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Trends in Indicators of Eutrophication in Western Long Island Sound and the Hudson-Raritan Estuary

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ABSTRACT: Significant improvements in water quality have been observed for several decades throughout much of the Hudson-Raritan Estuary, primarily as a result of regional abatement of municipal and industrial discharges. These improvements include area-wide, order-of-magnitude reductions in ambient coliform concentrations and significant increases in dissolved oxygen (DO) concentrations. In contrast to these improvements, DO in bottom waters of the western Long Island Sound (WLIS) appears to have decreased in the last two decades. Although there is no consensus as to why hypoxia in WLIS may have recently become more severe, several related hypotheses have been suggested, including an increase in eutrophication, increased density stratification, and changes in wastewater loads. To determine if eutrophication has increased in WLIS, trends in several indicators of eutrophication were examined from a long-term water quality data set. Since the mid-1980s surface DO supersaturation has increased, bottom minimum DO has decreased, and vertical DO stratification has increased in WLIS. Other areas of the Hudson-Raritan Estuary, such as Jamaica Bay and Raritan Bay, exhibit similar evidence of declining water quality and may be experiencing increasing eutrophication. Temporal changes in vertical density stratification indicate that surface to bottom temperature differences have increased to a greater extent and have had a more significant impact on bottom DO depletion in WLIS than in the shallower Jamaica Bay and Raritan Bay. Additional factors contributing to the observed decline in water quality include recent changes in wastewater loads and possible increases in upstream and nonpoint source loads.

Introduction

Long-term monitoring of the lower Hudson-Raritan Estuary indicates that many areas of New York Harbor have recently experienced dramatic improvements in several conventional water quality indicators (O'Connor 1990; Suszkowski 1990; Parker and O'Reilly 1991; Brosnan and O'Shea 1996a,b; O'Shea and Brosnan 1997). Many of these improvements have occurred since the passage of the Clean Water Act in 1972, when most of the region's water pollution control plants (WPCP) were significantly upgraded and expanded, and additional plants were constructed. As a result of these efforts, untreated municipal wastewater loads into the Estuary were reduced from approximately 197 $m^3 s^{-1}$ in 1970 to less than 0.044 $m^3 s^{-1}$ by 1993 (Brosnan and O'Shea 1996a).

In combination with the abatement of untreated municipal discharges, the improved capture of combined sewer overflows (CSO), decreased industrial discharges, and product bans (e.g., phosphorus in detergent, tetraethyl lead in gasoline, PCBs, DDT, and chlordane) have resulted in decreased loadings of several pollutants. These abatement measures as well as examples of their impact on the environmental quality of the Hudson-Raritan Estuary over the past 26 years are summarized in Table 1.

Untreated municipal and industrial discharges to the Long Island Sound (LIS) (Fig. 1) have also declined significantly since the Clean Water Act of 1972 (Interstate Sanitation Commission 1970, 1997; U.S. Environmental Protection Agency [EPA] 1994). Although coliform bacteria concentrations in WLIS have experienced order-of-magnitude declines, dissolved oxygen (DO) concentrations in bottom waters have not improved, and analyses of data from different monitoring programs in the region suggest that the frequency, spatial, and temporal extent of hypoxia (DO < 3.0 mg l^{-1}) may have become significantly worse in the past 20–30 yr (Parker and O'Reilly 1991; Swanson et al. 1991; O'Shea and Brosnan 1997). This is despite the elimination of raw sewage discharges, and the dramatic decrease in biochemical oxygen demand (BOD) and total suspended solids (TSS) point source loads from all communities discharging to LIS.

The Long Island Sound Study (LISS), initiated

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TABLE 1. Recent pollutant loading abatements and resulting impacts in the Hudson-Raritan Estuary.

Abatement Measure	Impact
Construction and upgrade of water pollution control plants (WPCP); disinfection of treated effluent; reduced bypassing of untreated wastewater; abatement of illegal connections; improved capture of CSOs and floatables (Brosnan and Heckler 1996; Brosnan and O'Shea 1996a).	• Order-of-magnitude reductions in total and fecal coliform bacteria concentrations (Brosnan and O'Shea 1996a). • Reduced beach closings (Swanson and Bortman 1994). • Re-opening of long-closed New York City beaches (Brosnan and Heckler 1996). • 68,000 acres of shellfish beds upgraded (Pathogens Workgroup 1990; Gottholm et al. 1993).
Improved capture of TOC, BOD, and TSS as a result of regional WPCP construction and upgrades (Ayres et al. 1988; O'Connor 1990; Suszkowski 1990; Brosnan and O'Shea 1996a).	• Increasing DO concentrations and decreasing ambient concentrations of TOC, BOD, and TSS (O'Connor 1990; Suszkowski 1990; Brosnan and O'Shea 1996b). • Re-infestations of wood pilings by marine wood-borers (van Allen 1989; Gruson 1993).
Decreasing industrial discharge of As, Cd, Cu, Pb, Hg, Ni and Zn (Ayres and Rod 1986; Smith et al. 1987; Rod et al. 1989; Brosnan et al. 1994).	• A 50–90% reduction in Hudson River and Jamaica Bay sediment concentrations of most metals and toxic organic compounds (Bopp and Simpson 1989; Bopp et al. 1993; Chillrud 1996).
Product ban on the use of PCBs and the insecticides DDT and chlordane (Bopp and Simpson 1989).	• Decreasing PCB tissue concentrations in striped bass (New York State Department of Environmental Conservation 1988; McHugh et al. 1990; Hogan 1995). • Relaxed fish consumption advisories (New York State Department of Health 1995). • Re-establishment of breeding populations of peregrine falcons, ospreys, herons, egrets, and other wading birds (West-Valle et al. 1991).

in 1988 as part of the National Estuary Program, has identified hypoxia as the most significant water quality problem affecting LIS (U.S. EPA 1994). Since 1986, hypoxia in bottom waters has been documented by the LISS and other monitoring efforts, affecting on average of c. 400 km^2, and persisting from 2 to 11 wk each year (Welsh et al. 1994; U.S. EPA 1998). Associated studies by the Connecticut Department of Environmental Protection (CT DEP) and others have documented the negative impact of these reduced DO levels on both the abundance and diversity of living marine resources in LIS (McEnroe 1992; Miller et al. 1992; Woodhead and McEnroe 1992; Howell and Simpson 1994). Recommended management plans for the point-source control of nutrients in LIS are estimated to cost approximately 650 million U.S. dollars (U.S. EPA 1998).

The New York/New Jersey Harbor Estuary Program, part of the National Estuary Program since 1988, similarly recognizes that hypoxia and novel algal blooms are significant impairments to the ecological health of parts of the Harbor. Areas of concern for excess eutrophication include Jamaica Bay, Raritan Bay, Sandy Hook, and the New York Bight Apex (U.S. EPA 1996).

This paper examines whether the apparent recent decline of DO in bottom waters of WLIS is associated with trends of any other indicators of eutrophication, including surface DO supersaturation, vertical DO stratification, nutrients, and Secchi transparency. To determine whether a regional change in indicators of eutrophication is occurring, trends in these indicators are also examined in two other eutrophic areas of the estuary: Jamaica Bay and Raritan Bay. Trends in both water quality and factors that may be contributing to declining water quality are presented. The source of most of the data for this analysis is the New York City Department of Environmental Protection's (NYC DEP) long-term water quality monitoring program, the Harbor Survey Program (O'Shea and Brosnan 1997).

Study Site

The Hudson-Raritan System, which includes New York Harbor, is a varied and complex coastal plain estuary. The system is dominated by a drowned river valley (the Hudson), with a network of tidal straits (Arthur Kill, Kill Van Kull, and the Harlem and East Rivers), open and enclosed bays (Raritan, Jamaica, and New York Bays), tidal mud flats, and beaches (Fig. 1). The estuary communicates with the Atlantic Ocean through The Race at the eastern end of LIS, and through the mouth of Lower New York Bay at Sandy Hook (Swanson et al. 1982); the tides are semi-diurnal throughout.

The Hudson River constitutes the largest single freshwater input into the Hudson-Raritan estuary, draining 34,600 km^2 of a total watershed of 42,188 km^2. Land use in the Hudson River basin is c. 62% forested, 25% agricultural, 8% urban, 2.6% open water, and 2.4% classified as other (Wall et al. 1998). The Hudson's average flow of 683 m^3 s^{-1} at

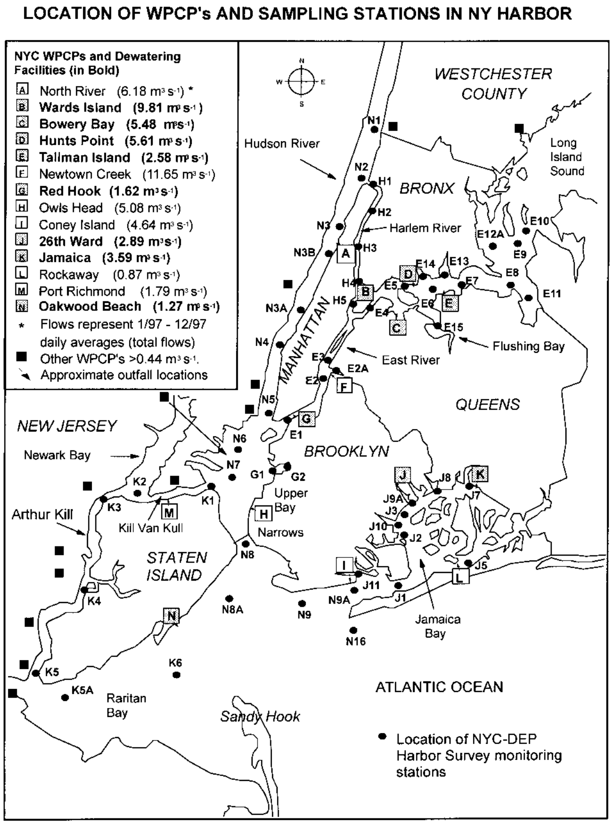

Fig. 1. Location of New York City Department of Environmental Protection's Harbor Survey Water Quality Monitoring Stations. Also depicted are the 14 New York City Water Pollution Control Plants and the 8 New York City sludge dewatering facilities (shaded boxes). Sandy Hook, lower center, is located at approximately 40°30′N, 74°W.

the Battery (the southernmost tip of Manhattan) constitutes approximately 87% of the total riverine flow into the estuary (Mueller et al. 1982). The Raritan, Passaic, and Hackensack Rivers in New Jersey drain most of the remaining watershed. The 7-d, 10-yr low flow (7Q10) and the 100-yr flood flow for the Hudson-Raritan system have been estimated at 82 $m^3 s^{-1}$ and 5,760 $m^3 s^{-1}$, respectively (National Oceanographic and Atmospheric Administration [NOAA] 1985). The volume of water entering the coastal system during the flood tide (i.e., the tidal prism) is approximately 7.49×10^9 m^3; the volume of water introduced from the tributaries is small (i.e., by a factor of 200 on a volumetric annual basis) compared to the amount introduced from tidal action, and tides dominate the density-driven estuarine circulation (NOAA 1985).

Jamaica Bay, part of Gateway National Recreation Area, is a semi-enclosed coastal embayment of 64.7 km^2 characterized by salt marsh islands in a tidal lagoon (Fig. 1). Jamaica Bay's present man-made watershed includes 148 km^2 of the boroughs of Brooklyn and Queens in New York City that drain to the bay through 41 CSOs, numerous storm sewers, and groundwater seepage from the Brooklyn/Queens aquifer. Six WPCPs discharging approximately 12.2 $m^3 s^{-1}$ of treated effluent to the Bay are the largest source of freshwater (Interstate Sanitation Commission 1997). An additional 4.2 $m^3 s^{-1}$ of flow is estimated to be discharged to the Bay from CSOs (2.0 $m^3 s^{-1}$) and storm sewers (2.2 $m^3 s^{-1}$) (O'Brien and Gere Engineers, Inc. 1993). Significant bathymetric alterations that have resulted from 80 years of dredging, filling, channelizing, bulkheading, and stormwater and wastewater collection activities have led to increased mean depths (1 to 5 m), reduced flushing (from 11 to 35 d), the obstruction of natural circulation, the destruction of peripheral marshland, and an increase in freshwater flow and its intensity (Swanson et al. 1982; West-Valle et al. 1991; Squires 1992).

Raritan Bay, in the lower Harbor, is an open bay of 81 km^2 that receives riverine flow from the Hudson River, primarily through the northeast end of the Bay, the Raritan River in the western end of the Bay, and the Hackensack and Passaic Rivers through the Arthur Kill tidal strait (NOAA 1988). As with Jamaica Bay, dredging, filling, channelization, and dumping activities have altered the natural bathymetry of Raritan Bay; mean depth has increased to 6 m and freshwater replacement times are reported to be 16–21 d (NOAA 1988; MacKenzie 1990). For both Raritan Bay and Jamaica Bay, no significant alterations to bathymetry have occurred since c. 1970s (NOAA 1988; MacKenzie 1990).

The East River is a 26 km long tidal strait that connects the Upper Harbor waters to the adjoining LIS. The residence time of the Lower and Upper East River combined is 1.25 d (Jay and Bowman 1975). Daily tidal excursions range from 16–21 km and average 70% of the East River's length. Tidal currents are driven by head differences at the two ends of the strait, with velocities exceeding five knots on the ebb tide at Hell Gate. Six WPCPs (36.75 $m^3 s^{-1}$) and numerous CSO and storm sewers discharge into this strait.

Long Island Sound is 93 km long and 34 km at its widest point with a total area of 3,390 km^2 (U.S. EPA 1994). Based on bathymetry and shoreline characteristics, LIS can be divided into several basins, of which the western Narrows and the upper and lower East River are here considered part of New York Harbor. Freshwater flow in LIS's drainage area is dominated by the Connecticut River, which discharges into LIS's Central Basin and drains approximately 65% of the 45,100 km^2 drainage area that extends as far north as Canada and also includes portions of New York City, and

TABLE 2. Relative contribution of major sources of flow and pollutant loadings (fecal coliforms, biochemical oxygen demand [BOD], total suspended solids [TSS], nitrogen, and phosphorus) into New York Harbor in percent, based on data from the late 1980s (HydroQual, Inc. 1991). WPCP = water pollution control plants; CSO = combined sewer overflows.

	Tributaries	WPCPs	CSOs	Storm Water	Total
Flow	81	15	1	4	765 $m^3\ s^{-1}$
Fecal Coliform	2	<0.1	89	9	$2.1 \times 10^{16}\ d^{-1}$
BOD	16	58	19	5	5.7×10^5kg d^{-1}
TSS	80	11	5	3	2.4×10^6kg d^{-1}
Nitrogen	29	63	2	2	2.8×10^5kg d^{-1}
Phosphorus	16	75	4	4	2.3×10^4kg d^{-1}

Westchester, Nassau, and Suffolk counties in New York State (U.S. EPA 1994). Exchange of water between LIS and New York Harbor through the East River has been estimated at 50 $m^3\ s^{-1}$ of near surface flow to LIS and 250 $m^3\ s^{-1}$ of near bottom flow to New York Harbor for a net flux of 200 $m^3\ s^{-1}$ toward New York Harbor (Blumberg et al. 1999).

New York Harbor's network of varying freshwater inflows, tides, and tidal flows produces vertical density stratification in some areas (the Hudson River and upper New York Bay) with seasonal and tidal variations in stratification. Average summer salinities range from 10–15 psu in the surface waters of the lower Hudson River to 15–25 psu in its bottom waters and 25–30 psu in the waters of the outer Harbor areas (O'Shea and Brosnan 1997). Vertical temperature stratification is normally less than 2°C. Because the difference between surface and bottom salinities in the estuary is generally less than 10 psu, the estuary is classified as moderately stratified and partially mixed (NOAA 1985; Clark et al. 1992). While water column stratification in the Harbor's outer bay areas is generally moderate, it is more pronounced than in the well-mixed tidal straits, e.g., the East River and Arthur Kill (Fig. 1).

On an estuary-wide basis, the relative importance of pollutant loadings associated with common point and nonpoint sources are provided in Table 2. Municipal point sources constitute approximately 14.9% of the average freshwater inflow to the Harbor, approximately 60% of the nitrogen and BOD loading, and 75% of the phosphorus loading to the Estuary (HydroQual, Inc. 1991). Presently, New York City processes 65.3 $m^3\ s^{-1}$ of domestic sewage. Thirteen of its 14 WPCPs, representing 81% of this flow, currently provide secondary treatment; the remaining plant (Newtown Creek) provides advanced primary treatment: 64% BOD removal and 77% TSS removal (O'Shea and Brosnan 1997). Another c. 43.8 $m^3\ s^{-1}$ of effluent is discharged into the Hudson-Raritan Estuary from 35 other WPCPs in the coastal counties of New Jersey and New York, and an additional c. 9.4 $m^3\ s^{-1}$ is discharged into LIS from 39 WPCPs in coastal counties of New York and Connecticut (HydroQual, Inc. 1996; Interstate Sanitation Commission 1997; Blumberg et al. 1999).

On a per unit area or volume basis, annual inputs of nutrients into New York Harbor have long been recognized as among the highest of any estuary in the country (Nixon and Pilson 1983; Jaworski et al. 1997). Although municipal point source loadings are the primary source of nutrients to the Harbor (Table 2), other sources that may be important locally include tributaries, oceanic loads, regeneration from sediments, atmospheric deposition, CSO, and stormwater (HydroQual, Inc. 1991). In LIS, municipal point source loads were the single largest source of nitrogen (3.35×10^7 kg yr^{-1}); nonpoint loads and oceanic inputs together account for 5.71×10^7 kg yr^{-1} of nitrogen, or 63% of the total nitrogen (TN) load to LIS (U.S. EPA 1998).

Materials and Methods

Sample Collection

As part of the New York Harbor Water Quality Survey, NYC DEP personnel presently monitor 53 stations in the Hudson-Raritan Estuary for a variety of water quality indicators (Fig. 1). Sampling frequency was approximately once every other week, for a total of 8–12 samples collected per station, per summer (June through September). The number of stations monitored as part of the Harbor Survey Program has steadily increased over the years, from 12 in 1909, to 26 by 1917, to 40 by 1956. Twelve more stations were added in 1984, for a total of 52 stations monitored from 1984–1996 (Fig. 1). In recent years, water samples were collected from onboard the NYC DEP's *HSV Osprey,* an aluminum-hulled, approximately 17-m long, twin-engine diesel craft. Stations were located in the field by latitude-longitude and line-of-site from historical station descriptions (O'Shea and Brosnan 1997); the same captain was in charge of locating sites from 1968–1996. Water samples were collected from surface waters (1 m below the water surface) and bottom waters (1 m above the sediment surface) using a 3.2-l Kemmerer sampler, or a 5.0-l Teflon-lined Niskin sampler. One chemist was in charge of sample analysis from 1968 until his retirement in 1988. Sampling and analyses since 1988 are consistent with prior years.

Dissolved Oxygen

New York City has been monitoring DO at various stations in New York Harbor since 1909. The frequency of summer (June–September) sampling at these sites has varied overall from five times or less during 1909–1930 to five for 1931–1950, six to nine for 1951–1970, twelve for 1970–1984, eight to ten for 1985–1989, and nine to fifteen for 1990–1999. In the first ten years of the survey, DO was measured using a procedure sometimes called the Albert Levy Method, in which DO depended upon the absorption of oxygen by ferrous sulphate in the presence of an alkali. The amount of ferrous sulphate not acted upon by the oxygen was then determined by acidifying and titrating with potassium permanganate (Metropolitan Sewerage Commission 1912). This method was certified at that time to produce results identical to the more familiar Winkler method; however, no information on split sampling efforts between this method and subsequent ones are available. From c. 1920 to 1984, the azide modification of the Winkler method was used. From 1985 through 1987, YSI (Yellow Springs Instruments Inc.) meters were also often used. These DO meters were calibrated and rechecked two to three times each sampling day using the azide modification of the Winkler method (American Public Health Association 1985). The Winkler method was once again used exclusively from 1988 through 1999. Duplicates were analyzed at one random station per day and reagents were changed at regular intervals.

Transparency

Secchi transparency has been recorded at each station 2–4 times per month since 1986 using a standard black and white quadrant Secchi disk, as per Standard Methods (American Public Health Association 1985). Measurements were recorded to the nearest half-foot (0.15 m).

Nutrients

From 1989–1999, the nutrients, dissolved ammonium-nitrogen (NH_4-N), dissolved nitrate and nitrite-nitrogen [(NO_3 +NO_2)-N], and total phosphorus (TP) were sampled two-four times per month. Dissolved orthophosphate (PO_4-P) sampling was conducted from 1989–1997. Total phosphorus samples were placed directly in sample-rinsed 125-ml polypropylene bottles, refrigerated at 4°C, acidified with sulfuric acid to pH < 2, and then placed in coolers for transport back to the Wards Island NYC DEP laboratory. For NH_4-N, (NO_3+NO_2)-N, and PO_4-P, 250-ml samples were immediately filtered on-board through pre-rinsed 0.45-μm membrane filters, placed in 250-ml bottles, refrigerated, preserved with sulfuric acid, and put in coolers for transport back to the lab. Prior to 1989, nutrient concentrations were measured using a Technicon Auto Analyzer II according to U.S. EPA's Methods for Chemical Analysis of Water and Wastes (U.S. EPA 1979). Since 1989, a TRAACS 800 using U.S. EPA approved Industrial Methods Nos. 780–86T (NH_4-N), 824–87T [(NO_3 +NO_2)-N], 787–86T (TP), and 781–86T (PO_4-P) has been used (Bran Luebbe, Inc. 1987a-c, 1988).

Quality Control

Daily quality control procedures for the above parameters were modeled after Standard Methods (American Public Health Association 1985). These included analysis of duplicate and blank samples, and comparison of results between analysts. Samples were analyzed at NYC DEP's Special Project Laboratory at Wards Island, New York, which is certified by the New York State Environmental Laboratory Approval Program (ELAP). Quality control charts and comparisons of Harbor Survey Program data with other monitoring programs are presented in an annual report (O'Shea and Brosnan 1997). Although NYC DEP has been collecting chlorophyll *a* (chl *a*) data since 1986, these data are currently undergoing a QC review by NYC DEP and so were not used in this analysis.

Trend Analyses

Temporal trends in DO, nutrients, and Secchi depth were determined by examining graphical presentations of these variables over time and quantified by estimating linear trends over time using OLS (ordinary least squares) regression methods. Linear trends were estimated by running Statistical Analysis System (SAS) procedure PROC REG or Statistica Pearson-Product Moment Correlations. The level of significance examined was $p < 0.05$. In SAS, Pearson correlation coefficients were obtained from procedure PROC COR.

Results and Discussion

Indicators of Eutrophication in the Hudson-Raritan Estuary

Eutrophication is a process where nutrients added to a body of water at excessive rates cause a dramatic increase in algal primary productivity and associated organic carbon. Coastal areas generally receive more nutrient inputs than other types of ecosystems, and are particularly threatened by eutrophication (Howarth 1993; Bricker and Stevenson 1996). Symptoms of eutrophication include elevated nutrient and phytoplankton concentrations, excessive macroalgal biomass, reduced transparency, and increased surface to bottom summer DO differences, i.e., surface DOs exhibiting supersaturation from excess algal photosynthesis, and bottom DOs depleted from the decay of algal organic matter.

The following sections examine the extent to which the recent decline in summer bottom DO in WLIS correlates with trends in other indicators

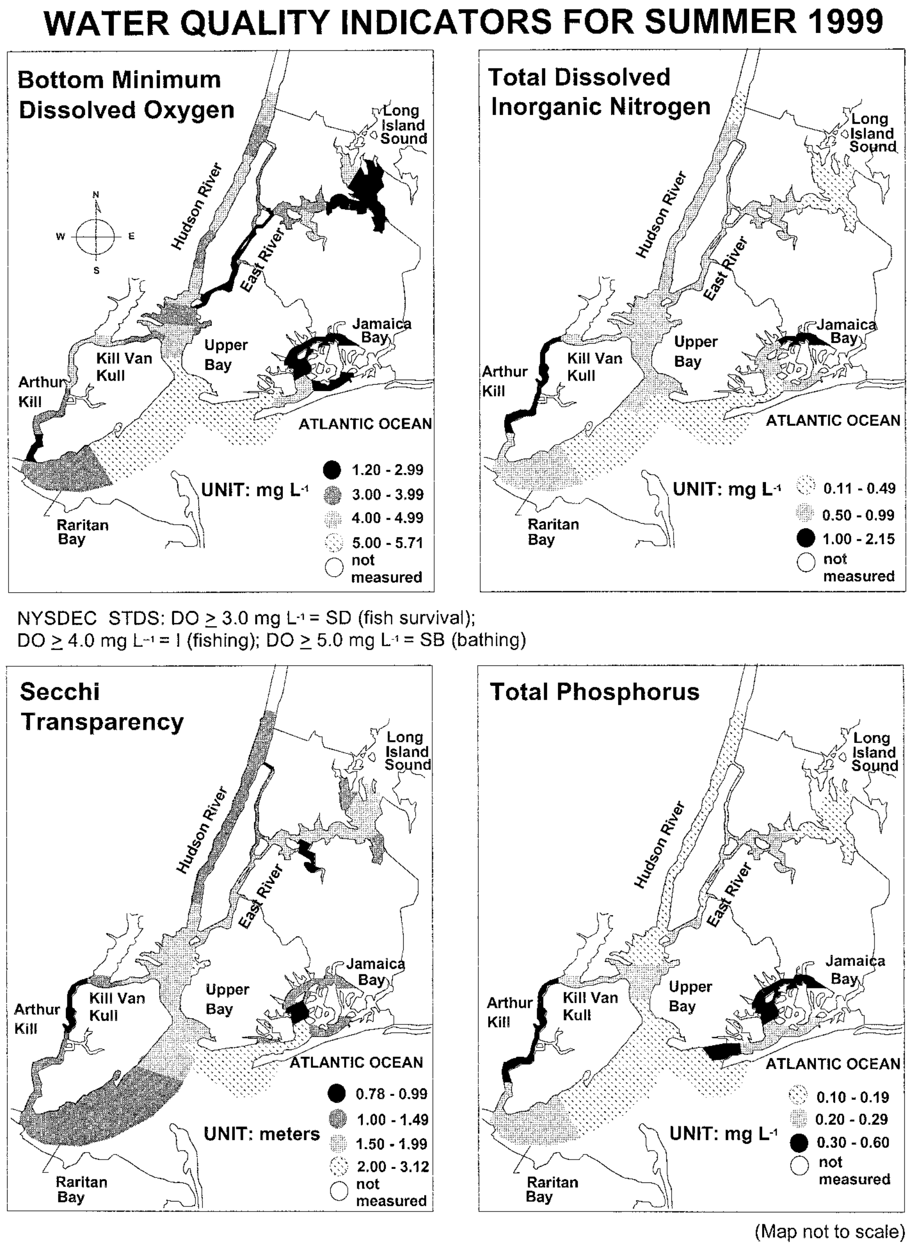

Fig. 2. Key water quality indicators for summer (June–September) of 1999. Depicted are: bottom minimum dissolved oxygen (upper left) in mg l^{-1}; summer average dissolved inorganic nitrogen [NH_3-N + (NO_3 + NO_2)-N] (upper right) in mg l^{-1}; summer average Secchi transparency (lower left) in m; summer average total phosphorus (lower right) in mg l^{-1}.

of eutrophication. In addition, trends in WLIS will be contrasted with trends observed in two other eutrophic areas of the estuary: Jamaica Bay and Raritan Bay (Fig. 1). Although the entire Hudson-Raritan estuary exhibits elevated dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorous (DIP) levels and limited light transparency, the lowest bottom summer DOs are typically recorded in these three areas (Fig. 2), along with Sandy Hook Bay in New Jersey (Brosnan and O'Shea 1996b; O'Shea and Brosnan 1997).

Dissolved Oxygen

Over the last several decades, DO has increased throughout the Hudson-Raritan Estuary in response to the abatement of untreated sewage discharges (O'Connor 1990; Brosnan and O'Shea 1996b). Trend analysis of long-term summer average monitoring data indicate that during 1968–1999, 73% of the 40 sites sampled continuously display increasing trends in summer (June–September) average bottom DO; for surface waters, 93% of sites display increasing trends. Trend analyses performed on raw, i.e., unaveraged, summer data indicate similar results. In contrast to these widespread improvements, two stations located in the WLIS (Table 3 stations E9 and E10; Fig. 3 station E9) exhibit a declining long-term trend in both summer average bottom DO and bottom minimum DO. Although these trends are not as significant as some of the improving trends observed elsewhere in the region (e.g., contrast trends between Hudson River and WLIS stations, Table 3), their significance has increased since Parker and O'Reilly's (1991) earlier analysis of the NYC DEP data set for WLIS. Areas of the Harbor that exhibit no significant trends include the bottom waters of Jamaica Bay and selected sites in Raritan Bay. Detailed discussion of DO trends for each water body of concern are presented below.

The East River and Western Long Island Sound. The abatement of untreated sewage discharges as a result of treatment plant construction, expansion, and upgrade activities starting in the 1930s, and later mandated by the 1972 Federal Clean Water Act, has incrementally reduced municipal BOD, TSS, and nutrient loadings to the East River and surrounding region (Interstate Sanitation Commission 1970, 1978, 1985, 1990–1997; Brosnan and O'Shea 1996a). For example, a comparison of typical East River point source loadings from the 1960s to 1990 indicates a decline in BOD, TSS, and nitrogen loadings of 61%, 64%, and 30%, respectively (Swanson et al. 1991).

In response to these decreasing loadings, DO has increased in the Lower and Upper East River (Brosnan and O'Shea 1996b). From 1915 to the early 1950s, average summer DO percent saturation typically fluctuated between 15–25% in the Lower East River, and between 20–40% in the Upper East River (Fig. 3 stations E5 and E2). Starting in the 1930s, surface and bottom DO saturation throughout the East River gradually rose from approximately 30% in the 1950s to over 60% in the 1990s. From 1968–1999, summer bottom DO minima increased in the East River by 0.04–0.06 mg l^{-1} yr^{-1} (Table 3) for a cumulative increase of 1.2–1.8 mg l^{-1} over the past 30 yr. Note that the East River is a well-mixed tidal strait that has not historically exhibited vertical density or DO stratification (Fig. 3 stations E5 and E2).

In contrast, average bottom summer DO in WLIS exhibited no improvement from 1920–1980 (Fig. 3 station E9). Beginning in the mid-1980s bottom DO decreased from approximately 70% saturation down to 50–60%, while average surface DO increased from approximately 70% to over 100%. Dissolved oxygen minima in the WLIS in the 1990s were typically less than 2 mg l^{-1} (Fig. 4 kms 25–30), while summer surface maxima typically exceeded 12 mg l^{-1}, implying increased algal production in these waters (O'Shea and Brosnan 1997). While periods of vertical DO stratification were observed intermittently in WLIS in the past, the period from 1985 through 1999 represents the longest and most intense degree of stratification observed since monitoring began in 1909 (Fig. 3 station E9). Since 1980, annual average DO stratification in WLIS has increased approximately 0.15 mg l^{-1} yr^{-1}, the largest increasing trend observed in the Harbor. From 1968–1999, while summer bottom DO minima increased in the East River by 0.04–0.06 mg l^{-1} yr^{-1} (Table 3 and Fig. 4 kms 0–20), it decreased in the WLIS by 0.06–0.07 mg l^{-1} yr^{-1} (Table 3 and Fig. 4 kms 25–30), for a cumulative decrease of approximately 2 mg l^{-1} over the past 30 yr.

Jamaica Bay. The long-term temporal pattern of summer DO in Jamaica Bay exhibits similarities to the East River and WLIS. Jamaica Bay's four largest WPCPs were constructed during the period of 1935–1952 (NYC DEP 1994). Three of the four WPCPs (representing 10.1 m^3 s^{-1} design flow) were upgraded to secondary treatment by 1979; the 4.4 m^3 s^{-1} Coney Island WPCP, at the mouth of Jamaica Bay, was upgraded to secondary treatment in 1994.

Dissolved oxygen improvements in response to the abatement of municipal discharges vary spatially throughout Jamaica Bay. Improvements to average surface and bottom DO percent saturation are much more pronounced among the northern shore stations (Figs. 1 and 5 stations J2, J3, and J7), rising from approximately 40–80% saturation in the 1930s to 80–110% saturation in the 1990s. Improvements along the southern shore (J5) and near the mouth (N9A and J1) are more modest, rising from 60–85% saturation in the 1930s to 90–100% saturation in the 1990s. Note that the counterclockwise circulation in Jamaica Bay is restricted considerably by islands, narrow channels, and other natural and man-made constrictions such that tidal currents and the flushing rate are considerably slower, and nutrient and chl *a* concentrations are significantly higher along the northern shore (O'Shea and Brosnan 1997). Similar to the WLIS, after 1980, vertical DO stratification along the northern shore (Fig. 1 stations J2, J3, J7, J8, and J10; Fig. 5 stations J2, J3, and J7) exhibits statistically significant increasing trends. In particular, as surface waters of northern Jamaica Bay exhibit increasing trends to supersaturated levels (Fig. 5 sta-

TABLE 3. Summertime (June–September) significant trends ($p < 0.05$) in: bottom average (DOB), bottom minimum (DOB min), surface average (DOT), and surface maximum (DOT max) dissolved oxygen in mg l^{-1} yr^{-1} (1968–1999); surface average dissolved ammonium-nitrogen (NH_4-N), surface average dissolved nitrite-nitrate-nitrogen [(NO_2 + NO_3)-N] and surface average total phosphorus (TP) in mg l^{-1} yr^{-1} (1989–1999); surface average dissolved orthophosphate-phosphorus (PO_4-P) (1989–1997); and average Secchi depth in m yr^{-1} (1986–1999). Trends indicative of declining water quality are in bold and nonsignificant trends are left blank.

		DOB		DOB min		DOT		DOT max		NH_4-N		(NO_2 + NO_3)-N		TP		PO_4-P		Secchi	
Branch	Site	Slope	r^2	Slope	r^2	Slope	r^2	Slope	r^2	Slope	r^2	Slope	r^2	Slope	r^2	Slope	r^2	Slope	r^2
Upper East River	E5	0.05	0.42	0.04	0.38	0.04	0.23							**4.1**	**0.38**			**−0.05**	**0.40**
	E6	0.06	0.61	0.06	0.47	0.07	0.51	0.09	0.28					**4.2**	**0.38**			**−0.04**	**0.35**
	E7	0.04	0.37			0.06	0.43	0.09	0.34	−17.4	0.68					−2.6	0.49		
	E8			**−0.04**	**0.19**	0.05	0.32	0.06	0.15	−11.6	0.45								
WLIS	E9	**−0.04**	**0.25**	**−0.07**	**0.34**	0.08	0.51	0.16	0.39	−17.4	0.54								
	E10	**−0.04**	**0.28**	**−0.06**	**0.26**	0.06	0.45	0.11	0.28	−6.7	0.42								
Hudson River	N2	0.06	0.38	0.09	0.59	0.07	0.48	0.08	0.15	−5.6	0.41								
	N3	0.08	0.53	0.10	0.67	0.08	0.57	0.11	0.34										
	N4	0.08	0.66	0.09	0.57	0.10	0.72	0.12	0.30										
	N5	0.09	0.73	0.08	0.57	0.09	0.68	0.10	0.42										
Arthur Kill	K3	0.15	0.85	0.16	0.84	0.14	0.82	0.10	0.66	−28.0	0.51								
	K4	0.13	0.83	0.12	0.74	0.13	0.78	0.13	0.66	−45.4	0.79								
	K5	0.05	0.48	0.04	0.18	0.08	0.67	0.14	0.35	−24.0	0.60								
Lower New York Bay	K5A	0.05	0.49	0.03	0.13	0.10	0.68	0.22	0.48	−21.1	0.41								
	K6	0.08	0.47	0.08	0.45	0.09	0.48												
	N16																		
	N9	0.02	0.14	0.03	0.14	0.02	0.16												
	N8	0.07	0.75	0.10	0.61	0.08	0.63	0.09	0.35	−8.0	0.45	−6.1	0.45						
Jamaica Bay	J1																		
	J10	0.09*	0.31*									**9.9**	**0.45**	**8.5**	**0.49**				
	J2					0.05	0.40	0.10	0.27					**9.0**	**0.50**			**−0.04**	**0.48**
	J3			**−0.03**	**0.14**	0.05	0.42	0.13	0.35			**12.0**	**0.44**	**9.5**	**0.49**	**5.3**	**0.80**	**−0.04**	**0.62**
	J5			**−0.04**	**0.16**			0.08	0.18									**−0.04**	**0.54**
	J7					0.08	0.43					**9.6**	**0.65**					**−0.03**	**0.52**
	J8					0.12*	0.31*			−19.6	0.37	**12.3**	**0.54**					**−0.04**	**0.56**
	N9A									−9.3	0.37								

* Dissolved oxygen trends for J10 and J8 are from 1984–1999.

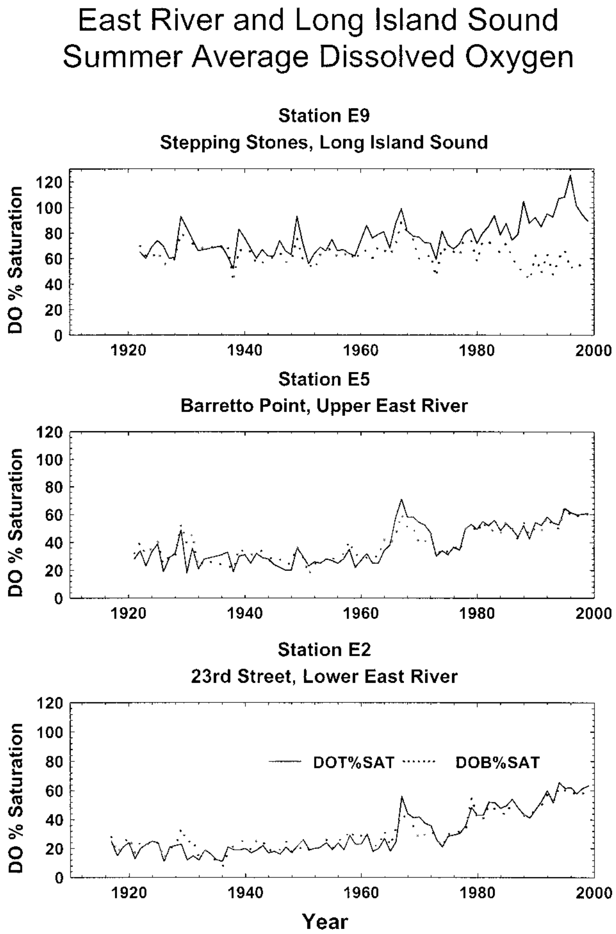

Fig. 3. Comparison of summertime (June–September) average surface (DOT% SAT) and bottom (BOB% SAT) percent saturation of dissolved oxygen (DO) for western Long Island Sound station E10 and East River stations E5 and E2: 1909–1999.

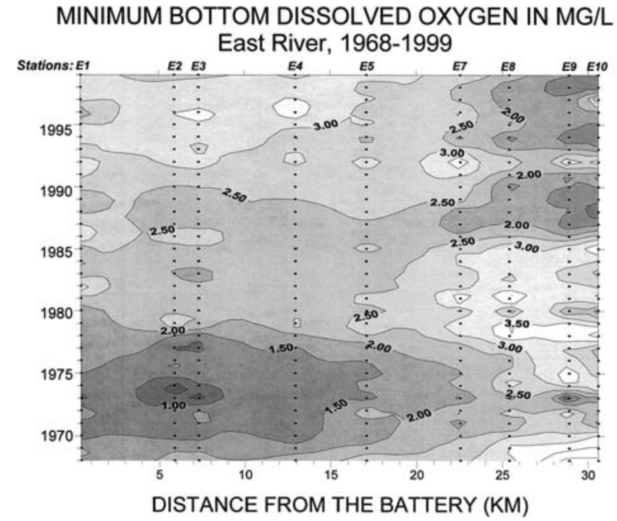

Fig. 4. Spatial and temporal plot indicating summertime (June–September) minimum bottom water summer dissolved oxygen (mg l^{-1}) in the East River and western Long Island Sound from 1968–1999. The x-axis indicates the kilometers from the Battery (the southernmost tip of Manhattan), as well as the locations of New York City Harbor Survey Water Quality Monitoring Stations in the East River and western Long Island Sound.

tions J2, J3, and J7), bottom DOs have not exhibited any significant increasing trend over the past 30 years and bottom average and minimum values of some stations indicate a declining trend (Table 3 stations J3, J5). In contrast, stations along the southern shore and at the mouth of Jamaica Bay (Fig. 5 stations N9A, J1, and J5) do not indicate any increase in vertical DO stratification, with both surface and bottom waters approaching 100% saturation.

The Arthur Kill and Raritan Bay. Dissolved oxygen in the Arthur Kill (a poorly flushed tidal strait connecting Newark Bay and Raritan Bay) and Raritan Bay has also increased following the abatement of municipal discharges, especially after the 1972 Clean Water Act. Improvements are most dramatic in the Arthur Kill, where DOs increased from less than 20–40% saturation during the 1940s into the early 1970s to over 60–80% saturation in the 1990s (Figs. 1 and 6 stations K4 and K5). Over the last 30 years, DO bottom minima in both waterways have increased by 0.03 to 0.16 mg l^{-1} yr^{-1}, depending on the station (Table 3). Neither vertical DO stratification nor supersaturation are evident in the historical record of the Arthur Kill.

Surface DO at the head of Raritan Bay (near the confluence of the Raritan River) has increased significantly from approximately 60% saturation in the early 1970s to over 100% saturation in the 1990s (Figs. 1 and 6 station K5A). Although bottom average DOs have also increased at statistically significant rates (0.05 mg l^{-1} yr^{-1}), surface average and surface maximum DO concentrations trends are some of the highest in the Harbor and the area exhibits increasing DO stratification (Table 3 and Fig. 6). The open waters of Raritan Bay (Fig. 1 station K6) have historically exhibited a degree of vertical DO stratification and supersaturation and the data indicate no trend in either surface maximum or DO stratification (Table 3); however, surface waters since the mid-1980s now routinely exceed an average of 120% saturation (Fig. 6 station K6).

Chlorophyll a and Phytoplankton

Algal biomass is highest in the outer Harbor areas, including Raritan Bay, Jamaica Bay, and WLIS where summer average chl *a* concentrations often exceed 30 $\mu g\ l^{-1}$ (Brosnan 1991; Cosper and Cer-

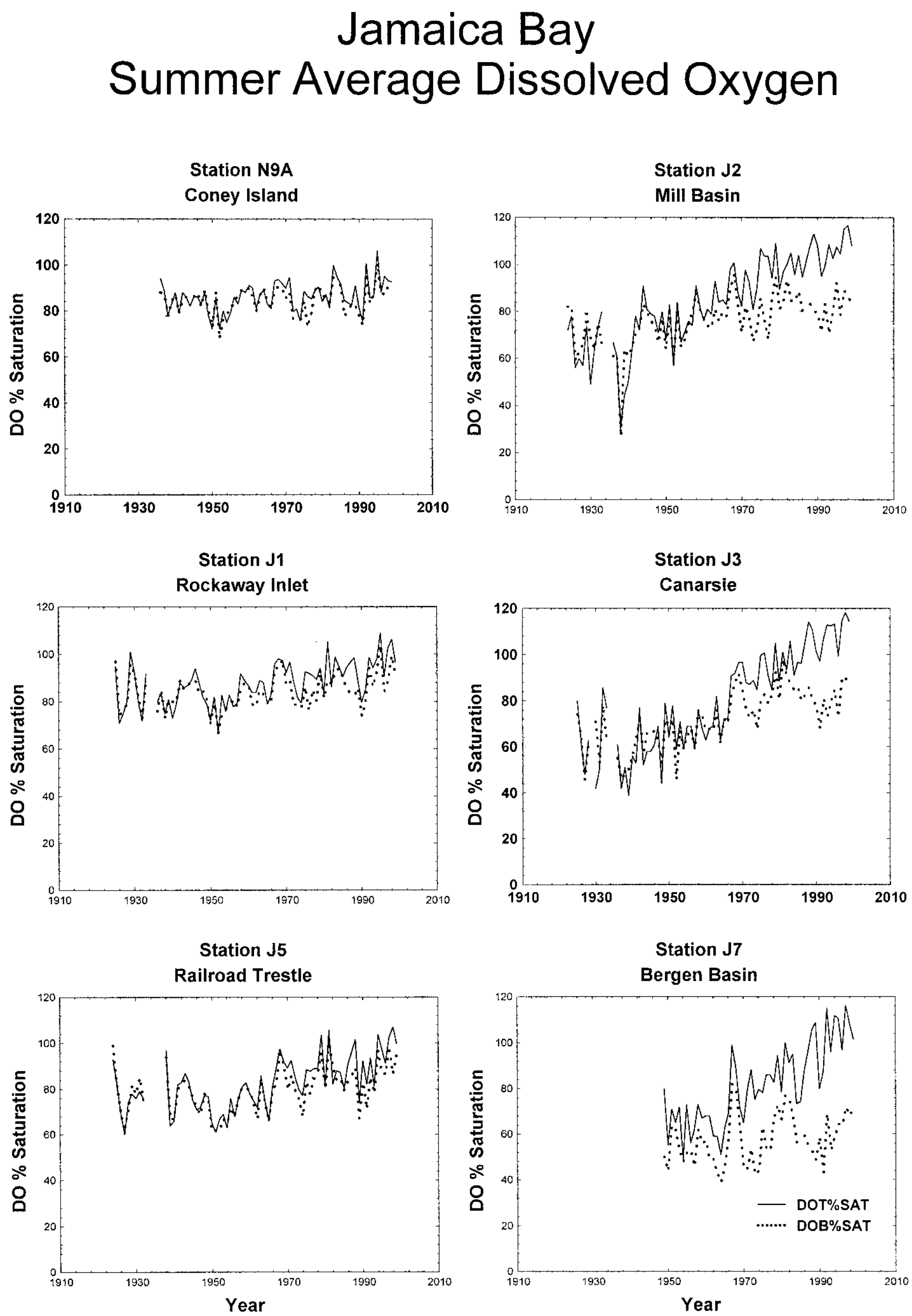

Fig. 5. Summertime (June–September) average dissolved oxygen percent saturation for surface (DOT%SAT) and bottom waters (DOB%SAT) in Jamaica Bay, 1909–1999. Stations on the southern shore are on the left, while the northern shore stations are on the right.

ami 1996). Monthly average summer cell counts of phytoplankton range from 5,200–85,000 cells ml^{-1} harborwide and follow a geographic distribution similar to that observed for chl *a* (Brosnan 1991; O'Shea and Brosnan 1997). In the WLIS, Jamaica Bay, and Raritan Bay, winter/spring and summer/fall blooms are primarily dominated by the diatom *Skelotonema costatum,* and the green alga *Nannochloris atomus* (O'Shea and Brosnan 1997). When NYC DEP's phytoplankton cell counts are converted to phytoplankton carbon, the dominant genera, on a biomass basis, include *Thalassiosira, Rhizosolenia,* and *Eucampia* (Cosper and Cerami 1996). Comparisons of phytoplankton species identified since

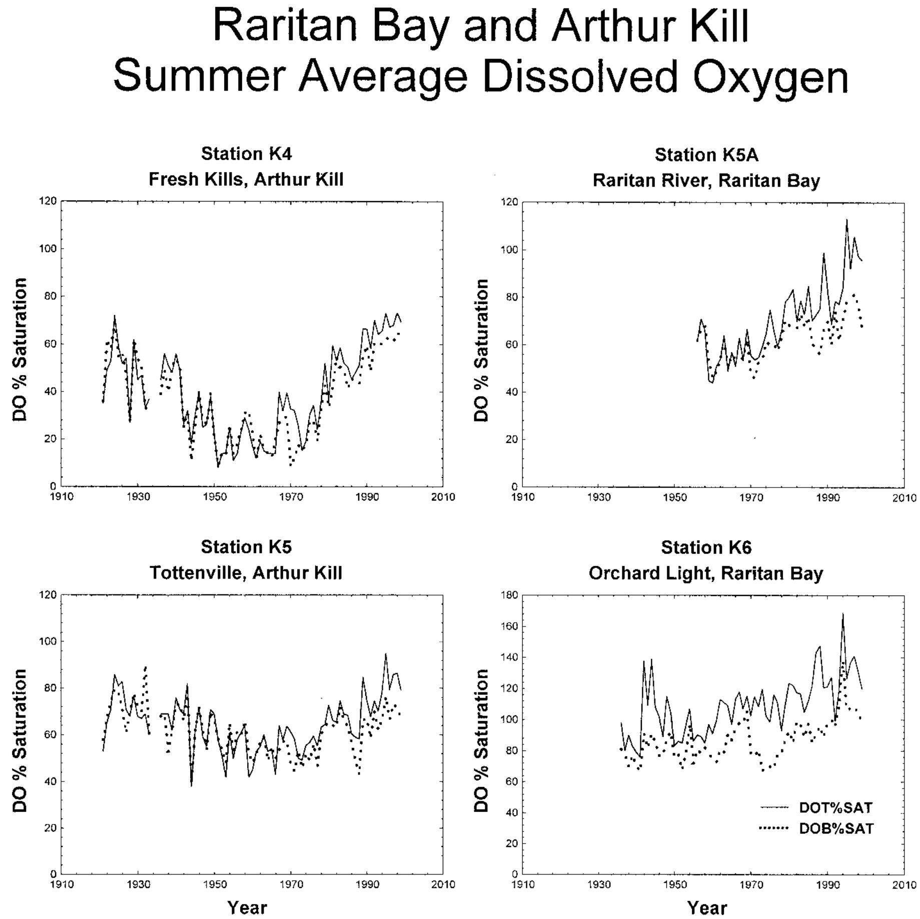

Fig. 6. Summertime (June–September) average dissolved oxygen percent saturation for surface (DOT%SAT) and bottom waters (DOB%SAT) in the Arthur Kill and Raritan Bay, 1909–1999.

the 1950s (Swanson et al. 1991) and since the 1970s (Cosper and Cerami 1996) indicate there have been no detectable changes in phytoplankton composition.

Preliminary analysis of long-term (1991–1997) WLIS chl *a* data collected by CT DEP as part of the LIS Ambient Water Quality Monitoring Program indicate a spring (February–April) and, to a lesser extent, summer (June–August) peak (Olsen unpublished data). Similar seasonal trends have been observed in both Raritan Bay and Jamaica Bay (HydroQual, Inc. 1999). Although the CT DEP data indicate a statistically ($p < 0.005$) significant decline in surface chl *a* concentrations of approximately 1 $\mu g\ l^{-1}\ yr^{-1}$ at the Throgs Neck in the upper East River (Fig. 1 station E8), this station exhibits a weak, albeit statistically significant, increasing trend in both summer surface maxima (Table 3 station E8) as well as surface DO% saturation ($r^2 = 0.10$; slope = 0.59% yr^{-1}; $p < 0.05$). No long-term productivity data are available for any of these areas.

Water Transparency

Recent summer average Secchi transparencies observed throughout the Harbor range from 0.5–2.5 m, with the lowest values typically observed in the Hudson River, Jamaica and Raritan Bays, and bays in the upper East River (Fig. 2). Reasons for reduced light penetration vary both temporally and spatially throughout the Harbor. Lower light transmittance observed in Flushing Bay (Fig. 1 station E15), Jamaica Bay, and Raritan Bay appear to be associated with high seasonal concentrations of phytoplankton. In these areas increasing chl *a* concentrations correlate with decreasing percent transmittance and light transmittance near the bottom is markedly better than at the surface (O'Shea and Brosnan 1997). This relationship is not observed in the Hudson, where reduced light penetration is most likely associated with higher concentrations of river-born suspended solids, and resuspension of bottom sediments from tidal scouring (O'Shea and Brosnan 1997).

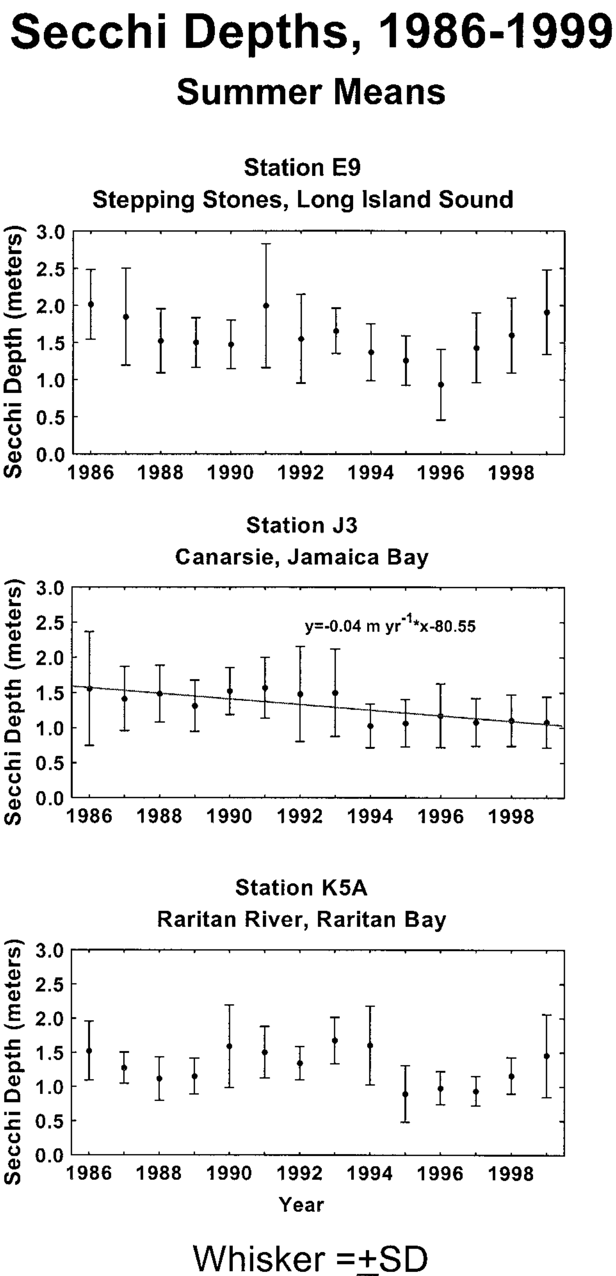

Fig. 7. 1986–1999 summertime (June–September) average Secchi transparency (m) trends for western Long Island Sound, Jamaica Bay, and Raritan Bay stations. Trends in summertime average concentrations are noted where significant ($p < 0.05$).

Summer average transparency in the WLIS has ranged from 0.9–2.0 m from 1986–1999 with 1997–1999 averages higher than the preceding three year period (Fig. 7). Although trend analyses performed in earlier years have indicated declining trends in transparency in the WLIS (Swanson et al. 1991; O'Shea and Brosnan 1997), trend analysis performed on the 1986–1999 data indicate no significant trends in this area at either $p < 0.05$ (Table 3) or $p < 0.1$. In Jamaica Bay, declining Secchi transparency is observed throughout much of the Bay (e.g., Fig. 7 station J3), with declines of 0.03–0.04 m yr^{-1} (Table 3 stations J2, J3, J5, J7, and J8). As in WLIS, no significant trends in Secchi transparency are observed in Raritan Bay (Table 3 stations K5A and K6 and Fig. 7).

Nutrients

The Hudson-Raritan Estuary is nutrient-enriched, with summer average surface concentrations of DIN $[(NO_3+NO_2)\text{-}N+NH_4\text{-}N]$ ranging from 0.08–2.10 mg l^{-1} (Fig. 2 upper right), PO_4-P ranging from 0.045–0.30 mg l^{-1} (O'Shea and Brosnan 1997), and total phosphorus ranging from 0.10–0.60 mg l^{-1} (Fig. 2 lower right). An analysis of temporal trends from 1989–1999 of summer average NH_4-N indicates declining concentrations in the East River and WLIS (Table 3 stations E7-E10), the Arthur Kill (Table 3 stations K3-K5), and selected stations in both Raritan and Jamaica Bay (Table 3 stations K5A, J8, and K5A; Fig. 8 left side). In contrast to the observed declines in NH_4-N, summer average (NO_3+NO_2)-N concentrations at some stations have increased, namely along the northern shore of Jamaica Bay (Table 3 stations J3, J7, and J8) where increases range from approximately 9–12 $\mu g\ l^{-1}\ yr^{-1}$, with higher summer average concentrations generally recorded after 1991 (Fig. 8 right side). Trends in summer average TP and PO_4-P are not observed in WLIS (Table 3 station E9 and E10 and Fig. 9 top plots); however, in several northern shore stations in Jamaica Bay (Table 3 station J2, J3, and J10), concentrations of TP, and to a lesser extent PO_4-P, show statistically significant increasing trends (Fig. 9 middle plots). Note that given the counterclockwise current in Jamaica Bay, stations J2, J3, and J10 are downstream of the 26th Ward and Jamaica WPCPs (Fig. 1). A somewhat similar pattern for TP, i.e., higher means observed after approximately 1992, is observed in Raritan Bay (Fig. 9), but is not statistically significant at $p < 0.05$ (Table 3).

Most areas of the Harbor exhibit average DIN concentrations that are one or more orders-of-magnitude higher than c. 0.02 mg l^{-1} (Fig. 2), the concentration potentially limiting to algal growth (Thomann and Mueller 1987; Fisher et al. 1988). Likewise, average PO_4-P concentrations exceed the half-saturation constant for PO_4-P of c. 0.01 mg l^{-1} by 4–30 times (O'Shea and Brosnan 1997). Although nutrient concentrations in the estuary are typically very high and far in excess of that needed for phytoplankton growth, nutrient concentrations do episodically appear to approach

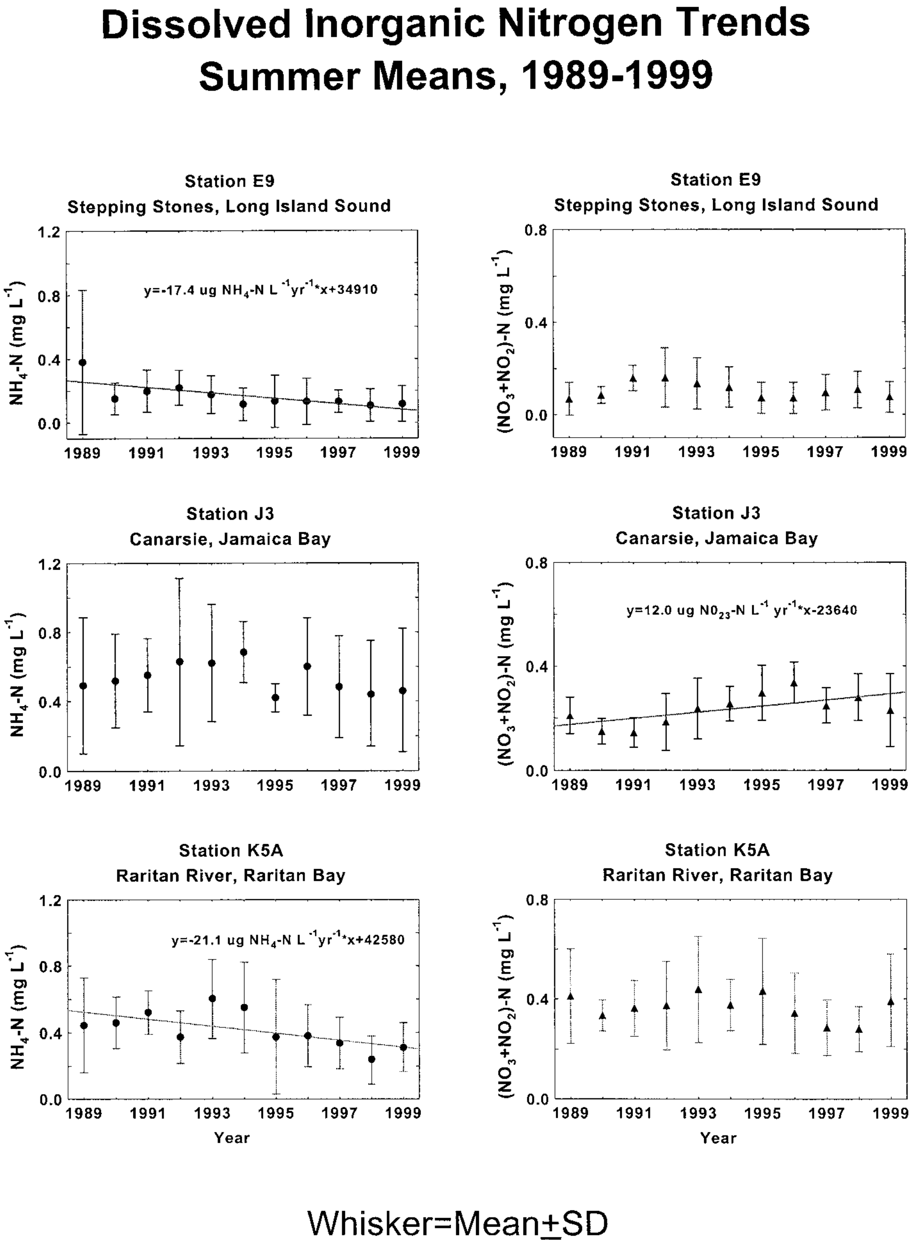

Fig. 8. 1989–1999 summertime (June–September) average ambient dissolved inorganic nitrogen [dissolved ammonium-nitrogen (NH_4-N) and dissolved nitrate- and nitrite-nitrogen (NO_3+NO_3)-N] concentrations (mg l^{-1}) for western Long Island Sound, Raritan Bay, and Jamaica Bay stations. Trends in summertime average concentrations are noted where significant ($p < 0.05$).

potentially limiting concentrations in some areas. In WLIS, minimum DIN concentrations display a spring-summer minimum when levels can episodically dip down to and below 0.02 mg l^{-1}, indicating that nitrogen availability could at times be limiting phytoplankton production in these areas (Fig. 10 station E10). In the nutrient-enriched Jamaica Bay and Raritan Bay, minimum nitrogen concentrations are generally one or more orders-of-magnitude higher than concentrations associated with nutrient limitation, even during summer when concentrations are normally at their lowest (Fig. 10). Minimum PO_4-P concentrations at or near limiting concentrations (0.01 mg l^{-1}) are observed in all three areas, but most frequently and for a longer period in WLIS (data not shown).

These observations are consistent with eutrophication modeling efforts that indicate that in waters beyond the western Narrows (Fig. 1 east of

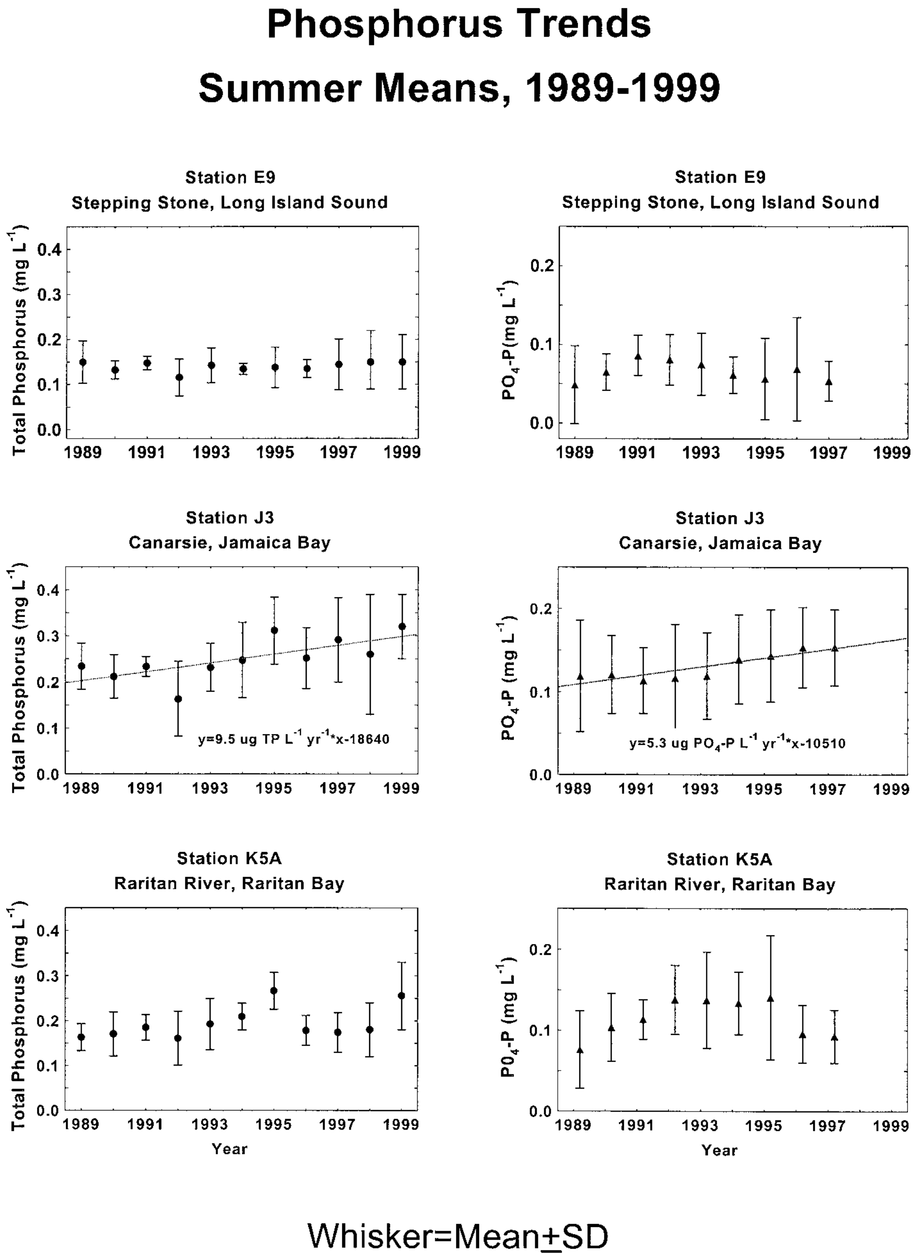

Fig. 9. 1989–1999 summertime (June–September) average ambient total phosphorus (TP) and dissolved orthophosphate (PO_4-P) concentrations (mg l^{-1}) for western Long Island Sound, Raritan Bay, and Jamaica Bay stations. Trends in summertime average concentrations are noted where significant ($p < 0.05$).

station E10), light and/or nitrogen and/or silica may all potentially limit algal growth (HydroQual, Inc. 1996). Harborwide- and system-wide eutrophication analyses undertaken by NYC DEP and others suggest that in the remainder of the Harbor, with the possible exception of Raritan Bay, non-nutrient factors such as turbidity, vertical mixing, flushing rate, zooplankton grazing, and meteorological effects most likely limit algal growth (Lee et al. 1982; Malone 1982; Mayer 1982; McLaughlin et al. 1982; Malone et al. 1985; St. John et al. 1996).

Factors Contributing to Eutrophication in WLIS and the Hudson-Raritan Estuary

Although WLIS has exhibited elevated nutrient concentrations for many years, severe, sustained hypoxia (DO < 3.0 mg l^{-1}) in bottom waters is a relatively recent phenomenon (Parker and Riley 1991; Brosnan and Stubin 1992; U.S. EPA 1994).

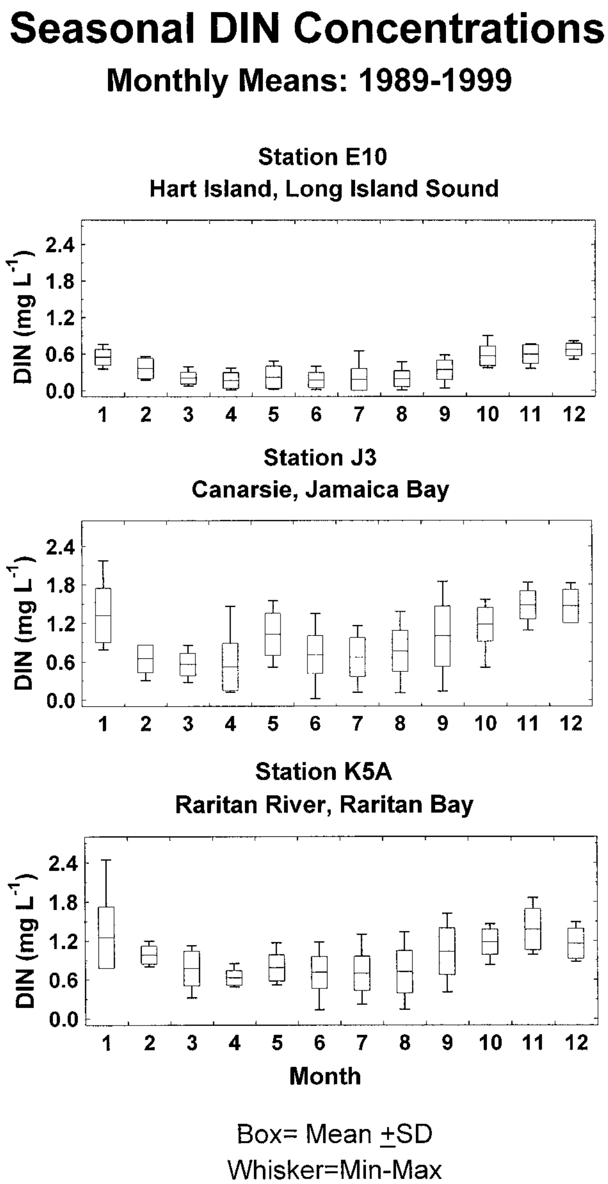

Fig. 10. 1991–1999 monthly average variability in ambient dissolved inorganic nitrogen [(NH_3)-N+$(NO_3$+$NO_2)$-N] concentrations (mg l^{-1}) for western Long Island Sound, Raritan Bay, and Jamaica Bay stations.

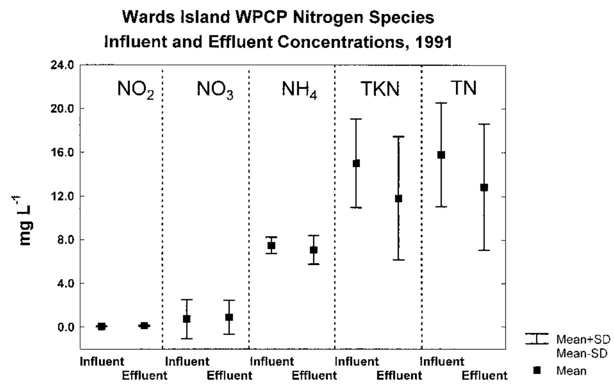

Fig. 11. Wards Island water pollution control plant pre-centrate (1991) influent (INF) and effluent (EFF) concentrations (mg l^{-1}) for dissolved nitrite-nitrogen (NO_2-N), dissolved nitrate-nitrogen (NO_3-N), dissolved ammonium-nitrogen (NH_4-N), total Kjeldahl nitrogen (TKN = NH_4-N + organic-N) and total nitrogen (TN).

One potential reason for this decrease in bottom DO is that WLIS is experiencing increasing eutrophication. Recent trends observed in WLIS, including decreasing DO minima, increasing surface supersaturation, and vertical DO stratification suggest that eutrophication has, in fact, increased in recent years (Table 3). Other areas of the estuary such as Jamaica Bay and Raritan Bay also indicate a potential increase in eutrophication, as evidenced by similar trends. Possible reasons why some indicators of eutrophication appear to be increasing include changes to both point and nonpoint BOD, TSS, and nutrient loads, and changes in vertical density stratification and residual circulation. The evidence supporting each of these explanations will be critically discussed in the following sections.

Changes to Point Source Loadings

Several studies have summarized the history of wastewater treatment in the New York Harbor and LIS region (O'Connor 1990; Suszkowski 1990; Parker and O'Reilly 1991; Swanson et al. 1991; Brosnan and O'Shea 1996a,b). Temporal changes in wastewater loads such as changes in the form of nutrients discharged, decreases in TSS and toxic compound loads, and post-1991 increases in nitrogen and phosphorus loads have all been hypothesized to have contributed to the decline of bottom DO in the WLIS. Analysis of declining water quality in LIS by Parker et al. (1986) and Parker and O'Reilly (1991) suggested that, following the 1972 Clean Water Act, increases in both the amount of wastewater treated and its level of treatment might have changed the form of nutrients being discharged, in particular increasing the percent of dissolved inorganic nutrients in effluent subject to secondary treatment. Parker and O'Reilly (1991) hypothesized that the higher proportion of dissolved nutrients discharged would result in a greater amount of wastewater nutrients being transported to the outer areas of the harbor, including WLIS, in a more bioavailable form. However, this suggestion is not supported by an analysis of typical New York City WPCP data that indicates there is no significant change in the discharge of either the fraction of inorganic dissolved nitrogen, or in the species (Fig. 11). While loadings of TN experience a c. 20% decrease after secondary treatment, most of this decrease is in the particulate organic nitrogen fraction that is removed incidentally as TSS. Therefore, there is no evidence that regional upgrades to secondary treatment resulted in an increased load of dissolved nutrients to WLIS.

Parker et al. (1986) and Parker and O'Reilly (1991) also suggested that reduced TSS loads might have increased phytoplankton productivity by increasing transparency and therefore, vertically extending the euphotic zone. This explanation has also been put forth for other estuaries similarly experiencing increasing algal productivity in the absence of increased nutrient loads (Stanley 1993, 1994). In the Harbor, this theory is supported by a comparison of typical East River point source loadings from peak concentrations in the late 1960s to 1990, indicating BOD, TSS, and TN loadings declined 66%, 66%, and 55%, respectively (Swanson et al. 1991). Water pollution control plant effluent loadings of toxic compounds have also declined significantly in recent years, possibly further contributing to more favorable conditions for phytoplankton growth. For example, from 1985 through 1993, loadings of eight metals dropped by 50–97%, with the phytotoxin copper load reduced by over 680 kg d^{-1} (Brosnan et al. 1994). However, Swanson et al.'s (1991) and HydroQual, Inc.'s (1995) examination of trends in effluent loadings and selected water quality parameters found no obvious linkage between water quality in LIS and changing sewage treatment processes or increasing water clarity. Although these analyses are somewhat qualitative (e.g., Secchi depth as a measure of water transparency does not distinguish between TSS and algal biomass), the potential linkage between the decrease in TSS and/or toxic compound loadings, algae, and the observed increase in DO stratification in WLIS, Jamaica Bay, and Raritan Bay should be investigated more thoroughly, perhaps with the recently calibrated system-wide eutrophication model (SWEM) developed by NYC DEP (St. John et al. 1996).

In the 1990s, several additional programs resulted in significant changes to NYC WPCP loads. These programs included: water conservation measures that reduced citywide sewage flows by c. 10% since 1991 (6.13 $m^3 s^{-1}$); the addition in November 1992 of a phosphate-based corrosion-inhibiting buffer to source water; a doubling of the capture and treatment of wet weather flows that enter the City's combined sewer system (Brosnan and Heckler 1996); and perhaps most significantly, an increase in nitrogen loadings from the dewatering of sewage biosolids.

This latter change occurred after the Ocean Dumping Ban Act of 1988 required several municipalities in New York and New Jersey to cease the ocean disposal of biosolids. To facilitate handling for land-based management of biosolids, New York City constructed biosolids dewatering facilities at eight of its WPCPs, including five on the East River, two in Jamaica Bay, and one in the Arthur Kill (Fig. 1). These facilities went on-line between December 1991 and June 1992, discharging their nitrogen-rich centrate, a product of the dewatering process, into the influent stream of the associated WPCP. At the time of construction, nitrogen loads were predicted to increase from 15–40% as a result of this increased load (Swanson et al. 1991). A comparison by the authors of pre-1991 and post-1993 centrate loadings data indicate that at the four WPCPs that receive centrate from more than one WPCP (i.e., 26th Ward, Wards Island, Hunts Point, and Oakwood Beach WPCPs), TN and TP loadings both increased on average 54%. At WPCPs that received centrate from only one facility (i.e., Jamaica, Red Hook, Bowery Bay, and Tallman Island WPCPs), TN and TP loading increases were on average 5% and 10%, respectively.

Figure 12 depicts changes to East River sewage loadings between 1990–1998. During this period, New York City loadings of BOD and TSS to the East River decreased c. 40% and 25%, respectively, while loadings from the four WPCPs located in the upper East River closest to WLIS, decreased 45% and 30%, respectively. Conversely, TN loads to the East River increased by c. 30% shortly after the implementation of dewatering and for the four WPCPs closest to WLIS, TN loads increased 46% between 1990 and 1993. Estimates of total LIS-wide loadings indicate the total anthropogenic point and nonpoint loads of TN to LIS increased 13% between 1990 and 1992, from 3.12×10^7 to 3.54×10^7 kg yr^{-1} (U.S. EPA 1998). Ongoing efforts by municipalities in Connecticut and New York to implement nitrogen removal technologies at their WPCPs have subsequently reduced nutrient loads, and 1996 Sound-wide loads were back down to precentrate levels (c. 3.1×10^7 kg yr^{-1}) (U.S. EPA 1998). As of 1998, NYC WPCP loads from the East River WPCPs were within 2% of 1990 levels; however, TP loads remain c. 35% higher than their 1991 low.

Depending upon the location and magnitude of load changes, the decrease in BOD loads and increase in nutrient loads could be expected to have different impacts on water quality. St. John (1990) estimated that while over 50% of the DO depression in inner harbor tidal straits (e.g., the East River) was caused by bacterial oxidation of point source organic carbon inputs (i.e., BOD), over 70% of the DO depression in outer harbor areas (e.g., LIS, Jamaica Bay, and Raritan Bay) is associ-

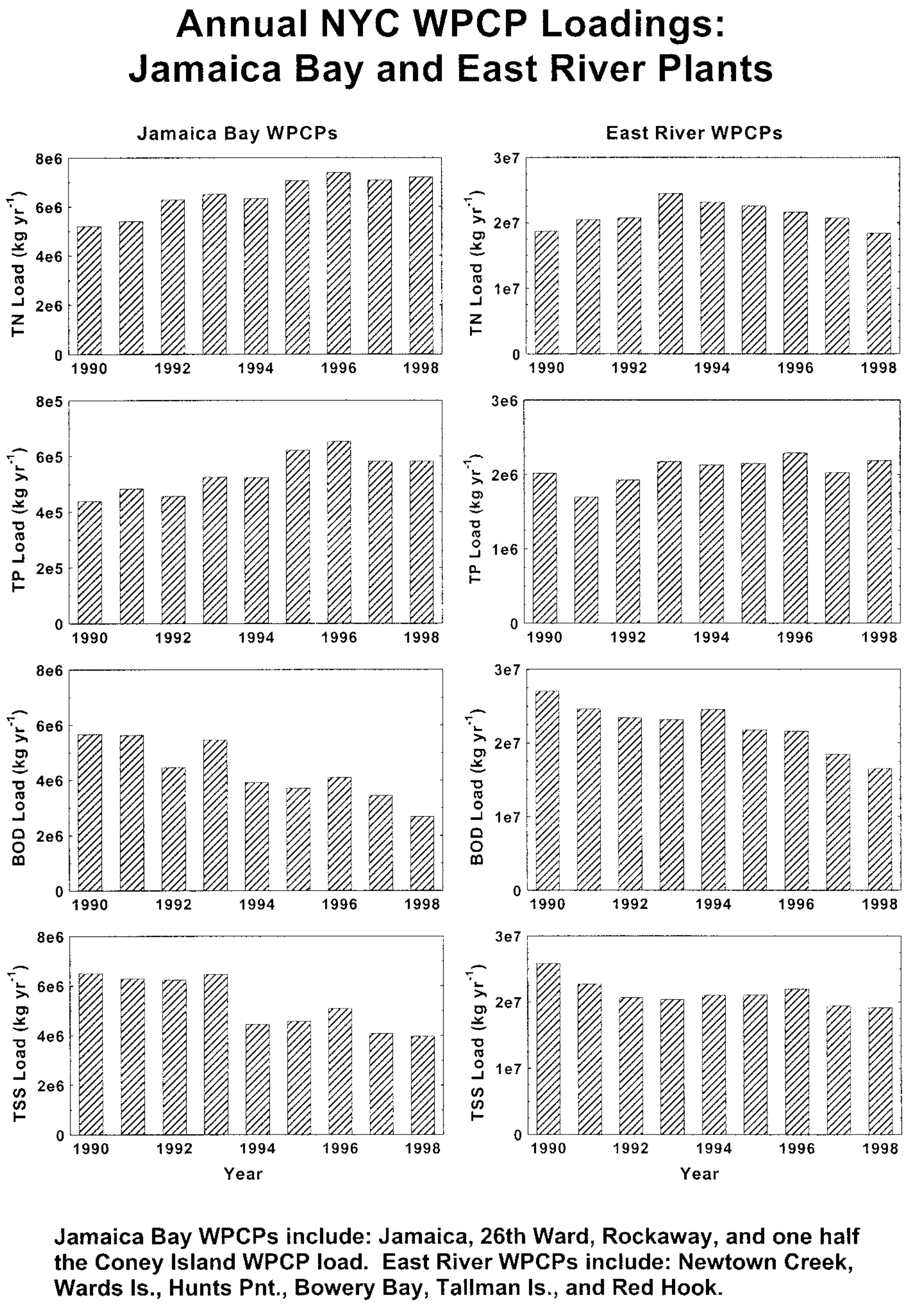

Fig. 12. 1990–1998 annual New York City water pollution control plant (WPCP) total nitrogen (TN), total phosphorus (TP), biochemical oxygen demand (BOD), and total suspended solids (TSS) loadings (kg yr^{-1}) discharging into Jamaica Bay and the East River. Jamaica Bay loadings are the summation of loadings from the Jamaica, 26th Ward, Rockaway WPCPs and one-half the loading from the Coney Island WPCP which discharges into the inlet of Jamaica Bay (Fig. 1). East River loadings are the summation of loadings from the Red Hook, Newtown Creek, Wards Island, Hunts Point, Bowery Bay, and Tallman Island WPCPs.

ated with nutrient-fueled algal respiration in the sub-pycnocline water column and associated sediment oxygen demand. Since point source loads of BOD dominate the DO dynamics in the inner harbor tidal straits, reductions in BOD inputs would be expected to improve DO in these areas. Decreases in BOD loads (Fig. 12) and increases in DO in the East River and adjacent Harlem River since 1990 support this statement (Fig. 3 stations E2 and E5).

In the WLIS, the 1993 observed increase in TN loadings from the four upper East River WPCPs closest to WLIS represented a c. 14% change to the total point source load to LIS. Long-term data suggest no evidence of increasing ambient nutrient levels, with concentrations generally lower after 1991 (Fig. 8 station E9). Although eutrophication modeling efforts suggest nitrogen may potentially limit algal growth in this area, preliminary analysis of chl *a* collected by CT DEP from the upper East River and WLIS indicate declining trends in both surface and bottom concentrations since 1991, the start of the increased TN loads from the City's biosolids dewatering efforts (HydroQual, Inc. 1996; Olsen unpublished data). The observed decline in bottom DO in the WLIS predates the onset of dewatering by several years (Fig. 3) and there is no evidence that DO has declined further as a result of increased WPCP nutrient loadings to this potentially nutrient limiting water body.

As in the East River, BOD and TSS loads to Jamaica Bay between 1990 and 1998 decreased by 52% and 39%, respectively, while TN loads increased by 38% and TP loads by 33%, with the highest loads recorded in 1996 (Fig. 12). In contrast to LIS, Jamaica Bay's total anthropogenic nutrient load originates primarily (c. 90%) from New York City WPCPs, making the impact of these increases on ambient concentrations potentially more significant (Fig. 12; West-Valle et al. 1991). Ambient concentrations of (NO_3+NO_2)-N, TP, and PO_4-P appear to have increased in northern shore Jamaica Bay stations since the early 1990s when both dewatered centrate and phosphate-buffered potable water were added to WPCP influent (Table 3 stations J2, J3, and J10; Figs. 8 and 9). Additionally, Secchi transparency in the Bay, which has been shown to be correlated with chl *a* levels, is noticeably lower after 1993 than in preceding years (Fig. 7 station J3; O'Shea and Brosnan 1997). As in WLIS, the onset of DO stratification and declining DO bottom concentrations in Jamaica Bay predate the loadings increases of the early 1990s and there is no evidence that the rate of DO decline has intensified as a result of increased nutrient loading (Fig. 5).

Raritan Bay exhibits a declining trend in ambient levels of NH_4-N with no evidence of increasing trends in other nutrient species, perhaps due to dilution from lower nutrient Atlantic Ocean waters that offsets any increases in nutrient loadings to the inner Harbor areas (Figs. 8 and 9). As is observed in Jamaica Bay and WLIS, the onset of DO stratification in Raritan Bay predates the inner Harbor loadings increases of the early 1990s (Fig. 6).

Due to modeling efforts that suggest a linkage between nutrient loading and hypoxia in the LIS, the states of New York and Connecticut are striving to reduce total anthropogenic point, nonpoint, and atmospheric sources from each of its eleven geographic management zones by 58.5% from 1996 levels over the next 15 yr (U.S. EPA 1994, 1998; HydroQual, Inc. 1996). Municipal point source reductions, which constitute 93.5% of the total target reduction of 59,000 kg yr^{-1}, will include full biological nutrient removal at five WPCPs on the East River, plus Oakwood Beach and 26th Ward (Fig. 1) with additional pilot tests on these and other WPCPs continuing (Stubin and Yao 1998) and similar nitrogen removal upgrades at the remaining Connecticut and New York WPCPs (U.S. EPA 1998). In addition, starting in the late 1990s, NYC DEP began partial centrate treatment at four of its WPCPs: Hunts Point, Wards Island and Bowery Bay in the East River, and 26th Ward in Jamaica Bay (Carrio personal communication), further reducing nitrogen loads.

Changes to Upstream and Nonpoint Source Loads

Loadings from upstream and/or nonpoint sources can be significant for some constituents. In the Hudson-Raritan Estuary, tributary loads of TSS and nitrogen account for 80% and 29% of total loads, respectively (Table 2). In LIS, 63% of TN loadings are from natural and anthropogenic nonpoint sources including: oceanic inputs through the Race, at the eastern end of the LIS, and the Battery at the southern end of the East River (34%); imports to tributaries upstream of the coastal management zones (8%); atmospheric deposition (8.3%); and other in-basin nonpoint sources (13%) (U.S. EPA 1998). Similar to point sources, long-term changes in these loadings could impact water quality in the Harbor and WLIS.

While a detailed assessment of trends in upstream and nonpoint source loads is beyond the scope of this paper, some analyses have recently been made that could increase our understanding of this issue. Analysis of nitrogen flux from 10 large watersheds in the northeast United States (including the Hudson River and Connecticut River) indicate that nitrogen loads from wastewater, atmospheric deposition, and agricultural runoff increased from c. 200 to 1,000 kg of TN km^{-2} yr^{-1} from 1900–1970, after which TN loads leveled off (Jaworski et al. 1997). Atmospheric deposition of nitrogen was the largest source, followed by agriculture and wastewater. A 3–8 fold increase in NO_3-N flux from these watersheds since the 1900s was directly related to the 5 fold increase in atmospheric NO_3-N emissions from the combustion of fossil fuels. Qualitative trends for 1970–1994 for the Hudson River and Connecticut River indicate a slight decline in atmospheric and agricultural peak loads of the early 1970s, with wastewater loads continuing to gradually increase (Jaworski unpublished data).

A recent trend analysis of 1974–1990 nitrogen and phosphorus concentrations at 16 tributary monitoring stations in Connecticut (U.S. Geological Survey 1997), indicates NO_3-N and TN increased at 8 of 16 sites with no sites exhibiting decreasing concentrations. Total phosphorus decreased at 13 of 16 sites. Insufficient data were available to detect a trend in NH_4-N for this period. However, examination of 1980–1992 data revealed decreases in NH_4-N at 11 of 16 sites. In the absence of loadings trends for which analysis is currently ongoing (Trench personal communication), the U.S. Geological Survey (1997) speculated that the observed increases in TN concentrations in these Connecticut tributaries might be due to an increase in population (up 9% from 1970–1990), increased atmospheric deposition, and increased fertilizer use. Decreases in NH_4-N and increases in NO_3-N might also reflect the oxidation of NH_4 to NO_3 by increased sewage treatment (U.S. Geological Survey 1997). Similarly for phosphorus, the widespread decrease in TP is attributed to WPCP upgrades, changes in agricultural fertilizer practices, decreases in agricultural land use, and elimination of phosphorus from some detergents. These trends reflect increasing nitrogen and decreasing phosphorus concentration trends in streams nationwide (Smith et al. 1987). An analysis of the magnitude of observed trends for Connecticut tributaries is not yet available; however, a U.S. Geological Survey analysis of a shorter time period, 1975–1988, for the Connecticut River at East Haddam (a 28,700 km^2 drainage area) indicate TN concentrations increased by 0.02 mg l^{-1} yr^{-1}, or 1.5% per year (Trench 1996). On the Housatonic River at Stratford (a 5,000 km^2 drainage area), TN concentrations from 1975–1988 also increased 0.02 mg l^{-1} yr^{-1}, or 1.9% per year (Trench 1996). Data appear insufficient for a similar analysis of trends in tributaries to the Hudson-Raritan Estuary (Phillips and Hanchar 1996).

Regional Hydrodynamics and Temporal Changes to Vertical Density Stratification

The recent covariance of vertical DO stratification and surface supersaturation among very diverse and isolated areas of the Harbor may suggest a widespread meteorological, hydrologic, or oceanographic explanation. Variations in stratification, wind mixing, tidal and gravitational circulation, and the quality of incoming waters have been shown to affect the frequency, duration, and severity of hypoxia in estuarine waters (Boicourt 1992; Diaz et al. 1992). Similarly, several studies have linked meteorological factors with observed water quality conditions within the Hudson-Raritan Estuary (Swanson and Parker 1988; Welsh and Eller 1991; Clark et al. 1995; Valle-Levinson et al. 1995; Torgersen et al. 1997).

In the Hudson River where freshwater replacement times can vary from less than 15 d during spring (high river flows) to 45–60 d during summer (low flow conditions), spring flows may be related to summertime DO minima (Clark et al. 1995). In the New York Bight, Swanson and Parker (1988) found that hypoxic and anoxic events observed since the mid-1970s could be correlated with increased stratification resulting from unusual meteorological conditions, including changes in the direction and intensity of prevailing winds, early warming, and high runoff conditions. Similarly, investigations into the cause of 180 algal blooms observed in the Hudson-Raritan Estuary from the 1950s–1980s found that 63% could be attributed to hydrodynamic or climatological factors (Olha 1990; Cosper 1991). Some of the warmest air temperatures on record in this region have been recorded in the late 1980s and 1990s, with 1990 and 1991 tied as the warmest years since 1869 (Swanson et al. 1991). The years 1987–1989 were notable for beach closings due to floatable washups on regional beaches and the mass die-off of dolphins along the coasts of several mid-Atlantic states (Swanson and Valle-Levinson 1990).

Hydrodynamic characteristics contributing to the water quality of the East River and LIS have been and continue to be studied (Jay and Bowman 1975; Swanson et al. 1982; Valle-Levinson et al. 1995; Blumberg and Pritchard 1997; Blumberg et al. 1999). WLIS has a natural predisposition for phytoplankton blooms and hypoxia. Moving from eastern to western LIS, tidal currents and estuarine circulation decrease, especially in summer when runoff is at a minimum (Bowman 1991). The surface area and wind fetch also decrease from east to west, resulting in diminishing wind-driven vertical mixing of phytoplankton and atmospheric DO exchange (Bowman 1991). Settling of organic matter increases in the WLIS, and stagnation of bottom waters is enhanced further by the Hempstead Sill, which acts as an impediment to intrusion of deep water to WLIS (Wilson 1991). Higher summer temperatures and diminished winds allow a weak stratification to persist until disrupted by storms (Parker and O'Reilly 1991; Welsh and Eller 1991). High respiration rates in the water column may enhance DO stratification by further restricting the vertical distribution of DO (Welsh and Eller 1991).

Recently the LISS investigated possible factors that may have contributed to the observed long-term decline of bottom DO in WLIS (HydroQual, Inc. 1995). Similar to the findings of Welsh and Eller (1991), this analyses of long-term (1963–1993) monitoring data found that periods of bottom water DO depletion were most strongly correlated with density stratification, in particular due to vertical temperature ($r^2 = 0.63$), rather than salinity ($r^2 = 0.15$), differences. When data were divided into two periods: pre-1985 and post-1985, the latter period exhibited greater vertical temperature differences and lower summer average bottom DO concentrations then the previous 1963–1985 period. Beginning in 1986, WLIS exhibited a substantial increase in vertical density stratification, with LIS appearing to stratify earlier in the summer and staying stratified longer, thus restricting oxygen replenishment from the surface. A similar increase was also observed at the Verrazano Narrows, but not in the Hudson River, suggesting the cause was more likely related to perturbations in coastal water characteristics, rather than changes in tributary runoff (HydroQual, Inc. 1995). However, an earlier analysis of water temperature data from the Hudson River 120 km upstream at Poughkeepsie, New York did detect a significant increasing trend of 0.12°C per decade from 1920–1990 (Ashizawa and Cole 1994). In contrast to these stratification effects, the HydroQual, Inc. (1995) study found no discernable change in other factors (e.g., nutrients, light extinction) that could cause, or relate to, the recent pattern of decreasing DO in WLIS, although this conclusion was partly based on the unavailability of long-term data for these parameters. Connecticut DEP's recent summary of DO data collected as part of the LIS Ambient Water Quality Monitoring Program elaborates on the importance of stratification in the development of hypoxia in LIS. Specifically, CT DEP has observed that since 1991, every year of observed hypoxia at their deepest station in the Narrows coincided with a vertical temperature difference of > 5°C and the maximum yearly area affected by hypoxia could be correlated ($r^2 = 0.83$) to the strength of the thermocline (Kaputa and Olsen 2000).

To investigate whether the observed correlation between temperature and dissolved oxygen stratification in the WLIS also exists for the other eutrophic areas of the Harbor, the authors updated HydroQual, Inc.'s (1995) correlation analyses of summer DO, temperature, and salinity data and expanded it to selected stations in Jamaica Bay and Raritan Bay (Fig. 13). As was found in WLIS, June–August averages of top-to-bottom temperature differences in Jamaica Bay and Raritan Bay correlate better with average DO stratification than similar measures of salinity stratification. However, the overall correlation in Jamaica Bay ($r^2 = 0.20$) and Raritan Bay ($r^2 = 0.26$) is significantly weaker than

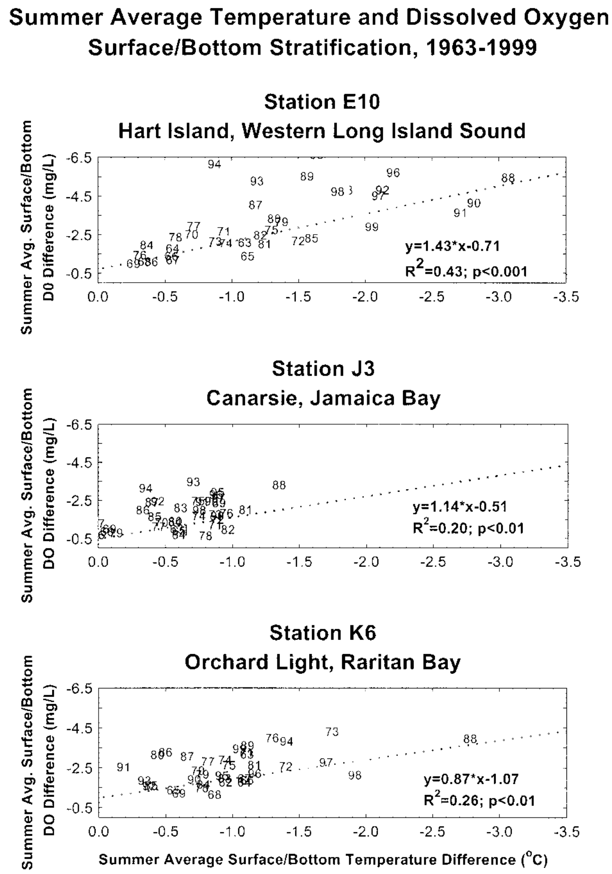

Fig. 13. Summertime (June–August) average dissolved oxygen stratification (surface-bottom; mg l^{-1}) expressed as a function of summertime (June–August) average temperature stratification (surface-bottom; °C) for the years 1963–1999 for stations in western Long Island Sound, Jamaica Bay, and Raritan Bay.

that observed in WLIS ($r^2 = 0.43$) and suggests thermal stratification may play a less significant role in other eutrophic areas of the Harbor. Note that the Jamaica Bay and Raritan Bay stations, at 9 and 6 m, respectively, are significantly shallower than the 28 m Hart Island Station and do not display the degree of temperature stratification observed in the deeper WLIS. An examination of trends in average summer temperature stratification indicates a significant ($p < 0.05$) increase in top-to-bottom temperature stratification in WLIS ($r^2 = 0.40$; slope = 0.043°C yr^{-1}) from 1963–1999, and a statistically significant albeit lesser trend ($r^2 = 0.13$; slope = 0.011°C yr^{-1}) in Jamaica Bay, primarily due to a statistically significant increase in surface temperatures (Fig. 14). No significant trend in ambient temperatures or temperature stratification is observed in Raritan Bay (Fig. 14). However, these three outer harbor areas do at times display similar temporal patterns of stratifi-

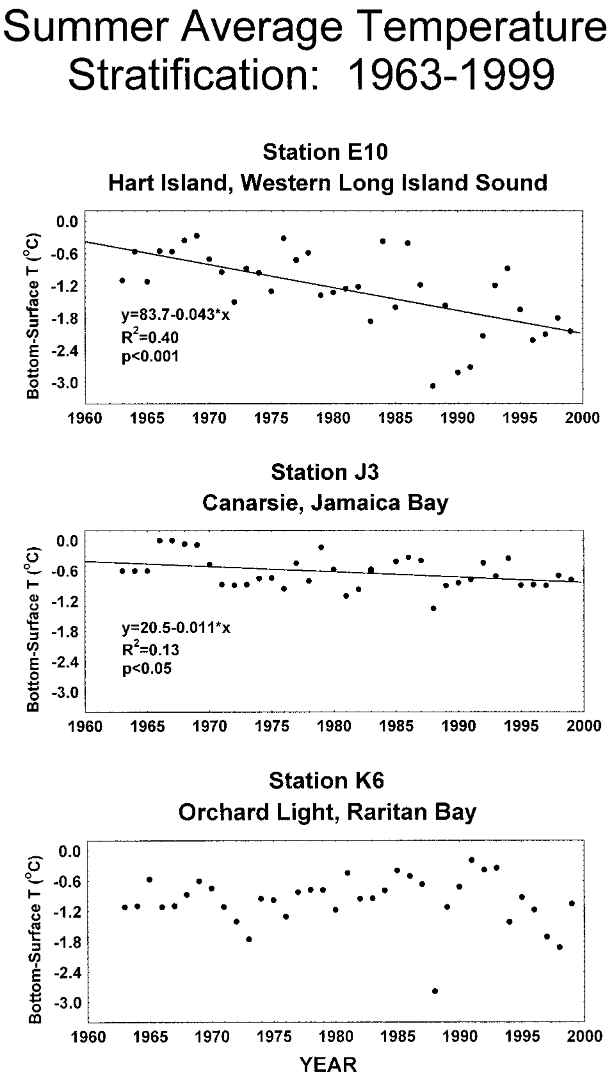

Fig. 14. Summertime (June–August) average temperature stratification (surface-bottom; °C) for the years 1963–1999 for stations in western Long Island Sound, Jamaica Bay, and Raritan Bay.

cation, e.g., the unusually high temperature stratification observed in 1988 (Fig. 13).

Conclusions

Long-term monitoring data indicate that bottom DO in WLIS has declined since at least the 1980s. This decline in summer bottom DO in WLIS is accompanied by increased supersaturation of surface DOs and increased vertical DO stratification. No trends were observed in Secchi transparency and although preliminary analyses of CT DEP data indicate a declining trend in chl *a* in the 1990s, NYC DEP summer surface DO% saturation and surface maxima data both indicate increasing trends. Although the exact causes of the decline in bottom DO is unclear, it seems more associated with increasing changes in vertical temperature stratification than changes in point and nonpoint loadings or other factors.

Similar to WLIS and also starting in the 1980s, stations in Jamaica Bay and Raritan Bay exhibit a dramatic increase in surface DO supersaturation and DO stratification. In addition, Jamaica Bay stations exhibit declining Secchi transparency and increasing (NO_3+NO_2)-N, PO_4-P, and TP concentrations. As in WLIS, some stations in Jamaica Bay exhibit a decline in bottom DO minima, albeit at rates half that observed in WLIS. No such declines are observed in Raritan Bay. Data suggest that increased density stratification due to warmer surface temperatures in the 1980s and 1990s has had a more pronounced effect on vertical temperature stratification and bottom DO depletion in the deeper WLIS waters than in the relatively shallow Jamaica Bay and Raritan Bay.

The role of long-term and short-term changes in loadings on these observations are not clear. The lack of improvement in bottom water DO in WLIS after decades of massive reductions of BOD from sewage is well documented and in stark contrast to the rest of the Hudson-Raritan Estuary. Previously hypothesized changes in the fraction or amount of dissolved nutrients discharged after the implementation of secondary treatment are not supported by examination of wastewater data. Although nutrient loads increased in the 1990s, summer bottom DO minima seem to have stabilized at approximately 1.5–2 mg l^{-1} in the far WLIS, and a further decrease in bottom DO associated with increased nutrient loads is not evident. However, the potential stimulatory impact of these recent increases in nutrient loads, as well as massive reductions of TSS and phytotoxic compounds like copper on phytoplankton in the estuary, are still open questions and may need to be examined in future modeling efforts. A more complete analysis and understanding of the impact of anthropogenic and poorly understood physical factors are needed to ensure that management strategies designed to correct symptoms associated with eutrophication in WLIS and New York Harbor are effective.

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