleum are toxic to fish and shellfish. Aromatic hydrocarbons such as benzene and toluene are water soluble and can kill fish larvae at concentrations far below those actually measured in the vicinity of oil spills. Benzene, comprising about 20% of the aromatics in crude oil, has been found to kill fish larvae and produce abnormal development in concentrations as low as 5 ppm (Struhsaker et al. 1974); concentrations up to 90 ppm have been measured in the vicinity of oil spills. Growth of juvenile fish was inhibited by benzene, and respiration adversely affected, above 10 ppm. One finding of the study was that 20–25% of larval herring hatched from eggs spawned in San Francisco Bay were abnormal, while a much lower proportion of abnormalities was observed in larvae from eggs spawned in relatively unpolluted Tomales Bay. This observation suggests early damage to mechanisms of normal larval development—damage which may have occurred before or after spawning.

The size of a year class of marine teleosts apparently is determined early in life. Many species have high fecundity and relatively small numbers of adults, so variations in larval survival are critical. Estimates of hatching and survival have been made for many species, including herring and flounders. With winter flounder, for example, Pearcy (1962) estimated that only 16% of spawned eggs hatch, while Saila (1962) estimated a hatch of only 10%. Mortality was estimated to be highest in newly hatched larvae and then declined with time, resulting in 18 survivors to age 1 of 100,000 newly hatched larvae. If pollutants produce added abnormalities, most of them lethal, in larvae, beyond those which occur naturally, this may have significant impact on population abundance.

Experimental exposure of the silverside minnow *Menidia* to crude oil and its fractions produced lesions in the olfactory organs (Gardner and LaRoche in press). Similar damage to sensory organs resulted from exposure to mercury and silver and the pesticide methoxychlor.

Petroleum pollution of hard-shell clam (*Mercenaria mercenaria*) beds in Rhode Island may cause irregular population densities, high mortality rates, and a generalized stress syndrome characterized by cessation of growth, mud blisters on the shell, changes in amino acid balances, and tar-like deposits in the kidney (Saila et al. 1967; Jeffries 1972).

Petroleum pollution can result in reduced market demand for fishery products which taste of the pollutant. The decline in value of Hudson River shad was attributed by Francis and Busch (1973) to the high proportion of fish said to taste like gasoline or kerosene; the off-flavor was attributed to oil spillage at unloading installations as well as to deliberate pumping of bilges of tankers. A similar report on "coal oil" tainting of shad in Newark Bay was made almost a century earlier (Goode 1887). Petroleum pollution and resultant tainting of fish and shellfish have been reported from many other places, at many other times, and for many other species (*see* e.g. Connell 1971; Blumer et al. 1971; Knieper and Culley 1975).

Some coastal waters near industrial centers have been degraded to a point where continued use for fish and shellfish production is difficult. Raritan and Newark Bays in the New York Bight are excellent examples of situations in which the sheer volume of various contaminants precludes most fishing activity. That conditions are not improving is underscored by the fact that coliform populations in Long Island Sound doubled during the decade 1959–1969 and that roughly 15% of the shellfish producing acreage in New York waters has been closed to harvesting because of microbial pollution (Francis and Busch 1973).

Nationally, there is an estimated 1.4% increase per year in acreage of shellfish producing areas closed because of pollution. It is worth noting, though, that the principal effect of pollution has been *closure* of beds rather than *disappearance* of stocks. There have been some localized reductions in abundance, but even in a heavily impacted area like Raritan Bay abundant hard- and soft-clam populations persist (Dewling et al. 1972).

Effects of domestic sewage contamination on fish and shellfish

Domestic sewage contamination may have multiple effects on fish and shellfish. Many toxic chemicals found in industrial effluents and agricultural runoff—pesticides, heavy metals, and petroleum residues—also occur in sewage treatment plant effluents or ocean outfalls of domestic sewerage systems. Additionally, microbial contaminants, bacterial and viral, may force closure of many areas to shellfish harvesting. Still another effect, potentially positive, but with some negative implications, is increased organic loading of coastal waters. This increment should increase productivity, but may contribute to oxygen depletion of water near the bottom, or to blooms of noxious phytoplankton, or of algal species too small to be utilized by shellfish, menhaden, and other filter-feeding animals.

Organic inputs to coastal and estuarine ecosystems from domestic sewage should lead to increased productivity—properly managed, such inputs could be positive factors. Without management, two events may occur: frequencies of algal blooms, some toxic, may increase; and populations of facultative aerobic and anaerobic bacteria may increase. Nutrient inputs, particularly of nitrogen from sewage treatment plant effluents, were identified by Segar and Berberian (1976) as the primary cause of oxygen depletion in the New York Bight apex.

Algal blooms on the coast of the New York Bight have been reported more frequently during the past decade. It is apparent that blooms are now annual occurrences off some parts of the coastline—the northern New Jersey coast in particular.

In 1972 and 1974 paralytic shellfish poisoning (PSP) occurred at dangerous levels on the central New England coast for the first time in history. The toxin is produced by the dinoflagellate *Gonyaulax tamarensis* (*excavata*), which bloomed widely in those years (LoCicero 1975). Similar shellfish toxicity appeared in mussels on the northeast coast of England in 1968 and affected 78 people. Previously such toxicity had been extremely rare in England; a monitoring program instituted after the 1968 outbreak has disclosed that mussel toxicity exceeded the accepted safe level in five of the seven subsequent years (Ayres and Cullum 1975). *Gonyaulax tamarensis* was implicated as the toxin source. This section of the English coast is heavily industrialized and populated, with great pressure for dumping at sea (Nunny 1975).

Evidence associating increasing dinoflagellate and other algal blooms with coastal pollution is circumstantial, but effects of toxic blooms on fish and shellfish are factual. Principal effects are fish mortalities, which occur in areas such as west Florida but only rarely in certain other areas (such as the New Jersey and New England coasts).

On the positive side, increases in marine productivity and in abundance of fish stocks may well be related to the fertilizing effects of domestic sewage and biodegradable detergents. Estimates of nutrient contributions from such sources have been made elsewhere. For example, the North Sea has been estimated by James and Head (1972) to receive 256 tons of phosphorus and 1,548 tons of nitrogen daily from sewage and runoff. Earlier studies of fish catches in the southern North Sea (Graham 1938; Hardy 1956) pointed out that high nitrates and phosphates outside the Thames estuary were derived from the sewers of London, and that the catch of fish was per unit area about double the corresponding catch for the rest of the North Sea, the English Channel, and the Kattegat-Skagerak region. The higher average catch was attributed to nutrients from London.

Effects of coastal pollution on marine aquaculture

Development of aquaculture in coastal waters is being paralleled—probably outdistanced—by increasing pollution. Effects of pollutants on fish and shellfish in coastal waters encompass a broad spectrum, including (but by no means limited to) mortalities caused by chemical pollution or by overenrichment of local areas by domestic sewage, physical and physiological abnormalities resulting from environmental stresses created by pollutants, viral and bacterial contamination, toxic phytoplankton blooms in culture areas, failures of larval development and setting in shellfish hatcheries as a consequence of poor water quality, and neoplasms and other abnormalities in polluted zones (Sindermann in press).

The major pollution-related problems in open system culture seems to be those produced by industrial chemical and domestic microbial contaminants. Two general problem areas have elicited some research and accumulation of specific information: public health problems, and problems associated with reproduction, survival, and growth of cultivated animals.

Public health problems—Public health aspects of ocean pollution can be categorized as those related to microbial and chemical contamination of food.

Microbial problems can be major deterrents to open system culture. Virus diseases such as typhoid and hepatitis have been transmitted by ingestion of raw shellfish from polluted waters (Mason and McLean 1962). Viruses have been found experimentally to have variable but often surprisingly long survival time in saline water (Metcalf and Stiles 1966). There is little information about their viability in sludges and in bottom sediments.

Viruses affecting humans, then, will constitute a critical problem for culture operations in coastal and estuarine areas where even limited domestic pollution exists—and this includes most of the areas now used or planned for use in marine aquaculture. Possible viral contamination will be an important issue where treated sludges or other fecal degradation products are used for enrichment of growing areas, until large-scale inexpensive techniques are available to ensure viral destruction.

While viruses constitute the most vexing public health problem in open system culture, pathogenic enteric bacteria form a continuing threat when raw or partially processed products are consumed. The role of a marine vibrio, *V. parahemolyticus*, in outbreaks of "food poisoning" in the Orient (and recently in the United States) related to seafood consumption is under study. This vibrio is a normal constituent of the inshore flora, but its abundance may be increased facultatively by the enrichment of coastal and estuarine areas. Other pollution-associated bacteria, such as *Clostridium*, *Salmonella*, and *Shigella*, should not be ignored since a single outbreak of disease related to any marine species can have drastic impact on markets for all the others.

An example of the potential danger to humans from bacterial contamination of cultivated marine animals occurred in late summer of 1973. Cholera outbreaks, traced to consumption of raw mussels from heavily polluted coastal waters, occurred in two Italian cities (Naples and Bari). The bacterium responsible is fragile and readily destroyed by normal sanitary methods. However, it can survive for weeks in seawater and can be accumulated by molluscs growing in polluted waters.

Wherever even one case of cholera occurs, the danger of shellfish contamination exists. In Portugal (Ferreira and Cachola 1975) *Vibrio cholerae* occurred in 38% of 166 samples of molluscs taken in 1974 from near Tavira, where a case of cholera was reported. This report was a sequel to an earlier one (Cachola and Nunes 1974) pointing out extensive pollution of shellfish growing areas on the southern (Algarve) coast of Portugal. [By coincidence, the day before I presented this paper, the *New York Times* on 2 November 1975 reported over 200 cases of cholera with three deaths in Coimbra, Portugal. Health authorities there attributed the outbreak to contaminated cockles from the Mondego River estuary.]

The horrors of Minamata Disease, caused by mercury contamination of fish and cultivated shellfish in a bay in Japan, have been widely publicized (e.g. Smith and Smith 1975). High mercury levels found in 1971 in swordfish effectively eliminated a developing fishery in the Gulf of Mexico and caused a ban on sale of swordfish in the U.S.

Certain processing methods, especially production of fish protein concentrate, result in higher concentrations by weight of mercury in the product than in the raw fish—concentrations that may exceed allowable limits for human consumption (Landis 1972). Mercury (and presumably other contaminants) can be removed (Regier 1972), but the cost may preclude removal and there is some problem with toxicities of extractants.

Possible toxic effects of pesticides in foods have been discussed often. Butler (1969*a*) stated that even though persistent pesticide residues were widespread in the world fauna, their magnitude was for the most part too small to have significant effect on human health. A U.S. program to monitor pesticide levels in estuarine fish and shellfish (Butler 1969*b*) has shown pesticide residues in all samples from 15 states involved in the program to be below levels considered hazardous to humans. Even though use of pesticides is coming under some measure of control in the United States, their use in other parts of the world is expanding. Because of their persistence in the environment, and their accumulation by successive levels of food chains, pesticides continue to threaten nearshore ocean areas, including those devoted to marine aquaculture.

Some petroleum derivatives are carcinogenic; this fact has been used, particularly in the public press, to argue against offshore oil drilling and establishment of offshore oil terminals (*see* e.g. Grossman 1972). Points made are that oil is persistent for long periods in the marine environment and that oil hydrocarbons are concentrated by food chain organisms and transferred to humans when they consume seafood.

But it should be clear that there is no evidence for direct association of carcinogens in seawater, fish, shellfish, or bottom sediments with cancer in humans. No case of human cancer has been unequivocally traced to chemical contaminants from the marine environment. However, this does not imply that there is no danger from contaminated marine sources.

Another public health problem is biotoxins—paralytic shellfish poisoning and ciguatera in particular. Seasonal toxicity to humans has characterized molluscan species from certain coastal areas, and toxicity of some species of fish from particular locations in tropical and subtropical waters has long been known. Changes in distribution of toxicity may be related to human modification of coastal or atoll waters. The spread of shellfish poisoning on the New England coast was discussed earlier. European reports of paralytic shellfish poisoning outbreaks seem related to very high nutrient levels in coastal and harbor areas (Korringa 1968).

Problems associated with survival, growth, and reproduction of marine animals—Estuarine and coastal species, including those important to marine aquaculture, survive in a variable environment. Effluvia from human terrestrial populations can further modify that environment beyond levels tolerable to certain species, or can modify environmental factors enough to place continuing stress on the species, or can introduce chemical compounds with which the species has had no previous evolutionary experience. Problem areas can again be generally categorized as microbiological and chemical.

The enrichment of nearshore waters by sewage discharge and sludge dumping can expand and modify the marine and estuarine microbial flora. Wherever organic content of water or sediments is increased, as it would be in most marine aquaculture areas, facultative bacteria such as vibrios, pseudomonads, and aeromonads may also increase. Concentrations of such bacteria may provide sufficient infection pressure on fish or shellfish so that disease and mortalities result. Bacterial fin rot (discussed later) which characterizes many species of fishes in heavily polluted waters is evidence of this.

Human bacterial pathogens may be able to infect fish. Janssen and Meyers (1968) demonstrated antibodies to a spectrum of human pathogens in fish taken from parts of Chesapeake Bay near human population centers. Fish taken from waters near sparsely populated areas did not contain the antibodies (Janssen and Meyers 1968; Janssen 1970). Human pathogens would probably not be virulent for fish, but the presence of antibodies could be used as pollution indicators in aquaculture areas.

The extreme sensitivity of molluscan larvae to pollutants has been used in the development of bioassay techniques, particularly on the west coast of the U.S. (Woelke 1967, 1968). Additional evidence of molluscan larval mortality after experimental exposure to heavy metals has been reported from Japan and Australia (Okuba and Okuba 1962; Wisely and Blick 1967). Davis (1961) and Davis and Hidu (1969) described severe retardation and lethal effects of 52 commercial pesticides, herbicides, etc. on oyster and clam embryos and larvae. Growth rates of adult oysters have been experimentally depressed by pesticides (Butler 1960), and residues of these compounds in oysters sampled from certain U.S. waters exceeded levels at which growth was retarded, indicating an existing deleterious impact on oyster populations. Furthermore, DDT-contaminated diets fed to shrimps, crabs, and fish (croaker and pinfish) caused significant and rapid mortalities (Butler 1969*a*). Effects of pesticides on reproduction of fish have been noted in other studies (Burdick et al. 1964) and it is likely that pesticide pollution causes significant increases in mortalities of sensitive stages of estuarine fish and shellfish populations. At low concentrations the effects would be continuous, but possibly at a level below that at which they would be readily noticed.

The implications of pesticide contamination in aquaculture areas are clear, but often overlooked. Sudden lethal intrusion of such chemicals (and heavy metals as well) from treated agricultural areas following heavy rains could cause mass mortalities, even eliminating entire crops, while lower levels could degrade otherwise excellent marine aquaculture areas to the status of marginal or unproductive for most species.

In addition to pesticides, industrial halogenated hydrocarbons, including PCBs, are widely distributed in the marine environment (Risebrough 1969; Duke et al. 1970). Toxicity to juvenile shrimps was demonstrated by Duke et al., suggesting an added negative factor for shrimp and possibly other crustacean marine aquaculture.

Nimmo et al. (1971*b*) and Nimmo and Bahner (1974) found that 1 ppm PCB kills 50% of adult shrimps in 15 days. Increased salinity stress hastens death.

Petroleum residues pose another chemical problem to marine aquaculture. Risk of massive contamination of growing areas by accidental spills will always be present, especially as ocean transport of petroleum increases. In addition to such isolated catastrophes, the long term contamination of coastal waters by air fallout, pleasure boat exhausts, and sewage treatment plant discharges can affect growth and quality of marine animals. Sidhu et al. (1971) found that Australian mullet took up petroleum hydrocarbons from coastal waters with very low levels of pollution. Fatty infiltration of liver and high lipid content of muscle characterized the affected fish. Kuhnhold (1971) found fish eggs and larvae to be sensitive to various crude oils. Mortality of eggs was proportional to petroleum concentration, and larvae were more sensitive than embryos.

Because bays and estuaries are particularly vulnerable to oil spills, effects of petroleum on early life stages of shellfish have been of particular interest. Wells (1972) found that Venezuelan crude oil was toxic to lobster larvae at 100 ppm, with sublethal effects at concentrations as low as 1 ppm. Renzoni (1973) found a significant decrease in fertilization rate of eggs of oysters and mussels when exposed to crude oil and crude oil-dispersant mixtures. Sperm were particularly vulnerable, suggesting a possible pathway to reduced fecundity of bivalve populations in oil-polluted estuaries and embayments.

Adult clams can be adversely affected by local oil spills and residual contamination of beds (Dow 1975). Production from contaminated areas declined 20% in 2 years, while adjacent uncontaminated areas increased nearly 250% in yield. Clams transplanted to contaminated areas declined 65% in growth after transplantation, and survival was significantly poorer than in uncontaminated control areas (12.8% vs. 78.1%).

Effects of coastal pollution on recreational fisheries

Coastal and estuarine habitats of many sportfish species are being degraded—particularly estuarine nursery areas—but there is yet no great amount of evidence that pollution has had a significant impact on abundance of sportfish species *over their entire range*. However, there is again adequate evidence for *localized effects*. Several New Jersey rivers—Raritan, Hackensack, Arthur Kill, Passaic, Newark, and upper Delaware —must be considered badly degraded.

In the Delaware River near Philadelphia, summer dissolved oxygen values drop to near zero preventing ascent of adult striped bass and descent of juveniles. Chittenden's (1971) report on the status of the striped bass in the Delaware River provides an excellent example of localized destruction of spawning runs and spawning-nursery areas by pollution. The species has declined drastically in the river since the late 1800s. Pollution of tidal freshwater has destroyed the river's potential as a spawning and nursery area and has prevented use of upstream waters as well. Chittenden found that 99% of young-of-the-year striped bass died at 0.5 ppm oxygen, and some would die at 2 ppm. He considered the minimum permissible oxygen level for normal existence to be about 3 ppm at 17°C.

Shad have also declined in the Delaware River since 1900, but effects of the pollution block in tidal freshwater have not been as severe as for striped bass. Shad move through the polluted area late in the fall when water temperatures are lower and oxygen concentrations are higher. Also, shad spawning areas have shifted upstream beyond the areas of greatest pollution impact. Striped bass have apparently not adapted in this way. These species, and others of recreational interest, are caught between degraded habitats for spawning and nursery areas and ever-increasing fishing pressure, both sport and commercial, on adult and juvenile stocks.

Coastal pollution influences the angler as well as his prey. Those who fish from shore are confronted with tar balls and other floatable debris and wastes. The catch may have fin rot, and the flavor may be less than appealing. Despite these inducements to stay away, the number of marine recreational fishermen is increasing at a phenomenal rate. This is particularly true in the New York Bight where, to quote from a recent issue of Eastern Airlines *Pastimes* (July/August 1974) "at no other place in the world are so many anglers out on the water at one time."

Demonstrated pollutant effects on sportfish and sportfishing must be categorized as primarily esthetic, with localized damage due to low oxygen, and some kills due to pollutant stress. For species such as striped bass, the positive effects of enrichment of spawning areas may be counterbalanced by negative effects on spawning and survival. The *net* effect in strategic estuarine areas (such as Chesapeake Bay, Delaware Bay, and Long Island Sound), when considered over the entire range of the species, should provide an estimate of human pollutant impact. Several species of great interest to the coastal recreational fisherman—striped bass, weakfish, and bluefish—have been very abundant during the past few years, and catches are exceeding those of the previous two decades.

Pollution-associated diseases of fish and shellfish

Marine animals are subject to a wide spectrum of diseases of infectious or noninfectious etiology (disease can be defined in the broad sense as any departure from normal structure or function of the animal). Disease usually exists in an enzootic form, weakening or disabling individuals and rendering them more susceptible to predators or other environmental stresses. Occasionally, though, epizootics and mortalities comparable to the great plagues of the Middle Ages may sweep through animal populations. In marine species such massive epizootics caused the great herring mortalities of the mid-1950s in the Gulf of Saint Lawrence and the extensive oyster mortalities of the 1960s in the Middle Atlantic States. These epizootics are triggered by interactions of pathogen, environment, and the host population. If we consider only the environmental aspects of such outbreaks, any departure from normal environmental conditions produces a degree of stress on the population, and many contribute to an increase in prevalence of a pathogen, or of facultative invaders. Some of these environmental factors are abnormal temperature change, lack of adequate food, or overcrowding. Resistance of the host animal to the disease is of course intimately related.

Human activity has introduced or has increased environmental stresses for fish in estuarine and coastal waters. We have, for instance, added pesticides and other synthetic chemicals which can, even in low concentrations, drastically affect the physiology of fish and shellfish, and with which the species has had no previous evolutionary experience. Heavy organic loads, in the form of sewage sludge and effluents, can produce anaerobic or low-oxygen environments. These wastes are often accompanied by other contaminants such as heavy metals, which interfere with enzymes of both fish and their food organisms.

Several diseases that appear to be related to pollutant stress in fish and shellfish have been studied. Probably the best known is a nonspecific condition known as "fin rot" or "fin erosion" in fish (Fig. 2), a syndrome that seems to be associated with degraded estuarine or coastal environments. Fin rot has been reported from the New York Bight (Mahoney et al. 1973; Ziskowski and Murchelano 1975; Murchelano 1975), Narragansett Bay (Levin et al. 1972), California (Young 1964; S. Calif. Coastal Water Res. Project 1973; Mearns and Sherwood 1974), Florida (Sindermann unpublished observations), Japan (Nakai et al. 1973), and the Irish Sea (Perkins et al. 1972). There are at least two types of fin rot: one occurring in bottom fish where damage to fins seems related to direct contact with contaminated sediments, and a second in pelagic nearshore species, characterized by predominant involvement of the caudal fin, with hemorrhagic lesions.

Fin rot is common (up to 38%) in trawled marine fishes from the New York Bight. Thus far, 22 affected species have been found. While bacteria of the genera *Vibrio*, *Aeromonas*, and *Pseudomonas* were frequently isolated from abnormal fish, a definite bacterial etiology has not been determined. Fin rot disease was significantly more abundant in the New York Bight apex—the area of greatest environmental damage—than in any comparable coastal area from Block Island to Cape Hatteras. High

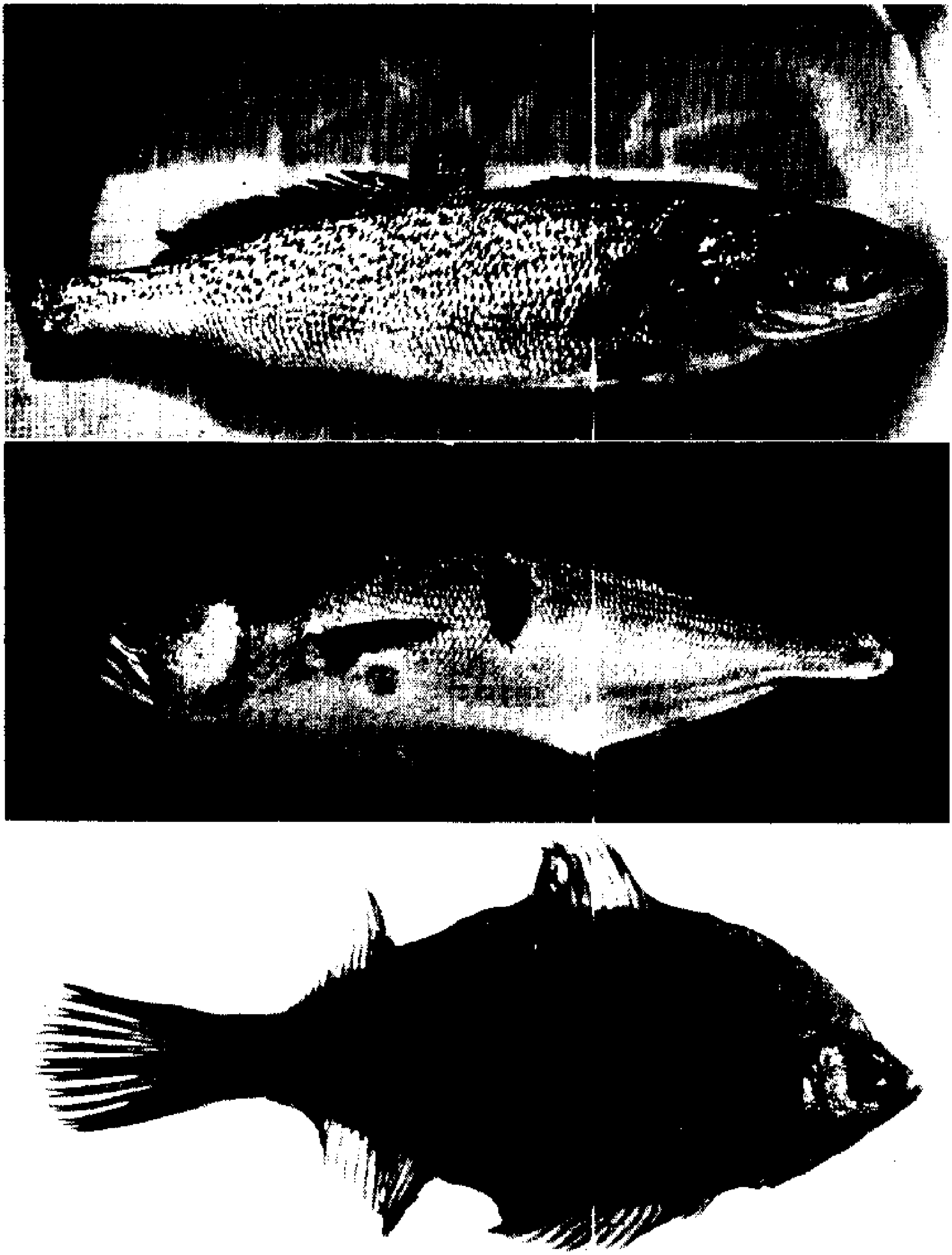

Fig. 2. Fin erosion in weakfish, bluefish, and winter flounder (in descending order). (From Am. Littoral Soc. 1972; Ziskowski and Murchelano 1975; Mahoney et al. 1973.)

fin rot prevalence may be associated with high coliform counts in sediments (Mahoney et al. 1973) and with high heavy metal levels in sediments (Carmody et al. 1973). The disease signs can be produced experimentally by exposure of fish to polluted sediments.

Information from southern California (S. Calif. Coastal Water Res. Project 1973) also strongly suggests an association of fin rot with degraded habitats; relevant statements are (p. 249): "The incidence of fin erosion was high in areas with high concentrations of waste water constituents in the sediments . . .", and (p. 424) "Although there is a definite association between fin erosion and waste water discharges, the causal factors are unknown. . . . Nearly half of the 72 species caught off the Palos Verdes Peninsula were affected with this syndrome" (eroded fins).

Fin rot, with associated mortalities, was reported by Couch (1975) in croaker (*Micropogon undulatus*) and spot (*Leiostomus xanthurus*) from Escambia Bay, Florida. The disease syndrome and mortalities have been observed for several years during periods of high temperature and low dissolved oxygen. Escambia Bay has been polluted by the PCB Aroclor 1754 for years (Duke et al. 1970). Probably the fin rot syndrome in fish includes chemical stress, possibly acting on mucus and epithelium, stress resulting from marginal dissolved oxygen concentrations, and secondary bacterial invasion in at least some instances.

Invertebrates such as lobsters, shrimps, and crabs from polluted areas of the New York Bight are also abnormal, with appendage and gill erosion a most common sign (Young and Pearce 1975; Gopalan and Young 1975). Disease signs were produced experimentally in animals held in aquaria containing sediments from sewage sludge dumping or dredge spoil disposal sites. Dead and moribund crabs and lobsters have been reported by divers in the New York Bight apex, and dissolved oxygen concentrations near the bottom during summer often approach zero (Young 1973).

Evidence for association of environmental contamination with other abnormalities in fish is increasing. High prevalences of tumors have excited interest in a number of studies—

1. Russell and Kotin (1957) found 10 of 353 white croaker (*Genyonemus lineatus*) from Santa Monica Bay, California, with papillomas of lips and mouth. Fish were taken 2 m from an ocean outfall. No tumors were found in 1,116 croakers from nonpolluted waters 70 km away.

2. Lucke and Schlumberger (1941) described 166 catfish (*Ameiurus nebulosus*) with epitheliomas of lips and mouth, taken from the polluted Delaware and Schuylkill Rivers near Philadelphia. Tumors of this type may result from mechanical, infectious, or chemical irritation. Catfish from other areas did not have a high prevalence of tumors. The investigators did not exclude the possibility that the lesions were induced by chemical carcinogens in waters. The lesions developed into epidermoid carcinomas, some of which were invasive.

3. Cauliflower disease (epidermal papilloma) has been increasing in prevalence in Baltic eels since 1957 (Koops et al. 1969). The pattern of spread and high prevalence suggests an infectious process (viral arrays have been seen) or progressive accumulation of industrial contaminants such as fuel oil and smelter wastes (known to contain carcinogenic hydrocarbons such as benzopyrene and heavy metals such as arsenic).

4. Halstead (1971) found that 12% of nearly 16,000 English sole from San Francisco Bay had as many as 33 tumors per fish. Highest incidence was in the northern part of the bay where the greatest concentration of petrochemical wastes existed.

5. Young (1964) found many small (4–6 inch) dover sole (*Microstomus pacificus*) from Santa Monica Bay with tumors. Fish above 10 inches long did not have tumors. Since 1956 numerous white croakers from Santa Monica and Los Angeles–Long Beach were found with papillomas of the lips, and papillomas were observed on tongue soles, cusk eels, and Pacific sand dabs. Such tumors were never seen on fish from unpolluted areas.

6. Carlisle (1969) found "growths" frequent on white croakers and dover sole

from Santa Monica—also fin rot in dover sole.

7. Sindermann (unpublished data) found warty tumors in mullet from Biscayne Bay, Florida. Such tumors are unknown in mullet from other coastal areas.

In southern California and in Japan, skeletal deformities and anomalies occur in fish sampled in areas of coastal pollution. Skeletal deformities occurred with greater frequency in samples from areas with significant pollutant stress (Valentine and Bridges 1969; Valentine et al. 1973). Exposure of fry to very low concentrations of DDT (<1 ppb) produced significant anomalies in fin rays (Valentine and Soulé 1973).

An association of disease and low-level chronic exposure to pollutant chemicals is being explored (Nimmo et al. 1971*a*; Couch 1974*a*). A virus disease of pink shrimp reached patent levels and caused mortalities of 50–80% in shrimps exposed to the PCB Aroclor 1254 and to the organochlorine insecticide Mirex (Couch and Nimmo 1974; Couch 1974*b*). Other experiments in which the shrimp were crowded, but not exposed to chemicals, resulted in similar enhancement of virus infections, suggesting that environmental stress may be an important determinant of patent infections.

Another series of experiments, using the spot *Leiostomus xanthurus*, resulted in experimental production of fin rot disease following exposure to 3–5 μg/liter of Aroclor 1254 (Couch 1975). Mortalities up to 80% were reported.

An association of high disease prevalence or disease enhancement in fish and shellfish sampled from thermal effluents has been made recently. Farley et al. (1972) described a lethal virus disease of oysters held in heated discharge water in Maine. The disease, apparently a low-level enzootic in oysters growing at normal low environmental temperatures (12–18°C summer temperatures) seems to proliferate in oysters at elevated temperatures (28–30°C).

Unpublished observations by personnel of the Sandy Hook Laboratory of the Middle Atlantic Coastal Fisheries Center point to high prevalence of lymphocystis disease in limited samples of striped bass overwintering in the heated effluent of a Long Island generating station. This virus disease is rare in striped bass (*Salt Water Sportsman* 1951; Krantz 1970), and its unusual abundance in a localized population may well be related to the abnormally high winter temperature regime in which the population exists or to abnormal crowding. As with the oyster virus disease, high temperatures may promote survival or transfer of the pathogen, or lower resistance of the host, resulting in grossly recognizable stages of infection. An additional area of concern about fish overwintering in heated effluents is that a focus of infection will be provided to infect incoming spring migrants. Bacterial fin rot of striped bass overwintering in heated effluents has also been reported recently by fishermen.

Conclusions

There are numerous examples of localized effects of pollutants on small segments of fish and shellfish populations, but no specific evidence of widespread damage to major fishery resource populations that can be attributed directly to pollution. Other factors, such as repeated year-class successes or failures, shifts in geographic distribution of fish populations, or overfishing (particularly by foreign fleets) may cause pronounced changes in fisheries—changes which could obscure any effects of habitat degradation.

It may be, of course, that coastal pollution is exerting some overall influence on certain resource species, but that this may be masked by increased fishing effort, or by favorable changes in other environmental factors which create a positive effect on abundance, outweighing any negative effects of pollutants.

A recent review of the fishery resources of the United States (Wise 1974) discussed the catches of all important marine fish and shellfish on a species-by-species basis for the past 20 years. Included are a number of Atlantic species usually considered estuarine-dependent or estuarine and nearshore inhabitants—species which might be expected to demonstrate some effects of estuarine and coastal pollution. With few exceptions, catches of these species have been relatively steady or have shown some increases in the past 20 years. Selected examples of catch trends are shown in Fig. 3. One of the summarizing statements in Wise's book is of particular note.

> In general there are no good fishing effort data available for those estuarine species whose catches have remained constant, so it is possible that maintenance of catch levels is due to constantly increasing fishing effort. Nonetheless, *the evidence from catch records of a substantial number of exploited estuarine species indicates that pollution and damage to estuaries have not yet shown any measurable overall effect on the part of the marine resource which might be expected to show the first effects* [italics added].

This, of course, discounts the possibility that anadromous or estuarine species may be resistant to some environmental changes and that species such as striped bass may actually benefit from certain manmade environmental alterations. Intensive foreign fishing, in the late 1960s and early 1970s exerted a much more severe impact on a number of major fish stocks in U.S. continental shelf waters than did pollution, insofar as we can determine such effects. However, most of the stocks affected by foreign fishing are not the ones which would be most directly affected by estuarine–coastal pollution.

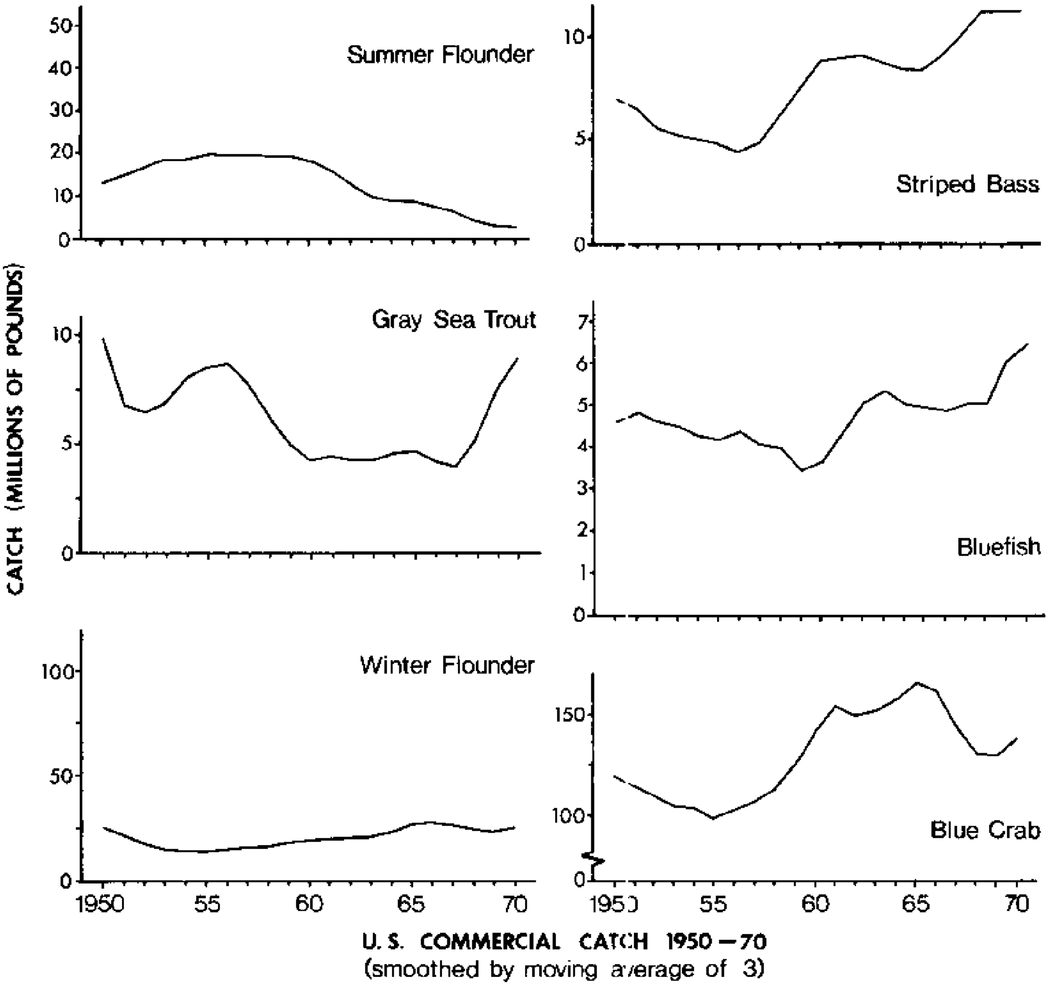

Fig. 3. Catches of selected estuarine–coastal fishes 1950–1970 (redrawn from Wise 1974).

Middle Atlantic catches of a number of estuarine–coastal species have increased sharply since 1970, including bluefish, weakfish, striped bass, blue crabs, menhaden, scup, and summer flounder. Catches seem to reflect at least temporary increases in population abundance, but causal factors are unknown. What is badly needed, of course, is more information about the abundance of marine species and about the factors, natural and manmade, which affect abundances. Only with this information can we expect to identify fully the effects of pollutants on living resources.

Considering fish catches in the New York Bight specifically, trends in New York and New Jersey total commercial food fish and shellfish landings have been generally downward for much of the past 25 years. A number of factors other than pollution were involved (McHugh 1972). These other factors were: lack of effective management, overfishing of some stocks, low prices and high costs, and natural fluctuations in abundance. It is important to note that statistics cited by McHugh do not include sport catches, which for some species far outweigh commercial catches. For example, it is estimated that the weakfish sport catch is three times the commercial catch, striped bass four times, and bluefish fifteen times the commercial catch (for the entire Atlantic coast). The coastwide trends in catches of estuarine–coastal species do not suggest immediate collapse, despite the rather glum picture in New York and New Jersey.

Mass mortalities of marine organisms due to pollutants or environmental stresses related to pollution may constitute only a small part of the total damage to coastal fish and shellfish populations. Sublethal effects such as spawning failure, poor survival of larvae, larval setting failure in the case of shellfish, reduced growth rates, and increased vulnerability to other environmental limiting factors, can have significant effects that may be less apparent. Greater understanding of such sublethal effects is needed to assess fully the influence of pollution on living resources.

Field investigations of possible relationships between coastal pollution and localized damage to fish populations usually yield evidence that must at best be described as circumstantial. Experimental evidence is factual, but is often difficult to relate directly to events in the sea, since in most studies one or a few environmental variables are artificially isolated and examined, while a host of possible synergisms, antagonisms, and other interactions are unavoidably overlooked.

After all the damage is done to marine populations, it is the potential problem of pollutant residues in harvested animals that is of ultimate concern. Public health aspects of seafood consumption—microbial and chemical—are of great importance. Management of living marine resources for optimum yields as food must include management of the chemical and microbial environments in which the animals are grown. Effective environmental management would ensure that fish and shellfish will be edible when they are harvested—that they will in fact be resources rather than potential health hazards.

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Middle Atlantic fisheries: Recent changes in populations and outlook

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Abstract

Groundfish survey data from 1963–1965 show that about 22% of the available finfish fishery resources were being harvested. In 1964–1967, standing crops decreased about 40%, indicating that harvesting was at near the maximum rate. All fishery resources were nearly depleted by 1975. Bringing back certain species to traditional levels of abundance would require reducing catches of other species so low as to create more serious problems. The ICNAF "second tier quota," or biomass approach, evolved from the need to deal effectively with these problems.

It is essential that the factors influencing recruitment be determined; these factors may be more density independent than many suppose. The concept of maximum sustainable yield ignores these factors.

We all know that the fisheries off U.S. coasts attract more than just Americans. Hardly a day passes without newspaper stories about the "foreign fishermen," or the progress of the "200-mile bill" in Congress, or concern over the health of the ocean. There has never, in my experience at least, been more interest in the ocean and its resources than there is today. Frankly I wish that there were a little less such interest.

I will describe what has happened to fishery resources in recent years and what progress has been made to bring order out of chaos in the management arena. Also I will discuss some ecological aspects of the marine ecosystem and suggest a somewhat different approach to evaluating the fishes' role in the ecosystem.

Except for Canadian vessels, foreign vessels operating off U.S. coasts beginning in the late 1950s were mostly survey or scout vessels. But by 1963 the foreign fleet was a significant presence. Initially these fleets were interested primarily in herring and silver hake and were mostly harvesting resources that were not the direct or immediate objects of American fisheries. The fleet grew quickly; as more vessels arrived the fisheries expanded their coverage in area, time, and resources sought. Total finfish landings from Georges Bank to Cape Hatteras reached the 1 million ton mark by 1966, roughly three times the amount taken annually from these same waters by Americans since 1950. From 1966 to 1975 total annual finfish landings have averaged about 1 million tonnes.

Clark and Brown (1975) reviewed the changes in finfish biomass and squid from the area for 1963–1974. They found that through 1968 there appeared to be little change in the predicted stock size despite the catch (Table 1). After 1968, the decline is obvious. When herring and mackerel are excluded, there is a steadier decline at the rate of about 6% a year. While reported landings averaged slightly over 1 million tonnes since 1968, fishing effort continued to increase. Other studies indicated that the fishing effort in 1968 was about the maximum that should be allowed, given the existing production capacity and the nature of the fishery.

Data obtained from fall groundfish surveys demonstrate the impacts of fishing. The unadjusted data (Fig. 1) for the stratified mean catch/tow of all species (finfish plus squid) and of haddock graphically illustrate Clark and Brown's main points. The overall biomass has consistently trended downward, with some irregularities principally due to substantial changes in abundance of pelagic fishes, such as mackerel and herring. Trends in demersal fishes have been more pronounced and steeper, with some curious exceptions.

Edwards (1968) reviewed the status of

Table 1. Yearly rate of change in the finfish and squid in ICNAF Subarea 5 and Statistical Area 6.

	Linear values				
		All data		Data for herring and mackerel excluded	
Year at midpoint	Landings tonnes $\times 10^3$	Predicted stock size tonnes $\times 10^3$	% change	Predicted stock size tonnes $\times 10^3$	% change
1963	651	5,240		2,886	
1964	786	4,564	−13	2,364	−18
1965	955	5,215	+14	1,895	−20
1966	989	6,849	+31	1,707	−10
1967	760	6,489	−05	1,650	−03
1968	943	6,952	+07	1,741	+06
1969	1,029	5,125	−26	1,690	−03
1970	840	4,904	−04	1,542	−09
1971	1,113	2,719	−45	1,519	−02
1972	1,115	2,690	−01	1,586	+04
1973	1,154	2,226	−17	1,512	−05
1974	983	1,592	−28	1,174	−22
		(1,731)*	$\overline{X} = -5.9$	(1,080)*	$\overline{X} = -6.0$

* Observed value.

the fishery resources of the North Atlantic area using groundfish survey data. The data presented were for the period 1963–1965. Biomass estimates were included for the 27 most abundant species at that time. Weighting coefficients used to estimate biomass were developed a priori, based on our knowledge of the fishes' behavior and distribution. Weighting coefficients developed since, by comparing independently developed estimates of biomass with the groundfish survey data, are generally comparable with those used in the 1968 document. Weighting coefficients for herring, mackerel, and some other pelagic species vary considerably with season, area, and apparently overall abundance. Bearing this in mind, I have calculated the biomass for these same species for the period 1972–1974 (Table 2). The value for herring is based on estimates obtained from virtual population analyses. Values for mackerel and alewife are probably suspect. The comparison between the two periods is, however, interesting.

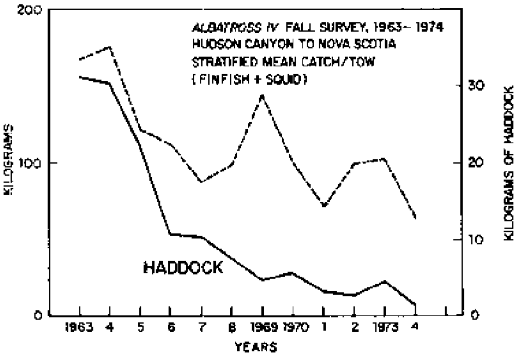

Fig. 1. Stratified mean catch/tow of fishery resources in the area from the Hudson Canyon to Nova Scotia. The dashed line is for all resources and the unbroken line for haddock only. The confidence limits are roughly half or double.

Alewives, haddock, herring, big skate, barndoor skate, ocean pout, and argentine show substantial declines. Few species have shown dramatic increases except mackerel, which is now relatively abundant. In the late 1960s mackerel was at least twice again as abundant as now, and it seems now to be decreasing rapidly. Some species like cod and redfish seem to have held their own. A few species not fished extensively, like white hake and scup, have apparently increased in abundance. However, such an analysis is useful only for comparing different periods. Some dramatic short term trends don't show up—for example, yellowtail flounder. Between 1972 and 1974 yellowtail abundance dropped from comparatively high to very low levels, because of fishing pressures and recruitment failures.

In 1963–1965, there was an estimated finfish biomass of about 14.2 billion pounds (6.5 million tonnes) with a harvest of about 2.6 billion pounds (1.2 million tonnes). In 1972–1974, from an estimated finfish biomass of about 8.8 billion pounds (4.0 million tonnes) about 2.8 billion pounds (1.3 million tonnes) was harvested. The basis for these estimates, while quite different from that developed by Clark and Brown, provides numbers similar to theirs.

By 1970 it was obvious that all was not well. In just a few years almost all species directly sought came under catch limitations. This was an insufficient control; other measures were considered, including effort control and a "second tier" or "total fish ecosystem" quota. The first direct approach to the need for management of the total ecosystem, by whatever means, was made at

Table 2. Estimated biomass of selected species for periods 1963–1965 and 1972–1974 (in lbs × 10^{-6}).

	1963–1965		1972–1974	
	Estimated standing crop	Average landings	Estimated standing crop	Average landings
Silver hake	2,084	638	1,215	312
Atlantic herring	4,000*	529	1,500*	774
Spiny dogfish	1,373	+	978	+
Haddock	980	337	161	42
Red hake	694	176	443	131
Pollock	596	54	576	72
Thorny skate	579	—	550	+
Cod	489	156	399	123
Redfish	399	54	406	51
Little skate	386	—	275	+
Butterfish	309	10	400	+
Big skate	295	+	84	+
Argentine	187	9	23	+
Blackback	185	32	113	+
Yellowtail	185	93	174	70
Barndoor skate	178	+	48	+
Ocean pout	171	12	6	+
Goosefish	142	+	140	+
American dab	125	9	66	(56)
Mackerel	100	+	1,800	794
Alewife	87	42	8	+
White hake	72	7	149	+
Longhorned sculpin	71	+	34	+
Sand flounder	68	+	100	+
Scup	66	10	183	+
Witch flounder	32	4	34	(56)
Fourspot flounder	16	+	12	+
Sea raven	10	+	5	+
All others	308	460†	455	386†
Totals	14,187	2,630	8,837	2,811

* Estimate not based on Edwards (1968).
† Includes landings of all species listed in table as well as species in "all others" category.

the June 1972 International Commission for the Northwest Atlantic Fisheries Meeting when data were presented to illustrate the overall biomass changes that had been observed, based on groundfish survey data (Grosslein 1972) and the relationship of total fishing effort to yield (Brown and Brennan 1972).

In October 1973, at a special meeting in Ottawa, Canada, the ICNAF nations agreed to an overall catch quota for finfish, including squid, but excluding certain other species such as menhaden, tunas, swordfish, and other high-seas pelagics. This quota, presently termed the "second tier quota" was set at 924,000 tonnes for 1974, 825,000 tonnes for 1975, and 650,000 tonnes for 1976.

E. D Anderson (personal communication) evaluated the impact of the 650,000-tonne total quota for 1976 on the various fish stocks within the area. Some stocks were in very poor shape, others in relatively good shape. With close adherence to the total quota, it will take no less than 5 years for the ecosystem to rebuild to where a harvest of about 1 million tonnes can be sustained under ideal circumstances. Some stocks, for example haddock, yellowtail flounder, and herring, are so depleted that full recovery cannot be expected in less than 7 years. And there is no reason to suppose that mackerel and herring can be maintained at maximum levels simultaneously. Nor can we expect any significant response from the ecosystem in less than 2 years.

The total ecosystem quota approach was developed for several reasons, including 1) the by-catch problem and 2) the desire to maintain the ecosystem in its most productive and useful state. The first of these lends itself to documentation and analysis; the second is intuitive and difficult to demonstrate.

Edwards and Hennemuth (1975) discussed the by-catch problem. Put simply, few if any fisheries and certainly no trawl fisheries are characterized by catches of just the species sought. The mixed species situation is pronounced in many areas of the continental shelf—certainly the New England continental shelf (Edwards 1958; Grosslein 1969; Grosslein and Bowman 1973). It can become critical wherever fishes are managed by species quotas and most of the available species are sought. The problem is further exacerbated in a trawl fishery when there are large differences in the size of meshes used. A small-mesh fishery may be mitigating the benefits of a large-mesh fishery for a different species by taking the small fish as by-catch.

Fisheries cannot be usefully classified as "haddock" or "herring" fisheries if the by-catch is ignored. The by-catch is ignored if it isn't documented, whether it be discarded or consigned to the fish meal plant. In many cases in ICNAF Subarea 5 and Statistical Area 6 the by-catch of some species has apparently equaled or exceeded the appropriate species quota. If a country or fishery has its quota taken inadvertently by another fishery, the situation is clearly unsatisfactory.

There are innumerable possible analytic solutions to dealing with the allocation of catches when by-catch is a problem; all require an inordinate amount of data and analysis. It is far simpler, more understandable, and probably more equitable to limit the overall catch and mandate cessation of fishing when the limit is reached, regardless of the species mix. This approach also encourages the cleanest possible fisheries. Individual species groups, or species, or stock quotas also need to be maintained to prevent overexploitation of particular groups (*see* Brown et al. 1973).

The second general rationale for the ecosystem quota—that it offers the best approach to date for maintaining the ecosystem in its most productive and useful state to man—is far less easy to demonstrate.

The late William Terry, when asked his basic philosophy about his part in fishery negotiations, stated that he "above all else should work toward maintaining all possible options for future generations." From our point of view, the overall quota or ecosystem approach would have pleased Bill Terry. The by-catch problem may be reason enough to go that way, but there is more to it than that.

I want to point out some obvious differences between land and ocean ecosystems and to suggest that earlier approaches to marine fisheries management may have been seriously inadequate. And I hope to convince you that a total ecosystem approach is the best one to date if we wish to maintain our options.

From man's point of view and from the standpoint of his technological capabilities, fishes in general and certain large invertebrates—lobsters, squid, and clams, for example—are the most useful form of protein provided by continental shelf ecosystems. There is neither the desire nor the capability, for example, to harvest amphipods, copepods, or euphausiids, except possibly in Antarctic waters. Certainly the standing crop of all such creatures exceeds that of the fishes. The decision to harvest these components of the ecosystem should only follow a deliberate decision to substantially reduce or eliminate fishes as a protein resource, since the invertebrates are directly or indirectly the fishes' food supply.

Ecologists generally believe that ecosystems "in balance" are more productive or efficient than those significantly disrupted or distorted by whatever mechanism. Dice (1952, p. 357) stated: "It may be assumed that the most efficient utilization of the resources of any given habitat is obtained when the community occupying the habitat is well organized and regulated so that no very violent fluctuations in the populations of the member species are produced, even by considerable variations in the weather or in the physical conditions of the habitat." The productivity of any ecosystem is determined by the particular constituents of that community and the rate at which they accrue, fix, or otherwise deal with the available energy. The term "biomass" has been used in this instance because the management measures developed came directly from analyses of biomass changes related to fishing effort.

All ecosystems change continuously. They tend toward a central theme, not point, of equilibrium—one most efficient in utilization of available energy. All ecosystems react and adjust, often very rapidly, to changes, including birth and death, growth, migrations, changes in physical conditions, and the introduction of new organisms, either de novo or from outside.

Marine "fish" ecosystems exist for long times in a recognizable state of dynamic equilibrium. Even though the species may vary among the North Sea, the Northwest Atlantic off New England and Nova Scotia, the Northwest Pacific, and even on the continental shelves of South Africa and South America, they have a marked ecological similarity.

We are far from being able to say that marine ecosystems are as stable as terrestrial ones; stability is used here in the sense of "robustness" or "resilience" to substantial, long term, natural change. If someone wants to argue that the fluctuations observed in Arctic mammal populations indicate a-less-than-robust ecosystem, I will agree that perhaps the marine fish ecosystems are not very stable. In any event there is still no good basis for quantifying "stability."

Land and marine ecosystems are very different. The principles are the same, but the scales of events are not. Terrestrial primary production is accomplished by macroscopic plants. Large herbivores can get one or more sizable mouthfuls without moving far. In the ocean primary production is accomplished by tiny plants, most of them not even visible to the unaided eye. Thus, the typical ocean herbivore is also very tiny. There are exceptions, of course, like some pelecypods and some fish. The typical primary carnivore in the ocean is also small, for example, chaetognaths and ctenophores. Those organisms that man finds useful are consequently hyper-carnivores, generally secondary carnivores.

Terrestrial ecosystems are relatively stable in their geographic boundaries and in their persistence. The beech-maple forest, once established, does not move much, and by its very existence creates microclimates allowing considerable specialization of animals and plants living within that forest. In the ocean, boundaries and distribution of ecosystems change constantly. Oceanic plants do not of themselves create opportunities for specialization based on microclimates. Feeding niches tend to be broad and seem to be based on size of prey as much as on kind of prey (Figs. 2 and 3).

Most terrestrial animals cease growing when they mature. Most also feed on the same food while growing as they do when mature. Fish, molluscs, and many other marine organisms continue to grow until they die. Some animals, particularly fish,

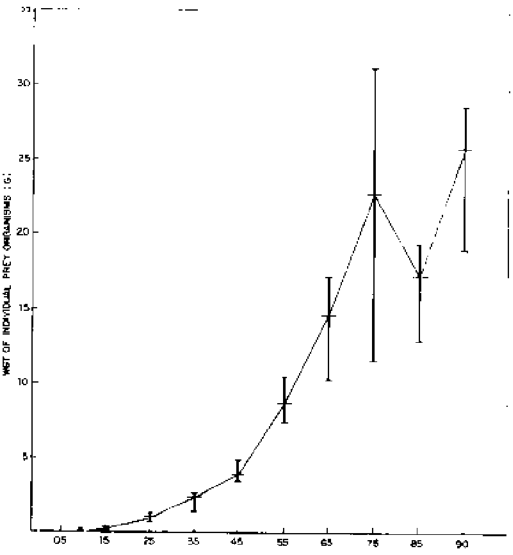

Fig. 2. Weight of prey organisms in food of cod. Based on data in Daan (1973). Full range of weights not shown as only averages for various groupings were provided, but approximate overall mean indicated by connecting line.

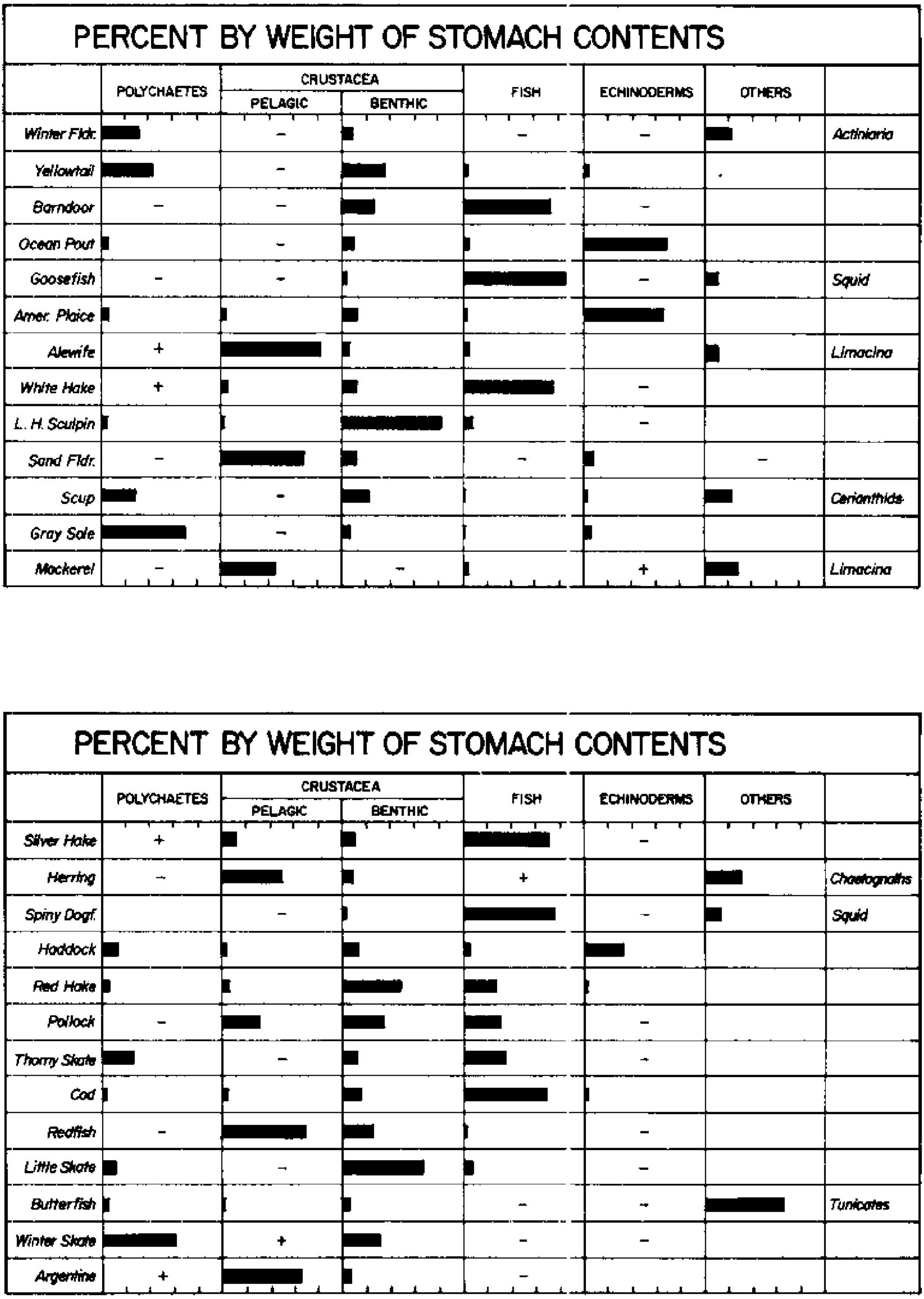

Fig. 3. The percent by weight of stomach contents of the commoner species of North Atlantic fish. Based on data from Maurer and Bowman (1975).

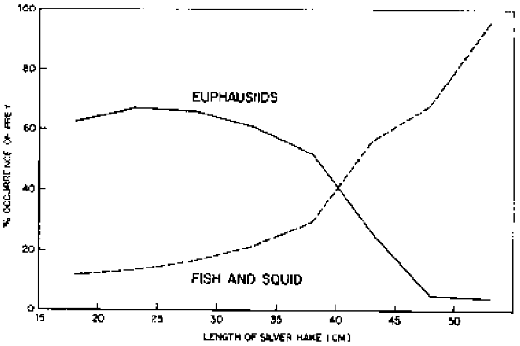

Fig. 4. The occurrence of prey in the stomachs of silver hake as it relates to fish size. (Based on data from Vinogradov 1972.)

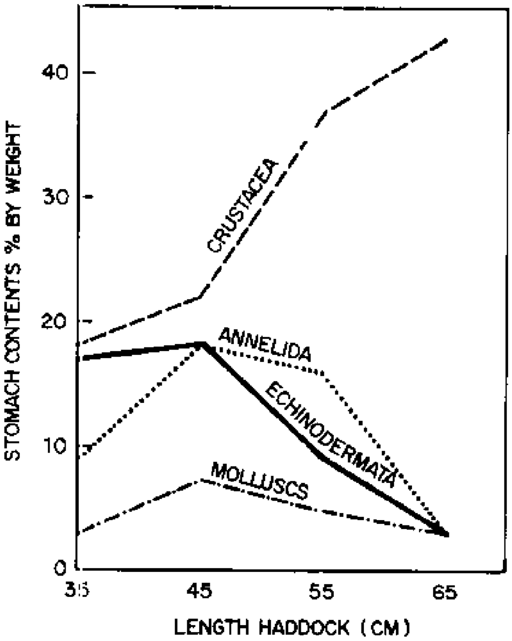

Fig. 5. Stomach contents of haddock. (Data from Wigley and Theroux 1965.)

feed on different groups of prey organisms as they grow (Figs. 4, 5, and 6).

Finally, fish and most other marine animals are *r* strategists in the extreme (cf. Miller and Botkin 1974). One would intuitively assume that this must be the case because so many of the biotic stabilizing forces of terrestrial ecosystems are lacking in the oceans. The adult fish has relatively little impact on the environment—both biotic and abiotic—that the very young fish

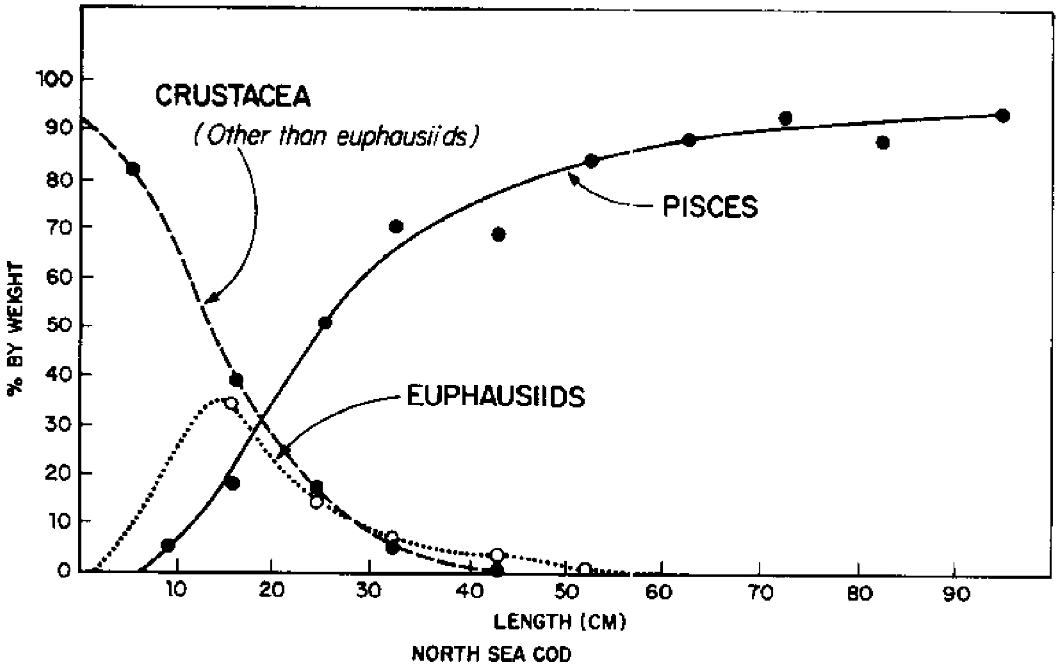

Fig. 6. Stomach contents of North Sea cod. (Based on Daan 1973.)

live in. Large numbers of eggs are produced with relatively little detectable preliminary adjustment to the conditions that the young will encounter. They are opportunists par excellence.

To put it differently, as a fish grows it tends to feed on larger and larger organisms, more or less proportional to its size; consequently it continually changes its feeding niche. This is almost as though the growing fish was evolving from one species into another and yet another. Adults are far removed from the niche occupied by the young, particularly in terms of food but often physically as well.

These thoughts can be expressed in terms of the familiar logistic growth equation

$$\frac{dN}{dt} = rN\left(\frac{K-N}{K}\right).$$

I have attempted to describe the life history of a fish by modifying this well known equation (Fig. 7). In all cases the appropriate coefficients are more directly related to weight than to number. N_0, the number of eggs or newly hatched larvae, is proportional to the weight of spawning adults. Since the carrying capacity, represented by K, is seemingly dependent on the available food of the right size, it, too, would tend to be proportional to the available weight of food, as would the various Ns.

All of these phases are not necessarily synonymous in time. Nor is time fixed in the sense that one can assign dates to the beginning and ending of phases. Time is flexible in that a fish changes its growth rate in response to changes in K. For example, we know that under crowded conditions fishes often grow very slowly. In other words, they have the option, to some degree, of growing slowly rather than dying. They apparently can also grow very rapidly under favorable circumstances with the ultimate penalty of dying a little sooner.

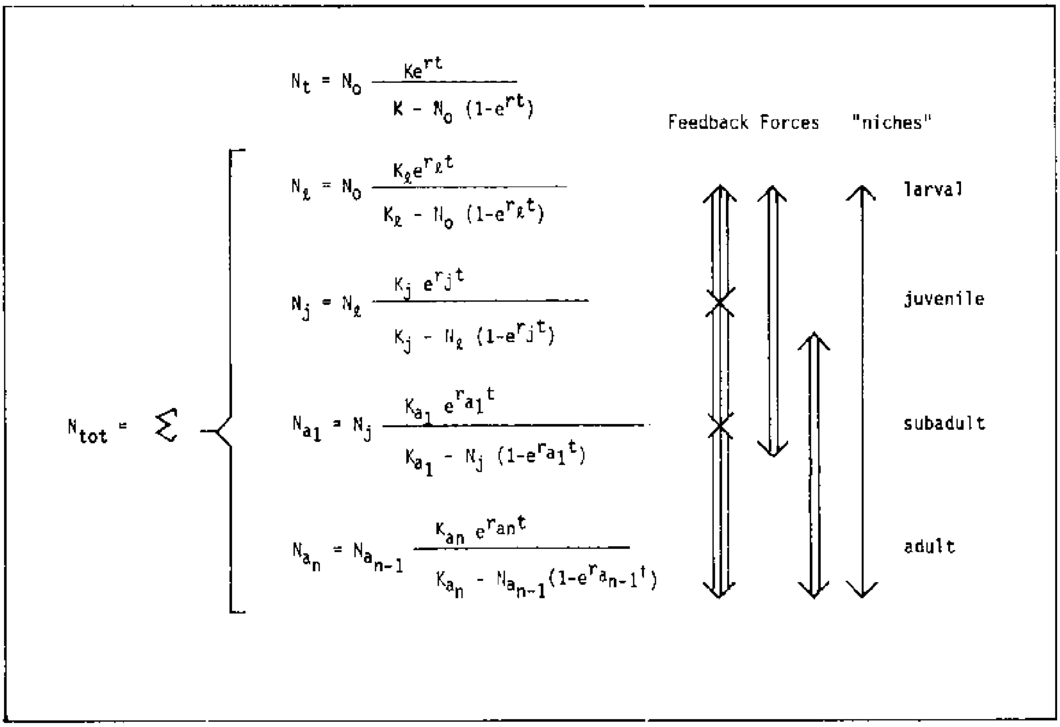

Fig. 7. The phases of a fish's life history expressed in terms of the logistic population growth equation, $\frac{dN}{dt} = rN\left(\frac{K-N}{K}\right)$.

This point was discussed recently by Kitchell et al. (1974).

The arrows in Fig. 7 indicate that the density-dependent feedback loops under these circumstances become less and less powerful as the fish climb the "niche ladder."

Since fish are flexible in their feeding habits—cod will turn to flounders when other fishes of the right size are not available—we might even use these equations as the basis for a model for the fish ecosystem by ignoring species differences and simply looking at obvious feeding niches such as the "pelagic feeding niche" etc.

It is a severe enough distortion when fisheries take all species more or less in proportion to their abundances, but other distortions that inevitably follow fishing could have serious, long term repercussions. There is growing evidence that spawning time in some species varies significantly with age of the fish. This strategy seems to increase the probability that at least some eggs will find themselves in optimal circumstances. Exploitation reduces the mean age of fish populations and could consequently further decrease the probability of successful spawning by those species.

Man's present fishing activities are comparable to adding a very large population of so-called apex predators, such as sharks and some whales and whale relatives. These are not very abundant now nor do they appear to have been so in the past. One is compelled to ask why. The obvious answer seems to be that there is no ecological basis for their continued, successful presence in the community at such population levels. This certainly raises the question, then, of the degree to which man can continue to harvest without seriously upsetting the ecosystem and reducing its efficiency, at least in his terms. Given all this, maximum sustainable yield seems a poor management tactic at best. One last point concerning what I regard as the "counterintuitive" consequences of part approaches to fisheries management: Much that has been done has not led to resolution of the problems. I am not saying that the scientist is the principal villain, although he surely has contributed. Symptomatic of this state of affairs is the recent intensive re-examination of maximum sustainable yield (Edwards and Hennemuth 1975), the manner in which it has been used, and the consequences of its use to achieve a rational basis for fisheries management. Rather than put the argument in my own words, I quote Kenneth Watt (1971, p. 71):

> In this paper the term "complex system" will be used to mean any process which is sufficiently complex that twenty or more state variables and a large number of feedback loops must be used in a computer simulation model in order to realistically mimic the process. Adequate research has now been done on such systems using computers to discover that they have one tremendously important feature in common. The word used by Forrester (1969) to describe this characteristic is "counterintuitive.' By this he means that complex systems give indications that suggest corrective action which will often be ineffective, or even produce the opposite effect to that desired. The reason for this counterintuitive feature is that each of us has learned through intuition an approach to life's problems which is almost invariably based on experience with simple systems; our experience with complex systems is typically too limited for us to have detected patterns of cause and effect. Simple systems have small numbers of simple feedback loops, as when we adjust the hot and cold faucets so that the bathwater will have the correct temperature. Our approach to complex systems is typically based on extrapolation from the simple systems with which we are experienced. Unfortunately, the large number of state variables found in complex systems, and the number and complexity of their feedback loops, gives them very different dynamic properties than those found in simple systems. For example in complex systems, cause and effect are often not closely related in time and place, but the cause of an effect may have occurred at a different place and far back in time. The optimal control strategy may not be a certain value of a particular variable, but rather an appropriate range of values for each of a large number of variables, applied in a particular temporal sequence. The right strategy for one time may be a terribly bad strategy for a slightly earlier or later time, and the appropriate variate value for one control variable may be totally dependent on the variate values taken by other variables in the system at the time of control.

I submit that marine fish ecosystems qualify as "complex" systems.

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Some results of fish surveys in the mid-Atlantic important for assessing environmental impacts

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Abstract

Knowledge of the distribution, biomass, and composition of the major biological communities is essential to the data base required for assessing the impact of environmental change on the marine ecosystem. Since 1967 the National Marine Fisheries Service has conducted regular trawl surveys in the mid-Atlantic, providing quantitative measures of density distributions of demersal fish populations. The surveys show that each species occupies wide areas of the shelf and that there is a high degree of species mixture, particularly in the New York Bight. Significant numbers of adult stages, planktonic eggs, and larvae can be found over the whole mid-Atlantic shelf all year. Thus there are no shelf areas free of risk from potential damage from waste disposal activities. Impacts of any change must consider a multispecies community and include all life stages.

Precision of large-scale surveys is relatively low, making it difficult to detect any but major changes. Broad-scale surveys must be augmented with detailed laboratory and field experiments on physiology and behavior (especially food chains) of selected organisms and communities to get insight into probable effects of sublethal factors. The complexity and scope of marine ecosystems require long term but well coordinated research programs to ensure proper integration of small- vs. large-scale studies and field vs. laboratory experiments.

Problems of assessing impact of man's activities on the marine environment are formidable because we lack adequate understanding of the dynamics of key elements, such as ocean circulation and the biotic and other abiotic processes controlling organic production. This is especially critical in coastal and shelf waters where biological productivity is high, circulation is poorly understood, and man's heaviest impact takes place. In addition a major logistic deterrent to acquiring quantitative knowledge for predictive modeling is the enormous range in the scales of time and space over which events occur. Understanding the life history of a single fish species usually requires years of sampling—in an area larger than the mid-Atlantic shelf from Cape Hatteras to Cape Cod, from estuarine to shelf-edge waters, and for planktonic eggs and larvae as well as demersal, juvenile, and adult stages—just to document density distribution. At the other end of the scale, experimental studies are needed on individuals of the species and on associated organisms to document physiological and behavioral responses over a very brief time—weeks, hours, even minutes—critical to estimating rates of feeding, digestion, growth, and reproduction. Ultimately, the ability to assess and predict long term changes in fish abundance, from both natural and man-induced causes, will require a quantitative understanding of the mechanisms controlling production in all major biological components of the marine ecosystem, including primary and secondary producers: in short, a predictive model of the whole system.

Essential to the necessary data base is a time series of observations on the distribution, biomass, and composition of the various biological communities. The National Marine Fisheries Service (NMFS) is developing a survey program called MARMAP (Marine Resources Monitoring Assessment and Prediction) to provide such a time series. On the east coast, initial emphasis has been on trawl surveys to provide annual inventories of the abundance, distribution, and structure of demersal fish populations to help assess the recent impact of tremendous increases in foreign fishing pressure. A long term objective is to acquire information on natural factors and processes critical to fish production in general. Recently, plankton sampling has been added to document zooplankton distributions.

This paper briefly reviews some results of MARMAP surveys relevant to detecting or predicting the impact of ocean pollution on marine fishes. Key aspects include geographic and seasonal distribution of fishes in the mid-Atlantic, including egg, larval, and adult stages, and the levels of accuracy attainable for population abundance estimates. Also included are some observations on feeding interactions involving fishes and some thoughts on the scope and nature of research needed to provide predictive capability.

I wish to thank W. Davis and T. Azarovitz for the use of data from their larval fish surveys and inshore trawl surveys. Also I acknowledge the assistance of R. Livingstone, Jr., for information on spawning of fishes in the New York Bight area.

NMFS trawl surveys in the mid-Atlantic

Regular trawl surveys by the Northeast Fisheries Center (NEFC) at Woods Hole, Massachusetts, have been conducted in the New York Bight since 1963 and in the mid-Atlantic from Cape Hatteras to Cape Cod since 1967. The surveys are done in autumn and spring with a standard bottom trawl and provide a valuable quantitative time series on the distribution, abundance, and population structure of over 100 species of marine fishes. A stratified random sampling design is used to facilitate construction of statistically unbiased abundance indices but still provide a fairly uniform distribution of stations throughout the geographic area for monitoring distribution (Grosslein 1969). Trawling was restricted to the area north of Cape Hatteras and to depths of 15–200 fathoms (about 27–365 m) until 1972 when the Middle Atlantic Coastal Fisheries Center (MACFC) at Sandy Hook, New Jersey, expanded the survey to inshore waters (from just off the beach out to the 15-fathom-stratum boundary) and southward to Florida. At present, south of Cape Hatteras is surveyed by the South Carolina Marine Resources Institute, from Cape Hatteras to Cape Cod by MACFC, and to the north and east of Cape Cod (as far as Halifax, Nova Scotia) by NEFC. In all these surveys, total weight and length frequency are recorded for all fish (and most shellfish) species caught in the trawls. Other biological data collected include samples providing information on age and growth, maturity, fecundity, spawning, and food habits of juvenile and adult stages. Sampling of zooplankton began in 1968 in a piggyback mode on the trawl surveys, with emphasis on fish eggs and larvae.

This paper discusses fish distribution in the area north of Cape Hatteras, which encompasses the range of most important commercial and recreational species of the mid-Atlantic. Data from the surveys south of Cape Hatteras have not been included.

Geographic and seasonal distribution of fishes in mid-Atlantic

Most fish species in the mid-Atlantic cover the whole region; many extend north of Cape Cod or south of Cape Hatteras. To illustrate average density distributions, plots of individual trawl catches for selected species are presented for the 1973 and 1974 surveys combined in Fig. 1. Only positive catches are plotted for each species, and the reader must keep in mind that the whole area was covered as shown in Fig. 1. In addition, direct comparisons of relative abundance between seasons cannot be made because availability of some species varies widely with the season and because a smaller trawl was used for autumn surveys (Yankee No. 36) than for spring surveys (Yankee No. 41). The main purpose of the figures is to illustrate density distribution of populations within seasons.

Red and silver hake, the two most abundant hake species, extend from Cape Hatteras to Nova Scotia, but with major population density in the New York Bight area (Fig. 2). Yellowtail flounder, one of the most important commercial species to U.S. fishermen, is found principally in the New York Bight and on Georges Bank where it is frequently caught with hake and other

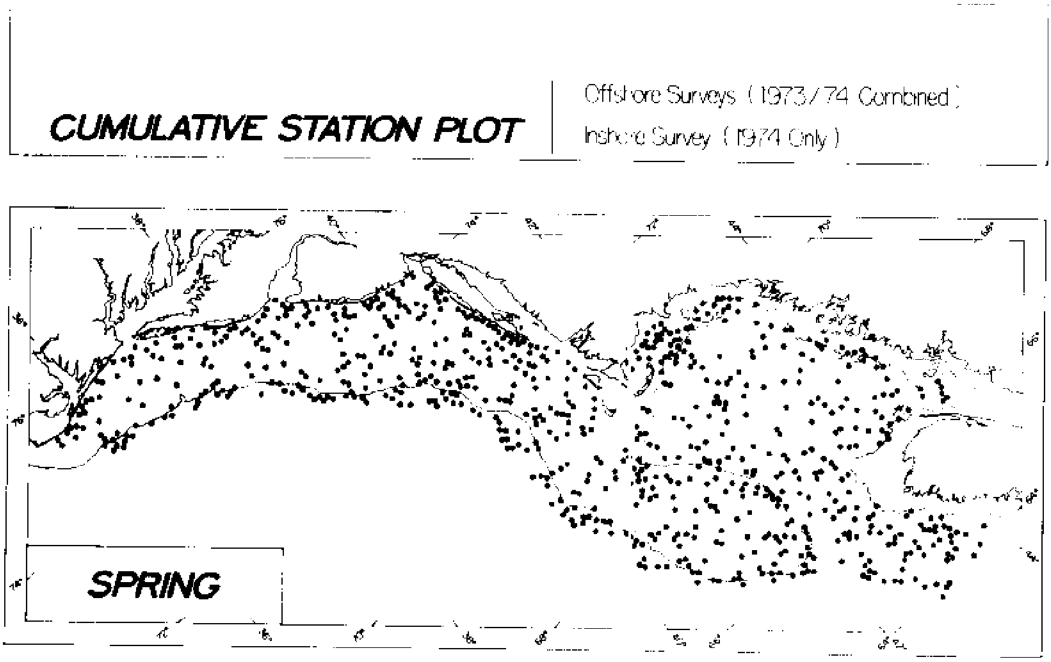


Fig. 1. Trawl stations occupied in spring 1973 and 1974 offshore surveys and spring 1974 inshore survey.

species in bottom trawl catches (Fig. 3). Little skates, also significant in trawl catches in the bight, are distributed like yellowtail. Hake tend to move offshore during winter but major aggregations of skates and yellowtail (as well as most other flounders) tend to remain in the same areas throughout the year.

Other species show more pronounced seasonal migrations, both inshore–offshore and north–south. For example, longfin squid and butterfish, found widely dispersed over the shelf in the warm seasons, tend to aggregate along the shelf edge in winter, although some butterfish are still found in shallow water (Figs. 4 and 5). Other species exhibit similar inshore–offshore movements, such as scup, summer flounder (Fig. 6), and lobster. Sea herring (Fig. 7), cod, and mackerel occur in the mid-Atlantic chiefly in winter and spring but are east and north of Nantucket during summer and fall. Weakfish and bluefish (Fig. 8) shift southward toward Cape Hatteras and below during winter and move north into the mid-Atlantic in summer.

Mixture of species on the mid-Atlantic shelf

The density distribution of the approximately 30 species making up the bulk of trawl catches in the mid-Atlantic indicates a heavy mixture of species throughout the region with virtually no place on the shelf where significant aggregations of a number of important species are not found at some time of year. The New York Bight area has the maximum density and mixture of species important for both commercial and recreational fisheries, since it tends to be the northern limit of tropical and subtropical migrants and the southern limit of boreal migrants. Seventy-two species were recorded in three autumn surveys from 1963–1965 in sampling strata 1–12, i.e. Hudson Canyon to Nantucket (Lux and Grosslein 1970). The same region has traditionally been the area exploited by the U.S. industrial trawl fishery, which in 1958, for example took significant quantities of 20 species, including the three principal hake species, three species of skates, four species of flounders, alewife, sea herring, scup, butterfish, ocean pout, sculpins, sea robins, sea

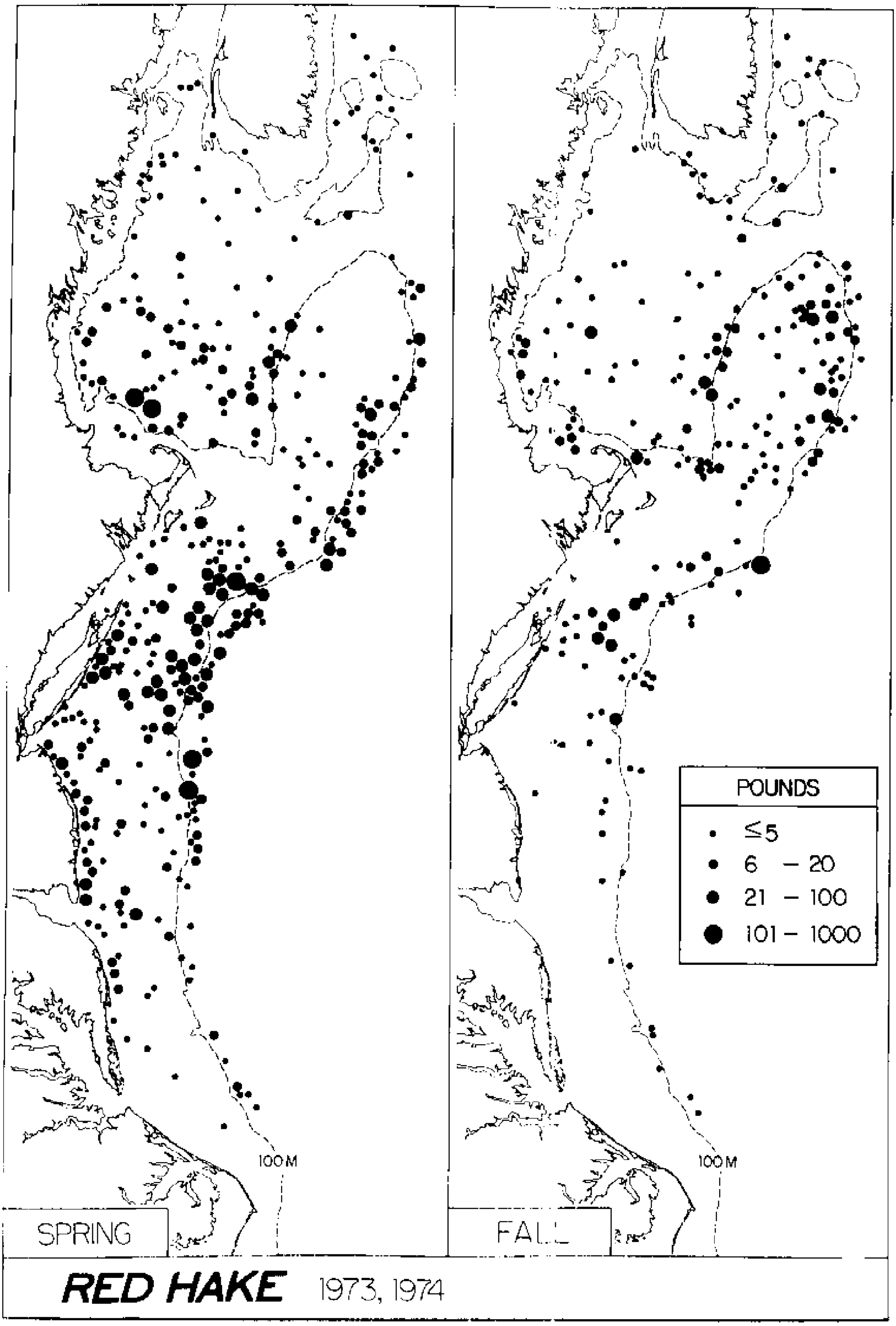


Fig. 2. Trawl catches of red hake in 1973–1974 MARMAP surveys.

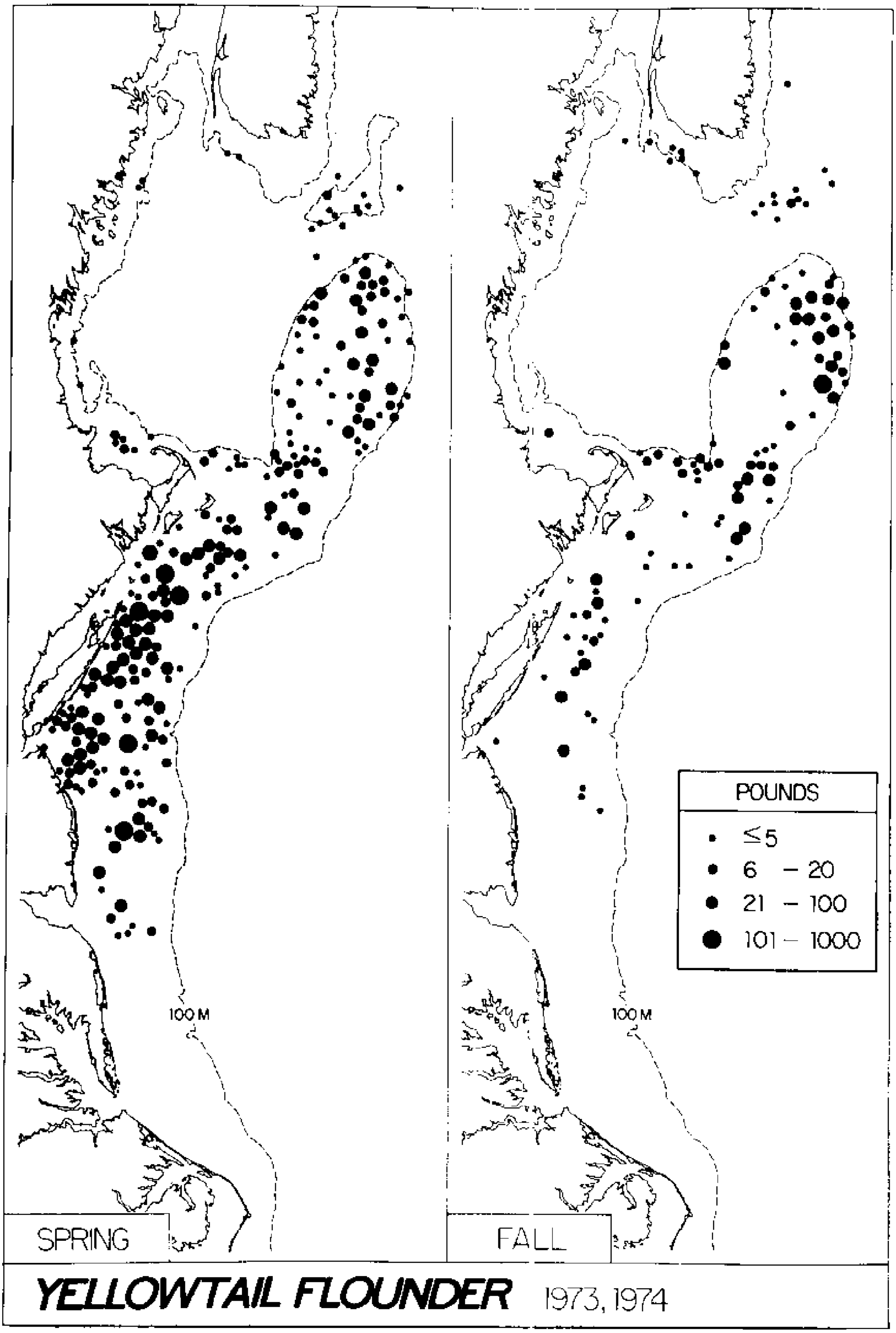


Fig. 3. Trawl catches of yellowtail flounder in 1973–1974 MARMAP surveys.

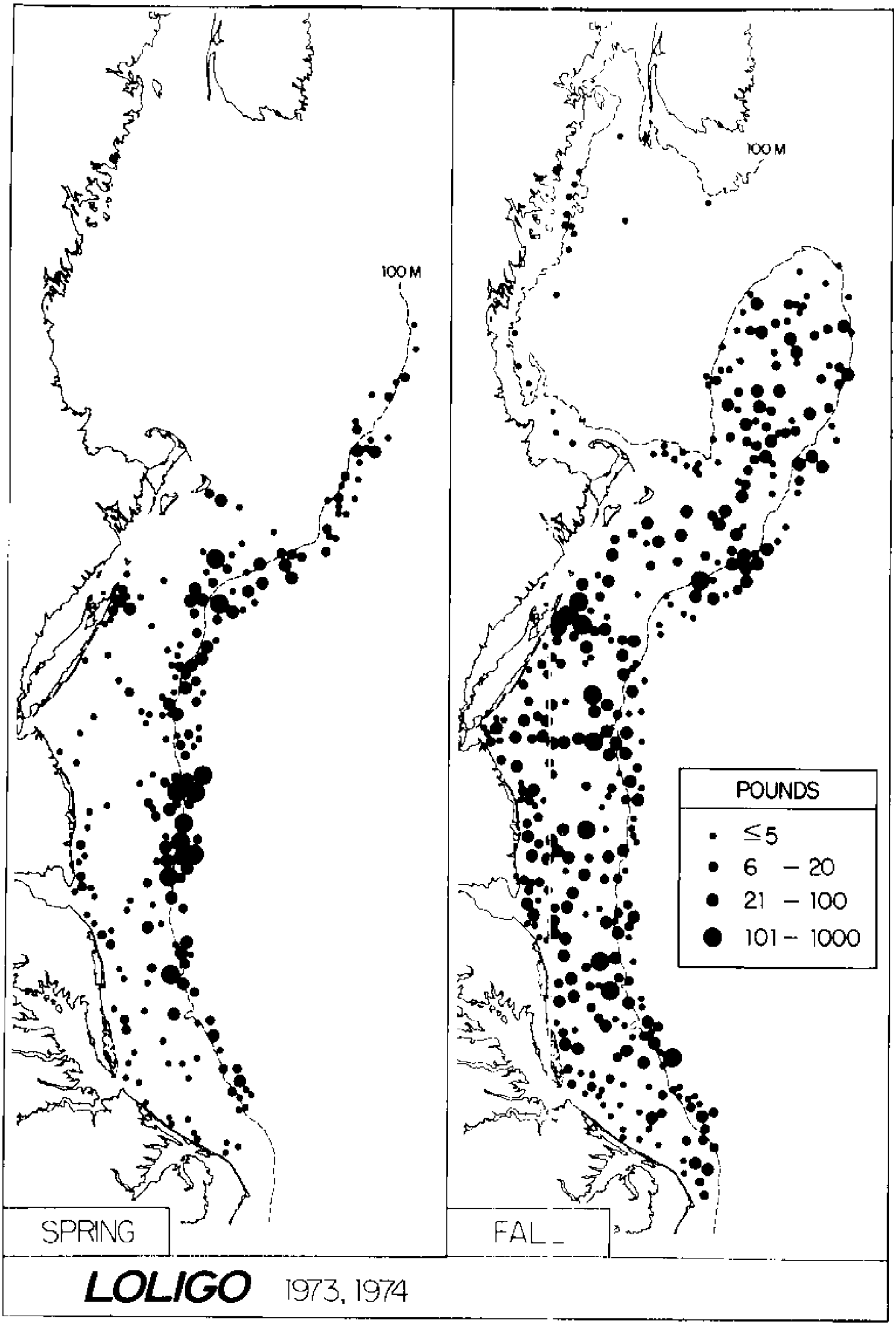


Fig. 4. Trawl catches of longfin squid (*Loligo*) in 1973–1974 MARMAP surveys.

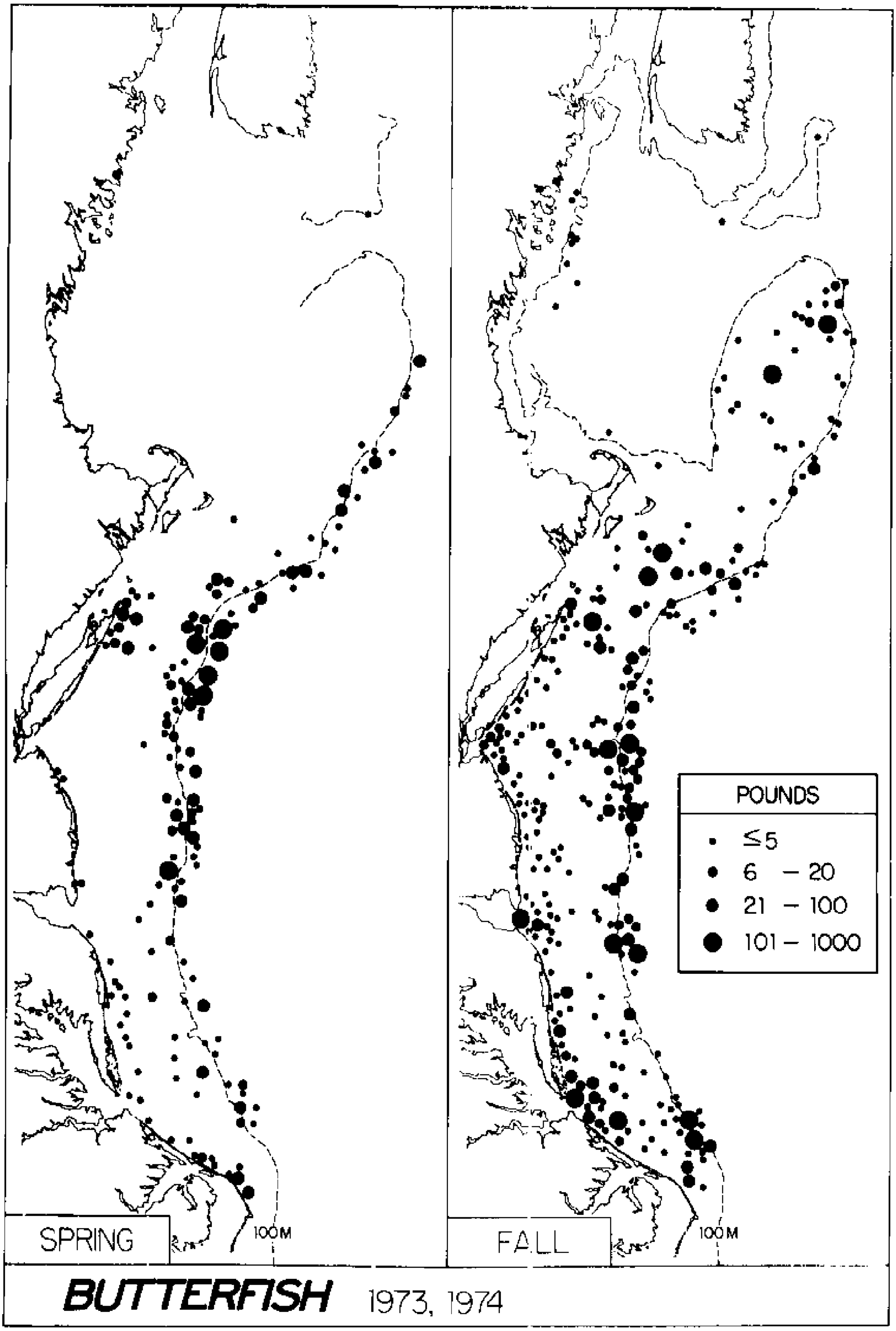


Fig. 5. Trawl catches of butterfish in 1973-1974 MARMAP surveys.

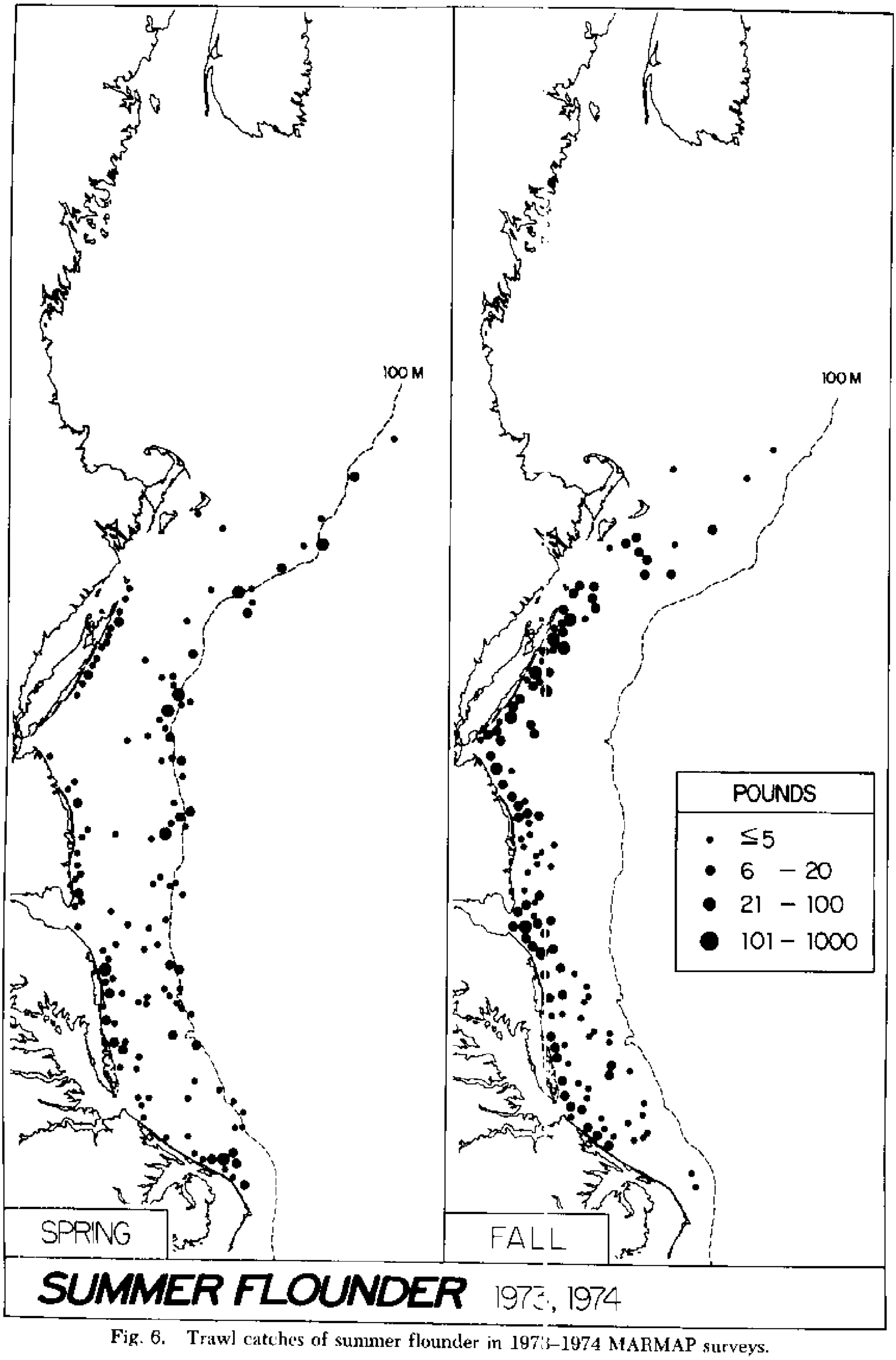


Fig. 6. Trawl catches of summer flounder in 1973–1974 MARMAP surveys.

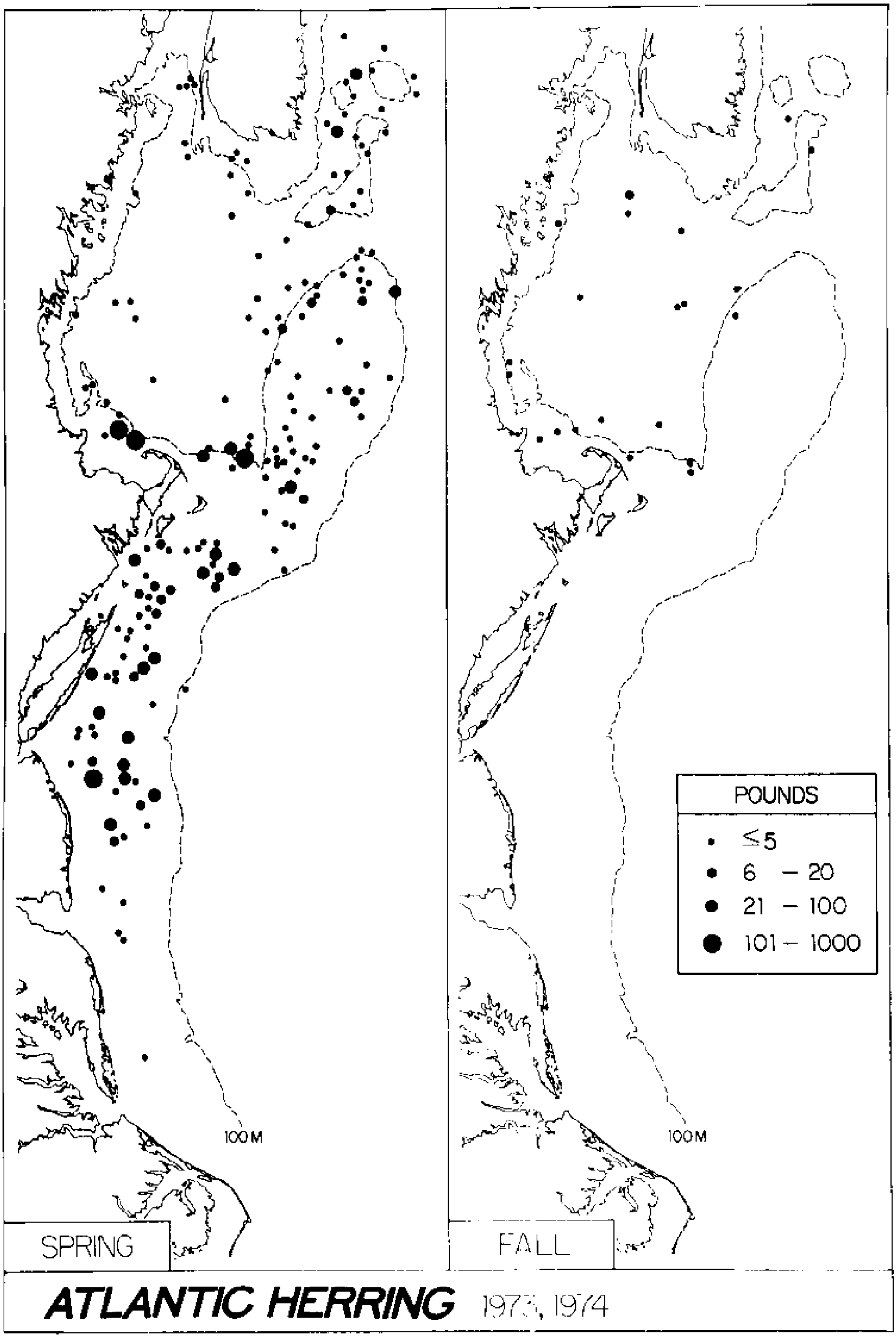


Fig. 7. Trawl catches of Atlantic sea herring in 1973–1974 MARMAP surveys.

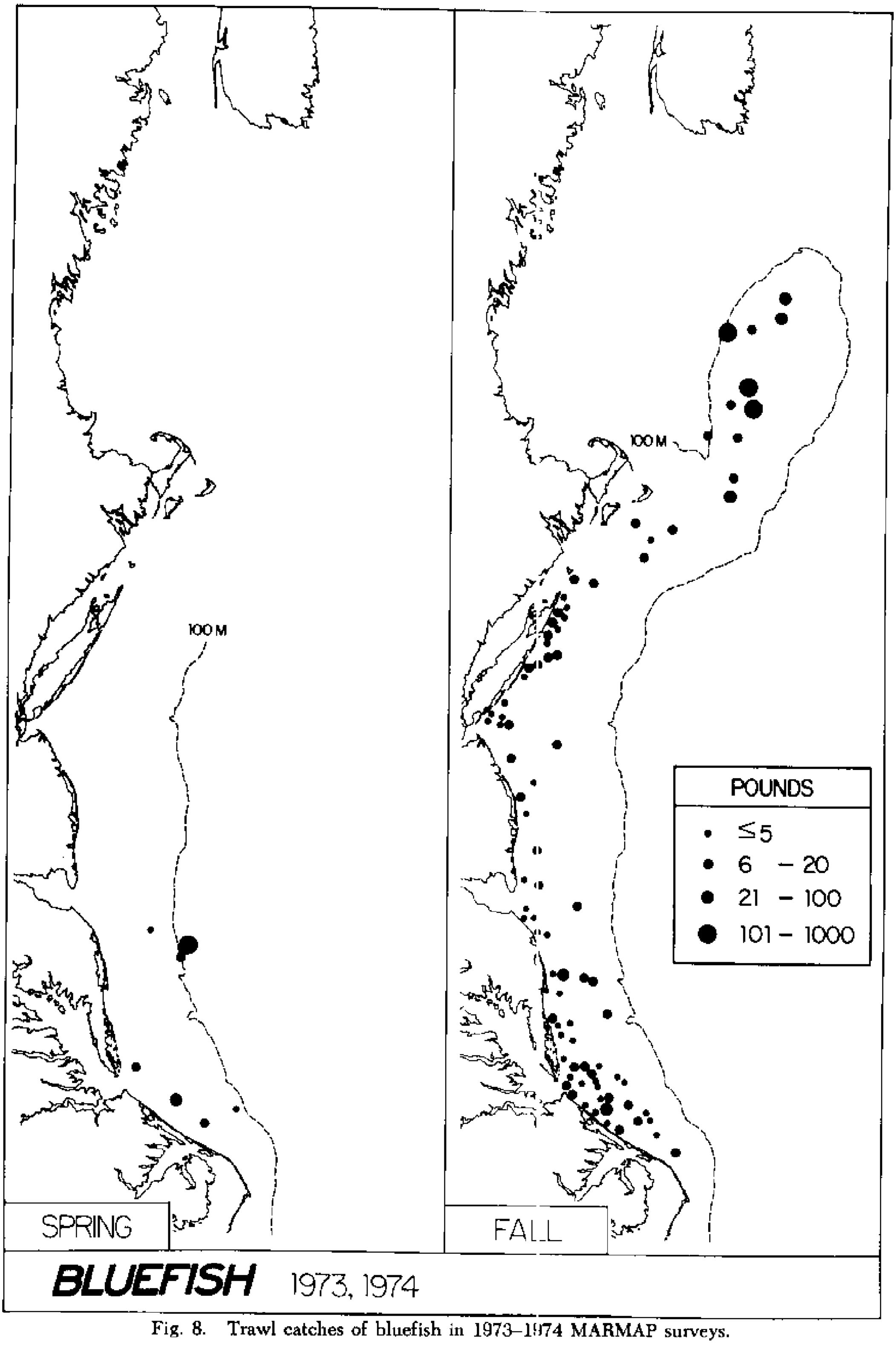


Fig. 8. Trawl catches of bluefish in 1973–1974 MARMAP surveys.

raven, and spiny dogfish; in the same area in 1971, NMFS research trawl hauls caught an average of 12 species per haul (Grosslein and Bowman 1973).

Some nonseasonal shifts in species distributions apparently have occurred since 1967, although a definitive analysis has not yet been completed. For example, before 1972 bluefish were extremely rare in survey bottom trawl catches east and north of Nantucket Shoals. Beginning in 1972 bluefish began appearing frequently in autumn trawl catches on Georges Bank and by 1974 they were caught in substantial numbers there (NMFS unpublished records). The recent "invasion" of the coast of Maine by bluefish is now well known, and there is no doubt they have been ranging farther north than usual. Part of these changes may be attributed simply to a large increase in population size, but part is also believed to reflect a real shift in the density center of the population, northward and offshore, perhaps related to a recent rise in Gulf of Maine temperatures (Davis 1976). A similar northward shift or expansion of green crab populations appears to have occurred. Whether these changes are major shifts in density centers, from time to time major shifts in population centers do in fact occur, they must be monitored if reliable measures of population abundance are to be obtained.

Distribution of spawning

Larvae are probably the life stage most vulnerable to environmental changes; hence they are likely to be most sensitive to pollution. It is especially important to document the geographic and seasonal distribution of eggs and larvae to improve chances of predicting effects of pollution on these life stages.

We are not as far along with larvae as with the distribution of adult stages. Plankton sampling has been done routinely on the trawl surveys but the data are not yet summarized. Also the once-a-season coverage of trawl surveys provides a spotty picture of the egg and larval distribution. Most species spawn over several months; thus a continuous series of synoptic cruises is needed to get a quantitative picture of egg and larval production and distribution.

We have one time series in the mid-Atlantic which provides a general multispecies picture of the occurrence of eggs and larvae throughout a whole year. The MACFC at Sandy Hook conducted 14 plankton transects on eight cruises from Block Island to just below Cape Hatteras, representing sampling in all months except March and July between December 1965 and December 1966 (Clark et al. 1969). From Cape Hatteras north, plots of the numbers of species of larvae per station indicate that from 1–10 species occurred at nearly every station in winter and spring 1966; during summer and autumn (June–September) at least 1–10 species occurred at every station and a significant number of stations yielded 11–20 (Fig. 9). Generally larger numbers of species were encountered at Cape Hatteras and just south, owing to the subtropical species there. Significant numbers of larvae were present throughout the year, with smallest numbers in winter, increasing in spring, and peaking in summer and autumn (Fig. 10). This pattern is consistent with estimated spawning times of major mid-Atlantic species (Table 1).

Detailed results of these cruises for individual species will soon be published by MACFC. We already know that egg and larval as well as adult stages of most species are widely distributed and mixed in the mid-Atlantic. Because of this large-scale distribution and overlap of species population (and life stages) it is virtually impossible to find a shelf location where there is no risk, or even a minimum risk, of environmental impact on fishes.

Problems of scale in environmental assessments

Measuring the impact of ocean pollution or predicting its potential effects on marine organisms will require studies ranging in scale from laboratory experiments on isolated single organisms to broad-scale surveys of the abundance and density distribution of populations over the whole continental shelf region and over years.

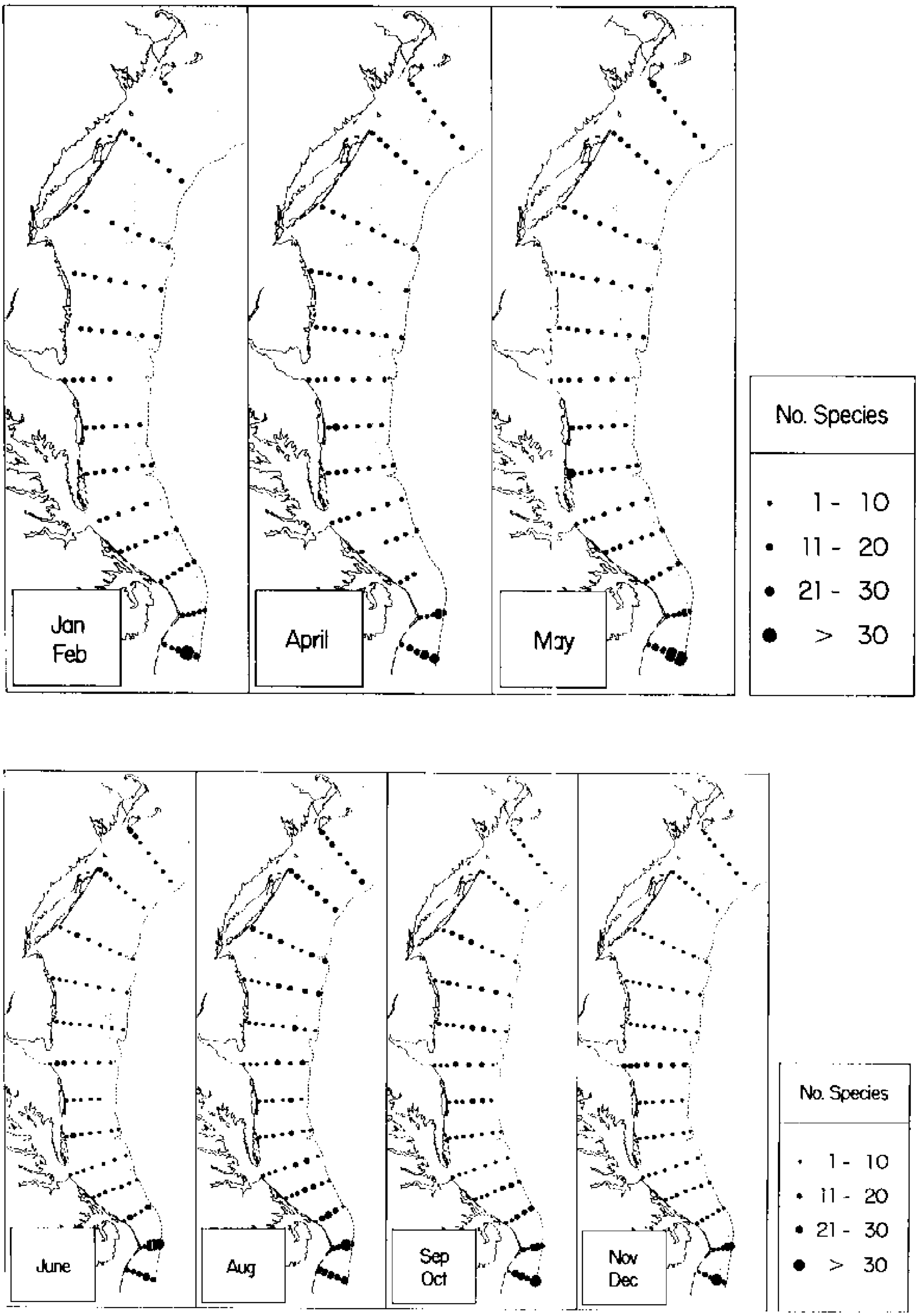


Fig. 9. Numbers of species of fish larvae per plankton haul on RV *Dolphin* cruises 1965–1966.

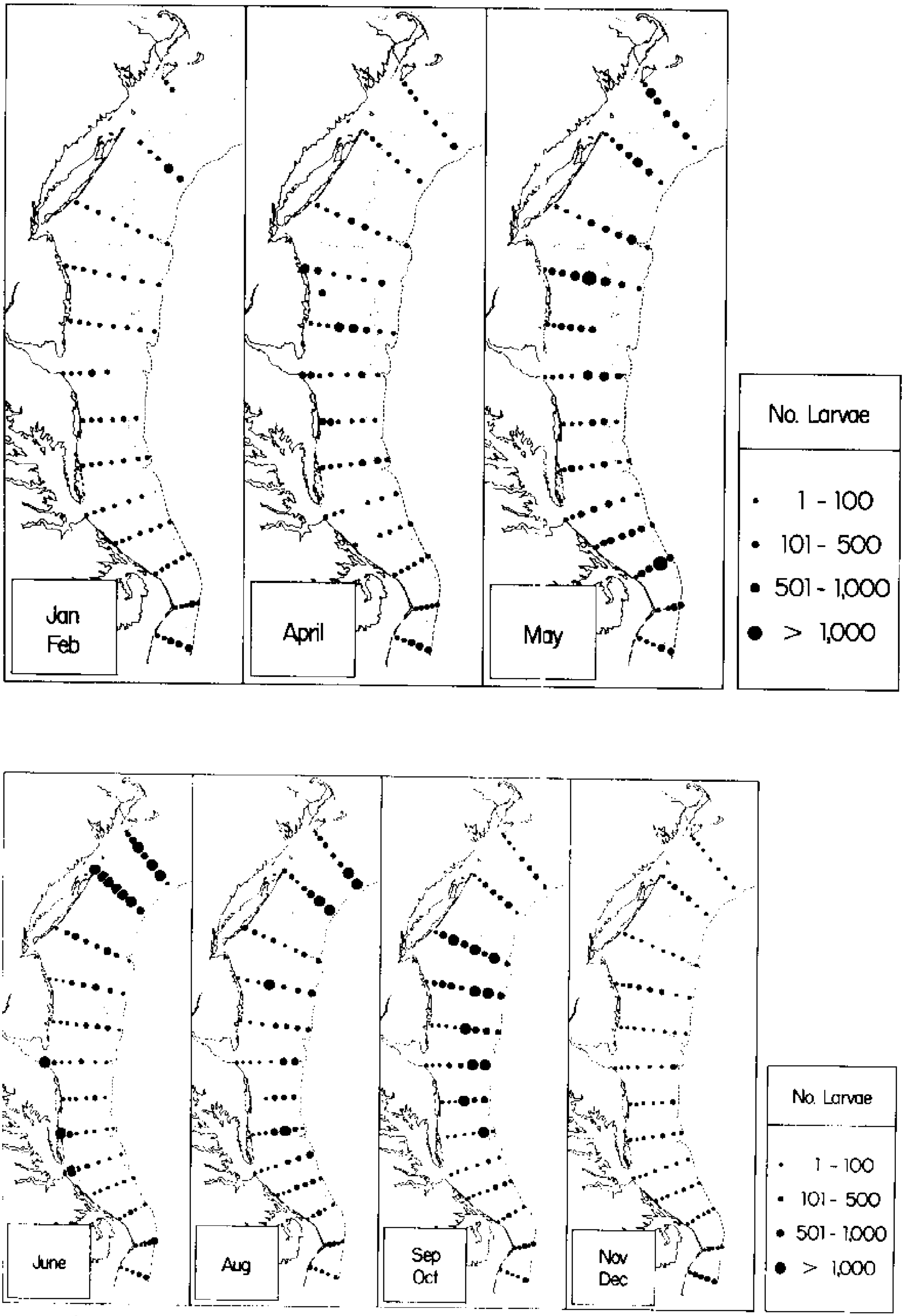
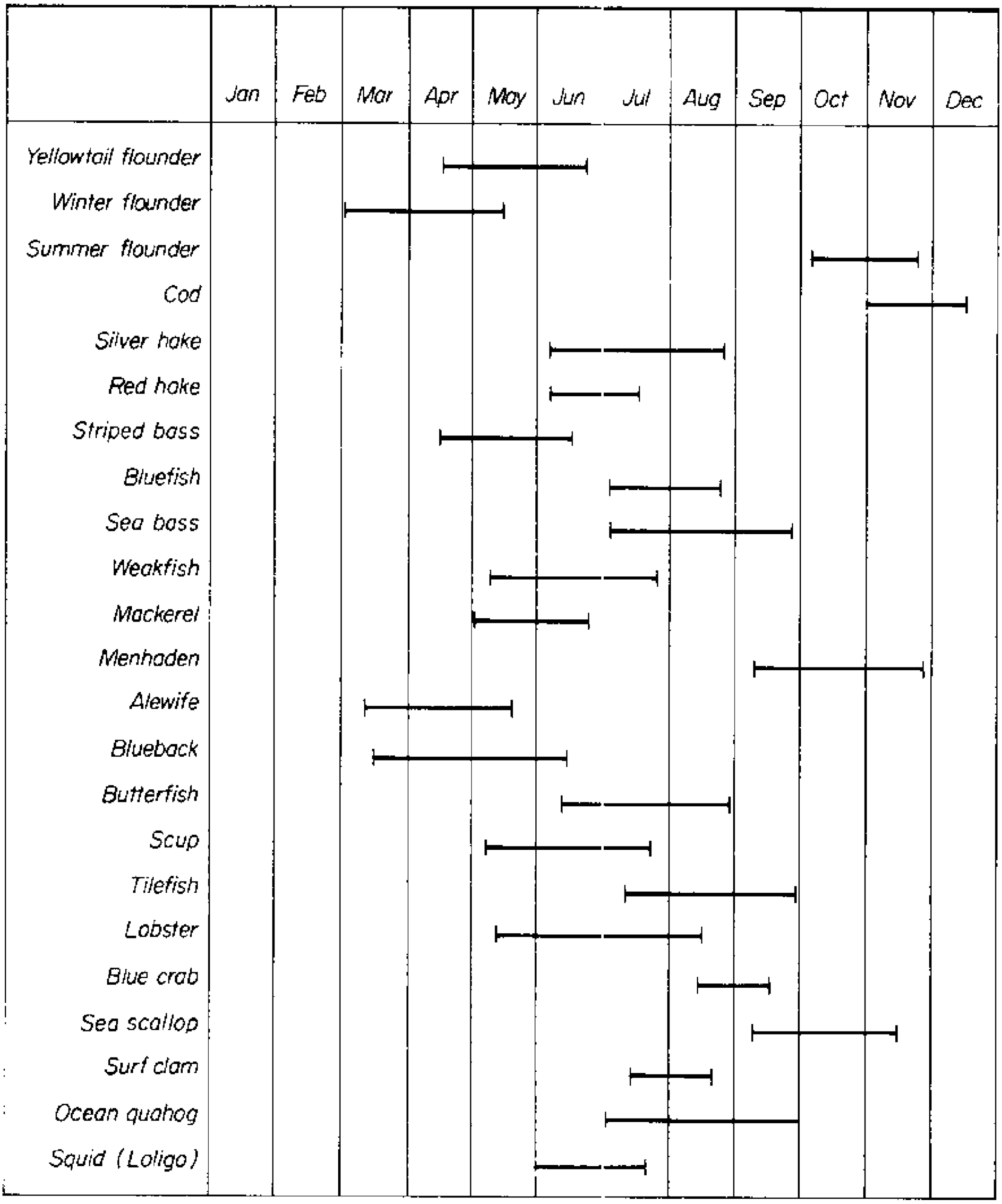


Fig. 10. Numbers of fish larvae per plankton haul on RV *Dolphin* cruises 1965–1966.

Table 1. Approximate spawning season of major commercial and recreational species in the New York Bight area.



Identifying modes of action and predicting levels of impact of specific pollutants on an individual organism will have to come primarily from laboratory studies and intensive localized experiments in the ocean involving contaminated vs. uncontaminated sites. However, inferences about impacts on the population and ecosystem will require knowledge of the abundance and density distribution of whole populations, communities, and trophic levels. Only large-scale synoptic surveys covering entire population ranges can provide such information. Unfortunately, the levels of precision attainable on large-scale surveys are low, the logistic requirements and costs very high. For these reasons it is not possible to detect other than major fluctuations or trends with such surveys. Yet it is essential to maintain large-scale monitoring of the status of the ecosystem as an integral part of the data base needed to separate natural from man-induced changes. Knowledge of the degrees of accuracy feasible is important in the development of effective and realistic research strategies. Some results on accuracy of NMFS trawl surveys and comments on the relative precision of plankton surveys are presented here.

Analysis of sampling errors in the NMFS trawl surveys shows that a relative biomass index for a single survey over an area the size of the mid-Atlantic has a 95% confidence limit of about ±50% of the mean (Grosslein 1971). This level of variability represents random sampling errors derived from an average sampling intensity of 80–100 standard hauls or about one haul per 300 mi^2 (about half the density of stations illustrated in Fig. 1). It does not include error introduced through year-to-year changes in availability coefficients due to environmental factors or cruise-to-cruise changes in efficiency of even a standard trawl.

Despite these problems, this level of precision is adequate to assess annual fluctuations exceeding a factor of two. It is also adequate for measuring trends such as the declines in fish biomass observed in the mid-Atlantic since 1967 caused by heavy fishing and on the order of 50% or more (Brown et al. 1975; Clark and Brown 1975). The decline in raw (untransformed) mean catch per haul for major species groups in the mid-Atlantic is illustrated for autumn surveys in Fig. 11; the bulk of the decline appears due to reduced abundance of sea robins and flounders; the majority of other species showed individual declines. The 1972 data are believed to have a large positive bias because of an unexplained increase in fishing power of the standard Yankee No. 36 trawl. Approximate adjustments for this bias show a more pronounced decline. Even smaller changes than these probably can be detected with a time series of annual surveys, given ancillary data on age-length compositions and accurate fishery statistics, but such additional data are available only for a few commercial species. For the majority of fish species, abundance changes substantially smaller than these would be difficult to measure reliably even with twice the intensity of sampling (Grosslein 1971). A point of diminishing returns is reached rapidly in improved precision per unit increase in sampling intensity because the fish are contagiously distributed.

To achieve the same levels of accuracy for plankton populations generally requires much more frequent sampling than for adult fishes because specific components of plankton populations are short-lived, changing rapidly over weeks instead of slowly over months or even years like long-lived fishes. For example, to monitor the production of mackerel larvae over a single spawning season in the mid-Atlantic with anything approaching a precision of ±50% would probably require at least four cruises from April through June, with each cruise covering a significant part of the area from Cape Hatteras to Nantucket and with up to 90 stations per cruise. This represents nearly four times the sampling intensity of one bottom trawl survey in the same area, and as much sampling as one vessel could do within that time. Also it yields information only on those planktonic forms present in that year from April to June, whereas the single trawl survey can provide a basis for estimating abundance of adult fishes for the next several years.

Thus, although the impact of a degraded environment is more likely to affect plankton communities first, it is extremely difficult to measure anything less than major changes in terms of abundance alone. A more meaningful index to environmental quality probably will require additional information on growth and condition of plankton organisms and general structure of plankton communities. But before we can develop such indices much intensive work

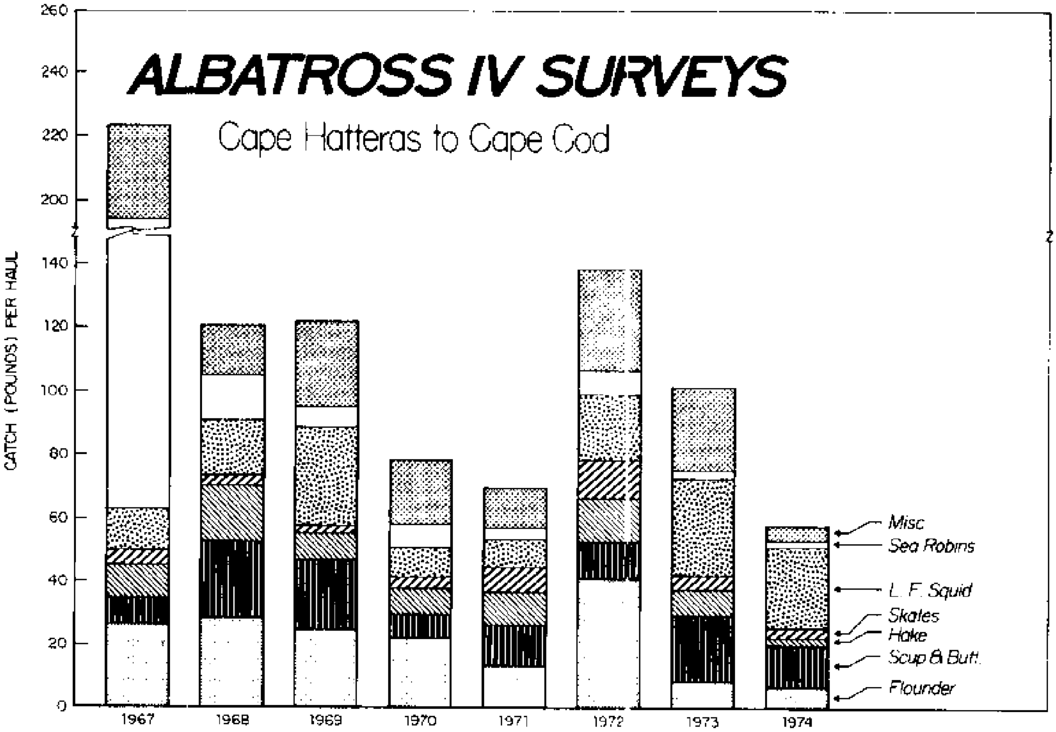


Fig. 11. Mean catch per standard 30-min haul of major species groups (spiny dogfish excluded) in autumn MARMAP trawl surveys from Cape Hatteras to Cape Cod, 1967–1974.

on plankton communities must be conducted along the lines of the MARMAP program. A key initial problem is to develop sufficient resources and coordination so that large-scale standard monitoring can be started to provide reasonably quantitative measures of the distribution, abundance, and structure of communities. Without this data base, it will be impossible in future years to judge whether there have been any significant changes in the system.

Feeding interactions

The most difficult problem will be to predict and assess the impact of environmental changes on food chains. An extensive inventory of the food habits of fishes has been completed by NEFC (Maurer and Bowman 1975). Stomach contents were analyzed from nearly 20,000 fish representing 80 species taken on the MARMAP trawl surveys from Cape Hatteras to Nova Scotia between 1969 and 1972. These data have not yet been analyzed in detail by season and separate regions, therefore only general observations are possible for 28 major species based on samples throughout the whole survey area.

Of these 28 species, most are significant components of the finfish biomass in the mid-Atlantic, including 4 species of hakes (red, silver, spotted, and white), 6 species of flounder, scup, butterfish, herring, mackerel, alewife, skates, ocean pout, spiny dogfish, bluefish, cod, and others. If we consider all 28 species as a group, fish represented 46% of the diet (with prey weight prorated to a ton of each predator) as compared with 54% for invertebrates, including 2% squid (Maurer 1975). Invertebrate prey was dominant in the diets of 25 of the 28 species, and the bulk of these were demersal crustaceans, molluscs, and polychaetes. Although fish accounted for nearly half the total diet weight for all predator species combined, less than a third of the species were significant fish predators with more than 10% of their diet composed of fish. The fish predators included some larger species, such as cod, pollock, silver hake, bluefish, spiny dogfish, and goosefish; the most common fish prey species were herring, mackerel, and silver hake.

A more quantitative analysis of prey-predator interactions by individual areas is in progress, including a larger set of species for each area and incorporating estimates of absolute biomass of each species. The present data are sufficient, however, to indicate the general complexity of the food chain involving primarily demersal fishes as predators. Better knowledge of these interactions will be required before we can extrapolate the ultimate consequences of an environmental pollutant. Even more critical will be the food chains involving plankton communities where the fate of eggs and larvae is decided. So far we have only rudimentary knowledge of these interactions.

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Fin rot disease studies in the New York Bight

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Abstract

Surveys from February 1974 through June 1975 showed the prevalence of fin rot disease in winter flounder (*Pseudopleuronectes americanus*) from the New York Bight apex was 3.9%, compared to 0.7% outside the apex. Prevalence of the disease in winter flounder from apex areas of low carbon deposits was 2.9%; whereas, in apex areas of high carbon deposits, disease prevalence was 5.1%. The prevalence of fin rot disease in summer flounder (*Paralichthys dentatus*) from the apex was 0.6% and in summer flounder from Sandy Hook-Raritan Bay was 3.4%. No summer flounder with fin rot were noted in Great Bay, a control area outside the apex. Although the numbers of diseased fish are small, incidence of fin rot disease in summer flounder from Sandy Hook-Raritan Bay increased monthly from June–November 1974. Attempts to induce fin rot disease in winter flounder in cages submerged in the sewage sludge area of the bight apex produced active fin lesions on the caudal fins more often than on the dorsal and anal fins. These flounder were in substantially worse condition than fish in cages at the control site.

Fish disease surveys were conducted in conjunction with temporal and spatial assessments of the abundance of demersal fish inhabiting the New York Bight. Of particular interest was the prevalence of fin rot disease in flatfish obtained from the sewage sludge disposal area of the bight (Fig. 1). Mahoney et al. (1973) initiated studies on the prevalence of fin rot, a disease characterized by progressive necrosis of fin tissue, in marine and estuarine demersal and pelagic fish species inhabiting the bight. From extensive trawl surveys, Ziskowski and Murchelano (1975) noted that winter flounder (*Pseudopleuronectes americanus*) with fin rot were found more frequently in an ecologically degraded area of the bight apex than in adjacent coastal areas.

From February 1974 through September 1975, prevalence of fin rot disease was determined in winter flounder and summer flounder (*Paralichthys dentatus*) from the bight. The prevalence of fin rot in winter flounder from outside the apex, from inside the apex, and from areas of low and high carbon-containing deposits in the apex (Fig. 2) was compared. The incidence of fin rot disease in summer flounder was determined approximately monthly in the apex, in Sandy Hook-Raritan Bay, and in Great Bay, New Jersey. Caged winter flounder were used to assess the quality of bottom sediments in the apex sewage sludge area. The results of these studies are presented here.

Methods

Almost all cruises to assess fin rot disease prevalence in the apex or other offshore areas were conducted in connection with fish assessment surveys, when a Yankee otter trawl with a 60-ft headrope and 80-ft sweep was used. In Sandy Hook-Raritan Bay and Great Bay, diseased fish were collected on small vessels using an otter trawl with 25-ft headrope and 30-ft sweep. The cod end of both nets was fitted with a 0.5-inch-mesh nylon stretch liner. The large net was towed for 30 min and the small for 15–30 min. All fish collected in the trawl were examined for fin rot disease and other anatomic anomalies.

Cages (5 × 4 × 2 ft) were constructed of a welded frame of 1-inch concrete reinforc-

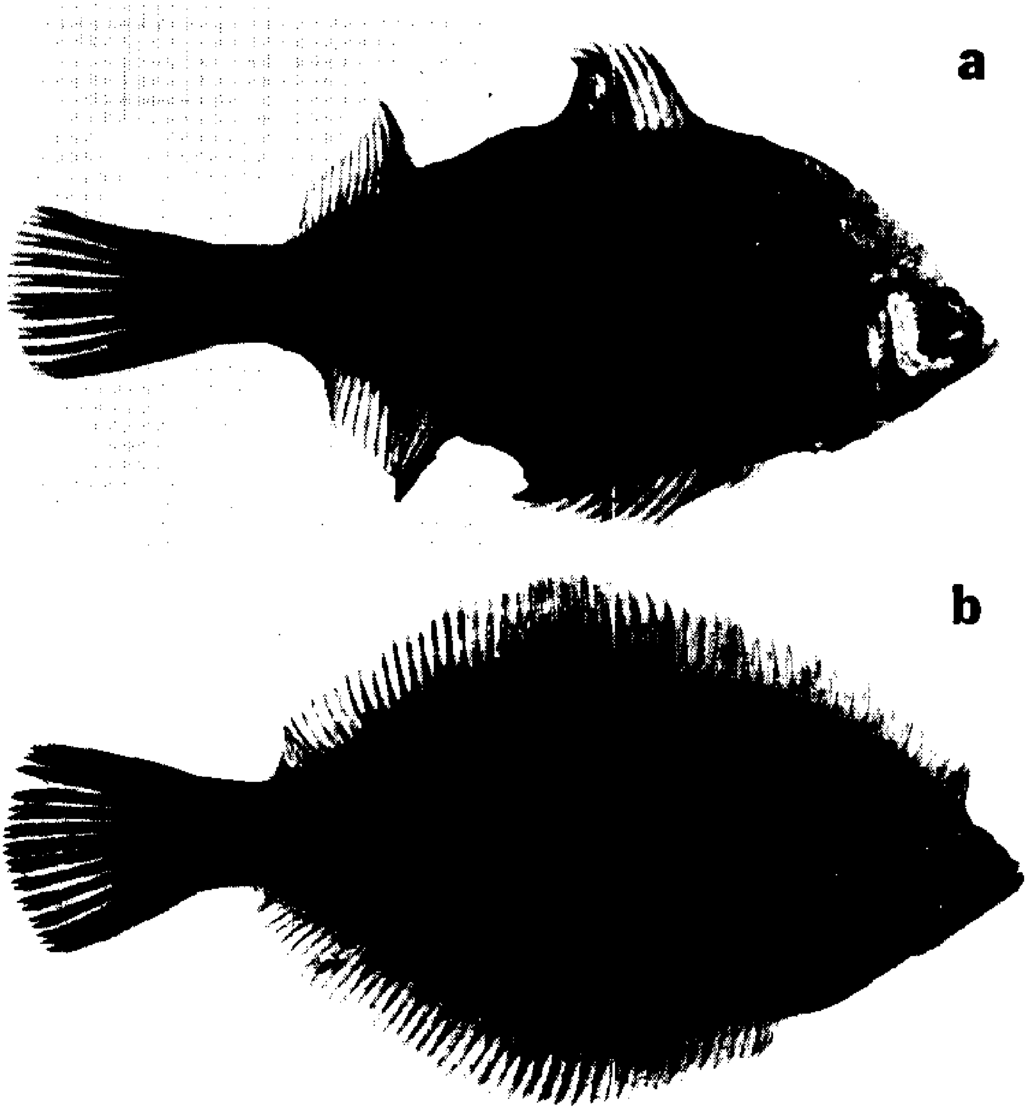

Fig. 1. Winter flounder with fin rot disease (a) and normal winter flounder (b).

ing bar stock enclosed with 2-inch stretch mesh nylon netting (Fig. 3). The cages were deployed in chains of two-three with anchors and buoys. When lowered to the bottom, they rested on the 5- × 4-ft side (20 ft^2). Sediments were soft enough to permit the nylon netting of the cage to penetrate the sediment interface. From 10–20 winter flounder (22–27 cm long) were placed in each cage; the cages were lowered at designated sites for 12–29 days. The sewage sludge site, at a depth of 27.4 m was about 5.4 km southeast of the Ambrose Light at 40°26.0′N lat, 73°47.5′W long. The control site, at a depth of 27.4 m was 22 km southeast of Jones Inlet, Long Island, at 40°26.0′N lat, 73°25.0′W long. Water temperatures and dissolved oxygen were measured whenever possible. Fish remaining in the cages after exposure were examined individually for fin rot and cage-induced damage.

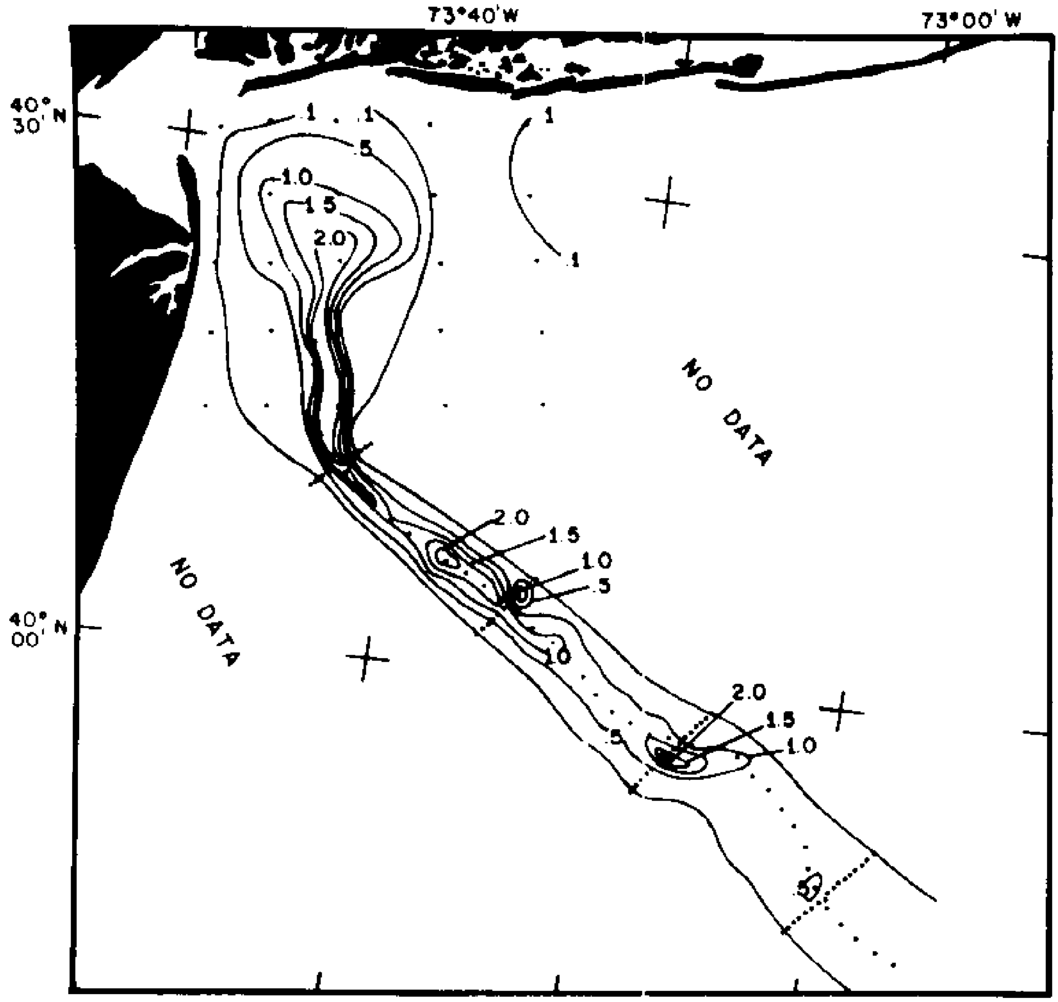

Fig. 2. Total organic carbon (% dry weight) in sediments of the New York Bight, August–September 1973 (from NOAA 1975).

Results

Fin rot disease in winter flounder—The prevalence of fin rot disease in winter flounder from the bight from February 1974 through June 1975 (Table 1) was 0.67% outside the apex and 3.9% in it. Disease prevalence was 3.0% in the low carbon area of the apex and 5.1% in the high carbon area. A total of 444 trawl collections were made and 2,689 winter flounder examined. Although the numbers of fish examined from outside (1,190) and inside (1,499) the apex were about equal, 58 diseased fish were obtained from the apex and only 8 from outside it. More fish were examined from the low carbon area (876) than the high carbon area (623), but more fish from the high carbon area (32) than from the low carbon area (26) had fin rot.

Fin rot disease in summer flounder—The prevalence of fin rot disease in summer flounder from the bight from January 1974 through September 1975 (Table 2) was 0.61% in the apex, 3.0% in Sandy Hook-Raritan Bay, and 0% in Great Bay. Altogether, 244 trawl collections were made and 2,537 summer flounder were examined. In the apex, only 7 of 1,145 fish examined (0.6%) were diseased, whereas from Sandy Hook-Raritan Bay 28 of 947 (3%) had fin

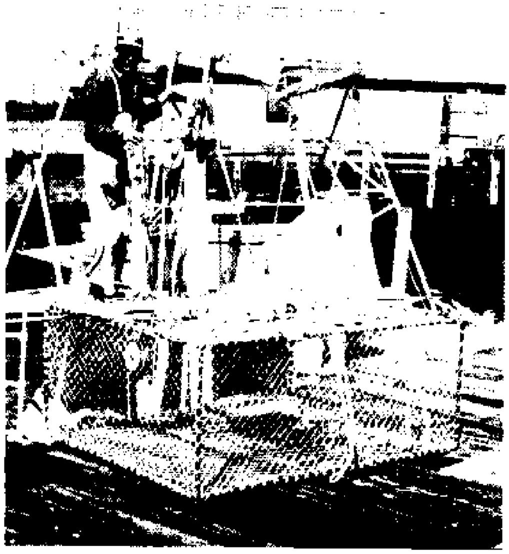

Fig. 3. Cage used in fin rot induction studies.

rot. Since no trawl collections were made in the apex during June and August 1974 and July, August, and September 1975, it is not possible to assess the incidence of fin rot disease in summer flounder from the apex during these times. Collections made during 1974 in Sandy Hook-Raritan Bay suggest that the incidence of diseased summer flounder increases from June (1) through November (8)—the period of onshore residence. Although the number of fish caught decreased during this time (due to migration offshore beginning in September), the percentage of diseased fish increased.

Caging experiments—On seven occasions from April 1975 to August 1975, caged winter flounder were placed at designated sites in the New York Bight. The results of these experiments are presented in Table 3. Cages were placed at the sewage sludge site six times and at the control site once. Water temperatures at the sewage sludge site ranged from 5.7°C in April to 17.0°C in August; in the one successful experiment at the control site, the water temperature was 17.0°C at recovery in August. Exposures ranged from 12 days in July to 29 days in May. The number of caged winter flounder in any one trial ranged from 20 to 80;

Table 1. Prevalence of fin rot disease in winter flounder (*Pseudopleuronectes americanus*) from the New York Bight, February 1974–June 1975. No stations were taken in March, June, August, or December 1974.

	Feb	Apr	May	Jul	Sep	Oct	Nov	Jan	Feb	Mar	Apr	May	Jun	All
Outside apex														
No. trawl stations	23	56	6	32	3	31	30	1	39	20	–	–	–	241
Total No. fish	112	484	130	80	0	46	120	170	9	39	–	–	–	1,190
Total No. diseased	1	7	0	0	–	0	0	0	0	0	–	–	–	8
% diseased	0.89	1.4	–	–	–	–	–	–	–	–	–	–	–	0.67
Inside apex														
No. trawl stations	27	27	–	9	19	9	5	5	6	5	18	29	44	203
Total No. fish	102	207	–	22	10	40	28	97	11	24	176	317	465	1,499
Total No. diseased	4	21	–	1	0	0	0	4	0	0	6	8	14	58
% diseased	3.9	10.1	–	4.5	–	–	–	4.1	–	–	3.4	2.5	3.0	3.9*
Low carbon area of apex														
No. trawl stations	17	19	–	6	16	8	4	4	5	3	10	18	31	141
Total No. fish	75	118	–	7	8	40	22	63	10	14	86	208	225	876
Total No. diseased	2	11	–	0	0	0	0	4	0	0	1	3	5	26
% diseased	2.7	4.0	–	–	–	–	–	6.3	–	–	1.2	1.4	2.2	3.0
High carbon area of apex														
No. trawl stations	10	8	–	3	3	1	1	1	1	2	8	11	13	62
Total No. fish	27	89	–	15	2	0	6	34	1	10	90	109	240	623
Total No. diseased	2	10	–	1	0	–	0	0	0	0	5	5	9	32
% diseased	7.4	11.2	–	6.7	–	–	–	–	–	–	5.5	4.6	3.7	5.1†

* Prevalence of fin rot disease is significantly higher ($P \leq 0.001$) inside the apex than outside it.

† Prevalence of fin rot disease is significantly higher ($P \leq 0.03$) in the high carbon area of the apex than in the low carbon area.

Table 2. Prevalence of fin rot disease in summer flounder (*Paralichthys dentatus*) from the New York Bight, March 1974–September 1975. No stations were taken in the apex in August 1974 or March 1975; none were taken in Sandy Hook-Raritan Bay in April, May, and December 1974 or in January, February, March, and July 1975; none were taken in Great Bay from September 1974 through April 1975 or in July 1975.

Apex	Jul	Sep	Oct	Nov	Dec	Jan	Feb	Apr	May	Jun	All
No. trawl stations	2	11	10	5	1	3	3	12	27	66	140
Total No. fish	2	31	127	40	8	3	6	25	220	683	1,145
Total No. diseased	0	0	2	0	2	0	0	0	0	3	7
% diseased	–	–	1.5	–	25.0	–	–	–	–	0.4	0.61

Sandy Hook-Raritan Bay	Mar	Jun	Jul	Aug	Sep	Oct	Nov	Apr	May	Jun	Aug	Sep	All
No. trawl stations	1	6	13	9	5	9	4	1	2	7	4	21	82
Total No. fish	2	11	245	93	89	41	11	1	2	132	144	176	947
Total No. diseased	0	1	2	0	1	5	8	0	0	5	3	3	28
% diseased	–	9.0	0.8	–	1.1	12.1	72.7	–	–	3.7	2.0	1.7	3.0*

Great Bay	May	Jun	Jul	Aug	May	Jun	Aug	All
No. trawl stations	4	4	4	3	1	3	3	22
Total No. fish	27	112	145	19	42	81	19	445
Total No. diseased	0	0	0	0	0	0	0	0
% diseased	–	–	–	–	–	–	–	–

* Prevalence of fin rot disease is significantly higher ($P \leq 0.001$) in Sandy Hook-Raritan Bay than in the apex.

the number of surviving fish ranged from 0 to 26. Fish recovered from cages placed in the sewage sludge area invariably demonstrated fin rot. Acute caudal fin lesions were observed more frequently than lesions of the anal and dorsal fins. The severity of the lesions was directly related to exposure time. Fin lesions were not significant in caged fish from the control site.

To compare the conditions of fish from traps in the sewage sludge and control areas, a "condition coefficient" based on values from 1–5 (1—normal; 5—dead or missing) was used. Condition coefficients for fish from the sewage sludge area ranged from 3.6 to 5.0. The condition coefficient was 2.8 for the one trial at the control location. It is apparent from Table 3 that the survival of caged winter flounder at the control site was appreciably greater than the survival at the sewage sludge site. Although the primary reason for placing

Table 3. Experimental variables and condition coefficients* of caged winter flounder (*Pseudopleuronectes americanus*) placed in two areas of the New York Bight.

	Trial No. Apex						Control
	1 15 Apr–14 May	2 29 May–11 Jun	3 27 Jun–9 Jul	4 9–25 Jul	5 25 Jul–6 Aug	6 6–26 Aug	1 24 Jul–13 Aug
Duration of exposure (d)	29	13	12	16	13	20	20
Water temp (°C)	5.7	10.0	9.5	17.0	13.1	15.0	17.0
Dissolved oxygen† (ppb)	9.7	—	—	—	5.4	5.4	8.1
No. fish							
Initial	80	54	40	36	20	20	20
Final	26	26	19	4	0	0	16
% survival	32.5	48.1	47.5	11.1	0	0	80.0
Condition coefficient	4.3	3.6	4.0	4.7	5.0	5.0	2.8

* Determined by assigning scores from 1 (normal) to 5 (dead or missing) to fish after exposure and dividing aggregate score by initial number of fish in traps.

† Winkler method.

caged winter flounder in the sewage sludge area was to determine environmental effect on induction of fin rot, it now appears that at any one time the number of surviving fish can be used to assess the effects of bottom water and sediments on the survival of caged fish.

Discussion

Surveys for diseased marine fish have documented the occurrence of fin rot disease in fish from the Gulf of Maine (Sindermann 1966), the Irish Sea (Perkins et al. 1972), the New York Bight (Mahoney et al. 1973), and coastal waters of southern California (Mearns and Sherwood 1974).

From data presented here and earlier by Ziskowski and Murchelano (1975), the prevalence of fin rot disease in winter flounder from the bight apex is significantly greater than from outside the apex. Winter flounder is the only numerically abundant fish caught with a high disease prevalence. Although the prevalence of fin rot disease is high in winter flounder from the apex, the fish may not have contracted the disease there. The disease may have been contracted elsewhere and the elevated prevalence in the area sampled merely reflects the movement of the fish into the survey area. Therefore, disease prevalence surveys alone cannot establish the specific area in the bight where the disease is contracted. The contiguous waters of Sandy Hook-Raritan Bay, although not direct recipients of barged waste disposal, certainly must be considered ecologically degraded. During their seasonal migrations, winter flounder spend 7 months (November–May) in the shallow estuarine environments throughout Sandy Hook-Raritan Bay and the lower Hudson River. At present, we can only speculate on the reason for the high prevalence of fin rot disease in winter flounder from the high carbon area of the apex. Analysis of 1973 data revealed that 312 of 1,603 (19.5%) winter flounder from the high carbon area had fin rot and 44 out of 724 (6.1%) winter flounder from the low carbon area had the disease (difference significant at $P=0.001$). The ability of sedimentary materials in the sludge disposal area to induce fin rot in any species of fish must yet be demonstrated conclusively.

It was hoped that the monthly assessment of fin rot disease in summer flounder would help define where in the bight the disease is contracted. Unlike winter flounder, which is present in the apex throughout the year, the summer flounder is an onshore migrant, abundant in the apex area only from June through October. If the incidence of fin rot disease increases monthly in any area sampled during the onshore period, that area may be where the disease is contracted. Since most offshore surveys were conducted in conjunction with fish stock assessment cruises, adequate collections were not made in the apex during 1974–1975 to determine fin rot disease incidence.

No conclusions can be drawn from the 1975 data on prevalence of fin rot disease in summer flounder from Sandy Hook-Raritan Bay until final collections have been made in November 1975. The increased incidence of fin rot disease in summer flounder from Sandy Hook-Raritan Bay from June through November 1974 suggests that the disease was contracted there. The sizes of some of the diseased summer flounder caught in November were consistent with 0-age fish. Since these fish had not yet migrated, one might assume that they contracted the disease in the area where they were caught. However, as suggested for winter flounder, the disease may have been contracted in another area and the high prevalence at any one time in the area sampled merely reflected the movement of fish to the area. Summer flounder may contract fin rot disease as they migrate through the degraded apex area on their way to Sandy Hook-Raritan Bay. The initiator of the disease may be present in the sludge-containing sediments of the apex area, but expression and progression of the disease may occur during the fish's residence in Sandy Hook-Raritan Bay. Although summer flounder are not present in Sandy Hook-Raritan Bay as long as winter flounder, their presence during periods of elevated water temperature would enhance microbial participation in disease pathogenesis. At present, it is not possible to favor either the degraded apex area or Sandy Hook-Raritan Bay as where summer flounder contract fin rot disease.

Caged winter flounder in the sewage sludge disposal area almost invariably developed fin erosion; in contrast, caged winter flounder at the control site did not develop the disease. The severity of the acute, caudal lesions observed on the fish in the sludge area depended on the duration of exposure in the cages. It is not known, however, if the lesions would continue to progress or remit if the cages were removed from the sludge area and placed in a clean environment. If the fish can survive the trauma of an additional relocation, moving cages containing diseased fish to a clean environment may be informative. If the progression of the disease depends on the combined stresses imposed by the environment at the sewage sludge area (low O_2, heavy metals, toxic inorganics and organics, high bacterial populations, etc.), moving the cages to a more suitable environment may promote resolution. If resolution does not occur, one may hypothesize that irreparable tissue damage has taken place or an aggressive infectious agent is present.

Experiments with caged winter flounder were intended only to assess the induction of fin rot disease by sediments in the sewage sludge area. It was, however, demonstrated that sediments or water at the site can profoundly affect the survival of caged fish. Percentage survival and condition were poor among caged winter flounder from the sludge area. Although only one exposure trial was conducted at the control site, the number of fish per cage was comparable to the number per cage at the sludge site. Fortunately the one exposure at the control site was made when the ambient water temperature was approaching its seasonal maximum; comparing the results of cage experiments conducted at the two sites at entirely different water temperatures would not be meaningful. The cage itself has an impact on the fish, which is influenced by water temperature. Self-inflicted, cage-induced injury from increased activity is more likely to occur when water temperatures are high. Elevated water temperatures may also affect the metabolic rates of potentially infectious microorganisms present in water and sediments and the solubility of dissolved oxygen. Survival of caged fish at the sewage sludge site may also be influenced by the concentration of dissolved oxygen at the sediment–water interface. Unfortunately, dissolved oxygen concentrations were not always measured in the initial exposure trials.

Winter and summer flounder from the New York Bight apex and from Sandy Hook-Raritan Bay have a higher prevalence of fin rot disease than fish from nonstressed biologically and hydrographically comparable areas. The pathogenesis of the disease remains unknown. Bacteriologic studies with diseased winter flounder (Murchelano unpublished) have not demonstrated the consistent presence of a dominant bacterium in the fin lesions cultured; immunologic studies (Robohm unpublished) have also failed to demonstrate consistently elevated agglutinin titers to bacteria isolated from fin rot lesions. Murchelano (1975) was unable to elucidate the pathogenesis of the disease from histopathologic studies of fin rot diseased winter flounder. It frequently is alleged that environmental stresses affect the well-being of fish (Snieszko 1974); however, the mechanisms whereby the stresses produce their effects are largely unknown. Demersal fish, especially flatfish in the degraded areas of the bight, are continuously exposed to many environmental stresses. Whether any single stress or several together initiate the lesions known as fin rot remains to be determined. The complexity of the problem makes its solution elusive. Determination of the specific etiology of fin rot disease still requires extensive study.

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Chromosome mutagenesis in developing mackerel eggs sampled from the New York Bight

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Certain heavy metals and pesticides are recognized mutagens and, along with some other major classes of marine contaminants, may have important implications in survival of fish populations (Longwell 1975). This may be particularly so for fish using the polluted New York Bight as spawning grounds. Mutagens can cause genetic damage at subtoxic levels. Many marine contaminants accumulate in the body tissues of fish and other marine species. Cadmium has been shown to be absorbed from seawater by postspawned fish eggs (Rosenthal and Sperling 1974; Westernhagen et al. 1974, 1975).

Cells in the meiotic divisions of gametogenesis are particularly sensitive to the damaging effects of mutagens. Early cleavage mitoses of the fertilized egg are even more sensitive. When fish eggs, often already carrying significant contaminant loads, are spawned in polluted waters they have only halfway completed these sensitive meiotic divisions with their intricate chromosome maneuvers. As components of the neuston in surface waters, fertilized fish eggs must then undergo repeated divisions of their chromosomes during the even more sensitive stages of early cleavage.

From studies of mammals, insects, and plants we know that most dominant lethal mutations which kill at early stages of development are associated with gross chromosome alterations. In an experimental study of incorporated radionuclides in eggs of commercial turbot (*Rhombus maeoticus*) and ruff (*Scorpaena porcus*), Russian workers found increased incidences of chromosome aberrations in normal and abnormal larvae associated with poor hatchability and reduced vigor of hatched larvae (Polikarpov 1972).

The chromosomes of developing fish eggs provide a sensitive test for genetically active substances, both experimentally and in polluted natural waters. Any new determination of mortality of early fish stages, irrespective of its cause, would have important bearing on the general theories of fluctuations of fish populations and on predictions of success of any year class of commercial fish.

It is not necessary to assign genetic damage to specific chromosomes, or to know details of the chromosomes of a fish species, or even to know numbers of the chromosomes of a fish to estimate rate of occurrence of such damage. Whether spontaneous or induced by radiation, chemicals, or physical stress, a significant portion of all chromosome breakage, as well as faulty distributions of the chromosome material, is easily observable and measurable by abnormal orientation, behavior, and configurations of the chromosomes in dividing cells. Many chemicals affect the mitotic (or meiotic) spindle apparatus itself and cause maldistribution, or nondivision, of the chromosomes because of their failure to orient together on the defective spindle.

With such test criteria in mind, as well as what may be the special susceptibility of fish eggs to genetic damage in nature, a study was made of fish eggs from 40 sample stations in the New York Bight. These were collected from surface waters during the 7–18 May 1974 cruise of the *Westward* (Sailing Educ. Assoc.) sponsored by the MESA Project (NOAA) and assisted by the Middle Atlantic Coastal Fisheries Center. The New York Bight is simultaneously one of the most heavily polluted coastal areas in the U.S. and the spawning grounds for large numbers of economically important fish (NOAA 1974).

Eggs of 15 different samples from 14 stations thus far studied have been identified by W. Smith as being almost exclusively those of the Atlantic mackerel *Scomber scombrus*.

Altogether 30,689 embryo cells were scored in 452 eggs from the 14 different *Westward* stations. Less than 20% of all the eggs had all their dividing cells free from chromosome and division abnormalities. A third of the 30,689 division figures scored for all the embryos were abnormal. By far the largest portion of the data came from the early embryo stage just after gastrulation and formation of the three primitive cell layers.

Abnormalities of the chromosomes extended through the entire range of radiomimetic effects on the chromosomes and their division apparatus, including extreme stickiness of chromosomes having obvious division difficulties and irregularities, failure of chromosomes to orient on spindles, with consequent loss of chromosomes, and chromosome breakage.

Cells with at least one chromosome or mitotic irregularity for the eggs at any one *Westward* station varied from a mean low of about 13% to a mean high of about 79%. This low of 13% is close to the frequency of abnormalities reported by Russian workers (Polikarpov 1972) as background aberrations in the turbot and ruff in their experimental study; they used a different scoring system and slightly earlier embryos still in gastrulation. Background mutation rate of fish is, of course, important in interpreting data. Waste disposal sites, station locations, and mean percent of early embryo cells with chromosome abnormalities are shown in Fig. 1.

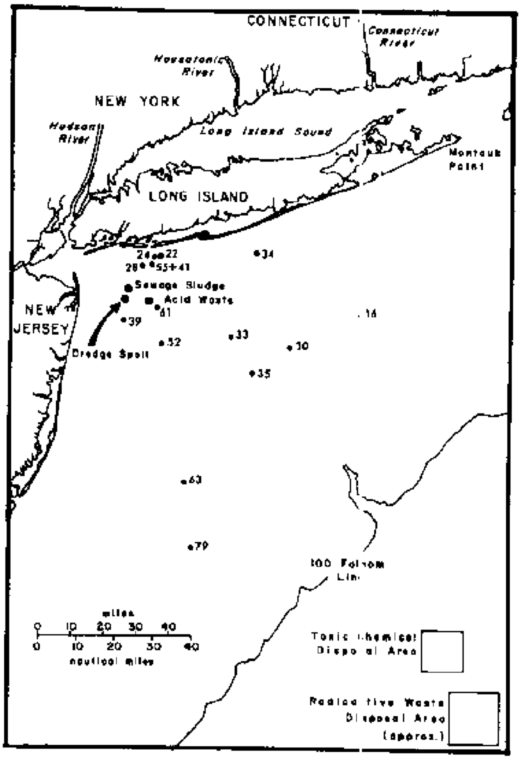

Fig. 1. The mean percent early embryo cells of Atlantic mackerel eggs with chromosome abnormalities is indicated next to the sample site. *Westward* cruise, May 1974. The second figure given for one station represents an overnight rerun sample.

Two stations just south of Long Island were among those with the lowest incidences of chromosome aberrations and mitotic irregularities. A third station farther up the Long Island coast had a slightly higher incidence.

A station about 5 mi southwest of the two coast stations in the direction of the sewage sludge and dredge spoil dump sites had an incidence of abnormalities only slightly higher than the two coastal stations. A second station, also about 5 mi closer to these dump sites and about 5 mi away from this other station, had a somewhat higher incidence of abnormalities.

Eggs sampled near the acid waste dump site showed an appreciably higher incidence of abnormalities, as did two stations a few miles to the south of it.

Away from dumping sites and the coast about halfway to the edge of the continental shelf four stations showed relatively low to very low aberration rates.

Another outermost station, the farthest point from the sewage sludge and dredge spoil dump sites and about 55 mi offshore, had the highest incidence of abnormalities. This station was closer (about 67 mi) to the toxic chemicals disposal area (NOAA 1975) and the radioactive waste disposal area than other samples. Another station relatively close to this one, about 45 mi off the New Jersey shore and 79 mi from the toxic chemicals disposal area, had the second highest incidence of abnormalities.

The station with the highest mean off the New Jersey coast was the only one with any significant observable mortality. On the basis of cell contrast and deterioration of the nuclei, 20 of 76 embryos (26%) were already dead. There was not yet any gross deterioration of the embryo or egg. Also, there were more instances of multiple chromosome abnormalities within mitosing cells of embryos from this station.

Winter movements of mackerel are not completely understood. There is a possibility that eggs sampled at the outermost stations came from fish that overwintered and underwent gametogenesis just off the continental shelf in the same general area where they spawned. Eggs studied from stations farther north, inshore, and along the Long Island coast are more likely to have been spawned by fish migrating north from overwintering areas farther south.

Statistical analysis of the data was done using both the *t*-test comparing each station in all combinations and Duncan's new multiple range test (Steel and Torrie 1960). The 0.05 level of significance was used. Results of the two tests agreed generally and revealed a similar pattern of statistically significant differences.

Stations in close proximity to each other did not differ significantly, nor did the day and night samples taken at one station. Stations with lowest means, such as the two Long Island shore stations and two of the four northern outermost stations halfway to the edge of the continental shelf, showed few significant differences with each other. They did differ significantly from stations near the sewage sludge and acid waste dump sites and the southernmost peripheral stations with the highest means. Stations with intermediate means, such as two of the four northern outermost stations and the one along the Long Island coast halfway to Montauk Point, differed in general only from stations with highest and lowest means. There were no significant differences among the four samples with highest mean aberration rates, that is, the samples from the two southernmost stations about 50 mi off the New Jersey coast, the one from near the acid waste dump site, and one of the two samples taken at the station southwest of the Long Island stations in the direction of the sewage sludge and dredge spoil dump sites.

A full presentation of this work is to be published as a MESA Technical Report.

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Section 7 Benthic processes

Bacterial flux in some New Jersey estuarine sediments[1]

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Abstract

From July 1972 through December 1974 ten sampling trips were made in Raritan Bay and Sandy Hook Bay to obtain estuarine sediments. The cores were subsectioned at approximately 10-cm intervals and each section was analyzed for pH, *Eh*, carbon, nitrogen, percent moisture, total aerobic colony forming units, and facultatively anaerobic colony forming units. Replica plate technique was used to determine the inorganic and organic nitrogen requirements of the cultured aerobic colonies. Sedimentary bacterial populations exhibit wide fluctuations in total numbers (up to five orders of magnitude), relative population components, and inorganic nitrogen requirements. These fluxes cannot be correlated overall with season, sediment particle size, nitrogen or carbon content, pH, or *Eh*. Although the general trend of high bacterial numbers in silt-clay sediments exists, the marked decrease in total and specific bacterial flora at station 15 makes it impossible to consider this an absolute principle for estimating bacterial numbers and potential for activity in marine sediments.

Although bacteria have been recognized as components of the marine sedimentary microflora since the Talisman expedition and Certes' report of their isolation in 1882 (Certes 1884), the factors influencing bacterial distribution in deep ocean and even coastal sediments are poorly understood. In the last century, Russell noted that the numbers of bacteria decreased with increasing distance from shore and with increasing depth in a core (Russell 1892*a*,*b*). At first this seemed logical because it was assumed that with increasing core depth, the oxygen supply would become deficient and this would lead to the death of the predominant aerobic bacteria which might be replaced by less numerous anaerobic forms. However, by 1940 ZoBell and his coworkers had amply demonstrated that although the numbers of organisms did indeed decrease with core depth, this was not necessarily due to decreased oxygen levels or to the presence of negative redox potentials; certainly the anaerobic isolates did not always become predominant—at least in the upper 40 to 50 cm of a core (ZoBell and Anderson 1936; Rittenberg 1940; ZoBell and Feltham 1942).

Meanwhile, Lloyd demonstrated that fluctuations in the bacterial populations of some Clyde Sea sediments occurred to depths of about 20–30 cm. He attributed these changes to fluctuations in phytoplankton blooms in the overlying water. Although he had only two samples, taken at approximately 6-week intervals, he cautioned that the range of fluctuation might not be due to food source or season but to reasons not then understood (Lloyd 1930–1931).

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More recently, examination of factors affecting bacterial distributions in sediments have centered primarily on lake and intertidal regions. In 1960, Oppenheimer reported that while it might appear that the carbon and nutrient contents of sediments were affecting bacterial distributions, it was equally likely that discontinuities observed in sediment properties were the result of microbial activity (Oppenheimer 1960). Later Volkmann and Oppenheimer (1962) noted that particle size appeared related to the total numbers of organisms present but that much of the organic matter was refractory to microbial attack; hence carbon and perhaps nitrogen content were not major determinants of bacterial distributions in sediments. More recently, Tsernoglou and Anthony (1971) noted a direct relationship between particle size and bacterial numbers for 10 Canadian lakes. These results have been confirmed by Dale (1974) for some intertidal sediments; he also noted a high degree of correlation between the bacterial numbers and the carbon and nitrogen contents of these sediments. However, in none of these cases has the possibility of bacterial fluctuations been considered.

If the roles of bacteria in sediments are mainly as primary degraders and as food sources for higher eucaryotic organisms, then it becomes important to understand the range of bacterial populations and activities in marine sediments, for these would naturally influence the concentration of the higher forms. Several questions still need to be answered regarding microorganisms in marine sediments—

1. To what extent do total numbers of bacteria fluctuate?

2. Are these fluctuations in the total population or selected nutritional, physiological, or generic groups?

3. Are there changes in the rates and degree of microbial attack on substrates?

4. Do any of the physical-chemical properties of the sediments greatly influence any of the potential fluxes of the microbial flora?

While recognizing the difficulties of repeated sampling at the exact same location in a marine environment, we have studied the range of bacterial fluctuations over a 2½-year period in sediments from Sandy Hook Bay and Raritan Bay, New Jersey. These are sufficiently shallow bays with good buoy markings and landmarks so that we can reliably return to the same general area. In addition, we examined selected chemical and physical parameters of these sediments to elucidate those parameters with long term importance for bacterial distribution, flux, and activity.

We gratefully acknowledge the assistance of J. Pearce and the masters and crews of the vessels of the Sandy Hook National Marine Fisheries Laboratory for providing ship-time and field assistance. In addition, we are indebted to J. O'Reilly for assistance in the carbon and nitrogen analyses, R. Loveland for his help in the interpretation of the statistical information, and D. Brown, D. Stein, R. Watanabe, and J. Zindulis for technical assistance during various phases of this study.

Materials and methods

Field studies—Locations of sediment sampling stations are shown in Fig. 1; the general circulation patterns and water characteristics have been previously described (Jeffries 1962). A gravity corer was used at stations 15, 61, and 62 where the sediment was more compact; a Smith-McIntyre grab sampler was used at station 2, which has a high sand content. Immediately after collection the following measurements were taken at each designated core depth: pH (microcombination electrode); *Eh* (platinum electrode with a separate calomel reference electrode); and temperature with a standard 76-mm immersion thermometer. For both the pH and *Eh* measurements, the instruments were allowed to stabilize before recording the appropriate end point (Whitfield 1969).

About 10- or 100-g sediment samples were aseptically removed and added to 90 or 900 ml of sterile artificial seawater at a salinity of 35‰. These 10^{-1} dilutions provided the material for all subsequent bacteriological studies conducted according to previously described procedures (Litchfield et al. 1975). Twenty-milliliter aliquots

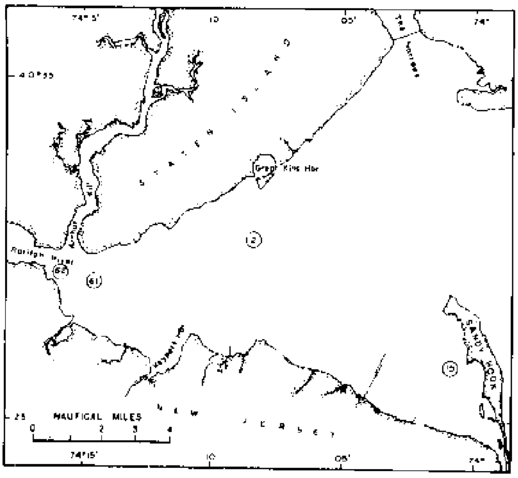

Fig. 1. Sampling stations in Sandy Hook Bay and Raritan Bay.

Table 1. Composition of media used for the culture of sedimentary microorganisms.

Medium designation*	Ingredients
Basal medium	0.05 g sodium glycerophosphate 0.5 ml glycerol 1,000 ml artificial seawater 20 g agar
ESW	1,000 ml basal medium plus 0.10 g proteose peptone No. 3 and 0.10 g yeast extract
Urea	1,000 ml basal medium plus 1.0 g filter-sterilized urea
KNO_3	1,000 ml basal medium plus 1.0 g KNO_3
$(NH_4)_2SO_4$	1,000 ml basal medium plus 1.0 g $(NH_4)_2SO_4$
Thioglycolate-casein	1,000 ml basal medium plus 0.01 g thioglycolate and 1.0 g casein

* pH was adjusted to 7.4±0.1 before autoclaving and checked after sterilization.

were removed to determine the dry weight of the sediment in the dilutions used for plating. These aliquots were filtered through dried, tared, glass fiber filters, which were then dried at 110°C until five constant weights (SD ±1%) were obtained. All chemical and bacteriological data were converted to a gram dry weight basis for more accurate comparisons between stations.

Microbiological analyses—Serial dilutions were prepared in artificial seawater and 0.1-ml quantities were surface spread onto the ESW medium described in Table 1. Three or five replicates of each dilution were inoculated and incubated at 15°C with counts taken at weekly intervals for up to 4 weeks. Replicate counts having 10% or less variation were averaged and the resulting numbers converted to colony forming units (CFU) per gram dry weight sediment.

Facultatively anaerobic bacteria were also incubated at 15°C but under anaerobic conditions using the BBL gas pack system after surface spread plating onto thioglycolate-casein or thioglycolate-ESW medium (Table 1). Because the inoculum originated from the same dilution bottles that provided the inoculum for the aerobic plates, the reducing conditions necessary for the isolation of obligately anaerobic bacteria were not possible (Hungate 1969). Colonies resulting from incubation under these anaerobic conditions are consequently considered facultative anaerobes. Random sampling did indicate a preponderance (60–90%) of catalase negative colonies, but this is not a priori proof of an obligately anaerobic metabolism (Hungate 1969). Enumeration and evaluation of the resulting colonies was similar to that used for the aerobic plates.

All plates were scored on the basis of colony size, color, and filamentous structure indicative of fungi or Actinomycetales. No attempt was made to distinguish between bacterial and yeast colonies although examination on four occasions did reveal the presence of yeasts on the ESW medium (Rake and Burstyn unpublished).

Evaluation of the nitrogen requirements of the isolates was accomplished by replica plating from plates containing from 20 to 100 colonies on the media listed in Table 1. The same aerobic incubation conditions listed above were used.

Other analyses—Values for total carbon were obtained on a LECO carbon analyzer (model 521-275); nitrogen was determined on a Coleman nitrogen analyzer (model 29A). The percent moisture was determined by drying at 75°C wet-weighed samples and calculating the loss in moisture. Particle size was calculated following wet-sieving and pipette analysis of the 64-μ-sized fraction (Krumbein and Pettijohn 1938) of selected samples. The resulting data were analyzed using multivariant and linear regression analysis as well as "Student's" *t*-test.

Results

Sediment at station 2 consisted of coarse-to-medium sand with about 94% of the material having a ϕ-sorting value in the range of 1–3. Particle size analysis of the core subsections at station 15 showed that 80% of the surface material had a ϕ-sorting value of 3, while deeper in the core it was coarse-to-fine silt (ϕ3). Station 61 had a coarse-to-fine sand with silt content increasing with depth in the core. Sediments at station 62 contained about twice as much silt as station 61 and correspondingly less coarse-to-medium sand.

Data from chemical and bacteriological analyses of these sediments in Sandy Hook and Raritan Bays are presented in Table 2. Fluctuations in the pH of the sediments were minimal regardless of the station location, season, or depth in the core, and the redox potential was always slightly positive, again independent of station and season. Thus the pH and *Eh* would not be limiting to the growth of aerobic and facultatively anaerobic bacteria. Wide fluctuations occurred in the carbon, nitrogen, and total CFU values during the sampling period and extended to all stations and depths within the cores, including the 30–40-cm section.

Fluctuations in the CFU/g dry weight during the study period for the different stations are presented in Figs. 2, 3, and 4; both the aerobic and facultatively anaerobic CFU are shown. Most noticeable in all three figures is that there is no apparent correlation between CFU and season. Fluctuations in the population occur even at 20–30 cm in the core, and although at station 15 (Fig. 3) they do appear to follow the changes in the surficial layer, this is not a consistent pattern. The flux at station 2 (Fig. 2) is dramatic if the single sampling time corresponding to December 1973 is included. However, if that value is removed from the overall data, we see that the general range of bacterial flux in these sediments is relatively small—only about two orders of magnitude. From the data in Table 2 it is not possible to cite any specific cause for this unusual dip in the CFU in December 1973 because generally there were no major alterations in the pH, *Eh*, carbon, nitrogen, or moisture content of this sample.

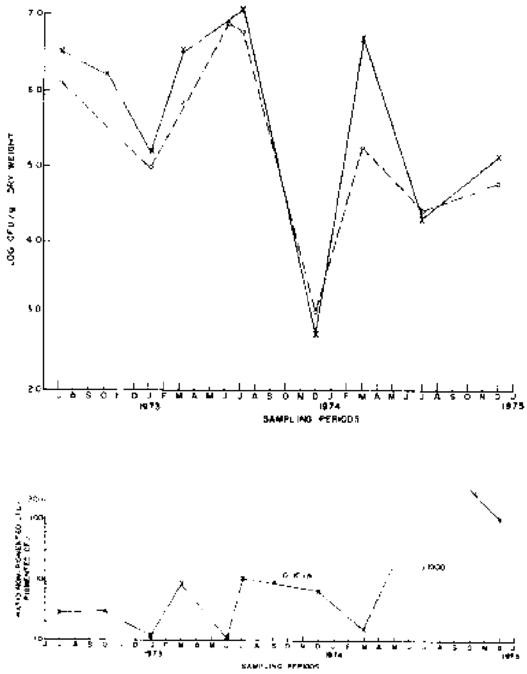

Fig. 2. Fluctuations in bacterial population at station 2, July 1972 through December 1974. Upper graph shows the variations in the total aerobic colony forming units (solid line) and the total facultatively anaerobic colony forming units (dashed line). Lower graph ratio of >1,000 occurred when no pigmented colonies were scored on the plates.

The CFU flux can be quite large as observed at station 15 (Fig. 3) where a difference of five orders of magnitude exists between the highest and lowest numbers reported. Also, there was a major decline in the CFU for station 15 during the sampling period, and the bacterial population remained at the 10^4 to 10^5 level even into this year (unpublished data).

Fluctuations in the sedimentary bacterial content at stations 61 and 62 fall in between

Table 2. Summary of data for sediments in Raritan and Sandy Hook Bays, 1972–1974. The carbon, nitrogen, and CFU values are all on a gram dry weight of sediment basis.

Depth in core (cm)	20 Jul 1972	26 Oct 1972	18 Jan 1973	3 Mar 1973	13 Jun 1973	12 Jul 1973	14 Dec 1973	15 Mar 1974	25 Jul 1974	6 Dec 1974
Station 2										
0–10 pH	—*	7.0†	7.7	—	7.55	7.2	8.7	6.5	4.7	6.0
Eh	—	—	470	—	355	525	185	10	65	—
% Moisture	2.5	52.9	44.4	26.4	20.0	41.8	20.3	26.2	16.2	25.7
mg C	15.77	27.16	30.50	4.77	0.535	12.6	0.383	7.0	0.250	1.5
mg N	0.658	2.524	1.054	0.103	0.220	1.46	0.306	0.470	0.315	0.141
Aerobic CFU	3.4×10^6, 8.1×10^7	1.7×10^6	1.5×10^5	2.7×10^4	1.17×10^7, 6.21×10^6	1.35×10^7	4.83×10^5	57.6×10^5	9.9×10^3	1.53×10^5
Anaerobic CFU	1.5×10^5, 1.3×10^6	C‡	1.0×10^5	7.4×10^3	8.33×10^6, 1.26×10^7	6.63×10^6	2.38×10^6	2.08×10^5	1.27×10^4	6.7×10^4
Aerobic: anaerobic	22.67, 62.31	C	1.5	3.65	1.40, 0.49	2.04	0.2	27.69	0.79	2.28
Station 15										
0–10 pH	8.0	7.0	8.0	7.9	7.45	7.9	7.5	—	7.4	7.2
Eh	—	—	445	—	515	485	815	240	600	—
% Moisture	19	45	59.8	45.9	50.0	50.8	53.6	54.5	46.3	50.8
mg C	51.84	22.8	31.35	37.23	20.87	19.9	45.29	24.5	28.12	14.3
mg N	2.118	2.294	1.670	1.013	1.530	2.00	2.746	2.400	2.293	1.157
Aerobic CFU	6.5×10^4	3.2×10^5	3.0×10^4	2.7×10^5	3.51×10^5	9.63×10^4	1.22×10^5	12.2×10^5	1.97×10^5	1.44×10^4
Anaerobic CFU	8.4×10^4	1.5×10^4	4.7×10^4	2.2×10^4	1.43×10^6	3.44×10^4	C	5.34×10^4	1.54×10^4	1.61×10^5
Aerobic: anaerobic	0.77	21.33	0.64	12.27	0.25	2.80	C	22.85	12.79	0.09
10–20 pH	—	7.0	—	—	7.65	8.0	7.5	—	—	8.3
Eh	—	—	—	—	530	—	785	455	—	—
% Moisture	—	40.1	—	—	51.9	51.7	47.9	50.2	—	49.7
mg C	—	17.83	—	—	17.6	21.5	28.12	19.5	—	20.5
mg N	—	1.709	—	—	1.500	2.07	2.558	1.720	—	1.228
Aerobic CFU	—	5.7×10^4	—	—	1.12×10^6	3.2×10^4	2.69×10^4	1.9×10^5	—	3.60×10^3
Anaerobic CFU	—	3.8×10^4	—	—	6.06×10^5	8.63×10^3	3.27×10^3	1.57×10^4	—	1.76×10^2
Aerobic: anaerobic	—	1.50	—	—	1.85	3.72	8.2	12.10	—	20.45

Table 2. Continued

Depth in core (cm)	20 Jul 1972	26 Oct 1972	18 Jan 1973	3 Mar 1973	13 Jun 1973	12 Jul 1973	14 Dec 1973	15 Mar 1974	25 Jul 1974	6 Dec 1974
20–30 pH	—	7.0	—	5.2	7.6	7.6	7.5	—	7.5	8.5
Eh	—	—	—	—	555	—	845	615	500	—
% Moisture	—	47	—	49.1	60.0	54.2	61.3	42.1	48.1	67.0
mg C	—	16.87	—	15.2	21.94	18.0	20.96	17.0	16.71	36.6
mg N	—	2.226	—	0.8830	1.720	2.03	2.426	1.780	1.494	2.806
Aerobic CFU	—	3.5×10^5	—	1.9×10^4	4.12×10^5	3.83×10^5	1.82×10^4	4.5×10^4	5.5×10^4	7.57×10^5
Anaerobic CFU	—	2.8×10^5	—	C	2.51×10^5	1.20×10^5	1.10×10^4	1.1×10^4	8.14×10^3	3.42×10^5
Aerobic: anaerobic	—	1.25	—	C	1.64	3.19	1.65	4.1	6.72	2.21
30–40 pH	—	—	—	7.8	7.6	—	—	—	7.4	—
Eh	—	—	—	—	555	—	—	—	485	—
% Moisture	14.8	—	—	18.9	56.4	—	—	—	47.2	—
mg C	17.43	—	—	15.2	19.8	—	—	—	15.40	—
mg N	0.3189	—	—	1.0269	1.910	—	—	—	2.487	—
Aerobic CFU	4.3×10^4	—	—	1.7×10^4	2.75×10^5	—	—	—	4.70×10^4	—
Anaerobic CFU	1.9×10^4	—	—	4.8×10^4	2.69×10^4	—	—	—	9.85×10^3	—
Aerobic: anaerobic	2.26	—	—	0.35	10.2	—	—	—	4.90	—

Depth in core (cm)	20 Jul 1972	26 Oct 1972	3 Mar 1973	13 Jun 1973	Depth in core (cm)	20 Jul 1972	26 Oct 1972	3 Mar 1973	13 Jun 1973
Station 61					Station 62				
0–10 pH	—	7.0, 7.0	7.8	7.25	0–10 pH	8.0, —	7.0	7.53	7.20
Eh	—	— —	—	310	*Eh*	—, —	—	—	335
% Moisture	23.2	50.4, 60.3	50.0	69.7	% Moisture	15.5, 19.4	70.9	69.4	69.8
mg C	17.19	32.89 —	5.1§	41.3	mg C	33.43, —	33.92	9.7§	41.7
mg N	1.448	3.271 —	0.686	3.530	mg N	3.029, 2.453	3.244	—	3.280
Aerobic CFU	1.4×10^5	5.7×10^7, 4.2×10^7	5.7×10^5	7.62×10^7	Aerobic CFU	1.1×10^5, 1.3×10^5	C	1.9×10^7	2.73×10^7
Anaerobic CFU	9.4×10^5	9.2×10^7, 2.5×10^7	3.5×10^5	4.65×10^7	Anaerobic CFU	8.3×10^5, 3.2×10^4	C	8.1×10^6	$>1.46\times10^8$
Aerobic: anaerobic	0.15	0.62, 1.68	1.63	1.64	Aerobic: anaerobic	0.13, 4.06	—	2.35	<0.19

Table 2. Continued

Depth in core (cm)	20 Jul 1972	26 Oct 1972	3 Mar 1973	13 Jun 1973	Depth in core (cm)	20 Jul 1972	26 Oct 1972	3 Mar 1973	13 Jun 1973
10–20 pH	—	7.0	—	7.20	10–20 pH	—	7.0, 7.0	—	7.10
Eh	—	—	—	305	*Eh*	—	—, —	—	360
% Moisture	16.5	67.9	—	67.2	% Moisture	19.4	64.9, 63.6	—	64.8
mg C	28.4	39.28	—	73.7	mg C	—	33.94, —	—	47.1
mg N	1.4754	4.07	—	3.940	mg N	2.4527	3.49, —	—	3.065
Aerobic CFU	6.5×10^4	1.5×10^6	—	1.58×10^7	Aerobic CFU	1.3×10^5	1.4×10^7, 1.0×10^8	—	4.24×10^6
Anaerobic CFU	1.1×10^5	1.1×10^6	—	9.01×10^6	Anaerobic CFU	3.2×10^4	3.1×10^7, 3.1×10^7	—	8.73×10^6
Aerobic: anaerobic	0.59	1.36	—	1.75	Aerobic: anaerobic	4.06	0.4, 3.2	—	0.49
20–30 pH	—	—	7.15	7.20	20–30 pH	—	—	7.35	6.00
Eh	—	—	—	305	*Eh*	—	—	—	350
% Moisture	—	—	50.9	50.6	% Moisture	—	—	65.9	58.1
mg C	—	—	17.87	42.7	mg C	—	—	35.26	56.6
mg N	—	—	1.4088	3.030	mg N	—	—	3.08	2.910
Aerobic CFU	—	—	5.4×10^5	9.21×10^5	Aerobic CFU	—	—	8.0×10^6	1.10×10^6
Anaerobic CFU	—	—	5.4×10^4	4.39×10^5	Anaerobic CFU	—	—	3.9×10^6	1.19×10^6
Aerobic: anaerobic	—	—	10.0	2.10	Aerobic: anaerobic	—	—	2.05	0.92
30–40 pH			7.7		30–40 pH			7.25	6.65
Eh	—	—	—	—	*Eh*	—	—	—	320
% Moisture	—	—	49.4	—	% Moisture	—	—	53.3	53.1
mg C	—	—	19.35	—	mg C	—	—	31.85	55.8
mg N	—	—	1.6046	—	mg N	—	—	2.5038	3.095
Aerobic CFU	—	—	1.3×10^5	—	Aerobic CFU	—	—	2.1×10^6	6.17×10^5
Anaerobic CFU	—	—	2.7×10^5	—	Anaerobic CFU	—	—	2.8×10^5	2.44×10^6
Aerobic: anaerobic	—	—	0.48	—	Aerobic: anaerobic	—	—	7.5	0.25

* Information not obtained.

† pH measurements in October 1972 taken with narrow range pH paper.

‡ Contaminated plates.

§ Carbon determined by wet oxidation technique.

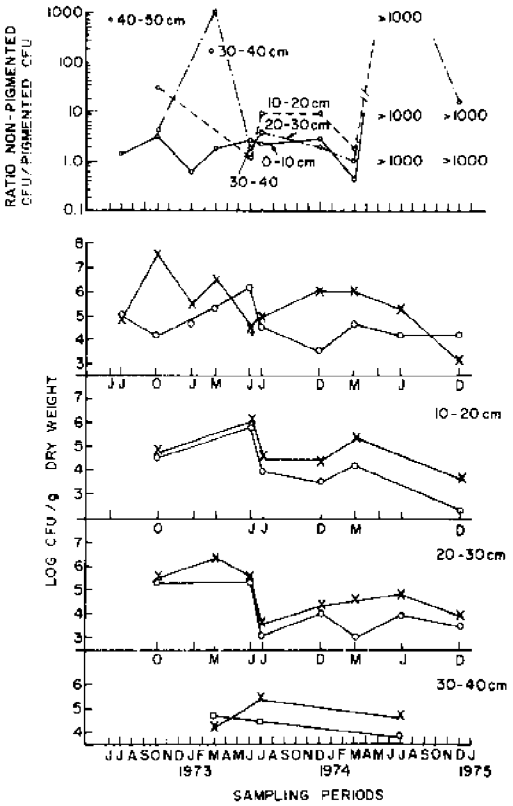

Fig. 3. Fluctuations in the bacterial population at station 15, July 1972 through December 1974. Upper graph >1,000 ratio resulted when no pigmented colonies were identified on the isolation medium. Lower graphs depict fluctuations in the log of the total aerobic colony forming units (X) and the total facultatively anaerobic colony forming units (O).

stations 2 and 15; i.e. there is a fairly consistent flux that approximates three orders of magnitude in absolute numbers and this change is not seasonally related (Fig. 4). We attempted to collect samples during the same portion of the tidal cycle, especially at these two stations, to avoid tidal-induced fluxes such as have been noted by Erkenbrecher and Stevenson (1975).

When the data were examined to determine whether the observed fluctuations were due to gross changes in the absolute population or whether the isolations reflected changes in the generic or species composition of the sediments, a slightly different pattern emerged. At station 2 the number of nonpigmented bacteria increased during the study period, culminating in July 1974 when no pigmented colonies appeared on the medium (Fig. 2). A "typical" ratio was restored by December 1974. There was no change in medium composition, incubation condition, or overall laboratory technique between the March and July sampling period. During periods of bacterial bloom at station 2, which has a predominantly sandy sediment, the ratio of nonpigmented to pigmented aerobic bacteria declined, indicating that the bloom includes proportionally more pigmented yeasts and bacteria, maybe members of the genera *Flavobacterium, Brevibacterium, Xanthamonas, Corynebacterium, Pseudomonas,* and *Serratia* (Buchanan and Gibbons 1974). Indeed, when selected isolates were examined according to the schema of Shewan et al. (1960), flavobacteria, brevibacteria, corynebacteria, and nonlactose fermenting enterics were isolated (unpublished data) as were several species of pink yeasts (Rake and Burstyn unpublished).

At station 15, however, the ratio of nonpigmented CFU to pigmented CFU closely paralleled the total CFU counts (Fig. 3). During July and December 1974, again no pigmented colonies were noted in the 0–10- and 20–30-cm core sections; in the 10–20-cm section the ratio returned to a characteristic range in December 1974.

Similar data for stations 61 and 62 are shown in Fig. 5. Although less frequently sampled than stations 2 and 15, there is a consistent, or persistent, ratio of nonpigmented to pigmented microbes at these two locations. This could indicate a more stable mixed bacterial population than found at the other two sites, perhaps reflecting the overwhelming impact of the Raritan River and the Arthur Kill.

In a further attempt to understand the fluctuations that might typify a sedimentary bacterial population, a series of nutritional experiments was undertaken. The initial isolation plates of ESW containing between 20 and 100 colonies were replica-plated onto media containing selected nitrogen sources: urea, nitrate, ammonium, and the original peptone medium. Following incu-

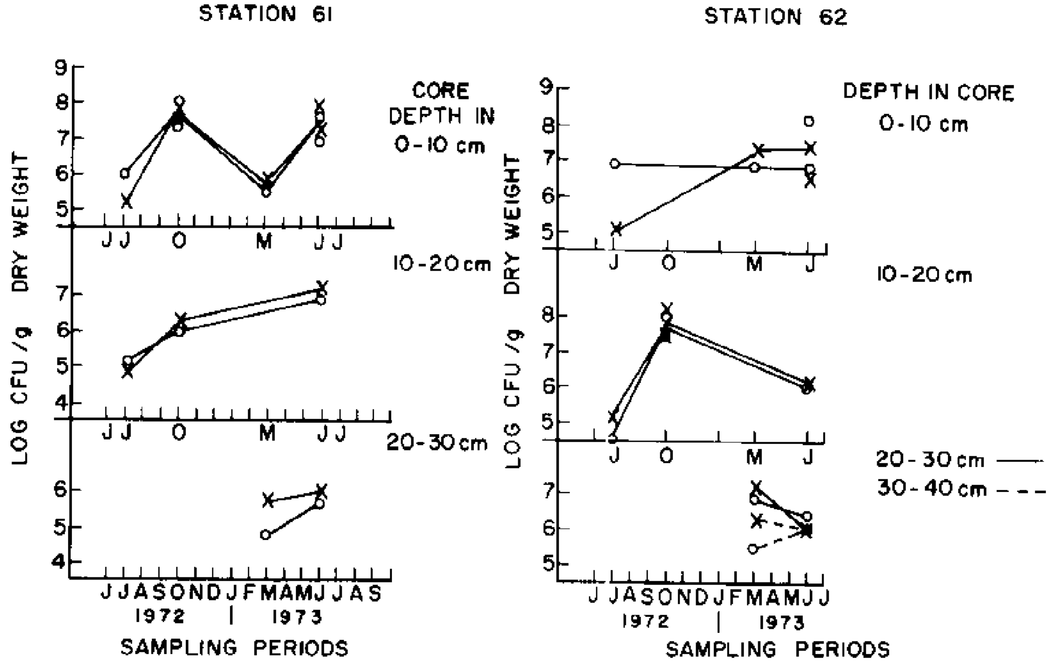

Fig. 4. Fluctuations in the bacterial population in cores from stations 61 and 62, July 1972 through June 1973. Log of the total aerobic colony forming units (X) and the total facultatively anaerobic colony forming units (O) is shown for each core subdivision.

bation for 21 days, the plates were scored for growth on the test nitrogen sources as well as on the control replicate ESW. Those organisms that failed to grow on the replicate ESW were not included in the calculations of the percent of isolates able to utilize urea, nitrate, or ammonium ion. The results (Table 3) show a wide versatility toward the selected nitrogen compounds. None of the test compounds could be consistently used by the entire heterotrophic population, however. What was surprising was the number of organisms that could grow on the urea medium inasmuch as urease is not considered a widely distributed bacterial enzyme.

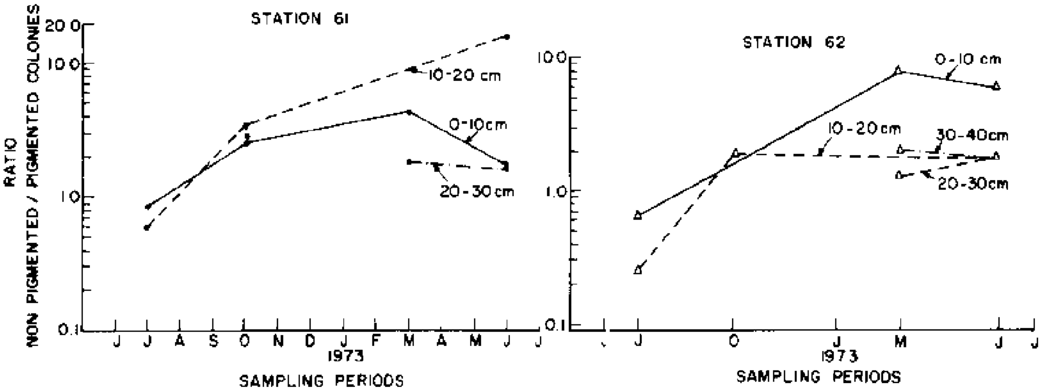

Fig. 5. Fluctuations in ratio of nonpigmented to pigmented aerobic colony forming units at stations 61 and 62, July 1972 through June 1973.

Table 3. Percent of aerobic isolates capable of utilizing the designated nitrogen sources.

Depth in core (cm)	KNO₃					$(NH_4)_2SO_4$					Urea				
	Jul 1972	Oct 1972	Jan 1973	Mar 1973	Jun 1973	Jul 1972	Oct 1972	Jan 1973	Mar 1973	Jun 1973	Jul 1972	Oct 1972	Jan 1973	Mar 1973	Jun 1973
Station 2															
0–10	67,50	36	27	80	—*	72,50	40	37	79.5	—	66,49	29	33	79.5	—
Station 15															
0–10	51	68	81	71	C†	13	51	76	71	C	61	61	74	76	C
10–20	—	0	—	99	27	—	27	—	99	22	—	96	—	99	50
20–30	—	85	—	47	0	—	77	—	47	—	—	71	—	47	—
Station 61															
0–10	76	50	—	86	99	C	44	—	82	99	C	99	—	84	99
10–20	85	30	—	—	81	62	28	—	—	83	77	39	—	—	83
Station 62															
0–10	67	67	—	96	39	62	67	—	96	39	58	71	—	98	52
10–20	50	79,75	—	—	56	86	81,75	—	—	40	26	85,77	—	—	69

* Data not obtained.
† Contaminated plates.

When the total CFU and the ureolytic CFU were plotted as a function of sedimentary urea content (values from Nakas and Litchfield in prep.), at low urea concentrations of $\lesssim$10 μg urea/g dry wt an inverse relationship apparently exists (Fig. 6), indicating limits of bacterial numbers but also allowing for a wide degree of variance within the data pairs. It would appear, then, that as the urea level reaches concentrations greater than 10–30 μg/g, the population is restricted to a narrower range of 10^5 to 10^6 CFU/g, whereas at the low urea concentrations greater fluctuations can exist, 10^3 to 10^8 CFU/g (Fig. 6).

When all the data were subjected to multivariant analysis, the overall correlation was extremely poor due to the wide variances in the data from the different stations. However, when the data for each

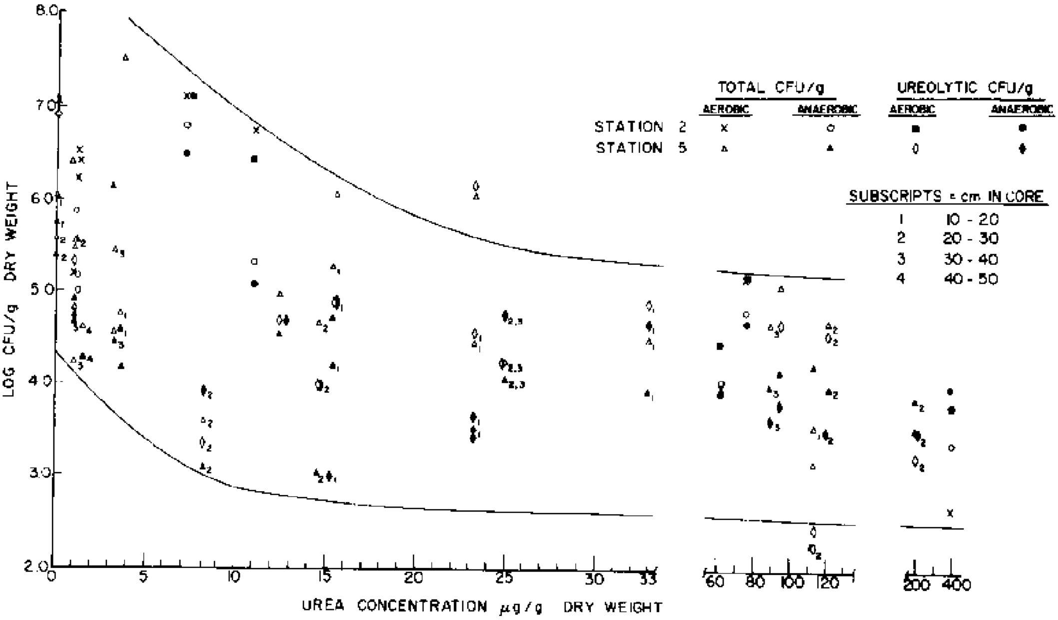

Fig. 6. Effect of in situ urea concentration on the bacterial population at stations 2 and 15, July 1972 through December 1974. Composition of the isolation media is listed in Table 1.

station and sediment section were used, a correlation matrix could be developed (Table 4). As Whitfield (1969) has demonstrated, unless special precautions are taken to ensure reproducibility in *Eh* and pH measurements, these measurements for estuarine sediments are only qualitatively reflective of the general sediment condition. For this reason, pH and *Eh* measurements were not included in the regression analyses. Even a cursory examination of Table 2 data and the seasonal plots of CFU reveals no correlation between lowered redox potentials and the proportion of facultatively anaerobic bacteria at any of the study areas. Whitfield also reported a generally positive redox potential within the first 3–4 m in the sediments, which is similar to our findings in this bay system (Table 2).

Table 4. Correlation matrices using sampling date as dependent variable.

	%*	C:N*	U*†	NP:P*	CFU*	D*
Station 2 0–10 cm‡ $r = 0.850$						
%	1.00					
C:N	0.62	1.00				
U	−0.12	−0.36	1.00			
NP:P	−0.11	−0.32	0.10	1.00		
CFU	−0.10	0.24	−0.75	−0.47	1.00	
D	−0.09	−0.03	0.56	0.39	−0.39	1.00
Station 15 0–10 cm‡ $r = 0.825$						
%	1.00					
C:N	−0.18	1.00				
U	0.14	−0.20	1.00			
NP:P	0.06	−0.19	0.98	1.00		
CFU	0.19	0.62	−0.06	−0.18	1.00	
D	−0.16	0.09	0.47	0.40	0.47	1.00
Station 15 10–20 cm‡ $r = 0.994$						
%	1.00					
C:N	−0.05	1.00				
U	0.09	0.56	1.00			
NP:P	−0.28	0.39	0.88	1.00		
CFU	−0.13	−0.49	−0.61	−0.53	1.00	
D	−0.48	0.76	0.48	0.62	−0.51	1.00
Station 15 20–30 cm‡ $r = 0.872$						
%	1.00					
C:N	−0.54	1.00				
U	0.48	−0.18	1.00			
NP:P	−0.35	−0.22	0.12	1.00		
CFU	−0.34	−0.10	−0.47	0.39	1.00	
D	0.63	−0.52	0.46	−0.41	−0.40	1.00
Station 61 0–20 cm‡ $r = 0.902$						
%	1.00					
C:N	0.13	1.00				
U	−0.06	0.41	1.00			
NP:P	0.45	0.04	−0.01	1.00		
CFU	0.66	0.49	0.53	0.22	1.00	
D	−0.06	0.54	0.22	−0.66	0.19	1.00
Station 62 0–20 cm‡ $r = 0.920$						
%	1.00					
C:N	−0.35	1.00				
U	0.48	0.37	1.00			
NP:P	0.63	−0.71	0.06	1.00		
CFU	0.77	−0.41	0.38	0.50	1.00	
D	−0.10	0.44	0.05	−0.69	0.13	1.00

* % = Percent moisture; C:N = carbon to nitrogen ratio; U = µg urea/g; NP:P = nonpigmented CFU to pigmented CFU ratio; CFU = total aerobic and anaerobic colony forming units/g; D = sampling date.

† Urea data taken from Nakas and Litchfield (in prep.).

‡ $p = 0.25$ for stations 61, 62, and 15; 0.01 for station 15 (10–20 cm); and 0.10 for station 2.

From the matrix (Table 4), we can note that moisture content, C:N ratio, and season have little bearing on the bacterial population in that sample. Also, the trend observed in Fig. 6 regarding an inverse relationship between the CFU/g and the urea content is confirmed by a generally negative correlation coefficient at the −0.06 to −0.75 level for stations 2 and 15; a positive relationship exists for the first 20 cm of the sediments at stations 61 and 62.

One other surprising feature was that for stations 2 and 15, down to 20-cm core depth, there was a direct and fairly high degree of correlation between the nonpigmented CFU to pigmented CFU ratio and the sampling date, although this relationship did not extend to the total CFU and sampling period. This indicates selective predation or nutrient factors controlling the bacterial flora in these areas. Just the opposite correlation existed at stations 61 and 62 and the 20–30-cm core section of station 15. Less variation in this NP:P ratio occurs at these locations than in the other samples, suggesting greater population stability. Within each matrix the overall correlation is good (Table 3), but the probability of predicting data for the stations is less reliable: $p = 0.1$ for station 2, $p = 0.01$ for station 15 at the 10–20-cm depth, and $p = 0.25$ for all other core sections and stations. This suggests, that within the upper 30 cm, the sediments and their microflora in Sandy Hook and Raritan Bays are heterogeneous and subject to multiple interactions.

Discussion

Bacterial fluctuations in the water column have been reported previously. Sieburth noted that not only did the absolute numbers vary by one–two orders of magnitude with time, but that there was an overlying flux in the ratio of flavobacteria to vibrios in Narragansett Bay waters (Sieburth 1968). This population shift could be related to the seasonal distributions of these genera, and he postulated that the flavobacteria predominated during periods of greater light intensity because of the photoprotection conferred by their carotenoid pigments (Sieburth 1968). At stations studied in this report, Secchi disk readings seldom extended below 2–3 ft, whereas the overlying water depth was 14–30 ft. Thus, it would seem unlikely that in our study area, UV light stress could explain bacterial flux in sedimentary populations. Certainly, selectivity by light could not account for the fluxes observed with depth in our sedimentary cores.

Diurnal fluxes in the bacterial population have been reported by Lloyd (1929–1930) for the Clyde Sea and by Taga (1968) for the Kuroshio Current. In neither case was the overall flux greater than two orders of magnitude. Unfortunately, no chemical or physical data were included in these reports so it is not possible to evaluate factors inducing such changes. Partial comparisons between bacterial numbers and water chemistry have been recently provided by Erkenbrecher and Stevenson (1975) who noted an order of magnitude change in the bacterial numbers and a 2–4-fold flux in the POC and ATP, respectively, for South Carolina estuarine waters during a tidal cycle. The change in POC they reported is close to the range of chemical flux reported by Alexander and coworkers during a study of the diurnal variations in the nutrient chemistry of Block Island Sound waters over a tidal cycle (Alexander et al. 1973).

There have been even fewer studies on the bacterial flux and nutrient chemistry flux in marine sediments. While investigating the seasonal rates of sedimentation, two maxima in the amounts of deposited carbon were observed by Stephens et al. (1967). They also detected a high degree of correlation between the amount of sedimented material and the total carbon. The amount of deposited nitrogen followed a similar pattern but increased pheophytin deposition was observed only in late spring. Based on particle size analysis and the seasonality of these depositions, they concluded that one portion of the sedimentary input was due to biological forces and the other to terrestrial runoff, providing two major periods of nutrient input for benthic flora (Stephens et al. 1967). Reported changes in the percent moisture, carbon content, and nitrogen content of Lake Washington sediments during the period 1958–1970 (Shapiro et al. 1971) are similar to the amount of fluctuation in these parameters reported in this paper. Unfortunately, in neither of the above studies is there any mention of the changes in the meiofauna or benthic macrofauna, which would enable us to determine to what extent the chemistry of the sediments is influencing the biological life, or vice versa.

Heterogeneity in foraminiferal populations has been reported by Schafer (1971) for the sediments in the Gulf of St. Lawrence. Variations in their distribution were found to occur at core depths of up to 10–20 cm. Whether this included a temporal shift in the populations was not discussed, although Vilks (1967) did note that seasonal as well as geographical factors were responsible for fluctuations in foraminiferal populations in sediments. Similar studies and results were obtained during investigations of annelids and echinoderms in St. Margaret's Bay, Nova Scotia (Hughes et al. 1972). Coull (1973) also demonstrated a relationship between benthic meiofauna and bacterial levels in sediments. In fact, he postulated a high degree of interaction between all components of the sediments. No attempt was made during the course of the study reported here to examine the sediments for bacteriovorous meiofauna. The data in Fig. 6, indicating a limitation of the bacterial population with increasing urea concentrations, could be explained by the increased presence of bacterial predators cropping the population to a minimal level and in turn excreting urea. The presence of such organisms could account for some variance in the data.

Recently, Boylen and Brock (1973) reported a two-order-of-magnitude difference in the total CFU for Lake Wingra sediments over a 10-day period. This level of bacterial flux is certainly comparable to what we have observed in these bay studies during a 30-month period. These data indicate the need for greater control of the sampling periodicity than is normally practiced. Indeed, these results make it imperative that more long term studies be undertaken both inshore and in deeper ocean sediments and that during these studies attention be focused on fluxes in the composition of the population as well as alterations in the nutritional characteristics of that population. Such examinations are essential if we are to gain any understanding about the role of bacteria as primary decomposers and as food sources for higher organisms in marine sediments.

In conclusion, we have demonstrated that extremely wide fluctuations in the bacterial flora of a marine sediment are not only possible but occur generally. These fluctuations extend to the overall generic composition of the bacterial flora as well as to the nutritional requirements of the microflora. Moreover, neither the total CFU flux nor the compositional or nutritional fluxes are in synchrony; and no general pattern has emerged regarding a correlation of these parameters with the usual physical and chemical properties of the sediments. The sedimentary microflora is dynamic and future studies must include composition and nutrition if the role of heterotrophic bacteria in marine sediments is to be evaluated and understood.

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Seabed oxygen consumption—New York Bight apex[1]

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Abstract

Seabed oxygen consumption rates, temperature, salinity, and dissolved oxygen were measured during five cruises in the New York Bight apex between March 1974 and August 1975. The area sampled included the waste disposal sites for sewage sludge, dredge spoils, and industrial acid wastes. Samples were collected and incubated on shipboard at in situ temperature during oxygen uptake measurements.

Total oxygen consumption by the seabed ranged from 1–68 ml O_2 m^{-2} h^{-1}. An average of 444×10^6 liters O_2 (183 tonnes C) are estimated to be consumed by the apex (1,577 km^2) seabed each day ranging from 288×10^6 in February to 689×10^6 in August. In winter the highest rates of uptake were measured in the Christiaensen Basin adjacent to the sewage sludge disposal site, in the topographically high dredge spoil disposal area west of the Christiaensen Basin, and in the Hudson Shelf Valley. In summer the highest rates were measured in the dredge spoil area. Rates in the Christiaensen Basin, however, were low compared with the surrounding areas and were more like winter rates. This difference may have been caused by differential sedimentation rates of oxidizable organic carbon to the seabed, mediated by the presence or absence of a thermocline. The highest rates were measured near a municipal sewage outfall off Asbury Park, N.J. No discernible effects on seabed oxygen consumption were observed near the acid waste disposal area.

Rates of oxygen uptake by the bottom water and by the entire water column were measured and compared with oxygen uptake rates by the sediment. Most (93%–98%) oxygen uptake in the apex occurs in the water column and not on the seabed.

Consumption of oxygen by the seabed has been used as a measure of benthic community metabolism to understand energy flow and carbon cycling in marine ecosystems (e.g. Pamatmat 1971*a*; Smith et al. 1972; Carey 1967; Kanwisher 1962; Teal and Kanwisher 1961). It has also been used to indicate the oxidation of organic matter and effect of organic pollution on benthic communities (Pamatmat et al. 1973; Pamatmat 1971*b*; Smith et al. 1973, 1974; Baity 1938) and to study rates of nutrient regeneration by the benthos (Rowe et al. 1975). These studies demonstrated the importance of understanding these rates and processes when considering use and management of natural resources and protection of the marine ecosystem.

The New York Bight apex receives wastes from disposal of sewage sludge, dredge spoils, and industrial acid wastes as well as waste input from the Hudson River and other lesser sources originating from the activities of about 15 million people living in the New York–New Jersey coastal zone (U.S. EPA 1975; Mueller et al. 1975). These wastes (except acid wastes) contain oxidizable organic material (Smith et al. 1974; NMFS 1972; Gross 1972; Horne et al. 1971) which upon oxidation consume oxygen.

Our study measured seabed oxygen consumption rates throughout the apex and examined the temporal and spatial distribution and magnitude of these rates to determine effects of ocean dumping on the benthic community metabolism, ability of the benthos to assimilate organic wastes, and the relative significance of seabed oxygen consumption in terms of total energy flow and the carbon cycle.

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Methods

Seabed oxygen consumption rates (i.e. oxygen uptake rates of both the bottom sediment and the bottom 5–15-cm (avg 12 cm) layer of the water column) and related bottom water hydrographic measurements (temperature, salinity, and dissolved oxygen) were obtained during five cruises in the apex (Fig. 1) between March 1974 and August 1975. Navigational control was accomplished using Raydist (Teledyne Hastings Raydist, Hampton, Va.). With few exceptions, all sampling took place within 0.5 km of the station point. Samples for seabed oxygen consumption were collected with a Pamatmat multiple corer and incubated on shipboard at in situ temperature (Pamatmat 1971*a*; Pamatmat et al. 1973). Four cores, occasionally replicates from the same cast, of sediment plus overlying bottom water were collected from each of approximately 60 stations per cruise except where coring was difficult.

A 6-liter Niskin water bottle with reversing thermometer was attached to the side of the multiple corer (Pamatmat 1971*a*; Pamatmat et al. 1973) to record temperature and collect water 20 to 50 cm above bottom for salinity and dissolved oxygen determinations. Salinity was determined using a Beckman model RS-7C salinometer. Dissolved oxygen concentrations were measured using the azide modification of the iodometric method (Am. Public Health Assoc. 1965) except that thyodene was used in place of starch and 0.025 N phenylarsine oxide (PAO) was used in place of sodium thiosulfate (Kroner et al. 1964; U.S. EPA 1974). In situ percent saturation of oxygen was computed from the equations of Weiss (1970).

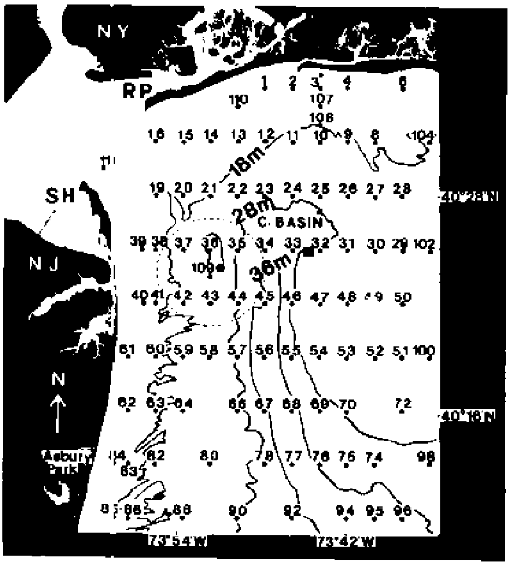


Fig. 1. SYMAP station numbers and locations of stations (dots) sampled for measurements of seabed oxygen consumption rates. Dredge spoil disposal site (large dot) is near station 109 at the center of encircled area. Encircled area affected by dredge spoils, but not synonymous with dredge spoil disposal site. Dredge spoil disposal site is defined as the area within a circle with a radius of 0.6 nmi about 40°24'N and 73°51'W (large dot). Hatched area bounded by 28-m-depth contour and stations 55, 56, 57 is designated Christiaensen Basin. Northwest corner of sewage sludge disposal site is black square (between stations 32 and 33). Sewage sludge drop area is between 40°22.30' and 40°25'N and 73°41' and 73°45'W (U.S. Environ. Prot. Agency, Edison, N.J.). RP—Rockaway Point; SH—Sandy Hook.

Equipment, materials, and procedure for measuring oxygen uptake rates are described in Pamatmat (1971*a*) and Pamatmat et al. (1973). Experiments lasted 6 h in summer and 12 in winter. Oxygen uptake rates measured outside the range of ±3.0°C from in situ for that station were not considered.

About 20% of the stripcharts from the August 1974 cruise were examined (BMD 02R regression program: Dixon 1973) and the change per time was linear in most cases. Consequently, initial and final oxygen concentrations determined in samples of water above the cores at the beginning and end of the experiment were used to calculate seabed oxygen consumption rates. These data are presented elsewhere (Thomas et al. 1976). Average coefficient of variability per station for rates of seabed oxygen consumption was 42.3% ($n = 318$ stations, 95% confidence limits, 39.6–45.0% using square-root transformation). This high value is due in part to spatial heterogeneity of the bottom observed by K. L. Smith (personal communication).

The sediment from all but one of the cores was sieved through a 1.0-mm-mesh screen and organisms recovered were noted and preserved. Sediment from one of the cores was stored frozen for carbon analyses (to be reported elsewhere).

At selected stations bottom water collected with a Niskin bottle was placed in two core tubes without sediment to determine oxygen consumption attributable to bottom water alone. During August 1975 at stations 34, 109, and 51 four water samples per depth were so incubated for about 6 h at in situ temperature to determine oxygen uptake in the water column. Also at selected stations samples of both sediment and water and water alone were investigated using Formalin to determine chemical oxidation rates (to be reported elsewhere).

Distributional maps and isograms for seabed oxygen consumption, dissolved oxygen concentrations, percent saturation of oxygen, temperature, salinity, and sigma-*t*, were generated using SYMAP program, version 5 (Harvard Center for Environmental Design Studies, Laboratory for Computer Graphics and Spatial Analysis).

Results

Temperature, salinity, and dissolved oxygen—During the cruises of March and December 1974, and February 1975, no thermocline was present. Bottom water temperatures (Fig. 2 T1,T3,T4) were uniform throughout the apex being only slightly warmer in the deeper areas. Temperatures ranged from 10°C in December to 5°C in February and March. Bottom water salinity (Fig. 2 S1,S3,S4) ranged from 30.00‰ to nearly 34‰ being lowest near the Hudson River entrance and highest in the deeper water of the apex (30–40 m in the Christiaensen Basin and upper Hudson Shelf Valley). Bottom water oxygen concentrations ranged from 8.00–10.80 ppm (Fig. 2 D1,D3, D4). Percent oxygen saturation of bottom water exceeded 86% throughout the apex except for a small area off East Rockaway Inlet with a value of 64%.

The cruises of August 1974 and 1975 are representative of summer conditions when a thermocline is present between 10- and 25-m depth. The thermocline acts to isolate bottom water from renewal of oxygen from both the atmosphere and primary production. Bottom water temperatures (Fig. 2 T2,T5) ranged from 8°C in the deeper water to 24°C in the shallow water. Salinities (Fig. 2 S2,S5) ranged from 28–33‰, being slightly lower than in winter. The lowest salinities occurred near the Hudson River entrance and down the New Jersey coast. The highest values were associated with the deeper areas of the apex.

Bottom water dissolved oxygen concentrations fell to 0.90 ppm in August 1974 (Fig. 2 D2). Low values (<4 ppm) occurred along a broad path extending from near the Shark River Inlet, N.J., below Asbury Park toward Atlantic Beach, L.I. Dissolved oxygen concentrations in bottom water during August 1975 (Fig. 2 D5) were not as low (3.09 ppm) as during the previous August. In August 1975, the lowest values occurred in the central or deeper parts of the apex and the highest values (up to 7.94 ppm) occurred in the shallower and warmer waters along the New Jersey and Long Island coasts.

Percent oxygen saturation (Fig. 3 P2) of bottom water in August 1974 ranged from 13–120%. The highest values occurred along the Long Island coast. During August 1975, percent saturation of oxygen in bottom water (Fig. 3 P5) ranged from 38–110%. The lowest values occurred at the northern edge of the Christiaensen Basin. Values less than 70% were generally associated with the Christiaensen Basin and the Hudson Shelf Valley, the deeper areas of the apex. The higher values occurred in the shallow water along the coast. The most

Table 1. Percent area of total apex (1,577 km^2) with corresponding values of various levels of percent saturation of bottom water dissolved oxygen for cruise D7409, 26 August–6 September 1974, and cruise D7512, 12–25 August 1975.

Bottom water DO (% saturation)	% Area of total apex*	
	Aug 74	Aug 75
<30	18	0
<50	48	2
<70	78	31
<90	90	89
<110	97	100
<120	100	100

* Only 76.6% of the total apex area was sampled during August 1975.

notable differences between August 1974 and 1975 are the much lower percents of saturation and the much greater area affected by saturation values less than 50% during August 1974 (48% of the apex) compared to August 1975 (2% of the apex) (Table 1).

Seabed oxygen uptake—In March 1974 (Fig. 2 C1) seabed oxygen consumption rates were low (2–20 ml O_2 m^{-2} h^{-1}), except for two areas. The larger area was centered over the Christiaensen Basin and upper Hudson Shelf Valley; the second area was near Asbury Park, N.J., where the highest rates occurred nearshore and decreased seaward. The Asbury Park, N.J., municipal sewage outfall (40°13′14″N, 73°59′52″W) which discharges 1.32×10^4 m^3 per day of primary sewage effluent with a BOD of 120 mg/liter (U.S. Environ. Prot. Agency, Water Facilities Branch, New York, N.Y.) is near this second area. No discernible effects on seabed oxygen consumption were observed near the acid waste disposal area (between 40°16′N and 40°20′N, and 73°36′W and 73°40′W) during the study.

In August 1974 (Fig. 2 C2) in apparent response to higher water temperatures, seabed oxygen consumption rates were generally higher throughout the apex (2.82–68 ml O_2 m^{-2} h^{-1}) except for the Christiaensen Basin and upper Hudson Shelf Valley just to the east of the dredge spoil disposal area. Oxygen consumption rates in this area were depressed (<10 ml O_2 m^{-2} h^{-1}) and more typical of winter conditions. The highest rates (40 ml O_2 m^{-2} h^{-1}) occurred on the periphery of the depressed area and decreased with distance away from it. A second area of elevated rates occurred off Asbury Park and included the highest values measured (up to 68 ml O_2 m^{-2} h^{-1}).

In December 1974 (Fig. 2 C3) the highest values again occurred at the dredge spoil area and eastern Christiaensen Basin (25 ml O_2 m^{-2} h^{-1}). In general, rates in the apex approached wintertime distribution and magnitude.

The distribution and magnitude of rates measured in February 1975 (Fig. 2 C4) mimicked those of December 1974. Major differences were the rates in both the dredge spoil and sewage sludge disposal areas in February (16 ml O_2 m^{-2} h^{-1}) were lower and the areas with higher rates were smaller than in December (also Fig. 4).

In August 1975 (Fig. 2 C5), apparently in response to increased water temperature, seabed oxygen consumption rates were generally higher throughout the apex (4–50 ml O_2 m^{-2} h^{-1}) except for several areas just to the east of the dredge spoil area in the Christiaensen Basin. This area exhibited depressed seabed oxygen consumption rates as in August 1974, which were more typical of winter values. The depressed area in August 1975, however, was smaller and not as depressed as in August 1974 (also Fig. 4). Again, the peripheral areas including the dredge spoil area exhibited the higher rates. In August 1975 the elevated effect of dredge spoil material on seabed oxygen consumption was located 5.5 km north (40°25′N) of the area with the highest rates in August 1974 (40°22′N). Seabed oxygen uptake along the New Jersey coast was generally higher in August 1975 than in August 1974.

Computer isographs of the distribution of seabed oxygen consumption rates for each cruise (Fig. 2 C1–C5) were integrated and the values obtained were used to construct a frequency distribution showing the percent of apex area having various rates of seabed oxygen uptake during the year (Fig. 4). During both summers over half the area (53 and 59%) of the apex had

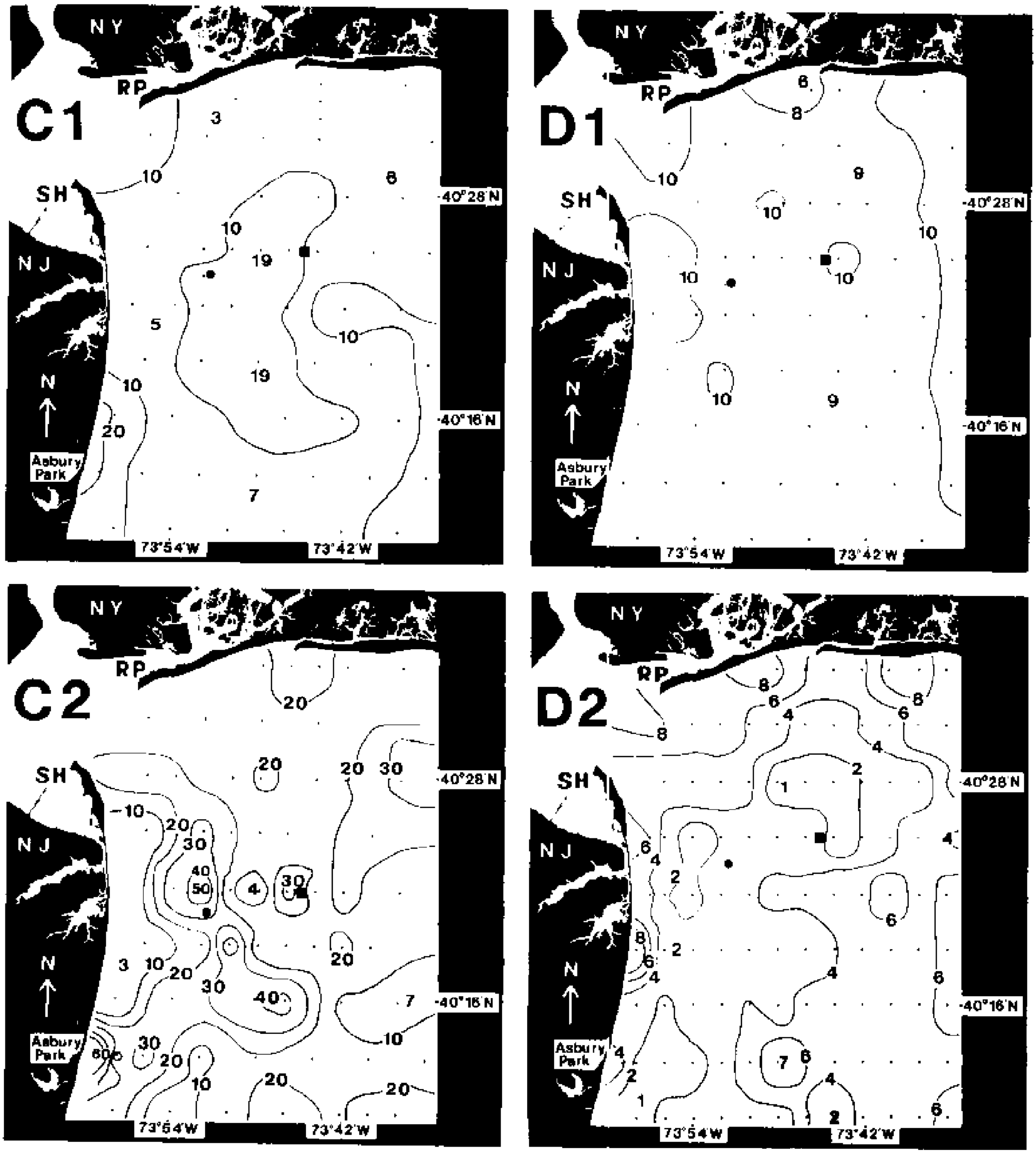
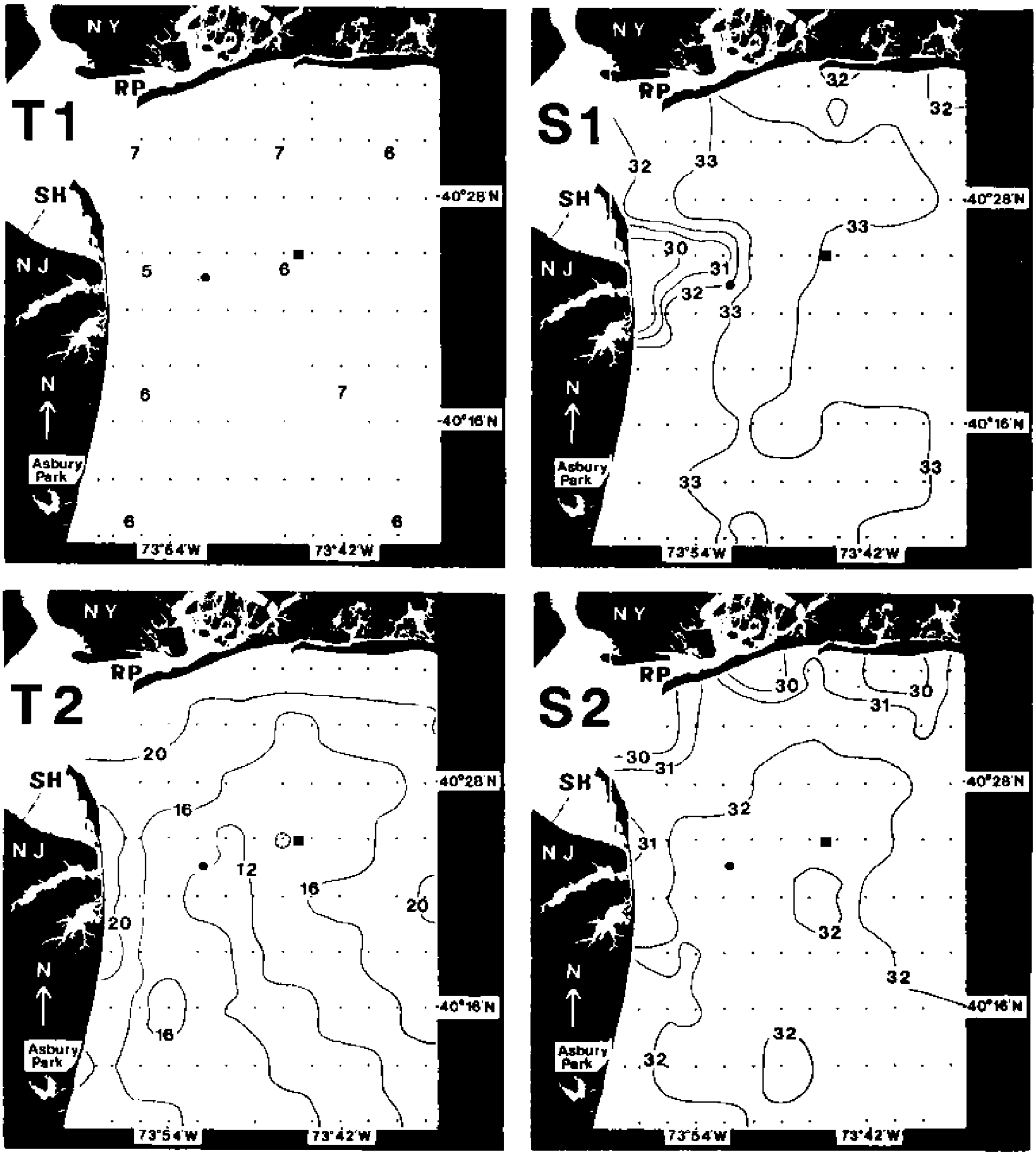


Fig. 2. Seasonal distribution of values for seabed oxygen consumption rates in ml O_2 m^{-2} h^{-1} (C1–C5), bottom water dissolved oxygen concentrations in ppm (D1–D5), bottom water temperatures in °C (T1–T5), and bottom water salinity in ‰ (S1–S5). C1, D1, T1, S1—21 March–4 April 1974; C2, D2, T2, S2—26 August–6 September 1974; C3, D3, T3, S3—2–15 December 1974; C4, D4, T4, S4—12–21 February 1975; C5, D5, T5, S5—12–25 August 1975.

Small dots—stations sampled; large dot—dredge spoil disposal site; black square—NW corner of sewage sludge disposal site. RP—Rockaway Point; SH—Sandy Hook.



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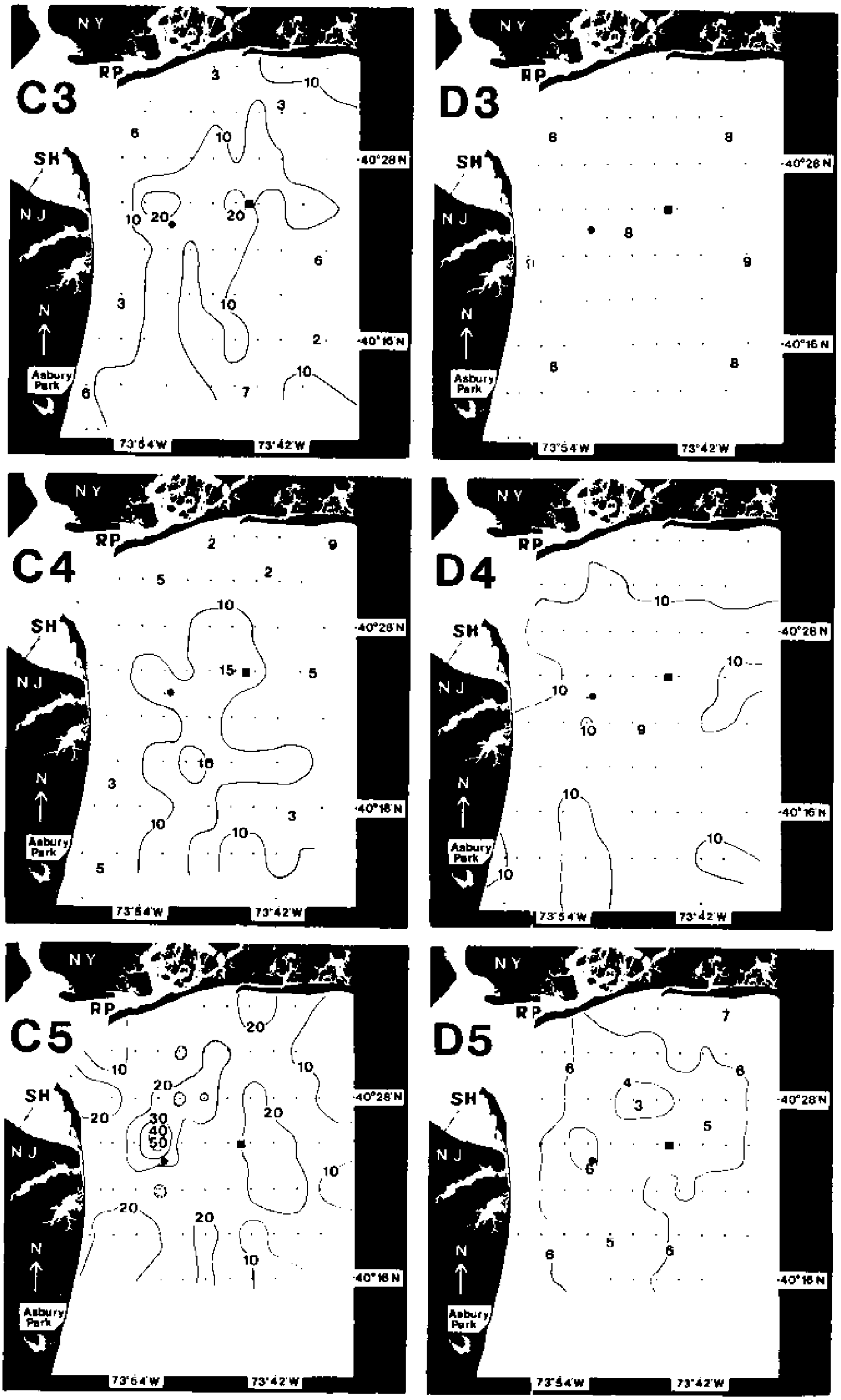

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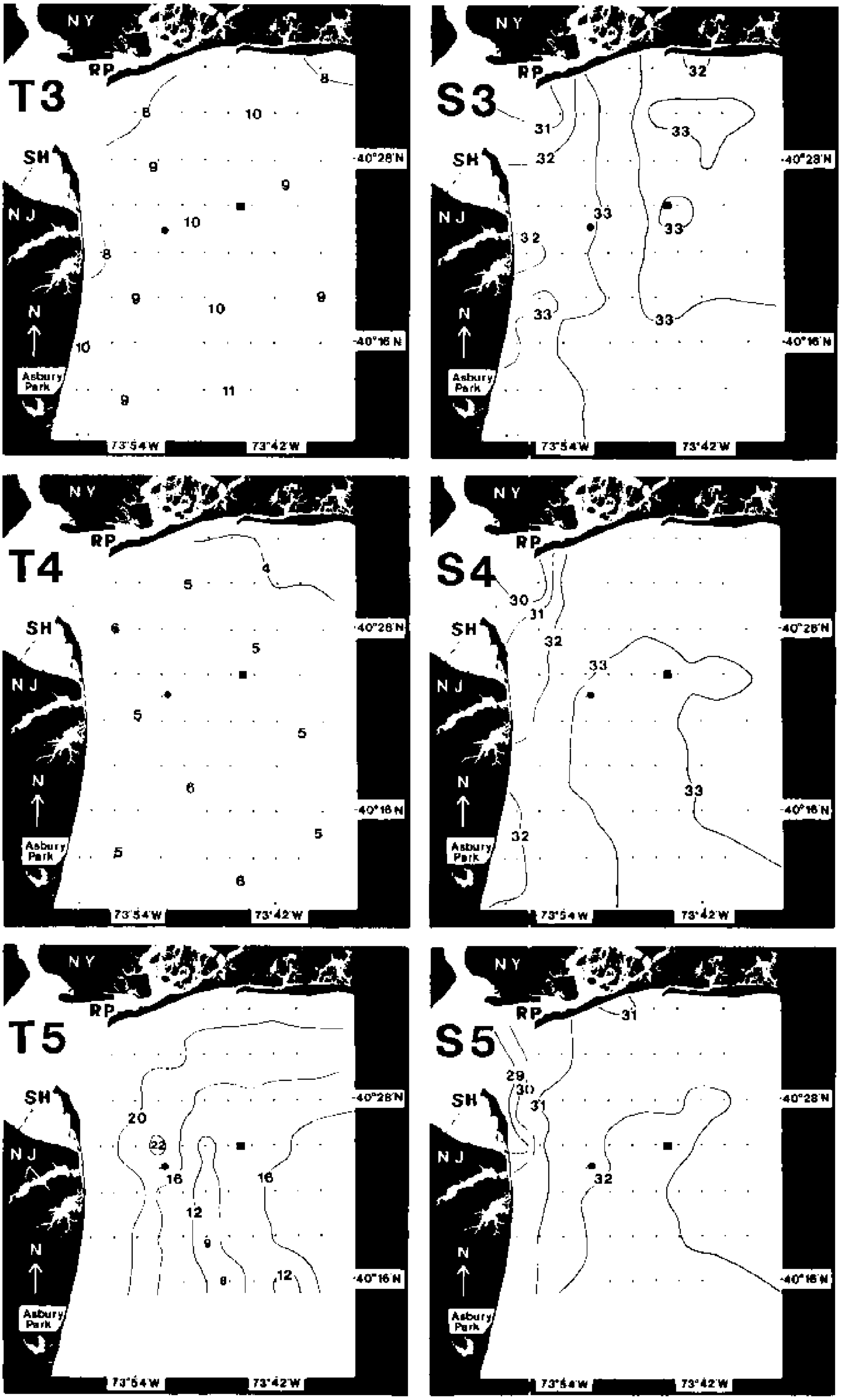

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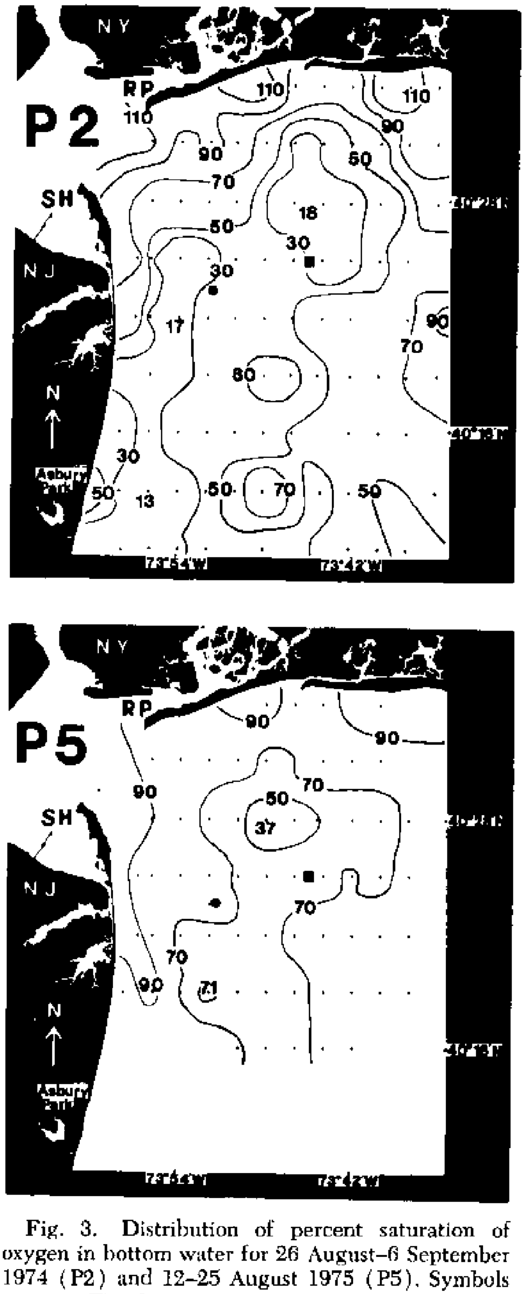

Fig. 3. Distribution of percent saturation of oxygen in bottom water for 26 August–6 September 1974 (P2) and 12–25 August 1975 (P5). Symbols same as Fig. 2.

seabed oxygen uptake rates between 10 and 20 ml O_2 m^{-2} h^{-1} (avg 18.2 and 16.6). During late fall 49% of the apex area had rates between 5 and 10 ml O_2 m^{-2} h^{-1} (avg 8.9). In winter 37% of the apex had rates between 0 and 5 ml O_2 m^{-2} h^{-1}, 31% had rates between 5 and 10, and 29% had rates between 10 and 15 (avg 7.6). In spring 60% of the apex area had rates between 5 and 10 ml O_2 m^{-2} h^{-1} (avg 9.3).

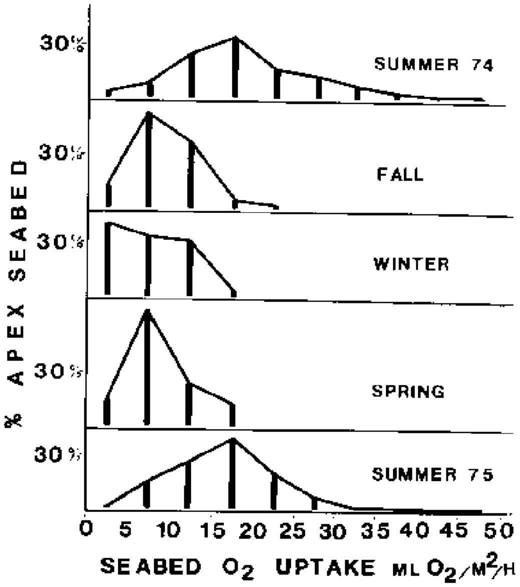

Fig. 4. Frequency distribution showing percent of apex area during the year having various rates of seabed oxygen uptake rates.

Annual curves of total seabed oxygen consumption and carbon oxidized, assuming respiratory quotient of 1.0 are shown in Fig. 5 for the entire apex (1,577 km^2) and in Fig. 6 for the Christiaensen Basin (107 km^2) and the area affected by dredge spoils

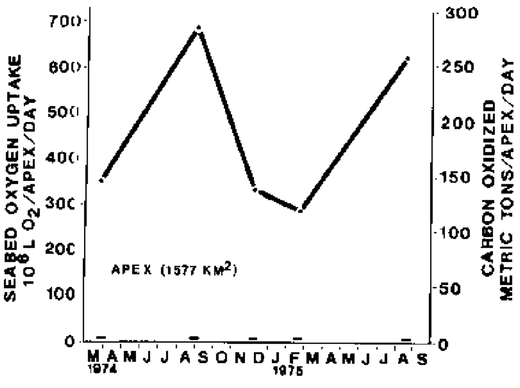

Fig. 5. Annual curve of total oxygen consumption by the seabed and equivalent carbon oxidized for the apex.

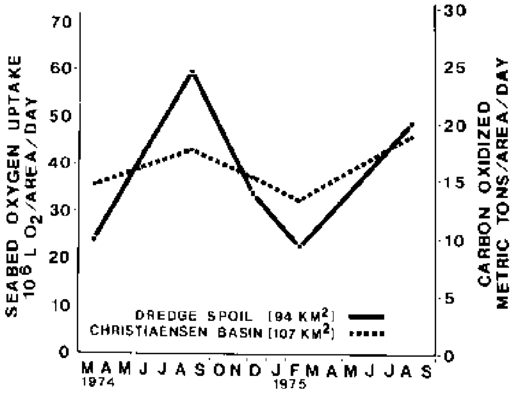

Fig. 6. Annual curve of total oxygen uptake by the seabed for the Christiaensen Basin and dredge spoil area shown in Fig. 1 (hatched and encircled areas).

(94 km²) shown in Fig. 1. The highest and lowest rates of seabed oxygen uptake were reasonably similar in both years in spite of environmental changes (i.e. bottom water dissolved oxygen concentrations). Over the annual cycle the temperature coefficient (Q_{10}) is 2.3 for the entire apex. From the integrated value of the area under this curve (Fig. 5) an average of 444×10⁶ liters of O_2 (183 tonnes C) are estimated to be consumed by the apex seabed each day. The amplitude of the annual cycle of seabed oxygen uptake rates in the Christiaensen Basin is less than half that of the dredge spoil area (Fig. 6) although they each oxidize about the same quantity of organic carbon per year—5,824 and 5,510 tonnes C. The dredge spoil area has uptake rates both higher and lower than those of the Christiaensen Basin and parallels the annual curve for the apex (Fig. 5). Uptake rates in the Christiaensen Basin approach a more constant level year-round. Temperature coefficients range between 1.7 and 3.3 for the dredge spoil area and between 1.7 and 1.9 for the Christiaensen Basin, depending on temperature. Rapid changes of at least 6°C are a normal and natural part of the system as evidenced by the change from 11°C to 17°C which occurred in the bottom water of the Christiaensen Basin between 29 August and 5 September 1974.

The percent of total oxygen consumption by the seabed (defined as bottom sediment plus the bottom 12 cm of water) due to sediment alone ranged between 0% and 100% and averaged 80.5% (73.2–86.9%, confidence limits at 95% using arcsine transformation, $n = 89$) for the entire apex over the year. The mode was very close to 100%. No distributional or seasonal pattern could be determined based on these 89 samples of bottom water taken throughout the apex during the year. The rate of oxygen uptake by bottom water was frequently near zero or negative and thus at or below the limit of sensitivity of method, suggesting that bottom water oxygen uptake rates are probably insignificant in many instances compared to the sediment.

Oxygen uptake in the water column—At three stations in the apex during August 1975 special studies were carried out, including measurements of the consumption of oxygen by the water column as well as by the seabed, to gain additional information on where the major portion of oxygen consumption or organic carbon oxidation takes place. At stations 34, 109, and 51, water samples were collected with a Niskin bottle just below the surface, 0.5 m above the bottom, and at depths above, below, and within the thermocline where possible. Vertical profiles of oxygen uptake, temperature, and oxygen concentrations in the water column are presented in Fig. 7. In general, the highest rates of oxygen consumption occurred in the upper layer or above the thermocline when one was present. Only at station 34 in the Christiaensen Basin, where perhaps a nepheloid layer (Eittreim et al. 1969) near the bottom may have contributed to the oxygen consumption, do we see high values of oxygen consumption 0.5 m above the bottom. Dissolved oxygen concentrations seem to parallel oxygen consumption rates. Compared with oxygen consumption by the total water column the contribution of the sediment ranged from 1.1–4.9% while the seabed contributed 1.2–5.2% (Table 2). Obviously the bottom 12 cm of water has a much smaller metabolic activity than the sediment, but it is signifi-

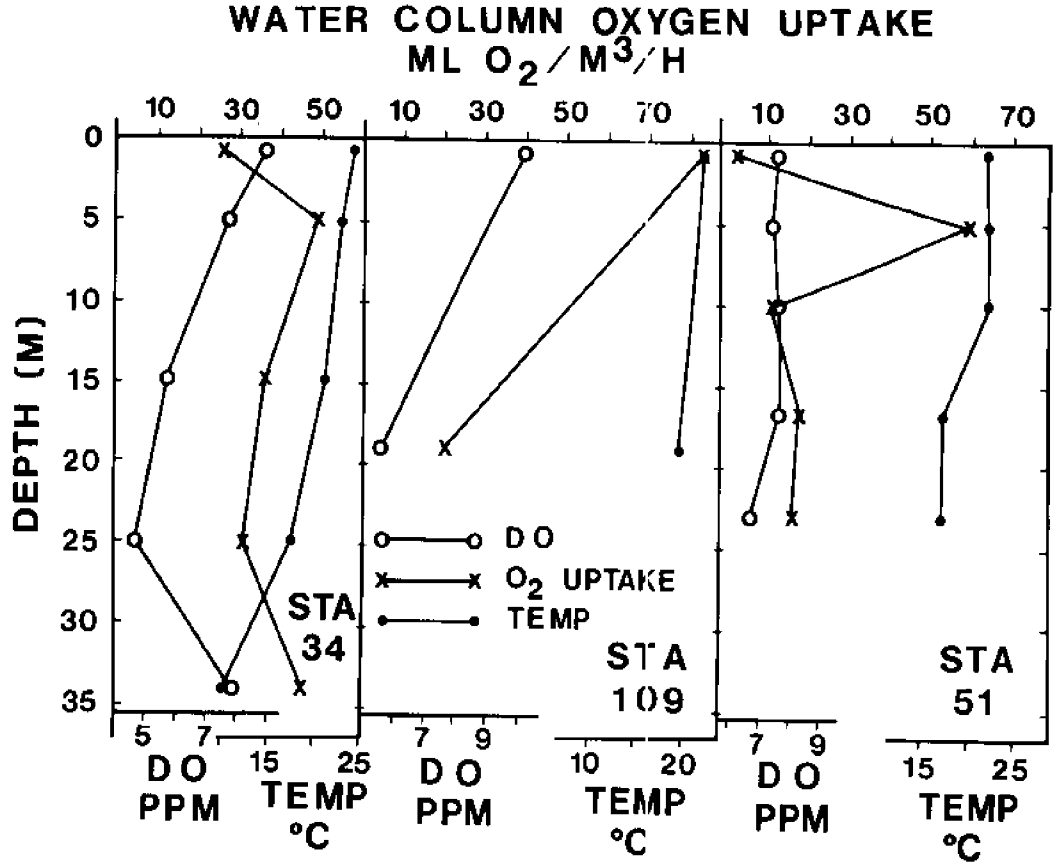

Fig. 7. Vertical profiles of total oxygen uptake, temperature, and oxygen concentrations throughout the water column, measured 20–21 August 1975 at stations 34 (Christiaensen Basin), 109 (dredge spoil area), and 51 (Cholera Bank).

Table 2. Total oxygen consumption measured throughout an entire water column plus bottom sediments at station 34 (Christiaensen Basin), 109 (dredge spoil) and 51 (Cholera Bank) 20–21 August 1975 (FRS *Delaware II*, cruise D7512).

	Sta. 34	Sta. 109	Sta. 51
A. Seabed and bottom water layer (ml O_2 m^{-2} h^{-1})			
Average core water height	8 cm	10 cm	11 cm
Total seabed (sediment and water) (M)*	26.7	27.1	11.5
Bottom water layer (M)*	3.5	1.7	0.9
Sediment (D)†	23.2	25.4	10.6
Percent total seabed due to sediment	86.8%	93.6%	92.6%
B. Integrated (measured) column (ml O_2 m^{-2} h^{-1})			
Water in column (M)*	1,293.0	495.0‡	951.0
Sediment (D)†	23.2	25.4	10.6
Total column	1,316.2	520.3	961.6
Percent total column due to sediment	1.8%	4.9%	1.1%
Percent total column due to seabed	2.0%	5.2%	1.2%

* Equals actual measured values.
† Equals value calculated by difference.
‡ Represents only one sample.

Table 3. Temperature coefficients determined at stations 34 (Christiaensen Basin), 109 (dredge spoil), and 51 (Cholera Bank), 20–21 August 1975.

Sta.	Total seabed oxygen uptake (ml O_2 m^{-2} h^{-1}) ± SD at experimental temperatures. Numbers of replicates (n)						Temp coefficient Q_{10}*
34	5.0°C	(n)	11.5°C†	(n)	21.0°C	(n)	
	17.33±11.11	(4)	26.85±4.69	(4)			2.0
			26.85±4.69	(4)	34.21±2.42	(4)	1.3
	17.33±11.11	(4)			34.21±2.42	(4)	1.5
109	9.0°C	(n)	19.5°C†	(n)			
	17.44±7.28	(4)	27.06±10.44	(8)			1.5
51	5.3°C	(n)	15.0°C†	(n)			
	1.73±1.79	(3)	3.85±2.77	(3)			2.3

* $Q_{10} = \left[\frac{\text{Rate at 11.5°C}}{\text{Rate at 5.0°C}}\right]\left[\frac{10°\text{C}}{11.5°-5.0°\text{C}}\right]$.

† Sea bottom temperature at that station.

cant in contrast to other areas (Pamatmat and Banse 1968).

Q_{10} of seabed oxygen uptake—Seabed oxygen consumption at stations 34, 109, and 51 was determined during August 1975 at different temperatures, including in situ, to determine temperature effects on the system. The temperature intervals were selected to include the annual range in the apex. The temperature coefficients (Q_{10}) were determined by averaging replicate samples at each temperature and ranged between 1.3 and 2.3 (Table 3).

Discussion

Comparison with other studies—Total oxygen consumption rates by the seabed in the New York Bight apex ranged from 1–68 ml O_2 m^{-2} h^{-1}, comparable to other coastal marine environments experiencing enrichment of the seabed. Carey (1967) measured rates of 3–70 ml O_2 m^{-2} h^{-1} in Long Island Sound, Pamatmat (1971*b*) found rates of 4–56 for Puget Sound, Smith et al. (1974) measured rates of 11.1–53.5 for two stations in the New York Bight apex, and Smith (1973) measured somewhat higher rates (53.9–92.7) for the nearshore coastal seabed off Georgia, a warmer area relatively unaffected by pollution but highly enriched organically (Thomas 1966). As a comparison, the highest known oxygen consumption rates are those measured by Odum and Odum (1955) for a coral reef community (746 ml O_2 m^{-2} h^{-1}). The lowest seabed oxygen uptake rates (0.5 ml O_2 m^{-2} h^{-1}) were measured by Smith and Teal (1973) at a depth of 1,850 m on the continental slope south of New England.

Studying the effects of organic enrichment on the benthos, Smith et al. (1973) investigated seabed oxygen uptake with an in situ bell jar technique both near the Woods Hole sewage outfall and at a control station in Buzzards Bay. They found elevated rates (67.8 ml O_2 m^{-2} h^{-1}) near the outfall compared with the control station (54.6 ml O_2 m^{-2} h^{-1}). Smith et al. (1974) also compared two areas widely separated geographically and affected by different kinds of organic enrichment. One area, off Baja California, was affected by natural enrichment caused by upwelling. The other area, in the New York Bight apex, was affected by organic enrichment from ocean waste disposal, estuarine runoff, and in situ production. They found little difference between the two areas: Baja California—36.7–39.1 ml O_2 m^{-2} h^{-1}; the New York Bight apex—37.4. In the New York Bight their sewage sludge station "A" (40° 26′N, 73°49′W) was located in the Christiaensen Basin to the north of our stations 33 and 34 and their control station "B" (40°24′N, 73°45′W) was located on Cholera Bank to the south of our station 32. None of their sampling times (April, July, and October 1971, and January and April 1972) overlapped ours, and, because of the relatively large changes in bottom water temperature and seabed oxygen consumption rates over short horizontal distances, only crude comparison can be made. In general, the two studies are probably in agreement even though we did not find as pronounced a seasonal cycle in the Christiaensen Basin (13.4–18.9 ml O_2 m^{-2} h^{-1}).

Our hydrographic data appear to be in good agreement with those of Ketchum et al. (1951) and the National Marine Fisheries Service (1972) who previously investigated the oceanography of the New York Bight. Our contribution is the increased density of sampling, especially at 0.5 m above the seabed within the apex. These data have demonstrated the occurrence and distribution of extremely low oxygen concentrations, lower than those previously reported by Ketchum et al. (1951).

Apex organic carbon budget—We have constructed an organic carbon budget for the New York Bight apex (Table 4). The value for primary production in the apex (370 g C m^{-2} yr^{-1}) was obtained from Malone (1976). The transect value was obtained from Mueller et al. (1975) based on their estimate of 1,500 tonnes of total organic carbon entering the apex each day across the Sandy Hook-Rockaway Point transect. The value for dredge spoils was also calculated from Mueller et al. (1975) using their values of 7×10^6 yd^3/yr and their bulk density estimate of 1.3 g/cm^3. The quantity of dredge spoils may vary considerably from year to year based on permits issued. Organic carbon in dredge spoil materials was estimated at 2% (Gross 1972). The sludge input to the system was calculated from Mueller et al. (1975) based on their value of 5.59×10^6 yd^3/yr. The solids were figured at 3.2% (Callaway et al. 1976). Organic carbon in sewage sludge solids was estimated at 35% based on analyses of samples of digested sludge from Newtown Creek and Tallman's Island treatment plants (Smith et al. 1974). This value is in close agreement with Duedall et al. (1975). Other organic imports by way of ocean outfalls or the atmosphere are minor (Mueller et al. 1975) and were ignored. The values of sediment oxidation and water column oxidation rates are based on this study.

Table 4. An organic carbon budget for the New York Bight apex demonstrating the relationship between input and loss components.

	Apex (1,577 km^2)	
	Bulk 10^6 m^3/yr	Organic C* 10^4 tonnes/yr
Inputs†		
1. Primary production		58.3
2. Transect		54.8
3. Dredge spoils	5.35	13.9
4. Sewage sludge	4.27	8.6
Total		135.6
Losses‡		
5. Seabed oxygen uptake (including bottom 12 cm water)§		6.7
6. Sediment oxygen uptake (80.5–94%)		5.4–6.3
7. Water uptake (bottom 12 cm of column, 6–19.5% of seabed)		0.4–1.3
8. Water uptake (bottom m^3 of column)		3.3–10.8
9. Water column uptake (24 m avg depth)		79.7–260.0
10. Total column (sediment plus water)		86.0–265.4
11. Percent total column due to sediment		2.0–7.3

* 1 ml O_2 = 0.412 mg organic carbon.

† 1. Malone (1976). 2. Mueller et al. (1975). 3. Mueller et al. (1975); Gross (1972), percent total organic carbon. 4. Mueller et al. (1975); Callaway et al. (1976), percent solids; Smith et al. (1974), percent organic carbon.

‡ 5–11. This study.

§ Average core water height.

Most oxidation takes place in the water column and not on the seabed. On the seabed itself, the bottom water has significant metabolic activity, probably arising in part from resuspension of particulate matter on the bottom.

Explanation of observed seabed oxygen uptake rates—Total oxygen consumption rates by the seabed "are evidently an integrated measure of the combined effects of temperature, oxygen tension, and size of the community, which in turn is determined by the amount of oxidizable organic carbon in the sediment and the flux of oxidizable organic matter to the bottom" (Pamatmat 1973, p. 367). In addition, the quantity and degree of toxicity of contaminants present in the sediment as well as the flux of these contaminants to the bottom may affect either the size and kind of community present or their oxygen uptake rates. Within such a framework we will attempt to explain the distribution and magnitude of seabed oxygen uptake rates we observed.

The coarser grained sediments along the New Jersey and Long Island coasts have seabed oxygen consumption rates which are low both summer and winter, except for the area off Asbury Park, N.J., which had the highest rates we measured in the apex in March and August 1974. These areas seem highly responsive to temperature and appear influenced by the flux of oxidizable organic carbon to the seabed.

The dredge spoil area appears as a relatively distinct and easily identifiable area based on seabed oxygen consumption rates. The rates there are elevated all year compared with the rest of the apex outside the Christiaensen Basin and are about twice as great in summer as in winter. The Christiaensen Basin has a nearly constant rate of oxygen uptake throughout the year, being equal to or slightly higher than the dredge spoil rate in winter and much less than it in summer.

Smith (1973) demonstrated that microbial populations dominate benthic community metabolism. There seems to be a correspondence between the distribution and abundance of coliform bacteria and seabed oxygen consumption rates in the apex (J. Graikoski personal communication). Previous data (NMFS 1972) show extremely high concentrations of sediment coliforms in the Christiaensen Basin and to a lesser degree high concentrations in the dredge spoil area.

Organic carbon in apex sediments is generally less than 4–5% of the dry weight of the sediment (Gross 1972; Smith et al. 1974; Hatcher and Keister 1976), which is surprisingly low considering the quantity of organic loading to apex waters. With such loading one would expect elevated rates of oxygen uptake to occur in both the dredge spoil area and the Christiaensen Basin year-round. However, we find elevated rates for the Christiaensen Basin only in winter. Preliminary data from 40 sediment samples suggest that sediment organic carbon concentrations may be higher in winter than in summer. Whether this net buildup is solely the result of slower biological oxidation processes in winter or the result of the combination of both physical processes (sedimentation rates) and biological processes together is uncertain. Visually, however, the sediments in winter are less cohesive and have a thicker oxidized layer at the sediment–water interface than in summer. As a consequence the picture is still not clear.

Portions of the Christiaensen Basin are colder in summer than other parts of the apex, yet have unexpected rates of oxygen uptake that are equal to or exceed those of the warmer areas of the apex. In addition, station 34 shows lower rates of uptake in summer than in winter when bottom temperatures are much lower.

Dissolved oxygen concentrations in the bottom water of the apex were quite different during the summers of 1974 and 1975. During August 1974 significant portions of the apex had dissolved oxygen concentrations of 2 ppm (25% saturation) or less (Table 1). During August 1975 dissolved oxygen concentrations exceeded 2 ppm yet oxygen uptake rates over the entire apex for the two summers were nearly identical. In fact the August 1974 rates were higher than for August 1975 for the dredge spoil area and only slightly lower for the Christiaensen Basin in spite of the extremely low dissolved oxygen concentrations during August 1974. These concentrations are the result of the rates of replenishment and utilization. We know only the rate at which oxygen is utilized. We require additional information on circulation and vertical mixing within the apex to determine rates of replenishment.

Heavy metal concentrations in sediments of both areas were about the same (Carmody et al. 1973; Greig and Thomas unpublished data), yet rates of oxygen uptake for the two areas were very different. Koditschek and Guyre (1974) have shown that microbial populations in these areas are frequently resistant to antibiotics and heavy metals. Thus neither bottom temperature, nor dissolved oxygen concentrations, nor heavy metal concentrations in the sediments explain the observed seabed oxygen consumption rates for the dredge spoil and Christiaensen Basin areas.

We believe that the combination of the size of the benthic community, the amount of oxidizable organic carbon in the sediment, and the flux of oxidizable organic matter to the bottom may regulate seabed oxygen consumption in the Christiaensen Basin to produce the observed depression in summer. Pamatmat (1971*b*) stated that the flux of labile organic matter to the benthos is really unknown. The relationship between organic carbon in the sediment and seabed oxygen consumption is often poor.

The flux of oxidizable organic carbon to the bottom may partly control seabed oxygen consumption in the Christiaensen Basin. Elevated summer rates of oxygen uptake in the dredge spoil area, but not in the Christiaensen Basin, may be indicative of differential sedimentation rates of labile organic carbon in the two areas while equal rates of oxygen uptake in winter could indicate nearly equal sedimentation rates in both areas. The seasonal difference in sedimentation rate may be occasioned by the presence or absence of a thermocline which could function as a density barrier for suspended particulate material. The major portion of the dredge spoil materials (bulk density 1.3 g/cm^3) should sink rapidly through the thermocline, if there is one, providing large quantities of organic carbon to the seabed and thus possibly accounting for the higher rates of seabed oxygen uptake in the area during summer. Sewage sludge, however, has a bulk density (1.014 g/cm^3) less than that of seawater and a solids density of 1.8 g/cm^3 (Callaway et al. 1976), considerably greater than that of apex water. Consequently the sludge is fractionated into floating, suspended, and sinking components. A portion of the floating component is visible. The fate and distribution of the other two fractions is uncertain. The sediments under the designated sludge disposal site appear relatively clean and nonreducing whereas sediment in the Christiaensen Basin, immediately to the north and west of the site, appears to be affected by sludge disposal (Charnell 1975). It is not known what proportion of the organic muds in the basin are of sludge, dredge spoil (down-sloughing from a topographic high), riverine, or autochthonous origin. The basin is a depositional area in the apex (Drake 1974; Freeland et al. 1976; Hatcher and Keister 1976) and sludge dropped at the site on the west part of Cholera Bank may be carried northwest by currents (Charnell and Mayer 1975) which then deposit relatively more of this organic material in the Christiaensen Basin during winter when no thermocline is present. In summer sinking rates of sludge particles may be retarded by the thermocline (K. L. Smith, J. B. Pearce, and others: personal observations) and only limited quantities of organic carbon may reach the bottom while the bulk is oxidized in the water column. Measurements of sedimentation and water column oxygen uptake rates in both areas, summer and winter, before and after dumping, might clarify areal differences in seabed oxygen uptake.

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Benthic–pelagic coupling in the New York Bight

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Abstract

Bottom sediment oxygen consumption can be used to estimate how much of the energy incorporated by the phytoplankton is utilized by the benthos, and such rates are directly related to temperature, organic matter in the sediment, availability of dissolved oxygen, and primary production in the water column. Nutrient flux out of sediments has been measured directly by incubating areas of bottom under bell jarlike chambers. Fluxes of major inorganic plant nutrients are often high, indicating that in most nearshore environments most regeneration occurs on the bottom.

In the New York Bight, oxygen consumption by the bottom was measured in four different seasons; we concluded it was high enough to oxidize a large fraction of the daily input of sludge. Bottom water ammonia gradients suggested too that remineralization rates were high on the bottom in the bight. Samples taken in August 1975, in Christiaensen Basin, along with measurements in situ of ammonia flux from the bottom, confirmed that the sediments enriched by sewage sludge are regenerating nutrients but at that time not at rates as high as our earlier predicted rates for the mid-Atlantic Bight.

Benthic–pelagic coupling is the interdependence between biological and chemical processes in the water column and on the bottom in a marine ecosystem. For example, the rates at which organic matter is produced in the water column from nutrients and sunlight by the phytoplankton (Fig. 1) controls the amount of energy eventually available to the benthic biota. On the other hand, the rates of remineralization on the bottom may affect how much dissolved nutrient is available to the phytoplankton for primary production. Rates of remineralization on the bottom are directly affected by the availability of dissolved oxygen, which in turn is supplied by photosynthesis and mixing from the surface. In the New York Bight our concern is with how unnaturally large supplies of nutrients and organic matter may have altered the typical benthic–pelagic coupling that characterizes nearshore ecosystems.

The benthic biota in marine systems depends for the most part on energy supplied from the water column; direct responses in the abundance and biomass of the benthos to variations in these supplies are to be expected. In the New York Bight very high biomass has been recorded along the axis of the Hudson Shelf Valley between the dredge spoil dump site and the sewage sludge dump site, suggesting that unusually large amounts of organic matter are available to the benthos (Vaccaro et al. 1973; Rowe 1971). The high biomass values were the result of extremely dense populations [with total abundances of the macrofauna (>0.42-mm-screen mesh sieves) up to over 200,000 individuals per square meter], rather than high growth rates of a few, long-lived species.

A large fraction of the organic energy available to the benthos is expended as heat rather than utilized for growth or reproduction. The conventional method of quantifying this has been to measure respiration on individual species or to measure the total oxygen demand of areas of the bottom, as reviewed by Hargrave (1973) and Pamatmat (in press). Pamatmat (1973) suggested that the sedimentation rate of readily utilizable organic energy can be estimated approximately from these total metabolic assessments. Other investigators used benthic oxygen consumption to estimate what fractions of the total primary production are utilized by bottom organisms (Pamatmat and Banse 1969; Hargrave 1973; Smith et al. 1973*a*,*b*; Davies 1975). Comparison of rates from numerous areas has led to the conclusion that total metabolism is proportional to the organic matter available (Hargrave 1973).

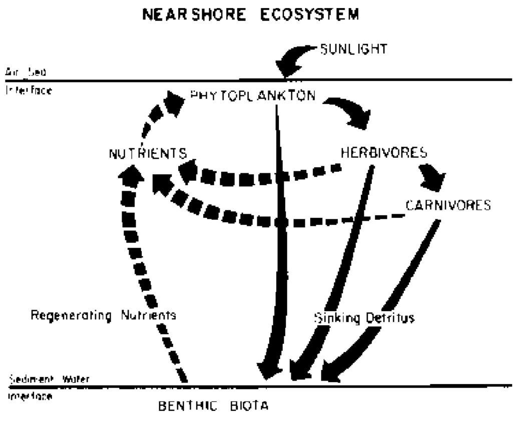

Fig. 1. Nearshore marine ecosystem components, with flows of matter represented by arrows. Dashed arrows are dissolved nutrients and solid arrows are organic matter.

Oxygen utilization by sediments of the New York Bight was measured during four different seasons in 1971–1972, at a location designated as the sewage sludge dump site and the Christiaensen Basin, about 2–3 nmi west of the sludge site (Smith et al. 1973*a*). That study's original intent was to measure oxygen demand on the organic-rich sludge at the sewage dump site. Visual reconnaissance of the bottom revealed a rippled sand rather than sludge, and it was presumed that the sludge moved west with prevailing currents to become trapped in the topographic depression of the Hudson Shelf Valley and the Christiaensen Basin—the landward terminus of the valley. The oxygen demand measurements indicated that about 20% of the input of sludge could be remineralized on the bottom daily, and this rather high rate of remineralization accounts for the absence of a severe buildup of organic matter in the sediments.

The earliest studies of nutrient regeneration in sediments were in freshwater (Mortimer 1942). Ocean sediments received early attention from Rittenberg et al. (1955), who investigated ammonia and nitrate in cores from the continental borderland basins off California. Harris (1959) attempted to balance the nitrogen budget in Long Island Sound with good success but did not attempt to separate microbial regeneration in the water from that in the sediments. In studies off Peru, it was discovered that the anoxic environment could be a nitrogen sink, through denitrification (Goering and Pamatmat 1971), rather than a site of regeneration. Seasonal variations in pore-water concentrations of ammonia in a small sound in Bermuda indicated that fluxes of nutrients out of the sediments could be a major source of nutrients for primary production (Thorstenson and Mackenzie 1974).

The above investigations, based on laboratory incubations or concentrations in the pore waters of sediment cores, have been followed by in situ incubations under bell jarlike chambers. Using such techniques, Nixon et al. (unpublished) found regeneration rates of nitrogen products to be very high in Narragansett Bay, Rhode Island; rates appeared to be independent of both the species composition of the macrofaunal community and the sediment type but were strongly influenced by temperature.

Regeneration rates in a Scottish sea loch (Davies 1975) were also found to be high, under an annular-shaped chamber in which rates of water movement around the chamber could be controlled. Current speed had little effect on nutrient release at low rates (3–11 cm s^{-1}), but with the onset of spring warming, nutrient release appeared to lag behind oxygen demand and organic input to the bottom, estimated with sediment traps.

Near-bottom ammonia gradients in a number of nearshore hydrographic casts collected from Cape Hatteras to Cape Cod have been used to estimate summer rates of nutrient regeneration of continental shelf sediments, with a finite difference equation (Rowe and Smith in press). The predicted rates agreed well with what would be estimated from using benthic oxygen demand as an indirect measure of the breakdown of organic matter (Smith et al. 1973*a*) and actual simultaneous measurements of oxygen demand and nutrient fluxes in Buzzards Bay, Massachusetts (Rowe et al. 1975).

The hypothesis that ammonia gradients or benthic oxygen demand can be used to estimate sediment nutrient regeneration lacked confirmation in the New York Bight. The stoichiometry assumed for the uptake of oxygen and release of forms of inorganic nitrogen is not yet well understood or predictable. For example, much more nitrogen was released experimentally from the sediments off northwest Africa (Rowe et al. in press) than Buzzards Bay sediments, with greater rates of oxygen uptake (Rowe et al. 1975). Maximum regeneration of ammonia appeared to lag behind the seasonal peak in oxygen consumption in the study by Davies (1975), while Nixon et al. (unpublished) measured a midsummer decrease in ammonia flux at a time when temperature and oxygen consumption were highest. These variations have been speculated to be caused by excretion of organic nitrogen (Nixon et al. unpublished), denitrification (Rowe et al. 1975), or the differential effects of current velocity on fluxes of oxygen and inorganic nitrogen compounds (Davies 1975; and Rowe et al. in press).

In an attempt to validate in the New York Bight the hypothesis that the bottom is a major site of regeneration, and to determine the relationship between bottom oxygen consumption and nutrient release, we initiated a brief study during August 1975 in Christiaensen Basin (Fig. 2) at the site of our previous measurements of oxygen consumption (Smith et al. 1973*a*).

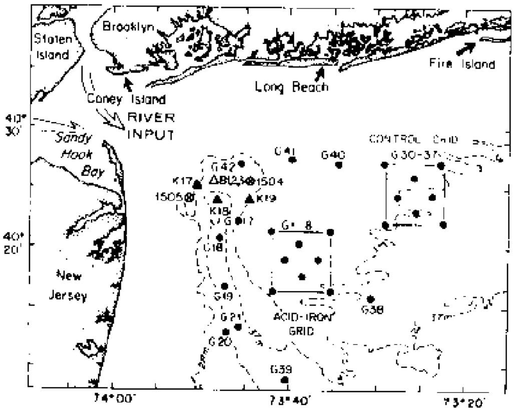

Fig. 2. New York Bight stations (from Rowe 1971). Regeneration experiments were carried out at RV *Balboa* stations—B1, 2, 3. Sewage sludge dump site—1504; dredge spoil dump—1505; stations labeled K or G were discussed in Vaccaro et al. (1973) or Rowe (1971).

Methods

Bell jarlike chambers were used to incubate 7.5 liters of bottom water over a 730-cm^2 area of bottom sediments. Polarographic oxygen electrodes were plugged into the sides of the jars, and the current, once electronically amplified, was recorded on a strip-chart recorder held to the bell jar in a watertight Plexiglas case. The jars were gently emplanted on the sediment surface at the base of, and tethered to, a buoy to the water's surface. Hypodermic syringes were used to withdraw water samples from the jars for dissolved nutrient analyses at the beginning and end of incubations, with the differences in concentration being used, based on jar volume and area covered, to calculate sediment water nutrient fluxes. On two occasions 20 ml of 10% Formalin was injected into the bell jars to estimate the chemical oxygen demand as opposed to respiration. Both opaque and transparent chambers were used with the hope that differences in light and dark uptake rates could provide an estimate of photosynthesis, or oxygen production, within the chambers. Nutrient concentrations were determined by the methods of Solórzano (1969), Strickland and Parsons (1968), Murphy and Riley (1962), and Wood et al. (1967).

Results

Average oxygen consumption (32 ml m^{-2} d^{-1}: Table 1) was high and similar to what would be expected in Christiaensen Basin at the same temperature (7.8°C), based on previous work there (Smith et al. 1973*a*). The rate is high compared to most other nearshore mud bottoms, however (Rowe et al. 1975). Chemical oxygen demand, or perhaps more properly, the rate of consumption within the bell jar after addition of Formalin, was about 50% of the total. Although light would be expected to have an effect on apparent rates of consumption, no difference was seen in light or dark chambers. This was probably because the turbid surface water above the thermocline

Table 1. Oxygen consumption (ml $m^{-2} h^{-1}$) and nutrient fluxes (mg $m^{-2} d^{-1}$) across the sediment–water interface in New York Bight, 3 nmi SE of Ambrose Light, at a depth of 35.5 m.

Date (1975)	Bell jar	O_2	NH_4	NO_2	NO_3	PO_4	Duration (h)	Formalin	Stirred	Total N
31 Jul	Dark	14.27	14.3	3.9	2.6	23	4	yes	yes	20.8
31 Jul	Light	32.46	21.8	1.7	13.5	22.1	4	no	yes	37
31 Jul (night)	Dark	29.76	—	—	—	—	5.5	no	yes	—
31 Jul (night)	Light	34.13	—	—	—	—	5.5	no	yes	—
31 Jul (night)	Light	17.51	—	—	—	—	5.5	yes	yes	—
31 Jul (night)	Dark	30.80	—	—	—	—	5.5	no	yes	—
1 Aug	Dark	33.27	33.9	−2.5	2.6	9.2	3	no	yes	34
1 Aug	Light	—	1.7	−5.1	5.1	5.8	3	no	no	1.7
1 Aug	Light	—	38.9	−3.1	−0.9	1.1	3	no	no	34.9
1 Aug	Dark	31.40	15.2	0.3	−0.1	−5.8	3	no	yes	15.4
		$\bar{x}=32$*	$\bar{x}=21$			$\bar{x}=9.2$				$\bar{x}=24$
			$\bar{x}=25$†							$\bar{x}=28.4$†

* Not including poisoned jars.

† Not including low value on 1 August.

prevented much light from getting to the bottom anyway.

Average nitrogen flux out of the sediments was 23.8 mg N $m^{-2} d^{-1}$, with a wide range to the estimates (1.7–37 mg N $m^{-2} d^{-1}$). Whether this variation is a result of small-scale variations in space or experimental error is not known. The pattern does not indicate that poisoning or absence of stirring affects nitrogen flux rates. Ignoring the one strikingly low ammonia flux value would increase the flux average for ammonia by 16%, which is less variation than the several separate studies reviewed had between them. Longer incubations might decrease experimental error. In two incubations (1 August, light chambers) there was a decrease in nitrite, suggesting denitrification was also occurring, which is to be expected at depths in the sediment where oxygen is depleted. Phosphate flux rates were much less than for nitrogen, of course, but the nitrogen and phosphate rates had a ratio of 2.6 : 1, somewhat lower than the ratio of nitrogen to phosphorus in phytoplankton (Redfield 1934). Other studies have found the ratio to be low (Rowe et al. in press; Nixon et al. unpublished) and have used it in regeneration to account for nitrogen limitation to photosynthesis and low N : P ratios in phytoplankton (Ryther and Dunstan 1971) in nearshore ecosystems (Riley 1941*a,b*).

The concentrations of nutrients went from strikingly high values in surface sediments to comparatively low values in the water (Fig. 3), indicating nutrients should flow down the gradients or from the sediment into the water. Ammonia was high in bot-

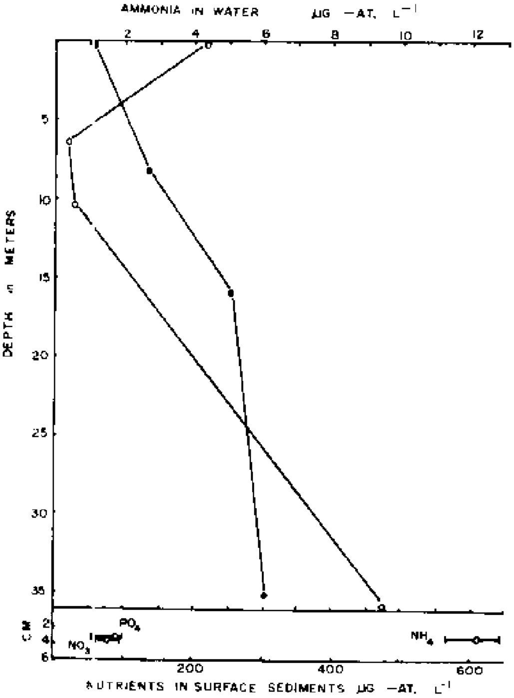

Fig. 3. Concentration of nutrients in sediment pore water (below) and concentration of ammonia in water column on two consecutive sampling days.

tom water and decreased up into the water column, with differing patterns above the thermocline.

Discussion

Rowe and Smith (in press) used average near-bottom ammonia gradients for the whole of the mid-Atlantic Bight to estimate nutrient fluxes; they used a finite difference equation and estimated that at depths of 20–40 m the average flux should range from 70–350 μg-atoms NH_4^+ m^{-2} h^{-1}, or from 23.5–118 mg m^{-2} d^{-1}. This compared reasonably well with the nitrogen regeneration that might be estimated from oxygen demand measurements, or 24.7 mg N m^{-2} d^{-1} at the sand bottom below the sewage sludge dump site and 42.5 mg N m^{-2} d^{-1} from the Christiaensen Basin mud bottom. Our actual measurements ($\bar{x} = 23.8$ mg N m^{-2} d^{-1}) were significantly below these predictions for Christiaensen Basin.

The question then remains: were the finite difference and oxygen–nitrogen models inappropriate, or do processes in Christiaensen Basin sediments differ radically from those in other sediments? The gradient of ammonia we found in the water column (Fig. 3) was markedly lower than the gradients Rowe and Smith (in press) used to make the predictions for the mid-Atlantic Bight, including the gradient they presented for the location of recent sludge dumping (station 1504: Fig. 2). Using our present data in the same finite difference formulation, we would predict the flux to be from 14.4–43.3 mg N m^{-2} d^{-1}, or values more in line with what we measured (Table 1). However, using the same conversion factors for oxygen uptake to nitrogen release (1 ml O_2 = 0.04 mg N), we would estimate that the average return of nitrogen to the water should be about 32 mg N m^{-2} d^{-1}, or about 50% greater than the measured flux average. This was assuming that the nitrogen regeneration was tightly coupled to that of carbon and that N and C would be liberated as NH_4^+ and CO_2 at rates proportional to their organic concentrations in the sediments (10:1). It might be more proper to use the ratio 5:1 or the relative concentrations of C and N in phytoplankton organic matter (Redfield 1934). If that ratio is used however the O : N model overpredicts the measured rates by a factor of two. Whichever model is used, one thing is obvious: unless dissolved organic nitrogen accounts for a major portion of the flux, nitrogen is lost from the system in the sediments in Christiaensen Basin.

The studies in Buzzards Bay (Rowe et al. 1975), Narragansett Bay (Nixon et al. unpublished), and a Scottish sea loch (Davies 1975) indicated that less nitrogen is evolved than would be predicted to return, based on either oxygen demand or regeneration of phosphate from the sediments. This was not the case in studies off northwest Africa where sediment oxygen demand was somewhat below that found on the U.S. east coast but was accompanied by higher rates of ammonia flux out of the sediments (Rowe et al. in press). Ammonia flux there was about three times what would be predicted from an oxygen-nitrogen relationship wherein 0.4 mg C, or 80 μg N (assuming a C:N ratio of 5), is liberated with the consumption of 1 ml of oxygen (1 ml O_2 = 0.08 mg N).

Overestimating flux with this model has been attributed to denitrification (Rowe et al. 1975) and flux of dissolved organic nitrogen (Nixon et al. unpublished), but underestimation, as was the case off Africa, is more difficult to explain. It may be due to the experimental techniques used. Davies (1975) found that while ammonia fluxes were not related to differences of 3–11 cm s^{-1} bottom current velocity, at the bed, oxygen fluxes seemed to increase with the higher velocity in summer. The work off Africa was done on a bottom characterized by relatively strong longshore currents, which created short, sharply crested ripple marks (about 10 cm long and 2 cm deep) in the bottom. The experimental chambers (the same as those used in this study) mix the bottom water at very slow rates and create no pattern of water movement other than what appears to be a slow, random turbulence. The oxygen uptake and dissolved nitrogen output might have been in a more or less steady state under natural conditions with strong bottom currents, but with the bottom isolated by the chamber and current velocity sharply decreased, the ammonia continued to escape at a high rate and oxygen demand decreased. If this occurs it might explain the return of more nitrogen than would be predicted from oxygen consumption.

If we consider the various measurements of oxygen uptake and nitrogen flux, a subtle, suggestive pattern emerges. In regions with organic-rich sediments covered with virtually anoxic water, the sediments act as a nitrogen sink through reduction of nitrate and nitrite to elemental dissolved nitrogen (Goering and Pamatmat 1971). Sediments bathed with high dissolved oxygen concentrations in the near-bottom water and intermediate concentrations of organic matter are able to return considerable inorganic nitrogen to the water (Rowe et al. 1975; Nixon et al. unpublished; Davies 1975), either as ammonia, nitrate, or dissolved organic nitrogen. The opposite end of the spectrum would be the sand and silt sediments off northwest Africa where, because of the availability of oxygen, denitrification rates are lower and less nitrogen is lost from the system (Rowe et al. in press).

Christiaensen Basin, along this gradient, would fall between Peru, characterized by anoxic bottom water over a mud bottom, and the intermediate conditions found in well mixed bays. On the two consecutive days of our experiments we measured values of 3.8 and 3.0 ml liter^{-1} dissolved oxygen in the bottom water, which would be 55% and 43% saturation—values certainly not low enough to be stressful to the benthic invertebrate fauna, but in general below the usual well oxygenated conditions characterizing Buzzards Bay and Narragansett Bay. Denitrification, we would presume, occurs at relatively high rates in the sediments and accounts for the loss of nitrogen from the system.

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Benthic colonization processes—a review and a proposed new model

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Abstract

The new model proposed here is based on the assumption of a constant environment or a priori knowledge of its dependence on time and recruitment into the area by motile juvenile forms. The model can accommodate any number of species for which data are available. The above assumptions are formulated into a linear immigration death process. To model species interactions, the immigration rate of each species is assumed to be a function of the total biomass in the area at time *t* and the carrying capacity. The distribution of the number of individuals in a species is assumed to be approximated by a Poisson distribution.

The model was implemented on a digital computer. The required input parameters are the initial number, immigration rate, death rate, and average biomass of organisms by species. An estimate of the environmental carrying capacity is also required.

The relatively great data requirements have limited further empirical testing, nevertheless, it is the best available tool for better understanding and predicting recovery from catastrophic events.

Predictive knowledge of biological recovery and community development of a denuded or significantly altered marine habitat would be useful for all activities that threaten or cause acute losses of marine organisms. Such activities include offshore oil and gas development and transport, offshore sand and gravel mining, and dredge spoil, sewage sludge, and other solid waste disposal. To measure the impact of such activities, predictions of the extent and frequency of biota losses and dispersal of introduced materials must be coupled with projections of ecosystem recovery processes and with the development of a predictive model. Specific emphasis is on benthic macroinvertebrate colonization on altered substrates.

Studies by MacArthur and Wilson (1967), Simberloff and Wilson (1969, 1970), and MacArthur (1972) of island recolonization have contributed to better basic understanding of evolution, succession, and dispersal; these studies can also be used for more applied investigations. Cairns et al. (1969) demonstrated that the formation and composition of protozoan communities on artificial substrates in a lake were the result of interactions comparable to those proposed by MacArthur and Wilson. Saila (1976) fitted the MacArthur-Wilson model to empirical data on recolonization of macroinvertebrates on dredge spoils dumped into Rhode Island Sound. From observational data, he predicted the new species equilibrium in the spoil area and estimated the time required to repopulate the area to within 95% of its predicted equilibrium population. Some limitations of the model were presented, and the application of delay-differential equations to realistically predict species equilibrium in benthic recolonization was introduced.

Schrader et al. (1974) utilized another approach to predicting equilibrium, involving individual species or very similar species groups; they modified the Leslie matrix (Leslie 1945), which is based on a life-table model of age-specific population growth and decay. From this, Schrader et al. computed, for a wide range of survivorship curves, the time to stable age distribution (recovery time), assuming an initially decimated population of benthic organisms. For survivorship curves typical of widely dispersed ubiquitous benthic species, the time needed to achieve a stable age distribution

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about equalled the longevity of the species. These predicted values were considerably lower than the times to reach equilibrium predicted by Saila with the MacArthur-Wilson model. The species-by-species approach using the matrix model probably does not take into account the interspecific effects that might delay recovery time considerably in a natural situation. Chirlin (1974) reviewed the assumptions and pitfalls of various mathematical representations of the biological environment. His conclusions emphasized the importance of constructing models from empirical findings; he believed that adequate fieldwork and analyses at specific sites are a requisite to realistic models and good prediction.

Characteristic of many recovery processes is the initial dominance of an area by species with efficient dispersal mechanisms. Populations of these opportunist species grow rapidly, in the absence of competition, until later arriving competitors and predators hold their numbers down; in many cases they become extinct. An example is the secondary succession of plants in abandoned farmland (Billings 1938). Island recolonization is somewhat different however. The opportunist species are more likely to persist since their populations are bolstered by immigration, making them more competitive (MacArthur and Wilson 1967). Defaunated island studies (Simberloff and Wilson 1970) showed a pattern indicating that the curve of the number of species present against time reached a maximum, then declined and leveled off. It was theorized that the increase curve showed the noninteractive phase of colonization when competition was at a minimum, whereas the decline was due to crowding pressures. This pattern is believed to be common in recovery processes, such as the colonization by benthos of altered marine environments.

Recovery model

For the model proposed here we assumed that the environment is constant and tolerable over the disturbed area and that members of each species are recruited by the settlement of motile juvenile forms. Examples include airborne pine seeds landing in abandoned farmland or burned out forest (Cypert 1973) dispersal by marine benthic organisms (Thorson 1950), and colonization of dredge spoils by marine benthic organisms (Saila 1976). The mean number of deaths within a species is assumed proportional to the number of individuals of that species.

For a given species, we can formulate the preceding assumptions as a linear birth-death process with constant immigration (Karlin 1969). If we call the immigration or the death of an organism an event, then we take $h > 0$ such that during a time interval of length h, the chance of two or more events occurring is negligible and is in fact $o(h)$. The probability of an individual immigrating during an interval of duration h, where $n(t)$ is the number of individuals present at time t, is given by

$$P[n(t+h) - n(t) = 1] = \lambda h + o(h),\ \lambda \text{ constant.} \quad (1)$$

This probability is independent of $n(t)$ since we are assuming that there are no births, just immigration. The chance of an organism dying is given by

$$P[n(t+h) - n(t) = -1 \mid n(t) = k] = \mu k h + o(h). \quad (2)$$

Note that this is a conditional probability, dependent on $n(t)$, where λ is the immigration rate and μ is the per capita death rate. In Eq. 1, λ is also the constant immigration rate as in Eq. 2.

The mean, $M(t)$, of this process is given (Karlin 1969) as

$$M(t) = \frac{\lambda}{\mu}(1 - e^{-\mu t}) + n(0)e^{-\mu t}, \quad (3)$$

where $n(0)$ is the initial number of individuals. Obviously in Eq. 3, the limiting value of $M(t)$ for large t is λ/μ.

The next assumption concerns interaction among species. It seems typical of recolonization processes that some early arriving species will not compete successfully with later arrivals. Therefore the model must reflect species interactions. This is done by taking the immigration rate for a given species as

$$\lambda[1 - B(t)/K],$$

where λ is as above, $B(t)$ is the total biomass of all individuals of all species present in the area, and K is the carrying capacity of the area, i.e. the maximum biomass that the area can support. To model species interactions we reduce the immigration rate of each species by the same proportion, but the species with lower death rates tend to displace those with higher death rates. Thus, for a given species, the probability of immigration during a brief time is

$$P[n(t+h) - n(t) = 1 \mid B(t) = b] = \lambda h(1 - b/K). \quad (4)$$

Note in Eq. 4 that the immigration probability is conditional, depending on $B(t)$.

Let us suppose we have N species, indexed 1, 2, . . . , N. The species indexed by i, $1 \leq i \leq N$, have an immigration rate λ_i, a death rate μ_i, and average biomass per individual α_i; $n_i(t)$ describes the number of individuals of that species present at time t. Define

$$\Lambda = \sum_{i=1}^{N} \lambda_i[1 - B(t)/K]$$

and

$$M = \sum_{i=1}^{N} \mu_i n_i(t),$$

where

$$B(t) = \sum_{i=1}^{N} \alpha_i n_i(t).$$

Then the probability that immigration of an individual from some species will occur during a time interval of length h is $\Lambda h = o(h)$. Similarly, the probability that an individual of some species will die during the same time is $Mh + o(h)$. Given that the last event occurred at time t, then the length of the interval to the next event, τ, has an exponential distribution (Karlin 1969):

$$P[\tau \leq x] = 1 - e^{-(\Lambda + M)x}. \quad (5)$$

To simulate the model described in Eq. 5, we start by generating the waiting times. This is done by generating a uniformly distributed (on[0,1]) random number u by the mixed congruential method (Mihram 1972) and transforming it (Mihram 1972):

$$dt = \frac{-\ln u}{\Lambda + M}.$$

Once the waiting time has been found, the following algorithm is used to determine the particular event that happens. A random number u_1, between 0 and 1, is generated and transformed to give $u_2 = (\Lambda + M)u_1$. If $u_2 \leq \Lambda$, let j be the smallest integer such that $u_2 \leq \sum_{i=1}^{j} \lambda_i$ and take the event to be the immigration of one member of the species indexed by j. If $u_2 > \Lambda$, let j be the smallest integer such that $u_2 \leq \Lambda + \sum_{i=1}^{j} \mu_i n_i(t)$, and take the event to be the death of one individual of species j.

Application of this model depends on data obtained from random samples taken in a disturbed area, so it was necessary to design a simulation model of the sampling process. Since we assumed that the environment is constant, we can suppose the individuals of each species are distributed uniformly over the disturbed area. Consequently, the distribution of the number of individuals of species i to be found in a sample is Poisson, with parameter m_i equal to sample area times density per unit area (Lentner 1972). This can be simulated by generating random numbers between 0 and 1, u_0, u_1, u_2, . . . until such time as for some r: $-\sum_{n=0}^{r} \ln(u_n) \geq m_i$ for the first time (Mihram 1972). Then r is the number of individuals of species i found in that sample.

The model just described was implemented on a digital computer (IBM 360/155) in C8 and FORTRAN. The program is available.

Several runs were made using arbitrary parameters, where the immigration and death rates of the species increased as their biomasses decreased. Table 1 illustrates one set of input parameters used for the model.

Table 1. Arbitrary input parameters used for a simulation of benthic colonization illustrated in Figs. 1 and 2. Number of species or species types is 20; initial number of individuals and biomass at $t = 0$ is zero.

Species No.	Avg. biomass/individual	Immigration rate	Death rate
1	0.01	1,000	0.5
2	0.01	8,000	0.5
3	0.01	5,000	0.1
4	0.01	2,000	0.01
5	0.01	2,000	0.01
6	0.025	1,000	0.015
7	0.025	1,000	0.015
8	0.025	1,000	0.015
9	0.02	1,000	0.01
10	0.05	333.3	0.005
11	0.05	333.3	0.005
12	0.05	335	0.004
13	0.05	335	0.001
14	0.05	335	0.001
15	0.075	200	0.001
16	0.05	200	0.005
17	0.1	100	0.001
18	0.1	100	0.001
19	0.1	100	0.001
20	0.25	50	0.003

A carrying capacity of $K = 2{,}000$ was used to plot total biomass against time (Fig. 1). In Fig. 2 the biomass for species number 2, 6, and 20 (Table 1) was plotted against time. Figure 3 is taken from the empirical studies of dredge spoil recolonization by Rhoads et al. (1975). Figure 4 shows the analog of the DT curve of Fig. 3 as reproduced by a computer run of this model using parameters extracted from the empirical curves in Rhoads et al. (1975). In Fig. 5 the individual curves for polychaetes and bivalves (deposit, suspension-feeding) at DT as plotted from the above computer run are shown. The carrying capacity was assumed to be a step function of time, reaching a maximum of 1,300 after 5 months and the average biomass of all species involved equal to 1. The last assumption limits the validity of the output since the variation in the average biomass of the species involved is a crucial assumption in implementing this model.

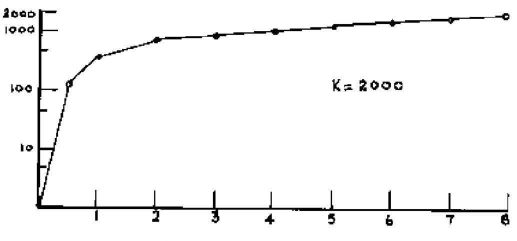

Fig. 1. Total biomass of species with input parameters given in Table 1, carrying capacity $K = 2{,}000$, with time.

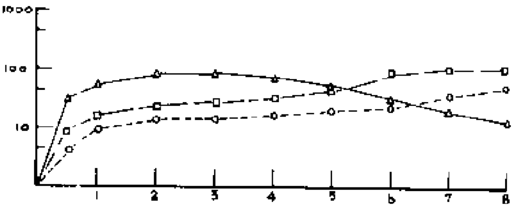

Fig. 2. The biomass of species numbers 2(△), 6(□), and 20(○) Table 1 with time.

Discussion

Use of this model implies knowledge of three input parameters for each species: the immigration rate, the death rate, and the average biomass per individual. The carrying capacity of the area must also be estimated. The model is intended for operation on a digital computer; it can be considerably simplified by classifying the N species into a smaller number of similar groups and assuming that the corresponding

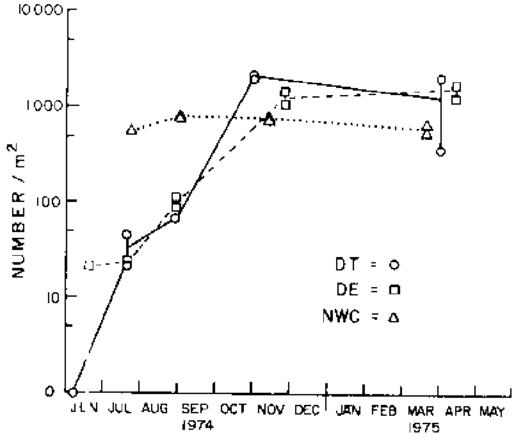

Fig. 3. From the empirical studies of dredge spoil recolonization by Rhoads et al. (1975).

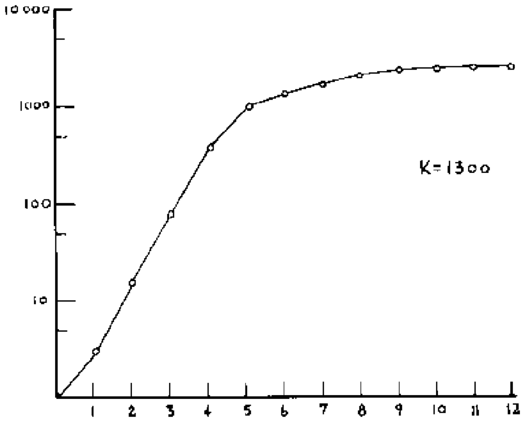

Fig. 4. The analog of DT curve of Fig. 3 as reproduced by a computer run of this model, using parameters extracted from the empirical curves in Rhoads et al. (1975).

parameters of species belonging to the same group are equal.

Some problems are involved in implementing this model as applied to the process of benthic recovery or colonization.

The input parameters for the immigration and death rates of individual species are difficult to measure. Saila's (1976) model does not require as much detailed information. However, that model is also not as realistic as the model presented here. Seasonal or external factors may affect immigration and death rates.

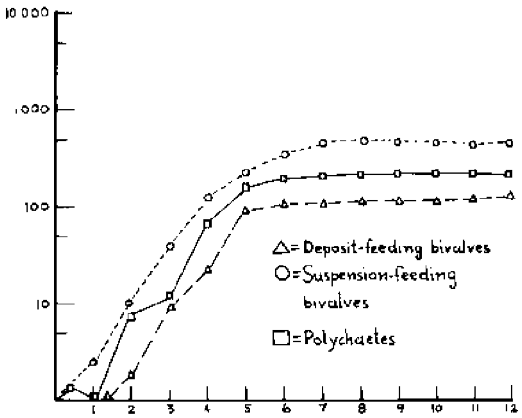

Fig. 5. The individual curves of polychaetes and bivalves (deposit, suspension-feeding) at DT as plotted from the computer run that produced Fig. 4.

The assumption that the average biomass of a species is representative of every individual of a species should be treated with caution; the weights of different individuals of a species tend to vary greatly, depending on the age of the individuals.

This model is also dependent on the assumption that the environment being considered can only support and maintain a certain optimal biomass. This is the carrying capacity *K*—a variable that depends on the environment and the external influences exerted on it. The accuracy of the model is limited by how accurately the carrying capacity can be estimated. We also assume a random immigration process, which should be tested empirically.

In summary, the application of the model is limited by the input data requirements and by the assumptions inherent in the model. The development of this model has clearly defined the nature of the biological data required for predictive modeling. It also is a useful conceptual tool to better understanding the recovery and colonization process.

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Antimicrobial resistant bacteria in the New York Bight[1]

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Abstract

Sediment samples and overlying water from stations around the sewage dump site in the New York Bight were analyzed for coliform and noncoliform bacteria resistant to mercury and antibiotics. Few or no coliforms could be found in these samples, but a pollution gradient was identified in a northeasterly direction, toward Long Island. About 1% of the sediment bacteria desorbed was consistently resistant to $HgCl_2$ 10^{-5} M and/or tetracycline 40 mcg/ml at MESA station 34. The antibiogram of a majority of the isolates from these sediments showed multiple antibiotic resistance.

Sediments from which antimicrobial resistant bacteria were isolated had high bound water content, much debris, little or no evidence of normal benthic macrofauna and a black, gelatinous consistency. In contrast, sediment taken from the Sandy Hook transect showed no resistant bacteria, and demonstrated no other parameters related to pollution. Analysis of water and sediments containing very small numbers of coliforms from a beach at Sandy Hook State Park revealed a significant number of bacteria resistant to seven or more antibiotics. These data suggest that techniques used in these studies may detect pollution gradients which are not measurable by coliform counts.

Analysis of water and sediments from the sewage dump site in the New York Bight (summer 1972) revealed the presence of coliforms resistant to antibiotics and selected heavy metals among the bacterial population present in the sediment sampled at the approximate center of the site. Some of the multiple resistant coliform strains were donors of R plasmids during conjugation with a recipient strain of *Salmonella gallinarum* (Koditschek and Guyre 1974*b*). (*R plasmid*—Synonyms: R factor, resistance factor—is plasmid DNA that carries genetic information for resistance to antibiotics, metals, or other toxic substances.)

The high incidence of noncoliforms resistant to $HgCl_2$ and to multiple antibiotics observed for the sediment samples examined, as in an earlier study on freshwater (Koditschek and Guyre 1974*a*), suggested that coliform counts may not be adequate indicators of pollution. Other investigators have questioned reliance on total and fecal coliforms to monitor public health hazards in aquatic environments (Dutka 1973; Cabelli unpublished; Grabow et al. 1974). It is conceivable that *Pseudomonas* spp. may act as ubiquitous plasmid vectors in soil and water since plasmid genes in this genus code for degradation of unusual, polluting substrates as well as resistance to antimicrobial agents (Nelson and Colwell 1973; Walker and Colwell 1974; Gunsalus et al. 1974). The fact that some antibiotic resistance in *Pseudomonas* spp. may be natural and chromosomal resistance in the species can be mobilized by plasmids (Gunsalus et al. 1974; Jacoby 1974; Stanisich and Holloway 1971) may be important in environmental dissemination of multiple resistance. Lacy and Leary (1975) reported the transfer of antibiotic resistance plasmids into phytopathogenic and phytosymbiotic pseudomonads as well as among bacteria resident in the soil.

The relationship of heavy metals and antibiotic selection of plasmids among hospital isolates, demonstrated by Summers et al. (1974) and Schottel et al. (1974), becomes more significant in relation to the report by

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[2] Samples were obtained with equipment and help from personnel at the Sandy Hook Laboratory. In particular, I thank J. R. Pearce. Very efficient technical assistance was rendered by P. Guyre, L. Dzurney, and J. Young. I appreciate help from J. Unowsky and R. H. Bordner.

Walker and Colwell (1974) and Vaituzis et al. (1975) that there is a high incidence of mercury resistance among petroleum-degrading bacteria in a marine environment. Although many gram-negative Hg^R bacteria detoxify mercurials by producing volatile Hg^0, Hamdy and Noyes (1975) have isolated strains of *Enterobacter* from the Savannah River which convert $HgCl_2$ to methyl mercury.

Isolation of bacteria simultaneously resistant to $HgCl_2$ and multiple antibiotics suggests a concentration of plasmid hosts, which though they may not be pathogenic, could disseminate resistances to a wide range of resident bacteria, including pathogens. Although R factors do not seem to enhance survival of host bacteria under optimal conditions, there is evidence for selection of R^+ bacteria under adverse conditions such as low oxygen tension (Brinton 1971), UV radiation, heavy metal concentration, etc. (Davies and Rownd 1972; Grabow et al. 1974; Kondo et al. 1974). [*R^+ bacteria* —Synonym: resistance transfer donors—are bacteria containing resistance plasmids and genes that code for conjugative transfer of the plasmids (RTF). On receiving R + RTF genes (DNA), the recipient becomes a donor. This type of conjugative transfer is very rapid (often less than 10 min) and a whole population may become donors by such "genetic infection" with specific DNA.]

Studies reported here investigated concentration of antimicrobial resistance bacteria in the New York Bight sewage dump site and the feasibility of using such assays to demonstrate pollution gradients of potentially toxic substances which may persist at the sewage dump site. As such, the emphasis was on analysis of *total bacterial populations* showing resistance to $HgCl_2$ 10^{-3}M and/or tetracycline 40 mcg/ml, rather than speciation, to provide a "rapid profile" of pollution gradients. In conformity with *Standard methods* (Am. Public Health Assoc. 1970, 1971), MPN and M-FC coliform counts were taken on all samples.

We examined samples from the MESA stations shown in Fig. 1 and also from Sandy Hook State Park. These analyses are detailed in Koditschek and Guyre (1975) and only summary data are given in part 1 of this report. During this work it became increasingly evident that basic studies on optimal desorption of bacteria resident in these marine sediments was necessary. Elutants to desorb sediment bacteria have been reviewed by Litchfield et al. (1975). Roper and Marshall (1974) reported optimal elution of *E. coli* from marine sediments with dilute seawater at ionic strength indicated by an inverse conductance (i.e. resistance) of about 450 ohms. We examined the conductance of commonly used elutants in relation to the recovery of a known recipient, *E. coli* K_{12} F^- Nal^R strain, which was inoculated into sterile sediment from MESA station 34. Initial results of these studies are available (Koditschek 1975); they are summarized in part 2 of this report.

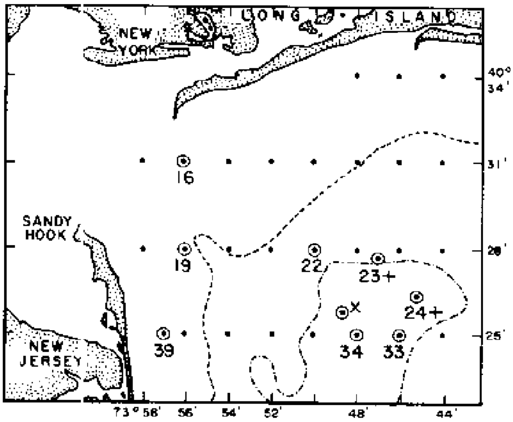

Fig. 1. Location of sampling sites. Numbers represent MESA grid for SYMAPS (*see Table 1*).

Materials and methods: Part 1

Sampling and microbiological techniques used in these studies have been described by Koditschek and Guyre (1975). Thus only methods of special interest are included here.

Buffered $HgCl_2$ was introduced into sterile media to give a concentration of 5×10^{-4} M for initial isolation on trypticase soy agar (TSA) and 1×10^{-3} M for initial isolation on M-FC or reisolation on TSA (sensitivity testing with 14 laboratory stock cultures used as controls indicated that M-FC agar required a higher concentration of

Hg^{2+} for full inhibition of control cultures). For antibiograms, antibiotic disks (BBL or Difco) were supplemented with disks impregnated with metal salts as follows: 10^{-2} M $HgCl_2$, 10^{-2} M $K_2CR_2O_7$, 10^{-1} M $Pb(NO_3)_2$, and 10^{-1} M $CuSO_4$. $Pb(NO_3)_2$ and $CuSO_4$ could not be adjusted to neutrality without precipitation. Therefore, control disks adjusted to the same pH were used to monitor acid toxicity.

Stock solutions of crystalline tetracycline (TE) or chlortetracycline (Chl) (Sigma Chemical Co. or Lederle) were introduced into sterile media to give a concentration of 40 mcg/ml.

Materials and methods: Part 2

Several "Kim-Kap" frozen cores (Koditschek and Guyre 1975) obtained at MESA station 34 were mixed for greater uniformity and autoclaved at 121°C for 15 min. Before inoculation all sediments were tested for sterility by streaking on TSA and inoculation into phenol red lactose broth. All sediments used in these experiments were from the same batch tested on 9 August 1974 (Table 1).

The *E. coli* $K_{12}F^-$ used for these recovery experiments was obtained from J. Unowsky (Hoffmann-La Roche, Inc.). This culture, which is widely used as a recipient strain (Jarolmen and Kemp 1969; Smith 1971) carries a chromosomal marker for resistance to nalidixic acid (Nal) and was maintained in Nal-supplemented TSA. It ferments lactose and produces colonies with a green sheen on EMB agar and blue colonies on M-FC agar (Am. Public Health Assoc. 1971). For sediment inoculation, overnight cultures were transferred into TSA broth and incubated on a water bath shaker at 35°C until the culture reached

Table 1. Incidence of Hg^R and antibiotic^R bacteria at selected stations.

Date	Sta.	MPN	M-FC*	Hg^R*	Chl^R* or TE^R	Bacterial density*
19 Jul–9 Aug 74 Sed. samples	MESA 39	<2–79	ND†–est. 3×10^3	ND	ND	2.4×10^4– 3.7×10^5
Water samples		13, <2	est. 32/100 ml	ND	Insig.	est. 1.8×10^4
29 Jul–9 Aug 74	BS‡	<2–2	ND	ND	ND– 2/10 ml SS	1.7×10^5– 3.1×10^6
	BW‡	7–49	28–56	4–6/100 ml	est. 35 lac+ 6.9×10^2 lac−	1.6×10^3– 1.6×10^4§
11 Jul–9 Aug 74 Sed. samples	MESA 34	<2 (LTB)‖	ND	4×10^3– 1×10^5	1.4×10^4– 3.6×10^4	2.7×10^6– 4.8×10^6
Water samples		<2–2 (LTB)‖	ND	est. 8	est. 14	4.7×10^3– 9.3×10^3
9 Aug 74 Sed. samples	MESA 23+ site between 23 & 24	<2	ND	est. 5×10^3– 8×10^3	est. 6×10^3– 4×10^4	1.3×10^6– 1.4×10^6
Water samples	"	<2	ND	ND	Insig.	1.9×10^3– 2×10^3
29 Jul–9 Aug 74 Sed. samples	MESA 24+ site between 24 & 33	<2	ND	3.4×10^3– 4×10^4	9.6×10^3– 3.3×10^4	1.6×10^6– 2.8×10^6
Water samples	"	<2–2	ND	ND	est. 27	3.3×10^3

* M-FC agar (*see Standard methods* membrane filtration). Hg^R = colony counts on TSA supplemented with mercuric chloride; Chl^R or TE^R = colony counts on TSA supplemented with chlortetracycline or tetracycline. Bacterial density = average range, colonies/100 g of sediment or 100 ml of water on TSA.

† None detected in 10 ml SS (0.25 g wet sediment).

‡ BS = surf sediment from several locations at Sandy Hook State Park where people were swimming. BW = water from same area.

§ Replica plating from TSA plates revealed a number of Hg^R and TE^R isolates (*see Table 3*).

‖ LTB = lauryl tryptose broth, to select against anoxic gas producers and other false positive reactors.

logarithmic phase (monitored by optical density). The centrifuged, washed cells were suspended in *Standard methods* buffer to give an optical density of 0.3–0.6 (Spectronic 20).

One milliliter of the washed buffered cell suspension (WCS) was added to 9 g of sterile wet sediment. After thorough mixing with a sterile spatula, 1-g aliquots of the inoculated sediment were weighed aseptically and suspended in 99 ml of the respective desorbing solution in a sterile, stoppered 250-ml Erlenmeyer flask. Each flask was shaken vigorously by hand 25 times (hand-mixed), sampled, and a portion of the sediment suspension (SS) was then agitated for 60 s in a blender. Thus both hand-mixed and blended SS represent a 10^{-3} dilution of WCS.

Viable counts were made from WCS and the SS by surface-plating 0.1 ml of appropriate dilutions on M-FC agar or by simultaneous membrane filtration (MF) of appropriate aliquots of each dilution. It is important to note that 0.45-mcg-pore size filters were used (HAWG-47SO, Millipore Corp.) since recent reports indicate that Millipore membranes may not permit optimum growth of coliforms on selective media (Green et al. 1975; Rose et al. 1975; Sladek et al. 1975). Control filters were introduced between the assay of each SS to monitor sterility during MF. Two to five replicates of each dilution were incubated at 35°C for 18–20 h, and the results were scored. On MF the colonies were usually smaller and less blue, assumed to indicate slower rate of growth compared with surface plating. Lower counts were consistently found on MF, compared with surface plating on M-FC.

The following sterile elutants were used for SS:

1. Distilled H_2O.
2. *Standard methods* buffer (SM).
3. 0.1% peptone solution (Bacto peptone). A low concentration of peptone was chosen to minimize nutrient supplementation in SS and minimize growth of inoculated bacteria in SS, and because the conductance of this peptone solution was close to the conductance of SM (*see Table 2*).
4. Artificial seawater (ASW) (obtained as a crystalline mix from Utility Chem. Co.) 2% and 4%.

Conductivity bridge model RC-16B2 (Beckman Instruments, Inc.) was calibrated for conductivity readings in micromhos and for resistance readings in ohms (Table 2).

Results: Part I

The highest incidence of resistant bacteria was uniformly found in black, gelatinous sediments, showing little visible evidence of life (Table 1). Water loss following desiccation of these cores of black sediment was consistently high (34–53%) compared to sandy sediment (16–18%). More than 1% of the bacteria eluted from these sediments were resistant to $HgCl_2$. Although coliform counts at station 34 were negligible, the average bacterial population resistant to chlortetracycline (Chl) or tetracycline (TE) was slightly less than 1% of the average bacterial density (Table 1); the highest number of resistant isolates was obtained at station 34 (Table 3). The incidence of Hg^R isolates was about double the antibiotic resistant isolates (40 Hg^R/24TE^R). Station 24+ showed a "pollution profile" similar to station 34, but not as severe, as evidenced by some life in the sediment, less bound water, greater variability and lower incidence in resistant bacteria found in different cores (3×10^4 Hg^R in one core compared with 6.4×10^3 Hg^R in another core from the same grab), and lower numbers of resistant isolates (20 Hg^R/8 TE^R). A comparison of the data for these "pollution parameters" in Tables 1 and 3 as well as the appearance of the sediment indicated a pollution gradient from station 34 to 24+ to 23+.

In spite of obvious anoxic conditions and black deposits in sediment from station 93 (alternate dump site, north, not shown in Fig. 1) assay results indicated no sewage population. Thus the pollution parameters used permitted differentiation from the "false positive" MPN reactions shown by anaerobes found in the clean sediment of station 93, contrasted with "false positive"

Table 2. Recovery and viability of *E. coli* $K_{12}F^-$ inoculated into sterile sediment from sewage dump site (*see control WCS* and text*).

Elutant used in SS	Resistance of elutant (ohms)	Hand-mixed Recovery CFU*×10^6	Hand-mixed % Recovery*	Blended Recovery CFU*×10^6	Blended % Recovery*
Distilled H_2O					
0 time	10×10^5	156	16	108	11
refrig. 1 d		18	2.8	3.4	<1
refrig. 2 d		5.6	1.4	0.6	<1
refrig. 6 d		$<10\times10^4$	<1	$<10\times10^4$	<1
Standard methods buffer					
0 time	15×10^3	113	11	93	8.4
refrig. 1 d		11	1.7	$<10\times10^6$	<1
refrig. 2 d		2.6	<1	$<10\times10^6$	<1
refrig. 6 d		ND†	<1	est. 4×10^4 blue est. 7×10^5 pink‡	<1
2% ASW					
0 time	37	192	20	148	15
refrig. 1 d		44	6.8	69	11
refrig. 2 d		4.6	1.1	18	4.2
refrig. 6 d		3.2	1.0	8	2.5
4% ASW					
0 time	22	188	19	108	11
refrig. 1 d		23	3.6	22	3.4
refrig. 2 d		6.5	1.5	4.5	1.1
refrig. 6 d		4	1.3	est. 4×10^5 blue est. 6×10^4 pink	<1
0.1% Peptone					
0 time	12×10^3	92	9.4	109	11
refrig. 1 d		ND	<1	ND	<1
refrig. 2 d		27×10^5	<1	13×10^4	<1
refrig. 6 d		$<10\times10^4$	<1	ND	<1

* CFU (colony forming units) represent average of 2–5 spread plate replicates.

Control, washed cell suspension (WCS)	CFU×10^6	Viability (%)
a) day of experiment (0 time)	975	100 (assumed)
b) refrigerated 1 day	640	65.5
c) refrigerated 2 days	427	44
d) refrigerated 6 days	318	32

Sediment suspension (SS) inoculated as described in text.

% Recovery = CFU of SS/CFU of WCS × 100 (and rounded to two digits). For each time of refrigeration, the new control count of WCS was used.

† None detected at 0.1 ml of SS or 10^{-4} dilution.

‡ Pink colonies = lac$^-$, i.e. not capable of fermenting lactose.

MPN reactions shown by polluted sediments from stations 34 and 24+.

Antimicrobial spectrum of isolates—Resistant colonies were picked at random from MF culture on selective media or from replica plating (Koditschek and Guyre 1974*a*). There was no proportional relationship between colony counts on selective media and number of resistant isolates per station. Nevertheless, where the overall incidence of resistant colonies was highest, i.e. station 34, a greater number of isolates were obtained. Table 4 shows the antibiogram of some tetracycline resistant isolates from station 34 that were also resistant to four or more antibiotics. Hg^R isolates were also resistant to Cu, Cr, and possibly Pb (results obscured by precipitation of reagents). Five of these Hg^R cultures on initial isolation were resistant to seven or more antibiotics.

Bathing beach sediment, which appeared clean and had little bound water, contained some resistant bacteria (Tables 1 and 3). Bathing water showed a much higher incidence of resistant bacteria than the bathing sediment (Table 1), as well as coliforms

Table 3. Number of isolates examined at each station.

MESA sta.*	$\mathrm{Hg^R}$	$\mathrm{Chl^R}$ or $\mathrm{TE^R}$
62	ND†	ND
93	ND	ND
16	ND	ND
19	7	2
22	ND	3
23+	12	4
24+	20	8
X	5	1
33	6	1
34	40	24
39	2‡	2‡
BS§	5	3
BW§	23	20

* From MESA SYMAPS.

† None detected.

‡ Found in only one sediment suspension by replica plating from TSA colonies only.

§ BS = sediment from surf area at Sandy Hook Park; BW = water from surf area at Sandy Hook Park.

(though few in number). This is in marked contrast to *water* from the sewage dump site, where neither coliforms nor significant numbers of resistant bacteria could be detected in the case of the small number of samples examined (Table 1).

The incidence of multiple resistance to antibiotics was much higher among either $\mathrm{Hg^R}$ or $\mathrm{TE^R}$ resistant isolates from the *bathing water* than it was from the sediment at station 34. In one sampling, replica plating of colonies from water isolates yielded a total of 13 $\mathrm{TE^R}$ isolates whose antibiogram showed that 12 of these isolates were simultaneously resistant to seven or more antibiotics (Table 5). In contrast, the antibiograms of resistant isolates from station 34 showed a lower percentage resistant to multiple antibiotics. Overall, 35% of the $\mathrm{Hg^R}$ isolates from station 34 showed intermediate or full resistance to tetracycline (as defined by the Kirby-Bauer method), and about 19% were resistant to Cu, Pb, and Cr salts.

Results: Part 2

Comparison of data on recovery and viability of *E. coli* $\mathrm{K_{12}F^-}$ in buffered washed cell suspension (WCS) and after inoculation into sediment and desorption with the specific elutants listed for SS preparation (Table 2) indicates that these elutants did not yield good recovery of the inoculated bacteria, even when assayed immediately after inoculation (0 time). In no case was recovery higher than 20% of the bacteria inoculated (2% ASW, hand-mixed SS). Variability was noted, but the data demonstrated the following:

1. There was no overall increase in recovery by blending SS, compared with hand-mixing. Thus, although in a few cases recovery after blending appeared to be better (2% ASW days 2–6), in other cases the recovery from blended SS was lower (distilled H_2O) and it can be concluded that such variations probably represent random fluctuations, although statistical analysis was not applied.

2. Best recovery was obtained with 2% ASW or lower (Tables 2 and 6). Although

Table 4. Selected TE resistant isolates from station 34 showing multiple antibiotic resistance.*

Isolate	Antibiotic resistance	Heavy metal† resistance
34 A 23	$\mathrm{NA^R}$, $\mathrm{TE^R}$, $\mathrm{A^R}$, $\mathrm{S^R}$, $\mathrm{AM^R}$, $\mathrm{C^R}$, $\mathrm{G^R}$	$\mathrm{Cr^R}$, $\mathrm{Cu^R}$, $\mathrm{Pb^{R\dagger}}$
34 A 29	$\mathrm{E^R}$, $\mathrm{TE^R}$, $\mathrm{A^R}$, $\mathrm{S^R}$, $\mathrm{AM^R}$	$\mathrm{Cr^R}$
34 A 45	$\mathrm{E^R}$, $\mathrm{TE^R}$, $\mathrm{A^R}$, $\mathrm{S^R}$, $\mathrm{AM^R}$, $\mathrm{K^R}$, $\mathrm{G^R}$	$\mathrm{Hg^R}$, $\mathrm{Cr^R}$, $\mathrm{Cu^R}$, $\mathrm{Pb^{R\dagger}}$
34 A 47	$\mathrm{TE^R}$, $\mathrm{A^R}$, $\mathrm{AM^R}$, $\mathrm{G^R}$	$\mathrm{Hg^R}$, $\mathrm{Cr^R}$
34 E 51	$\mathrm{TE^R}$, $\mathrm{A^R}$, $\mathrm{AM^R}$, $\mathrm{K^R}$, $\mathrm{C^R}$	$\mathrm{Cr^R}$
34 E 55	$\mathrm{NA^R}$, $\mathrm{TE^R}$, $\mathrm{A^R}$, $\mathrm{AM^R}$, $\mathrm{C^R}$, $\mathrm{G^R}$	$\mathrm{Hg^R}$, $\mathrm{Cr^R}$, $\mathrm{Cu^R}$, $\mathrm{Pb^R}$
34 A 40	$\mathrm{E^I}$, $\mathrm{NA^I}$, $\mathrm{TE^R}$, $\mathrm{A^R}$, $\mathrm{AM^I}$, $\mathrm{K^I}$, $\mathrm{C^I}$	

* Resistance to four or more antibiotics as defined by the "zone size interpretative chart" of the Kirby-Bauer method.

Sensi discs (BBL) of the following antibiotics were placed on the seeded plates: erythromycin (E), 15 mcg; nalidixic acid (NA), 30 mcg; tetracycline (TE), 30 mcg; aureomycin (chlortetracycline) (A), 30 mcg; streptomycin (S), 10 mcg; ampicillin (AM), 10 mcg; kanamycin (K), 30 mcg; chloramphenicol (C), 30 mcg; gantrisin (G), 1 mg.

† Small sterile filter paper disks (similar to BBL Sensi discs) were dipped in pH-adjusted solutions of metal salts to give concentrations as indicated in *Materials and methods*. (Precipitation of reagents presented some technical problems in pH adjustment with $\mathrm{Cu^{2+}}$ and $\mathrm{Pb^{2+}}$ salts, as well as some precipitation of metabolic products around disks after growth of bacteria.)

Table 5. Antibiograms of selected TE^R isolates from bathing water samples collected at Sandy Hook State Park.

Isolate No.†	Antibiotic resistance*									
	E	NA	TE	A	S	AM	GM	K	C	C
BWF 26	R	I	R	R	R	R	R	R	R	R
BWF 27	R	R	R	R		R		I	R	R
BWF 28	R	R	R	R	I	R			R	R
BWF 29	R	R	R	R	R	R		R	R	R
BWF 30	R	I	R	R	R	R		R	R	R
BWF 31	R	R	R	R	R			I	R	R
BWF 32	R	R	R	R				R	R	R
BWF 33	R	R	R	R	I	R			R	R
BWF 34	R	R	R	R	I	R	R	I	R	R
BWF 35	R	I	R	R	R	R	R	R	R	R
BWF 36	I	R	R	R		R			R	R
BWE 44	R	I	R	R	R	R			R	R
BWE 51	R		R	R	R	R		I	R	

* Table 4 footnotes and section on *Materials and methods* gives potencies of antibiotic disks. Symbols same as in Table 4.

† Isolates BWF 26-36 were Hg^R. Isolates BWE 44 and BWE 51 were also resistant to Triple Sulfa (Difco Trisulfapyrimidines USP 150 mcg/disk) as well as Cr^R and Cu^R.

some of the counts obtained with 4% ASW were similar to those with 2% ASW, there was a more rapid decline in recovery after 1 day of refrigeration with the higher concentration of salts. In repetitive testing, the 2% ASW consistently gave overall higher recovery. ASW solutions more dilute than 2% and corresponding to a resistance of 430 ohms failed to give the high recovery reported by Roper and Marshall (1974) who used diluted seawater of similar ionic strength. (The peak of desorption for *E. coli* in their experiments corresponded with an inverse conductance, 1/LS = 400–800 ohms; M. Roper personal communication.)

3. Poorest viability and recovery overall was found with 0.1% peptone and distilled H_2O, apparent in each of four runs.

4. Lac$^-$ colonies (pink colonies on M-FC media), apparently mutants of the inoculated lac$^-$ population, were observed; this suggests the presence of mutagenic agents in this sediment. Such lac$^-$ colonies were found at much higher incidence in precipitated sediment (Table 7). Additional evidence of the presence of toxic material in these sediments is found in the rapid decline in viability in SS. Thus, after 2 days of refrigeration, 44% of WCS was viable, but in SS viable counts ranged at 1% or lower except for SS with 2% ASW. After 6 days of refrigeration, 10^{-8} dilution of the WCS gave an average CFU = 318, but few if any bacteria could be detected in plating 0.1 ml of SS (i.e. a total dilution of 10^{-4}) except for 2% ASW (*see Table 2*). It would be of interest to explore the mechanism of increased viability of bacteria in such half-diluted ASW for possible relation to higher rates of viability of coliforms in estuary or other marine environments experiencing dilution of salt concentration.

Table 6. Recovery of *E. coli* $K_{12}F^-$.*

Elutant used in SS†	Resistance (Ω)	Recovery, CFU×10^6	Recovery, % of WCS
2% ASW	37	143	19
Dilute ASW (13 ml 2% ASW + 187 ml sterile dist. H_2O)	430	187	25
Standard methods buffer	15.4×10^3	23	3.1

* Control: WCS = 760×10^6 CFU.

† New core used, one experiment.

Recovery of bacteria not desorbed by first elution; second elution of SS sediment—Efforts were made to recover additional adsorbed bacteria in a second elution of precipitates from each SS. Attempts to obtain such precipitates by prolonged centrifugation were unsuccessful because ASW effectively dispersed sediment particles, making it impossible to obtain a packed sediment

Table 7. Second elution (SS_2) of bacteria from filter paper precipitation of sediment from 10 ml of SS_1.

Elutant used in SS_2	Filtration from hand-mixed SS_1		Filtration from blended SS_1	
	Refrig. 2d (CFU* SS_2)	Refrig. 3d (CFU* SS_2)	Refrig. 2d (CFU* SS_2)	Refrig. 3d (CFU* SS_2)
Dist. H_2O	0	0	2	0
Standard methods buffer	22	0	35 + 1 *pink*	8 + 1 *pink*
ASW 20 g/liter	390 + 17 *pink*	47 + 87 *pink*	657 + 8 *pink*	400 + 12 *pink*
ASW 40 g/liter	15	1	0	0
0.1% peptone	25	2	2	1

* All counts × 10^4.

pellet. Filtration of SS with sterile filter paper (Whatman No. 4 ect 4.25-cm diam in a sterile Buchner funnel) yielded a good sediment precipitate. After a sterile water wash, each filter paper with precipitated sediment from a 10-ml aliquot of SS_1 was introduced into a 50-ml sterile screw-cap Erlenmeyer flask containing 10 ml of the respective desorbing solution. The flasks were shaken vigorously (until the filter paper disintegrated), and the contents were plated by the spread plate technique. Results given in Table 7 demonstrate that 2% ASW was more effective in desorbing bacteria from the filtered sediment. Better survival of bacteria in sediment exposed to 2% ASW at refrigerator temperature (5-10°C) for 2 and 3 days was observed, compared with exposure to the other desorbing solutions tested. The appearance of pink (lac^- colonies) suggested that mutagenic substances may have been present in the sediment. To ensure that such lac^- colonies were not contaminants, selected mutant colonies were tested for nalidixic acid resistance (Nal^R is a chromosomal marker carried by this strain of *E. coli* K_{12}). It seems that such mutant bacteria are less viable in the presence of the other elutants, or they are desorbed effectively only by 2% ASW. These observations may have significance in survival and dissemination of mutant strains at the sewage dump sites.

These experiments suggest ionic strength may influence desorption of sediment bacteria, but the variety and balance of salts in ASW (as well as ocean water) may also be significant. The 0.1% peptone invariably gave lower recovery of the inoculated *E. coli* than did *Standard methods* buffer, although the ionic strength (resistance values) of these elutants is similar. The efficiency of desorption reported by Roper and Marshall (1974) was not seen. These differences may be due to differences in sediment, the strains examined, or in constituents of the diluted seawater and ASW. The composition of elutants, as well as the total ionic strength as measured by conductance (or resistance) may be of significance in optimal elution of bacteria (Marshall 1969). In summary, the dilute solutions of ASW were more effective in desorbing *E. coli* $K_{12}F^-$ from polluted sediments than other elutants examined.

Discussion: Parts 1 and 2

Although there is evidence of rapid die-off of sewage bacteria in marine waters, sedimentation and flocculation may result in accumulation of polluting bacteria in sediments (Rittenberg et al. 1958; Van Donsel and Geldreich 1971). Grimes (1975) reported release of sediment-bound sewage bacteria by dredging. Rittenberg et al. (1958) concluded that various pathogens and coliforms from sewage are sequestered in sediment, often some distance from the sewage deposit or point of discharge. There is evidence that the incidence of R^+ bacteria is not reduced in sewage effluents (Grabow et al. 1974), may be enriched in sewage maturation ponds (Grabow et al. 1973), and can be disseminated in marine environments (Feary et al. 1972; Smith 1971; Koditschek and Guyre 1974*a*).

Water samples collected from the Sandy Hook Park bathing area showed a higher incidence of multiple antibiotic resistant bacteria than the other water samples examined, although few could be detected in the surf sediment or in repeated sampling from an offshore site near the bathing area (sta. 39) (Table 1). The swimming area showed low coliform counts and clean surf sand, suggesting assay for antimicrobial resistant bacteria may be a useful indicator of recent or transient pollution in the bathing water.

Smith (1971) reported that ocean bathing water in England contained large numbers of bacteria showing multiple antibiotic resistance. Isolation from marine waters of R plasmid *E. coli* which transfer resistance to human pathogens has been documented (Feary et al. 1972; Grabow et al. 1974) and the transfer of R plasmids from fish pathogens (particularly plasmids coding for TE^R) has been reported (Aoki et al. 1971*a,b*; Watanabe et al. 1971). There is ample evidence that coliforms within the human alimentary tract could mobilize transfer of antibiotic resistance (Jarolmen and Kemp 1969; Smith 1971) to a wide variety of pathogens. It might be well to point out that *Aeromonas* sp. and many of the other opportunistic pathogens involved in nosocomial infections are gram-negative nonlactose fermenters which were not considered human pathogens until the recent increase in drug resistant infections became apparent (Grabow et al. 1974). (Nonlactose fermenters are excluded from routine examination under *Standard methods* procedures.)

Sewage treatment does not eliminate pathogens from the effluent (Rittenberg et al. 1958). Feary et al. (1972) found that about 1% of the coliforms in raw and treated sewage were multiple antibiotic resistant and that sewage treatment failed to reduce the incidence of R plasmid bacteria which transferred multiple antibiotic resistance. Studies of drug resistant coliforms in sewage maturation ponds indicate a marked increase ("threefold enrichment") in R^+ bacteria in the ponds, and the data suggest that a low level transfer of R factors occurs among bacteria in these toxic environments (Grabow et al. 1973). In an epidemic of shigellosis among swimmers in the Mississippi River (Shaeffer and Moriarity 1974), cultures isolated from several patients were identified as *Shigella sonnei* showing multiple antibiotic resistance. The antibiogram of the *Shigella* isolated from the patients was the same as that of *S. sonnei* isolated from the sewage disposal plant outflow 5 mi downstream. Multiple antibiotic resistance has been identified with other intestinal pathogens in water (Day et al. 1974; Grabow et al. 1974). Evans et al. (1975) emphasized that transferable plasmids in *E. coli* code for enterotoxin production and other virulence factors not yet fully identified.

It is likely that plasmid host strains of *Pseudomonas* are selected in polluted aquatic environments by virtue of their metabolic versatility in degrading polluting substrates and resistance to antimicrobials. The very characteristics of *Pseudomonas putida* that might render it a "safe host" for recombinant DNA experiments (Chakrabarty 1975), i.e. limitation of growth at the temperature of the human gut, may ensure its survival at low temperatures of the marine environment.

Previous analyses using *Standard methods* probably yielded less than optimal desorption of sediment bacteria. Improved methods of membrane filtration (Green et al. 1975; Rose et al. 1975; Sladek et al. 1975) may yield higher counts. Results reported here thus may represent only a fraction of the multiresistant populations resident at the sewage dump site. More effective elutant procedures and the use of other selective agents may yield more successful recoveries of sediment bacteria that are resistant to antimicrobial agents.

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Section 8 Shellfish and fisheries

Temporal and spatial distributions of benthic macroinvertebrates in the New York Bight

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Abstract

Benthic biota in the New York Bight apex is characterized by high spatial and temporal variability. A considerable reduction in numbers of individuals per sample was observed between August 1973 and August 1974; on an average, numbers of individuals per station decreased from 417 to 174. Low species diversity (< 0.500) was observed at some stations inside the Christiaensen Basin, an area characterized by high organic carbon values in sediment. Certain species, apparently tolerant of carbon-rich deposits of the Christiaensen Basin, were extremely abundant at some stations located in the basin. These included the anemone *Cerianthus*, four species of polychaete, and the bivalve *Nucula*.

Almost all quantitative data concerned with benthic populations of the New York Bight were gathered after 1960. Until 1968 few stations were sampled in the inner bight which might be affected by runoff from the Hudson River–Raritan Bay estuaries and ocean disposal of various domestic and industrial wastes and dredge spoils. Previous studies were generally concerned with distributions or occurrences of littoral species in New Jersey or New York metropolitan waterways (DeKay 1842; Hubbard and Smith 1865; Ingersoll 1881; Smith 1882; Smith 1887; Fowler 1911; Jacot 1920). The first comprehensive published investigations of the benthic fauna in waters adjacent to the bight were those of Dean and Haskin (1964) concerned with Raritan Bay. McGrath (1974) reported on continued studies of the benthic fauna in this bay.

Steimle and Stone (1973) discussed the distribution and abundance of benthic fauna off the southwest shore of Long Island in 1966–1967. Results of preliminary studies of the benthic fauna in and around solid waste disposal areas in the bight were provided by the National Marine Fisheries Service (1972) and Pearce (1972). Other data, in part gathered during the previous study and concerned with the distributions of toxic heavy metals, as well as the prevalence of certain pathological symptoms in fin- and shellfish, are given in Carmody et al. (1973), Young and Pearce (1975), Gopalan and Young (1975), and Ziskowski and Murchelano (1975).

Previous studies were concerned principally with the inner waters and sediments of the New York Bight usually referred to today as the bight apex. Pearce (1974), Leathem et al. (1973), Maurer et al. (1974), and Watling et al. (1974) reported on the benthic macrofauna off New Jersey south of the apex and off Delaware Bay. Wigley and McIntyre (1964) and Sanders et al. (1965) discussed the benthic fauna on the continental shelf northeast of the New York Bight. Hathaway (1966, 1967) provided limited qualitative data on benthic fauna of the northeastern continental shelf.

Since 1973 scientists of the Middle Atlantic Coastal Fisheries Center (MACFC) have investigated distributions, abundances, and temporal changes in benthic populations in the New York Bight and contiguous waters, including Long Island Sound, Raritan Bay, and those beyond the shelf–slope break. This paper discusses seasonal measurements and observations made at 64 stations located in the bight apex as well as single collections at over 100 stations located seaward of the general 40-m contour. The objectives of the studies include: determining distributions, abundances and seasonal variations in benthic populations in relatively unstressed areas to be able to assess change due to future environmental perturbations; detecting existing impacts of pollutants and measuring changes in distributions and abundances of benthic macrofauna; relating the distribution of benthic fauna to chemical/physical factors such as sediment type, measurable pollutants, and water column characteristics; synthesizing the various findings with the results of previous studies in the bight and other areas; and applying the results to problems of marine fisheries management.

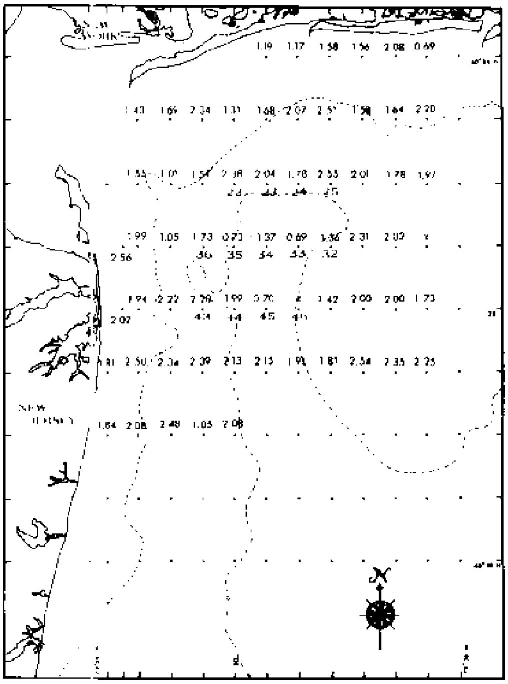

Fig. 1. Location of standard MESA-SYMAP sampling stations in the New York Bight apex. Diversity indices (H') calculated for the first grab sample at each station during the August 1974 cruise are given *above* the points indicating station locations. Stations discussed in Table 1 are indicated by arabic numbers located *below* specific points. Stations located in high organic carbon sediments have low diversity indices.

Materials and methods

Samples were collected during five quarterly cruises at MESA-SYMAP stations (Fig. 1) using a modified Smith-McIntyre grab sampler of 0.1-m^2 surface area (Smith and McIntyre 1954). Five replicate grab samples were collected at the first 66 stations and commencing with the third quarterly cruise at all 103 stations. Subsamples (taken with a 3.4-cm-diameter core to a depth of about 10 cm) were removed from the grab before discharge of sediments into plastic containers; subsamples were used for standard geological, heavy metals, and organic matter analyses. Subsamples were frozen until analyses were performed.

Sediments were washed through stainless steel screens (minimum aperture of 1.00 mm) to separate the biota. The biota was fixed for several hours in buffered Formalin ($\simeq$10%) and preserved in 70% ethanol containing 5% glycerine. All biological samples were accessioned, sorted, and identified to lowest taxonomic levels possible.

Data from each sample were then used to compute diversity as H' (Shannon and Weaver 1963), evenness, total individuals, and total species. All data, including species lists, are keypunched, verified, and machine listed. Biological data and data concerned with heavy metals abundance, sediment characteristics, seabed oxygen consumption, bottom dissolved oxygen, and other physical/chemical parameters collected at MESA-SYMAP stations are then ready for eventual multivariate analyses.

A combination of LORAN-A and Raydist precision navigation was used to locate and maintain station position while samples were being collected.

Results

Distribution and abundance of benthic organisms; August 1973—Preliminary analysis of single grab samples taken at MESA stations in coastal and inshore shoal waters indicate a fauna characterized by the bivalves *Tellina agilis* and *Spisula solidissima* and the sand dollar *Echinarachnius parma*. The faunas were similar to those found by Steimle and Stone (1973) off Long Island and by Pearce (1974) off southern New Jersey. They are characteristic of high energy coastal marine environments.

Samples taken in August 1973 and August 1974 at standard stations throughout the bight apex yielded most of the species found in an earlier study (NMFS 1972). Collections in August 1973 contained 146 species from 64 sampling stations. The number of individuals per station ranged from one (station 40) to 4,158 (station 35) and averaged 417. The number of species ranged from one (station 40) to 34 at station 44. The average number of species per station was 19.

Seven species occurred at 50% or more of the sampling stations. These were a rhynochocoel (found at 42 stations); the polychaetes *Glycera dibranchiata* (33), *Nepthys bucera* (35), *Spiophanes bombyx* (43), and *Tharyx acutus* (49); the bivalve *T. agilis* (46); and the brachyuran crab *Cancer irroratus* (46). In addition to the foregoing, eight species were found at 25 or more stations (40% of the total stations). These included the burrowing sea anemone *Cerianthus americanus* (29); an archiannelid, *Protodrilus* sp. (25); the polychaetes *Aricidea jeffreysii* (25), *Lumbrineris fragilis* (25), and *Asabellides oculata* (28); two bivalves, *Nucula proxima* (30) and *S. solidissima* (23); and the grass shrimp *Crangon septemspinosa* (29).

Finally, seven species occurred at between 19 and 24 of the stations sampled (30–39% of all stations). Included were the polychaetes *Hemipodus armatus* (20), *Spio filicornis* (21), *Prionospio malmgreni* (22), *Lumbrineris tenuis* (21), and *Pherusa affinis* (19); the bivalve *Ensis directus* (19); and the isopod *Edotea triloba* (21). Thus only 22 species were found at over 19 of the stations sampled during the August 1973 cruise.

Thirty-five of the species were found only at one sampling station. Several of these showed a highly restricted distribution, however. For instance the polychaete *Polydora caulleryi* was found only at station 7 but there were 197 individuals present in a sample of 0.1 m^2. Another polychaete, *Spirorbis borealis*, was found only at one station but was represented there by 162 individuals. Certain of the amphipods also showed a highly contagious distribution. For instance, although *Parahaustorius holmesi* was found only once, there were 25 individuals in a single grab sample. *Parahaustorius attenuatus* was also represented by 29 individuals at only one sampling station. Another haustorid amphipod, *Protohaustorius wigleyi*, was found at 11 sampling stations in numbers ranging from 2–15; one station, however, yielded 96 individuals. Again, the most widely distributed amphipod, *Unciola irrorata*, was usually represented by 1–12 individuals, but 108 were found at station 55. In an earlier study this species occurred in even larger numbers (NMFS 1972).

Other taxa also had species which showed highly contagious distributions. At 22 stations the protobranch bivalve *N. proxima* was represented by no more than 15 individuals but at six stations between 50 and 570 were found and at two stations (34 and 35) 1,439 and 3,650 were counted. The latter stations are characterized by highly organic sediments.

Distribution and abundance of benthic organisms; August 1974—Benthic samples taken during the quarterly cruise completed in August 1974 yielded 136 species from 64 sampling stations. The number of individuals per station ranged from zero (stations 29 and 46) to 1,826 (station 35) and averaged 174. The number of species ranged from zero (stations 29 and 46) to 39 at station 22. The average number of species per station was 15.

Only three species occurred at 50% or more of the 64 sampling stations: a rhynchocoel (found at 38 stations); the polychaete *T. acutus* (33); and the bivalve *T. agilis* (46). In addition to the foregoing, three species were found at 25 or more stations (40–49% of the total stations): two polychaetes, *G. dibranchiata* (29) and *L. fragilis* (27), and the bivalve *N. proxima* (27).

Finally, four species occurred at between 19 and 24 of the stations sampled (30–39% of all stations): the polychaetes *L. tenuis* (23), *Driloneris longa* (21), and *Ninoe nigripes* (20); and the anemone *C. americanus* (20). Thus during the August 1974 cruise only ten species were found at over 19, or 30%, of the stations sampled.

Thirty-six of the species were found only at one sampling station. Again, as was noted in samples taken in August 1973, several of these infrequently occurring species showed a highly contagious distribution. For instance, *Sphaerosyllis erinaceous* occurred only at one station but was represented by 91 individuals.

Nucula proxima again showed a highly contagious distribution. It occurred in numbers between 1 and 185 at 24 stations but at stations 34, 35, and 45 it was represented by 809, 1,568, and 592 individuals, respectively. Interestingly, this bivalve was represented in large numbers only at the same three stations in August 1973. Data for collections made in March 1974 indicate an almost identical pattern of distribution and abundance.

Samples collected at the same stations in March 1974 yielded a total of 136 species with an average of 293 individuals and 17 species per station. There is thus indication that the numbers of individuals were significantly reduced in August 1974 relative to August 1973 and March 1974. The distributions and abundances of certain species were considerably reduced. For instance, *C. irroratus* was found at 46 stations in 1973 but occurred at only 10 in 1974. However, in 1973 the average number of crabs per station was 3.7 whereas in 1974 the average was 2.1, a sizable reduction in the total numbers of crabs found in the bight apex. Other species showed a similar reduction. In fact, the average number of individuals per station decreased from 417 in August 1973 to 174 in August 1974 and the average number of species per station declined from 19 to 15.

A few species, including the deposit-feeding bivalve *T. agilis*, apparently remained more or less constant in their distribution and abundance in the interim between August 1973 and August 1974. Additional analyses and collections are being conducted to determine if large temporal change is characteristic of benthic fauna in the bight. Determination of natural variation is essential to understanding changes caused by environmental stresses.

Community structure—Samples from several stations were characterized by unusually low species diversity (H') in the areas most heavily impacted by sewage sludge and contaminated dredge spoils disposed of in the bight apex (Figs. 1, 2, and 3). Stirn et al. (1975) noted that when organic effluents such as sewage sludge are introduced into the marine environment a complex of primary effects may occur, including elimination of species or entire taxonomic groups which are intolerant of the pollution, dominance of detritophagous deposit feeders and certain groups of tolerant suspension feeders, and suppression of

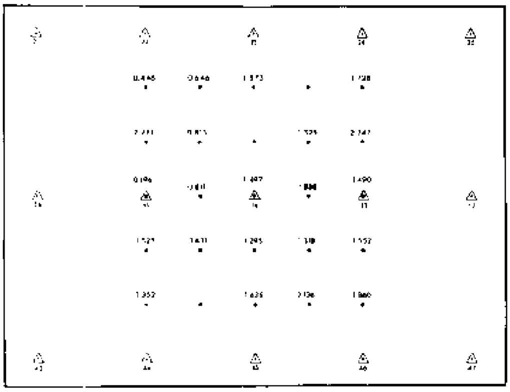

Fig. 2. Indicates the location of special sampling stations "nested" over the high organic carbon area, collected on 12–24 February 1975 within the grid of standard MESA-SYMAP stations; the MESA-SYMAP stations are indicated by triangles (*see Fig. 1*). Note the unusually low diversity indices generally observed at stations in the northwest sector of the special nested stations.

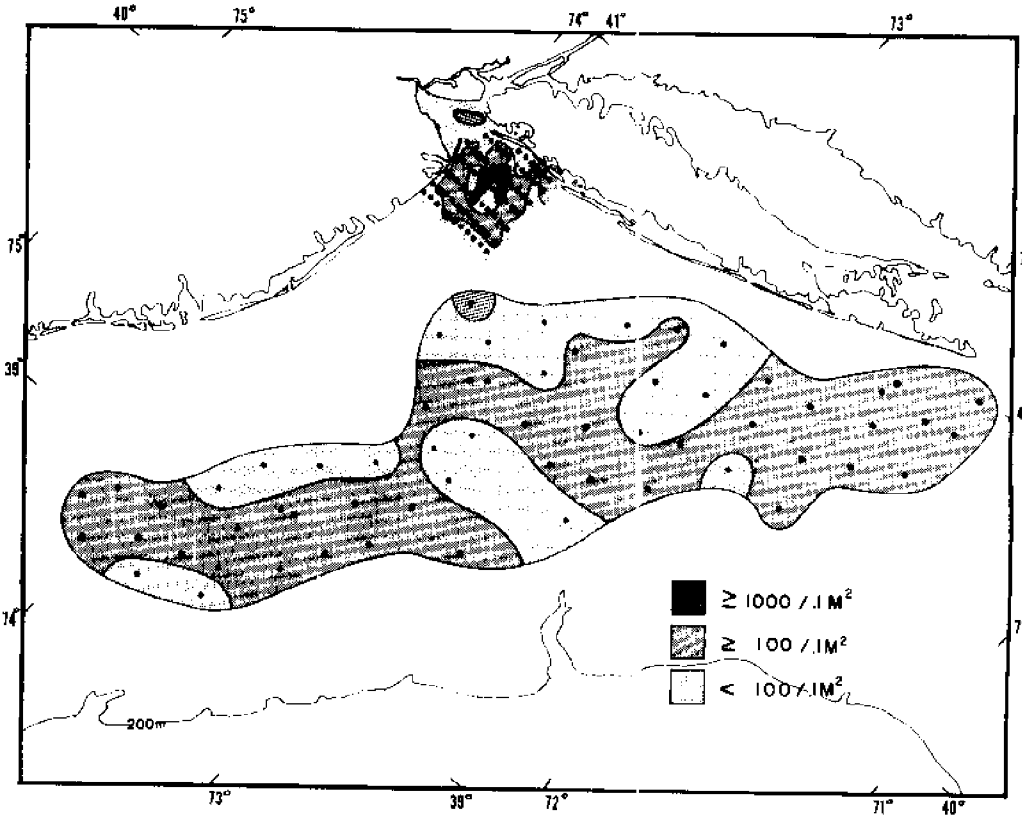

Fig. 3. Numbers of benthic macrofauna per 0.1 m² as estimated from Smith-McIntyre grab samples. Sampling stations are denoted by dots, with two grabs per station in the apex. Sampling density (78 stations) in Raritan Bay south of New York City does not permit indication of sampling locations. (After Pearce and Radosh in press.)

most carnivores and herbivore macrofeeders. Benthic communities in the highly organic sediments of the Christiaensen Basin of the bight apex resemble stressed communities described by Stirn et al. (1975) as characteristic of polluted marine ecosystems, i.e. diversity was sometimes reduced, entire intolerant taxonomic groups (i.e. Amphipoda) expected in samples were not present, certain deposit feeders occurred in great abundance and carnivores were absent. Species which occurred in extreme abundance in samples from carbon-rich deposits included the anemone *C. americanus*; the polychaetes *Capitella capitata, Nephtys incisa, Mediomastus ambiseta*, and *P. affinis*; and the protobranch bivalve *N. proxima*. The numbers of individuals of each of these species collected on three different sampling dates are given in Table 1. The stations are arranged in this table according to west–east transects, with stations 22–25 located along the northern margin of the high organic portions of Christiaensen Basin, 30–32 crossing the central portions, and 43–46 at the southern edges of the areas enriched by organic matter. These data, when considered with others, indicate abnormally high standing stocks of pollution-tolerant detritus and suspension feeders capable of utilizing the great amounts of organic matter present in the basin.

Equally important, several species usually found in the fine sands and muds of the New York Bight apex infrequently or never occur in those portions of the bight characterized by carbon-rich sediments. Samples from high organic areas of the Christi-

Table 1. Indicates the distribution and abundance of species of benthic macroinvertebrates diagnostic for high organic carbon areas within the New York Bight apex*. (*See Fig. 1 for station locations.*) These organisms generally were restricted to the high organic carbon areas which are believed to receive sewage sludge from ocean disposal. Stations 22–25 are on an east–west transect along the northern periphery of high organic carbon sediments. Stations 36–32 are in the high organic carbon sediments and stations 43–46 are south of the high organic carbon area.

Sta.	22	23	24	25	35	34	33	32	43	44	45
Cerianthus americanus											
Aug 73	4	57	42	12	27	14	5	3	2	1	32
Mar 74	0	19	30	19	27	29	0	3	4	2	10
Aug 74	6	31	22	19	31	25	0	2	2	8	32
Capitella capitata											
Aug 73	0	0	0	0	0	0	0	1,525	0	0	0
Mar 74	0	0	0	0	0	0	962	4	0	0	0
Aug 74	0	0	0	0	0	0	7	7	0	0	0
Nephtys incisa											
Aug 73	3	42	23	24	37	30	0	0	6	0	22
Mar 74	0	25	38	15	51	18	1	0	15	3	8
Aug 74	0	50	62	20	24	30	0	0	0	21	16
Mediomastus ambiseta											
Aug 73	0	255	436	0	15	24	0	0	8	17	281
Mar 74	0	15	6	3	0	45	0	0	0	18	3
Aug 74	19	8	0	3	0	70	0	0	2	1	1
Pherusa affinis											
Aug 73	9	74	48	16	26	16	0	0	12	3	114
Mar 74	0	4	23	18	23	68	2	0	4	2	28
Aug 74	10	12	13	13	13	15	0	0	0	4	2
Nucula proxima											
Aug 73	23	304	232	15	3,650	1,439	0	0	5	7	573
Mar 74	1	320	26	18	4,200	527	0	0	3	3	11
Aug 74	2	0	42	20	1,568	809	0	0	0	87	592

*Station 36 had 4 *C. americanus* in March 1974 and 1 *N. proxima* in August 1974. Station 46 had 1 *C. americanus* and 1 *M. ambiseta* in August 1973.

aensen Basin rarely contained members of the Amphipoda while similar sediments outside the high organic areas may contain hundreds of individuals per grab sample (0.1 m^2). Other taxa, the polychaetes *P. malmgreni* and *D. longa*, the bivalve *T. agilis*, and the canceroid crab *C. irroratus*, are found at stations 22–25 and 43–46 but not at stations 34–35 within the central portions of the high organic area. Finally, the extremely abundant and widely distributed polychaetes *S. bombyx* and *T. acutus* are reduced in numbers at stations 36–32 in the central portions of the Christiaensen Basin.

Outer shelf benthic communities—Samples collected from outer continental shelf and slope areas have been analyzed and reported (Pearce 1975; Pearce et al. 1975). The distributions of several species common to the bight apex are continued across the shelf and beyond the shelf–slope break. Moreover, stations farther offshore but on the shelf have about the same number of individuals and species as do the unpolluted or relatively unstressed stations found in the shallower bight apex (Figs. 3 and 4). Samples taken beyond the shelf–slope break had fewer individuals and species when compared to shelf stations.

It is interesting that certain species commonly found in the uncontaminated portions of the bight apex, for instance the bivalve *T. agilis* and the crab *C. irroratus*, were not found at shelf stations located in 40–70 m of water.

Discussion and conclusions

The bight apex benthic biota is characterized by high spatial variability. Sedi-

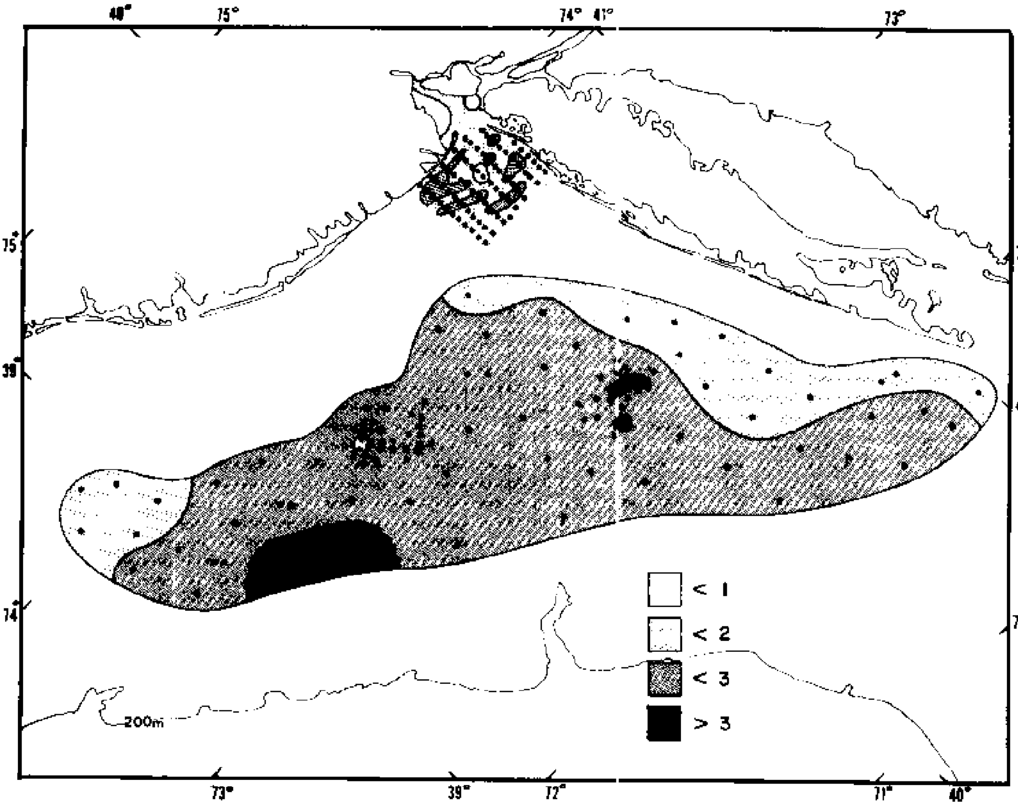

Fig. 4. Species diversity (*H'*) of benthic macrofauna as estimated from Smith-McIntyre grab samples. Sampling stations are denoted by dots, with two grabs per station in the apex. Sampling density (78 stations) in Raritan Bay south of New York City does not permit indication of sampling locations. (After Pearce and Radosh in press.)

ment types vary owing to relative scouring and deposition by wave and current actions. Deeper areas (30–35 m) tend to accumulate finer carbon-rich deposits. The distribution of benthic organisms and communities is closely correlated with sediment types (Sanders 1958); thus spatial change in sediments will be accompanied by variations in the composition of the benthic biota.

Temporal variability occurs in the biota of the bight apex. There was a considerable reduction in numbers of individuals per sample in August 1974 when compared to similar samples collected in August 1973. Although it is generally assumed that reproduction, colonization, survival, and subsequently, abundance are depressed during winter, our data suggest that there was relatively little change in species composition and abundance between August 1973 and March 1974; the greatest change appeared to occur between March and August 1974. Thomas et al. (1976) indicated that considerable variation occurs in seabed oxygen consumption at the MESA-SYMAP stations. Seasonal variation was also noted in heavy metals (Greig and Thomas unpublished).

Although there was a general decline in the numbers of individuals and species at each station between August 1973 and August 1974, the species composition at each station remained relatively consistent, i.e. stations characterized by high levels of organic matter and heavy metals as well as specific sediment type had similar populations dominated by pollution-tolerant species during the two sampling periods. As previously noted, certain species remained unchanged in abundance during the 1-year interim between sampling periods whereas others declined markedly in abundance and distribution.

Perhaps the most interesting observation was the reduced diversity observed at some stations in the carbon-rich deposits of the Christiaensen Basin. The highly organic sediments are thought to result from several factors, the most important probably being ocean disposal of solid waste. Several studies of heavy metals distribution (Carmody et al. 1973), microorganisms (Koditschek and Guyre 1974), human artifacts (Pearce 1971), and hydrocarbons (J. Thomas personal communication) suggest that the accumulated organic matter in the basin is largely the result of ocean disposal. As stated by Stirn et al. (1975) the usual response of the exposure of marine biota to organic pollution is a marked decrease in diversity with a great increase in the abundance of certain pollution-resistant species. Other investigators have reported similar effects when marine communities are faced with deteriorated conditions by sewage pollution (Grigg and Kiwala 1970; Jones 1973; Haedrich 1975). Littler and Murray (1975) also noted that certain macroinvertebrate species are often reduced in number at the center of pollution or stress but increase in abundance at the periphery of the waste discharge plume. These species were apparently those which can utilize particulate organic matter. Again, this same phenomenon has been observed in the bight, where ocean dumping has resulted in an irregular distribution of organic matter with extensive boundaries which serve as ecotones where detritus feeders can attain extremely high standing stocks. Similarities in chemical/physical and biological data collected during the present study and data and observations collected earlier (NMFS 1972) suggest that the situation may represent a stable condition, with only long term changes. We have other data, however, that indicate annual changes in biota and physical conditions may occur and the bight benthic ecosystem may be more dynamic than previously suspected. Littler and Murray (1975) and Stirn et al. (1975) both suggest pollution effects may duplicate the effects of naturally occurring events such as perturbations accompanying major storms. In both the "natural" and pollution-induced situations, periodic stress results in the elimination of intolerant species, with possible magnification of tolerant or opportunistic species due to increased particulate organic matter utilized as food. This is comparable to the maintenance of an early disclimax successional stage.

A major unknown in these situations is the impact of such conditions on the larger marine biome. Do significant reductions in macroinvertebrate biomass in anoxic environments such as observed in the bight result in any significant impact on major finfish and shellfish resources? On the other hand, does the augmentation of pollution-resistant, opportunistic, detritus feeders result in increased biomass available as food to the finfish? To date no conclusive evidence is available for the New York Bight. Grigg and Kiwala (1970) noted that an outfall off Palos Verdes Peninsula, California, caused localized siltation by organic matter with a concomitant reduction in epibenthic species and finfish. They attributed declines in finfish and certain macroinvertebrates to reduction in foods within certain portions of the normal food web.

It is known, however, that the presence of organic matter from sewage may have effects other than those associated with feeding activities. Elevated values for coliform and other bacteria as well as toxic heavy metals have been measured in the bight (Koditschek and Guyre 1974; Carmody et al. 1973). These organisms and toxic materials do have an impact on living resources; preliminary studies (Young and Pearce 1975; Gopalan and Young 1975; Ziskowski and Murchelano 1975) indicate unusual pathological conditions in finfish and shellfish from the bight apex and nearby polluted estuaries. Some forms affected are important as economic resources and others have an indirect, but important, effect on resources because of their role in the marine food web.

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Distribution and abundance of inshore populations of the surf clam *Spisula solidissima*[1]

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Abstract

The distribution and abundance of juvenile and adult surf clams were determined inshore of the 3-mi contour in July 1974 at 141 sites off Long Island, New York. Transects were located every 2 mi from Montauk to Rockaway Point, and samples were taken at 0.5, 1.5, and 2.5 mi offshore. Selected stations were revisited in July 1975.

East of Shinnecock Inlet adult clams occurred at densities between 0.5 and 3.0 bushels per dredge haul. West of Shinnecock adults declined gradually, reaching a minimum west of Jones Inlet to East Rockaway Inlet. Abundances increased precipitously off Rockaway Beach.

Highest densities of juveniles occurred at 0.5 mi and decreased rapidly farther offshore. Juveniles were more abundant at the west end of Long Island, particularly inshore at 0.5 mi. Higher densities of juveniles here may result from accumulation of larvae produced farther east and transported westward via longshore currents. The convergence of tidal and longshore currents may effectively "trap" larvae off western Long Island.

Clam stocks off eastern Long Island probably are older than 9 years and are mostly composed of 1–3 age classes. Clam stocks off the Rockaways appear younger. The commercial fishery is apparently dependent on massive settlements of larvae occurring irregularly and infrequently.

The surf clam *Spisula solidissima* is of great commercial importance along the mid-Atlantic seaboard and is also the basis of a small commercial fishery on Long Island. However, *Spisula* is ecologically significant beyond commercial value. It is the most characteristic macrobenthic suspension-feeding invertebrate occurring within the shifting sand habitats of the inner shelf. It is widely distributed and frequently abundant over most of the inner shelf south of Rhode Island to Cape Hatteras. Its occurrence is predictable where the proper combinations of sand, temperature, and water movement occur. Because it is commonly found over a wide geographical area within predictable and fairly narrow environmental limits, potential environmental changes affecting this species in the inner New York Bight will be discernible by comparison with nearby areas. Though the potential exists at present to use population data to recognize and evaluate such changes, our continued ignorance of many aspects of population dynamics and ecology of *Spisula* remains a drawback.

Distributions and abundance of stocks of adult *Spisula* off Long Island (beyond 3 mi) and throughout the Middle Atlantic Bight have been discussed elsewhere (Merrill and Ropes 1969; Ropes and Merrill 1971; NMFS 1974). The surf clam is frequently abundant in shallow water, however, and a significant proportion of total fishing effort for this species takes place inshore of the 3-mi limit (H. Haskin personal communication).

This paper deals with aspects of the distribution and abundance of inshore populations of *Spisula* south of Long Island from Montauk to Rockaway Point. Although this project was not undertaken specifically to investigate the impact of environmental changes on *Spisula* populations, the results provide a useful framework for interpreting environmental effects. The western third of the Long Island coast is apparently an important area for recruitment of juvenile *Spisula* and corresponds with the area most immediately endangered by spoils and sewage dumping.

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[1] This research was supported by NOAA, National Marine Fisheries Service, State-Federal Fisheries Management Program contract 03-4-043-355.

Methods and materials

Between 13 and 19 August 1974, 141 sites were sampled. Each station was at the center of a rectangular quadrat of about 2 mi^2. Stations were located on 47 transects normal to the shore at 0.5, 1.5, and 2.5 mi offshore. For analytical purposes, transects were grouped in fours, beginning at Montauk Point, Long Island. Thus 12 groups (designated A–L; group L contained only three transects) encompassed the entire coast (Fig. 1).

Adult clams (*Spisula* and *Arctica*) were collected with a 48-in (1.2 m) commercial hydraulic clam dredge operated from a commercial clamming vessel. The dredge was towed for 5 min at each station. Two sediment samples were collected at each station using a spring-operated Shipek grab sampler (sampling area = 1/25 m^2). Samples were screened immediately with a 1-mm sieve; the fraction larger than 1 mm was preserved in neutralized Formalin and taken to the laboratory where it was sorted for juvenile *Spisula, Tellina,* and other benthos.

Off the Rockaways 14 stations were sampled for juvenile surf clams and *Tellina* in July 1975. These samples were treated as above.

Measurements of juvenile bivalves were made using a calibrated Wild dissecting microscope. One-bushel samples of adult clams were measured at all stations where total catch exceeded 1 bushel. Along the eastern Long Island coast (transects 1–30: Fig. 1), clams per bushel ranged from 40–57, with a mean of 49 mm. West of transect 30, clams per bushel ranged from 78–125, with a mean of 86 mm. Adult clams were measured to the nearest 1 mm with a special clam board devised by the National Marine Fisheries Service.

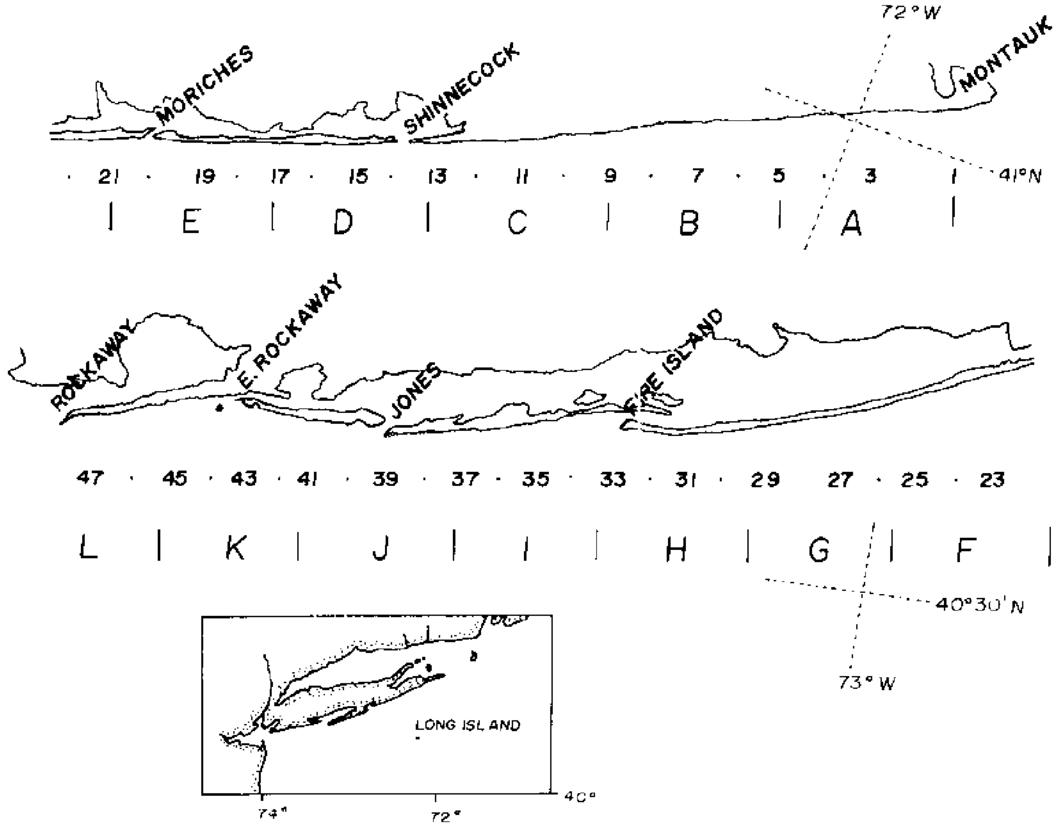

Fig. 1. Station locations. Numbers show transect locations; letters show transect groups.

Samples were retained from each station for sediment grain-size analysis. In samples containing significant quantities of clays, the silt-clay fraction was split from the total sample before dry-sieving the sand fraction, and the dry weight of the silt-clay fraction was included in the total weight (Holme and McIntyre 1971). The following sieve sizes were used: 4 mm ($-2.0\ \phi$), 2 mm ($-1.0\ \phi$), 1 mm ($0\ \phi$), 0.5 mm ($+1.0\ \phi$), 0.25 mm ($+2.0\ \phi$), 0.125 mm ($+3.0\ \phi$), 0.062 mm ($+4.0\ \phi$). The sediment characterizations used below refer to the modal particle size as determined from the size-frequency analyses for each station.

The size structure and inferred age structure of adult clam samples were determined by size-frequency analysis of all samples of adult clams yielding greater than 0.9 bushels per 5-min dredge haul (a total of 48 stations in this survey). Age inference from size data was based on a growth curve for Long Island surf clams (Westman and Bidwell 1946). The predictive value of this curve as well as all other growth curves for *Spisula*, based on shell growth ridges, has yet to be verified by independent analyses. Thus, the age structure discussed below provides, at best, only a rough approximation of the ages of clams dominating these populations.

Results

Distribution and abundance of adult Spisula—Beginning west of Montauk for about 24 mi (groups A–C), clams occur at relative densities of 1–1.5 bushels per dredge haul (Fig. 2). West of Shinnecock (beginning in group D), abundances decline gradually for about 56 mi until a minimum abundance is observed in group K (south of Long Beach). West of East Rockaway Inlet, abundance of adult clams increases sharply.

Vertical (inshore–offshore) patterns are similar at 1.5 and 2.5 mi, whereas the inshore pattern (0.5 mi) differs in the absence of clams west of Moriches Inlet and in the quantities of clams (0.5 bushels) west of Fire Island Inlet. The sharp increase off the Rockaways occurs both inshore and offshore. In contrast to the patterns in juvenile abundance presented below, the patterns of adults at 1.5 and 2.5 mi are not merely diluted reflections of a strong inshore pattern.

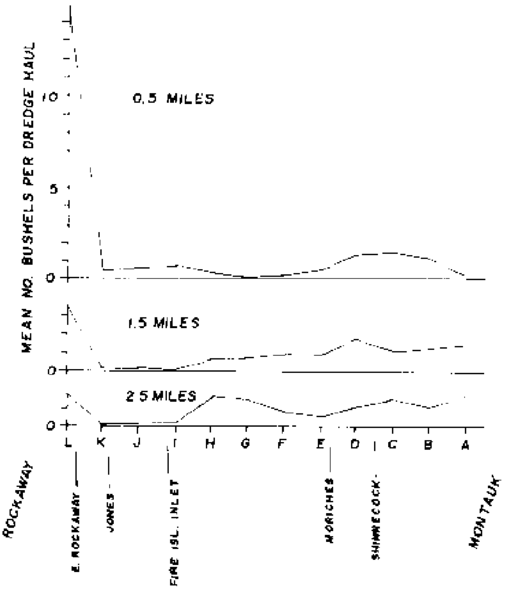

Fig. 2. Abundance of adult surf clams (*Spisula*) off Long Island.

Distribution and abundance of juvenile Spisula—The size-frequency distribution of juvenile surf clams from bottom grab samples is shown in Fig. 3. Most individuals are less than 5 mm, with the modal group in the 2.5–2.9-mm range. All clams in this distribution are assumed to be in their first season of growth, that is, zero-class animals. The distribution appears skewed toward small-sized individuals, but this probably results from the bias imposed by the 1-mm sieve. The tail of the distribution extending between 5 and 10 mm reflects the somewhat faster growth rates of a small proportion of individuals, such as may occur in unusually favorable locations. It could also result from earlier spawnings. There are no data to support either alternative.

Patterns in abundance of juvenile clams are shown in Fig. 4. Density is low near Montauk (group A) but increases gradually westward to a peak of 2.5 animals per sample at group E, just east of Moriches Inlet. Numbers decline westward, but in the area of group I, near Fire Island Inlet, num-

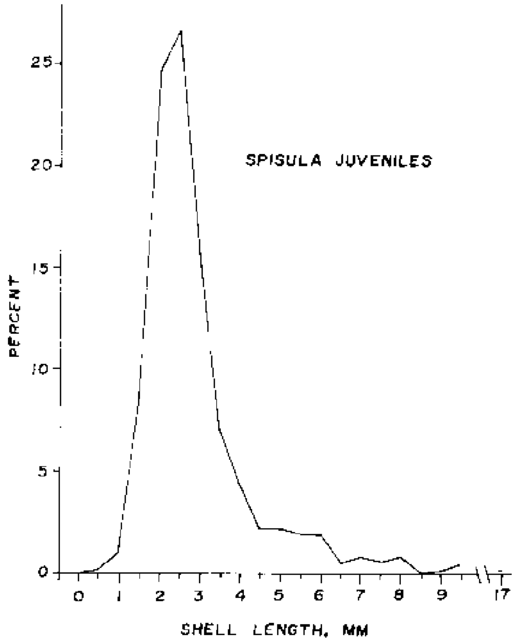

Fig. 3. Size-frequency distribution of juvenile surf clams based on all bottom grab samples from August 1974.

bers again increase dramatically and remain high (6–12 per sample) between Fire Island and East Rockaway Inlets.

In addition to the east–west pattern of abundance, there are also inshore–offshore abundance patterns of juveniles. Inshore, maximal abundance occurs in the region from D–F and declines westward only to increase sharply west of Fire Island Inlet and remain high to Rockaway Beach.

Abundance decreases rapidly with increasing distance offshore. At 1.5 mi (Fig. 4) the longshore patterns are still evident. At 2.5 mi, only the western accumulation of clams has a clearly defined pattern. Moreover, it is evident from Fig. 4 that the overall patterns described above primarily reflect the inshore patterns, where 71% of all juvenile clams were collected.

Distribution and abundance of associated macrobenthos—The molluscan species occurring most frequently in bottom samples containing juvenile *Spisula* was the bivalve *Tellina agilis*. Although *Tellina* and *Spisula* occur together in 58% of bottom grab samples, *Tellina* occurs alone in 32%. Chi-square analysis of the paired-species contingency table indicates a lack of significant association between these species ($\chi^2 = 0.541$). Size-frequency analysis of *Tellina* (Fig. 5) shows that although animals ranged from 1–13 mm, most individuals were less than 5 mm. It is uncertain whether the bulk of these small individuals comprise a single year class or a combination of several. However, since modal groups occurred at 2.5 mm, 4–4.5 mm, and 6–6.6 mm in both 1974 and 1975 (Fig. 5), it is possible that these represent year classes. Unfortunately, this cannot be confirmed from present data.

The abundance of *Tellina* is indicated in Table 1. The major variable affecting abundance is apparently associated with depth. In general, stations at depths >18 m (for example, groups A–F, 2.5 mi offshore) yielded mean estimated densities of 10–

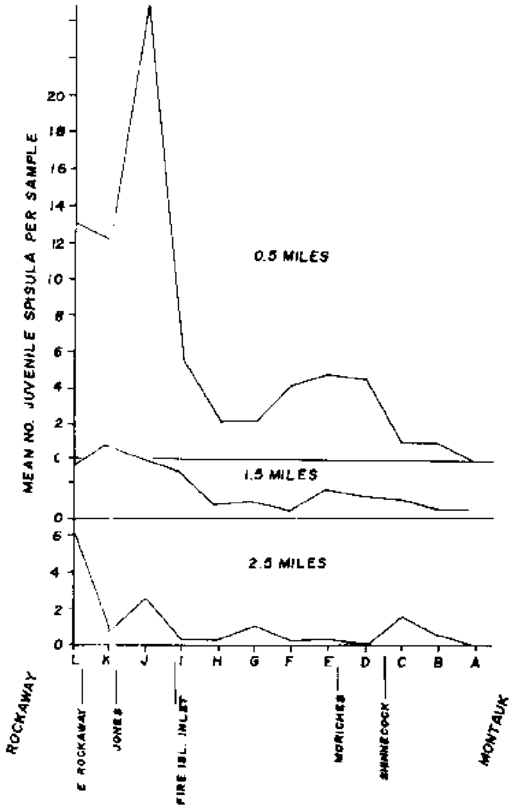

Fig. 4. Abundance of juvenile surf clams off Long Island.

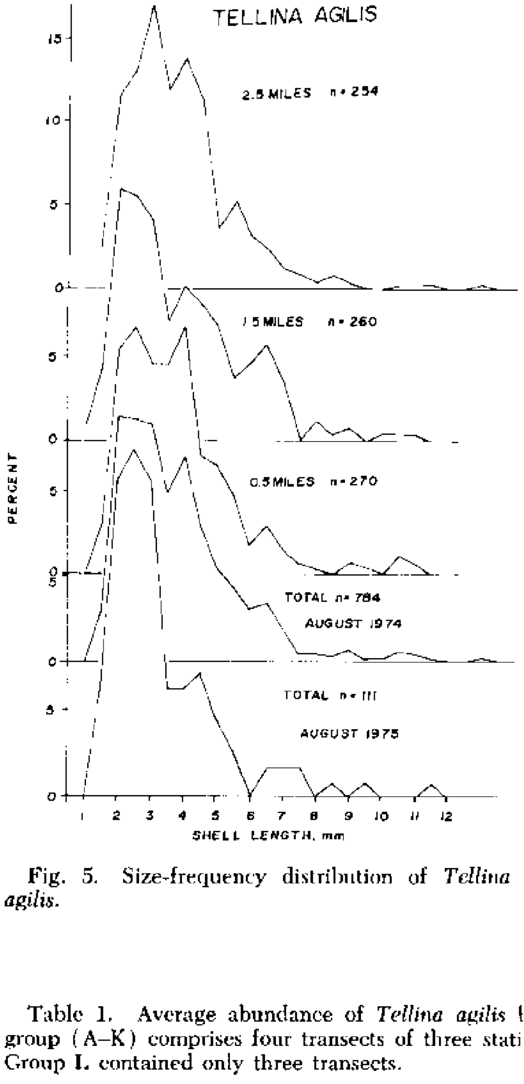

Fig. 5. Size-frequency distribution of *Tellina agilis*.

40/m². Stations in depths <18 m yielded densities of 80–140/m². However, variability among stations is high. A few stations in this survey had estimated densities of 700–1,200/m² (the low mean densities in Table 1 are a result of averaging data from 12 stations in each group).

The other bivalve species occurring frequently with juvenile *Spisula* in bottom grab samples was *Astarte castanea*. Chi-square analysis of association between *Spisula* and *Astarte* indicates the absence of statistically significant association between these species ($\chi^2 = 0.017$). *Astarte* collected in this survey ranged from 1–26 mm, that is, up to the maximum size of the species (Abbott 1974), although most specimens were less than 7 mm.

The sand dollar *Echinarachnius parma* occurred in bottom samples at 45 stations, mostly east of Moriches Inlet. Estimated densities ranged from 12–174/m², with an average of 36.6 (stations lacking animals not included in calculations). Animals ranged from 26–53 mm in diameter. The size-frequency distribution was bimodal with peaks at 34- and 36-mm diameter.

The ocean quahog *Arctica islandica* did not occur in any of the bottom grab samples. However, extensive populations of adult *Arctica* were observed off eastern Long Island, beginning at transect 24 (off

Table 1. Average abundance of *Tellina agilis* based on bottom grab samples, August 1974. Each group (A–K) comprises four transects of three stations each so mean values are based on 12 samples. Group L contained only three transects.

	0.5 mi			1.5 mi			2.5 mi		
Group	Mean/ sample	Est. mean/ m²	Mean depth (m)	Mean/ sample	Est. mean/ m²	Mean depth (m)	Mean/ sample	Est. mean/ m²	Mean depth (m)
A	0.2	4	11.6	0.5	10	16.8	0.7	14	21.0
B	4.0	80	10.4	6.0	120	17.4	1.2	24	21.6
C	6.2	124	9.1	5.0	100	17.7	2.2	44	23.2
D	4.5	90	11.0	16.0	320	18.0	2.0	40	23.8
E	7.7	154	8.2	5.0	100	18.6	1.0	20	23.5
F	7.7	154	10.1	2.5	50	18.6	1.5	30	22.3
G	2.7	54	10.1	3.0	60	14.3	4.5	90	18.6
H	6.5	130	8.8	11.0	220	15.6	7.2	144	18.9
I	5.0	100	7.3	6.0	120	11.6	28.2	564	15.2
J	4.7	94	6.4	4.5	90	10.7	5.0	100	13.1
K	14.2	284	7.6	4.0	80	11.9	6.5	130	13.7
L	3.7	74	6.7	3.3	66	9.1	6.0	120	10.4
$\bar{x}$		111.8			111.3			110.0	

Fire Island), and were present almost to Montauk. In this area, they often occurred with *Spisula* in the same dredge haul (*see Fig. 8*).

Sediment texture—Inshore bottom sediments off Long Island range from predominantly very coarse sand (<0 ϕ) to predominantly very fine (4 ϕ); medium-to-fine sands characterized most stations surveyed. The distribution of bottom sediments (Fig. 6) indicates, however, that fine sand predominates from east of Fire Island Inlet to Rockaway Point as well as inshore close to Moriches and Shinnecock Inlets. Fine sediments also prevail at outer, deeper stations off central Fire Island (transect 24), eastward to off Shinnecock and off eastern Long Island. Gray clay deposits sampled at several locations from east of Jones Inlet to Shinnecock, both inshore and offshore, are most likely Holocene lagoonal deposits. With few exceptions, these clays were not evident in the benthic grab samples but were noted, sometimes in massive quantities, in the clam dredge hauls.

Size and age structure of adult Spisula—Size-frequency and probable age-frequency distributions of adult clams are shown in Fig. 7. Size and age structure at stations 362–461 differ from the remaining stations in the predominantly small size and absence of large and presumably old animals. The area involved is from east of Jones Inlet to Rockaway Point, the westernmost portion of the survey area.

A marked change in size appears east of Fire Island Inlet between stations 362 and 310. To the east, all populations are broader in size range, due particularly to the presence of larger, older clams; most of these populations lack significant numbers of the young clams seen near the Rockaways.

The age-frequency distributions (Fig. 7B) are characteristically either unimodal or bimodal with a minor mode. An analysis of all the age distributions indicates that populations west of Jones Inlet exhibit major modes corresponding with ridges 5 and 7, whereas most sites east of Fire Island Inlet have major modes corresponding with ridges 1 –14. The analysis of major and minor modes for all collections suggests that recruitment at a rate sufficiently large to produce a recognizable mode occurred at one or more sites every year except the years corresponding to ridges 6 and 8. However, individual populations are usually dominated by one or two major modes.

Discussion

The size and age distribution data presented above imply that major settlements

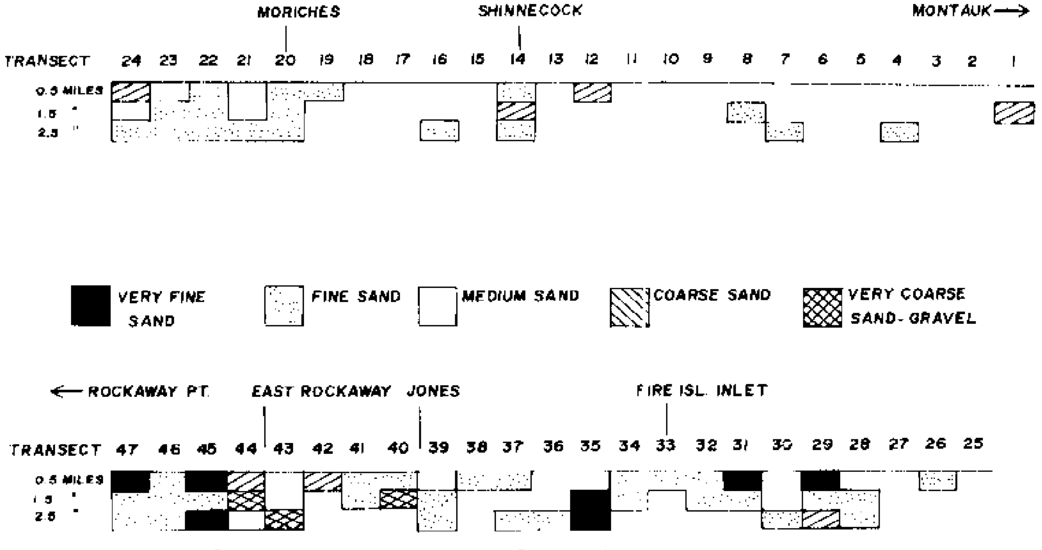

Fig. 6. Modal sediment particle size based on analysis of grab samples, August 1974.

of surf clams have occurred near the western end of Long Island more recently, compared with the eastern populations, which are predominantly older. Indeed, if the ridge numbers used in predicting age from size are approximately accurate, about 90% of the standing stocks of commercial-size clams off eastern Long Island are older than 9 years. Even with the serious difficulties involved in determining age in *Spisula*, it is evident that these eastern subpopulations are composed primarily of older animals with little evidence for recruitment of young animals into these populations. Even the populations of apparently young, small clams off the Rockaways seem to reflect one or a few major settlements and lack the component of old clams. Thus, the commercial fishery is apparently dependent on relative massive settlements of larvae that occur irregularly and infrequently.

Figure 7B suggests that between stations there are significant but apparently random differences in age structure, due primarily to differences in the relative contributions of a given year class; the success of recruitment of a given age class varies independently from site to site. Whether this variability is the result of differential rates of settlement or survival is unknown.

Comparisons of Figs. 2 and 4 show little correspondence between abundance patterns of juvenile and adult surf clams. Thus, the moderate though consistent patterns of adult clams east of Shinnecock occurred in an area where settlements of 1974 year-class juveniles were sparse indeed. At the other end of Long Island, relatively heavy concentrations of juveniles west of Fire Island occurred where adults were sparse. This apparent disparity in abundances probably reflects the different causes of mortality for juveniles and adults. Juvenile abundance patterns reflect recent events—

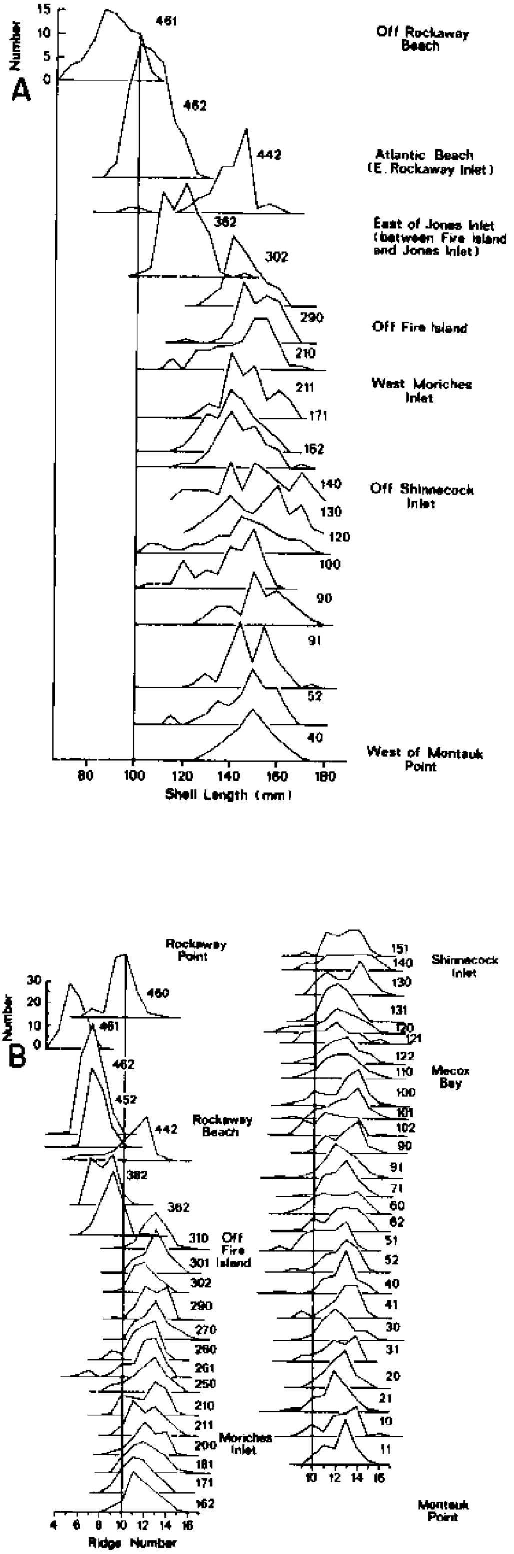

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Fig. 7. Distribution of selected populations of adult *Spisula*, August 1974. First two digits in each station number designate the transect; last digit designates vertical level: 0 = 2.5 mi; 1 = 1.5 mi; 2 = 0.5 mi offshore. A. Size-frequency. B. Age-frequency.

settlement and short term predation. Adult abundance patterns are the product of long term events—natural predation, fishing predation, and seasonal factors integrated over many years. Unfortunately, we cannot evaluate all factors causing these patterns.

Two important patterns in juvenile surf clam abundance are evident from Fig. 4: a tendency for high densities of juveniles near the west end of Long Island, and high densities of juveniles inshore at 0.5 mi, with decreasing abundance offshore. As noted earlier (Fig. 6), fine sediments seem to occur at the west end of Long Island as well as inshore near major inlets. Figure 8 shows the distribution of juvenile clams at concentrations of four or more spat per sample ($>50/m^2$) in July 1974 superimposed on the sediment texture distribution. Distribution of fine sediments is associated with the occurrence of aggregations of juveniles. Thus, the factors bringing about the deposition of fine sediments in this area may also play a role in the westward accumulation of juvenile surf clams. Tidal currents leaving coastal bays and inlets carry fine sediments, which are subsequently deposited inshore and usually to the west of the point of discharge (although ebb currents may deposit such sediments eastward of the point of discharge, as noted by Udell et al. 1974 for East Rockaway Inlet). The result is the entrapment, deposition, and accumulation of fine sediments inshore. Although much of this fine sedimentary material originates in the inlets, there must also be a contribution of fine sediments introduced farther east, carried in suspension by longshore currents. The convergence of longshore currents carrying sediments in suspension with tidal currents flowing out of tidal inlets may reduce current velocities in areas near inlets to permit deposition of fine sediments. If, in addition to suspended sediments, these longshore currents also carry populations of larval *Spisula*, then any factor slowing these currents or redirecting them would detain the larvae long enough to enhance the rate of metamorphosis. Thus, the convergence of longshore and tidal currents associated with inlets may effectively trap larvae as well as cause deposition of fine sediments. Whatever the specific mechanism, it is evident that the western end of Long Island, with its concentration of inlets

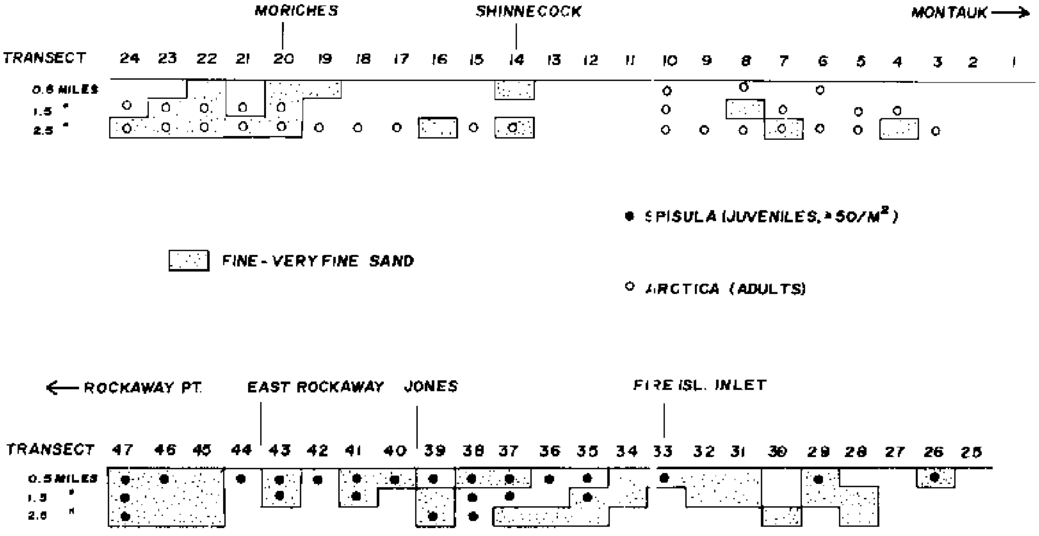

Fig. 8. Distribution of juvenile *Spisula* of estimated densities exceeding $50/m^2$ and adult ocean quahogs (*Arctica*) in relation to the distribution of fine–very fine sand. *Spisula* abundance is based on bottom grab samples; *Arctica* distribution is based on presence in commercial hydraulic clam dredge samples, August 1974.

and harbors, acts as a sink for the accumulation of juvenile surf clams that, as larvae, may originate far to the east.

The distribution of the ocean quahog *Arctica islandica* is correlated with offshore fine sand sediments off eastern Long Island east of Moriches Inlet. Indeed, *Arctica* is generally most abundant in silty sands and stable fine sand (Parker and McRae 1970) and is characteristic of the silt-sand fauna of the northwest Atlantic continental shelf (Wigley 1968; Pratt 1973). *Arctica* was collected in waters as shallow as 10–12 m at the innermost stations near Montauk (sta. 102, 82, and 62) although in general it occurred in depths >18 m.

The abundance and ubiquity of *Tellina* in a wide range of sediments from predominantly very fine sands to very coarse sands are interesting characteristics of this species, which have also been noted in Fishers Island Sound (northeastern Long Island Sound). Here, however, the dominance of bottom communities by *Tellina* is highly seasonal and is maximal in medium sand sediments. Seasonal declines in abundance of *Tellina* were less important in both coarser and finer bottom sediments (Franz 1976). This suggests that, in common with certain other tellinacean bivalves, such as *Macoma balthica* (Braefield and Newell 1961), *T. agilis* may be able to deposit feed or suspension feed, depending on the nature of the substrate in which it has metamorphosed.

The distribution and abundance of *Tellina* are different from juvenile *Spisula*, particularly in the absence of increased abundance inshore. This could result from an active selection of substrate by metamorphosing *Tellina*, compared to the passive response of *Spisula*. This possibility should be evaluated.

Results of this survey indicate that the New York Bight apex is a suitable environment for the natural development of surf clam populations and that the surf clam, both as a commercially significant species and a representative member of the shifting sand community of the inner shelf, should be considered in reviewing man's potential for future perturbation of the bight environment. Western Long Island's coast—the area most immediately susceptible to environmental degradation from ocean dumping—appears to play an important role in trapping larval *Spisula* populations originating farther east. Ocean dumping of sewage sludge has already modified a significant area of seabed from a potentially productive surf clam habitat into a less productive sludge-mud community. Damage has been done. It is not valid to argue that surf clam populations presently situated near New York Harbor must already by discounted because of pollution. Although these populations may, unfortunately, be presently unavailable for commercial harvest, this is an unacceptable argument for the status quo. With our present level of predictive knowledge of the population dynamics of *Spisula*, we cannot assay the long-range implications that could result from eventual destruction of Long Island surf clam populations. The major commercially important concentrations of surf clams now occur south of Long Island—off southern New Jersey, Maryland, and Virginia. Since southward-flowing longshore currents occur over this entire stretch of coast and since the source of the larval populations that eventually recruit into these areas is unknown, we must consider the possibility that future recruitment may require maintenance of adult clam populations farther north, including Long Island. Because we are almost totally ignorant in these matters, prudence in conserving the quality of the seabed environments of the New York Bight is required.

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Infectious diseases in commercial shellfish on the middle Atlantic coast

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Abstract

Several important molluscan and crustacean food organisms of the middle Atlantic coast of the United States are subjected to natural and man-caused perturbations and to infectious diseases.

Among the naturally occurring diseases, the protozoan oyster parasites *Minchinia nelsoni* and *Minchinia costalis* cause devastating oyster mortalities, particularly in Delaware and lower Chesapeake Bays. *Minchinia nelsoni* infections continue to be prevalent in Delaware Bay, with apparent resistance to the disease having developed there.

Several other infectious disease agents cause pathology or damage to these bivalve molluscs including viruses, bacteria, other protozoans, fungi, helminths, and crustacean parasites. Except for *Minchinia* and *Labyrinthomyxa marina* (=*Dermocystidium marinum*), none of these organisms is known to cause massive mortalities of adult bivalve molluscs. However, expression of herpes-type inclusions in hemocytes may be enhanced in oysters exposed to thermal effluents.

Proliferative cell conditions in marine shellfish have been reported with increased frequency recently and neoplasias of molluscs in epizootic proportions have been reported from both U.S. coasts. Gonadal neoplasms in soft-shell clams (*Mya arenaria*) have been found in areas associated with oil spills. Studies of benthic crustaceans from Sandy Hook Bay and the New York Bight show that the parasitic amoeba *Paramoeba perniciosa* occurs in rock crabs *Cancer irroratus* and American lobsters *Homarus americanus*.

Lobsters, shrimp, and rock crabs near sludge and spoil disposal grounds in the New York Bight had shell erosion and ulcers. Naturally occurring bacteremias in lobsters and blue crabs have been reported and may play a role in crustacean epizootics.

Viruslike infections in blue crabs from Chincoteague Bay have recently been noted. Polychlorinated biphenyls may activate production of a *Baculovirus* in shrimp exposed to sublethal levels of these compounds. Viruses may be latent in shellfish and activated by the presence of certain chemicals.

Several economically important molluscan and crustacean food species inhabit estuaries and marine coastal waters of the middle Atlantic coast. Among the molluscs are American, eastern, or Virginia oysters (*Crassostrea virginica*); hard clams or quahogs (*Mercenaria mercenaria*); ocean quahogs (*Arctica islandica*); soft-shell clams or mannose (*Mya arenaria*); and sea, surf, or skimmer clams (*Spisula solidissima*). Crustaceans include American, northern, or Maine lobsters (*Homarus americanus*); blue crabs (*Callinectes sapidus*); and the common rock crab (*Cancer irroratus*).

Until fairly recently, fishing mortality was considered as the only or the dominant factor in predicting population fluctuations and abundance and in developing fishing strategies and regulations for managing these wild fisheries. It is now recognized that these animals often are subjected to natural stresses (such as hurricane, drought, and flood damage) and man-caused perturbations (pollution stress) and to infectious disease. The effects of these infectious (contagious) and noninfectious agents may act directly or indirectly, additively, or synergistically with one another to cause epizootics of extensive proportions which in turn can result in massive mortalities or abnormalities. These effects may weaken the

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animals and render them more susceptible to predation, to other diseases, and to stress factors which ordinarily do not affect healthy organisms.

Sindermann (1976) reviewed the effects of coastal pollution (industrial, agricultural, and domestic) on fish and fisheries. The role of infectious diseases as they affect shellfisheries has also been reviewed (Sindermann 1970; Snieszko 1970). The research on infectious diseases of shellfish from along the middle Atlantic coast between 1970 and 1975 forms the basis of this report.

Molluscan disease problems

Protozoan diseases—A devastating oyster epizootic occurred in the United States in Delaware Bay in 1956–1957 and virtually eliminated the oyster industry there. Delaware Bay Disease was caused by the protozoan parasite *Minchinia nelsoni*, formerly called MSX (multinucleate sphere unknown) (Couch et al. 1966; Haskin et al. 1965, 1966). By 1960, the organism was associated with extensive oyster mortalities in lower Chesapeake Bay in Virginia (Andrews 1964, 1966) and, by 1967, extensive oyster mortalities occurred in upper Chesapeake Bay, Maryland, severely affecting the oyster industry (Sieling 1971). Mortalities caused by the parasite can occur throughout the year but the highest levels are reached during summer. The epizootiology, ecology, pathology, life history, and distribution of this parasite in oysters have been extensively reviewed (*see* Farley 1975).

Despite many laboratory attempts, consistent transmission or transplantation of the disease from one oyster to another or even to other animals (including mammals) has not been demonstrated. Attempts to culture the organism in vitro also have been unsuccessful. Possibly, insufficient numbers of infectious stages or inappropriate life history stages were used for these experiments. It is also possible that some stages in the life history of the parasite require development in an intermediate host before the organism can become infective.

Infectious stages are apparently waterborne, since infections initially occur in the gills, then spread to adjacent tissues (Farley 1968). Ultimately, all tissues except muscle are affected. Sporulation occurs infrequently, usually in fall, and most frequently in oysters less than a year old. Sporulation appears to take place only in the epithelial tissue of the tubules of the digestive diverticulae.

Season of introduction, temperature, and salinity apparently play a large role in the ecology of *M. nelsoni*. High prevalences of the disease in oysters and concomitant mortalities are more often observed in late summer and in waters with salinities consistently above 15‰. Other life history stages of closely related species of *Minchinia* have been found in mud crabs from epizootic areas (Perkins 1975; Rosenfield et al. 1969). This again suggests that an ecologically restricted alternate intermediate host or carrier may be involved; perhaps these other forms are the natural hosts for the parasite.

Mortalities in Delaware and upper Chesapeake Bays have now abated. Studies indicate *M. nelsoni* infections continue to be prevalent in Delaware Bay, with apparent genetic (inherited) resistance having developed there (Ford 1973; Haskin 1974). Few infections with *M. nelsoni* have been found in oysters from upper Chesapeake Bay since 1968.

The range of this organism extends from Wellfleet Harbor (Cape Cod) to the New River (North Carolina); the northernmost point in Massachusetts was apparently established inadvertently by introduction of heavily infected oysters from an epizootic area out of the state (Krantz et al. 1972). Management officials from many states interested in mariculture or shellfish farming are now deeply concerned over the possible promiscuous or thoughtless introductions of nonindigenous or exotic shellfish species into their coastal waters, believing that nonnative species may carry new disease agents, pests, and predators into areas now free of these organisms (Rosenfield 1969). For this reason, live and fresh shucked oysters from South Korea have been prohibited entry into U.S. waters since *M. nelsoni* or a closely related species and other kinds of parasites have been observed in them (Kern 1976).

Minchinia costalis (formerly *Haplosporidium costale*) (Sprague 1963, 1966; Wood and Andrews 1962) is another haplosporidan parasite held responsible for extensive oyster mortalities in bays of the Delmarva Peninsula with waters of oceanic or close to oceanic salinities. The disease is called SSO (seaside organism) in the older literature. Concurrent infections with *M. nelsoni* have been reported (Couch 1967). Vegetative or plasmodial stages of *M. costalis* occasionally have been observed in Long Island Sound and very rarely in other high salinity California coastal waters. It has a life history similar to *M. nelsoni*, with which it was at first often confused. But sporulation is more regular and takes place usually in late May and June in Chincoteague Bay, Virginia (Couch and Rosenfield 1968). Spores are more frequently observed in infected oysters than in the case of *M. nelsoni*. Sporulation apparently takes place in all tissues and is not confined to the digestive diverticulae. As with *M. nelsoni*, initial appearance of infections, patterns of infection, and mortalities depend more on season of introduction than length of exposure (Couch and Rosenfield 1968).

Fungal diseases—A highly infectious oyster disease caused by the fungus *Dermocystidium marinum* was described by Mackin et al. (1950) from oyster mortality areas on the gulf coast. Mackin (1951) also described histopathological changes in oyster tissues associated with *Dermocystidium* infections. He pointed out that massive infections develop in the host before death with early stages of infection characterized by inflammation followed by fibrosis and lysis of tissues. The most extensive damage is to connective tissue, adductor muscle, digestive epithelia, and blood vessels (Mackin 1951). For its detection, Ray (1952) developed a culture method that has now gone through several modifications (Quick 1971; Ray 1966). Andrews (1955) studied devastating oyster mortalities in the Rappahannock River, Virginia, and found several types of *Dermocystidium*-like fungi in several species of bivalve molluscs from Chesapeake Bay, Virginia.

It is probably not a single species but a complex of related species and has been found in several areas along the Atlantic coast including upper Chesapeake Bay, Maryland, and Delaware Bay. Transmission from infected oysters to noninfected can take place without the intervention of other hosts. However, Andrews and Hewatt (1957) could not transmit other *Dermocystidium*-like organisms from other molluscs to oysters. Mortalities caused by *Dermocystidium* occur in some gulf coast areas throughout the year. In more northerly latitudes, infections and mortalities occur primarily during summer, if oysters are crowded when water temperature and salinity are relatively high and in areas where water volume and flow over the beds are apt to be low. In transmission experiments in *Dermocystidium*-endemic waters, some populations appeared susceptible to infection while other populations were relatively resistant (Andrews 1955; Andrews and Hewatt 1957). Many reports have now resulted from studies of this organism (Perkins and Menzel 1967; Quick and Mackin 1971). Its name was changed to *Labyrinthomyxa marina* (Mackin and Ray 1966) but the colloquial name "Dermo" still persists among those working in the field. A *Labyrinthula*-like organism has recently been observed in association with mass mortalities of oysters (*C. virginica*) in Pearl Harbor, Hawaii (Kern et al. 1973). These observations raise questions regarding the development and management of the Hawaiian oyster fishery and the wisdom of using Hawaiian oysters for transplantation into mainland U.S. or other waters.

Virus diseases—Nonenteric viruses or viruslike particles causing pathology in molluscs and other marine invertebrates (*see* Rosenfield 1976) have been reported from cosmopolitan locations. The origin of these particles remains unknown, but it is doubtful that any are of human origin.

Farley et al. (1972) studied the effects of thermal effluents on oyster (*C. virginica*) growth and survival. They found that oysters from the Piscataqua River introduced into the Sheepscot River, Maine, and kept at elevated temperatures had higher mortalities than introduced animals in an adjacent area kept at lower ambient temperatures. Light transmission microscopy of oysters that died in heated water revealed frequent intranuclear inclusions within cells around the hemolymph sinuses. These inclusions were similar to those associated with herpes-type virus inclusions in other animals. Electron microscopy demonstrated typical herpes-type virus particles to be present within nuclei containing the inclusions. Herpeslike particles were also observed in later examination of other oysters from the Piscataqua River and of introduced oysters from the unheated control area in the Sheepscot River. However, prevalences were much lower than from the heated area. Farley et al. suggested that herpes-type virus infection is enzootic under ambient water temperatures and that the virus was introduced into the Sheepscot River by importation of infected oysters. They noted that the higher mortality of oysters held at higher temperatures correlated well with the higher prevalences of herpes-type virus inclusions. Furthermore, lower mortality also correlated well with lower prevalences of inclusion bodies in oysters from areas of lower ambient temperatures.

This study is relevant to energy-related developmental programs and aquaculture programs in the middle Atlantic and other coastal areas. It suggests that environmental modification, in this case elevated temperature, may favor the spread, activation, or enhancement of viral infection in aquatic animals.

Farley (1976) described another new virus disease of *C. virginica* from the Atlantic and gulf coasts of the United States. Gonadal tissue with abnormally large basophilic cells with enlarged Feulgen positive nuclei were observed in oysters infected with the virus. Electron microscopy revealed the presence of dense accumulations of 5- or 6-sided viruslike particles with an average diameter of 53 nm within the nuclei of these hypertrophied cells. Based solely on morphological features, the particles have tentatively been identified as a papilloma virus. R. Walker (personal communication) also observed very similar, if not identical, particles in a soft-shell clam from the U.S. Atlantic coast.

Otto et al. (1975) and Harshbarger et al. (1975), working on quahogs from Chincoteague Bay and oysters from Chincoteague and Chesapeake Bays, demonstrated amorphous blue bodies that were probably hypertrophied cells containing high concentrations of rickettsialike organisms. These were called blue bodies because they stained blue in fresh squashes with toluidine blue and in section they were strongly basophilic throughout. Electron microscopy of these bodies indicates that in some hard-shell clams from Chincoteague Bay the amorphous blue bodies consist of Feulgen positive organisms belonging to the Chlamydiaceae. Some of these *Chlamydia* sp. were in turn infected with a virus. This represents the first report in the United States of phagelike particle in rickettsialike organisms of aquatic invertebrates.

Viruses and epizootic neoplastic diseases in molluscs—Epizootics of proliferative cell disease tentatively classified as neoplasms have now been observed in several species of molluscs from several locations (Dawe and Harshbarger 1969; Rosenfield 1976; Sparks 1972). The etiology of tumors of neoplastic lesions in aquatic molluscs remains obscure; however, a few reports tend to implicate various physical agents and chemical compounds, including those present in fossil fuels (Yevich and Barry 1969) or coal tar derivatives and certain industrial chemicals or their byproducts used in commercial or agricultural operations (Christensen et al. 1974). It appears that in bivalve molluscs, as in many vertebrates, neoplastic disease may occur sporadically and infrequently as if random and unusual phenomena were involved or in epizootic proportions as if specific local factors, either genetically or environmentally mediated, were involved.

As indicated above, some viruses observed in molluscs are morphologically similar to some oncogenic viruses found in homeothermic animals. Thus, the discoveries of herpes viruslike particles in *C. virginica* and the papilloma viruslike particles in molluscs make the possibility more plausible that some molluscan neoplasms may be virus-induced. However, it is not known if the specific herpes virus and papilloma viruslike particles found in molluscs are the same as those which induce tumors in vertebrates or even if there is any causal relationship between molluscan viruses and known molluscan neoplasms. The range of infectivity of the molluscan viruses has not been determined and, in particular, it has not been determined whether they can produce disease in any vertebrate.

Diseases caused by helminths—Many metazoan diseases of molluscs have been described in the older literature including those diseases caused by helminths whose pathologic manifestations result in encystment, pearl formation, impaction, and even castration (*see* Cheng 1967; Sindermann 1970). More recently, however, some populations of surf clams (*S. solidissima*) from along the middle Atlantic coast were found to be infected with larval stages of a nematode worm of uncertain taxonomy (Perkins et al. 1975). The nematodes become apparent when they in turn are infected by a protozoan hyperparasite *Urosporium spisuli*. When the hyperparasites sporulate, the spores confer an orange-brownish coloration to the worm.

Crustacean disease problems

Protozoan diseases—In studies of blue crab mortalities, Sprague and Beckett (1966) described a disease syndrome associated with blue crab epizootics in seaside bays of Maryland and Virginia. The syndrome was referred to as the "gray crab" disease because of the translucent grayish coloration of the abdominal area of sick, dying, and recently dead crabs. The hemolymph of gray crabs always contained large numbers of amoebae which Sprague and Beckett (1968) described as the etiologic agent, probably belonging to the genus *Paramoeba*, later identified as *Paramoeba perniciosa* (Sprague et al. 1969).

Between 1966 and 1968 and occasionally since then, extensive summertime epizootics associated with declines of commercial catches of blue crabs occurred along the middle and south Atlantic coasts, particularly in relatively high salinity waters. Pesticides (DDT and its derivatives, toxaphene and Mirex), disease organisms, and hydrographic changes were suggested as possible causes. A large-scale cooperative study conducted among university, state, and federal investigators to test these theories did not implicate any single factor or combination of factors as responsible for the geographically widespread epizootics (Mahood et al. 1970). Gray crab disease was associated with these epizootics in nature as well as with mass mortalities of blue crabs held captive in "shedding tanks" as potential "soft crabs." Newman and Ward (1973) on the basis of their study of an epizootic in Chincoteague Bay believe *P. perniciosa* is the probable cause of many of the reported mass mortalities of blue crabs along the south Atlantic coast.

Sawyer (1976) and Bodammer (1976) investigated prevalences, causes, and histopathology of the "black gill" condition observed among benthic crustaceans captured in Sandy Hook Bay, the New York Bight apex, and adjacent areas. Sawyer showed that *P. perniciosa* or a very closely related parasitic amoeba of the same genus is present in both the rock crab and lobster. In both the rock crab and lobster, phagocytic response to *Paramoeba* results in the formation of multiple melanized nodules. Unencysted amoebae have been found only rarely and Sawyer suggested that the host response might therefore be immediate, providing little opportunity for amoeba proliferation and invasion of the hemolymph as in blue crabs.

Johnson and Newman (unpublished) investigated *P. perniciosa* in blue crabs collected along the south Atlantic coast; they reported infections for each month, but highest levels of infections occurred in May, June, and July and primarily from salinities of 28–36‰. They also reported *Paramoeba* sp. to be present in *Callinectes similis* and *Pinnixa chaeptopterana*. Farley (personal communication) has also observed *Paramoeba* sp. in the mud crabs *Rithropanopeus harrisi* and *Europanopeus depressus*. Sawyer (personal communication) has cultured, in vitro, free-living forms of *Paramoeba* from many locations including the New York Bight, but thus far has been unsuccessful in in vitro culture of parasitic *Paramoeba* derived from infected animals. An excellent annotated listing of protozoan parasites, hyperparasites, and commensals of decapod Crustacea has been published by Sprague and Couch (1971).

Bacterial pathogens—Studies related to microbial diseases of crustaceans have provided some interesting information concerning bacteremias in blue crabs and lobsters as possible normal conditions. Tubiash et al. (1975) investigating recurring summer epizootics along the middle and south Atlantic coasts undertook a seasonal survey of bacteria found in the hemolymph of freshly collected commercially acceptable blue crabs from Chincoteague Bay. They reported on an annual basis that hemolymph of only 18% of all crabs sampled was found to be sterile. They pointed out that injured crabs constituted 41% of the entire study sampling and that these showed lower percentages of sterility with higher bacterial counts when compared to intact crabs without missing appendages.

Earlier, Stewart et al. (1966) tested the hemolymph of lobsters captured from five areas of the Canadian Atlantic coast for the presence of the bacterium *Gaffkya homari*. Five percent (96 of 2,035) of the lobsters were infected with *G. homari*, with an approximate range of 2–40% depending on the area where the lobsters were captured, indicating that the organism may be resident in natural lobster populations.

In a companion paper, Cornick and Stewart (1966) indicated that 397 of the 2,035 lobsters were not infected with *G. homari* but had other bacteria in their hemolymph. The type or predominance of types varied with the area even though the same media was used for isolation. Thus, they reported the first occurrence of *Micrococci*, *Pseudomonas*, *Achromobacter*, and *Brevibacterium* in the hemolymph of lobsters. They pointed out that these genera are found in the marine environment and may have gained access to the hemolymph through injury to the digestive tract or integument. These same investigators believe that *G. homari* [= *Aerococcus viridans* (var.) *homari*] exists in many more marine species (Cornick and Stewart 1975). On the basis of their more recent experiments with red crabs (*Geryon quinquedens*) and snow crabs (*Chionoecetes opilio*), they suggest that these animals may serve as reservoir species for *A. viridans* (var.) *homari*.

While *A. viridans* (var.) *homari* was not found by Tubiash et al. (1975) in the blue crab surveys, bacteria of the genera *Vibrio*, *Pseudomonas*, *Acinetobacter*, *Aeromonas*, *Bacillus*, and *Flavobacterium* and coliforms were isolated. Other yellow-pigmented oxidative gram-negative rods were also found. *Vibrio parahaemolyticus* and related *Vibrio* sp. were the major groups found with respect to frequency of occurrence—21% of the total strains isolated (Sizemore et al. 1975). Kaneko and Colwell (1975*a*) believed that salinity, pH, and other ions present in seawater affect the ability for *V. parahaemolyticus* to adsorb onto chitin and copepods and should be considered as important relationships in determining the organism's natural habitat and ecological niche. *Vibrio parahaemolyticus* is apparently widely distributed in marine and estuarine environments and is known to be highly chitinoclastic (Kaneko and Colwell 1975*b*). Although the role of *V. parahaemolyticus* in hemolymph of blue crabs is unknown, one may speculate that it enters hemolymph through adventitious avenues and could also be responsible for integumentary lesions after adsorption.

Several types of shell disease or erosion and lesions are known (Rosen 1970; Sindermann 1970). Chitinoclastic bacteria, including *V. parahaemolyticus*, are ubiquitous in the marine environment; they are of course suspected to cause these conditions. The work of Cook and Lofton (1973) on blue crabs and penaeid shrimp tends to support this assumption. Mechanical damage to the exoskeleton was suggested as a possible route of infection by Rosen (1967) in his study of shell disease in blue crabs.

Young and Pearce (1975) and Gopalan and Young (1975) reported that lobsters, rock crabs, and shrimp (*Cragnon septemspinosa*) in and adjacent to the sludge and spoil disposal grounds in the New York Bight have various manifestations of shell erosion, rotting appendages, and ulcers. Lobsters and crabs held for up to 6 weeks in aquaria containing sewage sludge also developed ulcers and shell erosion. These conditions did not appear in controls held in sludge-free aquaria. These investigators also discussed the possibility that lesions of gills and exoskeleton may permit entrance of pathogens particularly in animals from areas, such as in waters of the New York Bight apex, having low oxygen concentrations and high numbers of bacteria. Tester and Barros (1975) studied exoskeletal erosion of spider crabs (*Chionoecetes tanneri*) and suggested that surface adsorption of microorganisms to the crustacean cuticle and the sequence of environmental events and conditions leading up to successions of microbial forms in shell lesions should be carefully studied. They found erosions to be the result of mixed bacterial populations, some of which were chitinoclastic. All bacterial cultures obtained from exoskeletal erosions were found to readily attach to and establish microcolonies within particles of chitin. Kaneko and Colwell's work (1975*a*, *b*) lends further credence to the concept of surface adsorption being a primary event leading to erosion.

Virus diseases—Shellfish harvested from domestically polluted coastal and estuarine waters have been associated with human enteric viruses and bacteria (Fugate et al. 1975). But before Vago's (1966) study, there were no published reports of viruses that infect tissues and cells of marine invertebrates. Since 1970, several papers have described viruses or viruslike particles infectious to marine invertebrates including crustaceans (*see* Rosenfield 1976). Of particular relevance is the finding by Couch (1974*a*) of a polyhedrosis virus in pink shrimp, *Penaeus duorarum*, whose northern range extends to the coast of Maryland. Couch observed pyramidal-shaped polyhedral inclusion bodies (PIB) in the cells of the hepatopancreas of shrimp exposed to polychlorinated biphenyls. He reported rod-shaped free and occluded viruses similar to *Baculovirus* associated with or within the polyhedral bodies. Couch (1974*b*) later observed an apparent increase in prevalence of patent infections with *Baculovirus* and with *Baculovirus* PIB when pink shrimp from wild populations with enzootic levels of virus are kept under crowded aquarium conditions. He suggested that increased prevalence could be due to increased transmission through cannibalism, decreased host resistance brought on by abnormal crowding, and gradual increase of polyhedral inclusion bodies in the sediment. Of great importance was his observation that Mirex and Aroclor 1254 (polychlorinated biphenyls) may activate the production of *Baculovirus* in shrimp exposed to sublethal levels of these compounds. These experiments suggest that viruses, including tumor viruses, are latent in shellfish and may be activated or effects enhanced in the presence of certain chemicals. These findings are important, for they could relate to PCBs and other chemical pollutants and energy-related activities in many coastal waters.

Two new debilitating virus diseases have been discovered in blue crabs from the middle Atlantic (Johnson 1976; Johnson and Bodammer 1975). One of the virus diseases appears to be a reovirus (Johnson personal communication). Sick crabs had discolored exoskeletons, progressive paralysis, and neither fed nor molted. Sick crabs also demonstrated necrotic hemopoietic tissues with extruded and disintegrated nuclei (karyorrhexis). Inclusion bodies consisting of rectangular crystals of Feulgen negative basophilic material were seen in the cytoplasm of young hyaline hemocytes and cells of hemopoietic tissues. Mild to severe destruction occurred to nervous tissue but no inclusion bodies were observed within these tissues. Electron transmission microscopy of infected hemopoietic cells showed the inclusion bodies were composed of viruslike particles.

Johnson (1976) has discovered another disease of blue crabs that may be caused by a herpeslike virus. This virus was found in a crab whose hemolymph was milky in appearance, did not clot, contained few hemocytes, and was filled with refractile granules. Most abnormal cells were associated with the hemopoietic tissue and had enlarged nuclei. Gill epithelium and some types of connective tissue cells were also affected, although gonad, heart, and skeletal muscle, epidermis, gut epithelia, and nervous tissue did not appear to be involved. These observations by Johnson represent the first findings of viruses in crabs in the United States and the second report of a herpeslike virus in invertebrates, the first being that of Farley et al. (1972) in oysters.

Summary and conclusions

Disease- and parasite-induced mortalities are paramount factors limiting the abundance of marine fish, crustaceans, and mollusks. Except for a few species, little is known about the causes of mortalities of eggs, larvae, juveniles, and adults of these resources. Point-in-time abundance assessments of marine animals for management purposes are only meaningful when natural mortalities can be estimated. Adequate knowledge of the health status of fishery stocks and of disease prevalence is fundamental to the significance of resource assessments.

Mass mortalities of aquatic animals grown under intensive culture conditions are often a consequence of infectious disease. In aquaculture, the indiscriminate transfer of animals from one location to another without regard to the disease entities they may harbor is a hazardous practice, the importance of which has not been adequately recognized. The diagnosis of infectious diseases of animals used in aquaculture is, therefore, vital to prevent disease-caused mortalities and to develop a fundamental competence in understanding the role of parasitism and disease as they affect both resident and introduced populations of marine animals.

Host susceptibility to disease is directly influenced by environmental stress, and there are no ways of knowing what stresses are significant in limiting populations without studying their effects on the animals themselves. Environmental barriers can substantially reduce the infection pressure from a particular pathogen and, with sessile invertebrates or geographically limited species, some measure of disease control can be effected. On the basis of known patterns of disease incidence, populations can be exploited before they are decimated by disease.

Until recently, the role of disease in studies of the ecology of marine poikilotherm animals has been virtually ignored and poorly understood. The high degree of interest that now exists has been brought about largely through increased development of operations involved with fish and shellfish aquaculture and farming practices, including transplantation and introduction of nonnative or exotic stocks; massive, poorly understood natural mortalities of marine and estuarine commercial species; increased awareness that man's activities and effluvia can have profound effects on living marine resources; and recognition that disease problems in "lower" animals can serve as study models for understanding and comparing disease processes in "higher" animals.

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Section 9 Public health

The impact of pollution on marine bathing beaches: An epidemiological study

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Abstract

As part of a national program to develop health effects criteria for marine recreational waters, the U.S. Environmental Protection Agency conducted a prospective epidemiological-microbiological study at bathing beaches in the vicinity of New York City, specifically at 20th St. on Coney Island and 67th St. and Riis Park at the Rockaways. The most consistent findings over the first 2 years of this study were that, for most of the water quality indicators examined, the mean densities at the Coney Island beach were appreciably and significantly higher than those at the Rockaways, and that the rate of gastrointestinal (GI) symptoms was significantly higher among swimmers relative to nonswimmers at the Coney Island beach but not at the Rockaways. When the data from two summers at both beaches (four points) were examined, good agreement was obtained between the mean *Escherichia coli* and enterococcus densities and the differential (swimmers minus nonswimmers) rate of GI symptoms. This preliminary finding addresses the objective of the study: relating illness as measured by symptomatology to some indicator of water quality.

As part of a national program to develop health effects criteria for marine recreational waters, the U.S. Environmental Protection Agency (EPA) conducted a prospective epidemiological-microbiological study at bathing beaches on Coney Island and the Rockaways, New York.

The overall program called for a 3-year study at the New York City beaches to develop a data base relating illness as measured by symptomatology to some potential microbial indicator of water quality (Cabelli and McCabe 1974). This will be followed by trials at some "subtropical" site to verify or expand the relationships obtained, by the development of a mathematical model and by a series of model verification trials at geographically distinct and representative locations.

The need for recreational water quality criteria is especially critical for coastal waters adjacent to large urban centers such as New York City since it is in these very areas—where swimming can represent a major source of outdoor recreation—that the safe disposal of sewage is most difficult. In response to this need, various states have adopted microbiological criteria and standards for the sanitary quality of marine waters used for bathing. These have been reviewed by McKee and Wolf (1963), by Senn et al. (1963), and most recently by Mechalas et al. (1972). The two values of interest are the mean total coliform density of 2,400 per 100 ml of water used by New York State and the fecal coliform criterion of 200 per 100 ml[1] recommended in 1968 by the National Technical Advisory Committee (NTAC) to the Federal Water Pollution Control Administration (1968). The latter value was based on findings from Stevenson's (1953) studies on Lake Michigan and the Ohio River. He reported that total coliform densities of about 2,000 per 100 ml were associated with epidemiologically detectable health effects. The U.S. EPA (1973) proposed a criterion essentially the same as that recommended by NTAC.

Shortly after its publication, the NTAC microbiological criterion was attacked by Henderson (1968) as too restrictive. He set forth several arguments against the promulgation of microbiological *standards* on a nationwide basis. These included the broad confidence limits of the most probable number (MPN) test used in quantifying total or fecal coliforms, temporal and geographic variability of pathogen and indicator levels, and the effect of differing sources of pollution (treatment plant effluents, stormwater runoff, farm-lot wastes, etc.). However, the thrust of his attack was the paucity of defined epidemiological data in support of the NTAC criterion. To the contrary, he used the British experience (Moore 1959), the observations from Santa Monica Bay, California (Garber 1956), and the lack of published morbidity and mortality data associated with swimming in "polluted" waters to support a much less restrictive microbiological standard for bathing beaches, or no standard at all.

The validity of Henderson's comments on the shortcomings of some water quality indicators and the paucity of epidemiological data notwithstanding, his conclusions about the *need* for recreational water quality guidelines and standards cannot be accepted. Recreational water quality criteria are required so that they can be translated into source (effluent) guidelines for designing and operating sewage treatment plants. In addition, health officials need the criteria to set target area (receiving water) guidelines and standards designed to restrict recreation to those areas considered "safe." Because social, economic, and political factors also may influence them, the two translations may result in different values. Without the extensive data base needed to develop the more appropriate criterion, a reasonable alternative at present is to use a guideline such as that developed by the NTAC.

A review of infectious diseases associated with recreational use of sewage-polluted marine waters, the etiological agents involved, and potential water quality indicators for indexing the health hazards is beyond the scope of this paper. This has been done most recently by Geldreich (1970) and Cabelli et al. (1974).

The data base for the development of microbiological criteria for recreational water can be sought from three sources: available mortality and morbidity statistics, including retrospective epidemiological analysis of case reports and disease outbreaks; output from predictive models; and findings from prospective, controlled, epidemiological-microbiological studies. Studies have been conducted using all three approaches (Stevenson 1953; Moore 1959; Mechalas et al. 1972). The strengths and weaknesses of each approach were reviewed in the development of the experimental design used in the present investigation and have been described earlier (Cabelli et al. 1974).

Our investigation at the New York City beaches, like Stevenson's, is a prospective epidemiological study. However, the experimental design was changed to increase the sensitivity and specificity of the epidemiological tool. Fortunately, advances in the state of the art with regard to methodology have removed many of the constraints placed on Stevenson in the design of his study.

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[1] As determined by multiple-tube fermentation or membrane filter procedures based on a minimum of not less than five samples taken over not more than a 30-day period, the fecal coliform content of primary contact recreation waters shall not exceed a log mean of 200 per 100 ml, nor shall more than 10% of total samples during any 30-day period exceed 400 per 100 ml.

Detailed epidemiological findings as they relate to the development of health effects criteria for recreational water quality will be presented elsewhere.

Experimental

The study at the New York City beaches was conducted in three phases. The first, a pretest of the epidemiological and microbiological methodology, was performed during summer 1973. The second phase was conducted during summer 1974. Its major objective was examination of symptom rates (by category, importance, and, where possible, demographic grouping) of swimmers relative to nonswimming controls at a "barely acceptable" as compared to a "relatively unpolluted" beach. A barely acceptable (BA) beach was defined as the most "polluted" beach available, not posted as unsafe for recreational use according to local criteria. A relatively unpolluted (RU) beach was defined as a beach subject to the least amount of pollution and at which the populations were demographically comparable to those at the BA beach. Because positive findings were obtained in the second phase of the study (i.e. statistically significant differences in symptom rates among swimmers relative to nonswimming controls at the BA but not at the RU beach), beaches on a pollution gradient (spatial variability) or trials at a given beach (temporal variability) were examined toward quantifying the relationship of symptomatology to some microbial indicator of bathing water quality (phase 3).

A basic premise of the experimental design was that the "important" illnesses and the "correct" indicators of the quality of the water would be treated as unknowns. Therefore illness information was sought in the context of symptomatology, and measurements were made for a number of potential water quality indicators. The experimental design for this study was described by Cabelli et al. (1974). Briefly, it consisted of a series of discrete trials in which (1) the potential participants (primarily as family groups) were contacted at the beach on weekends, (2) individuals who swam in the midweeks immediately before and after a trial were eliminated from the study, (3) measurements for a number of potential water quality indicators were made during the course of the trials at the test beaches, and (4) follow-up information concerning symptomatology and demography was solicited by phone some 8–10 days after a trial. The design differed from Stevenson's prospective study in that swimming was defined as significant exposure of the head to the water, and discrete trials were conducted over relatively short periods (1–2 days). Thereby, the effects of day-to-day variability in pollution level reaching the test beaches were minimized.

Beaches at Coney Island and the Rockaways were chosen as test sites for several reasons (Fig. 1). First, the beaches are heavily used, especially on weekends. Second, the beach area around 20th St. at Coney Island is immediately adjacent to an area posted as unsafe for swimming. In addition, beaches in the vicinity of 67th St. and Riis Park at the Rockaways could be used as the relatively unpolluted beaches. Third, a group of medical demographers capable of gathering and analyzing illness information was identified. Finally, there was considerable interest, assistance, and cooperation on the part of the New York City environmental health officials.

Three types of information were sought and obtained in the course of the beach interviews and subsequent follow-up by telephone: beach activity, symptomatology, and demography. The beach activity information was used to categorize the participants as swimmers (head actually in the water) and nonswimmers and to examine any effect of the time they were in the water. Information was not obtained on the period over which a given individual's head actually was exposed to the water because of the assumed unreliability of such information. The symptoms for which queries were made and the groups into which they were categorized are shown in Table 1. An "importance index" was obtained from information on whether the respondents remained home, remained in bed, or sought medical

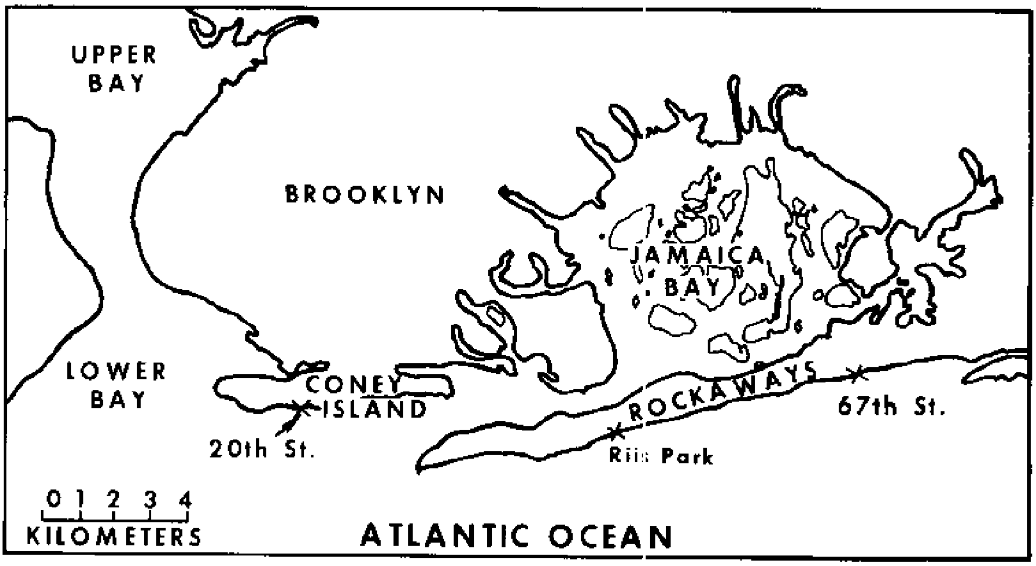

Fig. 1. Test beaches at Coney Island (barely acceptable) and the Rockaways (relatively unpolluted) in New York City.

advice. Demographic information included age, sex, ethnicity, and socio-economic status as determined from persons-to-rooms ratios.

For obvious reasons, the trials were conducted during summer. By conducting the trials on weekends, it was possible to eliminate individuals who swam during the midweeks before and after a weekend trial without severely limiting the size of the study population.

Table 1. Symptoms for which queries were made.

- Gastrointestinal
  - Vomiting
  - Diarrhea
  - Stomachache
  - Nausea
- Other
  - Fever (>100°C)
  - Headache (more than few hours)
  - Backache
- General
  - Sunburn
  - Skin rash, itching skin
  - Red, itchy, or watery eyes
- Respiratory
  - Sore throat
  - Bad cough
  - Chest cold
  - Runny or stuffed nose
  - Earache or runny ears
  - Sneezing, wheezing, tightness in chest
- Importance index
  - Home because of symptoms
  - In bed because of symptoms
  - Medical help because of symptoms

Water samples were collected periodically during the time of maximum swimming at the beaches. Two samples were collected from each beach at about 11:00 am, 1:00, 3:00, and 5:00 pm. The samples were collected in sterilized bottles about 10 cm below the water surface at about chest-high depth. They were placed in iced chests and returned to the laboratory where they were assayed within 6 h of collection. In keeping with the premise that the "correct" indicator would be treated as an unknown, several potential water quality indicators were considered and measurements for most of them were made sometime during the 3-year study at the New York City beaches. Enteroviruses were excluded because of methodology and logistic considerations. Shigellae were not quantified because of methodology problems. *Salmonella* densities were obtained during the pretest; however, due to the extremely low densities observed, *Salmonella* assays were abandoned during the last two phases of the program.

Total and fecal coliform densities were obtained using the MPN procedure (Am. Public Health Assoc. 1971). Total coliforms and the component genera thereof, *Escherichia*, *Klebsiella*, and *Citrobacter–Enterobacter*, were estimated using the mC procedure (Dufour and Cabelli 1975). *Salmonella* densities were obtained by the high volume sampling method (Levin et al. 1974). Enterococci (Levin et al. 1975), *Aeromonas hydrophila* (Cabelli 1973), *Pseudomonas aeruginosa* (Levin and Cabelli 1972), and *Vibrio parahaemolyticus* (Thomas et al. 1974) were counted using membrane filter procedures developed in this laboratory.

The symptom rates, categorized into respiratory, gastrointestinal (GI), other, and "importance," for swimmers and nonswimmers at the BA and RU beaches were analyzed statistically by Cochran's two-way, χ^2 analysis of association (Fleiss 1973).

Results

The results described here are those obtained during the first two phases of the overall study at the New York City beaches. The data from the third phase, conducted during summer 1975, are undergoing analysis and are not yet available.

Table 2. Mean indicator densities at the Coney Island and Rockaways beaches during 1973 and 1974 trials.

	Log mean recovery/100 ml			
	1973		1974	
Indicator	Coney Isl.	Rockaways	Coney Isl.	Rockaways
Total coliforms	983*	39.8	1,213*	43.2
Fecal coliforms	165*	21.5	565*	28.4
Escherichia coli	174*	24.8	15.3*	2.4
Klebsiella	122*	13.7	59.2*	3.5
Enterobacter-Citrobacter	530*	11.1	434	6.6
Fecal streptococci	91.2	21.8	16.4*	3.5
Pseudomonas aeruginosa	30.4	6.5	45.8*	3.1
Aeromonas hydrophila	25.3	26.5	9.6	4.9
Vibrio parahaemolyticus	ND	ND	54.5	32.8

* Significantly different at 95% confidence level.

Most of the microbiological findings obtained during the 1973 pretest were presented earlier by Cabelli et al. (1974). The mean densities of those indicators for which measurements were made during 1973 and 1974 are presented in Table 2. For most of the indicators, the densities at the Coney Island (BA) beach were significantly and appreciably higher than those at the Rockaways (RU) beaches. As expected from previous information, between-day (primarily due to rainfall) and within-day (primarily due to tide) variability in most of the indicator densities was considerable at the Coney Island beach but less at the Rockaways beaches. Within-day variability of the total coliform densities at the BA beach over a 6-h period as obtained by the mC procedure is illustrated in Fig. 2.

The symptom rates by category for swimmers and nonswimmers at both beaches over the first 2 years of the study are presented in Table 3. At both the BA and RU beaches, the rates were higher for swimmers than for nonswimmers. This is consistent with Stevenson's observation that, irrespective of the pollution level, there is

Table 3. Symptom rates in percent at Coney Island and Rockaways beaches during 1973 and 1974 trials (S—swimmers; NS—nonswimmers; Δ—difference).

	Rates in percent for symptom groups											
	1973						1974					
	Coney Island			Rockaways			Coney Island			Rockaways		
Symptom group*	S	NS	Δ	S	NS	Δ	S	NS	Δ	S	NS	Δ
N	474	167		484	197		1,961	1,185		2,767	2,156	
Resp.	12.9	10.2	2.7	18.0†‡	11.7	6.3	7.2	6.4	0.8	8.3	7.8	0.5
GI	7.2†	2.4	4.8	8.1	4.6	3.5	4.2†	2.6	1.6	3.9	3.5	0.4
Other	9.9	6.6	3.3	9.1	8.6	0.5	7.3	6.7	0.6	8.6	7.7	0.9
Important	5.9	4.2	1.7	6.0	5.6	0.4	3.8	2.9	0.9	3.0	2.6	0.4

* *N*—sample size; Resp.—respiratory; GI—gastrointestinal; for group composition, *see Table 1*.

† Significantly ($P = 0.5$) higher than nonswimmers.

‡ Significantly ($P = 0.5$) higher than other beach.

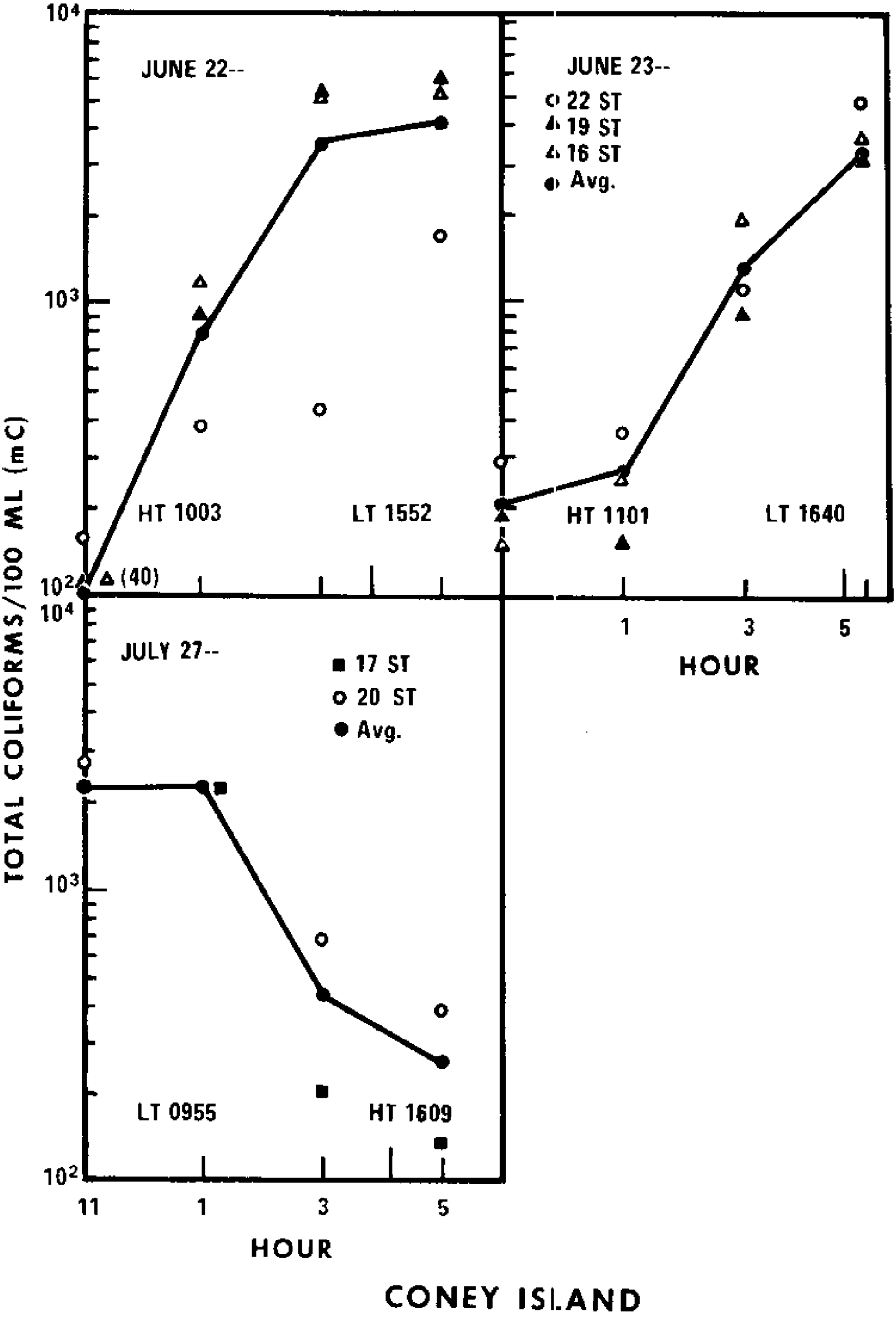

Fig. 2. Variability of coliform densities at the Coney Island test beach as a function of tide. ST—Street, HT—high tide; LT—low tide.

an association of symptomatology with swimming per se. Furthermore, in 1973 and again in 1974 the differential rates of symptoms (swimmers minus nonswimmers) were generally higher at Coney Island than at the Rockaways. But the only statistically significant, consistent finding during both summers was that rate of gastrointestinal symptoms among swimmers relative to nonswimmers was higher at Coney Island. Such was not the case at the Rockaways.

These findings point out the appropriateness of the epidemiological and microbiological methodology, the suitability of the test sites, and the sensitivity of the experimental design. However, these findings do not speak to the overall objective of the U.S. EPA program—that is, quantifying the relationship of illness as defined by symptomatology to some water quality indicator (criteria). A preliminary analysis of the data was attempted even though the data from phase 3 of the study were unavailable. The approach used was to analyze the data by years (summers). Thus, the symptom rates and associated mean densities for all the trials at each beach during a given summer were combined to yield a single data point. Four points were obtained for each indicator (Fig. 3). Since additional points should be obtained from the 1975 trials, a statistical analysis was not attempted. However, inspection alone confirms the close relationship of GI symptomatology to *E. coli* and enterococcus densities. This approach to the analysis of the data is consistent with an initial premise of the study—that is, that the "correct" indicator would be treated as an unknown and that the indicator of choice would be the one yielding data agreeing best with the epidemiological findings.

Regression lines for both total and important GI symptoms against *E. coli* densities are shown in Fig. 4. The differential rate for total GI symptoms associated with

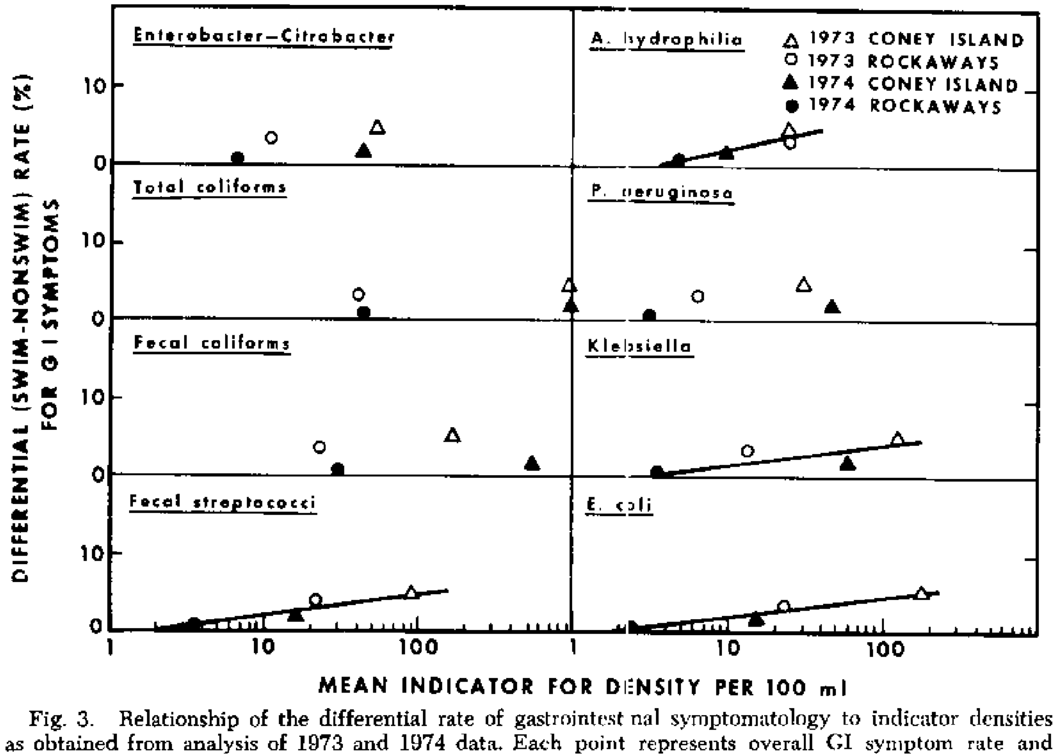

Fig. 3. Relationship of the differential rate of gastrointestinal symptomatology to indicator densities as obtained from analysis of 1973 and 1974 data. Each point represents overall GI symptom rate and mean indicator density for all trials conducted at the beach during that summer.

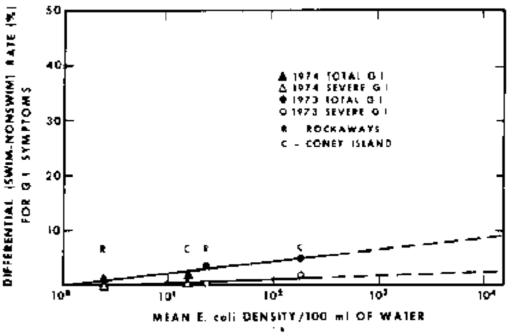

Fig. 4. Differential rates for total and relatively important gastrointestinal symptoms in relation to mean *E. coli* density as obtained from 1973 and 1974 data.

a mean *E. coli* density of 200 per 100 ml was 3.8%.

Discussion

Finding that the differential rate of gastrointestinal symptomatology was higher at the Coney Island than at the Rockaways beach was not unexpected. Existing information on the sources of pollution reaching the area, the hydrography, and indicator densities at the beaches along Coney Island and the Rockaways would have predicted such results. Of course, this assumes that, even in the absence of visible fecal contamination, there is a potentially quantifiable association between swimming in sewage-polluted waters and gastrointestinal disease and that ill individuals or carriers are discharging pathogens into the sewage systems. The point in doubt was whether the epidemiological methodology used was sensitive enough to detect such differences at two sites acceptable for swimming according to existing guidelines and standards. It would appear that it was. Nevertheless, the findings cannot and must not be interpreted as either justifying or condemning existing standards or their application at the test sites. This is beyond the scope of this investigation since other factors would enter into such a value judgment.

An attack rate for total gastrointestinal symptoms of 3–4%, or even about 1% for important GI symptoms, associated with swimming in waters having about 200 *E. coli* per 100 ml would appear disquieting, especially when projected to the large numbers of individuals who swim at such beaches during a season. However, even the severe symptoms did not result in hospitalization or protracted illness, although it is possible that more serious illnesses would have been recorded if the observation period had been extended beyond 8–10 days. In all probability few, if any, of these cases would have been reported to public health authorities except in an "outbreak" situation. Public health effects of posting or closing such beaches must be weighed against social and health consequences of denying the resource to the public, especially in large urban centers. On the other hand, given epidemic conditions in a community whose sewage is discharged into a body of water, a previously acceptable recreational water quality standard may not be applicable in spite of a built-in safety factor. Thus, recreational water quality guidelines and standards must be used in conjunction with and not to the exclusion of good public health surveillance and practice.

Finally, the data presented here suggest that there are measurable health effects associated with swimming in sewage-polluted waters. The overall program to develop health effects criteria for recreational water quality is far from complete, although the data obtained thus far are most encouraging. *Escherichia coli* and fecal streptococci appear to be the best indicators examined thus far. It should be pointed out that the expected output from the experimental design was an association of the indicator density to symptomatology and not a cause-and-effect relationship to a specific disease entity. Nevertheless, an association should be satisfactory for the purpose intended: providing a data base for the development of guidelines and standards for municipal sewage treatment plant effluents.

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Safe shellfish from the sea

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Abstract

The National Shellfish Sanitation Program was initiated in 1925 after a widespread typhoid fever outbreak caused by raw oysters. Specific recommendations made in 1925 established guidelines for classifying offshore waters for clam harvesting and stated that shellfish being harvested must not be exposed to fecal contamination. In 1974, sea clams accounted for 59% of all shellfish (oysters, clams, and mussels) harvested from United States waters. The Food and Drug Administration is responsible for classifying the offshore water beyond 5.5 km for harvesting shellfish. In recent years, the sea has been used increasingly for sewage sludge and industrial waste disposal. Depletion of the resources in the original major sea clam harvest area off New Jersey and increased demand (43 million kg were harvested in 1974) have caused the industry to expand its area of operation to the southeast in search of more productive shellfish beds. There too, heavy population and industrial growth have increased the amounts of chemical wastes, sewage sludge, and other waste materials being disposed of at sea. Warning notices to harvesters, closing areas of the New York Bight to shellfishing, have been posted since 1970. High bacterial levels are found in both the sewage sludge and dredge spoil sites.

The Food and Drug Administration (FDA) is concerned with ensuring the safety and wholesomeness of sea clams, including the surf clam and ocean quahog, which are commercially harvested in the New York Bight area. In 1974, sea clams accounted for 43.5 million kg (96 million pounds) or nearly 59% of the total United States harvest of clam and oyster meats (U.S. Dep. Commerce 1975). The FDA is concerned because sewage sludge and other contaminated wastes are discharged near commercially important sea clam beds. In addition, effluents from primary and secondary sewage treatment plants add discharges that contaminate inshore clam beds as the effluents move out to sea. Because of the potential health hazard, the FDA has delineated contaminated areas and published prohibited area closure maps for shellfishermen harvesting in the bight. FDA reviews ocean waste disposal operations and must be informed of any plans to relocate or alter existing disposal sites for sewage sludge or other wastes that may contaminate shellfish and be harmful to man.

History of the National Shellfish Sanitation Program

In 1925, the cooperative federal-state-industry National Shellfish Sanitation Program was initiated to ensure safe shellfish. The program provides sanitary control over shellfish culture, harvesting, and processing. It is the outgrowth of a widespread typhoid fever outbreak in the United States that began in winter 1924, caused by consumption of contaminated raw oysters (U.S. Public Health Serv. 1925*a*). The epidemic spread to 24 cities between 16 November 1924 and 7 January 1925. Over 1,500 cases and 150 deaths were reported, principally in Washington, D.C., Chicago, and New York City. The epidemic caused a loss of confidence in oyster and other shellfish industries and a reduction in the sale of all fishery products. The affected industries and control agencies of the shellfish-producing states met with the Surgeon General "to consider measures to insure the future safety, from the standpoint of health, of all shellfish" (U.S. Public Health Serv. 1925*b*).

The committee concluded that public health protection demanded action to stop transmission of preventable disease through shellfish and that the industry needed to have public confidence restored and maintained if it was to sell its product. The National Shellfish Sanitation Program (NSSP), currently administered by FDA, resulted from that meeting. The program's standards, criteria, and operation procedures are contained in the *NSSP Manual of operations.*

Biological and environmental considerations

Shellfish filter and concentrate harmful substances present in overlying waters, so they have a potential for transmitting disease to man; thus, they must be protected from contamination. Two factors contribute to contamination of sea clams in the New York Bight area.

1. The nearshore portion of the bight receives the combined wastes of the New York–New Jersey metropolitan region. These wastes affect water quality for about 32 km.

2. Shellfish pump large quantities of water across their gills. Particles and dissolved substances are removed and may be concentrated many times above levels in overlying waters. Bacteria can be concentrated six or more times, whereas metals, pesticides, or biotoxins can be concentrated to levels thousands of times greater than in the overlying waters.

Health considerations

Typhoid fever has been the principal disease associated with shellfish; the last case recorded in the United States was in 1954 (Verber 1972). Since 1961, infectious hepatitis has been the prime offender, the last large outbreak occurring in 1974 (Portnoy et al. 1975). *Vibrio parahaemolyticus*, a true marine bacterium causing summer diarrhea in Japan, has been reported as a probable cause of shellfish-borne gastroenteritis (U.S. Public Health Serv. 1970). *Salmonella* and other enteric pathogens have also been incriminated.

In Japan, 83 cases of severe mercury poisoning from fish and shellfish demonstrated that high levels of industrial wastes in areas adjacent to harvested shellfish are dangerous to human health (Kerland 1962).

Paralytic shellfish poison (PSP), a natural marine biotoxin produced by dinoflagellates such as *Gonyaulax tamarensis*, has never been found in the New York Bight, but occurs north of Cape Cod. Toxic levels of PSP have been found in both species of sea clams where toxic dinoflagellate blooms have occurred (Larkin 1975). Therefore, FDA must be alert to possible occurrences of toxic blooms in the bight area.

Growing-area classifications

The primary consideration in shellfish sanitation is the classification of growing areas based on water quality. The criteria established for estuaries have been adopted for classifying open ocean areas subject to pollution.

Closure of the bight to sea clam harvesting was required because:

1. Dumping of sewage sludge contaminated the area so that consumption of shellfish grown there might be hazardous.

2. Sewage treatment plants' effluents discharged into the Hudson and nearby water courses pollute the coastal waters of the bight.

3. The area is contaminated by discharges of industrial waste.

4. The failure of regulations and enforcement to control the precise disposal location of sewage sludge has resulted in uncontrolled dumping over a large area of the bight.

Ocean dumping and shellfish control

Since 1942, sea clam production in the U.S. has increased from 2.5 million kg (5.5 million pounds) to 43.5 million kg (96 million pounds) annually (Fig. 1). In the early 1960s, federal shellfish control authorities became concerned about the proximity of sea clam harvesting areas to dump sites in the bight and conducted a series of federal-state-industry discussions to develop and implement measures to ensure the sanitary quality of sea clams subject to potential contamination from ocean dumping. As a result of these discussions, a bacterial study of sea clams was conducted near the dump site in 1964 (Hunt 1964). Further investigations on harvesting practices were made the following year (Fester 1965). The 1966 New York studies of Buelow et al. (1968*a*) and the 1967 offshore Delaware Bay area studies (Buelow et al. 1968*b*) were reported at the January 1968 National Shellfish Sanitation Workshop and formed the

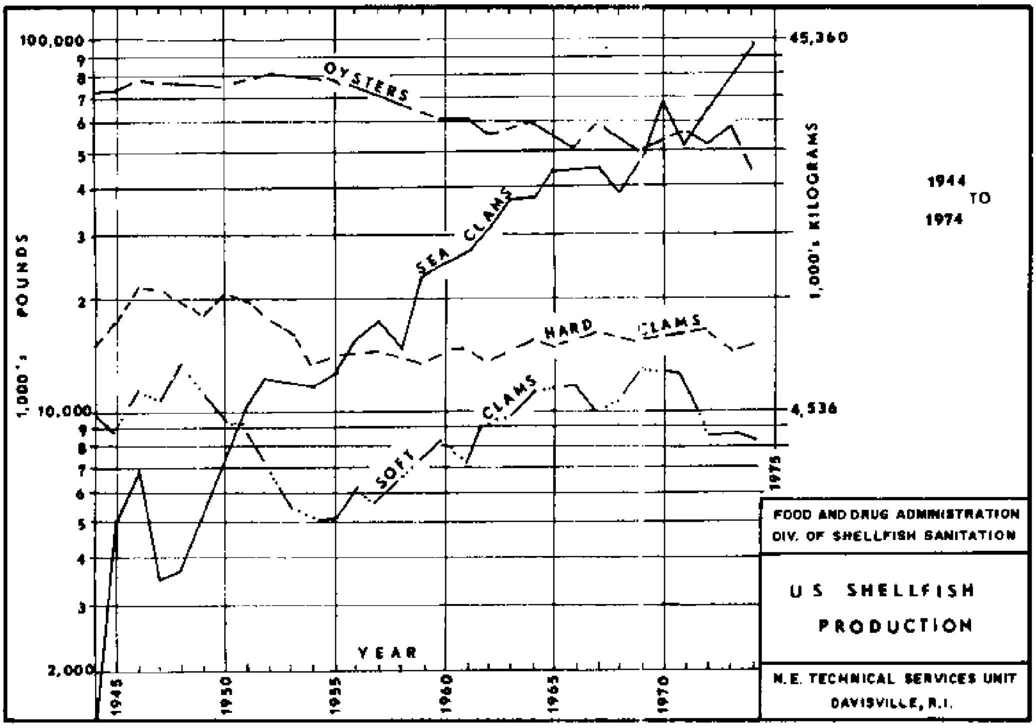

Fig. 1. U.S. shellfish production, 1944–1974.

basis for the first shellfish closures in the ocean proper. Because of these reports and the concern of the Food and Drug Administration over sanitary control of the sea clam, follow-up investigations were made in 1969 by the Northeast Technical Services Unit, FDA, assisted by the Sandy Hook Marine Laboratory (NOAA).

In early 1969, the FDA reviewed all ocean dump sites to determine the extent of pollution sources that might affect shellfish and other marine foods (Smith and Brown 1971). Following the initial studies, FDA began classifying and monitoring ocean dump sites near offshore commercial clam harvesting areas.

The 1966–1967 studies on waste disposal in New York and Delaware were only the beginning of offshore studies. Increasing national concern over ocean waste disposal led to a succession of investigations beginning about 1968.

In the 1969 review, FDA tabulated 280 ocean dump sites, many considered potentially hazardous. These sites, coupled with direct ocean outfalls of municipal and industrial wastes, posed a serious problem.

Waters affected by 128 dump sites, plus the municipal and industrial outfalls within 5.5 km, are classified by state shellfish control authorities; responsibility for classification of waters affected by 41 sites within 5.5–22.2 km has been assumed by FDA. The harvestable area beyond 22.2 km, some 20 sites, is in international waters and under semifederal jurisdiction.

As a direct outgrowth of the initial offshore studies, a broad range of investigations was conducted in areas adjacent to commercial harvesting. Bacterial, chemical, pesticide, and radionuclide studies were pursued, some only for general screening.

The magnitude of the problem can be better comprehended when we consider that sewage sludge dumping from the greater New York area has increased 75% between 1965 and 1975. The 1965 volume is expected to double by 1979.

Bacterial studies

In May and June 1966 and in June 1967, Buelow (1968) undertook the first dump site studies off New York and Delaware. He collected 165 samples of ocean water at 29 stations in the New York area and 85 samples at 19 stations in the Delaware area and monitored them for total and fecal coliforms (Am. Public Health Assoc. 1970). Significant findings related to the shellfish program of the New York study included:

1. Coliform MPN of surface waters, with no tanker in the area, exceeded 2,400/100 ml.
2. MPN seawater values in excess of 2,400,000/100 ml for both total and fecal coliforms could be found in the surface water after sewage discharge.

Buelow recommended—

1. An area 11 km in radius from any point at which sewage sludge discharge takes place should be prohibited to shellfishing.
2. The dump site should be adequately marked.
3. No relocation of the dump area was justifiable.

The recommendations in the Delaware report were similar to those for New York except that alternate dump sites were suggested, especially "where the shellfish population is small or non-existent" (Buelow et al. 1968*b*).

Ocean studies by FDA are made principally to monitor growing areas to determine the proper classification for safe shellfish harvesting. Based on early studies, FDA prohibited shellfishing in the offshore New York and Delaware sewage dumping areas in May 1970 (Fig. 2). The shellfish control agencies notified individual ship captains of the closures. In 1970, 101 ships were harvesting sea clams, according to data provided by the National Marine Fisheries Service (NOAA). Closed areas were routinely patrolled by the Coast Guard, who agreed to radio the FDA of any sea clam harvesting in the prohibited areas; FDA would then inform the state so appropriate action could be taken.

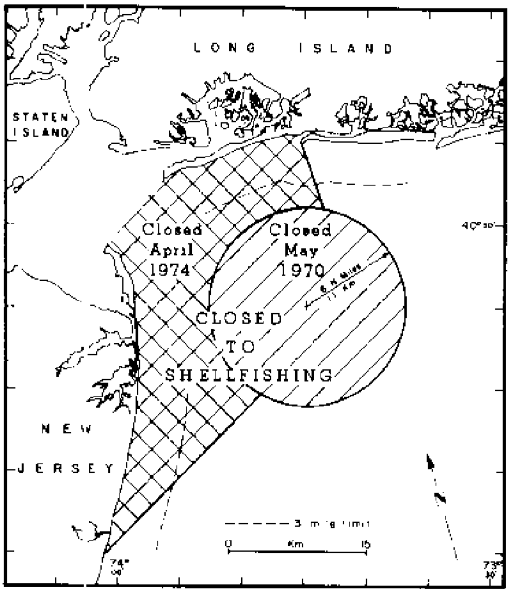

Fig. 2. Closure of New York bight areas to commercial shellfish production, 1970 and 1974.

The program has worked well, with only two reported violations by ships harvesting in the outer margins of the Delaware closure area during the initial phase of the program and before they received a notice. No action was taken except that these vessels were notified.

As conditions changed, so did the closure notices. When increasing bacterial pollution from the Metropolitan New York area expanded the area of contamination to the inshore waters of New York and New Jersey, FDA, in joint action with the states, extended the closure in April 1974 (Fig. 2). The extension was based on studies conducted by New York, New Jersey, and the Food and Drug Administration in 1972 and 1973.

In the offshore Delaware area, the FDA took action to rescind the closure when all threat of contamination was past. In May 1973, the Environmental Protection Agency closed the former Delaware site to sewage dumping and moved the dumping area more than 111 km offshore. After monitoring studies indicated that a bacterial problem no longer existed at the former site, a cancellation notice was issued in January 1975, and the area was reopened.

Table 1. New York Bight, coliform/100 ml, g.*

	Jun–Jul 1969			Mar 1971		Dec 1971	
Sta.	Sediment†	Water	Shellfish	Shellfish	Sediment†	Water	Sediment†
109	<20/<20			23,000/1,100	11,000/700	1,700/49	130,000/2,800
				79,000/2,200			
				23,000/1,400			
	220/130	70/7.8		49,000/7,900			
2007				3,300/45			
				3,300/20	3,300/270		
				3,300/110			
				1,300/40			
106	20/<20	<2.0/<2.0	45/<20	17,000/790		1,700/330	
		<2.0/<2.0		7,900/230		790/540	
				7,900/330		240/240	
				13,000/1,300			
48				7,900/1,700			
	40/20	<2.0/<2.0	20/20	3,300/490			
	330/20	2.0/<2.0		3,300/330			
				3,300/430			
101				13,000/430			
	<20/<20	<2.0/<2.0		7,000/790			
				7,900/330			
	<20/<20	<2.0/<2.0		3,300/1,300			
N-1				22,000/2,300			
				35,000/4,900			
				17,000/4,900			
				35,000/950			

		Apr 1972			Jul 1972		
Sta.		Water		Sediment†	Water		Sediment†
106	130/14	490/110	<1.8/<1.8	1,700/330	17/2		110/20
	33/13	11/1.8	4.5/2		2/<1.8		
	1.600/350	49/49			79/4.5		
	13/4.5	14/6.8			240/2		
48	9.2/6.8	49/33		330/78	79/14	<1.8/<1.8	130/130
	540/130	49/33			<1.8/<18	4.5/<1.8	
	17/7.8	<1.8/<1.8			70/2	2/<1.8	
		22/4			49/<1.8	79/<1.8	
101	14/4.5	79/17		1,300/<18	23/2	350/49	
	<1.8/<1.8	21/7.8			2/<1.8		
	27/17	<1.8/<1.8			4.5/<1.8	>1,600/240	
	11/8	2/<1.8			4.5/2		
	79/49						
	13/2						
N-1					21/4.5		
					2/<1.8		
					33/4.5		130/20
					2/<1.8		
					13/<1.8		
					1,600/920		

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* Total coliforms/fecal coliforms.

† Coliforms determined by the method of Presnell and Miescier 1971.

The new offshore dump site was not closed to shellfishing because studies have shown that it does not contain economically harvestable quantities of shellfish; however, the area is being monitored. Since 1969, 14 bacteriological investigations have been conducted under the program—as many as four per year. A broad range of stations is sampled in the New York area (Pearce et al. 1976). Bacterial levels in water samples at the dump sites do not closely correspond to values found in the shellfish or the sediments (Table 1). Where shellfish are found in the closed area, the bacterial counts of the shellfish frequently exceed those of the sediments. Results of the large tidal flushing action in the ocean, plus dilution from the vast expanse of water, is illustrated by the day-to-day fluctuations of bacterial levels as shown by the July 1972 water values at stations 101, N-1, or 106. This is typical of ranges found in the various studies. Results of bacteriological levels in sediments in the New York area for the December 1971 study are illustrated on Fig. 3. A very definite buildup of pölluted sediments extending toward Raritan Bay indicates that the sludge is moving toward the bay, that prevailing currents are moving the sludge-water column toward the bay, or that short dumping is occurring. Any combination of these factors has been suggested as a probable cause of the buildup, but short dumping appears to be most probable.

Details of the buildup extending to the southwest from the dump center at the outer margin of the closure was explored in 1970 (Fig. 4). Bacterially polluted dredge spoils were found spread over 65 km^2. Most of these contaminated dredge spoils were within the original 1970 closure. All this area is presently included in the 1974 closure. The median coliform and fecal coliform values (MPN/100 ml) for bottom sediments of station 70, the former center of the sewage dump sites, were 540,000/33,000 (total/fecal coliforms) determined by Presnell and Miescier (1971). The sediments here continually show the highest bacterial values in the bight area.

The coliform populations of sediments, either from dredge spoils or sewage sludge, appear to have a different die-off rate than is generally characteristic of coliforms in seawater. Coliform exposure to the marine environment has a 90% die-off rate in 2 days according to Harris (1974). The widespread bacteria occurrence and high bacterial values found in sediments in the bight could hardly represent dumping practices of a 2-day period. However, contaminated sediments, when deposited in sufficient amounts on the bottom, appear to form an effective barrier, insulating sediments from seawater exposure. The widespread areas of bacterial contamination effectively illustrate this principle and its long term implications. Others have found that coliform bacteria may persist in estuaries over 200 days (Buck et al. 1952).

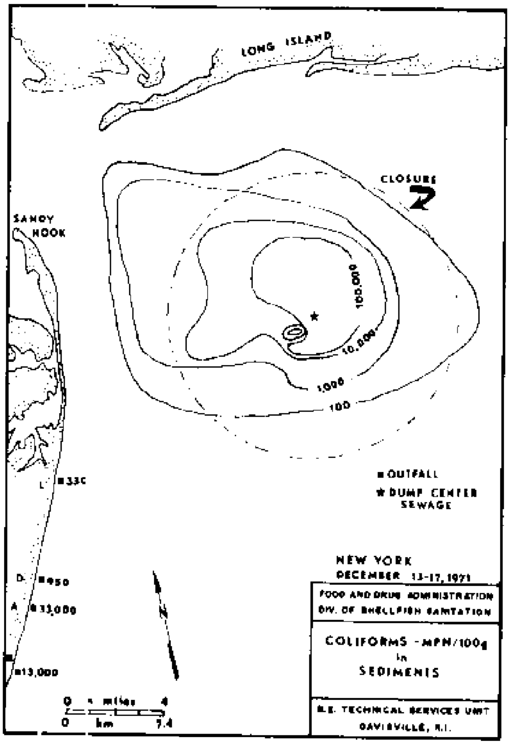

Fig. 3. Coliforms in deposits near disposal site for sewage sludge.

The acid-waste dump site, located 10 km southeast of the sewage dump center, did not show any bacterial levels in the sediments at two stations in the middle of the grounds. Efforts to find shellfish in the immediate area were unsuccessful. Since no shellfish were found and the water depths were nearly 30 m (the outer limit of sea clam habitats), no additional studies are being conducted in that section.

Short dumping

Sewage sludge may be dumped short of the intended dump center for several reasons—storms, ship malfunctions, navigational errors, etc. The problem is more serious in the Delaware region than in the New York region. Short dumping at the Delaware site could affect sizable inshore clam beds off the Delaware coast, which are not found in the New York area.

Past surveys in the former Delaware offshore site indicated that sediments contaminated with total and fecal coliforms extended from Delaware Bay out to the old dump site. Bacterial values in no way approached those of New York, as the number of sewage barge dumps was normally scheduled every 3 days (rather than daily) and the volume of sewage dumped was only a tenth that of New York. The extent of short dumping, however, appeared to be just as great. Since the site for dumping sewage in the Delaware area was moved farther offshore, the problem is greater because more harvesting area is traversed by the sludge barges.

Chemical studies

Periodically, shellfish and sediments are collected and analyzed for various trace metals. The last studies made were on shellfish collected from the new dump site being used by Philadelphia. The EPA ran chemical analyses for 13 metals: Ag, Al, Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb, Ti, V, and Zn (U.S. EPA 1971). The FDA analyzed for Hg on 9 samples of clams and 10 samples of scallops (Table 2). The Hg values are more than an order of magnitude lower than the FDA limit of 0.50 mg/kg (0.50 ppm) Hg for consumption. FDA and other agencies have found Hg concentrations to be

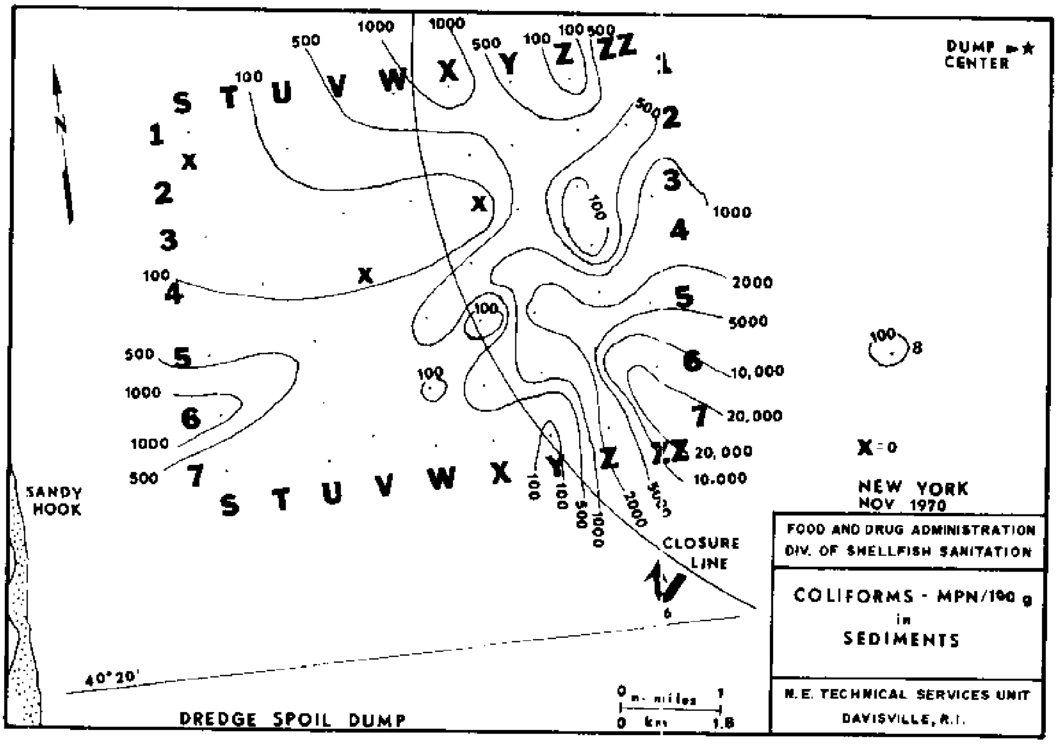

Fig. 4. Coliforms in deposits near disposal site for dredge material.

Table 2. Mercury concentrations in shellfish.

Sample	Species	Date collected (1974)	Hg (mg/kg)
558	Scallops	13 Aug	0.008
559	Scallops	13 Aug	0.008
560	*Arctica*	13 Aug	0.007
561	*Arctica*	13 Aug	0.008
562	Scallops	13 Aug	0.025
563	Scallops	13 Aug	0.025
564	Scallops	15 Aug	0.007
565	Scallops	13 Aug	0.026
566	*Arctica*	13 Aug	0.007
567	*Arctica*	14 Aug	0.007
568	Scallops	12 Mar	0.007
569	*Arctica*	12 Mar	0.012
570	Sea clam	12 Mar	0.005
571	*Arctica*	12 Mar	0.010
572	Scallops	14 Mar	0.011
573	Scallops	12 Mar	0.009
574	*Arctica*	12 Mar	0.008
575	*Arctica*	12 Mar	0.007
576	Scallops	12 Mar	0.017

<0.50 mg/kg in flounder, lobster, crab, and whiting from the New York area.

Radioactivity in sea clams at four stations was evaluated by tritium and gamma measurements. Gross gamma analysis by the Winchester Engineering and Analytical Center indicated no significant activity other than naturally occurring ^{40}K. Tritium values expressed as nCi/kg of tissue ±2 SD were 0.1±0.4 and 0±0.4, or barely in the range of detectability. Pesticide analyses of sea clams north of the New York sewage dump site—DDT, DDE, dieldrin, and heptachlor—showed only a trace, and organophosphates were not detected. DDE levels in sea clams from three stations in the former Delaware site varied from a trace to 0.19 mg/kg and 0.03–0.04 mg/kg.

Present and future programs

The Food and Drug Administration is watching the expansion of the present sea clam industry to the south and is continuing studies in the north. A plan is being considered to move the present New York Bight sewage dump site farther offshore to deeper water. The FDA is taking a cautious approach to relocating the dump because of the many problems entailed. The widespread areas of contaminated sediments illustrate what can be expected in a new area. The New York sewage sludge dump area has been a repository for grossly contaminated municipal wastes from both primary and secondary sewage treatment plants as well as for unknown quantities of industrial wastes for the past 50 years. The annual volume of waste has been 90% greater than that dumped by Philadelphia, which dumped inshore for only 12 years. Recovery of the area cannot be achieved in a few years; more than a decade may be required before a semblance of stability occurs. The long term solution to polluted shellfish or other seafoods is to cease dumping domestic and industrial wastes at sea.

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