

C. CHEMICAL CHARACTERISTICS OF THE ESTUARY

The chemical characteristics of the estuary are described based on analytical data collected in conjunction with the Hudson River Utilities Monitoring Program and from other sources. The Hudson River Utilities Monitoring Program was developed in consultation with the DEC. The sampling design and procedures for all aspects of the monitoring studies have been described in detail in annual reports (see Appendix V-1). A synopsis of the program is also included in Appendix V-2. Chemical concentrations reported in surface water and sediment collected from the Hudson River Estuary were also obtained from the following sources:

- **Water Data Storage and Retrieval Database (STORET):** The U.S. EPA-sponsored STORET database serves as a national repository for water quality and biological monitoring data. The database has been in operation since the 1960s and is currently undergoing a major modernization effort. While the STORET database is intended to contain all relevant data collected by state and federal agencies, data entry backlogs are considerable and vary among states and regions. For the Hudson River and the Hudson River Estuary, the most recent surface water chemistry data listed in STORET date from 1998. Data contained in STORET that were collected between 1963 and 1998 are included in this evaluation.
- **National Status and Trends (NS&T) Program:** As part of the NS&T Program, National Oceanic and Atmospheric Administration (NOAA) conducted chemical analyses of sediment contamination in the Hudson-Raritan estuary (NOAA 1999). A total of 53 surface sediment samples were collected from the Hudson River Estuary between 1986 and 1995. These samples were analyzed for PCBs, pesticides, PAHs, butyltins and total metals
- **Regional Environmental Monitoring and Assessment Program (R-EMAP):** A sediment quality assessment of the New York/New Jersey Harbor system, including the Hudson River Estuary, was conducted as part of the R-EMAP program (EPA 1998). Surface sediment was collected from a total of 56 locations in the Hudson River Estuary, of which 28 were sampled in Upper Bay and 28 in the Lower Bay. The sediment samples were analyzed for PCBs, pesticides, PAHs, total metals, recoverable metals, SEM and AVS, and butyltins. Twenty PCB congeners were analyzed.

In addition to the databases described above, data reported in the scientific literature on chemical contaminants, including heavy metals and organic chemicals, known to pose risks to the human population were also evaluated. Many of these data are not presented in detail because the presence of these substances is not related to the continued operation of the Hudson River power plants and extensive presentations are readily available from other sources. Data on constituents that have impacted the Hudson River biota, e.g., cadmium and PCBs, by altering commercial and sport fisheries, are presented.

1. **Salinity**

By definition, an estuary is that portion of a river where fresh water and marine water mix. Although this definition does not allow fixed geographic boundaries, the ultimate upper end of an estuary is that point where tidal effects are no longer present. For the Hudson River this point is indisputably Troy Dam. Within the estuary, salinity influences the distribution and abundance of species and biotic communities along a gradient from "fresh water" to polyhaline or "marine" water (Table V-9).

Estuarine organisms are adapted to live in the variable salinities characterizing estuaries. Most are able to live over a wide range of salinities (euryhaline). These species are most abundant in the stenohaline and mesohaline zones typical of the Hudson River. Marine species may also be able to live in estuarine waters. The more euryhaline an organism is, the farther upriver it may be found. Marine and freshwater organisms that are able to tolerate a smaller salinity range (oligohaline) are restricted to the upper and lower ends of the estuary.

Salinity not only influences the species of resident organisms, but interacts with other variables (e.g., temperature) to determine the solubility of other substances, such as dissolved oxygen. As salinity increases, the possible amount of oxygen that can be dissolved in the water decreases. Thus, salinity may indirectly affect the distribution and well-being of biota by altering parameters having direct physiological effects.

Salinity, expressed in parts per thousand (ppt), is conventionally viewed as salt concentration, but is actually a complex measure of many dissolved substances. The "salt front" is a transition zone where fresh water first meets the mixture of fresh and marine waters, traditionally defined as the 0.1 ppt concentration. The salinity zones move longitudinally up- and downstream with freshwater flow and are also influenced by tidal amplitude and by mixing caused by variability in the morphometry of the river. Thus, the amount of habitat available in each zone changes. As freshwater flow increases, the area of tidal fresh water

TABLE V-9
ESTUARINE SALINITY ZONES

ZONE	SALINITY (ppt)
Tidal freshwater region	0 – 0.5
Salt front	0.1
Oligohaline	0.5 – 5.0
Mesohaline	5.0 – 18.0
Polyhaline	18.0 – 30.0

expands; as fresh water decreases, the higher- salinity zones extend upstream. Concerns have been expressed about the possible effects of the Chelsea Pumping Station, which would withdraw fresh water, on the position of the salt front. The position of the salt front in miles can be estimated from the flow at Green Island using a model developed by Abood (1977). Average daily flow for the period 1947-1991 as measured by USGS at Green Island is compared with the estimated location of the salt front in Figure V-14.¹

High spring flows move the salt front down to the Tappan Zee region (mile point [MP] 27); summer low flows allow the salt front to intrude toward Poughkeepsie (MP 71). For most years the salt front remains below the Roseton Units 1 and 2 station. The Indian Point Units 2 and 3 and Bowline Point Units 1 and 2 area experiences seasonal variation from fresh water to mesohaline salinities. The variation in salinity measured during the utilities' Long River Sampling Program for 1991 is shown in Figure V-15 and for 1988 in Figure V-16.

As mentioned in the section on temperature, the intrusion of salt water from the ocean brings about stratification of the estuary. Denser, more saline water follows deeper areas of the Hudson River channel. Irregularities like sills in the river bottom or constrictions in shorelines cause changes in flow direction and velocity, resulting in mixing between fresh- and saltwater layers. The slower flows in shallow shoreline areas, often coupled with inflow tributaries, bring about lower salinities in shallow shore zones. The intrusion of salt from the ocean into the Hudson River is the primary cause of density-induced circulation in the estuary. This net nontidal movement of water seaward in the upper layer and landward in the lower layer of the salinity-intruded river affects the transport of energy, mass, and plankton through the Hudson River. For example, this phenomenon, coupled with diurnal vertical movement of many fish larvae, is believed to control the location of these early life history stages along the river's longitudinal axis.

2. **Nutrients**

Estuarine circulation distributes not only salt but other dissolved organic and inorganic compounds and minerals, including nitrogen and phosphorus, which are an essential part of the "macro-nutrients" (carbon, nitrogen, phosphorus, and silicon) that plants use as raw materials to produce new biomass through photosynthesis. In freshwater systems the lack of either nitrogen or phosphorus may limit primary productivity; excess amounts may

¹Green Island flow data for 1992 and 1993 were not used because they were influenced by a temporary dam installed immediately below the gauging station.

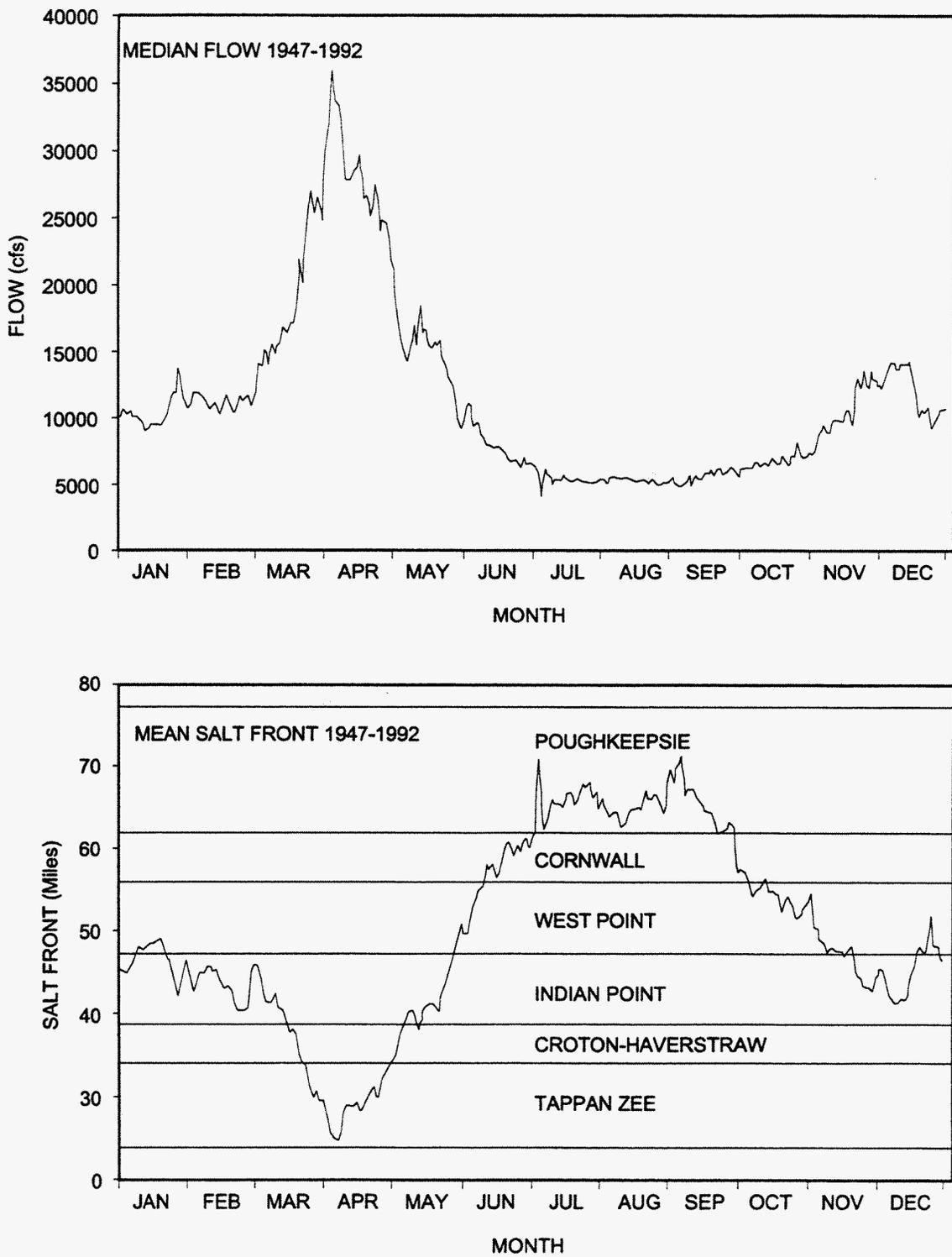


Figure V-14. Relationship between flow at Green Island and location of the salt front.

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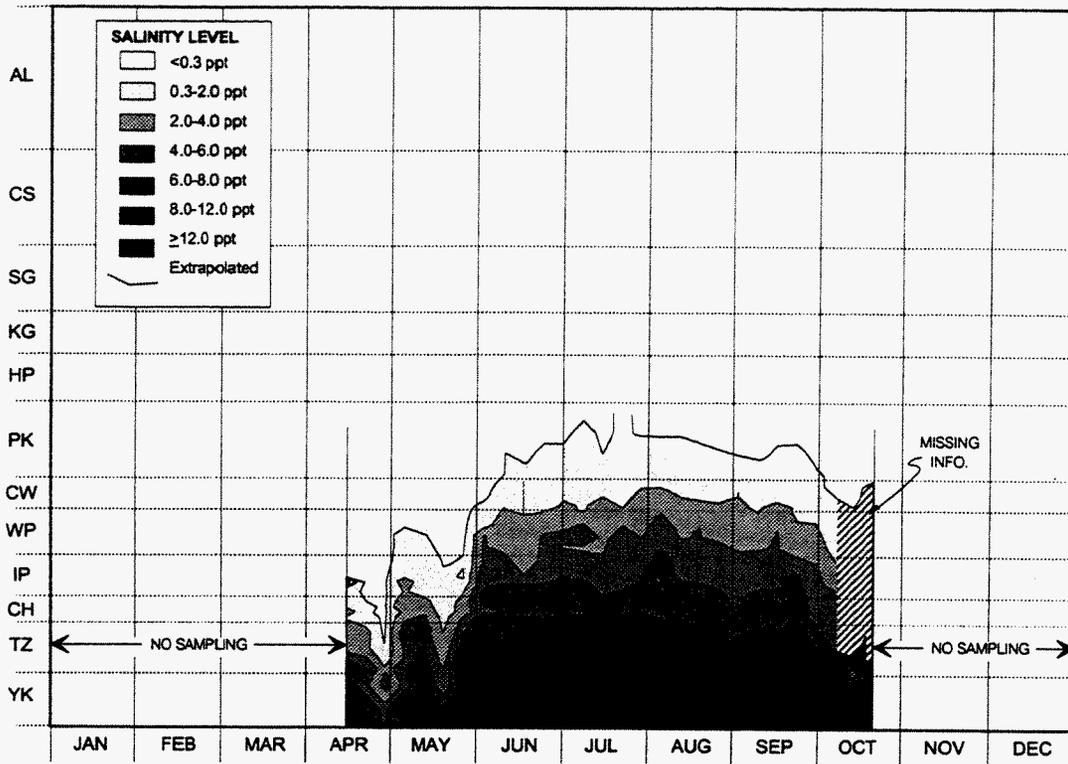


Figure V-15. Seasonal variations in salinity based on the 1991 Longitudinal River Ichthyoplankton and Fall Juvenile surveys.

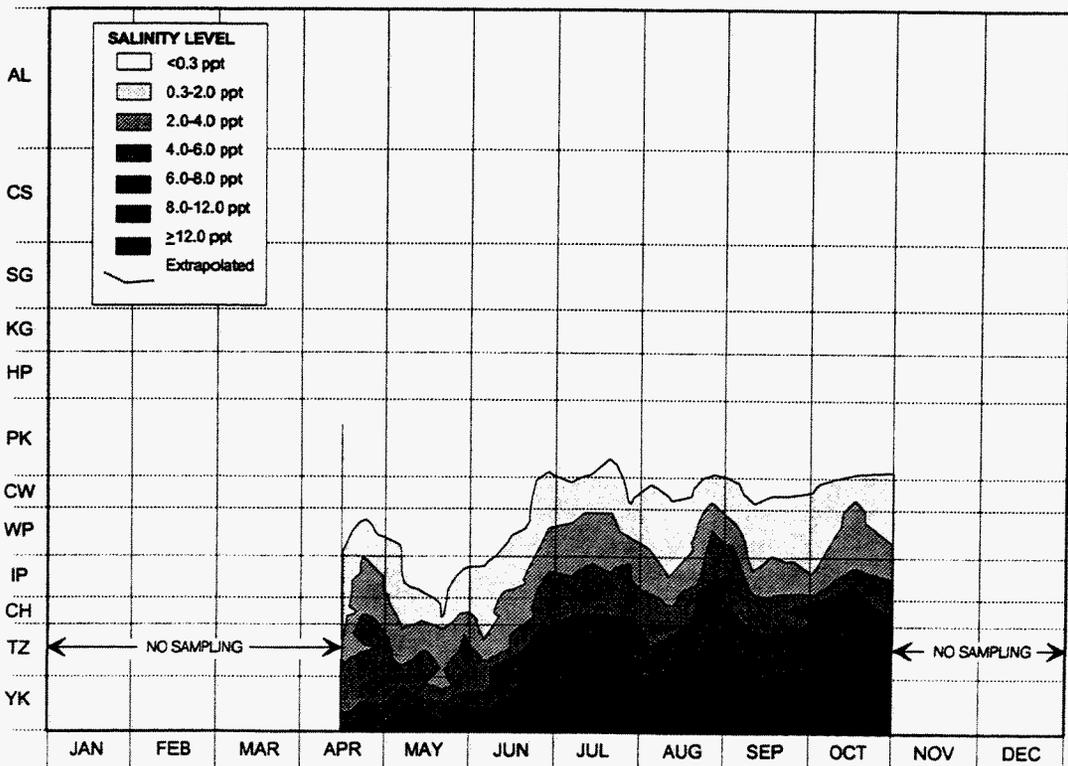


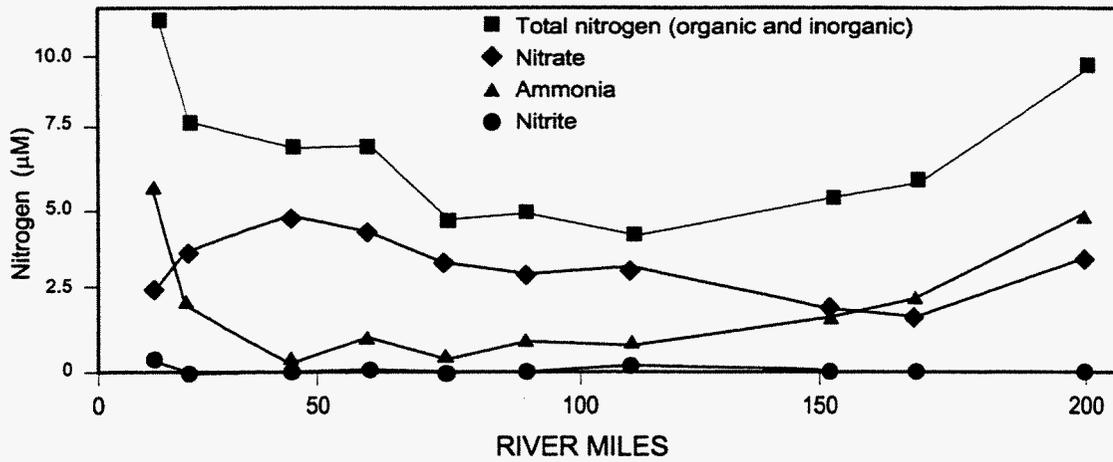
Figure V-16. Seasonal variations in salinity based on the 1988 Longitudinal River Ichthyoplankton and Fall Juvenile surveys.

contribute to eutrophication. Both undergo cyclic transformation between inorganic and organic states.

a. Nitrogen

Nitrogen has the more complex of the two cycles. Most nitrogen in estuaries comes from forest and agricultural drainage in the upper freshwater basins, from discharge of sewage treatment plants, and from storm runoff associated with urbanization. Nitrogen from the atmosphere may also be converted by lightning to a form usable by plants or by some blue-green algae species, while some additional nitrogen may be made available by sediment-dwelling bacteria. The most common forms of organic nitrogen in estuaries are amino compounds originating from plant and animal protein. Nitrogen in the form of ammonia, nitrites, and nitrates is used by plants and denitrifying bacteria. Inorganic nitrogen compounds may be released to the atmosphere, lost to bottom sediments, or flushed out to sea. Sources and cycling of nitrogen in the lower Hudson River is detailed in Carpenter *et al.* (1987) and Deck (1981). The nitrification of sewage-supplied ammonia has been studied by Deck (1986).

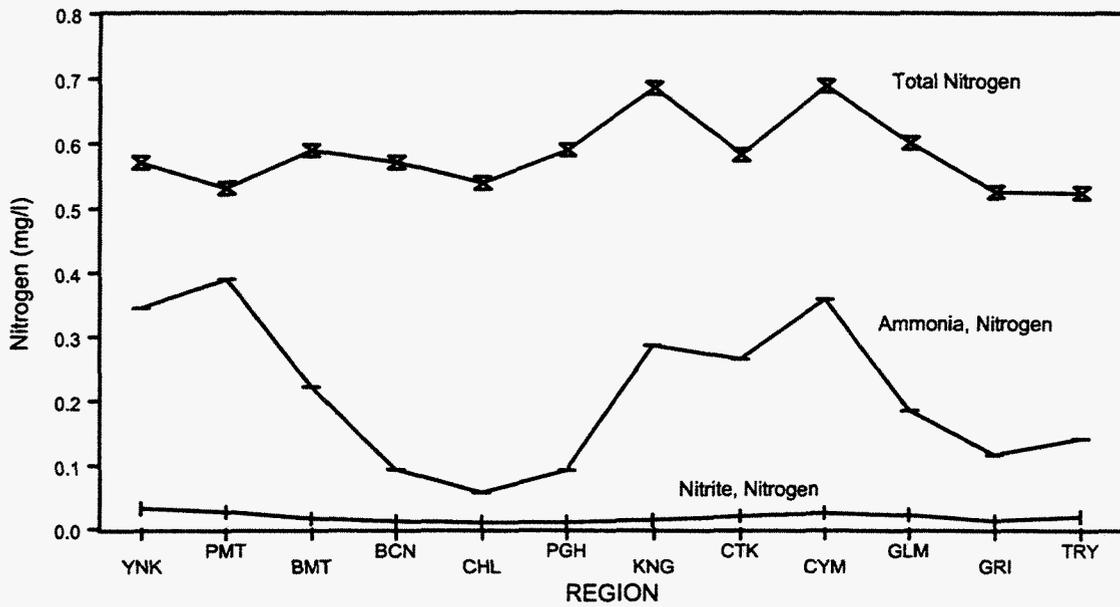
For most of the river, nitrate is the major contributor to total nitrogen concentrations. Data collected in 1974 by USGS over the length of the Hudson River reflect the influence of urban areas on both total nitrogen concentrations and the distribution among inorganic forms (Figure V-17). In the lower Hudson River about half the total inorganic nitrogen loading is attributed to wastewater treatment discharges and urban runoff (Fruci and Howarth 1990). The major source of ammonia is wastewater from the urban areas, especially New York City. This ammonia, in combination with nitrates coming from the upper estuary, accounts for 80% of the nitrogen load in the New York Harbor region (Deck 1981). Nitrate concentrations in the saline region are higher than can be attributed to known loading sources. Deck (1986) demonstrated that increases in nitrate concentrations are similar to decreases in ammonia, and also identified intense nitrification events around MP 18 that appear to be related to reduced circulation and increased residence time in the harbor. Data on total nitrogen, nitrate, and ammonia nitrogen concentrations from Yonkers to Troy are available from the EPA's STORET database. When averaged over the period 1975 through 1992, these data show small variations in total nitrogen concentrations (Figure V-18). Concentrations of total nitrogen at Glenmont (below Albany) and Poughkeepsie, both DEC water quality stations, are generally lower in recent years (Figure V-19). Concentrations of ammonia nitrogen at Poughkeepsie are available from data collected from 1965 to 1998. Average concentrations of ammonia nitrogen at Poughkeepsie declined sharply from 1967 to 1969, and data collected from 1969 through 1998 indicate a gradual decline in the mean concentration of ammonia nitrogen in surface water. The available data



NOTE: Each point represents a single water sample.

Source: USGS, cited in Limburg et al. 1986.

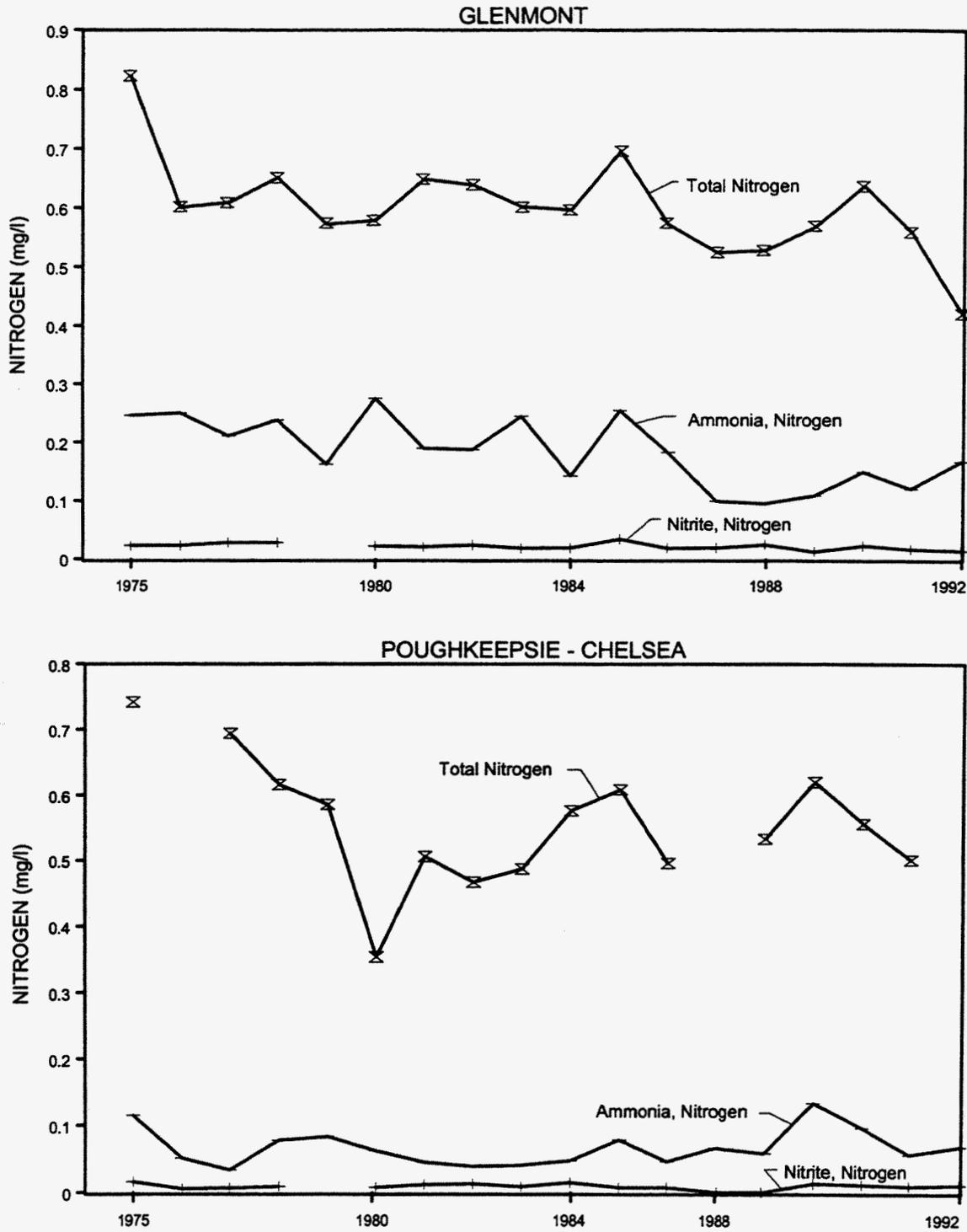
Figure V-17. Ammonia, nitrate, nitrite, and total nitrogen (unfiltered) concentrations in Hudson River water during August 1974.



Source: EPA STORET Database 1975-1992 (where available)

REGION					
YNK	Yonkers	BCN	Beacon	KNG	Kingston
PMT	Piermont	CHL	Chelsea	CTK	Catskill
BMT	Bear Mountain	PGH	Poughkeepsie	CYMY	Coeymans
				GLM	Glenmont
				GRI	Green Island
				TRY	Troy

Figure V-18. Nitrogen concentrations in the Hudson River, Yonkers to Troy, average values for each station, 1975-1992.



Source: EPA STORET Database 1975-1992 (where available)

Figure V-19. Average nitrogen concentrations in the Hudson River at Glenmont and Poughkeepsie-Chelsea, 1975-1992.

indicate that the input of ammonia, coupled with nitrogen's interconvertibility among species, results in overabundance of total nitrogen; thus in the Hudson River the availability of nitrogen does not limit primary productivity.

b. Phosphorus

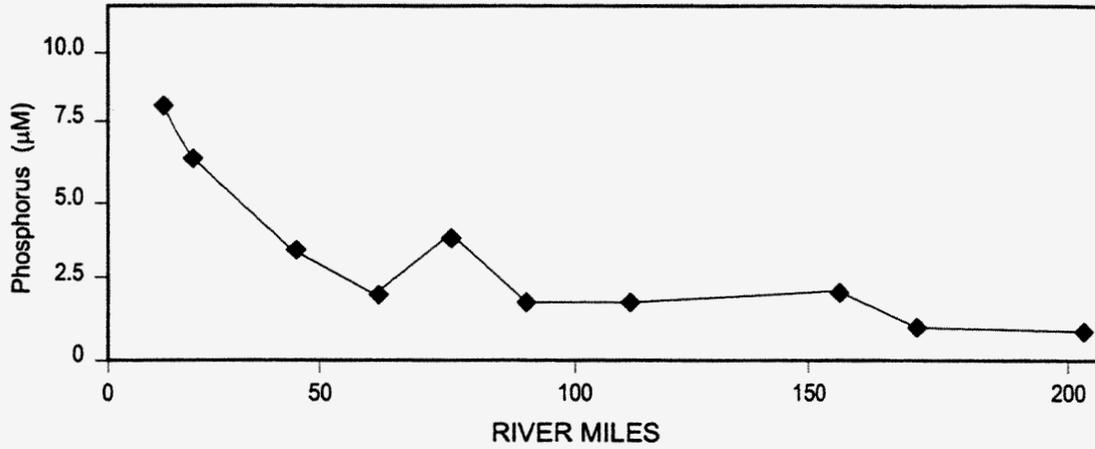
The phosphorus cycle is comparatively simple. Phosphates are leached out of rock formations and soil and enter the rivers and streams that flow into the estuary. Additional input also occurs from wastewater discharges. Inorganic phosphates are used directly by plants and are converted into organic forms used by animals. Organic phosphates released through decomposition of organic material or excretion are reconverted into inorganic forms by microbial action. Phosphates are lost from the estuary through flushing and loss to sediments.

Wastewater and runoff from urban areas contribute most of the phosphorus as phosphate. Deck (1981), in a comprehensive evaluation of nutrients in the lower estuary, attributed most of the phosphorus entering the ocean to urban sources below Yonkers. These data demonstrate concentrations of dissolved phosphate to be one to two orders of magnitude higher in the lower estuary (MP 18) than in the upper estuary. Data from the USGS 1974 survey reflect this input as a steep increase in total phosphorus below RM 55 (Figure V-20). The data available from EPA's STORET database, averaged from 1975 through 1992, show increasing concentrations from Bear Mountain Bridge to Yonkers (Figure V-21). From 1975 through 1992 data from Poughkeepsie show minor variations (Figure V-22), and STORET data from 1993 and 1994 indicate a slight decline in phosphorous concentrations. Data from Glenmont show higher concentrations in the period 1979 through 1982 as compared to the recent past. In general, all data indicate that phosphate concentrations are not limiting to plant growth in the freshwater reaches and reflect enrichment from artificial sources in the higher-salinity, lower region of the river. The effects of phytoplankton activity upstream of the salt front were observed in concentrations of soluble phosphate during summer low-flow conditions (Clark et al. 1992). No uptake by phytoplankton was observed in the harbor area, presumably because the changes caused by biological processes are overwhelmed by wastewater inputs of phosphorus.

c. Carbon

Rivers are the major conduits for flows of organic carbon from the terrestrial environment to the oceans (Howarth et al., 1992). High concentrations of suspended organic particles are a persistent feature of many large rivers and the upper reaches of

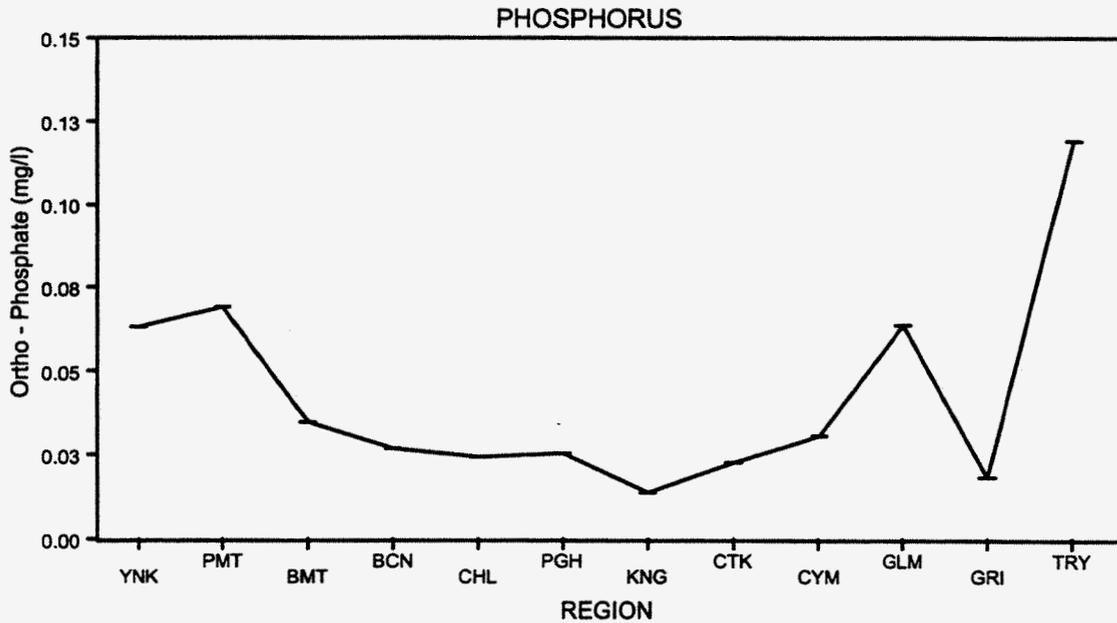
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NOTE: Each point represents a single water sample.

Source: USGS, cited in Limburg et al. 1986.

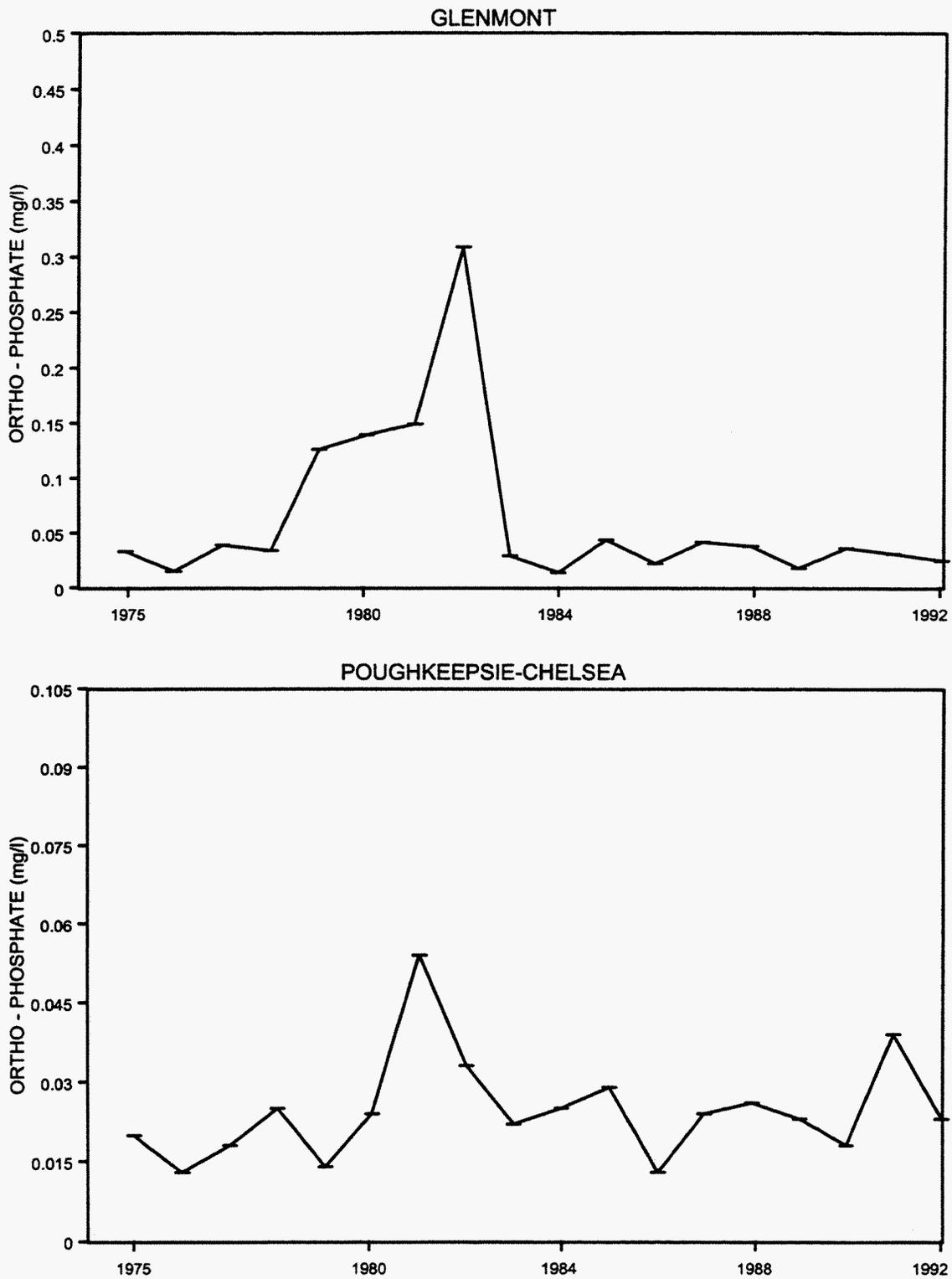
Figure V-20. Total phosphorus concentration in Hudson River water during August 1974.



Source: EPA STORET Database 1975-1992 (where available)

REGION			
YNK	Yonkers	BCN	Beacon
PMT	Piermont	CHL	Chelsea
BMT	Bear Mountain	PGH	Poughkeepsie
KNG	Kingston	CTK	Catskill
GLM	Glenmont	CYMY	Coeymans
GRI	Green Island	TRY	Troy

Figure V-21. Phosphorus concentrations in the Hudson River, Yonkers to Troy, average values for each station, 1975-1992.



Source: EPA STORET Database 1975-1992 (where available)

Figure V-22. Average ortho-phosphate concentrations in the Hudson River at Glenmont and Poughkeepsie-Chelsea, 1975-1992.

most estuaries (Findlay et al., 1991). In 1982, estimates of the flux of organic carbon in the world's rivers to the oceans range from 0.2 to $>1.0 \times 10^{15}$ gC yr⁻¹ (Mulholland and Watts, 1982). The Hudson River (at Green Island, NY) was estimated to export 7.3 gC m⁻² yr⁻¹ in 1977 and 5.0 gC m⁻² yr⁻¹ in 1978 (Mulholland and Watts, 1982).

Strayer et al. (1996) reporting on the effects of the 1991 invasion of zebra mussel (*Dreissena polymorpha*) indicated that phytoplankton production in the Hudson River was estimated to be about 50 gC m⁻² yr⁻¹. They noted that this was less than the estimated 120-150 gC m⁻² yr⁻¹ assimilation by the zebra mussel.

i. Estuarine/Tidal River Carbon Cycling

Detrital organic carbon in aquatic systems is often divided into two fractions by size, dissolved organic carbon (DOC) (<0.5 μm) and particulate organic carbon (POC) (>0.5 μm) (Mulholland and Watts, 1982). In large river systems, DOC is typically directly transferred to the coastal zone, whereas the particulate fraction tends to accumulate in the delta. The sediments deposited by large rivers tend to be organic-rich muds characterized by intense anaerobic processes. This is due to the flocculation of colloidal material, an increase in salinity, and the presence of large quantities of POC. Hence, a large fraction of the POC load accumulates in the delta and never reach the continental shelf (Gattuso et al., 1998). Rivers with macrophyte beds along its banks also trap the majority of the POC flowing downstream. In these situations, whole-system respiration in these macrophyte beds exceeds gross primary production (Howarth et al., 1992).

In theory, increased nutrient loads into large rivers lead to eutrophication, enhances net ecosystem production, and shifts the system toward increased autotrophy. However, increased nutrient loads often result in increased respiration of the organic carbon, shifting the system towards heterotrophy. When light becomes a limiting factor, respiration becomes the dominant process in the system (Gattuso et al., 1998). This has been seen in the Hudson River, where the freshwater portion is strongly heterotrophic. During the recent past, primary production was low and light-limited as a result of high turbidity and a deep water-column (Howarth et al. 1991). With the establishment of zebra mussel in 1991, this fundamental ecosystem component may be changing.

There is a positive, although weak, correlation between the concentration of organic carbon in running waters and flow rate (Mulholland and Watts, 1982). The combination of high stream discharge and high suspended organic matter concentrations during spring runoff or major storms are responsible for the bulk of the carbon transport in the Hudson River. Particulate organic carbon levels in this river exhibited maximum values in either the spring or during mid- to late summer from 1987 to 1989. Detrital material was a

significant component of the POC during this study period, with an average contribution of 63.3%, ranging from 50 to 80% of the total POC (Findlay et al., 1991).

Most of the carbon cycling studies conducted for the Hudson River, such as those cited above, have focused on the tidally-influenced freshwater portion of the river. These investigations have typically drawn a study boundary starting at the upstream end at a dam in Troy, NY, and extending downstream to the occurrence of saline water. The location of the saline water mass varies with season, year, and freshwater runoff from upstream. In years of extreme drought, the saline water mass extends further upriver than Newburgh, NY. The freshwater portion of the Hudson River would generally not contain materials associated with the brackish portion of the river since these constituents would only be advected as far upriver as the saline water mass (Howarth et al., 1996). Hence, most of the carbon studies in the open literature exclude the historical contribution of sewage discharge originating from New York City and should be interpreted as such (i.e., the mechanics and chemistry of carbon cycling in the estuarine portion of the Hudson River may be quite different from the freshwater portion of the river). In 1991, sewage inputs into the Hudson River from New York City were reported to be an estimated 5.3×10^9 g/year (assuming secondary treatment) to 35×10^9 g/year (assuming no sewage treatment) (Howarth et al., 1991).

ii. Benthic Utilization of POC

Most of the organic matter produced in, or entering, coastal systems is utilized by detritivores rather than grazers (Briggs et al., 1979). The benthic processing and cycling of autochthonous and allochthonous production is done almost entirely through detritus food webs (Diaz and Schaffner, 1990). Diaz (1989) reported a community dominated by *Macoma balthica* (bivalve), *Nereis succinea* (polychaete worm), *Tubificoides heterochaetus* (oligochaete worm), *T. brownae* (oligochaete worm), and *Leptocheirus plumulosus* (crustacean) in the immediate vicinity of a large sewage outfall to the James River estuary in Virginia. This community was similar to those found away from the vicinity of the outfall, although the standing crop and densities were much higher at the outfall, most likely due to the organic loading (Diaz, 1989).

The mere quantity of detritus in a coastal system, however, does not solely govern the utilization of the detritus by detritivores. The state of decomposition and the quality and form of the nutrients in the detritus are two of the major controlling factors in detritus utilization. Detritivores typically digest >15% of deposited organic matter. The bulk of detritus is composed of structural carbohydrates that some fungi and specialized bacteria can degrade, but which most detritivores do not have the enzymatic ability to assimilate. Marine bacteria make particulate organic matter in marine systems available to larger

detritivores (e.g., benthic macroinvertebrates) in a more usable form primarily through the attachment and metabolization. Thus, the microfauna community generally controls the enhancement of energy transfer from detritus to the detritivore (Briggs et al., 1979). Based on this information of microbial metabolism, the high rate of bacterial production compared to net primary production in the Hudson River suggests that allochthonous inputs are a significant source of energy to this food web (Howarth et al., 1992). In contrast to the view that microbial attachment and metabolization of detritus produces a more usable form of organic matter for detritivores, it has also been hypothesized that the microbial community associated with the detritus is itself serving as the real food source for detritivores. The detritus thus serves mainly as a carrier of this community (Day et al., 1989).

The periodic resuspension of detritus, either through currents or organism activity, also enhances the recycling of organic matter. Suspension feeders such as bivalves may use resuspended detritus, and microbial activity on the detritus may be increased more so than if these materials remained compacted in the sediment (Briggs et al., 1979). Filter feeders, such as *Guekensia demissa* (ribbed mussel), remove POC from the water column by surviving on suspended detritus (Chalmers et al., 1985; Findlay et al., 1991). The filtering capacities of the aggressive bivalve, *Dreissena polymorpha* (zebra mussel), in the tidally-influenced freshwater portion of the Hudson River has greatly reduced the amount of POC in the river (Howarth et al. 1996, Strayer et al. 1996, and Caraco et al. 1999). Within 17 months of its first detection, the zebra mussel population constituted over half of the heterotrophic biomass in the mid-Hudson River. This population explosion increased filtration by all filter-feeders from 0.2 to 8 m³ m⁻² d⁻¹ (Caraco et al. 1997, 1999). Consequently phytoplankton and small zooplankton biomass fell by 80-90%. This in turn, produced modest increases in water transparency and marked changes in water chemistry. In particular, PO₄ and NO₃ concentrations increased and dissolved oxygen concentrations decreased. Carbon assimilation during October of 1992 was estimated to be almost 8 gC m⁻² yr⁻¹ by 1993.

In terms of particle size, 93%, by volume, of particles retained by suspension feeders were found to be smaller than 4 μm; 95% were smaller than 9 μm (Haven and Morales-Alamo 1972 in Diaz and Schaffner, 1990). The estuarine copepod *Eurytemora affinis* has been observed feeding on relatively larger particles within the 2 to 60 μm size range (Heinle and Flemer, 1975).

3. **Toxics**

The potentially damaging effects of toxics in our estuaries and freshwater rivers was not clearly recognized until the late 1960s. Concentrations of these contaminants in the Hudson River were not measured systematically until the 1970s.

Toxics believed to pose the greatest risks to the Hudson River ecosystem fall into three major categories: (1) pesticides and herbicides, including DDT, DDE, aldrin, lindane, chlordane, endrin, heptachlor, and toxaphene; (2) heavy metals, including arsenic, cadmium, chromium, copper, mercury, lead, and zinc; and (3) other organic contaminants, including polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs). These contaminants may be acutely toxic or occur at levels that cause chronic or sublethal effects on biota. Toxins can also bioaccumulate, concentrating in the tissue of organisms at the upper levels of the food chain.

There are many potential sources for trace contaminants, including runoff from agricultural, industrialized, and urban areas. These contaminants enter the Hudson via its freshwater tributaries as well as along the main stem. Other sources include permitted municipal and wastewater discharges. Accidental spills and leachate from landfills or slag piles also contribute contaminants, although these sources are difficult to identify and measure. Many trace contaminants are persistent in the ecosystem. Some become associated with particulates and accumulate in bottom sediments. Due to the two-directional nature of the estuarine circulation, tidally driven bottom currents are created within the Hudson River Estuary resulting in extensive resuspension of contaminated sediments (Feng *et al.* 1998). Contaminants in the sediments may also remain sequestered for long periods of time, then resuspend in the water column when the sediments are distributed by dredging or scouring events. Some trace contaminants are soluble in fats and tend to accumulate in both biota and sediments. These substances may be transported through the food chain and bioconcentrate in components of the upper trophic levels, such as striped bass.

a. *Pesticides and Herbicides*

This class of substances is not believed to pose a present risk from continued input. Almost all the pesticides and herbicides in the Hudson River originated from nonpoint source runoff after applications to fields and orchards in the Hudson River basin. As shown in Table V-10 loading of most pesticides is believed to have declined since the mid-1970s when EPA restricted their use. These pollutants are not expected to impact the Hudson River ecosystem significantly in the future because they are controlled through restrictions on sales and

TABLE V-10
 ESTIMATED PESTICIDE LOADING INTO THE HUDSON-RARITAN BASIN
 (Metric tons per year)

YEAR	DDT	LINDANE	ALDRIN	CHLOR- DANE	DIEL- DRIN	ENDRIN	HEPTA- CHLOR	TOXA- PHENE
1945	1.1	0.10		0.2				
1950	2.5	1.70	0.001	3.1	0.003	0.004	0.014	0.037
1955	3.4	2.00	0.003	4.4	0.067	0.011	0.019	0.053
1960	5.0*	1.20	0.006	5.1	0.078	0.012	0.022	0.053
1964	4.7	0.64	0.010	5.5	0.066	0.010	0.023	0.054
1966	4.6	0.50	0.014	6.2	0.064	0.010	0.025	0.052
1971	3.5	0.28	0.010	8.5	0.030	0.007	0.020	0.050
1976	2.4	0.21	0.008	5.6	0.021	0.005	0.023	0.041
1980	1.9	0.08	0.001	2.8	0.005	0.001	0.025	0.020

*Peak occurred in 1957 when forests were sprayed to control gypsy moths.

Source: Ayers and Rod 1986.

application procedures. The majority of pesticides in common use today are biodegradable and do not bioconcentrate in biota.

b. Metals

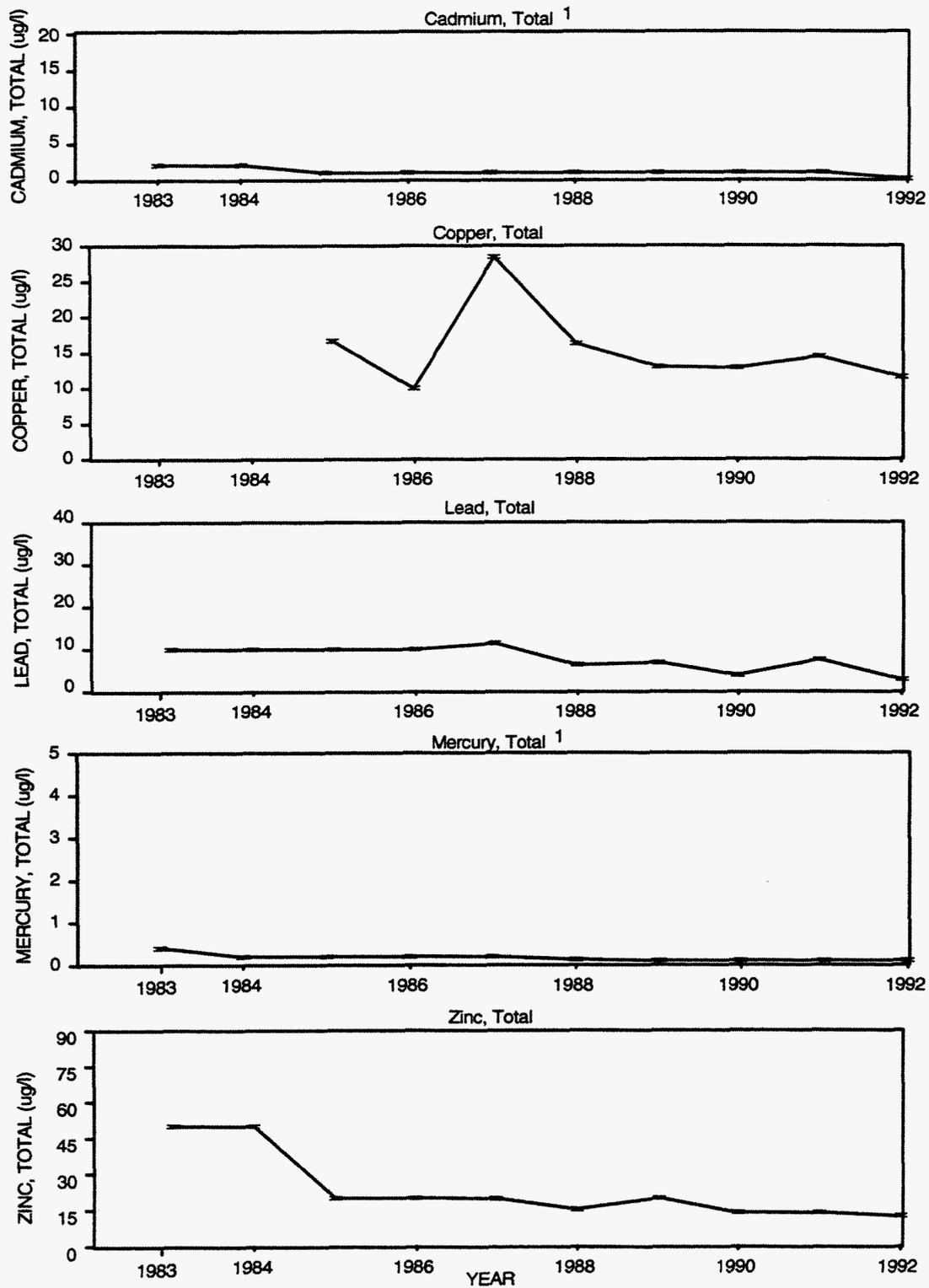
Trace concentrations of metals are a natural component of the aquatic ecosystem. Small amounts of some metals species are necessary for the physiological processes of organisms; in larger doses, however, metals may be toxic to ecological receptors.

Metals enter the estuary in runoff as a result of natural weathering from soils and rocks. Anthropogenic inputs occur in wastewater discharges from municipalities and industries and through leaching from areas used as slag piles or landfills. A substantial amount of metal enters the estuary in urban runoff and sewage effluents from the New York/New Jersey metropolitan area (Feng *et al.* 1998; Klein *et al.* 1974; Mueller *et al.* 1982). Studies to define the mass balance of metals in the Hudson estuary indicate that the discharges of some metals (cadmium, zinc, nickel, copper, and manganese) from wastewater in the New York Harbor were comparable to those from upstream riverine sources (Klinkhammer and Bender 1981).

The behavior of metals in the estuary differs by metal and form. Many metals become associated with particulates in the water column, then accumulate in bottom deposits through sedimentation. Anthropogenic cadmium, copper, and iron tend to be associated with particulates and sediment in the harbor (Klinkhammer and Bender 1981). Surficial sediments (0-3 cm) collected in the lower Hudson River in June 1994, November 1994, May 1995, and April 1996 showed the presence of anthropogenic trace metals (Feng *et al.* 1998; Gibbs, 1994). Trace metal contaminants in surficial sediments show distinct trends along the axis of the Hudson River Estuary. Silver, copper, and lead show an increasing trend down-estuary, with maximum values in New York Harbor sediments. Correlations of copper, lead, and silver suggest that urban down-estuary releases are primary sources of these contaminants (Feng *et al.* 1998). Cadmium and zinc however, show a decreasing trend down-estuary toward New York Harbor, indicating that primary sources are located upriver (Feng *et al.* 1998).

Although a number of studies have measured surface water concentrations of heavy metals in the Hudson River, these data must be evaluated and compared carefully because of differences in analytical techniques. Different studies measured different components, i.e., total metals versus the dissolved or particulate fractions. Equally important, analytical techniques for metals have improved since the 1970s and these improvements lowered the level of detection. Data for the Poughkeepsie area extracted from EPA's STORET database

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Source: EPA STORET Database 1983-1992

¹ Many data recorded as at or below detection limits.

Figure V-23. Metals concentrations in the Hudson River at Poughkeepsie, 1983-1992.

Table V-11

Concentrations of Chemicals in Hudson River Surface Water¹

	Roughkeepsie (pre-1992)				Roughkeepsie (since 1992)				Troy (pre-1992)				Yonkers (pre-1992)			
	n	Mean	Min	Max	n	Mean	Min	Max	n	Mean	Min	Max	n	Mean	Min	Max
Inorganics (µg/L)																
Cadmium	54	1.06	0	4	0	NA	NA	NA	43	1.63	0	18	30	2.15	0.5	21
Copper	56	17	0	50	0	NA	NA	NA	61	19	0	200	39	51	6	140
Lead	55	6.62	2	28	0	NA	NA	NA	61	10	0	70	33	24	0	520
Mercury	115	0.20	0.1	2.1	20	0.18	0.1	0.2	59	0.31	0.1	0.5	38	0.30	0.05	1.3
Zinc	53	27	10	340	0	NA	NA	NA	47	36	0.05	120	30	30	1	72
PCBs (µg/L)																
Total PCBs	5	0.26	0.1	0.8	0	NA	NA	NA	8	0.04	0	0.1	0	NA	NA	NA
PAHs (µg/L)																
Napthalenes, Polychlorinated	0	NA	NA	NA	0	NA	NA	NA	2	0.10	0.1	0.1	0	NA	NA	NA
Nutrients (mg/L)																
Nitrate Nitrogen	16	1.05	0	2.2	0	NA	NA	NA	37	1.90	0.01	3.8	0	NA	NA	NA
Nitrogen Ammonia (as N)	159	0.21	0	2.65	26	0.029	0	0.13	560	0.18	0	1.8	238	0.68	0	11
Nitrogen Ammonia (as NH ₄)	1	0.45	0.45	0.45	0	NA	NA	NA	18	0.13	0.02	0.3	0	NA	NA	NA
Total Nitrogen	33	1.23	0.75	2.06	0	NA	NA	NA	79	1.15	0.48	1.9	33	2.79	1.1	17
Total Orthophosphate	10	0.04	0.02	0.06	0	NA	NA	NA	25	0.02	0.01	0.06	48	0.12	0.05	0.42
Total Phosphorus	66	0.15	0.02	1.6	16	0.056	0.01	0.1	147	0.06	0	0.21	78	0.32	0.07	3.13

¹Data from USEPA's STORET database.

NA – Not Analyzed

show little variation in the concentrations of metals from 1983 through 1992 (Figure V-23). Detected concentrations of metals including cadmium, copper, and zinc exceed acute and chronic ambient water quality criteria. Additional mercury data from 1993 to 1997 are also available from EPA's STORET database. Overall, mercury concentrations in surface water collected from the Poughkeepsie area have declined slightly during the mid-1980s and have remained relatively constant throughout the late-1980s to 1997 (Table V-11).

Data from a year-long study, conducted between 1980 and 1981, indicate that measured concentrations of total and dissolved metals in the Hudson River (MP 11 to MP 155), showed higher values of some metals (iron, manganese, nickel, and zinc) in the New York harbor region (Table V-12). Most of the metals concentrations measured in the Hudson River are within water quality standards. Exceedances of the chronic ambient water quality criteria for cadmium, copper, lead, mercury, and zinc have been identified based on EPA's STORET database however (Table V-11).

i. *Lead*

At high concentrations under controlled conditions, lead adversely affects the survival, growth, reproduction, development, and metabolism of most species (Eisler 1988). Numerous physical, chemical, and biological variables substantially influence its effects. In general, organically complexed lead is more toxic than are inorganic lead salts. Food chain biomagnification of lead is negligible.

Lead has entered the Hudson River in wastewater and industrial discharges but also as road runoff. The largest historical source of lead in the environment has been attributed to particulates from automobile exhaust resulting from lead additives to gasoline. Regulations of gasoline additives in the 1970s lowered lead emissions substantially and the number of water quality standard violations has been reduced in the 1980s as compared to the 1970s (Rohmann et al. 1987).

ii. *Mercury*

Mercury, a widespread environmental contaminant, can occur in several valence (oxidation) states, including the monovalent mercurous ion (Hg^+), the divalent mercuric ion (Hg^{2+}), and uncharged elemental mercury (Hg^0). The ionic forms can combine with other elements to form inorganic salts of mercury, or they can combine with carbon to create a variety of organo-mercury compounds, including methylmercury (CH_3Hg^+). The conversion of inorganic mercury to methylmercury is the most important process affecting the ecological toxicity and bioaccumulation potential of mercury. Methylation is influenced by several factors, including dissolved organic carbon, dissolved oxygen,

TABLE V-12

AVERAGE TOTAL METALS CONCENTRATIONS BY STATION IN HUDSON RIVER 1980-1981^a

RM	As	Cd	Cr	Cu	Fe ^c	Pb	Mn ^b	Hg	Ni ^b	Se	Zn ^c	V
11	2	0.6	55	11	600	7	70	0.5	12	2	38	18
17-41 ^b	1	0.6	53	10	630	9	66	0.5	8	2	34	15
42	2	0.5	53	9	560	6	69	0.3	7	2	32	15
53-77 ^b	2	0.3	50	11	420	3	56	0.4	3	2	30	6
83	2	0.2	53	10	380	4	48	0.5	3	2	28	5
89-113	2	0.2	53	11	360	3	44	0.3	4	2	28	4
119	1	0.2	60	11	330	4	41	0.5	3	2	30	5
125-149	1	0.2	50	10	330	3	49	0.5	2	2	31	5
155	1	0.2	53	10	270	2	40	0.3	3	2	29	4

^aConcentrations are in µg/l.

^bComposite of five samples.

^cIndicates that ANOVA demonstrated significant differences by station at $\alpha = 0.05$.

Source: LMS 1982.

As, arsenic; Cd, cadmium; Cr, chromium; Cu, copper; Fe, iron; Pb, lead; Mn, manganese; Hg, mercury; Ni, nickel; Se, selenium; Zn, zinc; V, vanadium.

nutrients, pH, salinity, sulfides, and temperature. The environmental chemistry of mercury is quite complex, and the potential for ecological toxicity depends on a variety of site-specific factors. Mercury, particularly organically complexed forms, can bioconcentrate in organisms and bioaccumulate through food chains. Because mercury can accumulate in aquatic food webs, predation on mercury-containing aquatic organisms provides another pathway for potentially toxic exposures to mercury for both aquatic and terrestrial predators.

Mercury is a mutagen and teratogen and causes embryocidal, cytochemical, and histopathological effects at elevated exposures. Concentrations of total mercury that are lethal to sensitive wildlife species range from 0.1–2 µg Hg/L water for aquatic organisms, 2,200–31,000 µg Hg/kg BW for acute oral doses to birds, 4,000–40,000 µg Hg/kg food for dietary doses to birds, and daily doses of from 100–500 µg Hg/kg BW and 1,000–5,000 µg Hg/kg food for mammals (Eisler 1987a). For some birds, adverse effects (predominantly reproductive) have been associated with total mercury concentrations of 5,000 µg Hg/Kg feather and 900 µg Hg/kg egg. Sensitive mammals showed significant adverse effects of mercury when tissue concentrations exceeded 1,100 µg Hg/kg BW (Eisler 1987a).

Mercury enters the river in wastewater and industrial discharges but also as agricultural runoff from fields sprayed with mercury-based fungicides (Ayers and Rod 1986). Some mercury has also entered the ecosystem as leachate from marine paint that contained mercury as an antifouling agent. Data for the Poughkeepsie area extracted from EPA's STORET database indicates a slight decline in mercury from 1983 through 1994 (Figure V-23). Regulation of mercury has resulted in lower mercury concentrations in sediments of the Hudson estuary (Ayers and Rod 1986) during the 1980s as compared to the 1970s.

iii. *Cadmium*

Cadmium is a common element in the earth's crust and is commonly used as a protective coating for steel. Cadmium compounds are used in the production of nickel/cadmium batteries, as a stabilizer for PVC, and as pigments for plastics and glass. Additionally, cadmium is mixed with other metals to form alloys (WHO 1992a). In aquatic environments, suspended and dissolved forms of cadmium in surface water are readily accumulated by mollusks and other microorganisms. In sediments, cadmium commonly occurs as cadmium salts, such as sulfide, carbonate or oxide. These insoluble salts are generally not bioavailable to organisms. However, changes in physiochemical conditions may result in the conversion of insoluble salts to water-soluble forms (WHO 1992a; Eisler 1985).

Hudson River surface water concentrations of both the dissolved and particulate forms of cadmium are elevated by anthropogenic inputs associated with wastewater discharges and industrial discharges in the Foundry Cove area. Most of the cadmium is associated with suspended material that is deposited in sediments (Klinkhammer and Bender 1981). Substantial deposits of cadmium and nickel have accumulated in the sediments surrounding Foundry Cove, a backwater near Cold Spring, the site of a nickel/cadmium battery manufacturing facility (Feng *et al.* 1998; Kneip *et al.* 1974; Bower 1976). Between 18 and 23 tons of cadmium were estimated to be associated with sediments in Foundry Cove (Hazen 1981). Data collected in surficial sediments from the lower Hudson River in June 1994, November 1994, May 1995, and April 1996 show a decreasing trend in cadmium concentrations down-estuary toward New York Harbor; indicating that primary source(s) of cadmium is located upriver.

Klein *et al.* (1974) estimated cadmium concentrations of 10 µg/l in New York City sewage effluents and 25 µg/l in urban runoff. These concentrations in the untreated wastewater discharged to the harbor before treatment facilities were upgraded could have resulted in discharges of more than 40 tons per year to the harbor region. Many of the cadmium discharges in the city were believed to be associated with small manufacturing operations in New York City. Pretreatment requirements, upgraded facilities, and increased control of discharges has substantially reduced current inputs.

c. *Organic Contaminants*

i. *Polychlorinated Biphenyls*

PCBs are family of 209 compounds with range of physical properties from light, oily fluids to heavier, greasy, or waxy substances. PCBs were first synthesized over a century ago and their commercial production and use began in 1929. Because of their insulation capacity and their flame-retardant nature, they gained widespread use as coolants and lubricants. Additionally, PCBs were routinely used in the manufacture of a wide variety of common products such as plastics, adhesives, paints and varnishes, pesticides, carbonless copying paper, newsprint, fluorescent light ballasts, and caulking compounds. Manufacture of PCBs in the United States was stopped in October 1977 because of evidence that they accumulate in the environment and present potential health hazards to humans and a wide variety of other life forms (USPHS 1989).

In general, congeners of PCBs are lipophilic and have low water solubility, indicating a tendency to absorb to sediment and suspended organic matter, PCBs bioaccumulate in the fatty tissues of organisms and tend to bioconcentrate in the food chain (WHO 1992b). In

1973 the U.S. Food and Drug Administration (FDA) established a temporary tolerance level of 5 ppm (parts per million) for human ingestion of PCBs in fish and shellfish. This level was subsequently reduced to 2 ppm in a 1979 FDA ruling, judicially stayed until 1984 (FDA 1979).

PCB toxicity and carcinogenic potential has been extensively investigated. PCBs do not cause acute health effects in animals, except at extremely high doses (Flynn and Kleinman 1997). Experimental evidence does support, however, that PCBs can cause cancers in animals (IARC 1987, NTP 1989, ACGIH 1996, USEPA 1996a,b). Previously it was thought that chlorine content was the most important factor in determining the carcinogenicity with high chlorine congeners producing the highest cancer potential. However, recent studies (Brunner et al. 1996) suggest that the position of the chlorines within the PCB molecule is also important. Effects other than cancer on the liver, blood, immune system, nervous system, reproductive, and other organ systems have also been observed (ATSDR 1996; WHO 1993).

In 1973 the FDA established a temporary tolerance level of 5 ppm for human ingestion of PCBs in fish and shellfish. This level was subsequently reduced to 2 ppm in a 1979 FDA ruling, judicially stayed until 1984 (FDA 1979). The New York State Department of Health (NYDOH) requested FDA to consider a further reduction in the permitted tolerance level for the ingestion of PCBs in fish, stating that "new toxicological studies raise concerns about the potential human health risks from PCB exposure and the adequacy of the current tolerance and advisory levels" (DOH 1990a).

EPA currently considers PCBs to be a probable human carcinogen on the basis of animal tests. Until 1996, the human cancer risk factor was based on studies using rats fed high doses of Aroclor 1260, a commercial PCB containing 60% chlorine by weight (Norback and Weltman 1985). After 1996, EPA began using toxicity criteria specific to the medium of exposure (e.g., soil, air, and food) to account for different mixtures of PCB congeners in these different media. EPA also lowered their estimates of PCB cancer potency by 4- to 20-times, reflecting newer studies and risk assessment methodology (USEPA 1996b).

In addition to cancer, PCBs have been implicated in several other health issues. Recently, concerns have been raised over the possible impact of PCBs on the physical, mental, and motor development of babies born to mothers who consumed freshwater fish. Researchers in Michigan (Jacobson et al. 1990a,b) have demonstrated a correlation between (1) the estimated amounts of Lake Michigan fish (presumed to be contaminated with PCBs) consumed by mothers, (2) PCB levels in umbilical cord blood, maternal serum and maternal milk; and (3) physical measurements and neonatal behavior test scores. These studies have been highly controversial and criticized for several shortcomings. Among the criticisms,

and one acknowledged by the authors, is the fact that the behavioral deficits could have resulted from toxins other than PCBs.

A number of recent studies have also claimed that PCBs are endocrine disrupters, i.e., chemicals that cause adverse effects through changes in the endocrine system. These effects may be manifested in a variety of ways including reduced sperm counts or increased incidence of breast cancer. To date, links between PCBs and endocrine disruption have not been established. Safe (1995), based on laboratory studies concluded that PCBs themselves are not estrogenic. PCB metabolites were found to be both estrogenic and anti-estrogenic, but activity levels were about 40 million times lower than that occurring naturally in a normal diet. Studies reporting strong synergistic responses between weakly estrogenic chemicals and PCBs (Arnold et al. 1996) could not be replicated (Ramamoorthy et al. 1997) and have been formally withdrawn (McLachlan 1997).

A recently completed study by Kimbrough and Doemland (1999) appears to support EPA's 1996 lowered risk assessment categorization. Their study followed 7,075 men and women employed in the manufacture of electrical capacitors between 1946 and 1977. These workers experienced life time exposure levels far greater than that expected from environmental contamination. Although the subjects had significantly elevated blood level PCBs, they exhibited no statistically significant increase in cancer or any other disease.

PCBs in the NY/NJ Estuary. Historical releases of PCBs have resulted in the NY/NJ estuary having among the highest PCB concentrations in water, sediment, and biota along the coastal United States. As reviewed by Durell and Lizotte (1998), most of the PCBs currently in the estuary originated in PCB discharges into the Hudson River during a 25 year period from the early 1950s to the mid 1970s. The PCBs continue to be transported down the Hudson River as dissolved or particle-bound contaminants, ending up in the sediment or water column of the NY/NJ estuary which acts as a major sink for PCBs (Durell and Lizotte 1998). In addition to PCB influx from the Hudson River, PCBs also enter the NY/NJ estuary via other river systems (e.g. Passaic, Hackensack, Rahway, and Raritan) and other point and non-point sources (Durell and Lizotte 1998).

PCBs in the Hudson River. Between 1947 and 1977, 500,000 to 1.3 million pounds of PCBs are estimated to have been discharged into the Hudson River from two General Electric capacitor manufacturing plants at Fort Edward and Hudson Falls, New York (DEC 1990a). Much of this discharge was deposited in bottom sediments of the river and accumulated behind the Fort Edward Dam. When the Fort Edward Dam was removed in 1973, an estimated 850,000 yd³ of PCB-contaminated sediments was released and migrated downstream (EPA 1981). A 1978 departmental summary of PCB concentrations in Hudson River sediments indicated that the highest average concentrations were found several miles

V. Environmental Setting

downstream of the Fort Edward Dam at Thompson Island and Northumberland Dam, where overall concentrations measured 86.2 and 74.6 ppm, respectively, with individual hot spots as high as 163 ppm (DEC 1978). Lower river PCB concentrations decreased from an average of 60 ppm in the Albany turning basin to an average of 6 ppm in New York Harbor (DEC 1978).

According to the EPA, many of GE's PCBs remain concentrated in hotspots in the sediments of the upper Hudson River, but PCBs have polluted the entire stretch of the Hudson River below Hudson Falls (EPA 1999). In 1983, EPA classified a 200 mile stretch of the Hudson River from Hudson Falls to the Battery in New York City as a Superfund Site. In 1984, EPA made an interim no-action decision for the PCB contaminated sediments in the Hudson River. However, since 1990, EPA has been conducting a reassessment of its 1984 decision regarding PCBs in Hudson River sediments.

EPA's Reassessment Study has been grouped into three phases with the largest phase of the PCB Reassessment Study, Phase 2, being divided into six segments.

Report	Date	Subject
Preliminary Model Calibration Report	Oct. 1996	Groundwork for projections of future concentrations of PCBs in water, sediment and fish tissue; rationale for selection of calibration data sets; and projections for the Thompson Island Pool
Data Evaluation and Interpretation Report (DEIR)	Feb 1997	Contains geochemical analysis of data from water columns and high resolution sediment coring investigations; evaluation to determine relationships between parameters; evaluation of PCB sources.
Low Resolution Sediment Coring Report	Jul 1998	Addendum to DEIR. Contains geochemical analysis relating to comparison of sediment cores taken in 1994 from known hotspots. Addresses questions concerning burial and/or release of PCBs from hotspots.
Baseline Modeling Report	May 1999	Provides projections of future concentration of PCBs in water, sediment and fish tissue without remediation; interpretation of low resolution sediment coring data.
Ecological Risk Assessment	Aug 1999	Evaluation and interpretation of the ecological field data; furthers Phase 1 ecological risk assessment, presents ecological risk to certain organisms
Human Health Risk Assessment	Aug 1999	Presents human health risks, including cancer and noncancer risks from consumption of fish

The reassessment reports along with public comment review will assist EPA in the formulation of a Proposed Plan for the Hudson River Superfund Site. EPA's database for the Reassessment Study contains approximately 750,000 records consisting of historical data collected prior to the Reassessment Study and field data gathered during Phase 2 of the Reassessment. Data collected during Phase 2 of the study were used to characterize the nature and extent of the PCB loads in the Upper Hudson River and the importance of those

loads to the Lower Hudson River. The four major conclusions of the "Data Evaluation and Interpretation Report" are as follows:

- 1) The area of the Superfund Site upstream of the Thompson Island Dam represents the primary site of PCBs to the freshwater Hudson River. This area includes the GE Hudson Falls and Ft. Edward facilities, the Remnant Deposit area, and the sediments of the Thompson Island Pool.
- 2) A readily identifiable PCB homologue pattern dominates the water column load from the Thompson Island Dam to Kingston during low flow conditions which typically occurs 10 months out of the year.
- 3) The PCB load from the Thompson Island Pool originates from the sediments within the Thompson Island Pool.
- 4) EPA has determined that PCB sediment inventories in the Hudson River will not be naturally "remediated" via dechlorination. Evidence suggests that the extent of natural dechlorination is probably less than 10% loss from the original PCB concentrations.

Additionally, evidence in EPA's DEIR suggests that the Upper Hudson River PCB load can be seen as far downstream as RM 1.9. EPA estimates that the Upper Hudson River PCB load represents about half of the total PCB load to the entire New York/New Jersey Harbor (EPA 1999).

EPA issued the "Low Resolution Coring Report" in February 1997. This report is a companion to the DEIR report also issued in February, 1997. The Low Resolution Coring Report assesses the inventory of PCBs found in the Upper Hudson River sediments. Major conclusions of this report include the following:

- 1) Little evidence was found of widespread burial of PCB-contaminated sediment by clean sediment in the Thompson Island Pool. Burial of PCBs is seen at some locations in the Hudson River but more sites showed loss of PCB inventory from the 1976-1978 NYSDEC survey.
- 2) Sediment core site evidence shows that there has been a net loss of approximately 40% of the PCB inventory from highly contaminated sediments in the Thompson Island Pool.

- 3) Current sediment core sample evidence (1994) shows that there has been a net loss of PCB inventory in hot spot sediments sampled between the Thompson Island Dam and the Federal Dam at Troy since the 1976-1978 NYSDEC survey was conducted.

The Proposed Plan, in which EPA will issue its decision regarding possible clean-up of the Hudson River, and how that clean up will be done is also scheduled for release in December 2000. EPA estimates that a final decision regarding the PCB issues in the Hudson River will be reached by June 2001.

Twenty individual PCB congeners were measured in Hudson River sediment during NOAA's National Status and Trends study. Hudson River samples were collected only in the 1990s (Table V-13).

PCBs in Hudson River Fish and Shellfish. PCBs were first reported in Hudson River fish in 1975. Departmental studies of PCB contamination levels in Hudson River striped bass indicate that concentrations of total PCBs in the spring striped bass fishery have declined somewhat since 1978, the year annual monitoring commenced, when the average was 18.1 ppm. In 1980 concentrations averaged 6.1 ppm, dropping to 2.79 ppm in 1990, although individual fish continue to measure as high as 20.01 ppm. The highest concentrations in 1990 (an average of 6.90 ppm total PCBs), as in past years, occurred at the Albany/Troy location (RM 153) and decreased as sampling continued downstream. Lesser chlorinated PCB compounds have declined most rapidly, the more highly chlorinated and persistent PCBs predominating in Hudson River striped bass (DEC 1992a).

Because fish effectively bioaccumulate PCBs, they provide a pathway, frequently the predominant pathway, for human exposures to PCBs. Studies conducted on Michigan residents have established that those who regularly eat Lake Michigan fish have serum PCB levels up to 30 times greater than those who do not eat these fish (EPA 1992). Continuing concern for the adverse health effects of exposure to PCBs resulted in New York State issuing a ban on all fishing between the Troy Dam and Fort Edward. The state also imposed a ban on commercial fishing for striped bass in the Hudson River in 1976. This ban was extended to commercial fishing in New York Harbor and along both shores of western Long Island in 1985, ultimately extending to all waters of the state in 1986. In 1990, however, the state reopened a limited commercial fishery for striped bass on the eastern end of Long Island, permitting the harvesting of up to 17,260 adult fish 24 to 29 in. long. The state currently permits the harvesting of approximately 23,000 adult fish between 24 and 39 in. long (NYSR 1990 and 1992). DEC's reopening of the fishery was accomplished by a Negative Declaration under SEQRA, which concluded that commercial harvesting in excess of 17,000 adult striped bass was consistent with the Atlantic States Marine Fisheries

Table V-13

Concentrations of Chemicals in Hudson River Sediment¹

CHEMICAL	ENGLEWOOD CLIFFS (SINCE 1992)			
	N	MEAN	MINIMUM	MAXIMUM
PCBS ($\mu\text{G/KG}$)				
PCB 8	0	NA	NA	NA
PCB 18	3	26	24	28
PCB 28	3	89	65	110
PCB 44	3	24	21	27
PCB 52	3	38	32	43
PCB 66	3	25	19	34
PCB 101	3	21	17	25
PCB 105	3	4.3	4	5
PCB 107	0	NA	NA	NA
PCB 118	3	19	18	21
PCB 126	0	NA	NA	NA
PCB 128	3	5.3	5	6
PCB 138	3	20	19	21
PCB 153	3	34	24	44
PCB 170	3	5.3	5.0	6.0
PCB 180	3	10	10	11
PCB 187	3	7.3	7.0	8.0
PCB 195	3	0.9	0.8	1
PCB 206	3	4.3	3.0	5.0
PCB 209	0	NA	NA	NA
PAHs ($\mu\text{g/kg}$)				
Naphthalene	3	140	140	140
2-Methylnaphthalene	3	69	58	86
1-Methylnaphthalene	3	49	43	53
Biphenyl	3	86	80	89
2,6-Dimethylnaphthalene	3	55	44	62
Acenaphthylene	3	43	39	50
Acenaphthene	3	37	34	42
2,3,5-Trimethylnaphthalene	3	88	80	100
Flourene	3	383	350	430
Phenanthrene	3	203	190	210
Anthracene	3	103	100	110
1-Methylphenanthrene	3	900	840	960
Fluoranthene	3	953	890	1000
Pyrene	3	527	510	550

Table V-13

Concentrations of Chemicals in Hudson River Sediment¹

CHEMICAL	ENGLEWOOD CLIFFS (SINCE 1992)			
	N	MEAN	MINIMUM	MAXIMUM
PAHs (µg/kg)				
Benzo(a)anthracene	3	587	570	600
Chrysene	3	590	540	660
Benzo(b)fluoranthene	3	480	430	530
Benzo(k)fluoranthene	3	427	420	430
Benzo(e)pyrene	3	537	530	540
Benzo(a)pyrene	3	493	490	500
Perylene	3	420	380	470
Indeno(1,2,3-c,d)pyrene	3	110	100	120
Dibenz(a,h)anthracene	3	380	360	400
Benzo(g,h,i)perylene	3	4	3	5
Benzo(b,k)fluoranthene	0	NA	NA	NA

¹Data from NOAA's National Status and Trends benthic surveillance.

ND = Non-detect

NA = Not analyzed

Commission's striped bass management plan and FDA's PCB ingestion thresholds, and consequently would have no significant effect on the environment. NY DOH sampling data, however, indicate that any fish that could permissibly be caught, sold, and consumed under DEC's regulations may contain PCB levels as high as 12.225 ppm, more than six times the permissible federal tolerance level (DOH 1990b).

The future status of Hudson River PCBs is generally uncertain. The EPA is currently evaluating plans for removing or neutralizing PCBs that remain in the river's sediments. Some doubts remain as to the effectiveness of removal processes and the extent of new or unknown sources. However, there have been some recent preliminary considerations within NYSDEC regarding the reopening of the in-river striped bass fishery. NYSDEC has recognized that PCB concentrations in Hudson River striped bass have been declining over time and in 1997 were less than the 2 ppm than in previous years. The analytical data over the past 25 years indicate that PCB concentrations in striped bass from the traditional spring fishery (the lower estuary, downstream of Poughkeepsie) have declined to levels below 2 ppm, the FDA action level for PCB. Model simulations run on the 1997 striped bass data suggests a steady decline in PCB concentrations that is independent of fish size. The simulation revealed, however, that male fish would typically have a higher PCB content than female fish. This sexual dichotomy, coupled with an observed increase in the number of females over males, are some of the additional factors contributing to the decrease in striped bass PCB concentrations.

Since the declining trend in PCB concentration is expected to continue in future years, NYSDEC has begun to consider whether the in-river commercial fishery could be reopened and what regulations would be required to maintain low PCB levels in the harvest, and how to allocate the allowed in-river catch among recreational and commercial fishermen.

j. Polycyclic Aromatic Hydrocarbons (PAHs)

There are hundreds of PAHs, many of which commonly co-occur in complex mixtures. PAHs, products of incomplete combustion, are common components of petroleum, and as such, are prevalent in the environment from both natural and anthropogenic sources. Most PAHs have no known use except for research purposes. However, a few PAHs are used in medicines, dye production, plastics, and pesticides, while others are contained in asphalt used in road construction.

PAHs commonly enter surface water through atmospheric deposition and from discharges of industrial effluents, municipal wastewater, and improper disposal of used motor oil (ATSDR 1993). Microbial metabolism is the major degradation process for PAHs

in both soils and sediments. PAHs can accumulate in aquatic organisms from water, sediment, and food. Bioconcentration factors (BCFs; ratios of the concentrations of a chemical in organisms relative to the concentrations in surrounding waters) are available for several species of aquatic organisms. BCFs for fish and crustaceans range from approximately 10-10,000 (Eisler 1987b). Biomagnification has not been observed in aquatic organisms because of the tendency of many aquatic organisms to metabolize PAH compounds rapidly (Eisler 1987b). The PAHs can be grouped, according to molecular weight, into classes of chemicals with similar biochemical properties. The PAHs of interest for this assessment are classified as follows:

Low-molecular-weight compounds (3 fused rings; 152–178 g/mol): acenaphthene, acenaphthylene, anthracene, fluorene, and phenanthrene. Although 2-methylnaphthalene and naphthalene have only two fused rings, they are included with the low-molecular-weight PAHs based on generally similar environmental chemistry and metabolism.

High-molecular-weight compounds (4 or more fused rings; 202–278 g/mol): benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-c,d)pyrene, and pyrene.

PAHs in the Hudson River

Twenty-four individual PAHs were found in Hudson River estuary sediments during NOAA's National Status and Trends program (Table V-13).

Health Advisories

Various of the organic and inorganic contaminants have lead to the issuance of health advisories regarding the consumption of fish and shellfish from the Hudson river (USEPA, 1999). Under the current health advisory for sportfish no person is to eat no more than one meal (consisting of one-half pound) per week of fish taken from the state's freshwaters and some marine waters at the mouth of the Hudson River. The health advisory includes New York waters of the Hudson River Upper Bay of New York Harbor (north of Verrazano Narrows Bridge), Arthur Kill, Van Kull, Harlem River and the East River to the Throgs Neck Bridge. The health advisory is set to protect against eating large amounts of fish that haven't been tested or contain unidentified contaminants. The advisory excludes most marine waters.

NY DOH warns against eating deformed or abnormal fish noted as such by the presence of tumors, lesions, or other abnormal conditions of the fish skin or internal organs. Health

advisories for fish and shellfish caught in specific locations or stretches of the Hudson River are summarized below.

Hudson River Health Advisories for Sherman Island Dam to Feeder Dam

A health advisory is in place for PCBs in fish caught from Sherman Island Dam downstream to Feeder Dam at the South Glens Falls. NY DOH advises women of child bearing age and children under the age of 15 not to eat any fish caught in this stretch of the river. NY DOH advises other people to not eat more than one meal per week of fish other than carp and to not eat more than one carp meal per month.

Hudson River Health Advisories for Feeder Dam at South Glens Falls to Hudson Falls

A health advisory is in place for PCBs in fish caught from Feeder Dam at the South Glens Falls to Hudson Falls. NY DOH advises that people eat no more than one fish meal, regardless of species, per week.

Hudson River Health Advisories for Hudson Falls to Troy Dam

A health advisory is in place for PCBs in fish caught from Hudson Falls to Troy Dam. NY DOH advises that no fish of any species at any time caught in this stretch of the river be consumed.

Hudson River Health Advisories for Troy Dam south to the bridge at Catskill

A health advisory is in place for PCBs in fish caught from Troy Dam south to the bridge at Catskill. NY DOH advises people, other than women of child bearing age and children under the age of 15, to eat no more than one meal per week of American shad and for no one to eat all other fish species caught in this stretch of the Hudson River. Women of child bearing age and children under the age of 15 should not consume fish species caught in this stretch of the Hudson River.

Hudson River Health Advisories for the bridge at Catskill south to Verrazano Bridge including Arthur Kill and Kill Van Kull

A health advisory is in place for PCBs in fish and PCBs and cadmium in crabs caught from the bridge at Catskill south to Verrazano Bridge including Arthur Kill and Kill Van Kull. NY DOH advises people to eat no more than one meal per month of American eel, Atlantic needlefish, Blue Crab, bluefish, carp, goldfish, largemouth bass, smallmouth bass, rainbow smelt, striped bass, walleye, white catfish and white perch. Women of child bearing age and children under the age of 15 should not eat any of the previously mentioned fish species, crabs or any other fish species caught in this stretch of the river. People are advised not to eat blue crab hepatopancreas (mustard, tomalley, or liver) and should discard all cooking liquids. Other people should eat no more than one meal per week of all fish species not mentioned previously.

4. Dissolved Oxygen

The type of organisms present and their pattern of distribution within an environment are determined in part by the concentration of dissolved gases in the water. The most important of these is oxygen, which decreases in solubility with rising temperature and decreasing barometric pressure. The concentration of dissolved oxygen (DO) is determined by the interaction of several biological and physical processes. Some processes, including photosynthesis, tidal mixing, respiration, microbial decomposition, and chemical oxidation, act directly by increasing or decreasing the amount of oxygen produced or used in the water. Others - temperature, salinity, and atmospheric pressure - act indirectly by affecting the solubility of oxygen in water. The interaction among these direct and indirect factors typically results in a seasonal DO pattern of highest concentrations during the winter and lowest during the summer and a diel pattern of highest DO during daylight and lowest during darkness. In highly stratified estuaries, a gradient of decreasing DO with depth may develop due to the reduced interchange between the surface fresh water, where DO is enhanced by photosynthesis and diffusion across the water surface, and the more stagnant bottom seawater, where DO is reduced by biological respiration and chemical oxidation in the sediment.

Aquatic plants simultaneously produce oxygen during the process of photosynthesis and use it in respiration. In illuminated nutrient-rich surface waters, plants produce more oxygen through photosynthesis than they consume by respiration. Nonphotosynthetic organisms derive energy for their life processes from oxidation of energy-rich molecules. In poorly lighted or unlighted portions of water bodies and at night, aquatic biological systems consume more oxygen than is produced by photosynthesis. In the Hudson River the suspended particulate matter usually limits light penetration to only a few feet.

The organic materials in the water column and sediment exert additional oxygen demand. These materials originate from both natural and artificial sources. Aerobic decomposition of aquatic plants and organisms and surface runoff represent natural sources; industrial and municipal wastes discharged into the estuary are artificially introduced sources of oxygen demand. The addition of municipal wastes to the lower Hudson River has influenced longitudinal variations in DO content. If the oxygen concentration of an area is not renewed periodically (by photosynthesis or currents), anaerobic conditions will be established and most aquatic organisms will die. The New York State standard for dissolved oxygen concentrations in non-trout Class A waters stipulates that the minimum daily average not be less than 5.0 mg/l; and that no time should DO levels fall below 4.0 mg/l. For Class SB waters, DO levels must remain above 5 mg/l at all times and for Class D waters 3 mg/l.

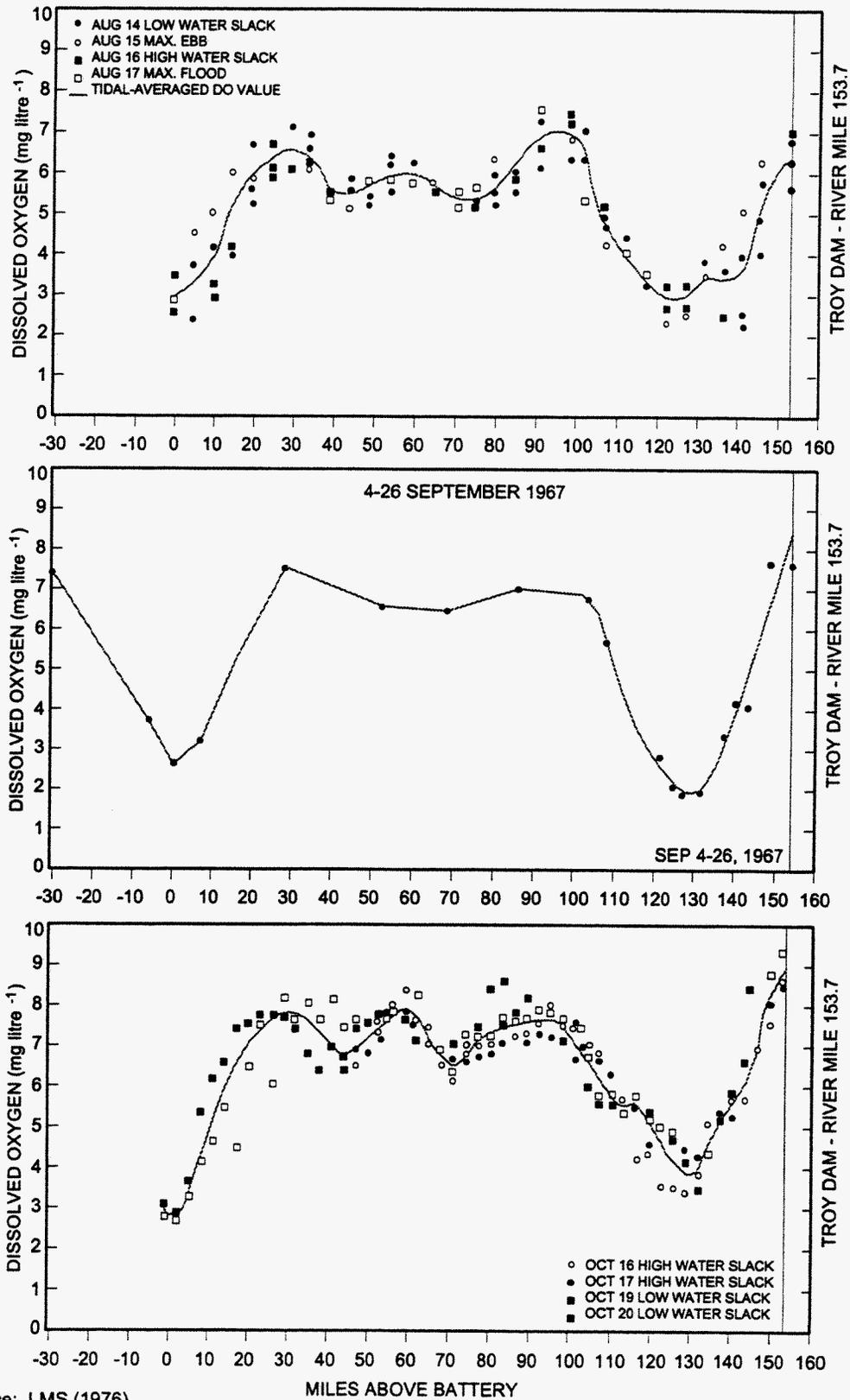


Figure V-24. Hudson River Estuary dissolved oxygen profile, 1967.

V. Environmental Setting

Many observations of Hudson River DO concentrations have been made at different locations and times by various governmental, state, and private institutions. These measurements provide the basic information on river DO levels. Three surveys conducted by Quirk, Lawler & Matusky Engineers in August, September, and October 1967 provide comprehensive historic DO concentration data for the Hudson River estuary. The longitudinal DO profiles in Figure V-24 are based on these 1967 data. DO was generally above 6 mg/l except in urban areas having significant wastewater discharges and New York City. Relatively high DO occurred from Saugerties to Croton Point. Downstream of this site, along Manhattan Island, DO concentrations were low.

Average regional DO values over the period 1974-1987 indicate low average concentrations in the Albany region (RM 125-152) and the Yonkers region (RM 12-23 [LMS 1989]). Only the Yonkers region, however, yielded summer DO values regularly below 4 mg/l, and then only prior to 1983. Highest average DO concentrations were observed in the regions from Kingston through Catskill.

The 1974-1987 data display the seasonal cycle of DO. Over the year, average DO concentrations were generally highest in February and April and lowest in July through September (Figures V-25 and V-26). In the Yonkers region peak DO concentrations averaged approximately 12 mg/l and decreased to about 5 mg/l in the summer. Peak DO concentrations were relatively constant at 12-13 mg/l throughout the river. Summer DO values tended to be higher upriver, approximately 7-8 mg/l in the Kingston region, although they were about 6.5 mg/l in the Albany region (Figures V-25 and V-26).

Analysis of residual DO values, i.e., the difference between the average weekly DO concentrations for the region over the period 1974-1987 and each individual observation from the corresponding week, indicates that the middle estuary regions are least variable in DO concentration. In the regions from Indian Point Units 2 and 3 through Hyde Park 95% of the DO values fell within approximately 1.6 mg/l of the weekly average. In the more variable Yonkers and Albany regions 95% of the observations fell within 2.3 and 2.5 mg/l, respectively, of the weekly average.

River-long variations in DO for different seasons are shown in Figure V-27 for 1988 and Figure V-28 for 1991. As measured in Long River surveys, DO values in the channel of the upper river regions are typically higher than those in the channel of lower regions, but this trend is not consistent through all years or all seasons. These figures demonstrate that the historic DO sag in the Albany region has been alleviated by improvements to the quality of wastewater discharges and that oxygen levels meet water quality standards. Levels are sufficient to support the indigenous biota.

V. ENVIRONMENTAL SETTING

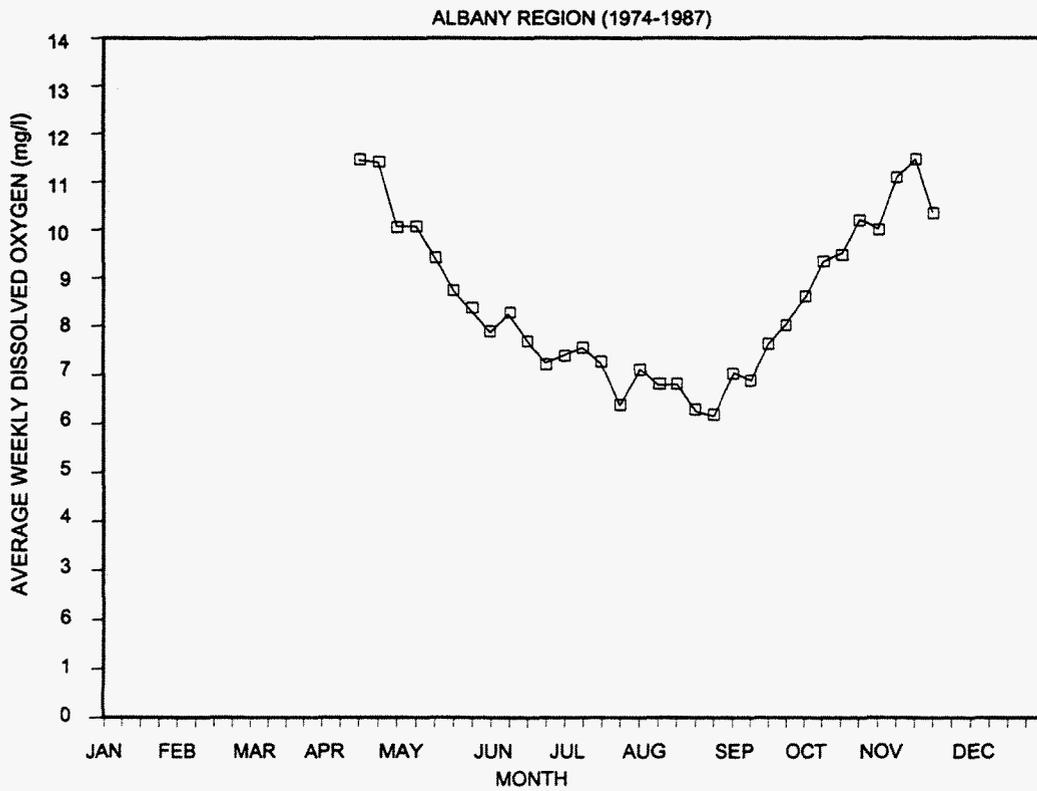
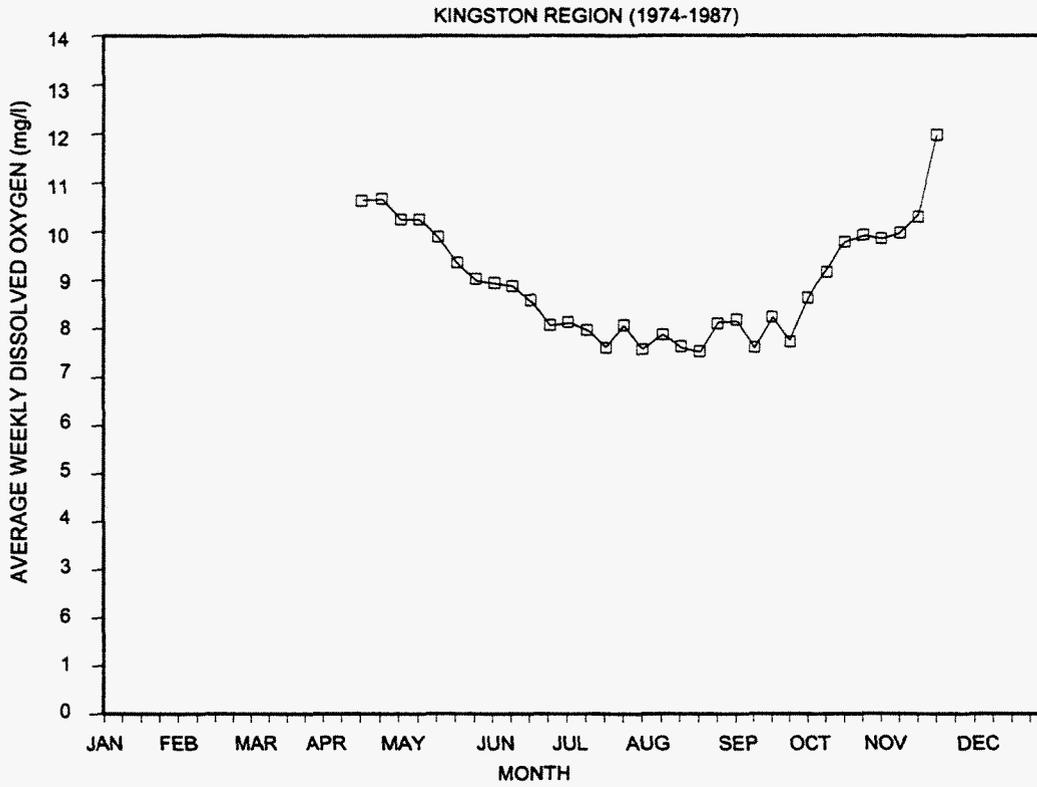


Figure V-26. Average weekly dissolved oxygen vs time.

V. ENVIRONMENTAL SETTING

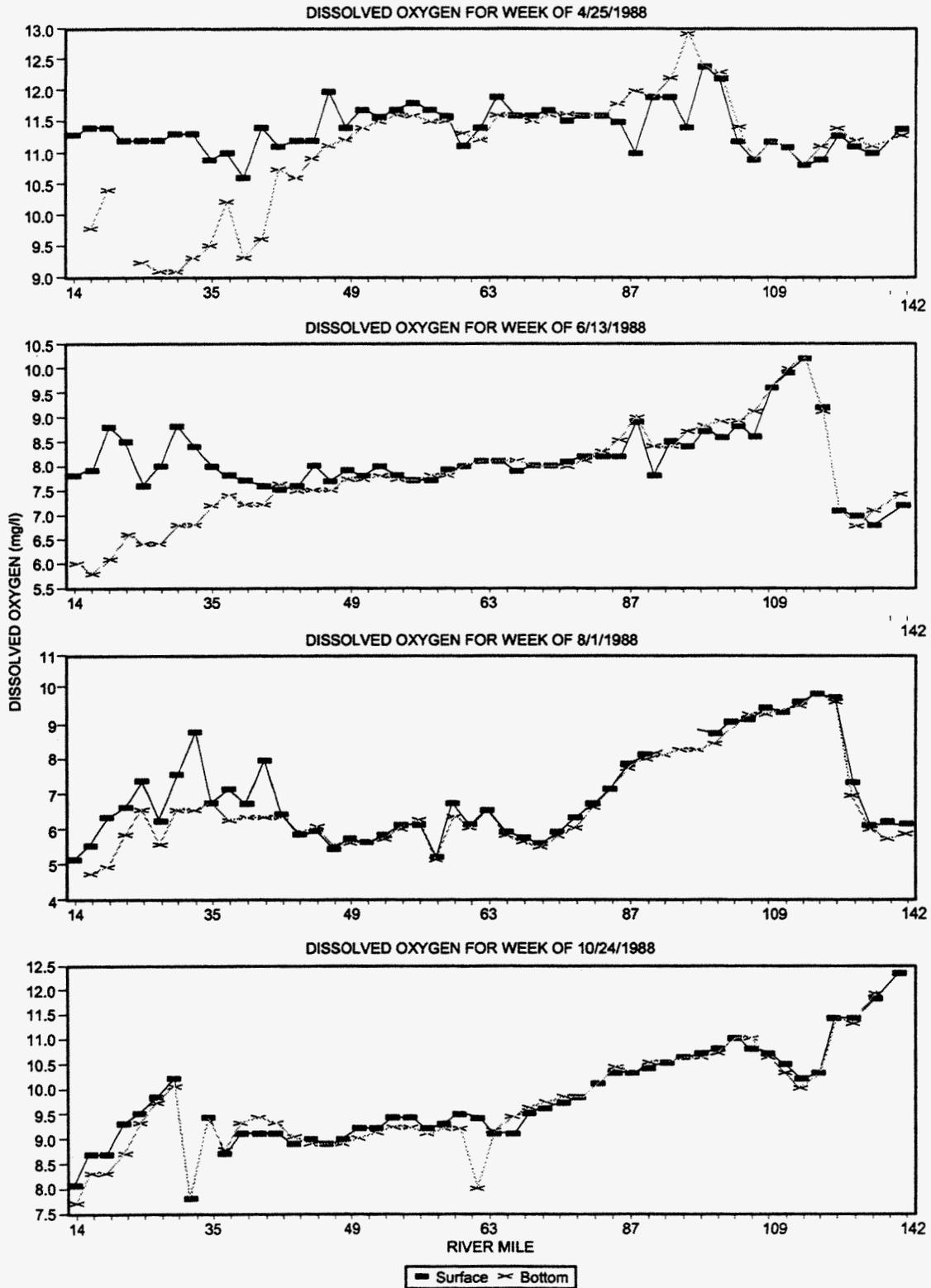


Figure V-27. 1988 longitudinal variation in dissolved oxygen.

V. ENVIRONMENTAL SETTING

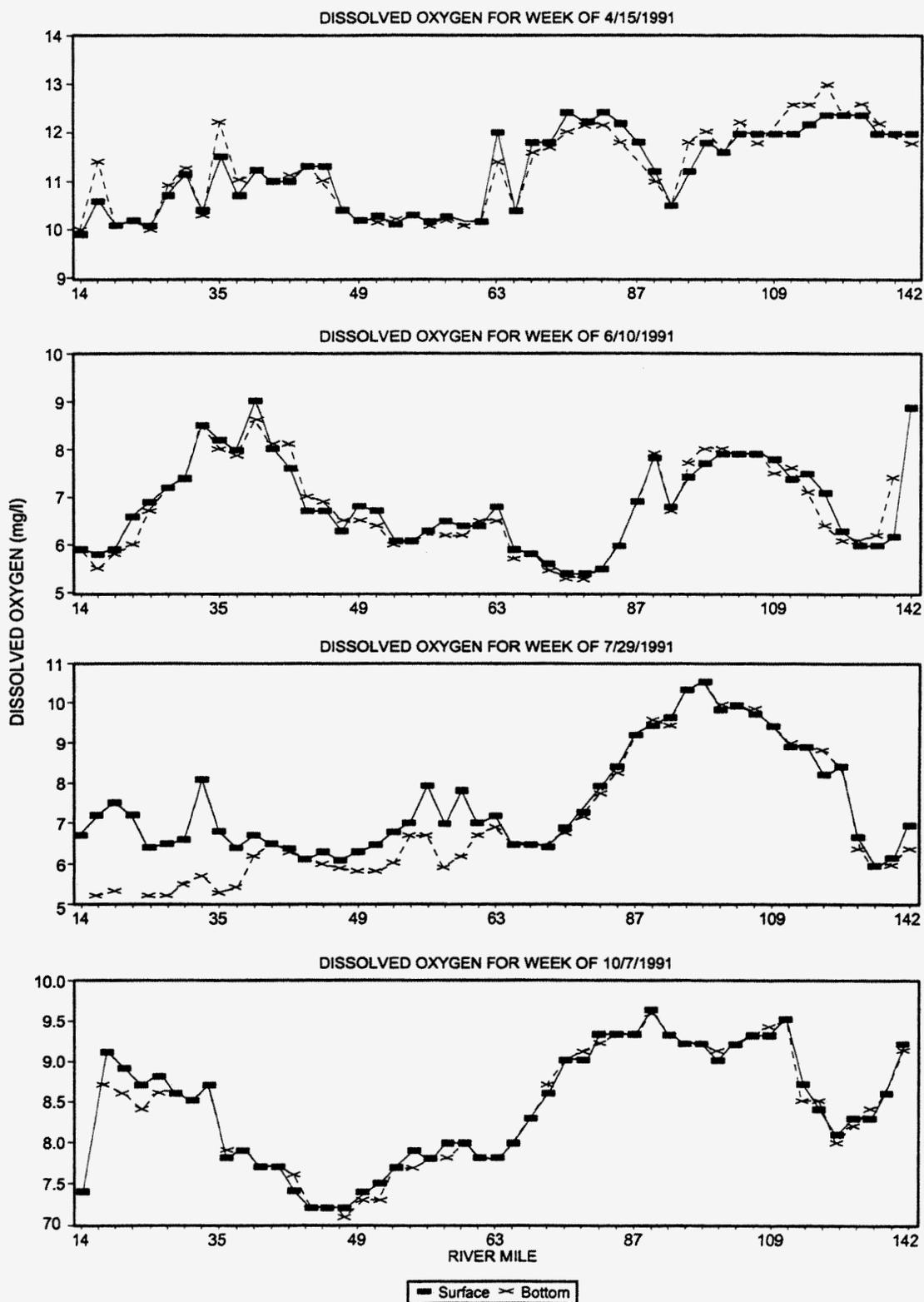


Figure V-28. 1991 longitudinal variation in dissolved oxygen.

V. Environmental Setting

DO and percent saturation are often higher in the shore zone than the deeper areas measured as part of the LRS/FSS program. Mean weekly DO reached peak values of approximately 10 mg/l on four dates in the BSS (weeks of June 24, September 2, and October 28, 1991). Saturated and supersaturated conditions existed during these and other periods. Previous reports have also noted supersaturation in the shore zone even in the summer months (MMES 1986; EA 1990, 1991). Turbulence from wave action and oxygen released as a by-product of photosynthesis could be two causes of this supersaturation.

DO may differ slightly from the surface to the bottom. This phenomenon, called stratification, is particularly noticeable in the lower river during low-flow months. Stratification in the Hudson is associated with the presence of higher-salinity ocean waters along the bottom.