Estuaries and Coasts: J CERF (2008) 31:85–100
DOI 10.1007/s12237-007-9010-y

Temporal and Spatial Variations in Water Quality on New York South Shore Estuary Tributaries: Carmans, Patchogue, and Swan Rivers

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Received: 9 April 2007 / Revised: 23 August 2007 / Accepted: 14 September 2007 / Published online: 3 January 2008

Abstract The chemical and biological impacts of anthropogenic physical modifications (i.e., channelization, dredging, bulkhead, and jetty construction) to tributaries were assessed on New York's Long Island South Shore Estuary. Water-quality data collected on Carmans, Patchogue, and Swan Rivers from 1997 to 2005 indicate no significant differences in nutrient levels, temperature, or pH among the rivers, but significant differences in light transmittance, dissolved oxygen (DO), salinity, and sediments were observed. Patchogue River (PR) and Swan River (SR) were significantly more saline than Carmans River (CR), PR and SR had less light transmittance than CR, and both exhibited severe warm season hypoxia. CR was rarely hypoxic and only at the lower layer of the deepest station in warm seasons. Deep stations on PR had hypoxic readings year round, but the shallower SR was well-oxygenated at all stations after the fall turnover. There were wide diel and seasonal variations in chlorophyll *a* on each river, and measurements were significantly higher at poorly flushed stations. In warm seasons, this often resulted in hyperventilation with supersaturated DO in the upper water column on sunny days, and suboxic conditions at nights and/or in deeper layers. PR sediments were anoxic, SR sediments ranged from normal to anoxic, and CR sediments were normal at all stations. Polyaromatic hydrocarbon concentrations in PR sediments were over three orders of magnitude higher than SR and CR sediments. Benthic invertebrate assessment of species richness, biotic index, and Ephemeroptera, Plecoptera and Trichoptera richness indicated that PR was severely impacted, SR ranged from slightly to severely impacted, and CR ranged from non-impacted to slightly impacted. Diversity and abundance of plankton were comparable on SR and CR, and were significantly higher than on PR. The data indicate that nutrients do not play a major role in hypoxia in these estuarine tributaries but that physical forces dominate. The narrow inlets, channelization, and abrupt changes in depth near the inlets of PR and SR foster hypoxic conditions by inducing salinity stratification that limits vertical mixing and by restricting horizontal water mass exchange with the bay. The study suggests that other tributaries with such physical modifications should be examined to assess the temporal and spatial extent of hypoxia.

Keywords Estuary · Dissolved oxygen · Tidal flushing · Stratification · Benthic · Sediment

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Introduction

The South Shore Estuary Reserve (SSER) of Long Island, New York, extends 120 km from its western boundary at Reynolds Channel in Hempstead Bay to its eastern boundary in Shinnecock Bay (Fig. 1). The estuary has a long history of economic, social, and ecological importance, and is located in a densely populated suburban area with approximately 1.5 million people living within its drainage area. It contains the most extensive acreage of tidal wetlands and the greatest diversity of habitat in New York State (Pataki and Daniels 1997), and serves as nursery and feeding grounds for a wide variety of fish and

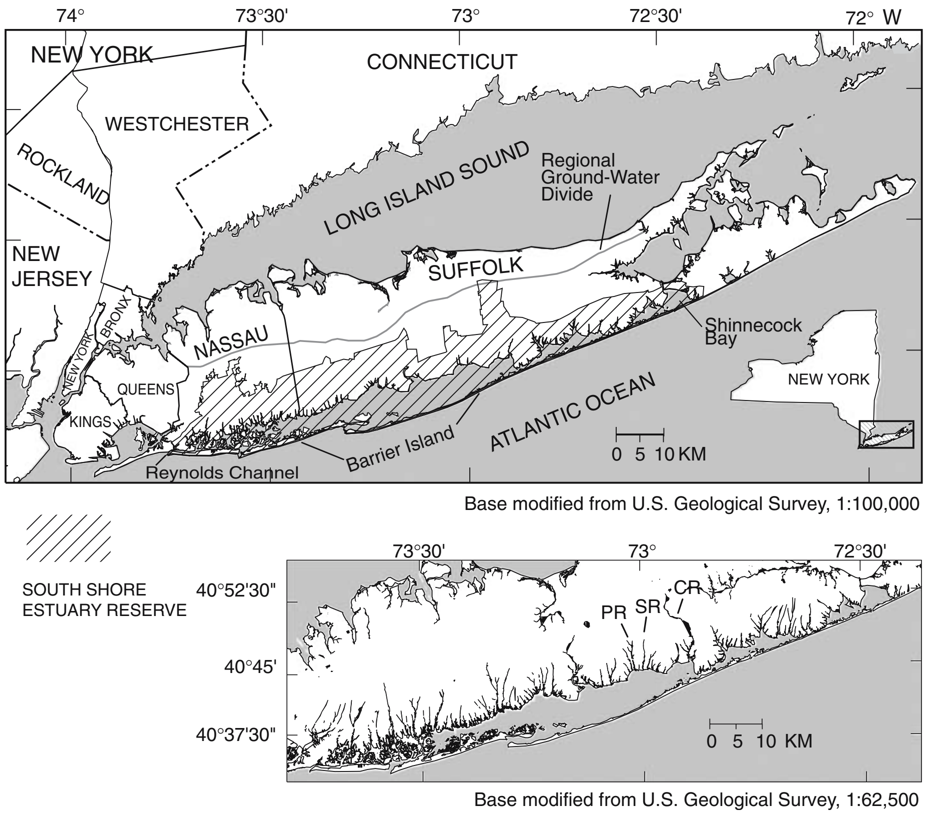

Fig. 1 The tributaries Patchogue River (*PR*), Swan River (*SR*), and Carmans River (*CR*) are located on Long Island in the New York South Shore Estuary Reserve. Modified with permission from USGS (Monti 2003)

birds, including threatened species such as Piping Plover, Least Tern, and Osprey. The Reserve extends from the mean high-tide line on the ocean side of the barrier islands, northward to its shallow interconnected bays, and upland to the headwaters of its tributaries. South Shore Estuary tributaries follow glacial meltwater channels that run southward from the Ronkonkoma moraine groundwater divide through the outwash plain to the bays and, today, contain groundwater outflow and surface water runoff.

To establish a scientific basis upon which management decisions may be made, the SSER Comprehensive Management Plan (CMP) calls for monitoring that includes identifying and assessing trends in water quality and living resources (Pataki and Daniels 2001). As tidal sections of tributaries are important as nurseries and as sources of nutrients and pollutants to the bays (Roman 2000), we sought to examine the status of three representative SSER tributaries. To our knowledge, there are no articles in the scientific literature that assess water quality and living resources in the tidal sections of SSER tributaries. One goal of this study was to obtain baseline data for the tributaries that could serve as benchmarks for future studies on other SSER tributaries. A second goal was to assess the impacts of tributary modifications such as dredging, channelization, and bulkhead construction on water quality and living resources. We hypothesized that such physical changes would reduce the species richness and diversity of aquatic biota by causing chemical changes in the tributaries. The data and assessment presented herein can be used to inform management and policy decisions concerning the effects of such tributary modifications on the exchange of water, dissolved oxygen dynamics, salinity, and aquatic biota. The study has implications for improving the ecological health and economic and aesthetic value of estuarine tributaries.

One of the most pressing threats to estuaries nationwide is diminished water quality as a consequence of urban growth and land use that increases the impervious surface area and non-point source runoff of nutrients and sediment (Choi and Blood 1999; Cloern 2001). Non-point sources in residential development areas can have larger negative impacts on water quality than urban point sources (Atasoy et al. 2006). Non-point source pollution from stormwater runoff is thought to be a leading cause of poor water quality in the SSER (Pataki and Daniels 2001). For example, the closure of 34,643 acres of hard clam beds and a 93% decrease in clam harvest was attributed to sediment, bacteria, contaminants, and excessive nutrients in runoff (Pataki and Daniels 1998). However, the major differences in water quality and living resources observed among the three rivers in this study are not caused by nutrient inputs or non-point source pollution but rather by tributary modifications that hinder tidal flushing and, thereby, foster hypoxia. Our data on the tidal sections reveal year-round hypoxia on Patchogue River (PR), warm season hypoxia on

Swan River (SR), and occasional warm season hypoxia at the deepest areas on Carmans River (CR). Hypoxia on PR and SR is persistent enough to severely impact bottom sediments and fauna. Although studies of hypoxic marine waters and benthic communities indicate that hypoxic environments occur naturally, hypoxia is becoming more prevalent in estuaries because of anthropogenic activities (Newcombe and Horne 1938; Seliger et al. 1985; Diaz and Rosenberg 1995; Karlsen et al. 2000). Tributaries with a two-layer pattern of circulation are particularly susceptible to hypoxia (Officer et al. 1984), and this is the case in SSER tributaries in which an upper layer of less dense fresh water flows toward the bay over a lower layer of much denser saline tidal water. The lack of wind and wave mixing on the tributaries fosters formation of thermoclines and haloclines, especially in the warm season, and traps hypoxic water at the bottom of such tributaries (Borsuk et al. 2001). Narrow inlets to the bay on PR and SR prevent tidal flushing of hypoxic river bottom water with oxygenated bay water and further exacerbate the problem. Our data indicate that the major factors contributing to low dissolved oxygen (DO) levels on these rivers are physical: deep channels that foster stratification and limit vertical mixing, and narrow inlets that hinder tidal flushing. Future dredging and inlet projects should consider the impact on DO and aquatic biota of increasing depths or narrowing inlets.

Study Location

The Carmans, Patchogue, and Swan Rivers are located within 8 km of each other near the center of the Reserve, had similar geomorphology before development, and are representative of the range of anthropogenic impacts found throughout the estuary. The major hydrogeographic differences today are because of dredging, filling, and construction of bulkheads and dams that modified their shorelines and their respective tidal zones, depths, and river mouth widths (Fig. 2). The CR tidal zone diminishes gradually, whereas PR and SR tidal reaches end abruptly at road crossings just north of station 1. The mouth of CR is over seven times wider than the modified inlet of PR (50 m), and the PR inlet is five times wider than the SR inlet (10 m). Only the CR inlet allows efficient tidal flushing.

CR has one of the last remaining undisturbed habitats in the estuary and provides baseline data to which other SSER tributaries may be compared. Much of the upstream reach is in Southaven County Park, and the tidal section meanders through the protected lands of Wertheim National Wildlife Refuge. CR is bordered by some residential property, primarily in the northern reach. Duck farms located near the river until the 1970s had a dramatic effect on water quality because of high nutrient levels from fecal matter (Pataki and Daniels 1999). A 1938 survey of the fresh-

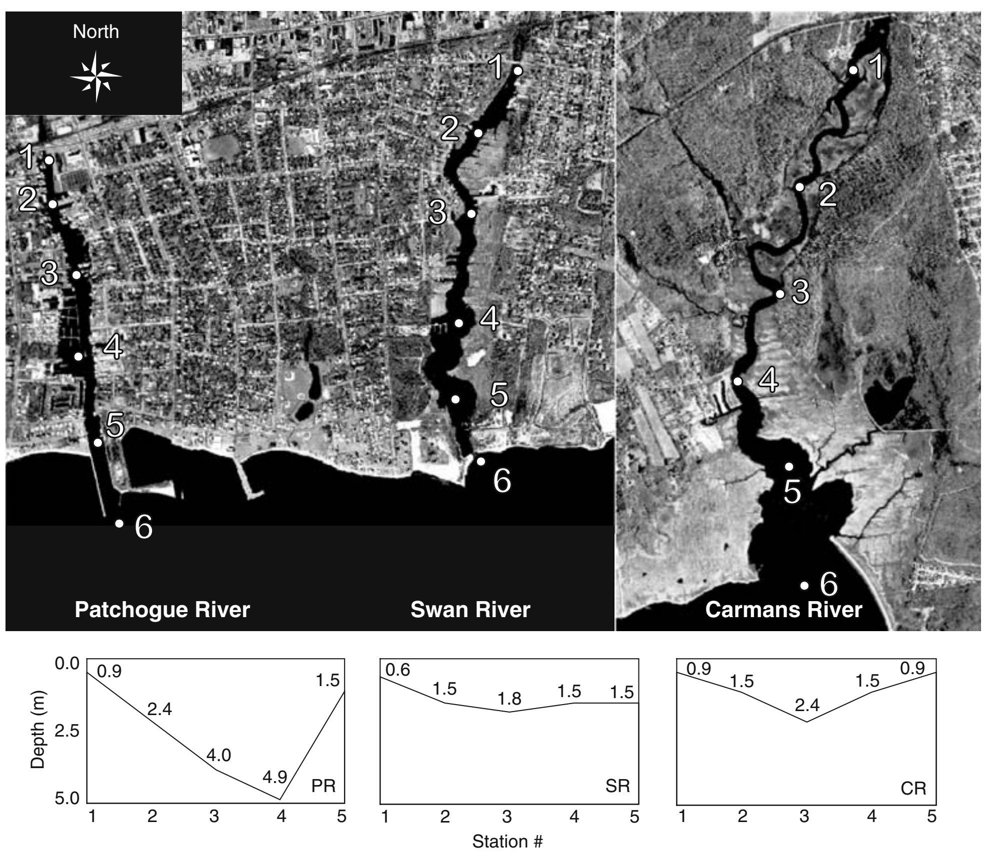

Fig. 2 Tidal sections of PR, SR, and CR; aerial digital orthophotos (New York State, 1996). *Below* Mean low tide river channel depth in meters at each tidal section station. Stations 6 have open bay characteristics

waters of Long Island reports a drop in DO in the CR tidal section surface samples from 10.3 ppm above the duck farms to 0.9 ppm below the farms and an increase to just 3.6 ppm above the mouth (Moore 1938). At 13 km from headwaters to bay, CR is the longest SSER tributary and has the largest drainage area (184 km^2). More than 90% of CR is considered by the New York State Department of Environmental Conservation (NYSDEC) to be wetlands and exhibits diverse native flora and fauna. The 11-km freshwater section has several lakes and bogs and is surrounded by pine–oak forest. The CR basin is home to over 100 species of plants, 40 species of fish, 240 species of birds, and 30 species of mammals (Borg and Shreeve 1974). Undeveloped marshes in the tidal section consisting of *Spartina alterniflora, S. patens*, and *Phragmites australis* provide plentiful nesting habitat for birds and a nursery for other marine life. Upland of the marsh is hardwood swamp, pine–oak woodland, and patches of grassland/shrubland.

PR has been a maritime center since colonial times and is listed as one of the six major maritime centers in the SSER CMP (Pataki and Daniels 2001). The tidal section of the river was significantly modified to accommodate large vessels when Patchogue was a major shipping port. It has been under federal jurisdiction since 1890, when the US River and Harbor Act required the US Army Corps of Engineers to dredge and maintain the waterway for use as a commercial center. There was heavy usage of the river in the early 1900s by building material transporters, paper mills, cotton factories, gristmills, fishermen, and coal transporters (Jones 1999). Remnants of this early industry remain, as coal particles are still present in river sediments today. Although industry on PR has diminished since the 1970s with the closure of the river's oil terminals, river usage is high, with a 20% increase in the number of boat slips available and used since 1977 (Jones 1999). Today, almost 100% of the 1.3-km-long tidal zone contains bulkhead, and no part of the river is classified as wetlands by NYSDEC. Bulkhead and jetty extend about 0.2 km into the bay beyond the shoreline. The surrounding community is a developed suburban residential and business area with a high percentage of impervious areas. From its 35 km^2 drainage area, the river receives point and non-point source pollution from residential and commercial areas and the Patchogue Village secondary treatment sewage plant. A second sewage treatment plant is located at an apartment complex near the mouth of the river. These plants operate under the State Pollution Discharge Elimination System (SPDES). Patchogue Village sewage treatment plant has an average daily flow of 1,230 m^3 per day, which is about 2.5% of PR average freshwater flow (Jones 1999). The plant generally operates within effluent limits with the exception of total residual chlorine, which is self-reported at 2.25–2.5 ppm on an occasional basis, exceeding the SPDES permit maximum of 2.0 ppm (SPDES permit NY0023922). Average total suspended solids discharge is 23 ppm, average biological oxygen demand is 9.0 ppm, and average total coliform is 11 mpn/100 ml (Suffolk County Department of Health Services 1998). The apartment complex sewage treatment plant discharges an average volume of 64 m^3 per day, average total suspended solids of 24.5 ppm, average biochemical oxygen demand of 11.5 ppm, and average total coliform of 1.0 mpn/100 ml (SPDES permit NY0080730). PR has been a significant source of coliform bacteria loadings to Patchogue Bay (Long Island Regional Planning Board 1982) that have kept the northern Bay closed to shellfishing since 1910 (Dennison et al. 1991). PR was designated a Priority Water Body in 2002 by the NYSDEC (2004).

The level of development on SR is intermediate between CR and PR. SR flows through mixed residential, commercial, and undeveloped areas with woods, bogs, and marshes, has a drainage area of 22 km^2, a narrow inlet that limits tidal flushing, and a narrow channel with an average maximum depth of 1.8 m. Impervious surfaces in the SSER drain directly into the estuary via stormwater drains. Although no data were available on the impervious surface area in the drainage basins of CR, PR, and SR, we estimated the relative impervious area by assigning PR a value of 1. The value for CR was approximately 0.1 and for SR about 0.4. SR follows its natural winding course over much of its length, and less than half of its shoreline has bulkhead. Sea-run brown trout spawn in the 2-km tidal section from September to November, and the freshwater section supports native brook trout, one of only a few such naturally reproducing populations on Long Island.

Materials and Methods

Five tidal sampling stations were designated on CR, PR, and SR from the northernmost tidal reach to the mouth of the river (Fig. 2). A sixth station was established in the bay near the inlet of each river. The mean low-tide depths at each tidal station during the 1997 to 2005 sampling period are indicated in Fig. 2. In the first year of the study, transect data collected at west (W), center (C), and east (E) locations at each station indicated that central readings provided representative data, with the exception of PR stations 1 and 4 at high tide on October 2, 1997. On that date, there were significant differences in transect values at station 1 in salinity (p=0.0012) and DO (p=0.0477), and station 4 in salinity (p=0.0224). A more thorough examination of PR transects was performed over the PR tidal cycle on October 18, 1998 and indicated no significant difference in west, center, and east salinity and DO values (p>0.05 in all cases). Hence, data in later years were obtained primarily at central stations, and central station data are used in many of the subsequent statistical analyses. To assess the quality of upstream freshwater coming into the tidal area, sampling was performed at established United States Geological Survey (USGS) gauging stations in the freshwater reaches: PR at Patchogue station #01306000, SR at East Patchogue station #01305500, and CR at Yaphank station #01305000 and at Southhaven station #01305040.

Chemical and Physical Measurements

Vertical profiles of temperature, salinity, and dissolved oxygen were obtained with in situ measurements at 0.3-m depth increments with a Yellow Springs Instruments (YSI) dissolved oxygen meter model 58 (accuracy of ±0.1 ppm) and a YSI model 335-C-T conductivity meter (accuracy of ±0.2 ppt). Oxygen measurements were corrected for salinity. The pH was measured with an Orion portable meter model 230 A. Meters were calibrated before analyses each day. The oxygen meter was calibrated using the air-saturation method daily in the field and periodically checked in the lab for correct calibration using the Winkler titration method (Clesceri et al. 1998). Oxygen levels were assessed based on New York State water quality standards, which require 5.0 ppm dissolved oxygen (NYSDEC 2000). Hypoxic conditions are defined in this study as <5 ppm and anoxic conditions as <2 ppm. Light transmittance was measured using a Secchi disk. Maximum tidal flood current values were determined from field measurements obtained midway between low and high tides.

Water samples were collected at the surface and near the bottom using a Van Dorn apparatus and were either analyzed immediately in the laboratory or were stored at 4°C until analysis of nitrate, nitrite, ammonia, phosphates, chlorine, and sulfide by colorimetric methods (Clesceri et al. 1998). Nitrite was converted to nitrate with cadmium before analysis. Samples for chlorophyll *a* determination were obtained with a Van Dorn bottle at the surface and at 0.6-m depths, transferred to brown bottles, and analyzed spectrophotometrically by the method of Clesceri et al. (1998). Sediment samples were collected using wide-mouth sediment corers and chilled to 4°C until analysis. Physical observations included particle size, shape, organic content, and sediment sorting. Sediments were soxhlet extracted with methylene chloride and concentrated with a Kuderna–Danish apparatus (Clesceri et al. 1998). Activated copper powder was used to remove elemental sulfur. The percent mass of extracted organics was determined, and preliminary analyses were conducted by ultraviolet and Fourier-transform infrared spectroscopy. Gas chromatography/mass spectrometry (GC/MS) analyses were performed on extracts from 15 g sediment samples in EI mode at 70 ev on a Varian Saturn 2100T GC/MS equipped with a DB-5 column. Deuterated internal standards were used for PAH quantification: d8-naphthalene, d10-acenapthene, d10-anthracene, and d12-benzo[a]pyrene. Compounds were identified by comparison to retention times and mass spectra of authentic samples obtained from the National Institute of Standards and Technology Standard Reference Materials 1491 and 1941a.

Biological Measurements

Plankton samples were collected at odd-numbered stations using a US sieve #29 phytoplankton net. Samples were analyzed immediately upon return to the laboratory or were preserved with 4% acidified Lugol's iodine solution until analysis. Cell counts were performed on each sample, and identification of zooplankton and phytoplankton was noted to order classification (Tomas et al. 1997). Benthic macroinvertebrates were collected in tidal areas with a grab sampler and wide-mouth sediment corer, and organisms were screened after preservation in 90% ethanol. The assessment scheme of Gray and co-workers (1992, 2002) was used to evaluate sediment samples as normal, slightly eutrophic, severely eutrophic, or anoxic. At freshwater stations, benthic invertebrates were collected with an aquatic net by the kick-sampling method using the travel-kick method for 5 min, moving 1 m/min (Abele et al. 1994). Samples were preserved in 95% ethanol and analyzed immediately upon return to the lab. Species richness, EPT richness (based on Ephemeroptera, Plecoptera, and Trichoptera), and biotic index (Hilsenhoff 1987) were determined and evaluated by the invertebrate criteria for slow sand/gravel streams developed by NYSDEC (Bode et al. 1990, 1991, 2000).

Statistical Analysis

Descriptive statistics of the measured variables and the correlations among those variables were generated using PROC MEANS and PROC CORR implemented with SAS software (SAS Institute 2003). Where appropriate, means were compared using one-way analysis of variance (ANOVA) with PROC GLM (SAS Institute 2003). The dependency of DO on salinity was evaluated with a regression analysis again using PROC GLM. The initial portion of that analysis used a covariance approach to see if there was interdependence of salinity and temperature class. Box and whisker plots of DO data were generated by Statistica (StatSoft 2006) to represent seasonal differences by river (Fig. 3a), station differences by river (Fig. 3b), and depth differences by river (Fig. 3c). Warm and cold seasons were established by separating the year into two equal parts, using the fall turnover that occurs in mid-October as a dividing point.

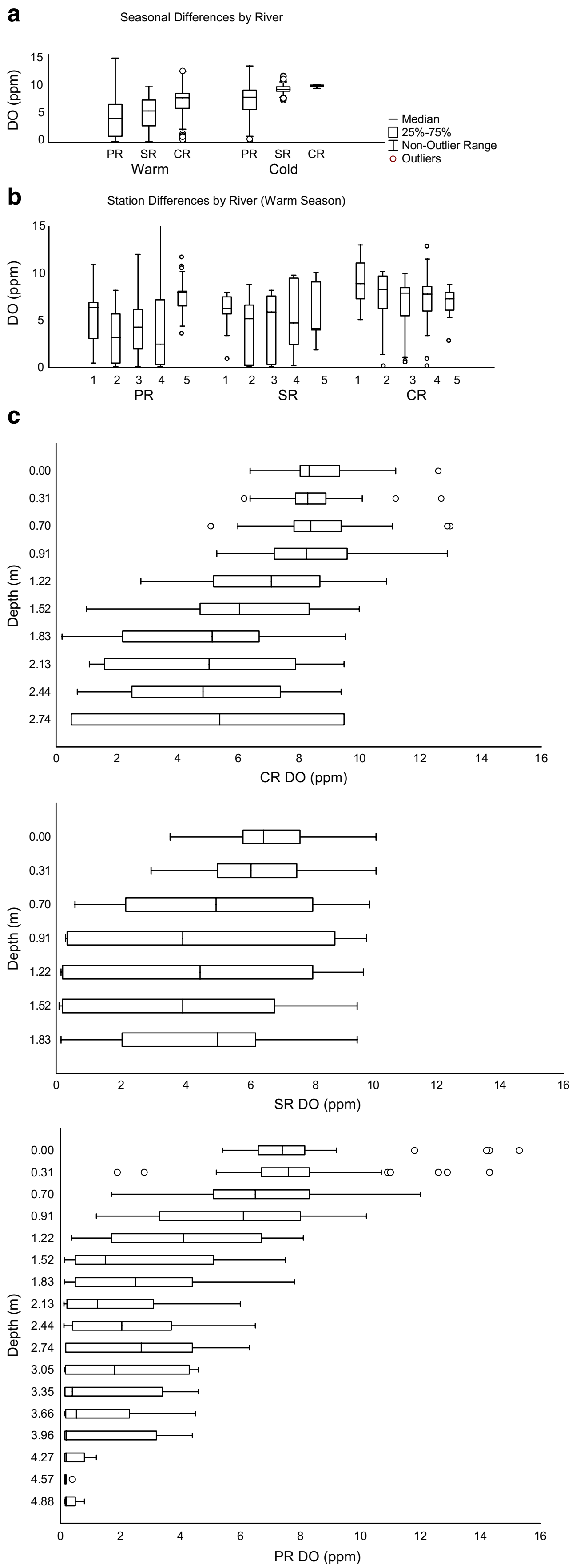

Fig. 3 *Box and whisker plots* of all observed dissolved oxygen values (DO in parts per million) on PR (n=625), SR (n=184), and CR (n=293) from 1997–2005 for **a** seasonal differences by river in warm months (April 15 to October 14) and cold months (October 15 to April 14), **b** station differences by river from fresh water (station 1) to tidal mouth (station 5) for warm season DO values, **c** depth differences by river for warm season DO values. Median is represented by the *horizontal line in the box*; the *box* represents the 25th to 75th percentile range; *whiskers* indicate data values within 1.5 times the interquartile range, and outliers are data values greater than 1.5 times the interquartile range

Visualizations

The visualizations in Fig. 4 were generated with the assistance of the Visualization Toolkit (Schroeder et al. 2002), a C++ class library used for developing custom visualization software systems. A total of 15 datasets were generated from the samples collected in the field. Along the vertical direction, samples were placed equidistantly in correspondence with the sampled data, which, in general, were taken at 0.3-m increments. To make use of several samples that were taken at 0.15-m depths, all data values were linearly interpolated along the vertical direction at 0.15-m increments.

Results

Chemical and Physical Measurements

A comparison of incoming freshwater and tidal water on PR, SR, and CR is shown in Table 1. The USGS gaging stations approximately 1 km north of PR and SR station 1 provided reliable estimates of freshwater influx, but the CR gaging station with the most continuous streamflow record is located about 5 km north of CR station 1 and under-estimates freshwater entering the study area. The maximum tidal flood current was highest at the narrow inlet of SR, and a large drop in current between PR stations 4 and 5 resulted from the wide basin at station 4. CR had the most efficient tidal flushing through its wide mouth. For all three rivers, nearby open bay stations typically had DO levels above 6 ppm in all seasons and at all depths because of efficient mixing of DO by wind and wave action in the shallow bay. Bay water entering the rivers with each incoming tide was usually well-oxygenated, as was incoming freshwater upstream of each station 1, which had DO levels above 6 ppm in all seasons and at all depths. An exception to this pattern occurred in the hottest days of summer, when measurements were taken in early morning after nights with calm winds and seas. In such cases, low morning DO levels rose substantially by afternoon. DO levels in Bellport Bay on August 22, 2005, for example, were representative of that pattern. The surface and bottom (1.5 m) DO values were only 4.9 and 1.8 ppm, respectively,

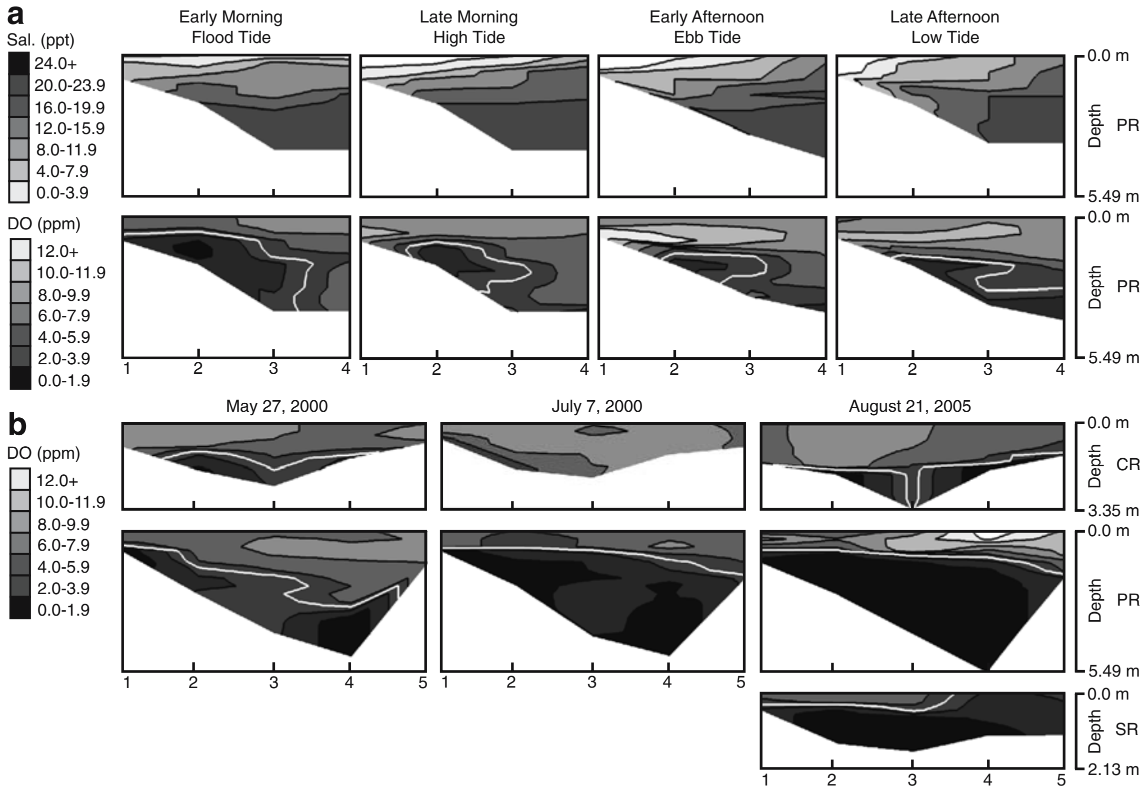

Fig. 4 Salinity and dissolved oxygen (*DO*) visualizations. Depth is on the *y*-axis, and station number is on the *x*-axis. The *white isoline* indicates the 5 ppm DO level. The *black regions* indicate water with a DO level of 2 ppm or lower. **a** PR tidal cycle salinity and DO collected on October 18, 1998. **b** Comparison of DO concentrations of CR and PR on May 27, 2000 and July 27, 2000, and on CR, PR, and SR in August, when DO levels are lowest

in early morning. By the late afternoon, the surface and bottom DO values had increased to 7.0 and 5.9 ppm.

DO patterns during warm seasons (April 15–October 14) were more complex and are presented separately from cold seasons data. The seasonal DO trends on each river are indicated in box and whisker plots in Fig. 3a. Mean DO levels and standard deviations across all stations and depths in the warm season for CR, PR, and SR, respectively, were 7.5±2.5, 4.3±3.3, and 5.3±3.2 ppm, and in the cold season (October 15–April 14) are 10.2±0.2, 7.6±2.7, and 9.7±0.8 ppm. The fall turnover that results in higher DO levels at lower depths of the tidal sections was observed by mid-October on CR and SR at all stations, and by the end of October on PR at all but station 4. CR and SR did not exhibit hypoxia at any depths during the cold season in any year, but PR hypoxia endured in the lowest quartile depths at station 4 throughout the winter. With this exception, there were no differences in DO by station on each river during the cold season.

Warm season differences by station and by depth are shown in Fig. 3b and c. CR occasionally experienced hypoxic conditions at stations 2, 3, and 4 at bottom depths in the warm seasons only, but most data indicate sufficient oxygenation. SR was hypoxic and anoxic in the summer months. Hypoxia at SR station 3 is exacerbated by the adjoining canal that harbors stagnant water with low DO. Throughout the summer, the canal had anoxic levels from 0.6 m below the surface to the bottom depth of 1.8 m. By the end of September, anoxia was only observed at the bottom, by mid-October oxic conditions were restored, and by May, the onset of hypoxia began again. The most

Table 1 Freshwater influx at stations 1 in cubic meters per second (±0.05) based on average of mean annual freshwater discharge values at upstream USGS continuous record gaging stations

Station no.	PR	SR	CR
1[a]	0.57[b]	0.35[c,d]	0.68[c,e]
4	0.16	0.33	0.32
5	0.34	0.35[f]	0.39

Average maximum tidal flood current at stations 4 and 5 in meters per second (±0.05).

[a] Based on Streamflow Statistics for the Nation at USGS National Water Information System (http://www.waterdata.usgs.gov/nwis).

[b] PR gaging station did not maintain a continuous record during the study period; value based on pre-study continuous record annual means.

[c] Based on 1997–2004 data

[d] Gaging station 01305500

[e] Gaging station 01305000

[f] Average value at the narrow SR inlet just north of station 6 was 0.78 m s^{-1}.

persistent low DO values were observed on PR at stations 2 and 4. May and June data indicated hypoxic conditions at all PR stations, with anoxic levels at PR station 4 from 3.3 m below the surface to the bottom depth of 4.8 m. By early July, there was usually a marked increase in anoxia, with PR stations 2 and 3 typically hypoxic at just 0.6 m below the surface and anoxic below 0.9 m, and station 4 hypoxic from depths of 1.2 to 1.6 m and anoxic from 2.1 to 4.8 m. In August, hypoxia was at its worst, with PR station 2 anoxic below 0.75 m, station 3 anoxic below 0.9 m, and station 4 anoxic below 1.2 m. The large volume of hypoxic water present throughout the year at PR station 4 affected DO at adjacent stations. PR station 1 often exhibited a sharp drop in DO near the bottom, although it is only 0.9 m deep. This may be attributed to the proximity to station 2, which was a source of dense, saline hypoxic water that was transported to station 1 during incoming tides. To the east of station 2, a short canal perpendicular to the shoreline had limited circulation with the main river flow and was a source of hypoxic water. PR exhibited a dramatic reduction in DO below the photic zone because of a sharp halocline that prevented vertical mixing of surface waters with bottom waters.

Pearson (product–moment) correlation coefficients (Table 2) indicated significant negative correlations between DO and depth, DO and salinity, and DO and temperature across all rivers, stations, and seasons. These negative correlations held true for data collected at four sampling times over the course of an entire tidal cycle on PR from before sunrise to after sunset (Tables 3 and 4), with the exception of DO and temperature, for which correlations varied from significantly negative to not significant to significantly positive. Mean values of DO and T showed significant differences over the tidal cycle driven by a mid-day increase in upper water column values because of an increase in incident solar energy and photosynthetic activity. As sampling on other dates was done mid-day, there was no significant temperature or photosynthetic difference during a given sampling period. The Tables 3 and 4 tidal cycle data indicate that our data overestimate DO levels because of mid-day sampling and do not reflect the lower values present in early morning after an evening of respiration (Breitburg 1990). Tidal cycle data also show no significant difference in salinity on PR over the tidal cycle and suggest that limited tidal flushing on PR contributes to the maintenance of the stagnant hypoxic reservoir of water. Hypoxic water present at the depths of stations 2, 3, and 4 during the low tide before sunrise was pushed northward with the influx of well-oxygenated saline water from the open bay, but hypoxic water at those stations did not get adequately flushed out with the ebb tide. An increase in DO was observed at station 4 during the flood tide, and the dense incoming bay water was diverted to the west and bottom at station 4 by the downstream southward counterflow of fresh water (Fig. 4a, flood tide). By high tide, incoming oxygenated water ameliorated hypoxia at station 3, but at station 2, incoming tidal water did not improve DO levels (Fig. 4, high tide).

There were no significant differences between west, center, and east transect values for depth, salinity, temper-

Table 2 Statistical comparisons of depth (*d*), dissolved oxygen (DO), salinity (*S*), and temperature (*T*) for CR, PR, and SR from 1997–2005 center channel data (n=745)

	R	*p* Value
DO, d	−0.5500	<0.0001
DO, S	−0.3538	<0.0001
DO, T	−0.4750	<0.0001
D, S	0.5174	<0.0001
S, T	0.0967	0.0085
D, T	0.0624	0.0886

r The Pearson correlation coefficient, *p* the probability that $|r|>0$

Table 3 PR tidal cycle: Pearson correlation comparisons of depth, dissolved oxygen, and salinity

Pearson correlation	Time	*r*	*p*
DO, d	1	−0.5751	0.0033
	2	−0.5749	0.0021
	3	−0.6396	0.0008
	4	−0.7910	<0.0001
	1 to 4	−0.6037	<0.0001
DO, S	1	−0.6629	0.0004
	2	−0.7272	<0.0001
	3	−0.6151	0.0014
	4	−0.5383	0.0009
	1 to 4	−0.5914	<0.0001
DO,T	1	−0.5587	0.0045
	2	−0.1251	0.5426
	3	0.7384	<0.0001
	4	0.6234	<0.0001
	1 to 4	0.3058	0.0012
D, S	1	0.8958	<0.0001
	2	0.8656	<0.0001
	3	0.8935	<0.0001
	4	0.7250	<0.0001
	1 to 4	0.8263	<0.0001

Tidal cycle from 6:30 A.M. to 6:45 P.M. on October 18, 1998. Sunrise was 7:15 A.M. and sunset 6:00 P.M.
Time 1 Early morning incoming tide (average=2.88 h before high tide), *time 2* late morning high tide (average=0.23 h before high tide), *time 3* early afternoon outgoing to low tide (average=3.42 h after high tide), *time 4* late afternoon low to incoming tide (average=3.96 h before high tide), *time 1 to 4* pooled data for all four times, *r* Pearson correlation coefficient, *p* the probability that $|r|>0$, *F* Fisher's ratio, *p* significant for values <0.05, *d* depth, *DO* dissolved oxygen, *S* salinity

Table 4 Least squares means comparisons and one-way ANOVA of data pooled over times 1 to 4

Least squares means	Time	LS mean ± SE	*F* Ratio	*p* Value
d	1	6.0±0.6		
	2	6.2±0.6		
	3	5.8±0.6		
	4	6.1±0.5		
	1 to 4		0.11	0.95
DO	1	5.0±0.5		
	2	6.8±0.5		
	3	8.5±0.5		
	4	7.5±0.4		
	1 to 4		8.20	<0.0001
S	1	14.3±1.4		
	2	13.4±1.3		
	3	11.6±1.4		
	4	14.8±1.1		
	1 to 4		1.14	0.34
T	1	15.8±0.1		
	2	15.9±0.1		
	3	16.4±0.1		
	4	16.3±0.1		
	1 to 4		8.50	<0.0001

Tidal cycle from 6:30 A.M. to 6:45 P.M. on October 18, 1998. Sunrise was 7:15 A.M. and sunset 6:00 P.M.

Time 1 Early morning incoming tide (average=2.88 h before high tide), *time 2* late morning high tide (average=0.23 h before high tide), *time 3* early afternoon outgoing to low tide (average=3.42 h after high tide), *time 4* late afternoon low to incoming tide (average=3.96 h before high tide), *time 1 to 4* pooled data for all four times, *p* the probability that $|r|>0$, *F* Fisher's ratio, *p* significant for values <0.05, *d* depth, *DO* dissolved oxygen, *S* salinity

ature, or dissolved oxygen on CR or SR. In the first year of the study, significant differences were observed in PR transect salinity values at high tide on October 2, 1997 at station 1 and station 4, and in DO values at station 1. The differences at stations 1 and 4 were thought to be driven by bulkheads perpendicular to the river's main flow. At station 1, a bulkhead marks the northernmost reach of tidal water, causing saline water to pool there at high tide, and oxygenated freshwater enters the study area from upstream just west of this bulkhead. At station 4, a bulkhead extends for 150 m west of the main river channel and is perpendicular to the flow of incoming saline tidal water. The more intensive data collected over the PR tidal cycle on October 18, 1998 were used to ascertain the importance of transect differences and tidal differences on the correlations and means. These data indicated no significant differences in transect values for DO and salinity. A significant difference in temperature (F=3.9, p=0.031) occurred at station 4 near sunset at time 4 and was driven by low temperature at station 4W. Data from other sampling dates throughout the study were not collected close to sunset and did not exhibit significant temperature differences over the time of sampling.

Early morning sampling was conducted on all three rivers in August 2005 to assess DO levels in summer after an evening of respiration. Results of one-way analysis of covariance of morning sampling on CR, PR, and SR in August 2005 indicated that early-morning hypoxia was significantly greater on PR and SR than on CR, with no significant difference in mean DO between PR and SR (Table 5). PR and SR were also significantly more saline than CR but did not significantly differ from each other. The same DO and salinity patterns among rivers were observed on other sampling dates. The dependence of DO on salinity by season was assessed by treating salinity as a continuous variable and temperature as a classification variable of warm and cold seasons. Two-way analysis of covariance was used to examine whether the regression of DO on salinity depended on temperature class (Table 6). The effects of season and salinity on DO were independent of each other on CR (p>0.574) and SR (p>0.389), but on

Table 5 Means and standard errors and results of one-way ANOVA for dissolved oxygen (DO) and salinity (S) for rivers sampled on the same dates

	DO (ppm) LS mean ± SE			S (ppt) LS mean ± SE		
	May 27, 2000	July 7, 2000	August 21, 2005	May 27, 2000	July 7, 2000	August 21, 2005
Means and standard errors						
CR	6.8±0.4	8.1±0.4	6.4±0.5	12.4±1.1	11.9±1.4	14.1±1.2
PR	5.4±0.3	3.7±0.3	4.0±0.5	20.5±0.9	21.3±1.0	19.3±1.0
SR			3.0±0.6			18.5±1.3
One-way ANOVA						
	p=0.0089 (F=7.16)	p<0.0001 (F=71.89)	p=0.0002 (F=9.38)	p<0.0001 (F=30.32)	p<0.0001 (F=30.93)	p=0.0044 (F=5.67)
CR/PR			0.0013			0.0046
CR/SR			<0.0001			0.0038
PR/SR			0.1913			0.6461

The August 2005 data were collected in early morning.

F Fisher's ratio, *p* significant for values <0.05

Table 6 Dependence of dissolved oxygen (DO) on salinity (S), season (Sn), and salinity by season ($S \times$Sn)

	All seasons			Warm			Cold		
River	Source	F ratio	p Value	F ratio	p Value	Slope ± SE	F ratio	p Value	Slope ± SE
CR	Season	3.40	0.0668						
	Salinity	1.07	0.3033						
	S × Sn	0.32	0.5743	17.29	<0.0001	−0.077±0.019	22.66	0.0021	−0.022±0.005
PR	Season	2.24	0.1354						
	Salinity	30.66	<0.0001						
	S × Sn	9.18	0.0026	91.56	<0.0001	−0.220±0.023	4.78	0.0323	−0.064±0.029
SR	Season	12.96	0.0004						
	Salinity	5.19	0.0243						
	S × Sn	0.75	0.3886	4.61	0.0347	−0.095±0.044	7.19	0.0100	−0.043±0.016

Two-way ANOVA for all seasons data combined shows that the effects of S×Sn are independent of each other for CR and SR but not for PR. One way regression analyses for warm and cold seasons separately indicate that the dependence of DO on salinity is negative on all three rivers, that the effect is more extreme in warm seasons than in cold, and that the effect is disproportionately greater on PR. Slope=ΔDO (ppm)/ΔS (ppt). F Fisher's ratio, p significant for values <0.05

PR, they were not (p=0.003). For PR, the analysis had to be broken down by season, and regression analyses indicated the dependence of DO on salinity for warm and cold seasons separately (Table 6). CR and SR were treated the same for consistency. For all three rivers, there was clearly a dependency of DO on salinity that is negative, but the effect was more extreme in warm seasons than in cold and was disproportionately greater on PR. At the mean water temperature of this study, the theoretical 100% saturation concentration of DO decreases by 0.05 ppm for every increase of 1 ppt in salinity. There is a lower theoretical 100% saturation DO concentration decrease at higher temperatures, and at the maximum water temperatures of this study (28°C), the 100% saturation concentration of DO decreased by 0.04 ppm for every increase of 1 ppt in salinity. However, DO levels on these rivers were typically below 100% saturation in warm seasons and were not driven by physical factors controlling gas solubility.

No elevated levels of nitrate, nitrite, ammonia, or phosphate were detected on PR, SR, or CR. Total nitrogen measurements were consistent with data obtained by USGS from 1971–1997 (Monti 2003) with median concentrations of 1.25 ppm (CR), 2.83 ppm (PR), and 1.92 ppm (SR). CR and SR had the lowest total nitrogen concentrations of the 13 SSER streams in the USGS report, and PR was just above the lower quartile. Phosphate was below detectable levels (<1 ppm) on each river. Values of pH on each river were in the normal range: from 5.0 to 7.5 for freshwater samples and from 7.5 to 8.5 for saline samples. GC/MS analysis revealed high sediment contamination at PR stations 1 and 3 to be primarily from PAHs, but PCBs and organochlorines were detected as well. Table 7 indicates the total PAHs identified in CR, PR, and SR sediments. Total PAHs in sediments on CR were 345 and 744 ppb, PR was 2,391 and 5,210 ppm, and SR was 135 and 2,682 ppb. High PAH values in PR station 1 sediments may be attributed to sealcoat applied to a parking lot adjacent to that station. Mahler et al. (2005) identified parking lot sealcoat as a source of PAHs and determined that a mean concentration of 3,500 ppm is present in runoff. High values in PR station 3 sediment may be a result of the proximity to former oil distribution terminals. Removal and remediation of terrestrial soil at these terminals is on-going, and the extent and origin of contaminants in PR sediments should be further examined.

Table 7 PAH concentrations in PR, SR, and CR sediments by GC/MS, and percent composition of extracted sediments

	PR station 1 (ppm)	PR station 3 (ppm)	SR at Franklin[a] (ppb)	SR station 01305500 (ppb)	CR station 01305000 (ppb)	CR station 01305040 (ppb)
Total PAH	2,391	5,210	135	2,682	345	744
Sediment type						
Clay	16%	23%	0%	0%	0%	0%
Silt	48%	52%	9%	26%	11%	18%
Sand	36%	24%	77%	57%	27%	68%
Gravel	2%	1%	14%	17%	62%	14%

[a] SR Franklin Ave. station is 1.5 km north of SR station 01305500.

Biological

Figure 5 presents the biological assessment of water quality according to the species richness, biotic index, and EPT index criteria for freshwater entering the study area. CR was non-impacted according to the biotic index and EPT criteria, and was slightly impacted according to the species richness criterion. The CR benthos was overwhelmingly dominated by Tricoptera, which requires high water quality, and Oligochaeta (Table 8). The level of impairment on SR was slightly impacted (EPT), moderately impacted (biotic), and severely impacted (species richness), and SR sediment was abundant in Gastropoda, Hirudinea, and Gammaridae. PR was severely impacted under all three criteria. No EPT organisms were found in any PR freshwater benthic samples, and the dominant PR benthic invertebrate groups were *Sphaerium* sp. and flatworms.

Pelagic organisms were not routinely observed in the PR water column, and they were relatively more abundant at PR stations 1 and 5. On CR and SR, there were frequent animal sightings including turtles, amphibians, muskrat, crabs, fish, and a wide variety of waterfowl and other avian species. Low light transmittance in the PR tidal section

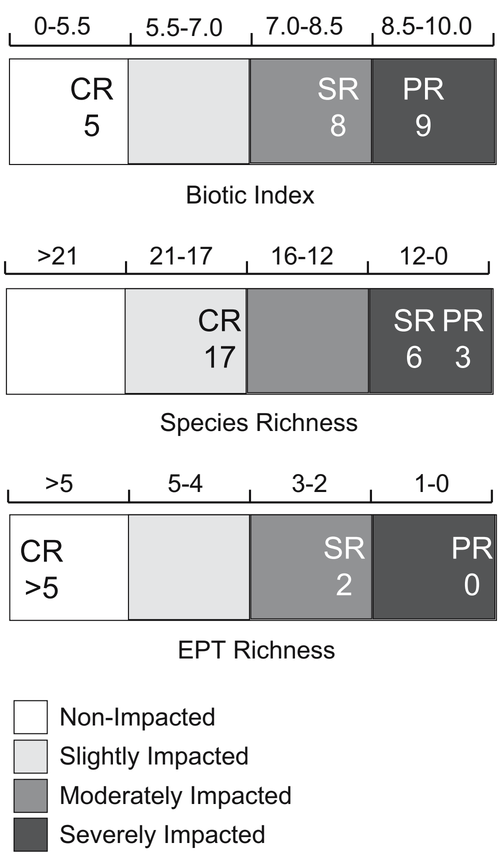

Fig. 5 Mean values for biological assessment of water quality at freshwater stations. Impairment ratings as established by New York State Department of Environmental Conservation (Bode et. al. 1990, 1991, 2000)

Table 8 Mean percent composition of benthic invertebrates and estimated sediment composition of 5 m kick sampling sites at freshwater stations

Invertebrate Group	PR	SR	CR
Ephemeroptera	0	0	0
Plecoptera	0	0	0
Tricoptera	0	2	68
Diptera	3	1	4
Odonata	0	0	0
Amphipoda[a]	4	15	4
Isopoda	2	0	0
Decapoda	0	0	0
Gastropoda	4	31	0
Pelecypoda	54	11	1
Oligochaeta	2	7	23
Hirudinae	1	32	0
Platyhelminthes[b]	30	1	0
Sediment type			
Silt	40%	25%	10%
Sand	40%	50%	20%
Gravel	20%	25%	70%

[a] Gammaridae
[b] Turbellaria

restricted plankton growth to near-surface depths; the average Secchi depth was 0.45 m on PR compared to 0.75 m on SR, and 0.9 m on CR. The water column in the PR and SR tidal sections was turbid, and sediments were fine-grained. There was considerable spatial and temporal variability in diversity and abundance of plankton on all three rivers. Plankton samples collected on the same dates at odd numbered stations on each river ranged from 3–10 times less diverse and 2–12 times less abundant on PR than SR and CR. SR and CR had comparable diversity and abundance. Chlorophyll *a* measurements varied by several orders of magnitude during the study, and during periods of phytoplankton blooms, DO levels in the upper water column exceeded 100% saturation during the mid-day samplings. This was most pronounced at PR stations 1, 4C, and 4W; SR stations 1 and 2; and CR stations 1 and 2. Chlorophyll *a* results were consistent with the study of Su et al. (2004) in which chlorophyll *a* phytoplankton concentrations were significantly greater in poorly flushed regions than in well-flushed regions.

CR tidal section sediments are coarse, dominated by organic material including wood, leaves, and aquatic plants, and support an abundant benthic community (>3,000 animals m^{-2}) of diverse crustaceans, molluscs, annelids, and burrowing fish. Non-stressed South Shore Estuary benthic environments were reported to have mean invertebrate sediment abundances of 5,402 animals m^{-2} (O'Connor 1972) and 4,445 animals m^{-2} (Cerrato 1986). However, the tidal PR and SR benthic fauna had both low diversity and abundance (no invertebrates at PR stations 2, 3, 4, and <200 animals m^{-2} at PR station 1 and SR stations 2, 3, 4). Just upstream of the PR tidal reach, 10 m north of station 1, Gammaridae routinely dominated the sediment with an average of 274 animals m^{-2}. Overall benthic fauna abundances on PR and SR were lower than those of the heavily stressed marine environments of Bowery Bay, Flushing Bay, and Raritan Bay, which had mean benthic invertebrate abundances of 127, 590, and 795 animals m^{-2}, respectively (Cerrato and Bokuniewicz 1985).

Tidal section sediments were analyzed according to the method of Gray et al. (2002) with cores taken to a depth of 0.2 m. CR sediments at all stations were classified as normal. SR sediments were normal at station 1, slightly eutrophic at station 5, severely eutrophic at station 2, and anoxic at stations 3 and 4. PR stations 1 to 4 were anoxic. PR station 5 could not be accurately assessed by this method because of frequent disturbance of the sediment by boat traffic. Throughout each PR sediment core, fine-grained particles (mean composition, 84% silt and clay) dominated. Oil was observed in samples from stations 1 to 4, and coal particles were visible in samples from stations 1 and 2.

Discussion

The USGS has monitored the freshwater sections of SSER tributaries including CR, PR, and SR since the 1940s, and Suffolk County Department of Health Services (SCDHS) has monitored open bay stations for decades. To address the lack of tidal section data for South Shore Estuary tributaries, this study focused on the tidal reaches of three centrally located representative tributaries. Our measurements at CR, PR, and SR stations 6 south of each tributary were consistent with data collected by the SCDHS marine environmental quality monitoring program, which only occasionally measured DO levels below the NY State standard of 5.0 ppm in SSER open bay waters (Pataki and Daniels 1999). Because the relatively shallow open bay is well-mixed as a result of wind and wave action, DO values less than 5.0 ppm are presumed to be uncommon but have been observed at stations in the northern reaches of the bay or near tributaries. We detected such low DO levels in the northern bay near CR, PR, and SR on summer mornings, especially after nights with low wind velocities. However, on most sampling dates, the open bay water entering the tributaries with the flood tide was well oxygenated.

The finding that DO and salinity on PR and SR did not differ significantly from each other, but that they did differ from CR, can not be explained by land use and non-point source pollution. One might predict that the PR tidal section would be expected to exhibit hypoxia on the basis of its high level of development and high percent of impervious areas and that CR would be well oxygenated because of its relatively pristine surroundings. However, the level of development on the tidal section of SR was approximately midway between PR and CR, and there were no significant differences in nutrient levels among the three rivers. The similarity of water quality on SR and PR may be attributed to physical processes rather than land use, point source, or non-point source pollution. The significantly higher levels (three orders of magnitude) of PAH pollutants in PR sediment compared to SR sediment did not result in a significant difference in water column quality between PR and SR. Neither did the point source outflow from the PR sewage treatments plants result in a significant difference in water quality on PR versus SR. The determining factor that PR and SR have in common is physical. The three physical transport processes responsible for water mass exchange are freshwater influx, density-induced circulation, and tidal flushing. Estuarine tributaries with low freshwater influxes, such as CR, PR, and SR, have their residence times dominated by tidal exchange processes (Pilson 1985). It has been shown that even when freshwater flow is high in estuarine tributaries, the influx does not result in significant flushing (Slinger et al. 1994). The second physical process, density-induced circulation, is limited during warm months because salinity-induced density stratification prevents vertical circulation. However, that stratification is overcome by temperature-induced density differences in the cold months when vertical circulation is favored. Therefore, in the warm months, when hypoxia is most extreme, tidal flushing should be the dominant of the three processes for water mass exchange.

However, as PR and SR had limited tidal flushing because of their narrow and shallow inlets, stagnant saline water was trapped below the upper layer of freshwater. Hence, a typical tidal cycle on PR and SR resulted in little water exchange. For example, during the incoming tide on PR, dense bay water entering at shallow station 5 sank at the lower depths of station 4E and was diverted westward toward deeper stations 4C and 4W. This set up an underlying clockwise pattern of circulation in the basin at station 4 that was reinforced in the upper layer by the tangent southward flow of freshwater in the main channel. The resulting vortex caused the saline water in the river to cycle rather than exit with the ebb tide, and it enabled the oxygenated upper freshwater layer to rapidly flow out to the bay. This pattern was observed and quantified in model harbors with geometry corresponding to PR station 4 and a narrow inlet like PR station 5, and the inlet width was found to have a significant effect on the characteristics of currents and flushing (Yin et al. 2004). The trapping of saline water in rivers with narrow shallow inlets is supported by our data indicating that PR and SR were on average 8 ppt more saline than CR, and they had a much sharper halocline than CR. Stratification such as this is known to foster hypoxia (Turner et al. 1987; Stanley and Nixon 1992).

The tidal flushing time on PR was estimated by the tidal prism method to be approximately 5 days on average (Jones 1999). However, it is expected to be much longer in the deep basin at station 4W and 4C, and in sidecuts such as at station 2E. The long residence times of water in these areas coupled with inefficient mixing with oxygenated surface water caused persistent hypoxic conditions on PR. In warm months, we often observed supersaturation of DO in the upper photic zone on PR and SR, and hypoxia or anoxia immediately below the photic zone. This phenomenon is attributed to algal photosynthesis and respiration by benthic and planktonic organisms (Kemp et al. 1992). Our nutrient data suggest that anthropogenic nutrient input was not a primary factor driving supersaturation or hypoxia on PR and SR but rather that stratification and tidal restrictions played the major role. The effects of tidal flushing on primary productivity, respiration, and diel DO values were examined by Beck and Bruland (2000) and Iriarte et al. (2003). They observed the greatest hyperventilation (supersaturation of DO during the day and anoxia at night) at times and locations where tidal flushing was low and sunlight was high. Hyperventilation was favored at higher water temperature, which increased both photosynthesis and respiration rates. Our data show that the mean temperature of PR and SR in the warm seasons was almost 2°C warmer than CR and that PR and SR had more hyperventilating episodes. Supersaturation on CR was only observed at station 1, which was least affected by tidal flushing. Our data are also consistent with other studies that indicate the importance of tidal flushing on DO levels (Breitburg 1990; Zajac and Whitlatch 2001; Su et al. 2004; Newton and Mudge 2005).

Many SSER tributaries have narrow inlets, have been dredged and channelized, and are likely to be hypoxic in warm months. However, DO data on SSER tributaries are sparse. A study of two SSER tributaries that examined the impacts of land use on environmental quality found bottom levels of DO below 2 ppm during warm months, but sampling was not conducted during cold months to determine if hypoxic conditions persisted (Koppelman and Davies 1990). By mid-October of each year, we observed a fall turnover (because of vertical mixing of the water column as a result of density variations) on CR and SR that resulted in well-oxygenated bottom DO that continued throughout the cold months. However, our data indicate a limited fall turnover on PR and that bottom-water hypoxia persisted in cold months on PR. One would expect hypoxia to be ameliorated during the winter on PR as well, but the higher salinity stratification on PR limited vertical mixing. Furthermore, the exchange of water between PR and the bay was restricted because of the narrow inlet. Hence, the deep and wide basin of PR station 4 remained a highly saline hypoxic lens throughout the winter. Buzelli et al. (2002) quantified the relationship between temperature, salinity differences, and hypoxia, and reported that estuarine waters with differences in surface and bottom salinity (ΔS) greater than 4 ppt experienced bottom-water hypoxia even in cold waters at 0–10°C. The ΔS effect is dramatic: At ΔS values of 0–2 and 2–4 ppt in waters between 0–10°C, bottom DO was 10 and 9 ppm, respectively. At ΔS values >4 ppt, however, bottom DO dropped to 3 ppm. At ΔS >5 ppt, about 60% of the bottom water samples had DO values <2 ppm. Because ΔS on PR was typically between 5–20 ppt at stations 2, 3, and 4, bottom water at those stations resisted vertical mixing and was prone to hypoxia. Buzelli et al. (2002) also found that the ΔS effect became much more pronounced at higher temperatures, and this is consistent with our data on PR and SR in the warmest months that indicated severe hypoxia below the photic zone and the halocline. CR exhibited higher DO year round and had lower overall ΔS values than PR and SR. The ΔS values on CR fluctuated greatly over the tidal cycle and indicated efficient tidal flushing, whereas PR and SR showed little variation in ΔS values over the tidal cycle or seasonally.

Ample dissolved oxygen is a major contributor to good water quality. The biological assessments of water quality performed by NYSDEC are based upon the principle that aquatic benthic species differ in their ability to tolerate variations in water quality (Llanso 1991, 1992; Llanso and Diaz 1994; Keister et al. 2000). NYSDEC assessments of the freshwater sections of nine South Shore Estuary streams indicate that CR is one of two streams minimally impacted, SR is one of three streams slightly impacted, and PR is one of four streams moderately impacted (Abele et al. 1994). Although low DO often affects primarily bottom waters and benthic organisms (Llanso 1992; Howell and Simpson 1994), the amount of suitable habitat in the water column is reduced when hypoxic bottom water mixes with upper layers and causes other organisms to suffer impediments to migration for spawning or feeding (Pihl et al. 1991; Roman et al. 1993; Stalder and Marcus 1997; Marcus 2001). When upper water layer temperatures are high in the warm season, fish may have little to no habitat within their optimal temperature, salinity, and DO range, and some species have been shown to become more sensitive to low DO at higher temperatures (Breitburg et al. 2001, 2003; Breitburg 2002). The DO threshold for finfish in northeastern US coastal waters is generally accepted to be 3 ppm (Stephan et al. 1985; Chapman 1986; Howell and Simpson 1994). However, some organisms experience effects on growth between 4.5–6 ppm, effects on other metabolism between 2–4 ppm, and mortality below 2 ppm (Gray et al. 2002). In the warmest months, hypoxia was most extreme in early mornings on SR, which has the least tidal flushing and the lowest freshwater influx. At these times, we typically recorded DO levels of 4 ppm or less near the surface on SR, levels less than 3 ppm at 0.6 m, and less than 1 ppm at 1 m and below. Hypoxia may be the cause of fish kills reported in the local media and observed by local residents on SR and similar SSER rivers in recent summers. On shallow and inadequately flushed rivers, such as SR, low DO may be exacerbated by high summer temperatures that further stress organisms (Niklitschek and Secor 2005). On PR, DO readings less than 2 ppm were typical below 1 m in the warmest months. However, the top 0.6-m layer on PR is significantly less saline than SR because of the greater influx of oxygenated freshwater on PR. Therefore, fish can escape hypoxia in the upper water layer of PR, which did not exhibit low DO in the warmest months.

To restore the abundance and diversity of aquatic organisms on PR and SR, it is recommended that dissolved oxygen levels be raised by interventions to improve vertical mixing and tidal flushing. For example, decreasing the depth of the PR station 4 basin and increasing the width and depth of the PR and SR river mouth inlets would improve DO water quality and the biota of the rivers and bay. Abrupt changes in depth and cut-ins to the shoreline of estuarine tributaries should be avoided to prevent the formation of hypoxic pockets. Dredging that creates sharp depth changes has been shown to prevent water column mixing, induce stratification, and trap saline water that becomes deoxygenated in the lower portion of the basin (Perillo et al. 2005). Water closest to the abrupt depth change was determined to have little to no velocity, but the reduced water transport of the lower water column was evident upstream as well. A restoration effort that involved dredging the lower reach of a tributary to remove a sill and increase tidal exchange, caused substantial ecological improvement (Zajac and Whitlatch 2001, and references therein). In that 3.5-year study, it was determined that increased tidal flushing resulted in a significant increase in species richness of macrobenthic communities and a decrease in polychaete pollution indicator species. Should future management actions improve the water mass exchange in PR or SR, our study provides a baseline to which to compare the chemical and biological and changes that occur.

National and regional data are insufficient to determine the extent of low DO areas, contamination in bottom sediments, and condition of bottom-dwelling animals in tributaries (Heinz Center Report 2002). Although hypoxia has been documented for decades in other Long Island estuaries (Parker and O'Reilly 1991; Schimmel et al. 1999), there has not been a systematic collection of such data in tidal sections of South Shore Estuary tributaries. The data presented in this paper over the period from 1997–2005 show that hypoxia is a significant problem on tributaries with limited vertical mixing and restricted tidal flow, and suggest that further examination of such tributaries is warranted. Although restoration to pristine conditions is not often practical in urban and suburban areas, management actions should include remedying the negative impacts of channelization, inlet narrowing, and bulkhead construction by improving tidal flushing where DO is low. On the SSER, the measure of water quality enhancement and restoration of ecosystem functions in response to management actions on SSER tributaries may be compared to the CR as a baseline for the expected natural state.

Acknowledgment The Dorr Foundation of New York, NY, awarded seed funding in 1997 to initiate the Chemistry in Action Research Program at Dowling College and to undertake this study. Students Jennifer Burke, Dan Dillon, Gary Falta, Rich Lemke III, Michael Melia, Andrea Mercier, Steve Strawgate, Tony Tierno, and E. Christopher Williams were funded by the Dorr Foundation. Beth Nagle was funded by Dowling College as a Graduate Assistant, and Dowling College provided release time for faculty research. The Idle Hour Flyfishers Association funded research fellowships for students Shawn Fisher, Richard Lemke III, Mia Jurjevic, Audra Selvaggio, and Nicole Stella. Nicole Stella and E. Christopher Williams were also supported on Robert Noyce Scholarships from the National Science Foundation under award no. 03-35799. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the authors and do not necessarily reflect the views of the National Science Foundation. John Egan (Emory University Law School) and Michael Nolan (University of Rochester) devoted substantial time to field and laboratory studies and to presentation of results to local governmental entities. Other students who contributed to field and laboratory studies were Rhonda Anjelly, Natasha Beria, William Capurso, Jennifer Cinquemani, Brihas Lara, Sean Madigan, Emily Montgomery, Anthony Monzon, Diana Muniz, Natalie Pedisich, Thomas Pettinato, and Sarah Porter. We appreciate the assistance of Bob Winowitch and Jack Monti of the United States Geological Survey with the taking of stream flow measurements and for providing electronic USGS map files for modification. We thank Jeff Kassner and Thomas Carrano of the Town of Brookhaven for the research vessel and staff to conduct field studies on Swan River.

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