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Response of the Habitat and Biota of the Inner New York Bight to Abatement of Sewage Sludge Dumping

Second Annual Progress Report - -1988

Environmental Processes Division, Northeast Fisheries Center

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Table of Contents

List of Figures

List of Tables

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EXECUTIVE SUMMARY

In July 1986, the Environmental Processes Division of the Northeast Fisheries Center initiated a study of the measureable effects on habitat and biota brought about by the cessation of sludge disposal at the 12-mile dumpsite in the New York Bight. This annual progress report highlights activities and findings during the second year of the study, focusing on data collected during surveys at the most heavily degraded station NY6, the enriched station R2, and the "cleaner" reference station NY11.

In summary, these findings are as follows:

1. Historical summary of physical oceanographic data of the inner New York Bight indicates the surface and bottom circulation to be strongly influenced by weather events. The Hudson plume is highly variable during periods of low runoff and is itself strongly wind influenced. Measurements of bottom currents indicate a close linkage to wind conditions. Fall and winter winds create an upward flow in the Hudson Shelf Valley. In the summer, flow is weak and down valley. "Northeaster" storms produce down-valley flows strong enough and frequent enough to cause a long-term net transport of fine-grained sediment toward and into the Hudson Canyon at the edge of the continental shelf. Indirect evidence of down-valley flow of bottom water was found during hydrographic surveys in the inner New York Bight in May 1987.

2. Preliminary analyses of sediment cores indicate significant recent particle transport down the Hudson Shelf Valley. Preliminary analysis of coprostanol in the most highly erodable sediments from the shelf valley indicates the presence of a fecal component in a significant fraction of these sediments.

3. Since reduction in the volume of sludge dumped at the 12-mile dumpsite began, dissolved oxygen minima in bottom water at NY6 have not been less than 4 mg l^{-1} (125μM). Before this reduction in sludge, values less than 0.5 mg l^{-1} (15μM) were observed in summer months.

4. Sediment redox potential has generally increased at NY6 and R2 since sludge input was reduced by 30 percent and the amplitude of seasonal redox cycles has diminished. There appears to be a convergence of values toward the level observed at NY11.

5. Contaminant metal concentrations in surficial sediments at NY6 in November 1986 (disposal level reduced by 30 percent) and November 1987 (60 percent reduction) were comparable to concentrations observed in August 1982 when sludge volume approached 8 million wet tons per year. However, by June and July 1988, after total cessation, contaminant concentrations were significantly lower.

6. Seasonal cycles in nutrient concentrations in bottom water have been described from analyses of samples collected from July 1986 to February 1988. Average concentrations of phosphate, as well as maximum concentrations and variability, parallel the station order from the presumed most contaminated to the least (NY6>R2>NY11).

7. Seabed oxygen consumption rates within the sludge dumpsite and at NY6 responded almost immediately to the cessation of sludge dumping by decreasing significantly to rates measured in coastal areas not under the influence of sludge dumping.

8. In the 10 cm of water overlying sediments polluted with sewage, total plankton respiration rates decreased in response to decreases in sludge dumping down to 4.0 million wet tons, but have remained constant throughout the remainder of the phaseout.

9. Numbers of bottom-living invertebrate species at NY6 showed no apparent recovery between summers of 1986 and 1987, but rose and became similar to R2 values in 1988. The polychaete *Capitella* sp., which had been the dominant feature of the NY6 assemblage, was scarce in 1988. Other benthic faunal variables examined showed no clear response to sludge phaseout.

10. Four species of fish, little skate, winter flounder, ocean pout, and spiny dogfish, dominated biomass at all three replicate stations as did rock crab for the megainvertebrate fraction. During the phaseout of dumping, the proportion of fish to invertebrates in trawl catches (weight per tow) increased, total biomass decreased, and differences among these three stations diminished.

11. Early results from food habit studies indicate principal prey items to be generally similar for all stations. The exception was the occurrence of *Capitella* in winter flounder diet at NY6, reflecting the dominance of this polychaete in that area.

12. Fish disease observations suggest a level of finrot (0.6 percent) similar to that found in 1983. Incidence of chitinoclasia in lobsters sampled from August to December 1987 ranged from 0 to 33 percent with overall prevalence averaging 20 percent for the study area.

13. Winter flounder tagged in the 12-mile dumpsite area were recovered by recreational fishermen inshore from Long Island waters west of Great South Bay to Lower Bay and Sandy Hook Bay in New Jersey. Five of over 1500 flounder released from this area exhibited exceptional travel eastward, some as far as to New England waters.

INTRODUCTION

BACKGROUND

Since 1924, sewage sludge from some 200 sewage treatment plants in the New York Metropolitan Area was dumped at a site in the New York Bight approximately 12 nautical miles from Sandy Hook (Fig. 1).

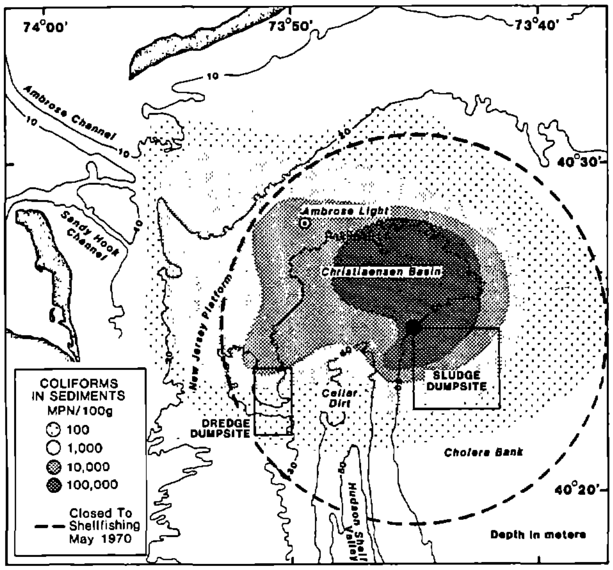

Figure 1. Location of the 12-mile sewage sludge dumpsite, dredge materials dumpsite, and area closed to commercial shellfishing in the New York Bight (after Verber 1976).

Although the number of municipalities using this site declined over time, the volume of sludge dumped increased as waste treatment facilities were upgraded. In 1974, approximately 4.2 x 10^6 wet tons were dumped, while by 1983, the volume had almost doubled, reaching 8.3 x 10^6 wet tons (Fig. 2; Santoro 1987). The volumes dumped during the early 1980s were larger than at any other sludge dumpsite in the world (Norton and Champ, in press).

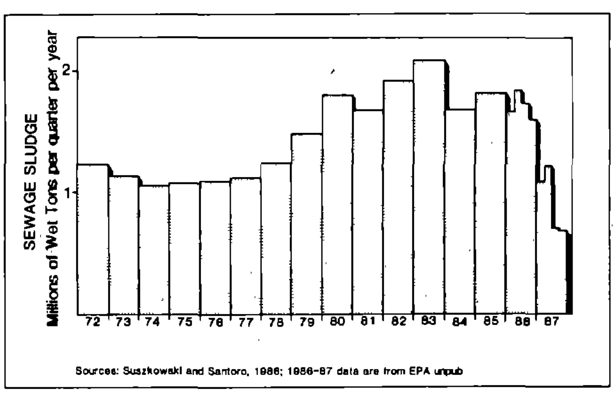

Figure 2. Estimated volumes of sewage sludge dumped in the New York Bight [annual means for 1972-85; quarterly totals for 1986-87 (after Swanson *et al.* 1985; Santoro 1987)].

Although the dumpsite is in a somewhat dispersive area with little evidence of sludge accumulation, in 1970 the U.S. Food and Drug Administration (FDA) closed an area within a radius of 11 km of the 12-mile dumpsite to commercial shellfish harvesting after finding elevated levels of coliform bacteria in sediment and shellfish (Fig. 1; Verber 1976). Additionally, indications that sewage sludge dumped at the site was the major source of sewage-related contaminants in the adjacent Christiaensen Basin and Hudson Shelf Valley led the U.S. Environmental Protection Agency (EPA) to deny further applications for dumping after December 1981 (Erdheim 1985; Santoro 1987).

Following unsuccessful litigation by New York and New Jersey to continue use of the 12-mile dumpsite, in April 1985, EPA essentially closed the site by denying requests for its redesignation (U.S. Environmental Protection Agency 1985) and selecting Deepwater Dumpsite 106 as the alternate location. EPA adopted a schedule to phase out use of the 12-mile dumpsite beginning in early 1986, and all dumping was discontinued by the end of 1987 (Fig. 3; Santoro 1987).

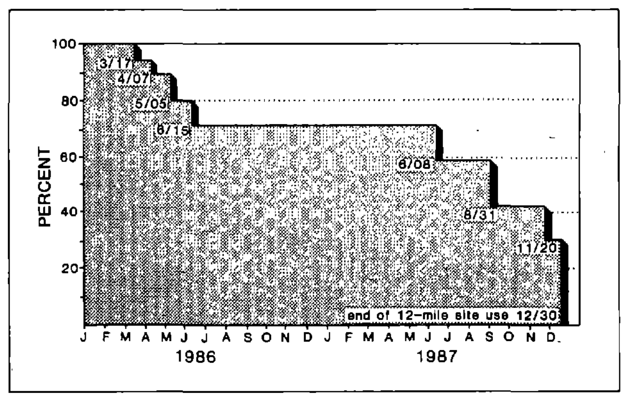

Figure 3. Phase-out schedule for sewage sludge dumping at the 12-mile dumpsite (after Santoro, pers. comm.[1]).

Since little is known about the recovery of dumpsites, discontinuation of the use of this site has provided marine scientists an opportunity to determine the response of this area to the removal of a major waste loading. Beginning in the summer of 1986, the Environmental Processes Division (EPD) of the Northeast Fisheries Center (NEC), National Marine Fisheries Service (NMFS), National Oceanic and Atmospheric Administration (NOAA), developed and initiated a study in which biological, chemical, and physical oceanographic approaches are integrated to document changes in living marine resources and their habitats (Environmental Processes Division 1988).

Results will document the response and extent of recovery of the site and its environs, which will provide information useful for:

1. Defining when and if the area can be reopened for shellfishing;
2. Assessing the changes in distribution and abundance of resource species in the vicinity of the site;

[1] **Personal communication from E. Santoro (USEPA, New York, New York 10278).**

3. Determining probable sediment resuspension and transport of associated sludge components out of the Christiaensen Basin;
4. Quantifying the response of the area to the changes in pollutant loading in a way which will permit the use of this information for predicting response rates for other proposed temporary dumpsites; and
5. Evaluating the management decision made by EPA to divert the dumping of sewage sludge to Deepwater Dumpsite 106.

DEVELOPMENT OF THE STUDY

Several factors had to be considered in the development of a practical and efficient study. The first relates to the long history of environmental degradation in the area which precludes the use of predumping conditions as a baseline. Data from the first year's sampling, during which the volume of sludge dumped was reduced by 30 percent (Fig. 3; Santoro 1987) serves as a "baseline" and assessment of changes will be based on comparisons of this baseline with later sampling. Other information collected from as far back as the late 1960s does exist as well for several of the variables under study and will also be used to assess changes.

A second difficulty pertains to the separation of pollutant effects from various sources. It is estimated that on average, 16 percent of the contaminant load in the New York Bight Apex is traceable to sewage sludge, with most contributed by the Hudson-Raritan plume (27 percent), and to dredged material which is dumped on the western edge of the Christiaensen Basin (Fig. 1; Stanford and Young 1988). Nevertheless, by conducting synoptic surveys and initiating the study before a significant curtailment of dumping occurred, evaluation of the effects of abatement should be possible. Possible changes may include the following:

1. Dispersion of sewage sludge and cleansing of the Christiaensen Basin will be influenced by local windfield conditions and resultant changes in bottom-water circulation in the upper Hudson Shelf Valley.
2. Changes in water and sediment chemistry should occur, including reduction of sediment trace metals in sludge depositional areas. However, concentrations of certain organic compounds (*e.g.*, PCBs) may be more stable. Seabed oxygen consumption and nutrient regeneration rates should be reduced as benthic community metabolism and organic loading decrease.
3. Microbial concentrations should be reduced and bacteria indicative of sewage contamination should decrease to acceptable levels and permit shellfish beds to be reopened for harvesting.
4. In areas which are presently heavily polluted, numbers of benthic macrofaunal species should increase; populations of *Capitella capitata*, an opportunistic polychaete found in disturbed areas, should decrease.
5. While dumping continues, abundance, distribution, and species composition of finfish and invertebrate communities should differ among areas which are bathymetrically similar, but represent a gradient of sludge influence. Following cessation of dumping and expected shifts in sediment contaminants and benthic forage species, these spatial differences should be reduced.

SAMPLING STRATEGY

The study is developed around two complementary sampling series: (1) a replicate survey; and (2) a broadscale survey, each conducted in alternate months, except for August when both are conducted to monitor rapid changes likely to occur at that time of year (Fig. 4; Environmental Processes Division 1988).

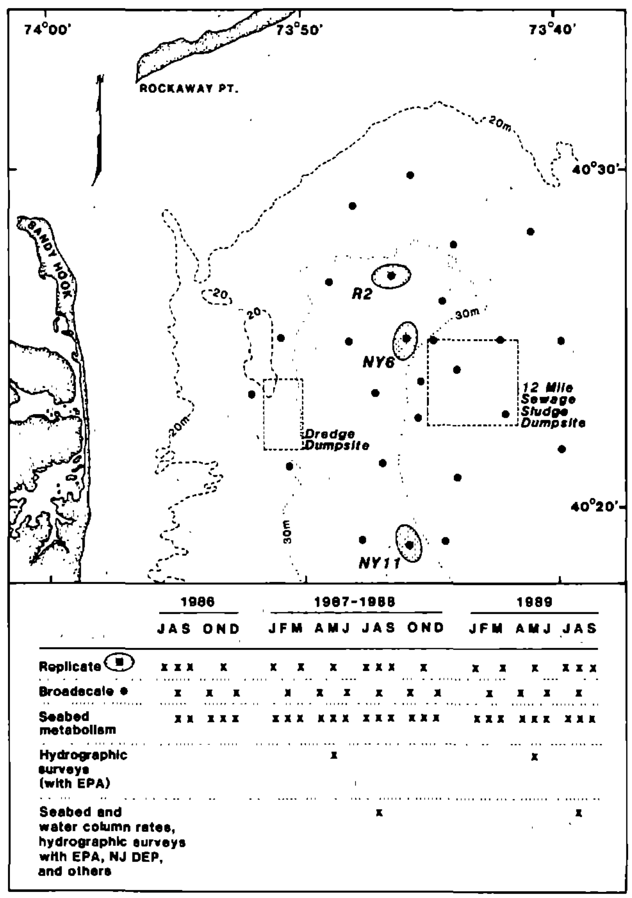

Fig. 4. Replicate, broadscale, and seabed metabolism stations (also Fig. 19) and survey schedule of the 12-mile dumpsite study.

In the replicate survey, repeated measurements (n=8) for most variables (Table 1) are made at three stations (Fig. 4) which are similar bathymetrically and for which historical data exist, but which represent different levels of sewage sludge accumulation and effects. The most heavily

Table 1. Variables measured during the 12-mile dumpsite study

Habitat		Biota
Water	**Sediments**	
Bottom Water	Chemistry	Resource species
Dissolved oxygen (R,B)[1]	Heavy metals (R,B)	Distribution/abundance (R,B)
Temperature (R,B)	Organic contaminants (R,B)	Diet (R)
Salinity (R,B)	Sulfide, pH profiles (R)	Winter flounder
pH (R,B)	Redox potential (R,B)	Red hake
Sulfide (R,B)	Sediment BOD (R)	Silver hake
Nutrients (R,B)	Chlorophyll pigments (R,B)	Lobster
Turbidity (R,B)	Total organic carbon (R,B)	Gross pathology (R)
		Winter flounder
Water Column	Characteristics	Lobster
Temperature	Grain size (R,B)	Tissue organics (R)
Salinity (CTD)	Erodibility	Winter flounder
Oxygen		Lobster
Current measurements	Rates	Migration (tagging) (B)
(moored meters)	Seabed oxygen consumption	Winter flounder
	Sedimentation	Lobster
		Benthos
		Macrofauna abundance/diversity (R,B)
		Meiofauna abundance/diversity (R,B)
		Bacteria - sediments
		Fecal and total coliform (R)
		C. perfringens (R)
		Vibrio spp. (R)
		Total count (R)
		Bacteria - shellfish

[1] R = Replicate survey
B = Broadscale survey

degraded, NY6, is located approximately 1.6 km downslope from the corner of the dumpsite where the heaviest dumping has occurred; it is thought to have the greatest accumulation of sludge constituents. A second station, R2, is about 3.4 km north of the dumpsite on the north edge of the Christiaensen Basin. Benthic communities in this area are presumably "enriched" with a high biomass of tolerant, though not necessarily pollutant-indicator, species. The third replicate station, NY11, is nearly 10 km south-southwest of the dumpsite center on the eastern shoulder of the Hudson Shelf Valley. It is considered to be the least polluted of the three sites, with the lowest concentrations of sediment contaminants and with benthic macrofauna typical of upper shelf valley sediments.

The broadscale survey consists of single, nonreplicated bimonthly measurements made every other month at 25 stations covering most of the inner bight and including all major habitat types. Station selection was based on considerations of bathymetry, known distribution of contaminants, patterns of environmental variables, dumping and dispersion patterns of sewage sludge, and existing historical benthic data. To add to our data base for interpreting the effects of sludge abatement, we have collaborative support from several agency and university researchers.

To date, all replicate and broadscale surveys have been completed on schedule; however, backlogs in sample processing and data analysis in some areas may delay integration of results.

RESULTS

Results presented in this year's progress report will focus on findings from the three replicate stations; *i.e.*, the heavily degraded NY6, "enriched" R2, and the reference station, NY11, concentrating on the period through December 1987 during which dumping was phased out. Where preliminary information is available for the following months, it is also included. Since many of the anticipated changes at the dumpsite are expected to be influenced by hydrographic processes, a review of the physical oceanography of the bight provides a background against which these changes will be assessed.

PHYSICAL OCEANOGRAPHY OF THE NEW YORK BIGHT -- A REVIEW

For the purposes of this summary, the inner New York Bight is defined as that area bounded on the south by 40°00' North latitude, on the east by 73°15' West longitude, on the north by the Long Island shore, and on the west by the New Jersey shore (Fig. 5). This area of the bight is roughly a square about 30 nautical miles on each side of generally shallow (<50 m) water, except for the Hudson Shelf Valley which has depths of 40-80 m and extends diagonally from the northwest corner of the area. The northwestern half of the area contains three dumpsites, including the recently deactivated 12-mile dumpsite for sewage sludge (Fig. 5).

Circulation of the water in this portion of the New York Bight is strongly influenced by that of the whole bight (Montauk Point to Cape May) and by that of the Middle Atlantic Bight (Nantucket Shoals to Cape Hatteras). However, there are some local influences which have significant effects on the circulation, principally the bathymetric configuration of this corner of the bight and the flow from the Hudson-Raritan estuary.

Water-mass properties of this portion of the bight are influenced by circulation, weather, and effluents reaching the coastal marine waters. Contaminants enter the water column by way of estuarine effluents, dumping of wastes, and atmospheric fallout.

Upper Water Circulation and Estuarine Effluent Plumes

Circulation of the surface layer of the New York Bight is strongly influenced by short-term (multiday) weather

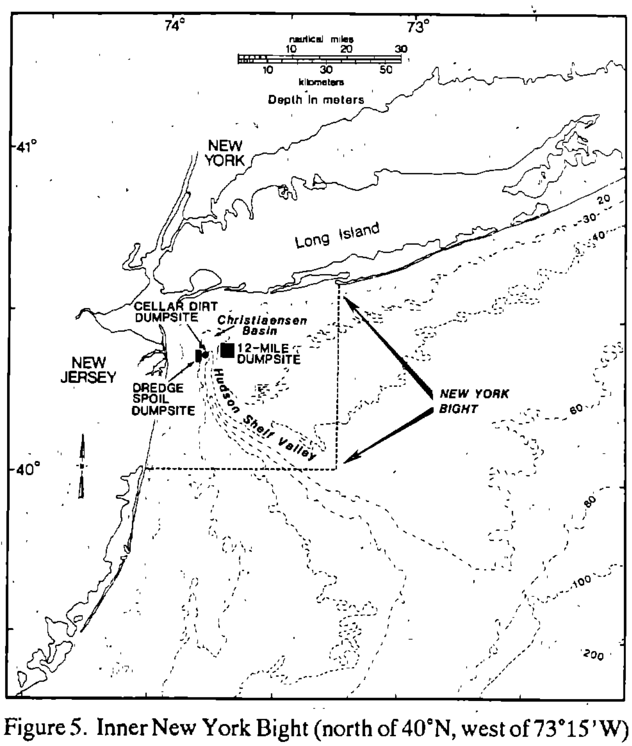

Figure 5. Inner New York Bight (north of 40°N, west of 73°15'W) showing features of interest in the inner New York Bight.

conditions. Beardsley and Boicourt (1981) point out that synoptic scale (greater than two days and/or 500 km) disturbances are responsible for most of the surface wind variance over the shelf and open ocean. This is manifested in the Middle Atlantic Bight as frequent, intense cyclones (low pressure systems); 2.5 per month in summer, five per month in winter, and more intense in winter than in summer. The mean surface wind stress is eastward-southeastward except in summer when it is northeastward, and the mean stress is generally stronger offshore (2-8 times at the shelf edge) and veers cyclonically (up to 30°) with increasing distance offshore.

Ketchum *et al.* (1951) provided the first extensive hydrographic survey of the apex of the New York Bight. Their salinity and temperature data showed that the volume of river water in the plume entering the bight from the Hudson River rarely exceeded one percent of the total volume of the bight apex. They concluded that the residence time of river water in the bight apex was only 6-10 days, indicating relatively rapid dispersion of the plume. Han and Niedrauer (1981) extended Ketchum's analysis with more recent data and obtained residence times ranging from 5.5 to 12 days with an average of 6.8 days.

Bight waters are dominated by spring season runoff (Charnell and Hansen 1974; Bowman and Wunderlich 1977). During these high discharge periods, the Hudson River plume is generally parallel to the New Jersey coast due to a southwest deflection caused by the Coriolis effect and shelf currents. Winter mixing due to storms can completely mix the entire water column rapidly; re-establishment of the river plume in the bight takes approximately two days. During periods of lesser discharge (summer), location of the plume becomes highly variable and strongly wind influenced (Bowman 1978). Mechanisms by which the plume is mixed into the surrounding shelf waters are not well understood.

Young and Hillard (unpubl. ms.) found suspended particle fluxes highest (about 30 x 10^{-6} g s^{-1} cm^{-2}) during periods of spring runoff as a consequence of increased flow, not increased concentrations of suspended particulate matter. At other times, the quantity of particulate matter carried by the Hudson plume appeared to be "... negligible compared with discharge through other adjacent coastal sectors ...".

Also found in the surface plume of the Hudson River were suspended particulates, present in concentrations usually higher than in surface waters off Long Island or bottom waters off New Jersey. The greatest concentrations, except in high runoff periods, were in bottom waters off western Long Island, assumedly due to resuspension of sediments by wave action. In April, during high runoff, the particulate plume extended broadly southward along the New Jersey coast at least 70 km. Drake (1974, 1977) and Nelsen (1979) found average surface concentrations of suspended particulate matter in the plume to range from 1 to 10 mg l^{-1} and the edge of the plume generally lying 5-10 km offshore. Fedosh and Munday (1982) analyzed 86 cloud-free images from satellite sensors for indications of surface configuration of the Hudson-Raritan plume. In the mouth of the estuary, the turbidity plumes were typically lobed, filled the entire entrance, and curved southward in the alongshore drift. Those plumes which extended farther seaward tended to be more linear in form and spread farther laterally. Plumes extended farthest seaward under the influence of northwesterly winds. Thermally mapped plumes had longer frontal regions than turbidity plumes. They found 51 percent of the plumes extended to the dredge spoil dumpsite (25 km) and 12 percent reached the sewage sludge dumpsite (35 km). None of the 82 plumes studied reached the acid waste dumpsite (45 km).

Surface circulation in the Middle Atlantic Bight was summarized by Beardsley and Boicourt (1981). They found the annual mean flow in the surface layer was southwest at 2 cm s^{-1} or less. Standard deviations about the annual mean speeds ranged to over 20 cm s^{-1}; assumedly the variance was the result of wind-driven currents associated with passage of storms.

Runoff through estuaries onto the shelf raises the sea level near shore, producing an offshore pressure gradient which results in a flow parallel to the coastline, oriented so that the shore is on the right when looking in the direction of flow. From April to September, winds generally from the southwest oppose this flow, but only rarely is it reversed by the wind. Bumpus (1969) reported on a period from 1962 to 1966 when reversals were common during the late spring and summer months. Bumpus believed relatively low rainfall and runoff to be responsible for the reversals of surface currents, because of the reduction of the normal offshore pressure gradient.

Density Stratification

The seasonal development of a surface mixed layer of low-density water and underlying pycnocline influences the dispersion of pollutant materials introduced into the surface waters. Those pollutants, which are dissolved or suspended, remain in the mixed layer above the pycnocline which acts as a virtual barrier to vertical mixing. Pollutant components which are slightly more dense than sea water in the mixed layer sink slowly to the pycnocline and accumulate there (Orr *et al.* 1980) because of the increased water density. As a consequence of these characteristics, the water column involved in the dispersion of dissolved or suspended pollutants is shallower in the late spring and summer when stratification occurs, than in the winter when the water column is vertically isothermal in the upper 100-200 m.

In the Middle Atlantic Bight, the water volume is divided into a narrow, coastal, low-salinity band affected primarily by outflow from three major estuaries, a shelf region ranging between 20 and 100 m depth, and the outer shelf region at the shelf break which receives intrusions of more saline, warmer slope water and is characterized by the shelf-slope front.

The shelf region is occupied by low-salinity water ranging from 30 ppt near shore to 35 ppt at the bottom along the shelf break. The average salinity over the shelf is around 32.5 ppt at the surface to 35 ppt along the bottom. Water temperature ranges seasonally from 2°C in the nearshore zone during February-March to 30°C in late summer, (generally averages 25°-26°C). In winter, the water is vertically homogeneous with temperature increasing seaward at all depths. The water begins to warm by late April with a thermocline established by early June and as the surface waters continue to warm, the thermocline deepens and increases in intensity, reaching a maximum in mid-August to early September. The warmer surface waters then begin to mix downward increasing the bottom temperature. Maximum bottom temperatures are reached approximately one month after the surface temperature has reached a maximum (Ketchum and Corwin 1964). Cooler air temperatures by this time have produced fall overturn which breaks down thermal stratification and enhances vertical mixing.

The intensity of the thermocline varies as a function of wind and solar heating. With an early spring onset of southerly winds and atmospheric heating, a strong thermocline can reduce vertical mixing to near zero. In addition, large freshwater runoff from the estuaries creates large salinity differences and further intensifies stratification. Strong southerly winds can also provide a mechanism for invasion of warm high-salinity slope water onto the shelf at mid-depths or along the bottom, compensating for offshore transport in the surface layer. During the winter when this occurs, a weak thermohaline stratification appears with slope water moving in along the bottom. During the summer, the warmer, more saline water appears to move along the shelf into the vicinity of the 10 m isobath (Boicourt and Hacker 1976).

Seasonal variations in runoff, mainly in the Hudson and Connecticut Rivers, also influence the time of onset and intensity of density stratification of the water column in the inner New York Bight. The period of high discharge for both rivers is March-May, with the peak flows occurring usually in April (Koppelman *et al.* 1976; Bowman and Wunderlich 1977). Minimum surface salinities in the nearshore area occur in May-June, when the Hudson-Raritan estuarine plume is most pronounced off northern New Jersey. Surface salinities then increase slowly during the summer and fall until the overturn (about November), then remain high during the winter until decreasing during runoff in spring.

The seasonal cycles of salinity, temperature, and density are apparent in a scatter plot of weekly observations made over 5 years (1983-1987) off Long Branch, New Jersey[2] (Fig. 6) and in an analysis of 11 years (1977-1987) of MARMAP survey data collected in the inner New York Bight[3] at a station 6 km away at the same depth of 21 m (Fig. 7). The annual cycle of sea-surface temperature appears to be reasonably regular, but the surface salinity cycle is more variable, as are cycles for bottom temperature

[2] **Personal communication from John O'Reilly (NOAA, NMFS, Sandy Hook Laboratory, Highlands, NJ 07732).**

[3] **Personal communication from David Mountain (NOAA, NMFS, Woods Hole Laboratory, Woods Hole, MA 02543).**

and salinity. Changes in the vertical gradient of density, as indicated by the difference between top and bottom density (Fig. 7), follows a pattern similar to surface temperature and salinity.

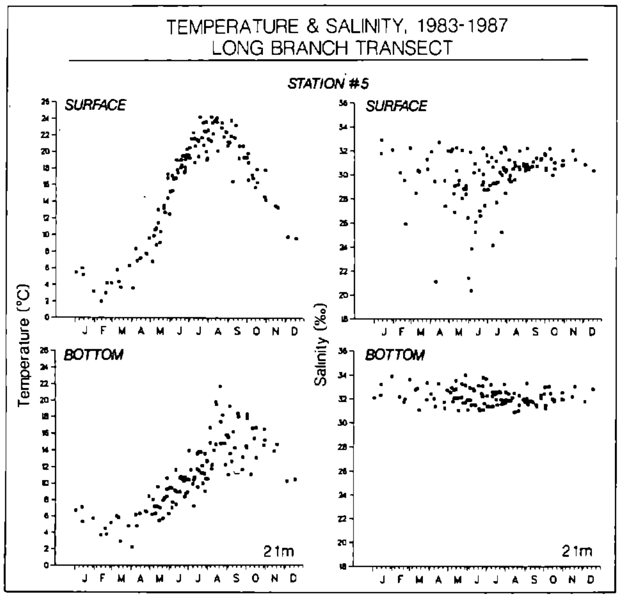

Figure 6. Scatter plots of temperatures (°C) and salinities (ppt) measured on a transect off Long Branch, New Jersey (40°18.2'N; 73°53.4'W).

Upwelling and the Cold Pool

During periods of prevailing southwesterly winds, mostly in summer months, upwelling should occur along the New Jersey-Virginia coast. An analysis of surf temperatures, wind velocities, and nearshore temperature sections off Monmouth Beach, New Jersey (Hicks and Miller 1980), demonstrated the linkage between southerly winds and shoreward motion of cold bottom water, often into the surf zone.

In the analysis of data and events involved in the development of anoxic conditions in the bottom water during the summer of 1976, some evidence of upwelling was uncovered. Mayer, Hansen, and Minton (1979) found indications of upwelling off northern New Jersey in 5 of 13 months and off western Long Island in 8 of 12 months from current-meter data. In each case it was persistent enough to register in monthly mean current vectors. Analysis of windfield observations for the inner New York Bight for February-June 1976 by Diaz (1979) produced monthly mean offshore transports (computed Ekman) in the surface layer each month, but strongest off New Jersey in February. Steimle and Sindermann (1978) reported observations of dead and dying fish and hydrogen sulfide odors in the surf zone of the south central coast of New Jersey in late July and early

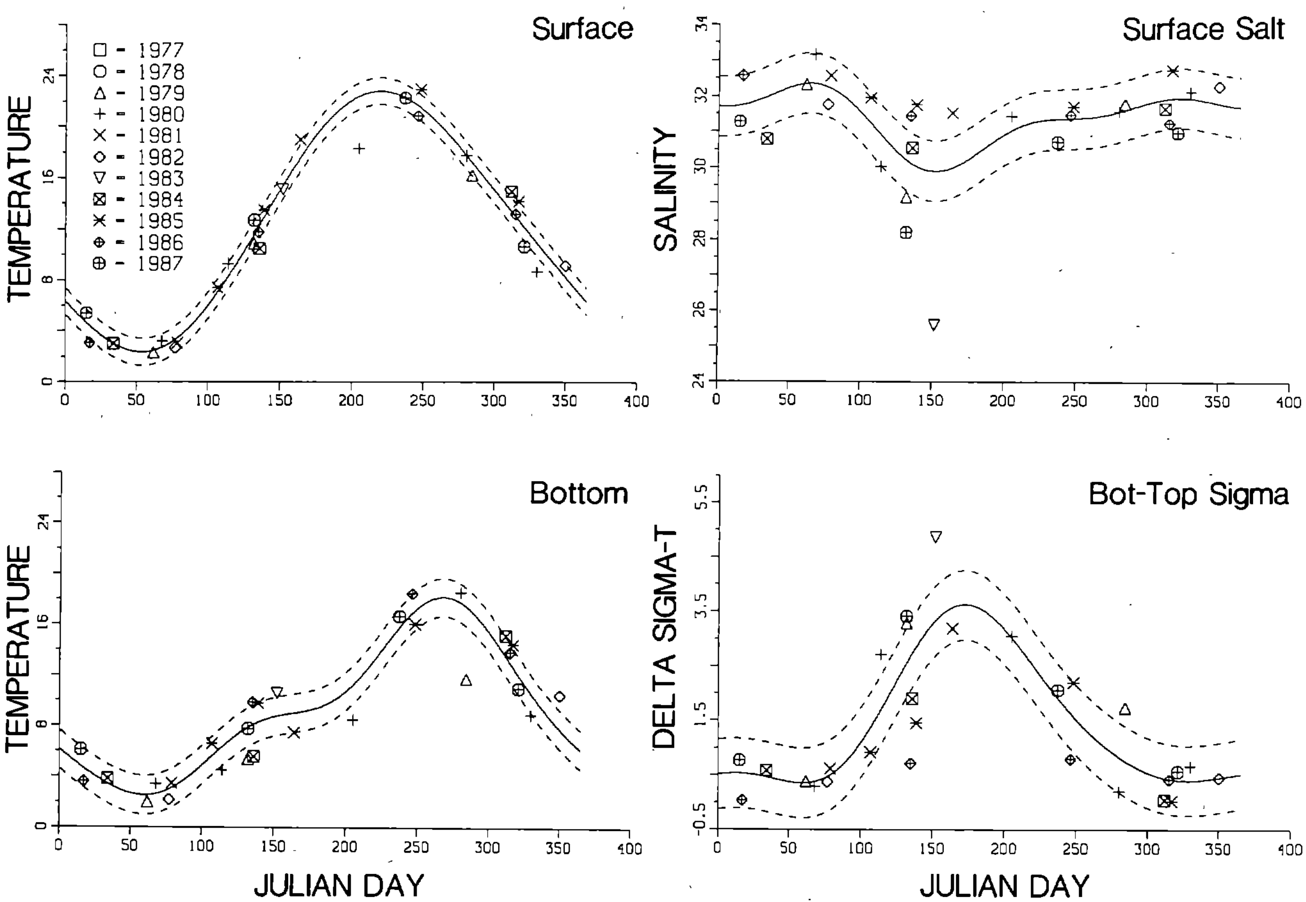

Figure 7. Scatter plots and annual trend lines for data collected on MARMAP Station 187 during 1977-87. Temperatures in °C, salinity in ppt, and density differences in sigma-t units.

August, indicating upwelling of the anoxic bottom layer clear into the surf zone. Nelsen *et al.* (1978) and Mayer *et al.* (1982) have identified a relationship between local wind stress and onshore flow of bottom water in the Hudson Shelf Valley (upwelling) in the apex of the New York Bight (for further details see Swanson and Parker 1988).

Field studies collecting current-meter, hydrographic, and wind data mainly along a 10 km transect of 20-30 m depths off eastern Long Island's south shore (about 72.6°W) were reported by Churchill (1985). He found coastal boundary-layer water flow (about 0-10 km offshore) was dominated by upwelling-downwelling episodes at subdiurnal frequencies. These cross-shelf events were generally wind driven, but in late summer offshore flow was measured though negligible wind stresses prevailed. The mean longshore flow over the period studied (four months: September 1975, May 1977, March 1978, and August 1978) was to the southwest and not driven by wind, but by a longshore pressure gradient.

The cold pool, so named by Bigelow in 1933, is a continuous subsurface water type on the continental shelf bottom extending from Georges Bank to Cape Hatteras. This feature was referred to as remnant "winter water" by Bigelow (1933), Ketchum and Corwin (1964), and Whitcomb (1970) who believed it formed from the winter cooling of mixed Middle Atlantic Bight shelf water. With hydrographic data portrayed by Colton *et al.* (1968) and Limeburner *et al.* (1978), the feature can be traced to the north along the eastern edge of Georges Bank and into the Gulf of Maine, where temperature and salinity relationships suggest a possible source of replenishment (Beardsley *et al.* 1976; Hopkins and Garfield 1979). To the south, the cold pool feature has been traced past the offing of Chesapeake Bay to Cape Hatteras where three major water masses (shelf, slope, and Gulf Stream) meet.

The cold pool becomes an identifiable feature each year beginning with spring surface warming and the onset of thermal stratification (generally in late April to early May) and lasts throughout the summer into early fall, until the normal seasonal overturn (from October to December, but usually November) mixes away the vertical density structure and the water column becomes vertically isothermal. It normally covers the bottom between the 40 and 100 m isobaths (20-60 fm), over an area of approximately 88,000 km^2. It averages about 35 m in thickness and extends up to the base of the seasonal thermocline (within 20-30 m of the surface). This represents a volume of 3,200 km^3, or about 30 percent of the total volume of shelf water in the Middle Atlantic Bight.

By virtue of its bathymetric location, the cold pool is a relatively slowly changing feature when compared to the more active zones of lateral mixing seaward and vertical mixing shoreward, with a long-term average flow of about 1-3 cm s^{-1} southwestward (Mayer, Hansen, and Ortman 1979). However, it is in a state of constant change, and can be acted on by several processes either singly or simultaneously. Some of these include:

1. **Wind events** which can cause upwelling nearshore, moving the shoreward leading edge of the cold pool toward the beach and sometimes into the surf zone (Hicks and Miller 1980). Other wind events, if persistent, can influence the offshore edge of the cold pool, forcing it and the shelf water-slope water front to move seaward (Csanady 1978).
2. **Gulf Stream rings** which migrate along the shelf edge to the southwest through the Middle Atlantic Bight causing perturbations in the subsurface shelf-slope front and the seaward edge of the cold pool by advecting upper slope water (greater than 12°C) shoreward over the edge of the continental shelf. These rings can even move the cold pool off the bottom and cause it to bulge seaward, leading to "calving" (Whitcomb 1970; Wright 1976).
3. **Bathymetric features** such as the Hudson Shelf Valley and submarine canyons which underlie the outer edge of the cold pool at several locations, from Corsair Canyon at the eastern edge of Georges Bank to Norfolk Canyon at the southern end of the Middle Atlantic Bight can affect the cold pool. Flow in the canyons can cause movement in the position of the cold pool. According to Mooers *et al.* (1979) down-canyon transport of cold-pool water was observed along the southwestern side of Wilmington Canyon. In contrast, Han and Niedrauer (1981) observed that cold pool water did not sink into the Hudson Shelf Valley, but rather stayed at the same level as on the surrounding shelf. Nevertheless, bottom-water flow in the canyons and shelf valleys should act to displace the cold pool either onshore or off.
4. Ou and Houghton (1982) developed a one-dimensional (along-shelf) model to explain temperature features of the cold pool in terms of variable flow and heat transfer. One of the features elucidated is a relative temperature minimum just north of the Hudson Shelf Valley produced by the increase in heat gain experienced by the cold pool (from beneath) as it passes over the valley.

Bottom Currents and Sediment Transport

Sediment transport in the Middle Atlantic Bight appears to be dominated by strong, infrequent storm-induced events separated by periods of little transport. Finer sediments may remain in suspension for a long enough period of time to move shoreward in the estuarine-like gravitational circulation apparent, at least, on the inner shelf. Some of these fines may find their way into estuaries.

Seabed drifter studies have provided an inexpensive way to study the net bottom currents on the continental shelf. Bumpus (1973) released thousands of seabed drifters with about 16 percent recovered from regional beaches or waters. These data showed a general onshore (northward) bottom movement south of Long Island and New England at speeds of about 1-2 cm s^{-1}. Off the New Jersey coast, currents converge towards the coast between about 38.5°N and 40°N latitude. Hardy *et al.* (1976) and Charnell and Hansen (1974) found similar results in other long-term seabed drifter studies. They also noted a strong near-bottom estuarine-like circulation into Long Island Sound and New York Harbor at 2 cm s^{-1}. They found an apparent divergence along the Hudson Shelf Valley; east of the valley bottom drift was northward while west of the valley more westward bottom drift was observed.

Long-term near-bottom currents were studied in the Hudson Shelf Valley and Canyon system by Keller *et al.* (1973) and Nelsen *et al.* (1978). Current meter measurements showed flow reversals up and down canyon with velocities of 8-15 cm s^{-1}, with a maximum of 27 cm s^{-1} in the upper central portion of the canyon. Sediment texture and organic carbon content of the sediments, however, indicate a long-term down canyon (seaward) transport of fines to the continental rise. Nelsen *et al.* (1978) concluded from current-meter data and wind records that upchannel near-bottom flow was related to westerly winds driving surface waters seaward during the winter, while during the summer seaward near-bottom flow prevailed.

Biscaye and Olsen (1976) measured suspended particulate concentrations in the New York Bight and found local resuspension of fine bottom sediments due to erosional currents. These resuspended sediments were limited to near-bottom waters during the stratified seasons due to limited vertical mixing across the pycnocline.

Simultaneous measurement of near-bottom currents and sediment transport or sediment bedforms, although difficult, has been conducted by some workers. McClennen (1973) reported that sediments off the New Jersey coast in four water depths between 59 and 143 m were being reworked based on ripple marks and sedimentary structure analyses. Mean currents varied between 11.8 and 19.5 cm s^{-1} measured at 1.5-2.0 m off the bottom. Based on critical erosion velocities, McClennen determined that shelf sediment may be eroded and transported as much as 30 percent of the time by currents and eroded eight percent of the time by wind waves at 30 m depth. Freeland *et al.* (1976) discussed similar results for linear bedforms indicating sand movement.

Swift *et al.* (1976) showed the importance of storm events in causing large sediment transports with sustained southwestward near-bottom currents of greater than 50 cm s^{-1} for approximately 12 h. These short, efficient, storm-related, large-scale transports are separated by longer periods of quiescent minimal transport. Data indicated westward transport off Long Island and southward transport off the New Jersey shore.

Two large depositional areas located within the Middle Atlantic Bight are quite different from the more common shelf relict ridge and swale topographies discussed by Swift *et al.* (1976) and Freeland *et al.* (1976). One, the so-called Christiaensen Basin, is located within the apex of the New York Bight (Fig. 5). This topographic low has much higher levels of clay and silt than surrounding sand areas. Although the sewage sludge (12-mile) dumpsite is located near the Christiaensen Basin, the basin has not shown any measurable bathymetric changes which could be attributed to dumping of sludge (Freeland *et al.* 1976). The area outside the Christiaensen Basin also contains a few mud-sand scour patches with linear bedforms indicating sand movement. The Christiaensen Basin may act as a "sediment trap," trapping fine sediments which are only aperiodically resuspended by major storm events.

Tidal currents have regular semidiurnal (12 h) periods and are generally strongest near the coast. The tidal currents are generally described as rotary, constantly varying in direction, thus describing what are known as tidal ellipses. Beardsley *et al.* (1976) and Patchen *et al.* (1976) noted from current-meter data that tidal currents in the Middle Atlantic Bight are related to the M_2 and K_1 tidal components. The tidal currents decrease in magnitude away from the coast and also decrease with depth. Scott and Csanady (1976) noted moderately strong (20 cm s^{-1}) tidal currents 11 km south of Long Island during September 1975.

Subtidal currents are mostly barotropic (caused by changes in sea level) and constant with depth. The sea-level changes are due to meteorological forcing caused by either local winds or the propagation of a shelf wave from some distant oceanic or meteorological event (Scott and Csanady 1976; Beardsley and Flagg 1976; Bennett and Magnell 1979; Ou *et al.* 1981). Beardsley and Butman (1974) proposed a conceptual model to explain simultaneous measurements of sea-level changes along the coast and current-meter measurements on the continental shelf. They proposed that offshore winter storms dominate the shelf circulation by causing strong westward wind stresses south of New England which drive water shoreward (north) due to Ekman transport. This causes sea level to rise along the coast which sets up a cross-shelf (seaward) pressure gradient force causing a flow to the west due to the geostrophic force balance. This current parallels the westward wind stress which caused the initial setup. Storms located over the land, however, cause eastward wind stresses setting up a large alongshore pressure gradient force which causes large current oscillations, but little net alongshore flow. These storms cause Ekman transport away from shore and cause sea levels to fall along the coast. Bishop and Overland (1977) showed that during the winter season, wind-driven circulation predominates on the shelf while during the summer, density-driven currents are more common.

Many workers have found that these meteorologically forced subtidal currents increase in magnitude away from the coast and decrease near bottom. Boicourt and Hacker (1976) showed near-bottom currents of between 12 and 36 cm s^{-1}, increasing in a seaward direction. They also showed that cross-shelf currents may be enhanced by near-bottom onshore flow along the outer shelf in response to offshore Ekman flow at the surface under certain wind conditions.

Beardsley *et al.* (1976) showed that currents veered shoreward closer to bottom, suggesting that near-bottom materials may move shoreward, in agreement with the earlier seabed drifter work already discussed.

Seasonal stratification due to the development of a pycnocline in shelf water has several implications for the current structure, response, and sediment transport of the region. Patchen *et al.* (1976) determined for the apex of the New York Bight that weak stratification resulted in the entire water column responding strongly to the wind, while response was limited to the upper stratified layer when a thermocline was present. Han *et al.* (1980) showed a coupling of currents above and below the pycnocline in the New York Bight, with near-bottom currents varying from northward to southwestward. Shonting (1969) reported similar results for Rhode Island Sound where the surface flow above the seasonal pycnocline was strongly isolated from the lower layer.

Nearshore current-meter data in the apex of New York Bight showed a strong onshore transport component (Swift et al. 1976). A similar onshore transport was noted by Han and Mayer (1981) on the Long Island inner shelf near the 37 m isobath in late fall. Current records showed both tidal and subtidal forcing, while onshore near-bottom flow appeared to respond to the net offshore surface flow resulting in a zero net cross-shelf flux. Only strong onshore or easterly winds produced an offshore near-bottom component.

Annual average bottom-water flows in the Middle Atlantic Bight summarized by Beardsley and Boicourt (1981) were approximately along-shelf toward the southwest with speeds of about 0.5 cm s^{-1}, or less. The standard deviations about the annual means ranged up to over 15 cm sec^{-1}, assumedly reflecting storm-induced currents. The bottom-current vectors, however, were more onshore in direction than the upper-water vectors at the same locations.

Mayer *et al.* (1982) summarized what was previously known of the circulation in the Hudson Shelf Valley and analyzed long-term current-meter records from moorings placed principally along the axis of the valley (Fig. 5). They found average current velocity in the valley (below the depth of the surrounding shelf) is up-valley, about 2 to 5 cm s^{-1}. In winter, and with passage of cyclonic storms, the several-day average up-valley flow can reach 20 cm s^{-1}, and up to 50 cm s^{-1} for shorter periods. Down-valley flow also can result from storm passage, depending upon the path of the storm. Weak (1-2 cm s^{-1}) down-valley flow is associated with calm (no-wind) periods and is the expected consequence of the general, all-season weak flow of shelf water to the southwest interacting with the bathymetry of the valley.

Tidal bottom (1 m) current ellipses (M_2 components) are presented by Moody *et al.* (1984, p. 98) for two locations in inner New York Bight: Station 15 at 40°26'N, 73°28'W (14 n mi southwest of Fire Island Inlet) and Station LTG at 40°08'N, 73°38'W (19 n mi east of Manasquan Inlet on the eastern edge of the Hudson Shelf Valley). Maximum tidal excursions for bottom water computed from the ellipse parameters for these locations follow:

Sta. 15: 0.99 km along 302° - 122° max speed 7 cm s^{-1}
0.14 km along 212° - 32° max speed 1 cm s^{-1}
Sta. LTG: 0.57 km along 318° - 138° max speed 4 cm s^{-1}
0.14 km along 228° - 48° max speed 1 cm s^{-1}

In summary, the long-term mean bottom currents in the inner New York Bight are approximately parallel to the shoreline (westward off Long Island, southward off New Jersey), but with a definite shoreward component. Bottom currents in the Hudson Shelf Valley follow a different pattern and seem to be closely linked to wind conditions. During periods of westerly (eastward) winds, including most of the fall and winter months, flow is up the valley toward the Christiaensen Basin. During periods of weak winds, mainly in summer, flow is weak and down the valley. During storms, flow can be much stronger either up or down the valley, the direction depending on the presence of westerly or easterly wind components, which in turn depends on the track of the low-pressure area causing the storm. A low which follows a track parallel to the coast but offshore usually yields strong northeasterly (southwestward) winds long enough to cause strong down-valley bottom currents. Episodes of down-valley flow apparently are strong enough and frequent enough to cause a long-term net transport of fine-grained sediment down the valley and into Hudson Canyon at the edge of the continental shelf.

Note: Parts of the foregoing text were taken directly from NOAA Technical Memorandum NMFS-F/NEC-17 (Ingham *et al.* 1982).

Results from the 12-Mile Dumpsite Study

Near-bottom current meters have been moored at nine locations in the vicinity of the 12-mile dumpsite and upper Hudson Shelf Valley for varying periods with varying success. At this time, the first-order processing of data has been completed by contractor on just four of the meters, and no analyses have been conducted yet. Four more current meters will be retrieved during June 1989, then all the remaining meter data records (5) will be processed by a contractor, and analyses will be undertaken. Also at this point, copies of the data records will be forwarded to EPA/Environmental Research Laboratory, Narragansett, RI, for use in development of a 3-D dispersion model.

Interpretations of temperature, salinity, and dissolved oxygen data collected on two cruises in 1987 (May and August) at 324 hydrographic stations in the vicinity of the 12-mile dumpsite and Hudson Shelf Valley revealed the following information (from Ingham *et al.*, unpubl. ms.).

Stratification (May)

Vertical sections of temperature, salinity, and density during the cruise show that density stratification existed because of heating of the surface layer (weak stratification) and because of the presence of the Hudson-Raritan estuarine plume (HREP) in the nearshore region (strong stratification). Throughout the study area, thermal stratification produced vertical density differences of about 1.0 sigma-t units between surface and bottom. Superimposed on this weak structure in nearshore waters off northern New Jersey was a structure due to the HREP which produced differences of up to 3.5 sigma-t units between surface and bottom.

Changes in windfield conditions which occurred during the cruise led to vertical mixing and reductions of stratification seen in the second phase of the cruise (17-21 May) as compared to the first phase (13-16 May). Such changes were particularly evident in the shoreward to middle portions of the western-most transects occupied off western Long Island during the two periods, where density stratification disappeared on some stations. Average wind speed during the first occupation (15-16 May) of these sections was about 10.9 kn (scalar average speed), but increased to about 13.6 kn before and during the second occupation (18-19 May). Although this windspeed increase seems slight (1.25 times greater than the first occupation), the resulting increase in mixing energy, which is proportional to the cube of the wind speed, would be considerable (1.95 times greater).

Stratification along the axis of the Hudson Shelf Valley was weak and relatively unchanged during the two occupations of stations there, except for the shoreward end where the HREP was encountered. Changes in the HREP extension dominated the density field at the shoreward end of the transect and obscured any differences which may have occurred because of the changes in the windfield.

Bottom Water Dissolved Oxygen (May)

Even though the cruise was conducted early in the stratification season, some areas of oxygen depletion in bottom water were detected. During the first survey (12-17 May), an area of about 55 km^2 showed dissolved oxygen levels below 80 percent saturation, extending from the northwest corner of the dumpsite toward the New York Harbor entrance, lying under the HREP. About four days later during the second survey (17-20 May), the area of less than 80 percent saturation had grown to about 300 km^2 in two patches mostly lying over the shallow shelf shoreward of the dumpsite, again mostly lying under the HREP. The larger patch of the two lay across the upper end of the Hudson Shelf Valley, but did not extend down the valley.

Hudson-Raritan Estuarine Plume (May)

The HREP, defined here as water with salinity of 31 practical salinity units [(psu), numerically equal to ppt] or less, extended out to the 12-mile dumpsite during the first survey (12-17 May) and to within 1 km of the site during the second survey. Water of less than 31 psu lying in the Long Island nearshore area may not be part of the HREP, but instead part of a low-salinity band which generally stretches from Montauk Point at the eastern end of the island to the apex of New York Bight. Thickness of the HREP ranged generally from 10 to 17 m, with the thickest portions found on the shoreward end of the NJ-NY transect.

Maps of the thickness of the HREP reveal that there was considerable change in the plume in the sampling area from one survey to the next. Although the area thicker than 14 m did not change very much, part of this portion of the plume was up to 10 m thicker (26 m) during the second survey.

During a 24 hr survey of the HREP (20-21 May), it was found to extend at least 80 km southward along the New Jersey coast, from Sandy Hook to south of Barnegat Inlet. Horizontal gradients of sea-surface temperature associated with the HREP were too weak to be detected by satellite infrared sensors, so the full extent of the plume at that time cannot be determined from satellite data.

Hudson Shelf Valley Bottom Water (May)

Two synthetic transects (stations not occupied sequentially) were developed for the Hudson Shelf Valley, one for each survey period. The transects show that the bottom water in the valley remained unchanged in temperature and changed little in salinity (in the shallow end) from the first survey to the second. Steep inclination of the density isopleths toward the bottom found in both survey periods implies down-valley flow of bottom water, especially pronounced in the latter period (17-19 May).

Stratification (August)

This is the period of the year when stratification caused by solar input reaches a maximum, and this is also a time of weak winds, generally from the S-SW. The overall influence of the HREP should be diminished during this time, 4 mo after the spring freshet, tending to reduce the stratifying effect of low-salinity water. The effects of the sun and reduced summer winds can be seen from the contoured values of relatively warm temperatures at the surface. Except for stations close to the Long Island coast, temperatures were generally between 23°C and 24.5°C. The influence of freshwater was not reduced as expected. Except for the southeast corner of the area sampled, salinity values at the surface were below 31 psu, indicating that the HREP was widely distributed across the region.

The low salinity and high temperatures contributed to surface-to- bottom stratification that was generally 3.5 sigma-t units over the continental shelf and 4.5 sigma-t units over the Hudson Shelf Valley.

In general, the thermocline varied between 5 m and 10 m of depth throughout the area, with the shallowest values in the northern and western portions near the coast.

Bottom Water Dissolved Oxygen (August)

There was no severe depletion of oxygen observed within the inner New York Bight during August 1987. One station near the southwest corner of the dumpsite had a value of 60 percent saturation at the bottom. This generally corresponded to the area of lowered oxygen saturation observed during the second hydrographic survey in May. Changes in oxygen percent saturation between May and August were not consistent in the area, indicating that there were no trends toward severe oxygen depletion.

Hudson-Raritan Estuarine Plume (August)

A wide distribution of low values of salinity (less than 31 psu) indicated that the Hudson-Raritan outflow was widely spread across the region at the surface. Because processes at the oceanic/atmospheric boundary tend to diffuse properties, the contoured values of salinity at 10 m of depth were examined to remove influence of some of the near-surface processes such as wind and rain. Water with salinities characteristic of the plume was found at this depth over a more limited portion of the area in a band parallel to the New Jersey coast. This band extended across the southern portion of the 12-mile dumpsite area, but was generally found west of the Hudson Shelf Valley. The southern extent of the plume was not determined during the cruise.

The layer thickness of the HREP was generally between 6 and 10 m. There were two areas in which the plume was greater than 12 m thick, one to the west of the 12-mile dumpsite and the other at the southwestern corner of the area sampled near the New Jersey coast. Caution should be used in interpreting the low-salinity layer just south of Long Island as the HREP, since another potential source is available as discussed earlier.

Hudson Shelf Valley Bottom Water (August)

The section along the Hudson Shelf Valley created from the stations sampled during the cruise gave a consistent pattern, indicating that there were no large changes during the period of the cruise. Below the rim of the valley the temperature was between 8° and 10°C and the salinity between 32.25 and 32.75 psu. The density field was generally horizontal at greater than 25 sigma-t units, implying little flow in the bottom water for this period.

SEDIMENT RESUSPENSION

One of the essential components in determining the extent to which the area responds to cessation of dumping is an assessment of the fate of sludge-contaminated sediments. Through collaboration with EPA's Environmental Research Laboratory in Narragansett, Rhode Island, cruises are conducted to survey transects of the inner bight, the Hudson Shelf Valley (HSV) and Hudson Canyon (Appendix A). Three variables measured include: (1) erodibility of surficial sediments (resuspension with respect to shear); (2) bottom shear with respect to time (estimated from current-meter records); and (3) chemistry and microbiology of surficial sediments.

Results and Discussion

Recent efforts are focusing on three issues: (1) sediment erodibility and chemistry prior to and following cessation of sludge disposal; (2) a final expression of data from erosion studies; and (3) the estimation of temporal shear from current-meter records.

Results from the 1987 *Albatross IV* cruise, 7 mo prior to cessation of dumping (dumping reduced 30 percent), showed a reservoir of fine sediment along the Hudson Shelf Valley-Christiaensen Basin transect. The location of peak sediment erodibility was at station 39 (upper HSV), decreasing both to the northwest (Christiaensen Basin) and to the southeast (Hudson Canyon entrance) (Fig. 8a,b). An earlier survey of *Clostridium perfringens* spore data (Cabelli and Pedersen 1982) and limited PCB data for sediment and infaunal (primarily polychaete) tissues for 1987[4] are illustrated in Figure 8c. Thus, while the potential sediment erodiblity increases to a peak 20 km southeast of NY6 (located between stations 71 and 83), the sewage sludge content of these sediments, measured as spore counts, is less than half of the sediments from NY6.

Results of analyses of metals [copper (Cu), chromium (Cr), nickel (Ni), lead (Pb), and zinc (Zn)] from the Hudson Shelf Valley (May 1987) indicate that concentrations decrease approximately exponentially with distance down the valley (Fig. 9). There is no statistically significant departure from linearity for Log_{10} transformed Ni concentrations as a function of distance. However, for the other metals, quadratic effects are statistically significant($P = 0.05$).

Coprostanol concentrations in the range of 12 to 27 µg/g at Station 39 (the site with the most highly erodible sediments) indicate fecal origin for a significant fraction of this material (Fig. 10).

Preliminary erodibility data measured 6 mo after cessation of dumping indicate changes in the sediments of the Hudson Shelf Valley transect (Fig. 11). Recent erodibility measurements (dashed lines) indicate an approximate 70-90 percent reduction of erodible fines in the lower Christiaensen Basin and upper Hudson Shelf Valley (10-30 km) while the mid-valley area shows a corresponding increase in erodible material. It is not clear if this represents down-valley transport and deposition. The chemistry and spore analyses of these sediments in conjunction with current-meter data analysis will help the interpretation.

[4] **Personal communication from James Lake (USEPA, Environmental Research Laboratory, Narragansett, RI 02882).**

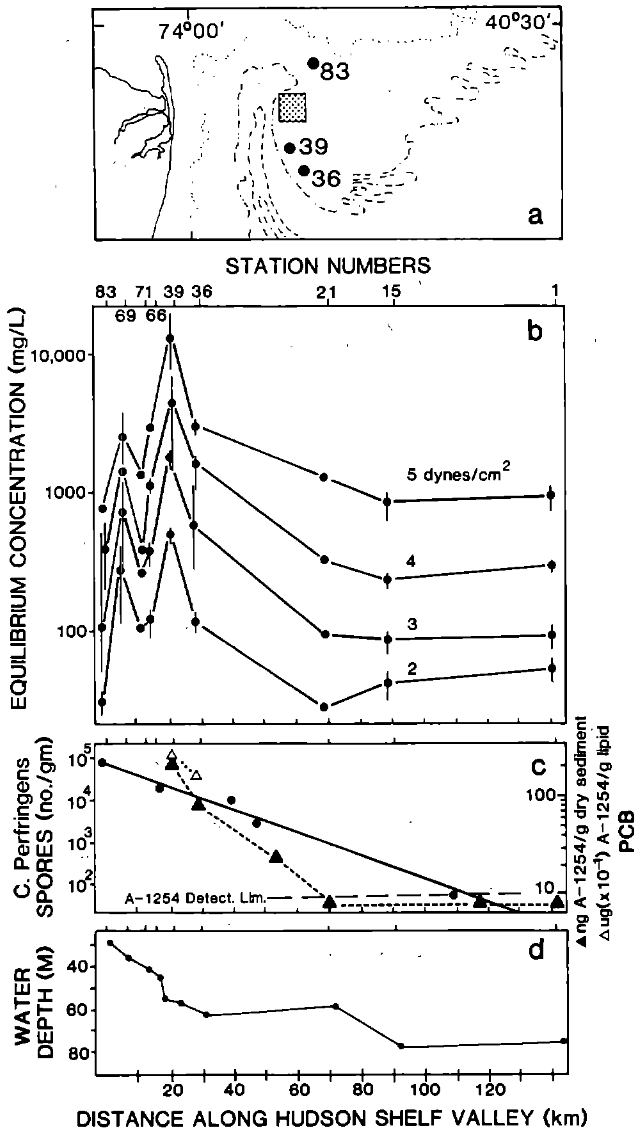

Figure 8. (a) Station locations; (b) resuspension potential for sediments along the Hudson Shelf Valley axis at several levels of resuspension; (c) distribution of *Clostridium perfringens* spores (1982) and PCBs in sediments and infaunal tissues from 1987 HSV surveys; and (d) water depth.

To address questions about resuspension and transport in the vicinity of the Christiaensen Basin and along the axis of the Hudson Shelf Valley, it will be necessary to use the Particle Entrainment Simulation (PES) data base in conjunction with current-meter records. The intention is to use a shear-resuspension rate relationship with current-meter data to estimate both entrainment rate as a function of time and direction of transport. A first approximation to estimate the mass entrained is to assume that total mass entrained is linearly related to the duration at a given shear. However, there is strong evidence of diminishing returns. A vertically homogeneous (grain size and water content) core from Station 39 was tested and retested four times. The suspended particulate phase was discarded between tests, and fresh sea water was added. Suspended solids concentrations measured at steady state were observed to decrease on successive runs of the experiment. Thus, there appears to be a finite reservoir of fine particles that can be eroded at a given shear. These findings suggest that entrainment rate calculations should incorporate a diminishing returns component.

The sediment chemistry results from the May 1987 cruise are consistent with a near-coastal source of contaminants and subsequent transport and dispersal down-valley, resulting in contaminant concentrations that decrease approximately exponentially as a function of distance down-valley. The highly erodible sediments found at Station 39 where the valley deepens (Fig. 8) were found to have coprostanol concentrations consistent with a significant amount of material of fecal origin. The sediment chemistry samples up-valley of Station 39 collected in March 1987 and June 1988 have not been analyzed. These results will help characterize changes in the sediment chemistry following cessation of sewage sludge dumping.

The PES results indicate that changes in sediment erodibility have occurred in the Hudson Shelf Valley. The changes may be consistent with resuspension and redistribution of contaminated particulates along the axis of the Hudson Shelf Valley, but definite conclusions cannot be made until the sediment chemistry and current-meter data are analyzed.

Long-term changes in the sludge-sediment reservoir will be a complex interaction of various transport processes such as particle flux and in-sediment mixing (bioturbation). These processes are illustrated and simplistically formulated in Figure 12. The key process categories are: (1) sediment-water particle flux of the thin interface sediment (entrainment and deposition); (2) vertical transport of particles (exchange of deeper sediment with the above thin flux zone); (3) deposition pit formation (rapid deep deposition of sediment); and (4) solute exchange (between pore and overlying water) including oxygen and sulfate.

SEDIMENT TRANSPORT

One difficulty, identified in the design of the study (Environmental Processes Division 1988) pertained to the separation of effects of pollutants from various sources. It has been estimated that only a small fraction (an average of 16 percent) of the major contaminant loadings of the New York Bight apex resulted from sewage sludge dumping; that the majority was contributed by the Hudson-Raritan plume and dredged material dumpsite west of the Christiaensen Basin (Fig. 1; Stanford and Young 1988). To help identify the sources of pollution to the bight, investigators from Lamont-Doherty are conducting collaborative studies with EPD to:

1. Provide net sediment and pollutant accumulation rates at depositional sites in the study area (Fig. 13).
   NOAA studies indicate that fine-grained sediment depo-

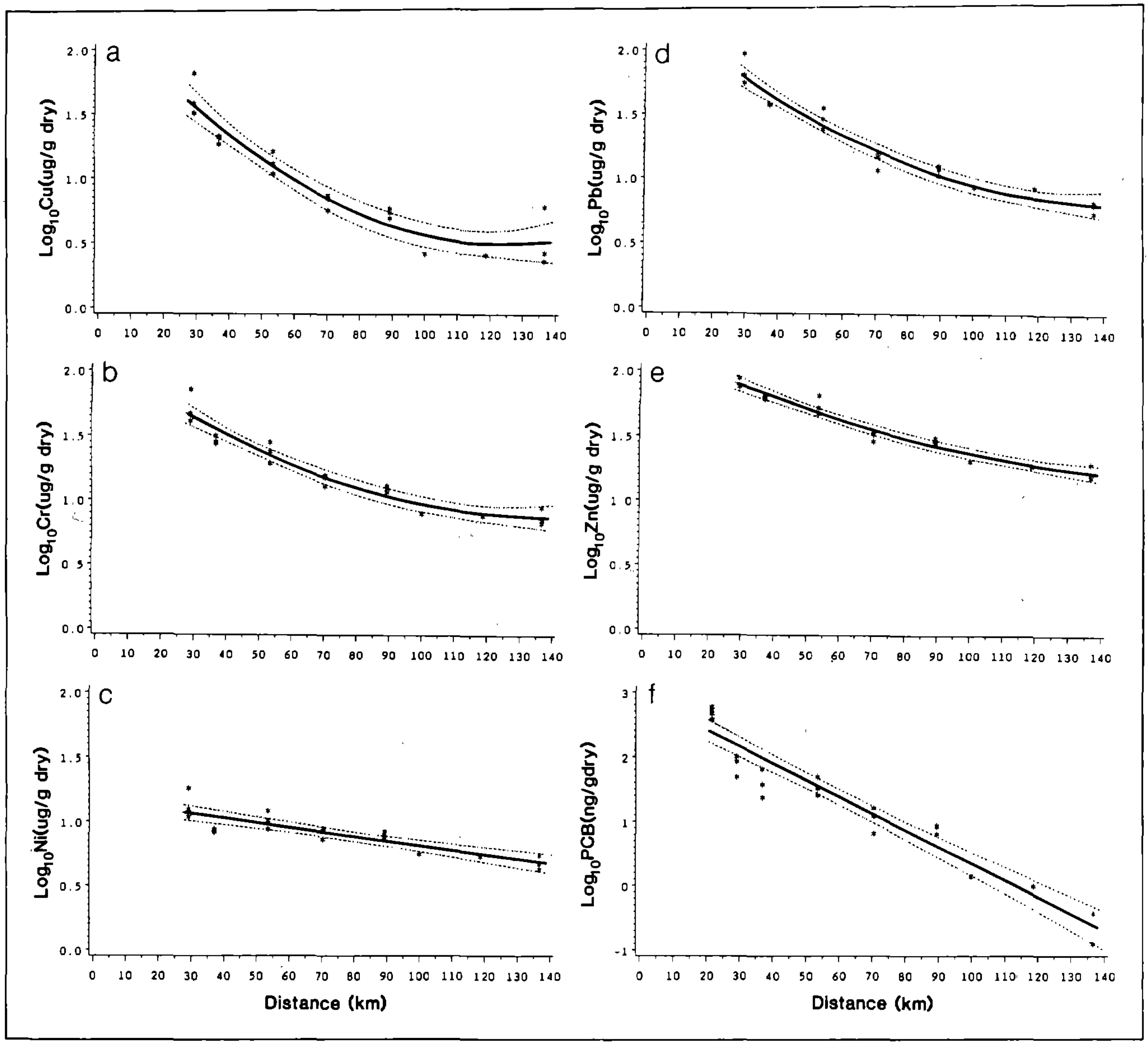

Figure 9. Concentrations of heavy metals (a - Cu; b - Cr; c - Ni; d - Pb; e - Zn) and PCBs (f) from Hudson Shelf Valley sediments collected 7 mo prior to cessation of dumping at the 12-mile dumpsite. Ninety-five percent confidence intervals are illustrated by dashed lines; see Fig. 8 for distance reference.

sition in this region occurs primarily in the Hudson Shelf Valley. Radionuclide analysis of sediment core sections will be used to determine net rates of sediment accumulation. A fallout radionuclide, Cesium-137, allows us to distinguish post-1954 deposition, and a short-lived natural radionuclide, Beryllium-7, unambiguously identifies sediment deposited within about a year of collection.

Samples are analyzed for chlorinated hydrocarbons including PCBs, chlordane, and DDT-derived compounds. Archived subsamples will be suitable for analysis of other persistent organic contaminants and trace metals. Our goal is to assess the importance of shelf valley depositional areas as ultimate reservoirs of persistent pollutants in the nearshore environment, particularly those derived from dredge spoil and sewage sludge dumping.

2. Provide direct measurements of chlorinated hydrocarbons actively being transported in the water column.

To our knowledge, there are no reported measurements of persistent pollutants on suspended particles in this area. A main reason for this is the difficulty of collecting large (gram-sized) samples of suspended matter. This problem has been solved through the use of a unique large-column, *in situ* pump and filtering apparatus. Samples collected during 1987 are described below. They will be analyzed and archived in the same manner as sediment samples.

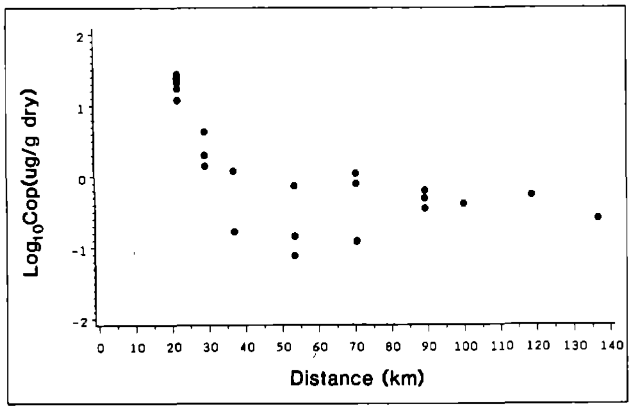

Figure 10. Concentrations of coprostanol from Hudson Shelf Valley sediments collected 7 mo prior to cessation of dumping; see Fig. 8 for distance reference.

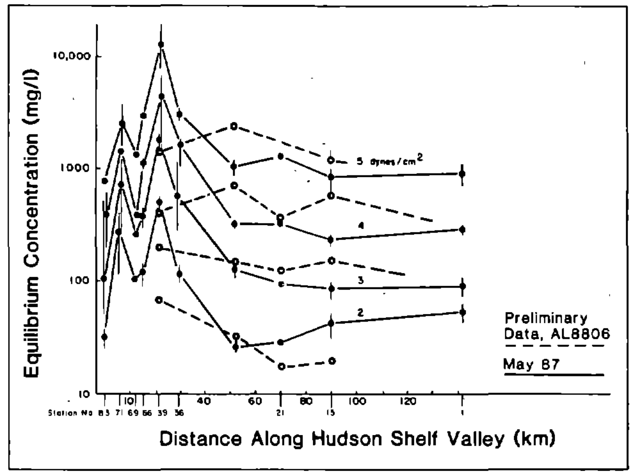

Figure 11. Resuspension potential for sediments along the Hudson Shelf Valley axis at several levels of resuspension. Stress measured 7 mo prior to and 6 mo after (dashed line) cessation of dumping.

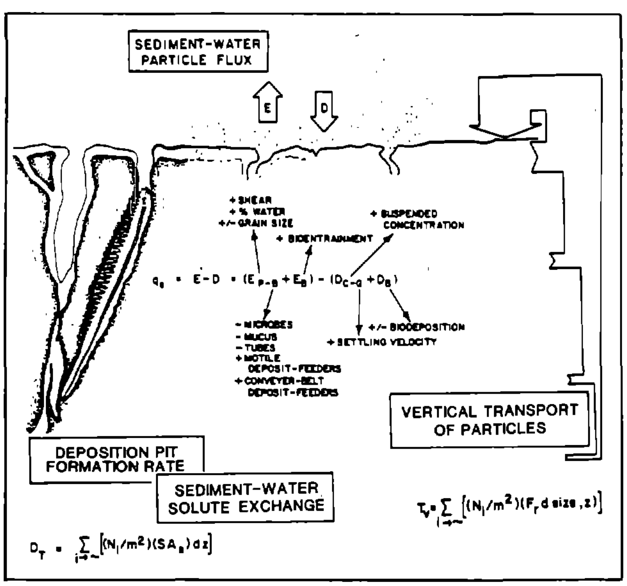

Figure 12. Transport processes affecting the sludge-sediment reservoir.

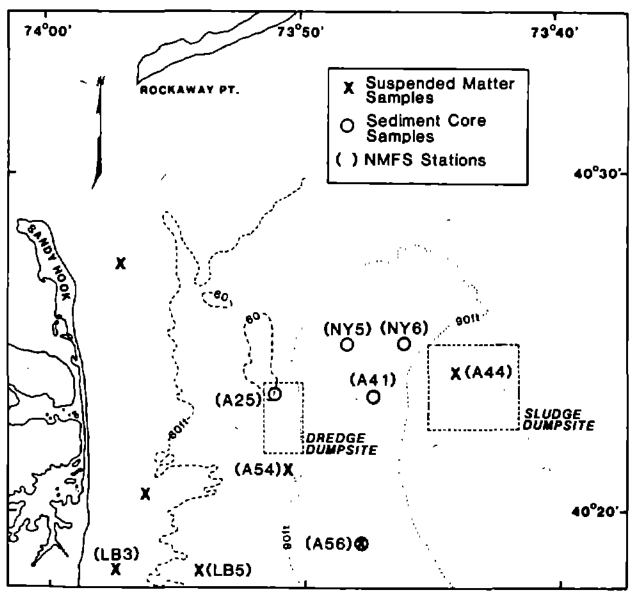

Figure 13. Sampling locations occupied in 1987 for suspended matter and long core samples. The Hudson Shelf Valley lies within the 90-ft (30-m) contours. NOAA stations are shown in parentheses. X = suspended matter; 0 = sediment core samples.

Table 2. DDT-to-DDD ratios in particle samples

Sample	pp 'DDT/pp 'DDD	# of Samples
Sediment samples (characteristic of dredge spoils)		
New York Harbor sediments	<0.02	30
Newark Bay cores	0.07	14
Suspended particle samples (characteristic of Hudson/Raritan discharge)		
Raritan Bay (GFF 1055, 1981)[1]	0.63	
New York Harbor (GFF 1060, 1981)	0.22	
Arthur Kill (GFF 1057, 1981)	0.19	
Newark Bay (F 1049, 1985)	0.22	
Sandy Hook (KN88, sediment trap)	0.77	

[1] **Lamont Control No.**

3. Interpret chlorinated hydrocarbon signatures on suspended particles as tracers of pollutant sources.

   This technique has been used quite successfully in the Hudson River and in Newark Bay. One particular problem faced by this study will be distinguishing Hudson-Raritan estuarine-borne particles and pollutants from those derived from resuspension of dredge spoils. The problem arises from their common origin -- the dredged material disposed at the nearshore dumpsite consists primarily of Hudson-Raritan particles that had sedimented in shipping channels of New York Harbor, Newark Bay, Raritan Bay, and adjacent systems. We will employ a unique tracer that we believe will allow us to distinguish between these two sources. That tracer is the ratio of pp'-DDT to pp'-DDD. In anaerobic sediments, much of the DDT present is dechlorinated to DDD. Thus, dredge spoils would be expected to have a lower DDT-to-DDD ratio than Hudson-Raritan suspended matter. Our past measurements indicate that this is indeed the case. We observed a ratio of about 0.4 on Hudson-Raritan suspended matter and about an order of magnitude lower on average bottom sediments of these systems, which is characteristic of dredge spoils (Table 2).

4. Provide preliminary information on the distribution and transport of dioxin in the study area.

   As demonstrated by our recent work with the New Jersey Department of Environmental Protection, recent sediments from the lower Passaic River and Newark Bay are significantly contaminated with dioxin. Dioxin may be transported to the study area both with dredge spoil disposal and via Hudson-Raritan estuary discharge. Thus far, monitoring of dioxin in the area has been limited primarily to biological samples with crustaceans collected at the mud dumpsite having notable concentrations. Our study, through a subcontract to Dr. Michael Gross of the University of Nebraska, will provide the first significant data on the accumulation of dioxin in sediments of the New York Bight and its transport on particles.

Results and Discussion

Most effort to date has been directed toward sample collection and some preliminary analyses. Sampling sites were chosen to: (1) study the Hudson-Raritan discharge; (2) follow the plume along the New Jersey shore; (3) investigate resuspension from the dumpsites; and (4) where possible, duplicate NOAA study sites (Fig. 13).

The goal of our sediment coring was to locate areas of fine-grained sediment accumulation, quantify net accumulation rates, determine levels of associated pollutants and ultimately infer rates of particle and associated pollutant transport from the dumpsites. Our core sampling (Fig. 13) was guided by NOAA data which show surficial fine-grained sediments concentrated along the axis of the Hudson Shelf Valley and not at the dumpsites. Preliminary analysis of the core at station A56 (Fig. 13) yielded net accumulation rates on the order of 1 cm yr^{-1}. This rapid sedimentation suggests significant recent particle transport to this site, perhaps from the dumpsites. We hope to support this hypothesis via contaminant analyses of the sediments. Earlier this year, additional cores were collected at three sites up to 5 mi further down the axis of the shelf channel. The only available preliminary data, from the furthest down-channel core, also indicates quite rapid net sediment accumulation (about 1 cm yr^{-1}). For example, results from NY6, the most degraded area, indicated it was one of the

Table 3. Nutrient concentrations (µM) measured in near-bottom water from selected 12-mile dumpsite stations (n = number of samples)

Nutrient		Stations NY6	R2	NY5	NY11
	n	74	37	24	29
PO_4^{-3}	min	.252	.292	.233	.285
	max	29.120	2.868	1.556	1.429
	$\bar{x}$	1.353	.765	.678	.591
	SD	3.567	.486	.289	.256
	n	55	30	22	24
NH_4^+	min	.323	.566	.542	.291
	max	8.895	10.059	13.766	6.365
	$\bar{x}$	2.643	3.569	3.638	2.833
	SD	1.632	1.678	3.547	1.556
	n	74	37	24	29
NO_2^-	min	.060	.098	.078	.069
	max	.822	.699	1.268	.718
	$\bar{x}$	.291	.321	.272	.298
	SD	.164	.177	.230	.159
	n	74	37	24	29
NO_3^-	min	.083	.097	.261	.069
	max	3.786	4.561	4.778	1.906
	$\bar{x}$	.985	1.142	1.321	.807
	SD	.803	.978	1.193	.496
	n	74	37	24	29
SiO^{-4}	min	1.212	1.417	1.044	1.303
	max	9.581	14.313	10.370	14.044
	$\bar{x}$	4.657	4.923	4.476	4.582
	SD	2.574	3.129	2.467	2.573

less depositional sites sampled; beryllium, indicative of sedimentation over about the past year was not detected; Cesium-137 dating indicated about 14 cm of sediment deposition since 1954. In contrast, A56 apparently had 2 cm deposited over the last year and greater than 42 cm since 1954. Surface sediments from these sites will quickly reflect any changes in pollutant accumulation caused by the recent cessation of sewage sludge dumping in the study area.

WATER AND SEDIMENT CHEMISTRY

Significant changes in sediment and water chemistry are expected to be linked closely to changes in the input of sewage sludge. One major component of the study is the measurement of a wide range of chemical variables (Environmental Processes Division 1988). The present report will focus on three variables: (1) water column nutrients; (2) redox potential as an indicator of sediment biogeochemistry; and (3) heavy metal concentrations in sediments.

Water Column Nutrients

Bottom-water samples have been collected from four stations, NY6, R2, NY5, and NY11, since July 1986 and analyses have been made through February 1988 for ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), phosphate (PO_4^{-3}), and reactive silicate (SiO_4^{-4}).

Results and Discussion

The greatest variability in phosphate (PO_4^{-3}) concentrations, as shown by the standard deviation, occurred at the most degraded station, NY6 (Table 3). Along with the mean and maximum PO_4^{-3} concentration, the decrease in variability parallels the station order of presumed most contaminated to the least, *i.e.*, NY6>R2>NY5>NY11. Except for five samples, PO_4^{-3} concentrations averaged 0.69 µM with a maximum of 1.66 µM and a minimum of 0.23 µM. Of the five exceptions, 2.87 µM was found at station R2 in October 1986 (Fig. 14) and the rest occurred at NY6 (2.72

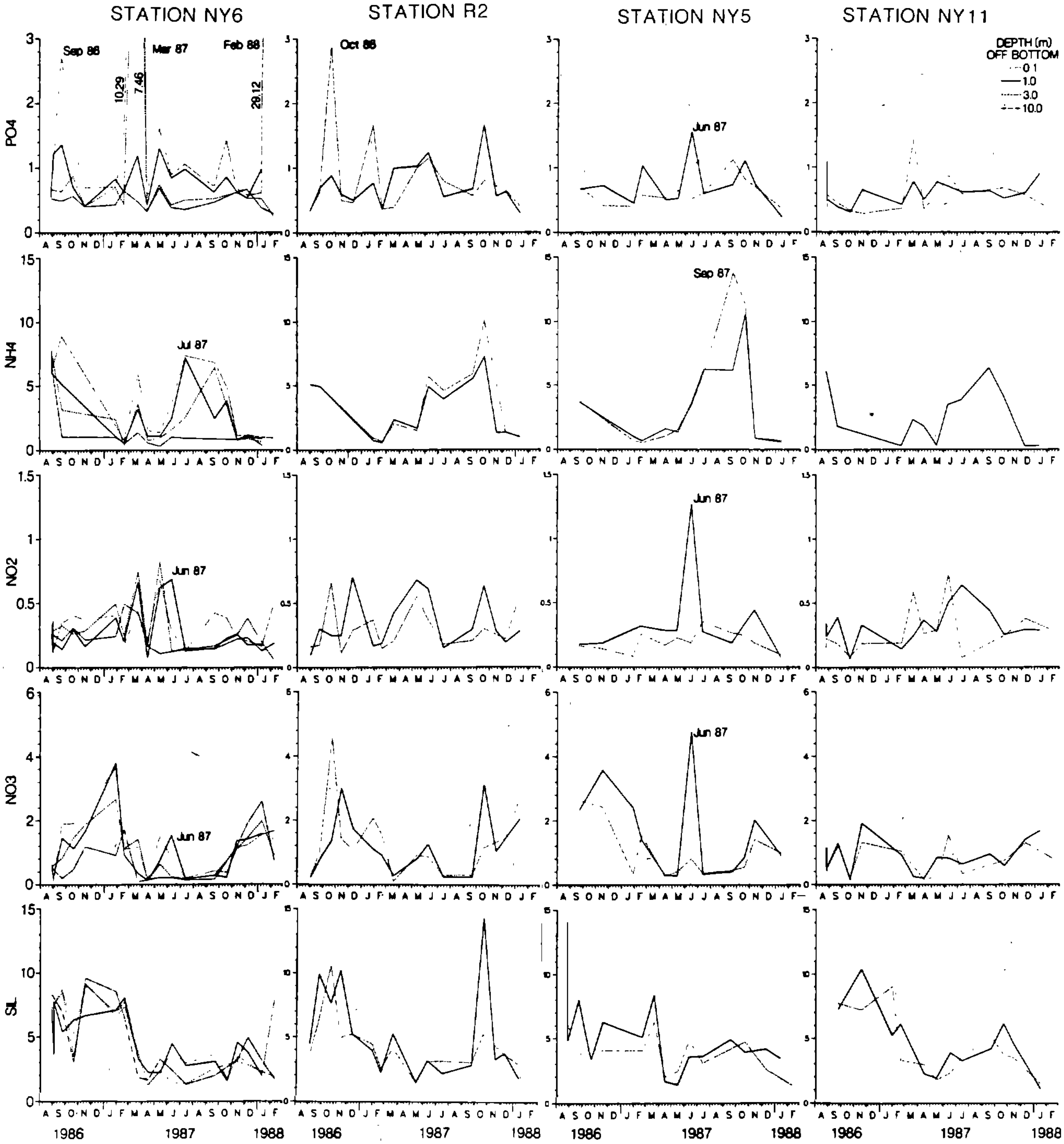

Figure 14. Nutrient concentrations in bottom water -- August 1986 to February 1988.

μM in September 1986, 10.29 μM and 7.46 μM in March 1987, and 29.12 μM in February 1988) (Fig. 14). These values range from 2 to 20 times the average. These unusually high values are probably related to certain sediment biogeochemical processes which will be understood better when other analyses are completed and data become available.

Concentrations of ammonium (NH_4^+) ranged from a maximum of 13.77 μM at NY5 in September 1987 to a minimum of 0.3 μM at a number of stations in winter and early spring (Table 3; Fig. 14). The seasonal maximum appears to have occurred earlier (July) at NY6 than at other stations, perhaps indicating an association with the level of sediment microbial activity; however, the maximum concentration of NH_4^+ at NY6 (2.64 μM) was less than at NY11 (2.83 μM), R2 (3.57 μM), or NY5 (3.64 μM).

Variability in concentrations of nitrate (NO_3^-) ranged from about 1.5 to 2.5 times lower at NY11 (the station furthest from the dumpsite) than at the other stations (Table 3). In June 1987, maximum concentrations of NO_3^- and NO_2^- and high values of PO_4^{-3} were observed at NY5; during the same period, peaks in NO_2^- and NO_3^- concentrations were recorded at NY6 (Fig. 14). These observations coincided with two unexplained high values for total plankton respiration rates in June 1987 at Stations 33 (NY6) and 32 at the north boundary of the designated dumpsite (see page 20 -- Sediment Metabolism).

Water-mass movement, particularly periodic flow up and down the Hudson Shelf Valley, may be a more important factor in seasonal changes in NO_3^- and NH_4^+ concentrations than effects from sludge dumping. Of the four stations sampled, the depth at NY5 (34.5 m) is about 15 percent greater than depths at NY6 (31 m), NY11 (30 m), or R2 (29 m). Since movement of the bottom-water mass due to wind stress would have the greatest influence on the deepest station, this could explain why the highest mean concentrations of NO_3^- and NH_4^+ are found at NY5.

One point of similarity among stations for silicate (SiO_4^{-4}) concentrations occurred in April-May 1987, when minima were observed (Fig. 14). By evaluating other variables when available, it may be possible to learn whether a diatom bloom at this time could have depleted silicate in the water. Although information concerning silicate concentrations may not be important by itself, an understanding of the dynamics of the system at the 12-mile site will provide a baseline for the interpretation of unusual events, *i.e.*, the exceptional values of PO_4^{-3} concentrations at station NY6 mentioned above.

In summary: (1) variability, average concentration, and maximum concentration of PO_4^{-3} were found to parallel the station order from the presumed most contaminated to the least; (2) the seasonal maximum for NH_4^+ occurred earlier at station NY6 than at the less-polluted stations; and (3) peaks observed in NO_2^-, NO_3^-, and PO_4^{-3} concentrations in June 1987 (Fig. 14) coincided with high values for total plankton respiration rates at Stations 33 and 32.

Sediment Biogeochemistry

A key chemical response variable in the sediments, which will provide insight on the relationship between sediment biogeochemistry and benthic community structure, is the oxidation-reduction (redox) potential.

Redox potential (measured in samples as E_h) is taken in the sense summarized by Whitfield (1969). It functions as an index that can be compared with discrete chemical measurements and animal community composition over the period of dumping and during ecosystem response. E_h is an extensive variable that indicates the potential of the chemical compounds present in a matrix to give up or take up electrons. It is determined by the sum of the electron affinities or contributions of the substances present (dissolved oxygen, ferrous and ferric iron, manganese, nitrate, nitrite, sulfide, carbon dioxide, hydrogen, etc.), but their effect on the measurement is a function of the concentration. Redox potential is a theoretical measure of the electron demand or supply of a matrix, while E_h is the potential actually measured in a sample which includes junction potentials or effects of poisons, and is reported relative to the standard hydrogen electrode.

Values were measured at NY6, R2, and NY11 before and during cessation of dumping (Environmental Processes Division 1988). Only values for the 0.5 cm depth in the sediment are presented since that depth reflects the annual variability at NY6 (in the area of sewage sludge accumulation) and because 0.5 cm is the depth for which a computer model has been run. Measurements at other depths will be considered in future analyses.

Results and Discussion

With few exceptions, sediment E_h was lowest at station NY6 and highest at NY11, with values at R2 intermediate (Fig. 15). The lowest values for NY6 occurred in the summers of 1983-85, prior to the cessation of dumping, when E_h was less than 0 mV for several months. In the summers of 1986 and 1987, E_h was less than 0 mV during only 1 mo. The highest value thus far observed at NY6 was measured in February 1988.

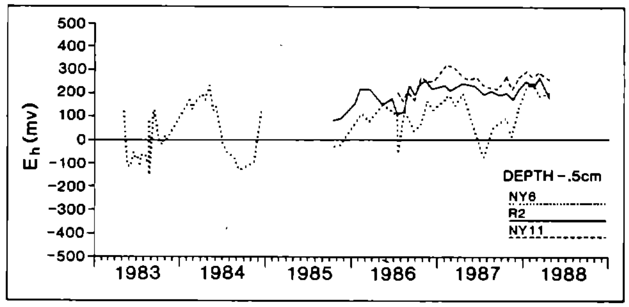

Figure 15. Mean redox potential (E_h) values for stations NY6, R2, and NY11 recorded at a depth of 0.5 cm. The mean is based on 12 E_h profiles (3 grabs x 4 profiles).

The lowest value at R2 was recorded in October 1985 with the second lowest value occurring in the summer of 1986 (Fig. 15). The highest value was observed in March 1988. At NY11, the lowest value was recorded in the summer of 1986 and the highest in January 1987. The amplitude of the seasonal cycle in E_h at both NY6 and R2 appears to have diminished after a 30 percent reduction in dumping in June 1986.

There is the suggestion of a change in the sediment chemistry at station R2 as well. As with NY6, the highest E_h value at R2 was in the winter of 1988 (Fig. 15). The lowest value available for R2 was recorded in October 1985. Summer minima were higher in 1986 and 1987. If this 1985 value represents an upper limit for precessation summer potential at 0.5 cm at R2, E_h observations at this station also reflect a release from the intense reducing conditions caused by the addition to the sediment of labile carbon from sewage sludge dumping, as at NY6.

The large apparent change at station NY6 and the smaller change at R2 in conjunction with no change in the consistently more oxidizing sediment at NY11, suggest that the E_h values at the three stations are converging, over time, toward the values currently observed at NY11.

Sediment Metals

Samples collected at the three replicate stations (NY6, R2, and NY11) in November 1986, November 1987, and May-July 1988, following reductions in dumping levels by 30 percent in June 1986, 60 percent in August 1987, and 100 percent in December 1987 (Fig. 3), were analyzed to determine whether a 30-60% decrease in sludge disposal volume resulted in changes in sediment metal concentrations.

Data from samples collected in August 1982 (Northeast Monitoring Program, New York Bight Survey DL 82-06, unpublished) are included for comparison and, for this initial estimate, are considered to represent maximal dumping levels.

Two sediment strata were investigated, based on the August 1982 results; surface (0-1 cm layer) sediment and buried (4-5 cm layer) sediment, hereafter referred to as layers 1 and 5, respectively. Surface samples provide data on recent inputs of metals at each station and, in conjunction with buried sediment samples, provide information on the vertical distribution of metals in the sediment column. Vertical chemical distributions provide information on the depositional history of the sediment. Chromium (Cr), copper (Cu), nickel (Ni), lead (Pb), zinc (Zn) and iron (Fe) were determined. The first five metals are contaminants commonly associated with sewage sludge, while iron is a natural sediment component generally indicative of sediment texture.

Principal components analysis and ANOVA (for α>0.05) were used to compare data from six strata (three stations, two depths). Data from August 1982 were not included in the statistical analysis, but are provided for qualitative comparison.

Results and Discussion

Distributions of the contaminant metals, chromium (Cr), copper (Cu), nickel (Ni), lead (Pb), and zinc (Zn), were similar to each other, but different from the distribution of iron (Fe) (Fig. 16). This is consistent with the correlations among these six metals. The contaminants group were highly intercorrelated (N = 81, R $\geq$0.90), but none was highly correlated with iron (N = 81, R $\leq$0.40). In general, highest levels of contaminant metals were found in layer 1 (surface) sediments from NY6 in November 1986 and 1987. By May-July 1988, levels in this layer had decreased. Concentrations in layer 5 samples from NY6 and R2 were next highest and were intermediate in magnitude. Lowest concentrations were found in layer 1 sediments from R2 and layers 1 and 5 sediments from NY11.

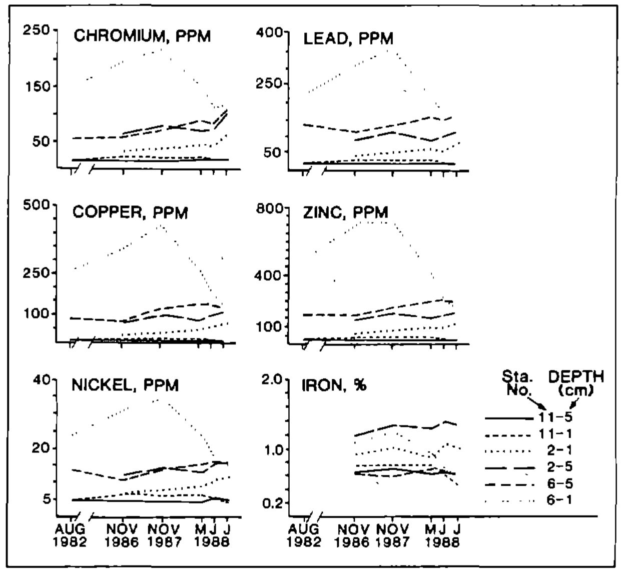

Figure 16. Mean concentrations of metals in two sediment strata (1- and 5-cm depths) at NY6, R2, and NY11. Code: station-layer (*e.g.*, 6-5 = NY6 and 5-cm depth).

Iron levels varied independently of the other metals, highest levels occurring in buried (layer 5) sediment at R2, and in surface (layer 1) sediment at NY6 in November 1986 and 1987. By May-July 1988, NY6 levels had decreased. Surface (layer 1) sediment at R2 was intermediate in iron content, while lowest iron levels were found in sediments from layer 5 at NY6 and layers 1 and 5 at NY11.

Results of principal-components analysis are consistent with the observations noted above (Fig. 17). Factor 1, the X-axis (90 percent of data variance), is composed of the contaminant metals Cr, Cu, Ni, Pb, and Zn, while factor 2, the Y-axis (nine percent), is composed of iron (a "sediment matrix" element), whose concentration is independent of the levels of the other metals. The relationship between contaminants and matrix can be considered representative of natural conditions at NY11 and R2, compared to NY6. At NY6, sediments contain far more contaminants than expected from the levels of iron (matrix element) present.

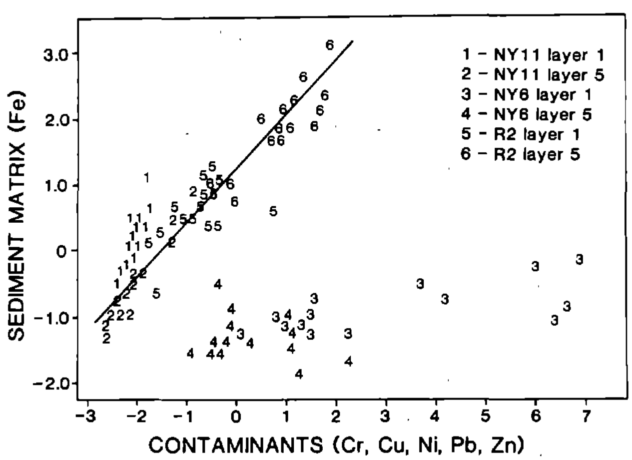

Figure 17. Principal-components analysis; x axis = contaminants; y axis = sediment matrix.

Layers 1 and 5 from NY11 and layer 1 from R2 were similar with respect to levels of all the metals (both axes), while layer 5 from R2 contained higher levels of both contaminant and matrix metals, although in proportions consistent with other NY11 and R2 samples.

At R2, layer 5 contaminant levels resembled those of NY6 layer 5 sediments, as evidenced by their similar positions along the contaminant axis.

At NY6, layer 5 samples can be considered either moderately contaminated or relatively depleted in iron, while layer 1 samples can be considered either highly contaminated or greatly depleted in iron.

Concentrations of zinc and iron in surface (1-cm) and buried (5-cm) sediment layers at individual stations are compared in Fig. 18 a,b. The zinc distribution is generally representative of the contaminant metals. Certain features are common to both plots:

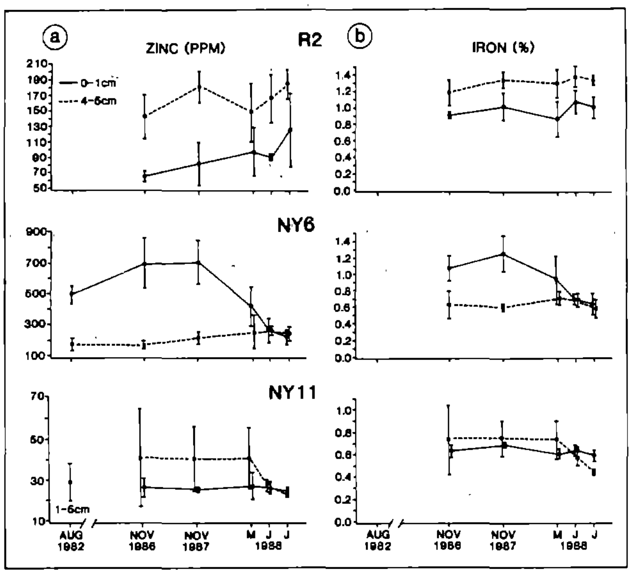

Figure 18. Mean (n = 3) concentrations (±1 SD) of: (a) zinc, and (b) iron in layers 1 and 5 at replicate stations R2, NY6, and NY11; x axis not to scale.

1. At NY11, metal levels in layers 1 and 5 were indistinguishable at all sampling times except July 1988 (iron). Also, in each individual layer, levels did not differ significantly over time.
2. At R2, contaminant metal levels were significantly higher in layer 5 than in layer 1 in 1986, 1987, and June 1988. In May and July 1988, layer 5 levels were higher but this difference was not statistically significant. Iron levels in layer 5 appear significantly greater on all sampling dates. The magnitudes of these differences were considerable for the contaminant metals, but small for iron. As with NY11, metal levels in individual layers did not differ significantly over time.
3. At NY6, metal levels were significantly higher in layer 1 than in layer 5 in 1986 and 1987, but not in 1988. Layer 5 metal levels did not differ significantly over time, but layer 1 concentrations decreased significantly after November 1987. By May 1988, concentrations in the two strata were indistinguishable.

ANOVA was used to investigate possible statistically significant differences in metal concentrations in sediment layers at each station. At the 95-percent confidence level using the Duncan Multiple Range test, the results were very similar to those described by Figure 18 a,b (Table 4).

The magnitude and statistical significance of the decrease in sediment contaminant metal levels since the complete cessation of sludge dumping, as evidenced by changes in contaminant metal concentrations in NY6 surface sediments, are the most important aspects of these preliminary results. Although the number of observations (cruises) examined to date is small, it appears that 1986-87 metal concentrations at NY6 were comparable to those found in 1982, but that 1988 levels were significantly lower. In addition, it appears that a 30-60 percent reduction in sludge volume was not detectable as a change in sediment metal concentrations at NY6 during this period, but a 100 percent reduction was. Also, since dumping ended in December 1987, it appears to have taken approximately 6 mo after complete cessation for this difference to be manifested. Finally, because of the small number of observations and because maximal contaminant concentrations were found in the fall while minimal concentrations were found in the spring, seasonality in sediment metal concentrations in this system must be investigated.

Of similar interest are findings at R2 and NY11. Concentrations of sediment metals at NY11 have been relatively constant, and low, in all samples analyzed, including 1982 samples. In contrast, R2 metal levels are variable and appear to have increased (although not significantly) after complete cessation, levels in layer 5 sediment approaching post-dumping levels at NY6. It is possible that R2 is influenced by NY6 (after some as-yet undetermined time lag) and that the northern Christiaensen Basin is very dynamic in nature, being heavily influenced by estuarine processes as well as by activities at the dredged spoil dumpsite. Analyses of additional samples are expected to provide information related to these and other questions.

SEDIMENT METABOLISM

As described previously (Environmental Processes Division 1987), seabed and water-column oxygen demand integrate a variety of oxygen-consuming processes. Consumption of oxygen by the seabed and water column has been used as a measure of benthic community metabolism to understand energy flow and carbon cycling in marine ecosystems. Oxygen consumption processes, coupled with the physical dynamics of the system, have the ability to cause oxygen depletion, often resulting in hypoxia or anoxia. Seabed oxygen demand measurements are also used to indicate the oxidation of organic matter and the effect of organic pollution on benthic communities. Understanding these rates and processes is important for considerations of use and management of natural resources and protection of the marine ecosystem.

Table 4. Significant differences between metal concentrations in surface (layer 1) and buried (layer 5) sediment at each replicate station, as derived by ANOVA for $\alpha = 0.05$

		Metals					
Station	Date	Cr	Cu	Ni	Pb	Zn	Fe
NY11	Nov 86	--	--	--	--	--	--
	Nov 87	--	--	--	--	--	--
	May 88	--	--	--	--	--	--
	Jun 88	--	--	--	--	--	--
	Jul 88	1	1	--	1	--	1
NY6	Nov 86	1	1	1	1	1	1
	Nov 87	1	1	1	1	1	1
	May 88	--	--	--	--	--	--
	Jun 88	--	--	--	--	--	--
	Jul 88	--	--	--	--	--	--
R2	Nov 86	5	5	5	5	5	5
	Nov 87	5	5	5	5	5	5
	May 88	--	--	--	--	--	--
	Jun 88	5	5	5	5	5	--
	Jul 88	--	--	--	--	--	--

1 = Significant difference, layer 1 concentration greater
5 = Significant difference, layer 5 concentration greater
-- = No significant difference

Aperiodically, between 1974 and 1985, various parameters were sampled at six stations on an east-to-west transect across the northern extent of the 12-mile dumpsite and Christiaensen Basin (Fig. 19). Since 1985, these stations have been occupied monthly. Measurements of seabed oxygen demand (SOD) (Phoel 1983), total plankton respiration (TPR) rates (Robertson 1983), benthic sulfide production, dissolved oxygen concentrations (Robertson 1983), salinity, temperature, and ambient nutrient concentrations have been obtained. Also collected were samples for total and particulate organic carbon, benthic nutrient regeneration, sediment grain size, chlorophyll pigments, and water-column photometry. Seabed oxygen demand rates for each station were averaged by calendar year and season (winter = November-April; summer = May-October) and compared with the annual volume of sludge dumped and seasonal and annual mean temperatures. Upper and lower confidence limits, at the 95-percent level are included about the means.

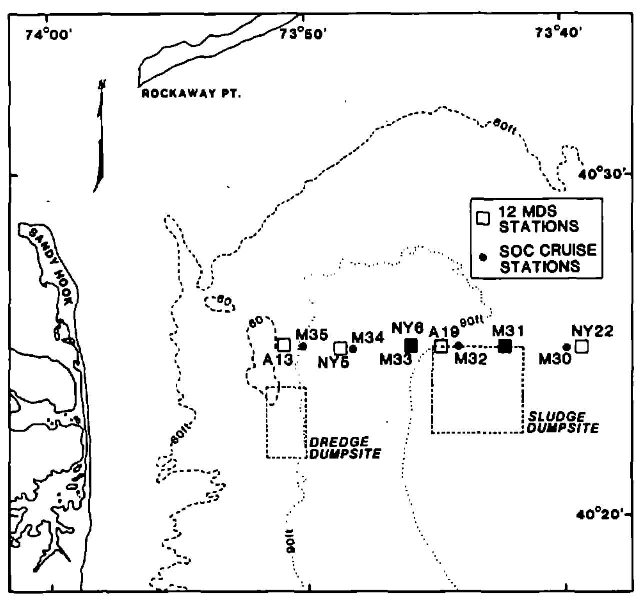

Figure 19. Location of stations occupied for measurement of seabed oxygen consumption (SOC) and other parameters.

Results and Discussion

The mean SOD rates at Station 30, which is located 2.3 km east of the designated dumpsite area, have exhibited no significant response to changes in the quantity of sludge dumped (Fig. 20 a). The data suggested that during the early months of 1987, after the 30 percent decrease in sludge dumping (Fig. 3), the SOD rates were decreasing. Subsequent measurements, however, indicate a return to earlier values with SOD rates ranging between 12 and 18 ml $O_2 m^{-2} h^{-1}$ ($\bar{x} = 15$). This range is close to values measured at other stations with similar sandy sediments under coastal influence, both within and outside the New York Bight.

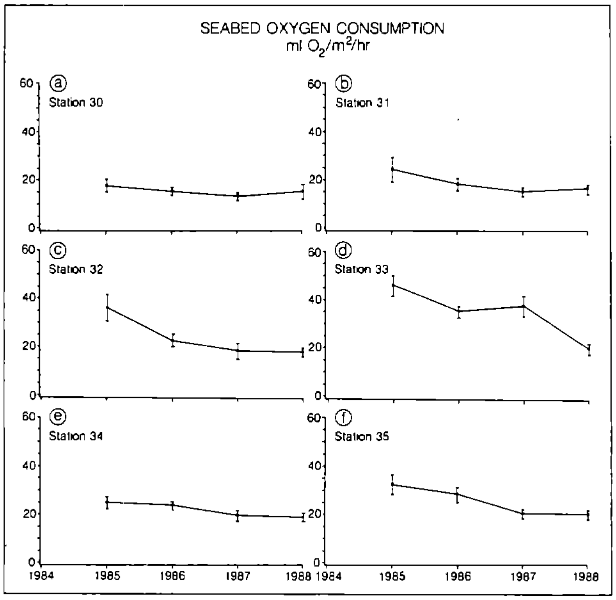

Figure 20. Seasonal and annual fluctuations in seabed oxygen demand at: (a) Station 30; (b) Station 31; (c) Station 32; (d) Station 33; (e) Station 34; and (f) Station 35. Mean and ± 1 SD indicated.

Station 31 has sediments similar to Station 30, but is 0.5 km inside the dumpsite's eastern boundary and therefore has received relatively more input of sludge. There is a suggestion that SOD rates declined significantly between 1985 and 1987 and remained at that level through the summer of 1988 (Fig. 20b). The relatively large decrease in SOD rates apparent between 1985 and 1986 is probably due to the early cessation of sludge dumping by Nassau County, New York (March-June 1986). Nassau County was the only dumper in the northeast corner of the dumpsite. In effect, the majority of sludge dumping affecting Station 31 was halted by mid-1986 and this is apparently reflected in the change in SOD rates.

The station in the dumpsite which received most of the dumped sludge is Station 32, which is located in the northwest corner of the site (Fig. 19). Although the sediments are sandy and somewhat similar to those of Stations 30 and 31, the mean SOD rates reflect the high organic input and historically were consistently higher (Fig. 20c). The decline in the SOD rates at Station 32 was precipitous after the phase-out of dumping began and levelled off to background rates as dumping ceased.

Station 33 (NY6), lies on the eastern slope of the Christiaensen Basin, just outside of the western boundary of the dumpsite and receives large inputs of organic material from the site. This has been documented by divers from the Sandy Hook Laboratory who, while performing *in situ* SOD experiments at this station were engulfed by a plume of sewage sludge from a dump observed about one half an hour earlier. The sediments at Station 33 are soft black mud as are those at the other stations in the Christiaensen Basin. Total organic carbon (TOC) in the sediments at this station was extremely high in late 1985 and the first half of 1986, reaching 14 percent carbon by weight (Environmental Processes Division 1987). After the 30 percent reduction in dumping, the TOC was significantly reduced, albeit still higher than the sandy stations (30, 31, and 32). Apparently in response to the 30 percent reduction in sludge dumping, the SOD rates at Station 33 fell from the relatively high rate of ~46 ml $O_2m^{-2}h^{-1}$ to ~37 ml $O_2m^{-2}h^{-1}$, where the rate remained until 1987 when it was significantly lowered to background levels as dumping ceased (Fig. 20d).

In the center of the Christiaensen Basin, Station 34 is polluted not only by sewage sludge but by a variety of other natural and anthropogenic inputs as indicated by the relatively high TOC content (~three percent by weight). Interestingly, the mean annual SOD rates at this station (Fig. 20e) were not as high as those at Stations 32, 33, and 35 (Fig. 20c,d, and f), possibly because the carbon reaching the center of the basin had been mineralized extensively during transport and represented refractory carbon or the 'ashes of the fire.' Only a very small decrease in SOD rates is evident from the data. This is not unexpected however, as the annual mean SOD rates prior to the cessation of dumping were not extraordinarily high (Fig. 20e).

Station 35, on the western slope of the Christiaensen Basin, is quite near and down slope from the dredge spoils dumpsite. Therefore, it is affected by dredge spoils to some substantial, but as yet unquantified, degree. The SOD rates at this station also declined with the cessation of sludge dumping (Fig. 20f). However, in the absence of data to the contrary, it is unlikely that this station is greatly influenced by sewage sludge dumping due to its physical location (7.5 km west of the sewage sludge dumpsite and on the other side of the Christiaensen Basin). More likely, the decrease in SOD rates is due to the 75 percent decrease in dredge spoil dumping between 1985 and 1986.

Correlation coefficients of mean annual SOD rates on mean annual wet tons of sludge dumped were calculated for each station. The stations in the Christiaensen Basin with mud sediments had higher correlation coefficients ($r = .90$, .92, and .85 for Stations 33, 34, and 35, respectively) than the stations with sand bottoms ($r = .64$, .66, and .71 for Stations 30, 31, and 32 respectively). The correlation coefficients must be viewed with care, since only four means were used for each station calculation and factors other than sewage sludge dumping may influence the SOD rates (as indicated above for Station 35).

Samples taken to determine rates of sulfide production by the sediments indicate that no detectable sulfide has been produced during the past three years. This lack of sulfide production is probably due to the failure of hypoxic water to affect the sampling area significantly.

Unlike SOD rates, preliminary analyses of total plankton respiration (TPR) rates from 50 cm off bottom do not indicate a response to the cessation of dumping as previously suggested (Environmental Processes Division 1987). Through March 1987, data from 10 cm off bottom suggest that the TPR rates at Stations 32-35 decreased, apparently in response to the cessation of dumping. Subsequent TPR measurements, however, indicated anomalously low rates at both 50 and 10 cm off bottom during May-July 1987. A large rebound effect occurred in the fall. By November 1987 and continuing through February 1988, the winter 1988 TPR rates at all stations were similar to their respective winter 1987 rates.

SOD rates at Stations 32, 33, and 34 (Fig. 20c,d, and e) appear to have declined significantly in relation to the phase-out of sludge dumping at the 12-mile site. The decline of SOD rates at Station 35 (Fig. 20f), while appearing to be responding to the cessation of sludge dumping, is probably actually responding to the decline in dredge spoil dumping. The SOD rates at Station 31 (Fig. 20b) appear to have also declined with the decrease in sludge dumping albeit to a lesser degree than the others. No such decrease was observed at Station 30 (Fig. 20a).

Mean temperatures of the cold season (November-April), the warm season (May-October), and the whole year were compared with SOD rates for those seasons and years. Although the annual mean temperature was a maximum 3°C cooler in 1988 than in 1986 (1985 and 1987 were intermediate), temperature did not appear to be a major cause of the declining SOD rates.

At every station there was a marked decline in SOD rates between 1985 and 1987 when sludge dumping was reduced from 7.2 to 4.0 million wet tons per year (Fig. 2). With the exception of Station 33, the mean 1987 SOD rates ranged between 13 and 20 ml $O_2m^{-2}h^{-1}$ and remained constant at these levels through July 1988 despite the continued phase-out of dumping. Although the SOD rates at Station 33 were reduced substantially between 1985 and 1987, the decrease between 1987 and 1988 was dramatic, declining from 37 ml $O_2m^{-2}h^{-1}$ to 19 ml $O_2m^{-2}h^{-1}$. Since Station 33 (NY6) was the most heavily polluted by sewage sludge, it may have taken more time for the SOD rates at this station to reflect the phase-out of dumping. The mean annual SOD rates have been reduced, apparently through the cessation of sewage sludge dumping, approaching the rates of other coastal areas which have not been polluted by the ocean disposal of sludge (*e.g.*, Block Island Sound, Buzzards Bay, and the Rhode Island coast) (Phoel 1982).

Further investigations into the SOD rates of these stations should indicate that either: (1) the rates continue to decrease over time to some natural background level; or (2) they do not decline further and the present rates are the background levels for this ecosystem.

Total plankton respiration rates in the bottom water were expected to decline with the phase-out of sludge dumping as the biological oxygen demand (BOD) of the dumped material would have an immediate effect on the water column. At no time were differences in TPR rates 50 cm off bottom observed between the stations on the transect. Rates measured in late 1986 from water 10 cm off bottom indicated that Stations 30 and 31 had similar rates (~5 ml $O_2m^{-3}h^{-1}$), but were significantly lower than those of the other four stations which averaged around 35 ml $O_2m^{-3}h^{-1}$. By spring of 1987, the difference in TPR rates between Stations 30 and 31 and the others was significantly lower (~10 ml $O_2m^{-3}h^{-1}$). After the aforementioned reduction and subsequent rebound in rates between May and October 1987, the 1988 winter rates steadied at about the 1987 winter values. This suggests that, like the SOD rates, the water 10 cm off bottom responded to the decrease in sludge dumping to 4.0 million wet tons, but did not respond to any further decrease in dumping over the long term. *In situ* experiments have indicated that sewage sludge exerts an extremely intense BOD to the bottom waters, but this demand is of relatively short duration.[5] It can be suggested that before dumping was phased out, the continual input of sludge maintained the high TPR rates at those stations which were polluted, but as dumping declined to 4.0 million wet tons per year, the number of dumps per unit time also decreased (the volume of sludge dumped per barge load remained the same). While the BOD of the sludge on the water during each dump was, perhaps, as high as ever, the time between dumps may have permitted the bottom waters to return to more natural background rates.

Continued investigations into the bottom-water TPR rates should indicate if these rates will decline further or if they have reached their lower limits.

BENTHIC MACROFAUNA

One of the important indicators of effects of cessation of sewage sludge dumping is expected to be changes in the structure of benthic communities (Environmental Processes Division 1988). The assumption is that the greatest effects of sludge on benthos will be imposed by summer factors such as high temperatures, low dissolved oxygen, presence of sulfide, and possibly increased additive effects of contaminants. We have concentrated on the summer (July, August, and September) samples at replicate stations R2, NY6, and NY11. Data are available for all eight samples taken at each station on each summer 1986 cruise. For summer 1987, only the three grabs from the center of each station ellipse have been processed, and three or fewer central grabs have been completed for summer 1988. Presentation of results is therefore limited to grabs 1-3 unless otherwise noted. Qualitative observations through March 1989 samples are included, as are historical data from NY6. Field notes on sediment type (Table 5) are presented since grain-size analyses have not been completed.

Results and Discussion

This analysis concentrates on several aspects of the data (total, crustacean, and molluscan species richness, and numbers and biomass of certain "indicator" taxa) which are useful for establishing baselines and detecting changes related to abatement.

[5] Personal communication from William Phoel (NOAA, NMFS, Sandy Hook Laboratory, Highlands, NJ 07732).

Table 5. Field observations of sediment types at 12-mile-dumpsite replicate and broadscale stations (see Fig. 4 for station locations)

Station	Sediment Type
A6	fine sand - compact
A13	medium sand; some clay 6/88
A19	silty-fine sand
A20	soft, sludgy mud
A41	very soft, soupy mud
A43	medium sand
A44	fine-medium sand
A50	medium-fine sand w/shell
A54	medium-fine sand w/shell; mud 6/88
A56	very soft, soupy mud
M31	fine-medium sand w/shell
M49	silty fine sand - compact
M54	silty fine sand - compact
M109	medium-fine sand w/shell; coarse w/mud 6/88
NY3	silty-fine sand - compact
NY5	compact mud
NY6	medium soft sludgy mud
NY11	silty fine-medium sand - often black sticky mud (even clay) below
NY22	silty fine sand - compact
R1	silty sand - compact
R2	sticky, sludgy mud with silty sand with large shell
R3	fine sand - compact
R4	silty fine sand - compact
R5	medium sand
R6	compact, sticky mud

Number of species (S) is a relatively clear indicator of environmental stress at a given station (Green 1977). Within a habitat type (*e.g.*, shallow/sandy, deep/muddy), S is generally lower in areas of natural or man-made stress, whereas variables such as faunal density and diversity may show increases or decreases depending on the nature and severity of the stress.

As expected, the three replicate stations continue to be quite distinct in numbers of species, with NY6<R2<NY11 (Fig. 21). Also, variability within sampling sites remained low relative to other benthic variables examined. These features make the number of species among the most promising measures of faunal recovery. As sludge levels decrease at NY6, numbers of species may increase to about the numbers found at R2, which are typical of most of the Christiaensen Basin. Number of species at NY11 may be higher due to factors such as different sediment type and a general trend toward greater species richness in offshore waters of the bight (Reid *et al.* 1982); values at NY6 and R2 may remain lower than at NY11 in the absence of sludge influence.

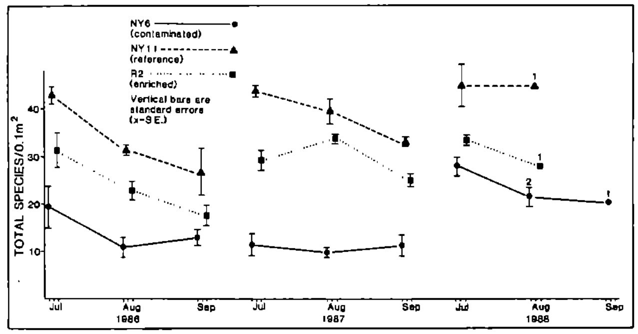

Figure 21. Means and standard errors for numbers of species (S) per 0.1 m² for replicate grabs 1-3 from each replicate station, summers 1986-88. Where less than three grabs have been processed, numbers completed are indicated above error bars.

Number of species at NY6 did not indicate any recovery between summer 1986 and 1987; values were in fact slightly lower in 1987 (Fig. 21). There were substantial decreases at NY6 and R2 between July and August in 1986, but not 1987. This could possibly indicate less deterioration of water/sediment quality over the summer in 1987 than in 1986. NY11 had similar steady declines over both summers. In summer 1988, values showed little change at R2, increased slightly at NY11 and more at NY6, which approached the R2 values. This is taken as a response to the sludge phase-out at NY6.

Numbers of species of crustaceans and molluscs, which are related to overall S, are also presented since these taxa are relatively sensitive and should respond to sludge phase-out. Trends in crustaceans (Fig. 22) paralleled those for total species (Fig. 21). Station NY6 had the fewest crustacean species, and NY11 the most. The declines at all three stations over summer 1986 show that crustaceans were responsible for much of the change in overall species richness that summer. Crustacean S values decreased again at NY11 over summer 1987, but remained level or increased at NY6 and R2. By September 1987, numbers of crustacean species at the three stations were similar. The preliminary 1988 data indicate increases at all stations, with the similar values at NY6 and R2 being another possible indication of recovery. Molluscan S values (Fig. 23) were also least at NY6 and greatest at NY11, and the temporal trends resembled those in overall and crustacean S.

The polychaete *Capitella capitata* is widely used as an indicator of organic enrichment, although it also reaches high densities in response to other disturbances, such as defaunation (Eagle and Rees 1973). [*Capitella capitata* is actually a complex of several species which are morphologically similar, but have distinct genomes and life histories (Grassle and Grassle 1976; Grassle *et al.* 1987). The

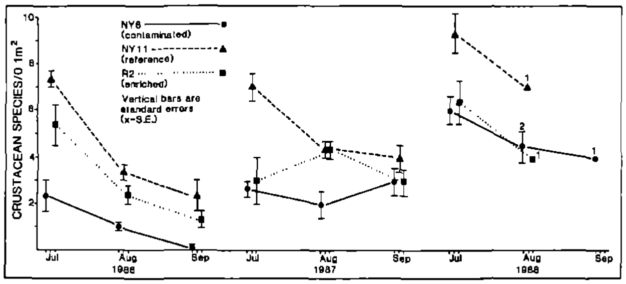

Figure 22. Means and standard errors for numbers of crustacean species per 0.1 m^2 for replicate grabs 1-3 from each replicate station, summers 1986-88. Where less than three grabs have been processed, numbers completed are indicated above error bars.

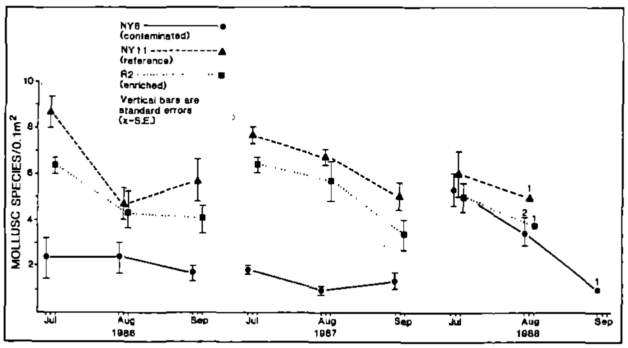

Figure 23. Means and standard errors for numbers of molluscan species per 0.1 m^2 for replicate grabs 1-3 from each replicate station, summers 1986-1988. Where less than grabs have been processed, numbers completed are indicated above error bars.

differences may be important in interpreting population fluctuations and in using *Capitella* as a pollution indicator. The *Capitella* spp. from the inner New York Bight appear to resemble species 1a of Grassle and Grassle (1976). Species produces a relatively large number (200-2000) of small eggs which remain in the plankton for several days, whereas the planktonic phase is shorter or absent in most other *Capitella* species examined to date. Since our specimens have not been positively identified, they will be referred to as *Capitella* sp.]

Station NY6 has apparently been the center of a "highly altered" faunal assemblage occupying a small portion of the Christiaensen Basin since at least the first benthic samples were collected there in 1968 (Pearce 1972; Pearce *et al.* 1981; Boesch 1982; Steimle *et al.* 1982; and Steimle 1985). High densities of *Capitella* sp. have been the most distinctive feature of this assemblage (Pearce *et al.* 1981). In NOAA's Northeast Monitoring Program sampling between 1979 and 1985, *Capitella* was the numerical dominant in eight of the 13 collections at NY6, with a peak mean density of 5604 individuals/0.1 m^2. *Capitella* was rarely the most abundant species at any of the other 47 New York Bight stations, and maximum density never exceeded 678/0.1 m^2 elsewhere.

There may be a seasonality in abundance of *Capitella* at NY6 (Fig. 24). In winter samplings from 1979 through 1983, *Capitella* was the top dominant only once (November 1983), and mean densities ranged from 1 to 415/0.1 m^2. Conversely, *Capitella* was the top dominant in all summer samplings from 1980 through 1985, though abundances were quite variable ($\bar{x}$ = 87-5604/0.1 m^2). Where there were multiple samplings in a given summer, shorter-term fluctuations were observed. There were large **increases** in abundance between July and August 1983. The monthly summer sampling in the present study has shown a **decrease** from very high densities ($\bar{x}$ = 2150/0.1 m^2) between July and August 1986, with little recovery in September 1986; and an **increase** from low numbers in July 1987 to a mean of 2187/0.1 m^2 in September 1987. Numbers were low (less than 10 per grab) in all 1988 samples.

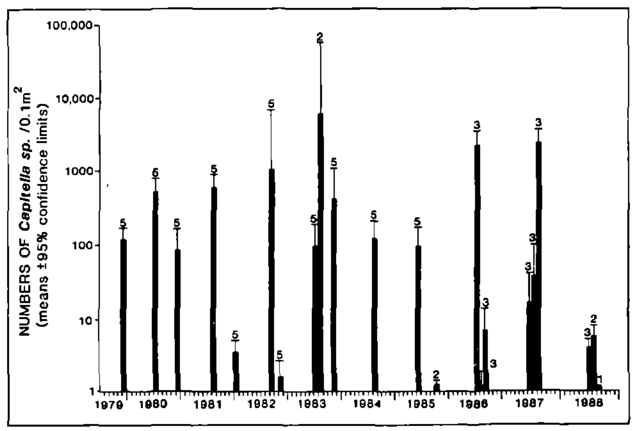

Figure 24. Mean and 95-percent confidence limits for numbers of *Capitella* sp. per 0.1 m^2 at NY6 from 1979 to 1988. Numbers of samples analyzed are given above confidence limits.

One hypothesis to explain the sudden changes is that *Capitella* populations are limited by deteriorating water quality (*e.g.*, high temperature, low dissolved oxygen, and concentrations of hydrogen sulfide), for variable periods during at least some summers. Recurrent hypoxia, and occasional anoxia and sulfide generation, have been documented at NY6[6]. *Capitella* has been reported to have only a moderate tolerance for hypoxia (Reish 1970). This theory will be examined by comparing *Capitella* densities to physical/chemical data (*e.g.*, temperature, oxygen, redox, and sulfide) being collected in the present study, and also to the limited historical data available.

There are of course other possible causes for the observed variability, perhaps acting in concert with changing water quality. Natural populations of *Capitella* sp. are known to have oscillations in abundance which have been linked to declining food supplies and overshooting an environment's carrying capacity (Chesney 1985). Much of the variability is undoubtedly due to the combination of spatial patchiness in *Capitella* densities and inexact relocation for multiple sampling, as shown by the wide confidence limits (Fig. 24). The variability of *Capitella* densities makes it difficult to use in monitoring changes following phase-out of dumping. It may be most informative to focus on the peak densities of perhaps greater than or equal

[6] **Personal communication from Andrew Draxler (NOAA, NMFS, Sandy Hook Laboratory, Highlands, NJ 07732).**

to 1000/0.1 m² that recent past levels of organic enrichment have supported. If and when such densities are no longer found in future summer sampling (as they were not in summer 1988), and concurrent chemical measurements also indicate a lessening of sewage influence, "recovery" of this component of the benthos may be inferred.

Rhynchocoels, or ribbon worms, appear characteristic of the NY6 area, and more consistently abundant than *Capitella*. Rhynchocoel biomass at NY6 is typically an order of magnitude or more higher than at NY11 or R2, although the August 1986 biomass at R2 was nearly as great as at NY6 (Fig. 25). A slight decrease was apparent at NY6 in 1988, but also at NY11 and R2, so there was no clear response to the phase-out.

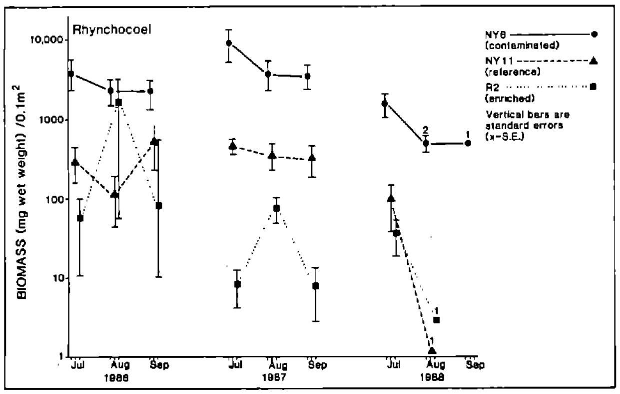

Figure 25. Means and standard errors for biomass of rhynchocoels per 0.1 m² for replicate grabs 1-3 from each replicate station, summers 1986-88. Where less than three grabs have been processed, numbers completed are indicated above error bars.

Pherusa affinis, a large flabelligered polychaete, is the dominant species in biomass over much of the Christiaensen Basin, including Station R2. Its biomass at R2 has consistently been high (Fig. 26). It is hypothesized that biomass at NY6 will approach that at R2 as sludge influence wanes. There does not appear to be any "recovery" of *Pherusa* biomass at NY6 between 1986 and 1988; lowest values occurred in July 1987 and September 1988 (Fig. 26). *Pherusa* is relatively long-lived and has little mobility, so if biomass increase at NY6 depends on larval settlement and growth, the process could take several years. Biomass at NY11 has been about an order of magnitude lower than at R2, with both stations showing increasing trends over the study period. Field observations indicate the high biomass at R2 has persisted at least through March 1989.

Tomato seeds have been used as indicators of sludge presence in past New York Bight studies (Pearce 1971), and also in Great Britain. Seeds were not quantified in the first grabs processed, so some 1986 data are missing or suspect. In 1987 and 1988, mean seed numbers were always highest at NY6 (Fig. 27). The apparent increase in counts there in 1988 is puzzling, but it is becoming obvious that the seeds are not easily broken down or winnowed out of the area. Numbers were always negligible at NY11.

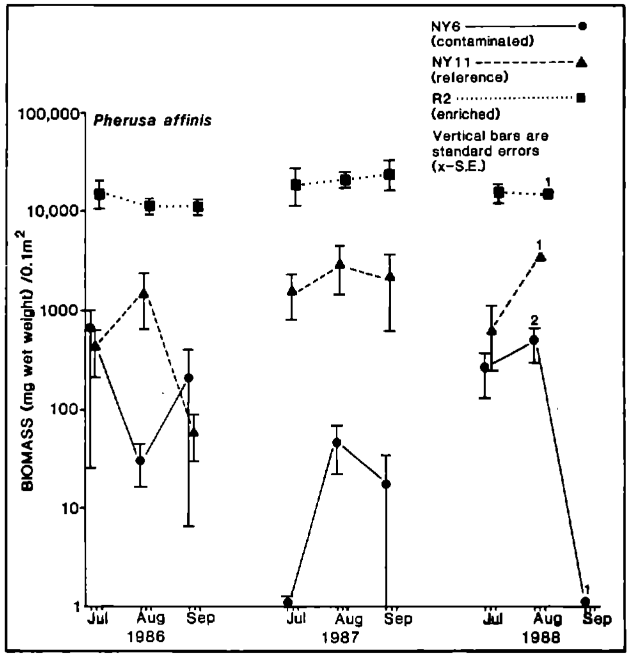

Figure 26. Means and standard errors for biomass of *Pherusa affinis* per 0.1 m² for replicate grabs 1-3 from each replicate station, summers 1986-88. Where less than three grabs have been processed, numbers completed are indicated above error bars.

There are other sources of seeds to the Christiaensen Basin (*e.g.*, dredged material, Hudson-Raritan plume), so counts at NY6 and R2 may never drop to the NY11 levels. Counts at NY6 and R2 should decrease and become similar, however, and this information may be useful to augment that from other indicators of sludge fates (bacteria, coprostanol, heavy metals, etc.).

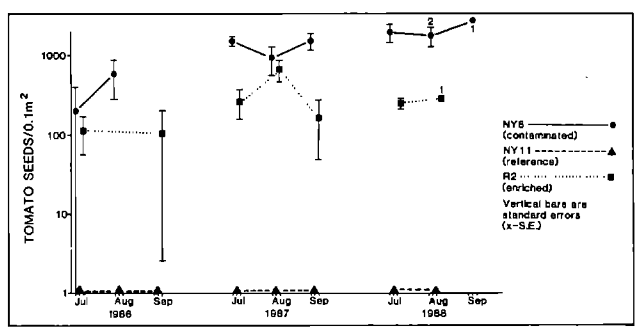

Figure 27. Means and standard errors for numbers of tomato seeds per 0.1 m² for replicate grabs 1-3 from each replicate station, summers 1986-88. Where less than three grabs have been processed, numbers completed are indicated above error bars.

In summary, there is little evidence of a benthic macrofaunal recovery between the summers of 1986 and 1987 with a somewhat greater apparent response (in numbers of species and of *Capitella* sp.) at NY6 in summer 1988. NY6 can also be distinguished from NY11 by the abundance of rhynchocoels, *Pherusa affinis*, and tomato seeds, but there were no clear indications that NY6 is becoming more like the other stations in these variables.

FISH AND MEGAINVERTEBRATES

The New York Bight is best described as a transitional area for the fish fauna, with significant overlap of cold- and warm-temperature species. The fauna exhibit dramatic seasonal shifts, a feature which has implications for any study intended to measure effects of natural fluctuations and man's impacts on fish abundance.

Characterization of the finfish and megainvertebrate (shellfish/crustacean) communities during the dumping phase at the 12-mile dumpsite are drawn from data taken during 19 mo of sampling at the replicate stations (NY6, R2, and NY11). Data were derived from a series of standard tows, gear and techniques which have been described previously (Environmental Processes Division 1988). Similarities of species composition at the three stations, seasonal patterns of species distribution, dominance hierarchy, and relative standing crop comprise this characterization.

Results and Discussion

Species Composition, Distribution, and Abundance

There were similar numbers of finfish species in the two years of the study -- 36 in 1986-87 and 39 in 1987-88 (Tables 6 and 7). Little skate, ocean pout, red and silver hakes, and winter flounder were the most significant in demersal biomass both years, with spiny dogfish added to the group during 1987-88. During this second year, four species, little skate, ocean pout, spiny dogfish, and winter flounder made up 81 percent of fish biomass at R2, 66 percent at NY6, and 81 percent at NY11. Similar patterns were evident in megainvertebrate species composition for the two years -- 13 in 1986-87 and 14 in 1987-88 (Tables 6 and 7). Rock crab dominated the biomass: 73 percent at R2, 75 percent at NY6, and 57 percent at NY11. There was a decline in biomass for both finfish and megainvertebrate

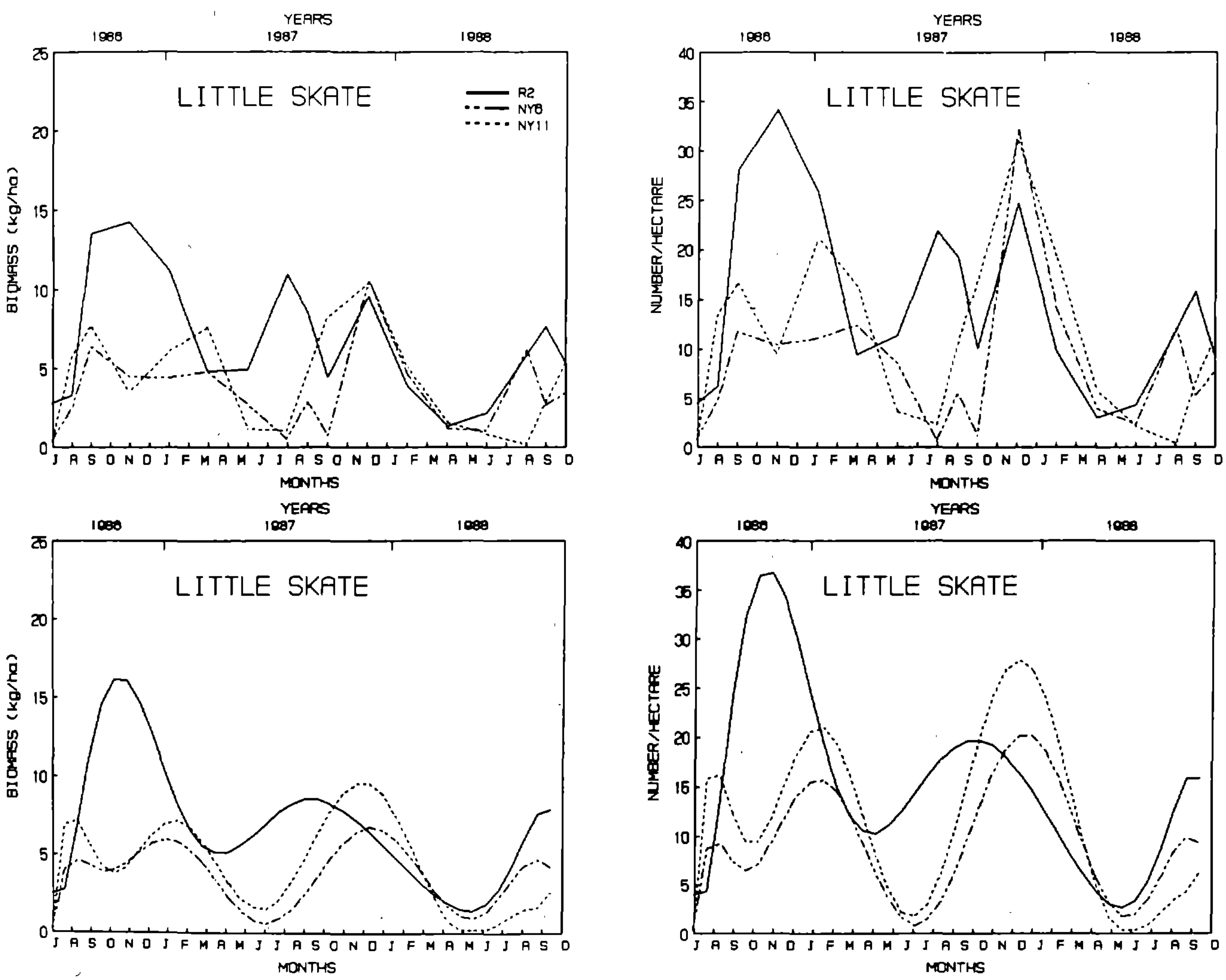

Figure 28. Trends in biomass as kilograms per hectare [(a) raw and (b) smoothed], and in numbers per hectare [(c) raw and (d) smoothed], for little skate, *Raja erinacea*, collected at replicate stations from July 1986 to September 1988. The data were fitted to a ninth-order polynomial.

Table 6. Finfish and megainvertebrate species composition, biomass, and numbers recorded from each of the three replicate stations, R2, NY6, and NY11, from July 1986 through May 1987; weights and numbers normalized to a 1-km tow; occ = occurrence

Finfish Species	Station R2 (56 Tows)					Station NY6 (56 Tows)					Station NY11 (56 Tows)				
	Occ	Wt per Occ	Wt per Tow	No per Occ	No per Tow	Occ	Wt per Occ	Wt per Tow	No per Occ	No per Tow	Occ	Wt per Occ	Wt per Tow	No per Occ	No per Tow
Alewife	9	0.16	0.03	2.6	0.4	5	0.13	0.01	1.5	0.1	2	0.04	0.00	1.9	0.1
American shad	4	0.04	0.00	1.5	0.1	1	0.05	0.00	1.0	0.0	1	0.04	0.00	0.8	0.0
Atlantic herring	3	0.22	0.01	1.7	0.1	6	0.39	0.04	1.8	0.2	1	0.15	0.00	1.5	0.0
Atlantic menhaden	0	0.00	0.00	0.0	0.0	1	0.16	0.00	0.8	0.0	0	0.00	0.00	0.0	0.0
Bigeye	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0	1	0.04	0.00	0.9	0.0
Black sea bass	4	0.15	0.01	0.7	0.1	3	0.25	0.01	0.8	0.0	2	0.12	0.00	0.8	0.0
Blueback herring	2	0.06	0.00	1.5	0.1	0	0.00	0.00	0.0	0.0	4	0.11	0.01	4.0	0.3
Bluefish	1	0.04	0.00	0.8	0.0	1	0.60	0.01	0.9	0.0	0	0.00	0.00	0.0	0.0
Butterfish	30	0.69	0.37	27.7	14.8	33	0.37	0.22	22.5	13.3	18	0.15	0.05	7.2	2.3
Clearnose skate	1	1.38	0.02	0.8	0.0	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0
Cunner	5	0.36	0.03	1.8	0.2	13	3.29	0.76	12.1	2.8	7	2.23	0.28	14.9	1.9
Fourspot flounder	19	0.90	0.30	4.6	1.6	19	0.93	0.31	4.4	1.5	26	0.38	0.18	2.4	1.1
Goosefish	5	2.03	0.18	1.1	0.1	10	0.16	0.03	1.0	0.2	7	0.92	0.12	1.1	0.1
Gulfstream flounder	10	0.04	0.01	1.5	0.3	21	0.11	0.04	2.9	1.1	15	0.13	0.03	3.3	0.9
Haddock	9	0.06	0.01	1.6	0.3	11	0.06	0.01	1.1	0.2	7	0.04	0.01	1.4	0.2
Little skate	53	6.18	5.85	14.1	13.3	42	4.09	3.06	11.4	8.6	49	5.24	4.59	14.4	12.6
Longhorn sculpin	7	0.34	0.04	1.1	0.1	2	0.23	0.01	0.7	0.0	2	0.26	0.01	0.8	0.0
Northern puffer	0	0.00	0.00	0.0	0.0	1	0.09	0.00	0.9	0.0	0	0.00	0.00	0.0	0.0
Northern searobin	6	0.13	0.01	1.3	0.1	6	0.06	0.01	1.0	0.1	0	0.00	0.00	0.0	0.0
Ocean pout	16	4.48	1.28	10.3	2.9	12	0.72	0.15	1.2	0.3	11	4.93	0.97	5.7	1.1
Red hake	28	0.62	0.31	4.9	2.5	40	0.30	0.21	4.5	3.2	6	0.08	0.01	1.3	0.1
Rough scad	0	0.00	0.00	0.0	0.0	1	0.04	0.00	0.9	0.0	0	0.00	0.00	0.0	0.0
Round herring	0	0.00	0.00	0.0	0.0	2	0.47	0.02	33.8	1.2	0	0.00	0.00	0.0	0.0
Scup	13	0.94	0.22	9.7	2.3	11	0.75	0.15	7.4	1.5	10	0.95	0.17	11.6	2.1
Sea raven	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0	5	0.87	0.08	1.3	0.1
Seahorse	3	0.05	0.00	0.9	0.0	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0
Silver hake	27	0.26	0.12	4.9	2.4	24	0.41	0.18	10.3	4.4	19	0.17	0.06	6.3	2.1
Smooth dogfish	3	3.55	0.19	1.7	0.1	2	5.04	0.18	2.2	0.1	1	3.45	0.06	0.8	0.0
Spiny dogfish	8	10.70	1.53	6.4	0.9	7	19.94	2.49	12.8	1.6	6	4.08	0.44	2.6	0.3
Spotted hake	7	0.34	0.04	1.7	0.2	6	3.88	0.42	1.7	0.2	4	0.14	0.01	0.9	0.1
Striped searobin	4	0.31	0.02	1.5	0.1	3	0.12	0.01	1.3	0.1	2	0.05	0.00	0.9	0.0
Sturgeon uncl.	0	0.00	0.00	0.0	0.0	1	4.63	0.08	0.8	0.0	0	0.00	0.00	0.0	0.0
Summer flounder	4	0.73	0.05	1.6	0.1	9	0.37	0.06	1.0	0.2	3	0.44	0.02	0.8	0.0
Tautog	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0	1	0.28	0.01	1.4	0.0
Weakfish	1	0.04	0.00	0.8	0.0	1	1.08	0.02	0.8	0.0	0	0.00	0.00	0.0	0.0
Windowpane	23	0.74	0.30	5.8	2.4	33	0.95	0.56	5.7	3.4	17	0.31	0.10	1.7	0.5
Winter flounder	47	1.89	1.58	9.0	7.5	48	1.56	1.34	7.9	6.8	42	0.88	0.66	4.0	3.0
Winter skate	6	0.96	0.10	1.8	0.2	3	4.14	0.22	0.8	0.0	7	2.44	0.30	1.7	0.2
Yellowtail flounder	8	0.06	0.01	1.0	0.1	15	0.17	0.05	2.5	0.7	14	0.32	0.08	2.5	0.6
Finfish subtotal			**12.66**		**53.32**			**10.67**		**51.78**			**8.23**		**30.00**
Invertebrate species															
American lobster	19	0.31	0.11	1.3	0.4	29	0.55	0.28	1.9	1.0	16	0.43	0.12	1.3	0.4
Blue crab	2	0.12	0.00	0.9	0.0	1	0.04	0.00	0.8	0.0	0	0.00	0.00	0.0	0.0
Hermit crab uncl.	16	0.05	0.01	1.3	0.4	10	0.11	0.02	1.6	0.3	2	0.06	0.00	0.8	0.0
Horseshoe crab	16	3.48	0.99	1.9	0.6	9	1.38	0.22	0.9	0.2	0	0.00	0.00	0.0	0.0
Jonah crab	13	0.28	0.07	1.6	0.4	12	0.37	0.08	2.7	0.6	7	0.05	0.01	0.9	0.1
Ladycrab	6	0.05	0.01	0.9	0.1	5	0.05	0.00	0.9	0.1	0	0.00	0.00	0.0	0.0
Longfin squid	31	0.41	0.23	12.2	6.8	27	0.24	0.12	12.5	6.0	28	0.61	0.30	16.8	8.4
Moon snail uncl.	1	0.04	0.00	1.7	0.0	3	0.06	0.00	0.7	0.0	2	0.19	0.01	0.8	0.0
Ocean quahog	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0	1	0.46	0.01	0.9	0.0
Red shrimp	3	0.05	0.00	0.9	0.0	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0
Rock crab	50	4.72	4.21	53.0	47.3	45	2.93	2.35	21.7	17.4	50	1.06	0.94	8.0	7.1
Shortfin squid	5	0.75	0.07	16.1	1.4	7	0.26	0.03	5.0	0.6	4	0.31	0.02	8.7	0.6
Spider crab uncl.	0	0.00	0.00	0.0	0.0	1	0.04	0.00	1.7	0.0	0	0.00	0.00	0.0	0.0
Starfish uncl.	46	0.12	0.10	4.8	4.0	13	0.04	0.01	1.5	0.3	53	0.24	0.23	8.0	7.6
Invertebrate subtotal			**5.80**		**61.42**			**3.13**		**26.56**			**1.64**		**24.28**
All species			**18.46**		**114.73**			**13.80**		**78.35**			**9.87**		**54.28**

Table 7. Finfish and megainvertebrate species composition, biomass, and numbers recorded from each of the three replicate stations, R2, NY6, and NY11, from July 1987 through May 1988; weights and numbers normalized to a 1-km tow; occ = occurrence

Finfish species	Station R2 (54 tows)					Station NY6 (58 tows)					Station NY11 (56 tows)				
	Occ	Wt per occ	Wt per tow	No per occ	No per tow	Occ	Wt per occ	Wt per tow	No per occ	No per tow	Occ	Wt per occ	Wt per tow	No per occ	No per tow
Alewife	5	0.09	0.01	1.4	0.1	11	0.26	0.05	2.3	0.4	6	0.32	0.03	3.8	0.4
Atlantic herring	8	2.79	0.41	16.9	2.5	8	1.53	0.21	9.9	1.4	4	14.64	1.05	96.8	6.9
Atlantic mackerel	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0	1	0.25	0.00	1.7	0.0
Black sea bass	0	0.00	0.00	0.0	0.0	1	0.31	0.01	1.0	0.0	3	0.15	0.01	0.8	0.0
Blueback herring	4	0.26	0.02	5.0	0.4	2	1.78	0.06	25.2	0.9	0	0.00	0.00	0.0	0.0
Butterfish	27	0.55	0.28	24.3	12.2	34	0.40	0.23	25.4	14.9	22	0.42	0.16	29.7	11.7
Clearnose skate	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0	2	1.09	0.04	0.9	0.0
Cunner	6	0.30	0.03	2.0	0.2	14	0.74	0.18	5.4	1.3	4	1.38	0.10	10.8	0.8
Fourspot flounder	20	0.63	0.23	3.7	1.4	24	0.51	0.21	3.1	1.3	28	0.45	0.22	3.3	1.7
Goosefish	4	0.69	0.05	0.9	0.1	11	0.74	0.14	1.2	0.2	3	5.37	0.29	1.0	0.1
Gulfstream flounder	3	0.06	0.00	1.1	0.1	9	0.08	0.01	2..0	0.3	6	0.04	0.00	0.9	0.1
Haddock	3	0.05	0.00	1.1	0.1	3	0.06	0.00	1.7	0.1	1	0.08	0.00	0.8	0.0
Hickory shad	2	0.06	0.00	0.8	0.0	1	0.04	0.00	0.8	0.0	0	0.00	0.00	0.0	0.0
Little skate	51	8.52	8.04	18.5	17.5	50	4.25	3.66	9.7	8.4	45	5.49	4.41	13.6	11.0
Longhorn sculpin	4	0.29	0.02	0.8	0.1	4	0.42	0.03	1.0	0.1	1	0.78	0.01	1.6	0.0
Northern kingfish	1	0.08	0.00	0.8	0.0	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0
Northern searobin	3	0.16	0.01	0.9	0.1	1	0.05	0.00	1.1	0.0	5	0.09	0.01	1.4	0.1
Ocean pout	20	4.66	1.73	11.4	4.2	17	0.91	0.27	2.8	0.8	13	1.93	0.45	2.3	0.5
Planehead filefish	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0	3	0.06	0.00	0.8	0.0
Pollock	0	0.00	0.00	0.0	0.0	1	0.12	0.00	1.2	0.0	0	0.00	0.00	0.0	0.0
Red hake	30	0.78	0.43	6.0	3.3	38	1.80	1.18	10.6	7.0	8	0.27	0.04	2.2	0.3
Scup	11	0.18	0.04	5.8	1.2	12	0.24	0.05	9.6	2.0	12	0.14	0.03	8.8	1.9
Sea raven	3	0.33	0.02	1.0	0.1	1	1.11	0.02	1.6	0.0	5	0.23	0.02	0.9	0.1
Silver hake	39	0.38	0.27	3.6	2.6	51	0.49	0.43	8.0	7.0	29	0.22	0.12	3.7	1.9
Silverside	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0	1	0.04	0.00	0.8	0.0
Smooth dogfish	8	3.23	0.48	1.7	0.3	4	1.61	0.11	0.8	0.1	1	1.22	0.02	0.9	0.0
Spiny dogfish	6	7.91	0.88	1.8	0.2	2	4.72	0.16	1.6	0.1	1	10.31	0.18	2.7	0.0
Spotted hake	8	0.21	0.03	1.7	0.2	11	0.31	0.06	2.5	0.5	3	0.11	0.01	0.8	0.0
Striped anchovy	0	0.00	0.00	0.0	0.0	1	0.04	0.00	0.9	0.0	0	0.00	0.00	0.0	0.0
Striped searobin	5	0.05	0.00	1.5	0.1	8	0.07	0.01	1.4	0.2	2	0.18	0.01	0.9	0.0
Summer flounder	11	1.28	0.26	2.9	0.6	16	0.54	0.15	1.5	0.4	14	0.58	0.15	1.9	0.5
Tautog	0	0.00	0.00	0.0	0.0	1	0.20	0.00	1.0	0.0	1	0.78	0.01	1.6	0.0
Windowpane	19	0.62	0.22	2.0	0.7	22	0.42	0.16	2.2	0.8	12	0.38	0.08	1.3	0.3
Winter flounder	51	1.61	1.52	7.4	7.0	46	1.46	1.16	6.1	4.9	44	0.64	0.50	2.7	2.1
Winter skate	1	1.41	0.03	1.0	0.0	0	0.00	0.00	0.0	0.0	1	0.37	0.01	0.9	0.0
Yellowtail flounder	5	0.55	0.05	1.8	0.2	8	0.84	0.12	2.2	0.3	16	0.56	0.16	1.6	0.4
Finfish subtotal			**15.08**		**55.29**			**8.67**		**53.33**			**8.13**		**41.10**
Invertebrate species															
American lobster	19	0.74	0.26	2.1	0.7	29	0.75	0.37	2.8	1.4	27	0.97	0.47	2.6	1.3
Blue crab	1	0.14	0.00	0.7	0.0	2	0.18	0.01	0.8	0.0	0	0.00	0.00	0.0	0.0
Hermit crab uncl.	24	0.10	0.05	1.8	0.8	20	0.20	0.07	3.4	1.2	3	0.10	0.01	0.8	0.0
Horseshoe crab	19	3.69	1.30	2.0	0.7	20	2.06	0.71	1.3	0.4	5	2.57	0.23	1.4	0.1
Jonah crab	15	0.23	0.07	1.4	0.4	20	0.24	0.08	1.7	0.6	8	0.12	0.02	0.9	0.1
Ladycrab	10	0.29	0.05	4.3	0.8	3	0.05	0.00	1.0	0.1	2	0.10	0.00	1.7	0.1
Longfin squid	24	0.85	0.38	21.5	9.6	31	0.38	0.20	13.4	7.2	35	0.86	0.54	14.5	9.1
Moon snail uncl.	3	0.27	0.01	1.1	0.1	1	0.22	0.00	0.9	0.0	2	0.42	0.02	1.3	0.0
Ocean quahog	1	0.25	0.00	0.8	0.0	0	0.00	0.00	0.0	0.0	2	0.19	0.01	0.9	0.0
Rock crab	51	3.59	3.39	27.3	25.8	49	5.91	4.99	52.3	44.1	47	1.76	1.48	12.3	10.3
Shortfin squid	1	0.04	0.00	2.3	0.0	2	2.10	0.07	61.2	2.1	5	0.39	0.04	24.1	2.2
Spider crab uncl.	2	0.38	0.01	0.8	0.0	0	0.00	0.00	0.0	0.0	0	0.00	0.00	0.0	0.0
Starfish uncl.	25	0.06	0.03	2.0	0.9	37	0.12	0.08	11.0	7.0	45	0.18	0.15	4.4	3.6
Invertebrate subtotal			**5.56**		**39.82**			**6.59**		**64.13**			**2.95**		**26.76**
All species			**20.63**		**95.11**			**15.27**		**117.46**			**11.08**		**67.86**

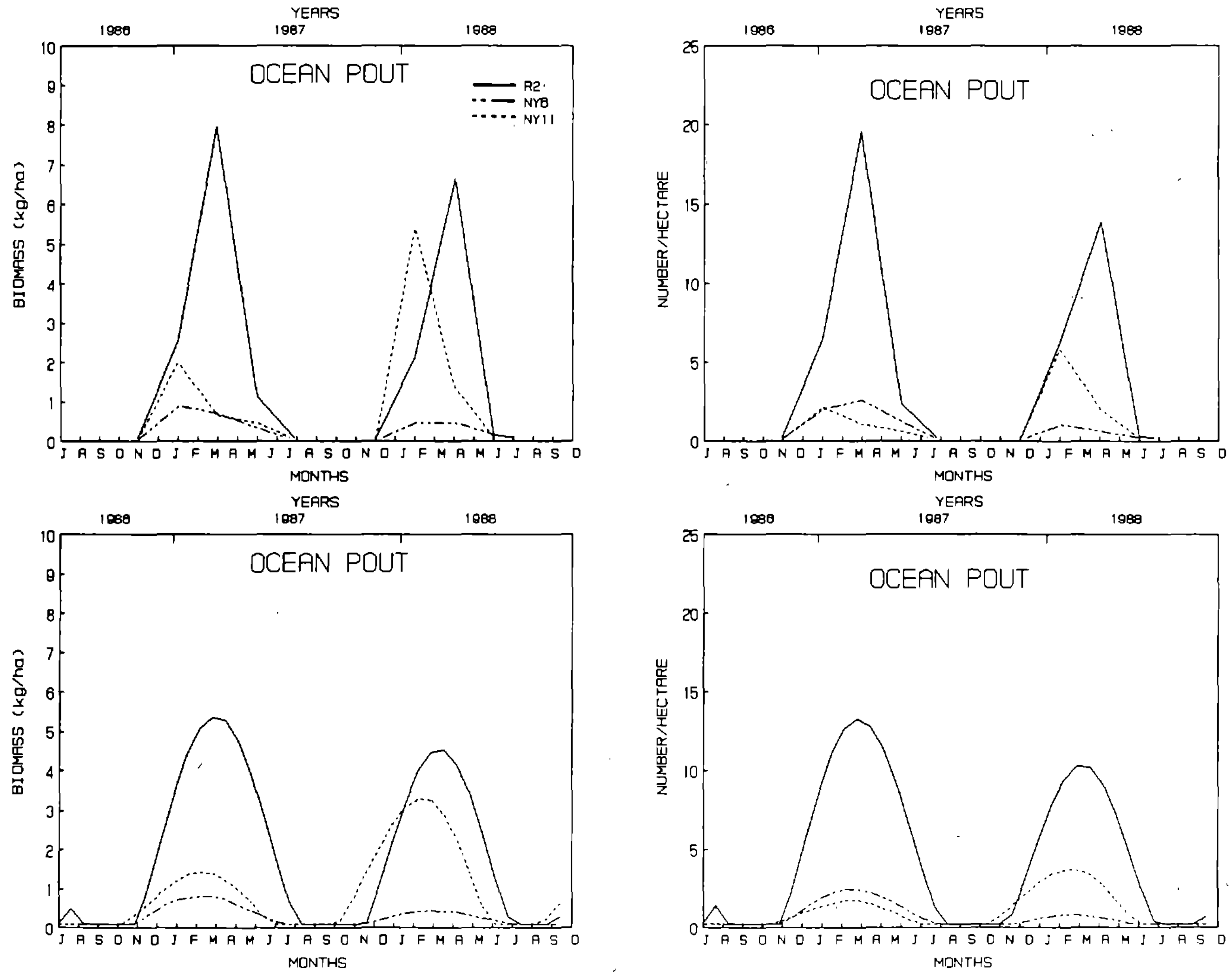

Figure 29. Trends in biomass as kilograms per hectare [(a) raw and (b) smoothed], and in numbers per hectare [(c) raw and (d) smoothed], for ocean pout, *Macrozoarces americanus*, collected at replicate stations from July 1986 to September 1988. The data were fitted to a ninth-order polynomial.

biomass southward from R2, with this station producing 50 percent more biomass than NY11.

The distribution of the predominant fishes and megainvertebrates at the three replicate stations, as expected, showed definite seasonal patterns as follows.

Little skate, *Raja erinacea*, occurred throughout the year at all stations, but were generally most abundant during fall and winter (September-January, Fig. 28a,b,c, and d). There appears to be a closer correspondence in distribution between NY6 and NY11 with skate moving into the more northern site (R2) in mid to late summer in 1986 and 1987, peak biomass occurring about 2 mo earlier than at the more southern stations (NY6 and NY11). Station rank in weight and number was R2>NY11>NY6.

Ocean pout, *Macrozoarces americanus*, showed distinct seasonal peaks from January through May (Fig. 29 a,b,c, and d). Comparisons between 1987 and 1988 indicate similar patterns, with the largest biomass found at enriched station R2. The only major difference between the two years was a distinct increase in 1988 winter biomass at NY11. Station rank in weight and number was R2>NY11>NY6.

Silver hake, *Merluccius bilinearis*, predominantly a cold-water species, exhibited a multimodal distribution (Fig. 30 a,b,c, and d). A numerical peak in August-September with low biomass, followed by a biomass peak in May, indicates an influx of young. Station rank in abundance and biomass was NY6>R2>NY11.

Red hake, *Urophycis chuss*, another cold-water species, followed a pattern similar to silver hake. An influx of young in August was followed by principal peaks in abundance during January and March, primarily at NY6 (Fig. 31 a,b,c, and d). R2 exhibited a peak in numbers from January to March. A biomass peak occurred at NY6 in March 1987, but was not as evident in 1988. Station rank in biomass and abundance was R2>NY6>NY11.

Winter flounder, *Pseudopleuronectes americanus*, generally occurred throughout the sampling period and seasonal fluctuations were generally similar in both numbers and biomass (Fig. 32 a,b,c, and d). There were synchronous

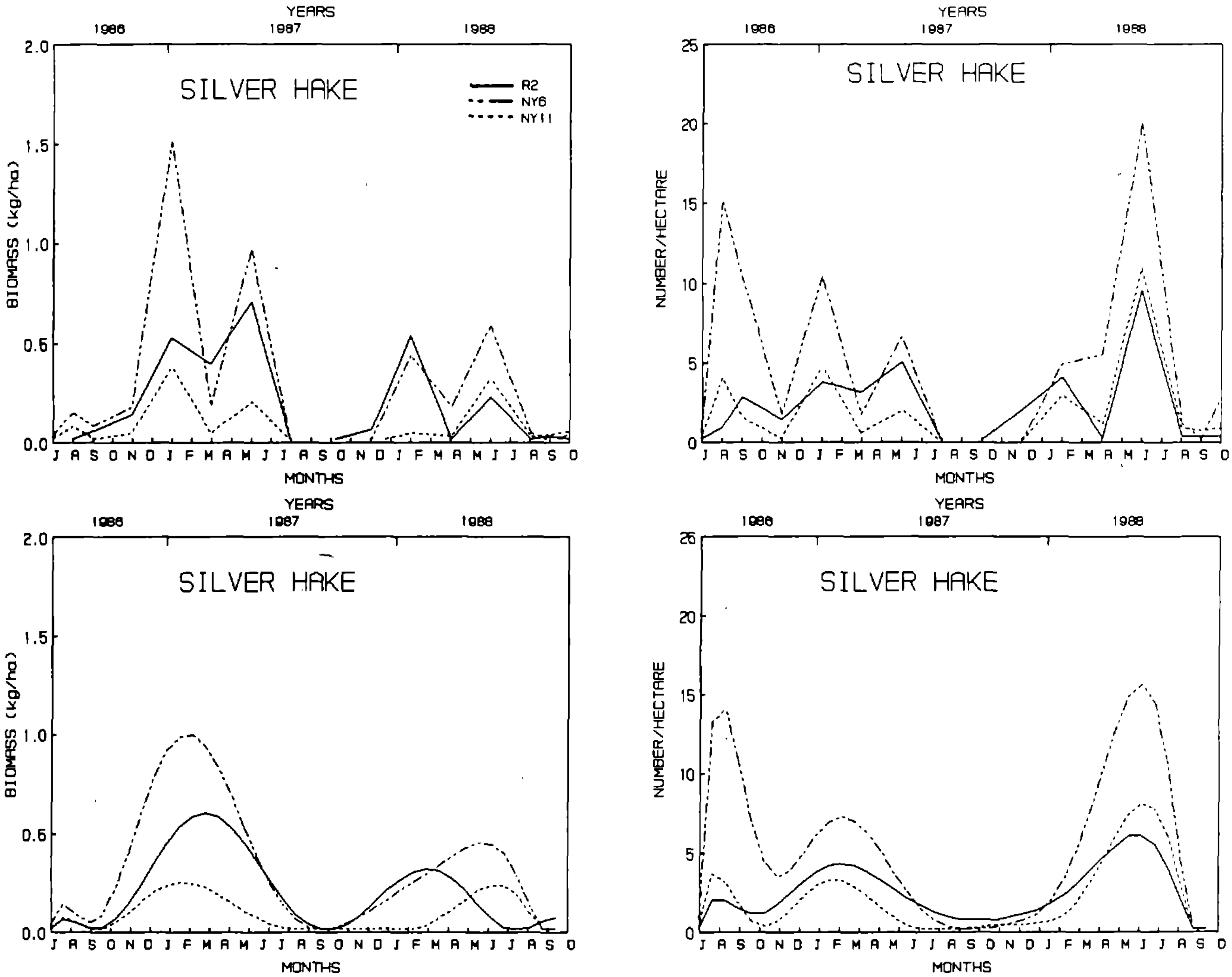

Figure 30. Trends in biomass as kilograms per hectare [(a) raw and (b) smoothed], and in numbers per hectare [(c) raw and (b) smoothed], for silver hake, *Merluccius bilinearis*, collected at replicate stations from July 1986 to September 1988. The data were fitted to a ninth-order polynomial.

extended peaks at all stations. Stations ranked as follows: R2>NY6>NY11, with a 3:1 ratio between the northern and southern stations.

Windowpane flounder, *Scophthalmus aquosus*, were also available throughout the year (Fig. 33 a,b,c, and d). There were no extreme seasonal peaks. There was a modest increase in numbers and biomass from September to March. Station trends, at a reduced level, were similar to that of winter flounder. Biomass ranked as follows: NY6>R2>NY11, with a 6:1 disparity between NY6 and NY11.

Fourspot flounder, *Paralichthys oblongus*, displayed distinct protracted summer peaks in both numbers and biomass (Fig.34a,b,c, and d). Fourspot flounder were absent from the sites from January through March. Numbers and biomass were similar among all stations, essentially equal at R2 and NY6. Biomass ranked NY6>R2>NY11.

Rock crab, *Cancer irroratus*, was the dominant megainvertebrate. Although available year-round, the principal peak of abundance and biomass occurred from May to September (Fig. 35 a,b,c, and d). R2 and NY6 were the sites of highest numerical abundance and biomass. Biomass ranked R2>NY6>NY11 with a 5:1 ratio between R2 and NY11.

American lobster, *Homarus americanus*, was considerably less abundant than rock crab (Fig. 36 a,b,c, and d), but supports an inshore fishery in the surrounding area. Patterns of numbers and biomass were parallel at NY11 and R2 with peaks in August-November and March. The peaks at NY6 occurred in November and March. NY6 led other stations in frequency of occurrence, count, and biomass. Biomass ranked NY6>NY11>R2.

In general, the sampling area has presented a two-season pattern of species abundance with boreal species in winter being replaced by warm temperate forms in summer. Even quasi-resident benthic species displayed inshore-offshore movements. For example, winter flounder typically move inshore during colder months to spawn in estuaries in the spring, moving out as temperatures rise in the late spring. The time lag in availability can be inferred by comparing station peaks (Fig. 32). They are generally replaced by summer flounder inshore in the summer, a species which overwinters in deeper water.

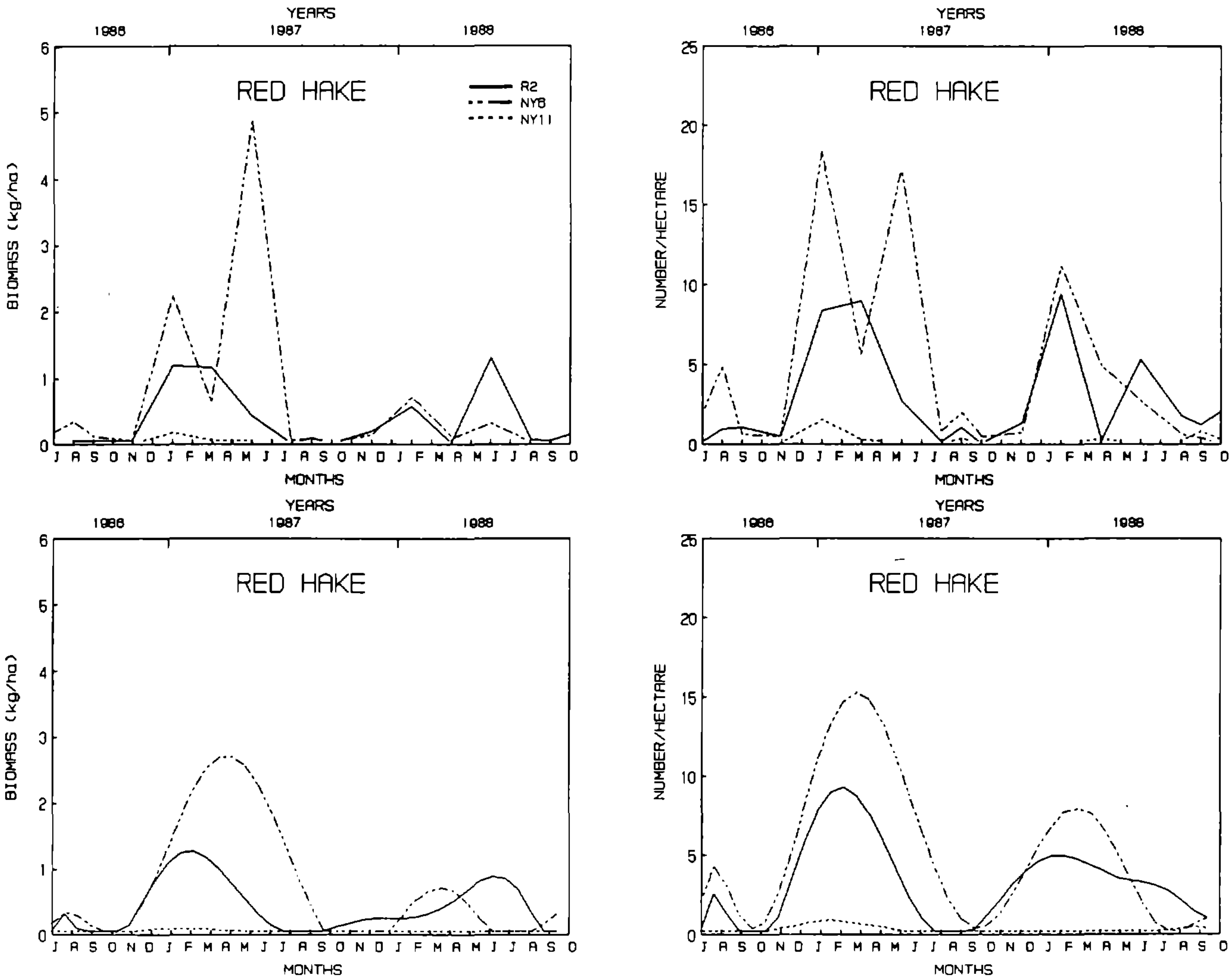

Figure 31. Trends in biomass as kilograms per hectare [(a) raw and (b) smoothed], and in numbers per hectare [(c) raw and (d) smoothed], for red hake, *Urophycis chuss*, collected at replicate stations from July 1986 to September 1988. The data were fitted to a ninth-order polynomial.

The collections from July to November from 1986 and 1987 are available for determining the range of variability of two comparable periods. Summary comparisons were made from the normalized catch (Table 8).

In terms of biomass from year to year, there is a high degree of consistency in both contributing species and relative abundance (Table 8). Biomass for the period varied as much as 100 percent at each station from one year to the next for fish, and in one instance, nearly 300 percent for invertebrates. For the late summer-autumn period available for comparison, the north-south differences are less dramatic in 1988 than in the 1987 annual summary (Fig. 37), but there still remains a bias for greater biomass at the enriched stations for both finfish and megainvertebrates. Variations in biomass up to ca. 10:1 have occurred in consecutive samplings during the early study phase. Although it is too early for a trend analysis, there is a suggestion of a shift in the fish-to-invertebrate catch ratios. During the phase-out of dumping, the portion of fish has increased while macroinvertebrates has decreased. Shifts in one or two dominant species (*i.e.*, spiny dogfish and little skate) can account for these differences.

Food Habits of Fish and Lobster

One aspect of the dumpsite study addresses the question of whether there would be a significant change in the stomach contents of several benthophagous fish species and lobster. This question was posed because the benthic macrofauna and toxic chemicals near the 12-mile site are quantitatively and qualitatively different from other adjacent areas, indicating an impact (Steimle 1985). Additionally, during the summer, hypoxic conditions are common in the same area. It was speculated that as the influence of sewage sludge dissipates, conditions would change and be reflected by alterations in food habits.

This report is a preliminary summary of the stomach-contents data for the period in which sewage sludge disposal was still occurring. No attempt is made to compare and correlate with other concurrent studies, *e.g.*, benthic community structure or habitat contamination, or with

Table 8. Summary comparisons of catch at the three replicate stations, R2, NY6, and NY11, for comparable summer-fall periods, 1986-87; weights in kilograms[1]

	R2		NY6		NY11	
	1986	1987	1986	1987	1986	1987
No. species						
Finfish	16	24	22	25	22	19
Shellfish	12	12	11	11	11	6
Rank as wt./tow (fish)						
1	LS	LS	LS	SpD	LS	LS
2	SpD	SpD	WF	LF	WF	WF
3	WF	WF	B	WF	B	SpD
4	B	B	SpD	C	FF	WS
Rank as no./tow						
1	B	B	B	B	B	LS
2	LS	LS	SH	LS	LS	B
3	WF	WF	LS	WF	S	S
4	SH	W	WF	W	WF	WF
Catch as wt./tow						
All fish	142.0	15.2	6.0	12.2	6.0	8.7
All invertebrates	5.4	7.8	8.0	4.2	3.4	2.0
Catch as no./tow						
All fish	53.4	62.9	49.8	53.5	41.1	30.6
All invertebrates	53.5	92.9	90.5	36.4	36.0	31.6

[1] **Species key: B = butterfish; C = cunner; FF = fourspot flounder; LS = little skate; S = scup; SpD = spiny dogfish; SH = silver hake; W = windowpane flounder; WF = winter flounder**

historical or other studies of the feeding habits of the predators considered here. This will be considered in subsequent reports and in a final study summary.

A summary of the results from the 11 surveys of this initial phase of the study, by predator and station, is presented (Table 9). (See Appendix for a description of the methodology.) Some preliminary observations follow.

At the enriched station, R2, the sample size per cruise was frequently less than adequate (less than 30) for all predators with the exception of winter flounder. Considering only this species, there did not appear to be any obvious trends in the results, although the relative stomach fullness index was highest (0.07-0.13) in the late fall-early spring. Despite conventional wisdom about this species fasting from November to March, (Klein-MacPhee 1978), the percent empty stomach values were actually lower (1-13 percent) during this period than during other times (10-39 percent) at this station. This could be a response to the slightly warmer (+1-2°C) bottom temperatures in the coastal study area during the winter than in estuaries, as suggested by Bowman (1981). Most winter flounder seasonal feeding studies have been reported for estuaries, *e.g.*, Klein-MacPhee (1978). As expected, the most frequently occurring prey varied, to some degree, with each predator. Some prey were almost exclusive to a predator while others were shared. The large polychaetes, *Pherusa affinis* and *Nephtys incisa*, were prominent in the diets of red hake and winter flounder. The decapod shrimp, *Crangon septemspinosus* and *Dichelopandalus leptocerus*, were prominent in the diets of both red and silver hake. Lobster and red hake ate *Cancer* crabs (mostly *C. irroratus*), with the hake utilizing mostly juveniles and the lobsters, larger crabs. Winter flounder also ate the burrowing anemone, *Ceriantheopsis americanus*, and another large polychaete, *Lumbrineris* spp. (mostly *L. acicularum*).

The station closest to the actual sludge disposal site, NY6, had adequate sample sizes intermittently for all predators. Empty stomachs appeared more frequently in late spring-summer for red hake and winter flounder and were variable for the other species. The relative stomach fullness index did not suggest any temporal pattern for any predator, other than winter flounder; this pattern was similar to that of Station R2, with a peak in the late fall-early spring. The major prey in this degraded area differed from those found at Station R2, with the exception of the isopod, *Edotea triloba*, and the tube-dwelling amphipod, *Unciola* sp., which were important in the diets of most predators (Table 9). The opportunistic polychaetes, *Asabellides*

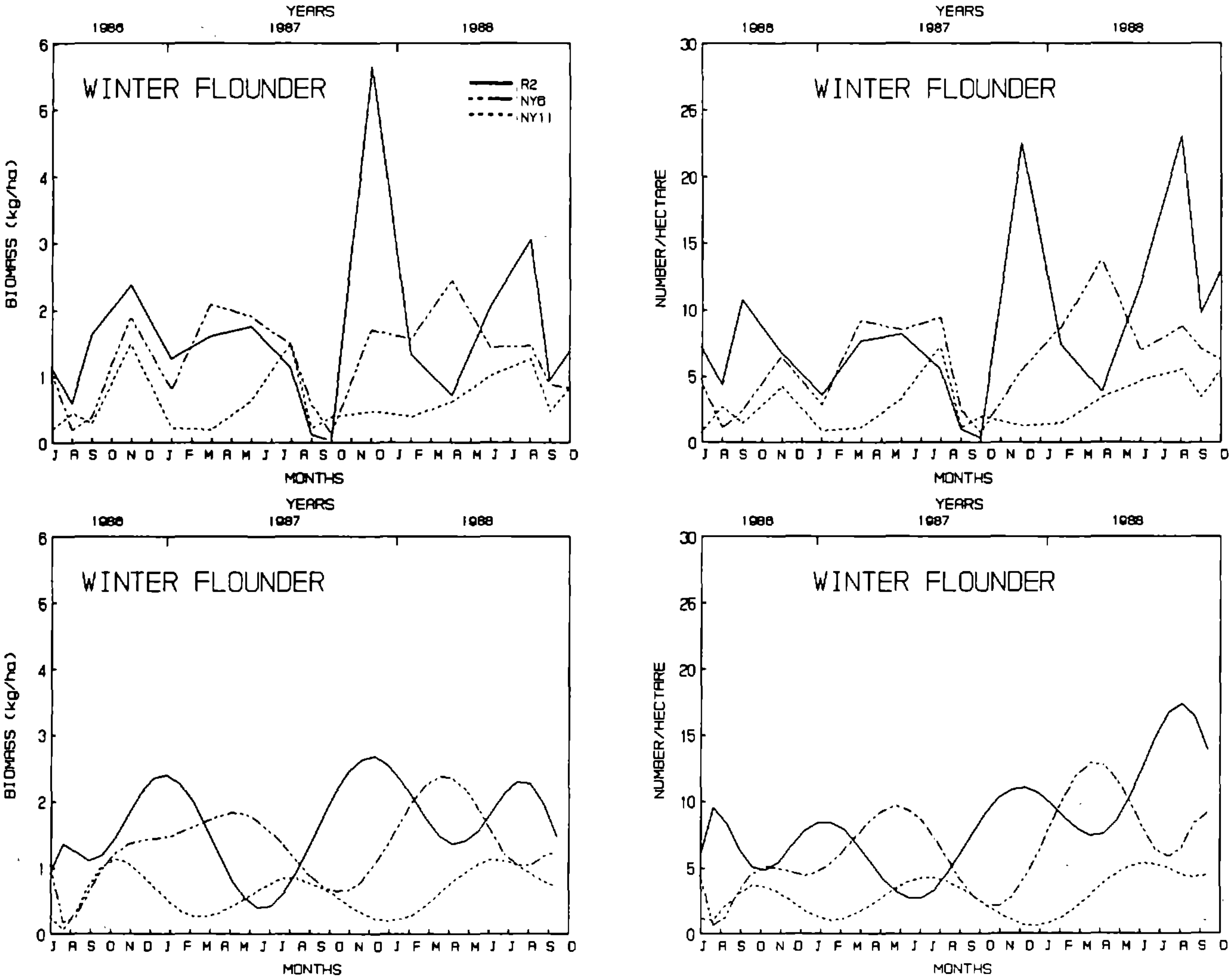

Figure 32. Trends in biomass as kilograms per hectare [(a) raw and (b) smoothed], and in numbers per hectare [(c) raw and (d) smoothed], for winter flounder, *Pseudopleuronectes americanus*, collected at replicate stations from July 1986 to September 1988. The data were fitted to a ninth-order polynomial.

oculata and *Capitella capitata*, were important to winter flounder and the small nut clam, *Nucula proxima*, to the lobster.

The southern reference station, NY11, also had less than adequate collections per cruise for all predators. The empty stomach factor was variable or inconclusive, except for winter flounder with the same temporal trend as noted in the other stations. The frequency-of-occurrence variable again supported no obvious trend for any predator. The principal prey were again similar, but reduced in relative diversity for red hake. The amphipod, *Leptocheirus pinguis,* appeared at NY11 as important prey for red hake and winter flounder.

A preliminary comparison of the overall stomach contents information for each predator per station (Table 9) suggests: (1) the percent-empty stomach parameter at NY6 was higher for red and silver hake, and variable to intermediate for winter flounder and lobster; (2) the relative fullness index suggested a slightly lower (probably insignificant) value at NY6 for red hake (however, the NY11 sample size was small) and little variation for winter flounder, lobster, and probably silver hake; and (3) in general, for all predators, the principal prey (based on frequency of occurrence) appear to be similar at all stations. For example, common prey at all three stations for red hake were *Pherusa affinis, Crangon septemspinosus, Nephtys incisa*, and *Dichelopandalus leptocerus,* and just for NY6 and R2, included juvenile *Cancer irroratus*. Common prey at the three stations for silver hake were predominantly *Crangon*, *Dichelopandalus*, and small fish; also mysid shrimp (mostly *Neomysis americanus*) at R2 and NY6. Winter flounder had a relatively diverse diet, but ate *Cerianthеopsis americanus*, rhynchocoels (mostly *Cerebratulus*), and *Pherusa* at all stations. Only capitellid polychaetes, a stress tolerant or opportunistic group, were special in the winter flounder diet at NY6 and commonly

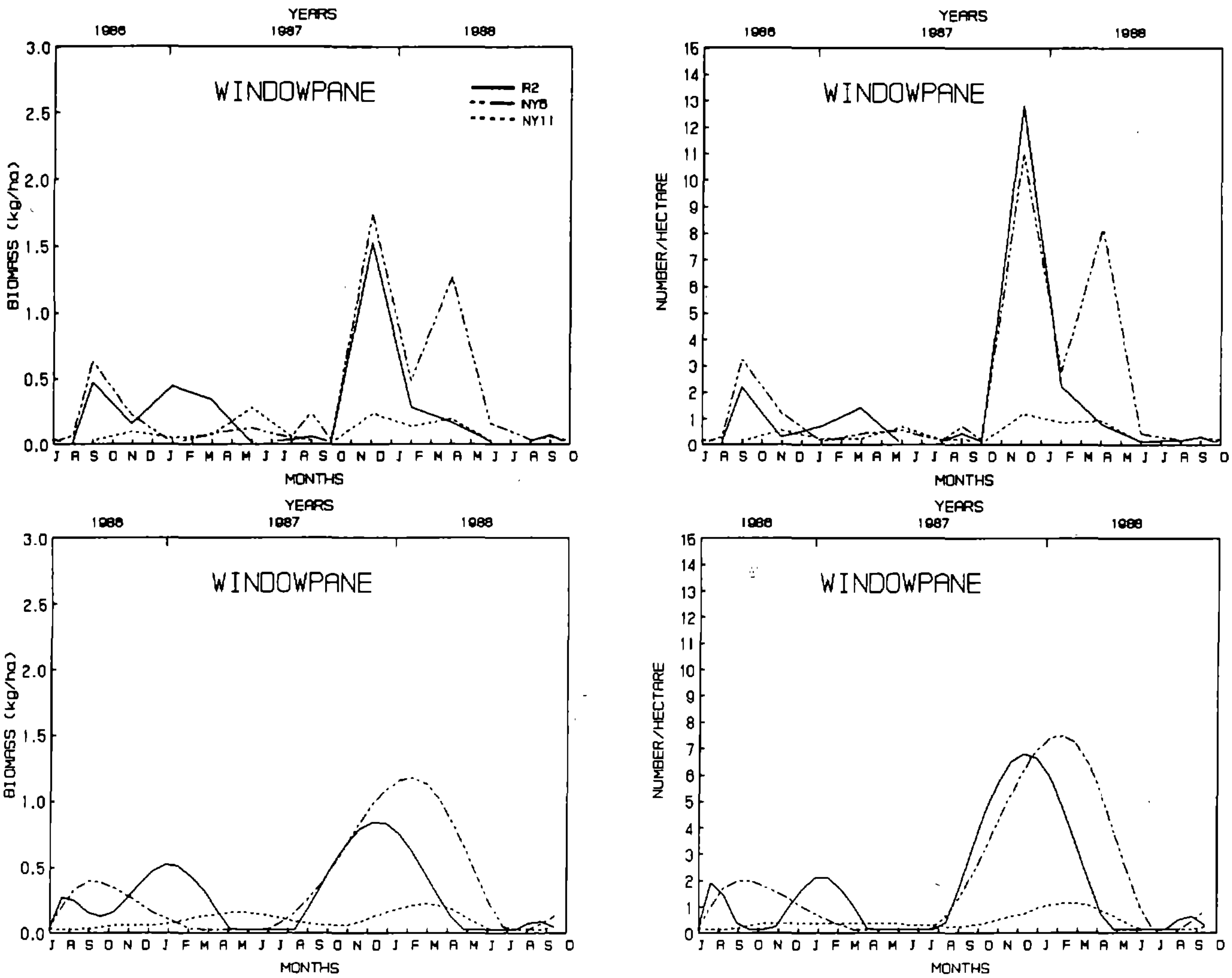

Figure 33. Trends in biomass as kilograms per hectare [(a) raw and (b) smoothed], and in numbers per hectare [(a) raw and (b) smoothed], for windowpane flounder, *Scophthalmus aquosus*, collected at replicate stations from July 1986 to September 1988. The data were fitted to a ninth-order polynomial.

consumed. The lobsters concentrated on larger *Cancer* crabs and fish at all sites.

These comparisons, although preliminary, suggest there may be only minor differences in the feeding variables examined among stations. A review of the sparse historical information on common benthic prey types since the late 1940s and the diets of some fish since the late 1950s also suggests little long-term change (Steimle *et al.*, in preparation). Thus, if sewage sludge disposal has had an effect of concern to fishery managers, it is not clearly manifested in benthic energy flow. This point was previously suggested by Steimle (1985) and supported by a directly measured benthic production study (Steimle *et al.*, in review). It is possible, but unlikely, that further, detailed analysis of the data might change this general conclusion. More information on gastro-intestinal evacuation rates, growth rates, and other benthic energy flow related variables in the apex would allow further examination of the validity of this preliminary conclusion. Also, as suggested by Steimle (1985), there is a serious need to understand benthic contaminant flow to fishery resource species and effects at the resource population level.

Fish and Lobster Pathology

Within the past decade, increasing numbers of marine fish and shellfish have displayed diseases attributed at least in part to pollution. Ulcers, fin rot, skin tumors, and skeletal anomalies have all been identified by the International Council for the Exploration of the Sea as appropriate for monitoring pollutant effects. The choice of diseases useful in pollutant-associated decision making considers both their seriousness and the likelihood of causation by pollution.

In the New York Bight region, fin rot has also been

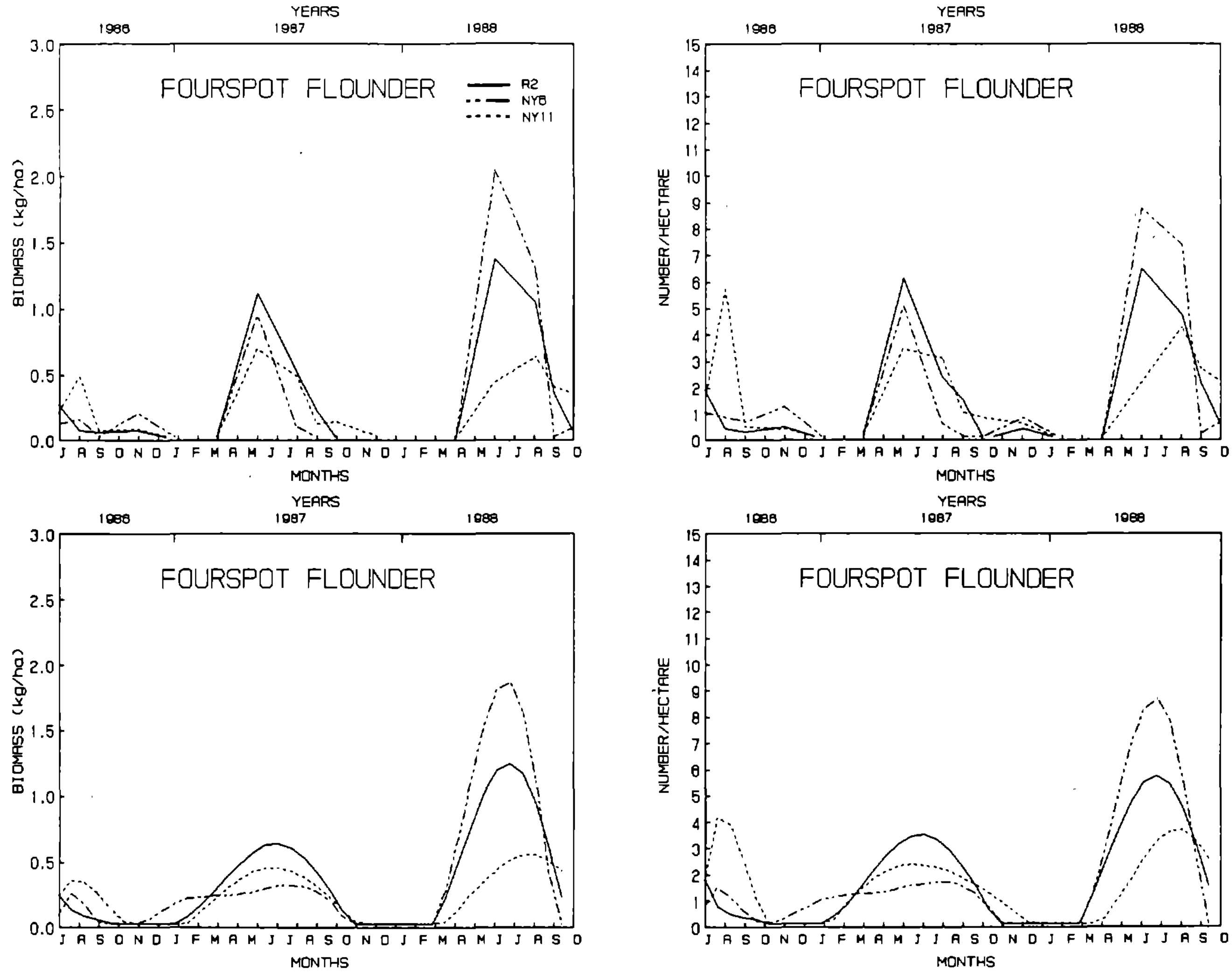

Figure 34. Trends in biomass as kilograms per hectare [(a) raw and (b) smoothed], and in numbers per hectare [(c) raw and (d) smoothed], for fourspot flounder, *Paralichthys oblongus*, collected at replicate stations from July 1986 to September 1988. The data were fitted to a ninth-order polynomial.

identified as a condition apparently caused by persistent exposure to toxicants in sediment and water. The characteristic progression of the disease involves death of fin tissue and fin erosion. In extreme cases, fins are completely eroded. Fin rot in winter flounder has been used to establish a pollution index (O'Connor *et al.* 1987) and presents a relatively easy and inexpensive measure of impact, and has the best long-term database of incidence. It should be noted, however, that observed incidence of fin rot has decreased since the early 1970s (Table 10). Lymphocystis and *Glugea* are also known from the area, but do not have a comparable database.

Results and Discussion

Pathology observations were recorded for winter flounder collected during the replicate cruises prior to closure of the dumpsite (July-December 1987). The fin rot incidence of 0.6 percent is similar to the level recorded in 1983 (Table 11). There is a suggestion that lymphocystis and *Glugea* have a higher incidence at R2 and NY6, the enriched and highly altered stations, as compared with reference area NY11. The occurrence of cysts, ulcers, tumors, bentfin and ambicoloration are of such low frequency that differences among stations are not discernable.

A total of 77 lobsters were examined for signs of shell disease from August to December 1987. Sixteen cases of chitinoclastic shell disease were identified. Disease incidence at specific stations where lobster were captured ranged from 0 to 33 percent, with an overall average occurrence of 20 percent for the study area. The incidence of shell disease appeared to be an issue of significant concern and a lobster tagging program has been initiated (December 1987) to provide needed field data in this area.

Migration of Winter Flounder

Since July 1986, winter flounder, *Pseudopleuronectes americanus*, have been tagged to determine the extent of movements between the dumpsite and surrounding inshore areas. A Peterson disc tag was applied to all winter flounder over 180 mm long captured during the broadscale

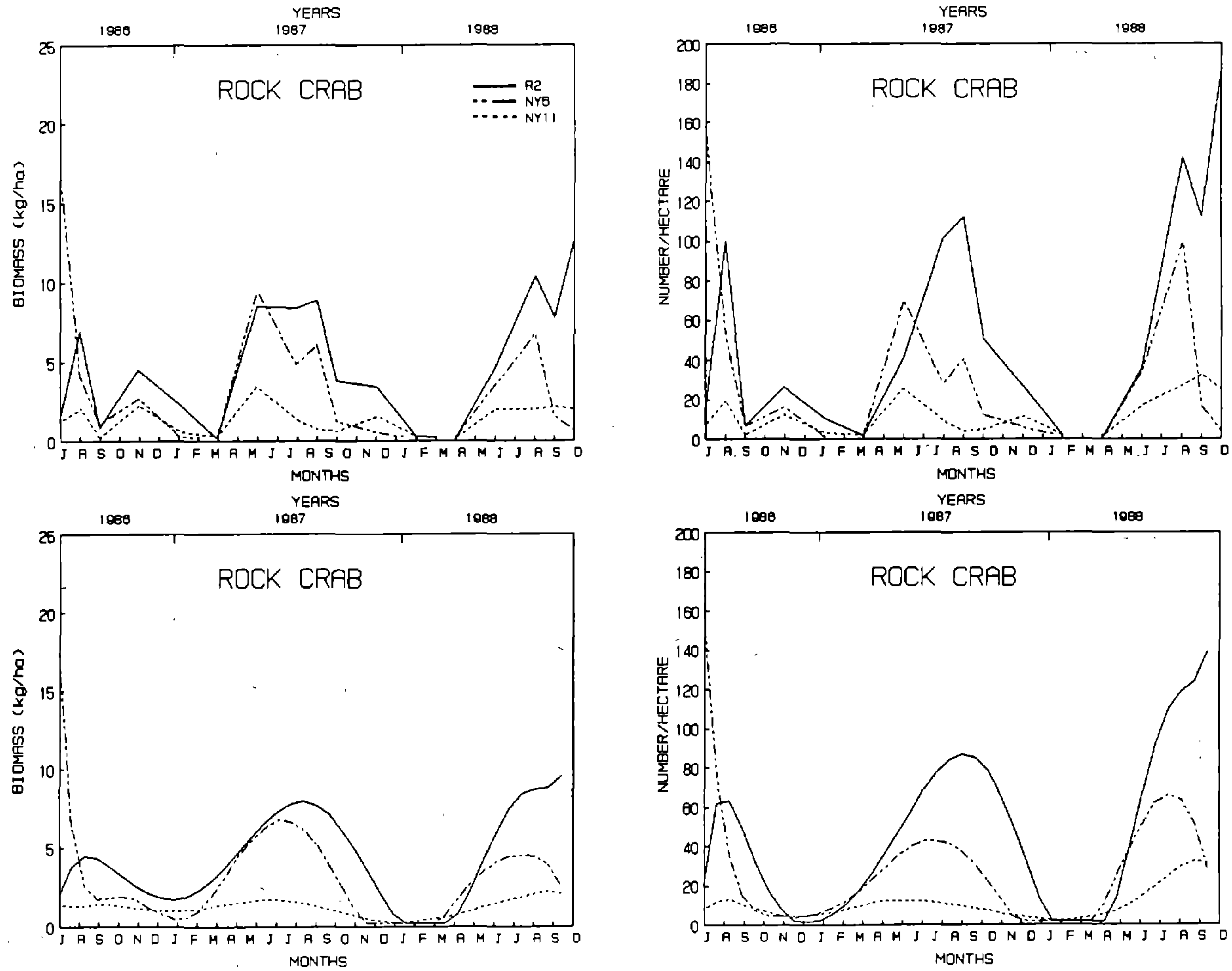

Figure 35. Trends in biomass as kilograms per hectare [(a)raw and (b) smoothed], and in numbers per hectare [(c) raw and (d) smoothed], for rock crab, *Cancer irroratus*, collected at replicate stations from July 1986 to September 1988. The data were fitted to a ninth-order polynomial.

survey. Fish were released immediately near the capture site. Fish were also tagged and released in bay and nearshore waters from special collections in those areas.

As of June 1988, 4223 winter flounder have been tagged (1596 dumpsite area fish and 2627 Sandy Hook Bay area fish). The 96 tags recovered to date result in a total return rate of 2.3 percent. Most returns (86.5 percent) were made by recreational fishermen, followed by research vessels (8.3 percent) and commercial vessels (5.2 percent).

Based on the number of returns, several trends are suggested:

1. Most returns were from inshore during the spawning season when winter flounder are concentrated and experience heavy fishing pressure by recreational fishermen. Most of these returns were represented by fish that were at large for less than 2 mo and recaptured within 4.0 km of the release site. However, one-fifth of these returns were at large a year or more and were also recaptured near the release point (Fig. 38). These recaptures indicate that some winter flounder return yearly to the same river systems to spawn.
2. Several returns indicate that many fish in this area follow the generally accepted patterns of migration, offshore into deeper, cooler waters in late spring followed by an inshore movement for spawning in early winter (Fig. 38). However, offshore movements may not be limited to deep ocean areas, as adult winter flounder are frequently found in the deep channels of estuaries during warm months. Lobell (1939) speculated that many winter flounder are "bay" fish, spending all their lives in estuarine systems, but moving into deeper waters as the shallows become too warm.
3. It appears that local populations within the New York Bight may not be discrete. Evidence of intermixing comes from five fish tagged in Sandy Hook Bay area which were subsequently recaptured in inshore Long Island waters (Fig. 38). Lobell (1939) first described intermixing between Long Island and New Jersey populations; however, later studies (Perlmutter 1947; Saila 1961; Howe and Coates 1975) concluded that winter flounder populations are geographically discrete.

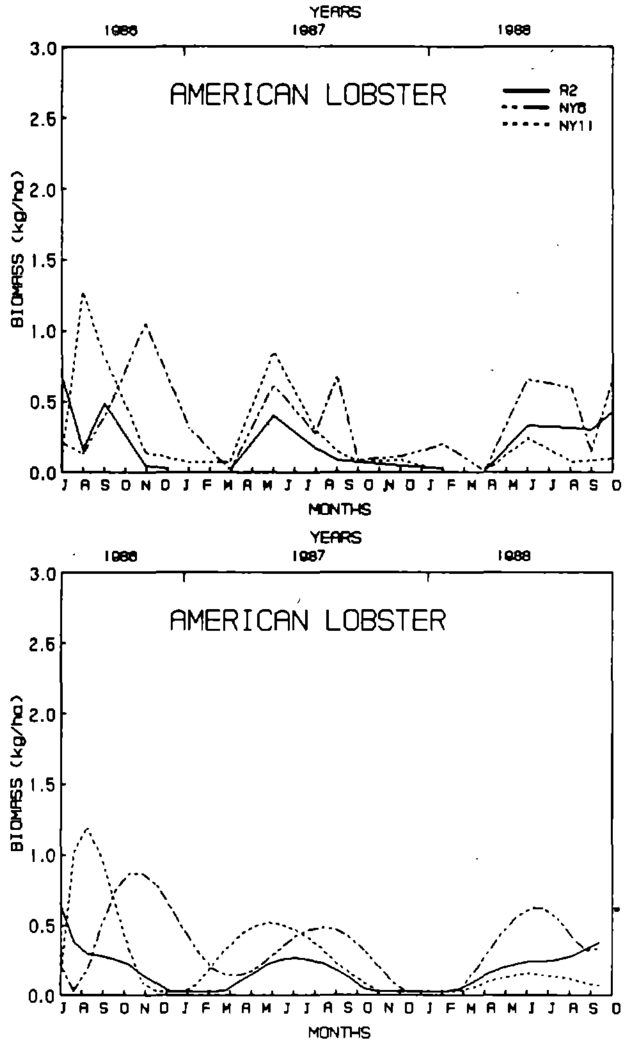

Figure 36. Trends in biomass as kilograms per hectare [(a) raw and (b) smoothed], and in numbers per hectare [(c) raw and (d) smoothed], for American lobster, *Homarus americanus*, collected at replicate stations from July 1986 to September 1988. The data were fitted to a ninth-order polynomial.

4. Five fish exhibited considerable movement beyond that associated with normal seasonal migrations (Fig. 39). While similar long-distance movements by winter flounder have been previously described (Lobell 1939; Scarlett and Schneider 1985; NUSCo 1986), the significance of these migrations requires further study.

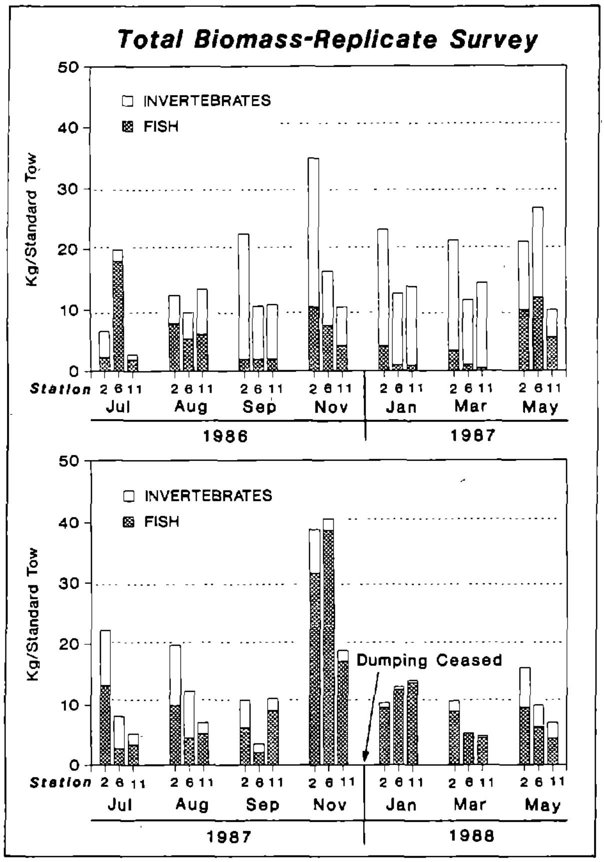

Figure 37. Mean biomass (kg/tow) of fishes and invertebrates trawled at the three replicate stations, R2, NY6, and NY11.

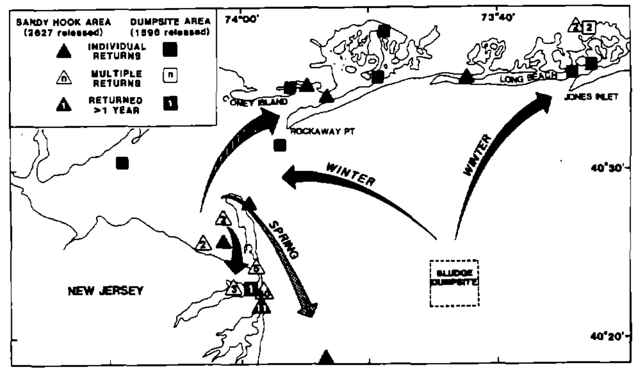

Figure 38. Winter flounder, *Pseudopleuronectes americanus*, tag returns indicating spring (stippled arrow), winter (solid arrow), and Long Island (striped arrow) migrations.

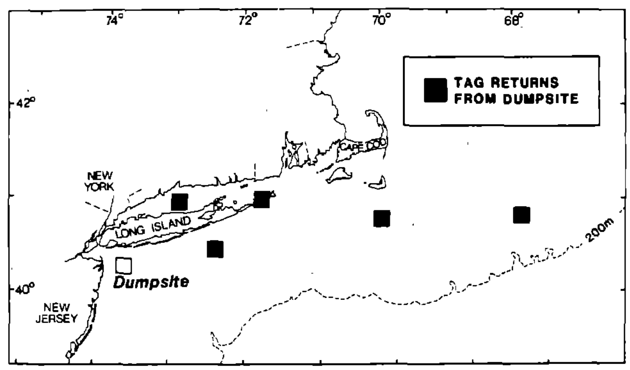

Figure 39. Winter flounder, *Pseudopleuronectes americanus*, tag returns showing long-distance recapture locations.

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Table 9. Comparisons of food habits data for winter flounder, red hake, silver hake, and lobster measured during the period when sewage sludge dumping was occurring

Winter Flounder		**Station**		
		R2	**NY6**	**NY11**
n		653	511	251
Percent empty		13	12	29
Relative fullness		0.07	0.06	0.06
Major prey	Type[1]			
Ceriantheopsis	A	X	X	X
Rhynchocoels	N	X	X	X
Lumbrineris sp.	P	X		X
Pherusa affinis	P	X	X	X
Nephtys incisa	P	X	X	
Capitella sp.	P		X	
Edotea triloba	C		X	X
Unciola sp.	C		X	X
Asabellides sp.	P			X
Leptocheirus	C			X
Echinarachnius	E			X

Red Hake		**Station**		
		R2	**NY6**	**NY11**
n		192	432	23
Percent empty		7	13	9
Relative fullness		0.08	0.04	0.09
Major prey	Type[1]			
Pherusa affinis	P	X	X	X
Nephtys incisa	P	X	X	X
Dichelopandalus	C	X	X	
Crangon	C	X	X	X
Cancer sp.	C	X		
Edotea triloba	C		X	
Unciola sp.	C		X	
Leptocheirus	C			X
Fish	V	X	X	

Silver Hake		**Station**		
		R2	**NY6**	**NY11**
n		194	453	119
Percent empty		24	55	29
Relative fullness		0.06	0.06	0.03[2]
Major prey	Type[1]			
Mysids	C	X	X	
Crangon	C	X	X	X
Dichelopandalus	C	X	X	X
Fish	V	X	X	X

Lobster		**Station**		
		R2	**NY6**	**NY11**
n		50	128	109
Percent empty		8	10	11
Relative fullness		0.11	0.11	0.09
Major prey	Type[1]			
Algae		X		
Cancer sp.	C	X	X	X
Fish	V	X	X	X
Nucula proxima	C		X	
Edotea triloba	C		X	

[1] **Types: P = polychaete; C = crustacean; M = mollusc; A = coelenterate; N = nemertean; E = echinoderm; V = vertebrate.**
[2] **Excludes one unusually high value.**

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Table 10. Summary of incidence of fin rot in winter flounder from the New York Bight apex (from O'Connor *et al.* 1987)

Year	Incidence of fin rot (%)	n
1973	13.4	1943
1974	6.1	570
1975	1.6	1637
1976	0.7	667
1977	3.2	1159
1978	2.0	2561
1979	--	--
1980	0.0	102
1981	1.6	314
1982	1.7	357
1983	0.4	241

Table 11. Summary of the pathological condition of winter flounder collected at the three replicate stations from July through December 1987

	Station			
	R2	NY6	NY11	%
Number observed	241	154	109	
Condition				
Fin rot	0	2	1	0.6
Ulcers	1	0	0	0.2
Cysts	11	9	15	6.9
Tumor	0	1	0	0.2
Lymphocystis	5	4	1	1.9
Glugea	11	5	2	3.6
Bentfin	6	3	4	2.6
Ambicoloration	2	0	4	1.2

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APPENDIX

SEDIMENT RESUSPENSION

Status

Cruises Completed in 1988

AL88-06 was a 7-day *Albatross IV* survey of erodibility, chemistry, and microbiology of sediments 6 mo after cessation of dumping. Transects were made of the inner bight, Hudson Shelf Valley (HSV), and Hudson Canyon.

Grab samples were taken for chemistry (metals, PCBs, and coprostanol), microbiology (*Clostridium perfringens* spores) and for erosion tests on board. Cores were subjected to physical stress using the Particle Entrainment Simulator and resulting resuspension was quantified as equilibrium suspended solids concentration as a function of shear.

All grab sampling and erodibility experiments were performed aboard the *Albatross IV*. Sediment chemistry will be performed by the EPA ERL-Narragansett Chemistry Branch and spore analyses will be performed by the FDA Laboratory (Davisville, Rhode Island).

Sample and Data Analysis

AL 87-03: All stress-resuspension data are processed (with the exception of entrainment rates); lower HSV cores are chemically analyzed; sediments for spore analyses were apparently lost after transfer to the EPA Edison Laboratory.

AL 88-06: All stress resuspension data are processed (with exception of entrainment rates) and stored on electronic media. Sediment samples collected for spores, chemistry, granulometry, and water content are frozen and await analyses.

METHODS OF FOOD HABITS ANALYSIS

The sampling stations used are those indicated as replicates in Figure 4. These stations were selected to include areas of similar habitat characteristics, except for organic carbon and toxic chemical levels. The Environmental Processes Division's Study Plan (Environmental Processes Division 1988) included the area generally considered to reflect the greatest effect of the sludge disposal (Station NY6), a site considered to be moderately affected or enriched (Station R2), and a site considered to be less polluted (Station NY11). Beginning in July 1986 and continuing every other month, but including August because of a tendency for hypoxia development, trawls sampled these three stations during the daylight hours (1000-1500) to collect the following predator species: red and silver hake, winter and yellowtail flounder, and the lobster. At each station per collection period, a total of eight replicate, 15-min trawls were made, distributed over an equidistant array of tow transects intersecting the center of the station and providing an equal number of morning and afternoon collections. These replicates were usually allocated so that only two tows were made at any station on any given day, to minimize the potential influence of the initial tow at the station on those that followed. All replicate trawl tows were normally completed within eight days, weather permitting. Before or after the tows at a station, bottom water and sediment samples were collected to provide environmental and benthic community data for each tow area.

The five predators being studied were selected because of their fishery value and expected abundance based on previous fish studies in the area, *e.g.*, National Marine Fisheries Service (1972) and Wilk *et al.* (1977). One predator, yellowtail flounder, did not occur as frequently or abundantly as expected after 16 mo of sampling, and was excluded from the analysis. The remaining four predators are known to be closely associated with the seabed, although with silver hake this association is strongest at the smaller sizes. Demersal species, such as these, are considered to be relatively stationary, *i.e.*, do not move far during the day, except during special seasonal or spawning migration periods. Thus, a major assumption of this study is that the items found in the stomachs of the predators collected on a station most probably were from feeding at or near the station. This is fully reasonable for species that feed primarily during the daylight, *e.g.*, winter flounder (Klein-MacPhee 1978; MacDonald *et al.* 1982), but becomes more questionable with species which feed nocturnally, such as silver hake, red hake, and lobster.

Upon collection, the fish predators were immediately segregated, measured, and usually had their stomachs removed and immediately examined. If there was a large catch, samples which could not be processed immediately were preserved in 10 percent buffered formalin for later analysis. A cut was made in the visceral cavity to allow the preservative to rapidly penetrate the stomach. All lobsters were preserved for later laboratory analysis because they chew their food into small fragments and closer analysis is necessary. Besides length, sex, and maturity state, internal and external observations of disease (reported elsewhere) were recorded. Lengths recorded were fork lengths for fish and both rostral and eye socket-thorax lengths for lobster.

To analyze stomach contents we used the semi-quantitative method, currently standard in the Northeast Fisheries Center (Langton *et al.* 1980). This is based on a visual, macroscopic examination of the stomach contents with volumes estimated by a direct comparison to a set volume of variable diameter of volume calibrated cylinders; volumes were also measured intermittently by water displacement in a graduated cylinder to estimate and correct any observer bias. Prey were mostly, easily discernable macroscopic common species that require only limited experience and training to identify without the aid of a microscope. If a substantial part of the diet of any predator examined consisted of small prey whose identity was questionable, a sample was collected and preserved for later identification in the laboratory. The estimated proportion (percentage) each prey made to the total stomach contents volume, a count of the number of each prey, mean prey size, and estimated digestive state were all noted, along with any special observations or comments. This approach, although requiring more subjective decisions about volume variables than we would like, has been used with relative success in the past (Larimore 1957; Hysop 1980). Bowman (1982) has demonstrated that the actual error introduced, when compared to doing a laborious, fully quantitative, laboratory, microscopic analysis, may be slight. The methods allow the development of a reasonably accurate feeding habits and diet information base, when personnel resources or time are limited. The entire study involved only two investigators so any biases were probably relatively constant.

The analysis of stomach contents data includes determination of: percent empty stomach, percent frequency of occurence of a prey or item, the relative fullness index, and most common prey. Other variables are available, *e.g.*, percent volume or count, but these are at least partially redundant in information value with the frequency-of-occurrence variable (MacDonald and Green 1983). The percent-empty stomach variable was examined to see if it can suggest feeding inhibition at any station; *i.e.*, a response to the low abundance of suitable prey at any station or presence of obnoxious environmental conditions. For example, Fletcher *et al.* (1981) reports reduced feeding rates for winter flounder exposed to freshly oiled sediments and Kramer (1987) discusses feeding responses of fish to hypoxia. The frequency-of-occurrence variable can provide insight to prey preference or availability. The relative fullness index is the product of the total stomach contents of volume divided by the predator length and can also provide insight into overall availability of suitable prey or feeding inhibition.

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