Limnol. Oceanogr., 51(1, part 2), 2006, 690–701

Factors contributing to hypoxia in rivers, lakes, and streams

Michael A. Mallin,[1] *Virginia L. Johnson, Scott H. Ensign,*[2] *and Tara A. MacPherson*[3]
Center for Marine Science, University of North Carolina Wilmington, Wilmington, North Carolina 28409

Abstract

We investigated physical, chemical, and biological variables contributing to biochemical oxygen demand (BOD) in 17 North Carolina lotic and lentic water bodies affected by mild to severe hypoxia. Phytoplankton production created the dominant reservoir of labile carbon driving BOD, and subsequent hypoxia, in a Piedmont river subject to algal blooms, three urban streams, a set of anthropogenically affected tidal creeks, and two urban lakes. Autotrophic phytoplankton production contributed to the BOD load in some rural streams. Autochthonous heterotrophic processes, stimulated primarily by phosphorus and secondarily by nitrogen loading, were the major influences on BOD in two large black water rivers and some rural black water streams. Inputs of biochemical oxygen-demanding materials from storm water runoff contribute to BOD in some urban and rural streams and black water rivers. We suggest that reductions of hypoxia can be better achieved by a system-specific approach based on an array of factors that potentially influence BOD, including both autochthonous and allochthonous variables. In some circumstances targeting the nutrient(s) stimulating phytoplankton blooms will suffice to reduce hypoxia, but in other situations targeting nutrient(s) limiting bacterial production will be necessary. Reduction of non–point source inputs of biochemical oxygen-demanding materials derived from urbanization or other land disrupting activities will be critical in some cases.

Hypoxia is a commonly recognized symptom of eutrophic waters (Bricker et al. 1999; Burkholder 2001). While acute hypoxia or anoxia can be caused by organic waste loading from allochthonous sources (Van Dolah and Anderson 1991; Mallin et al. 2002), chronic hypoxia is often caused by autochthonous processes within a water body (NRC 2000; Burkholder 2001; Wetzel 2001). Chronic hypoxia has been well documented in large ecosystems such as Chesapeake Bay (Officer et al. 1984; Boesch et al. 2001) and the Gulf of Mexico (Rabalais et al. 2001). Consequences of anoxia or hypoxia include fish and invertebrate kills, loss of habitat for resident organisms, enhanced susceptibility to disease, and changes in predator–prey interactions among affected species (Diaz and Rosenberg 1995; Breitburg et al. 1997; Lenihan and Peterson 1998).

Biochemical oxygen demand (BOD) is a measure of the dissolved oxygen (DO) required by the microbial community in decomposing the organic matter present in a water sample by aerobic biochemical action (Clark et al. 1977; Boyd 2000). This measurement is commonly used by environmental engineers to determine the carrying capacity of streams for organic wastes in the design of wastewater treatment facilities. BOD is also used to measure the strength of pollutant loads such as sewage effluents or spills (Clark et al. 1977; Boyd 2000) or spills from industrialized animal production operations (Mallin 2000). Elevated BOD loads to streams, rivers, lakes, and other enclosed water bodies can cause severe hypoxia problems (NRC 1993).

In terms of eutrophication, BOD loads caused by the decomposition of algal blooms and other excessive organic matter are a widespread and increasing problem in large open coastal waters, as well as more enclosed systems such as lakes and streams (NRC 1993, 2000; Boesch et al. 2001; Burkholder 2001). In rural areas non–point source runoff from concentrated animal feeding operations (CAFOs), as well as traditional agriculture, contribute nutrients to streams (Mallin 2000; Arbuckle and Downing 2001). Non–point source runoff from urban and suburban lawns, gardens, commercial landscaping, and golf courses contribute anthropogenic nutrients to streams and lakes (Paul and Meyer 2001), as does runoff of manure from pets and urban wildlife. Point and nonpoint nutrient inputs stimulate phytoplankton growth and can significantly contribute to a given system's chronic BOD load (NRC 2000). However, recent research has demonstrated that BOD can also be increased in some water bodies by direct stimulation of heterotrophic microbial flora by anthropogenic nutrient loading (Mallin et al. 2001, 2004*a*). Thus, while state and federal regulations concerning the discharge of oxygen consuming wastes (as required by U.S. Environmental Protection Agency under the National Pollution Discharge Elimination System) limit the primary contribution of BOD to a waterbody, inorganic nutrients can stimulate secondary sources of BOD that are often not regulated. This disconnect between water quality management objectives and the regulated discharge of waste products can result from outdated water body classification systems. As an example, the North Carolina Division of Water Quality requires a water body be specially designated as ''nutrient

---

[1] Corresponding author (mallinm@uncw.edu).

[2] Present address: Institute of Marine Sciences, University of North Carolina at Chapel Hill, Morehead City, North Carolina 28557.

[3] Present address: Division of Water Quality, Groundwater Section, North Carolina Department of Environment and Natural Re sources, Wilmington, North Carolina 28405.

Acknowledgments

Funding was provided by the Lower Cape Fear River Program, the city of Wilmington, New Hanover County, the town of Carolina Beach, the North Carolina General Assembly, and the North Carolina Academy of Sciences. For field and laboratory assistance we thank Alex Croft, Jen Hardin, Matthew McIver, Douglas Parsons, Amy Roberts, Ellen Wambach, and Heather Wells. Two anonymous reviewers provided helpful suggestions that improved the manuscript.

Table 1. Sampling site characteristics.

Site	Sampling period	Total samples collected	Mean depth (m)	Tidal range (m)	Major land use
Cape Fear River	Jul 1998–Jun 2003	58	10.0	0.8	mixed agriculture/urban
Black River	Jul 1998–Jun 2003	58	4.0	0.6	CAFO/agriculture
Northeast Cape Fear River	Jul 1998–Jun 2003	58	10.0	0.8	CAFO/agriculture
Browns Creek	May 2000–Jun 2003	38	0.7	0	agriculture
Hammond Creek	May 2000–Jun 2003	38	1.3	0	agriculture
Six Runs Creek	May 2000–Jun 2003	38	1.6	0	CAFO/agriculture
Little Coharie Creek	May 2000–Jun 2003	38	1.1	0	CAFO/agriculture
Great Coharie Creek	May 2000–Jun 2003	38	0.9	0	CAFO/agriculture
Colly Creek	May 2000–Jun 2003	38	0.8	0	undeveloped forest
Smith Creek	Feb 2001–Sep 2003	28	1.2	0.6	suburban/urban/forest
Motts Creek	Feb 2001–Sep 2003	28	1.0	1.4	urban/suburban/forest
Barnards Creek	Feb 2001–Sep 2003	28	1.4	1.4	urban/suburban/forest
Tidal Creeks	June 2001–Aug 2003	70 (5 stations)	0.5–1.5	1.3	suburban/urban
Greenfield Lake	Oct 2000–Jul 2001	12	1.5	NA	urban/suburban
Carolina Beach Lake	Apr 2003–Jun 2004	16 (4 stations)	1.5	NA	urban/suburban

CAFO, watershed dominated by concentrated animal feeding operations; agriculture, watershed dominated by traditional row crop agriculture; urban, watershed dominated by mixed commercial and moderate- to high-density residential land use; suburban, watershed dominated by moderate- to low-density residential with low commercial development; NA, not applicable.

sensitive waters'' to incur nutrient discharge limitations. While ammonia discharge limits are required for toxicity and BOD considerations, total nitrogen and phosphorus limits may not be imposed on point sources in many watersheds where BOD can be caused by nutrient loading.

In systems receiving nutrient and organic loading from varied sources, such as rivers and streams draining mixed-use watersheds, hypoxia can be a problem with no clearly defined cause. DO measurements provide an instantaneous assessment of present environmental conditions, but these conditions may have been created by physical, chemical, or biological factors occurring elsewhere (upstream) or at an earlier time period. Whereas the BOD analysis has the disadvantage of delayed results (by definition, 5-d, 20-d, 90-d, etc.), an advantage of using BOD in limnological and estuarine assessments is that it is a standard method that is easily performed, repeatable, and widely recognized geographically and across disciplines. When measured in conjunction with various physical, chemical, and biological parameters it provides insights on the conditions that have led to hypoxia in a given system. In this research we used BOD measurements to assist in assessing conditions contributing to eutrophication-induced hypoxia in 17 streams, rivers, and urban lakes in southeastern North Carolina. The data set was used to assess the variables contributing to BOD and subsequent hypoxia in this diverse range of aquatic ecosystems.

Sampling sites and characteristics—Data were collected from three rivers, six rural streams, three urban streams, three estuarine tidal creeks, and two urban lakes (Table 1). All sites were located in southeastern North Carolina.

The Cape Fear River is the largest river entirely within the state of North Carolina. It is a sixth-order stream that begins in the Piedmont near Greensboro, North Carolina, has a drainage basin encompassing 23,310 km^2, and empties into the Atlantic Ocean at Cape Fear (33°53.281N, 78°00.473W). It can be very turbid (Table 2) and carries a significant anthropogenic nutrient load from the Piedmont and upper Coastal Plain (Table 3; Dame et al. 2000). The lower river and estuary are considered to be moderately eutrophic (Bricker et al. 1999) and classified as impaired by low DO concentrations (NCDEHNR 2000). At the sampling location, 40 km upstream of the city of Wilmington and 5 km downstream from a lock and dam complex (Fig. 1), it was approximately 10 m deep. The river's flow is impeded by three lock and dam complexes upstream of the sampling location, which allow for phytoplankton bloom formation in the limnetic environments above the dams. Samples used in this analysis were collected monthly from July 1998 to June 2003 ($n = 58$).

The Black and the Northeast Cape Fear Rivers are both fifth-order streams that arise within the North Carolina Coastal Plain and join the mainstem Cape Fear River upstream of the city of Wilmington (Fig. 1). These black water rivers are well mixed and undammed, and they do not host phytoplankton blooms (Table 2), but they drain watersheds containing traditional agriculture and large numbers of CAFOs (Table 1), both of which contribute anthropogenic nutrients to the river systems (Mallin 2000; Mallin et al. 2002). Both rivers suffer from chronic summer hypoxia throughout the water column (Table 2; Mallin et al. 2004*a*) and have experienced acute hypoxia from waste loading incidents (Mallin 2000; Mallin et al. 2002). The Black River station was 4 m deep on average and located 45 km upstream of Wilmington, and the Northeast Cape Fear River station was 10 m deep on average and located 20 km upstream of Wilmington (Fig. 1). The sample period was identical to that of the Cape Fear River ($n = 58$ each).

The six rural creeks (Fig. 1) are second- to fourth-order tributaries of either the Cape Fear River (Hammond and Browns Creeks) or the Black River (Great Coharie, Little Coharie, Six Runs, and Colly Creeks). These streams pri-

Table 2. Physical and biological water quality data collected for the sampled sites during this study, presented as mean ± standard deviation/range.

Site	DO (mg L^{-1})	FC (CFU 100 ml^{-1})	Turb (NTU)	TSS (mg L^{-1})	Chl *a* (μg L^{-1})	BOD5 (mg L^{-1})	BOD20 (mg L^{-1})
Cape Fear River	8.6±2.3	48±177	25±30	13.9±27.6	4.0±3.8	1.2±0.5	3.2±1.2
	2.8–13.4	1–1370	3–167	1–191	0.4–15.2	0.6–2.4	1.8–8.6
Black River	6.9±2.5	48±33	4±2	1.7±1.1	0.7±0.7	0.9±0.4	2.5±0.7
	1.2–12.3	13–200	0–8	0–5	0.1–3.8	0.4–2.8	1.3–4.9
Northeast Cape Fear River	6.0±2.5	54±174	5±3	3.0±1.9	1.2±2.3	1.0±0.5	2.8±1.1
	0–11.7	6–1355	1–21	1–10	0.1–17.1	0.4–4.0	1.2–9.0
Browns Creek	9.0±1.9	157±383	8±13	4.5±8.2	1.8±2.0	1.0±0.4	2.7±1.0
	5.0–12.2	11–2310	2–85	1–51	0.4–11.7	0.3–2.4	1.4–6.2
Hammond Creek	7.1±2.5	242±701	10±19	6.2±17.3	3.0±4.3	1.4±0.7	3.6±1.6
	1.4–11.3	7–4360	4–125	0–109	0.2–17.7	0.4–3.4	1.2–9.1
Six Runs Creek	8.0±2.4	117±283	7±5	4.1±4.6	1.0±1.7	1.1±0.5	2.9±1.0
	2.8–12.2	11–1750	2–28	0–23	0.1–10.5	0.4–2.4	1.6–6.2
Little Coharie Creek	8.0±2.3	53±41	5±3	3.2±2.8	0.9±0.9	1.0±0.5	3.0±1.3
	4.3–12.1	8–195	1–15	0–12	0.2–5.2	0.4–2.3	1.2-6.8
Great Coharie Creek	6.3±2.7	72±187	5±5	3.9±4.2	1.9±3.1	1.0±0.7	3.1±1.7
	0.8–10.9	9–1170	1–20	0–18	0.3–19.7	0.3–3.6	1.2–9.1
Colly Creek	7.2±1.9	74±94	3±3	2.3±2.7	2.0±3.4	1.1±0.8	2.9±1.3
	4.3–11.2	7–449	0–16	0–11	0.1–14.8	0.3–3.5	1.2–7.5
Smith Creek	6.9±2.2	251±404	22±11	21.0±12.6	9.4±8.8	2.0±1.2	6.9±4.1
	4.0–11.4	18–1840	8–54	2–49	0.3–32.4	0.2–5.5	2.4–20.4
Motts Creek	6.2±2.1	785±1260	28±33	22.3±25.6	14.7±25.0	2.3±1.7	7.0±4.1
	3.3–11.4	24–6000	8–155	3–119	1.2–115.3	0.2–7.9	2.2–23.2
Barnards Creek	6.8±2.4	217±369	33±35	36.7±45.5	7.3±6.8	1.9±1.3	7.2±4.3
	2.9±12.1	13–2000	6–203	2–191	0.9–22.3	0.4–6.1	2.5–21.6
Tidal Creeks	4.7±2.1	NA	NA	21.3±15.3	6.8±8.4	2.4±1.5	NA
	0.7–10.3			1.1–80.5	0.2–51.3	0.1–7.6	
Greenfield Lake	9.2±3.6	1119±1946	15.6±20.3	7.2±4.8	43.3±52.7	4.0±2.6	NA
	5.3–18.9	14–6000	1–53	1–15	0.3–169.0	1.3–9.0	
Carolina Beach Lake	8.1±3.3	46±30	4±2	NA	10.5±5.4	2.9–0.9	6.5±1.7
	3.3–13.0	6–98	1–8		6.3–22.5	1.5–4.2	4.2–8.9

DO, dissolved oxygen; FC, fecal coliform bacteria; Turb, turbidity; TSS, total suspended solids; Chl *a*, chlorophyll *a*; BOD5, 5-day BOD; BOD20, 20-day BOD; NA, no data available.

marily drain watersheds containing traditional agriculture and/or CAFOs, except for Colly Creek, which mainly drains undeveloped wetlands (Table 1). Great Coharie, Six Runs, and Hammond Creeks in particular have been affected by severe hypoxia (Table 2). Depth at the sampling sites was highly variable depending on season, stream, and stream discharge, ranging from as shallow as 0.2 m during a severe drought to as deep as 4.7 m during high water. Samples for this analysis were collected monthly from May 2000 through June 2003 ($n = 38$ each).

The three urban streams are tidally influenced, but are fresh to oligohaline. Smith Creek is a third-order stream that drains suburbs north of the city of Wilmington and flows into the Northeast Cape Fear River before it joins the mainstem Cape Fear River (Fig. 1). The sampling station was located in an oligohaline stretch and was approximately 2 m deep. Barnards and Motts Creeks are third- and second-order streams, respectively, that drain suburbs south of Wilmington and enter the upper Cape Fear River estuary (Fig. 1). Sampling sites on both streams were in oligohaline waters about 1.5–2 m deep. The streams are primarily black water that has been chemically influenced by considerable urban and suburban surface runoff. Moderate hypoxia periodically occurs in these streams (Table 2). Samples were collected from February 2001 through September 2003 ($n = 28$ for each creek).

The three mesohaline tidal creeks are located in New Hanover and Pender Counties and discharge into the Atlantic Intracoastal Waterway (Fig. 1). These creeks (Futch, Hewletts, and Pages Creeks) drain urbanizing watersheds (Table 1) that range from about 10% impervious surface coverage in Futch Creek to approximately 22% impervious coverage in Hewletts Creek. These creeks have a tidal range of ca. 1 m, and streamside vegetation ranges from wooded uplands in the headwaters region to salt marsh vegetation in the lower reaches. The headwaters areas of these creeks flow into oligohaline to mesohaline marsh areas with open canopies. The upper reaches of these systems receive nutrient loading from the watershed and show responses to this loading in the form of algal blooms that vary by creek according to intensity of watershed development (Table 2; Mallin et al. 2004*b*). During summer these systems are affected by periodic hypoxia, especially in bottom waters (Table 2). During the present research one site was sampled in Futch Creek and two sites were sampled in both Hewletts and Pages Creeks, in mesohaline to polyhaline waters. Station depths were variable according to tide, ranging from 0.5 to 1.5 m. Samples used in this research were collected monthly from

Table 3. Nutrient data for the systems sampled during this study, presented as mean ± standard deviation/range.

Site	Ammonium N (μg L^{-1})	Nitrate N (μg L^{-1})	Total nitrogen (μg L^{-1})	Orthophosphate P (μg L^{-1})	Total phosphorus (μg L^{-1})
Cape Fear River	86.4±40.7	708.8±249.2	1332.0±328.5	105.4±60.4	200.5±84.2
	5.0–220.0	190.0–1580.0	780.0–2330.0	30.0–261.0	90.0–400.0
Black River	59.7±36.3	159.8±157.5	795.2±198.2	27.5±15.3	70.0±32.0
	5.0–200.0	5.0–870.0	460.0–1780.0	5.0–80.0	20.0–150.0
Northeast Cape Fear River	71.8±44.3	224.2±199.6	968.8±228.1	44.1±21.3	88.8±32.8
	5.0–240.0	5.0–990.0	550.0–1590.0	5.0–108.0	40.0–190.0
Browns Creek	60.8±33.8	168.2±195.4	641.8±237.1	23.9±11.7	93.4±91.5
	10.0–200.0	5.0–1130.0	260.0–1350.0	0.1–47.0	30.0–600.0
Hammond Creek	76.4±42.3	59.6±61.3	700.3±310.2	43.7±19.9	145.0±63.1
	5.0–170.0	5.0–200.0	300.0–1790.0	10.0–72.0	40.0–290.0
Six Runs Creek	85.5±48.4	354.6±200.0	1005.0±261.0	39.9±21.1	111.8±61.1
	20.0–240.0	5.0–890.0	610.0–2090.0	1.0–100.0	30.0–310.0
Little Coharie Creek	69.3±42.6	173.6±191.4	857.1±209.3	21.8±16.5	57.6±28.6
	5.0–190.0	5.0–880.0	380.0–1500.0	0.1–70.0	10.0–120.0
Great Coharie Creek	77.4±55.0	175.1±272.1	924.5±351.4	96.4±89.1	247.9±286.9
	10.0–270.0	5.0–1030.0	490.0–2140.0	10.0–380.0	20.0–1400.0
Colly Creek	83.8±73.1	54.7±192.0	988.1±358.4	10.0±9.1	34.6±29.0
	5.0–280.0	5.0–910.0	620.0–2270.0	0.1–38.0	5.0–180.0
Smith Creek	78.6±66.2	200.0±147.9	1151.1±0.822	59.5±27.4	108.1±56.5
	5.0–300.0	5.0–510.0	190.0–4405.0	12.0–130.0	40.0–290.0
Motts Creek	70.7±59.6	115.2±99.8	913.5±496.7	48.3±48.9	241.5±823.6
	4.0–229.0	6.0–400.0	80.0–2133.0	1.0–254.0	2.0–4590.0
Barnards Creek	109.4±87.8	229.2±153.5	1138.2±578.1	81.5±67.0	138.3±98.2
	5.0–340.0	2.0–560.0	310.0–2502.0	11.0–377.0	50.0–608.0
Tidal Creeks	38.8±47.7	20.3±35.2	NA	12.1±6.4	NA
	8.3–288.4	0.1–173.8		1.9–32.1	
Greenfield Lake	75.0±160.0	10.6±8.8	632.5±626.6	39.4±33.4	148.9±97.3
	5.0–500.0	5.0–100.0	70.0–2100.0	5.0–120.0	10.0–270.0
Carolina Beach Lake	42.5±36.8	13.7±13.0	765.1±169.1	8.9±6.6	62.7±31.6
	0.1–104.3	2.2–53.8	451.2–986.7	2.1–21.1	19.9–161.4

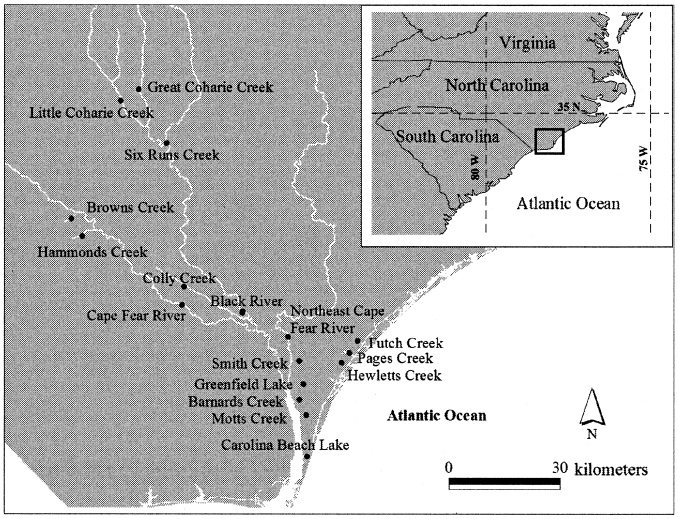

Fig. 1. Location of sampled systems in southeastern North Carolina, and location of study area in the southeastern United States.

July 2001 through August 2002 (MacPherson 2003). Because these tidal creeks behave similarly in terms of nutrient distribution and nutrient limitation (Mallin et al. 1999, 2004*b*), data for these three creeks were combined for this analysis ($n = 70$).

Greenfield Lake is a black water urban lake located in Wilmington, North Carolina (Fig. 1). It has a surface area of approximately 37 ha, it is approximately 8,530 m around the shoreline, and is about 1.5 m deep at the sampling location, located off a dock in an urban park. This lake is considered to be nutrient enriched (NCDEHNR 2000) and supports periodic phytoplankton blooms in spring, summer, and fall (Table 2). Some of the most frequent bloom-forming taxa are the cyanobacterium *Anabaena cylindrica* and the chlorophytes *Spirogyra* and *Mougeotia* spp. The free-floating macrophyte *Lemna* sp. (duckweed) is frequently observed on the surface, and below a *Lemna* bloom in August 2004 DO concentrations at the park station registered 0.4 mg L^{-1}. In situ monitoring instruments have demonstrated that DO concentrations can decrease by as much as 45% at night compared with daytime DO measurements. BOD data were collected in this lake monthly or semimonthly from October 2000 through July 2001 ($n = 12$).

Carolina Beach Lake is an oligohaline lake (mean salinity

= 1) located approximately 200 m from the Atlantic Ocean in the town of Carolina Beach, North Carolina (Fig. 1). It is approximately 1.5 m deep and drains a 243 ha urban and suburban watershed. It hosts periodic phytoplankton blooms and at times also has significant coverage (>25% of lake surface) of the submersed macrophyte, *Potamogeton pectinalis* (Sago pondweed). During midday, surface DO readings have been measured as low as 3.3 mg L^{-1} (Table 2). Sampling was conducted at four stations during April, July, and October 2003, and January 2004 ($n = 16$).

Methods

Field physical parameters (water temperature, pH, DO, turbidity, salinity, and conductivity) were measured at each site using a YSI 6920 multiparameter water quality probe (Sonde) linked to a YSI 610D display unit. YSI Model 85 and 55 DO meters were also used on occasion. The instruments were calibrated prior to each sampling trip to ensure accurate measurements. Samples for total suspended solids (TSS) were collected in 500-ml containers, placed on ice, and analyzed using Method 2540-D according to *Standard Methods for the Examination of Water and Wastewater* (APHA 1995).

For nitrate + nitrite (hereafter referred to as nitrate) and orthophosphate assessment, three replicate acid-washed 125-ml bottles were placed ca. 10 cm below the surface, rinsed, filled, capped, and stored on ice until processing. In the laboratory the triplicate samples were filtered simultaneously through 25-mm Gelman A/E glass-fiber filters (nominal pore size 1.0 μm) using a manifold with three funnels. The pooled filtrate was stored frozen until analysis. Samples for ammonium were collected in duplicate, field-preserved with phenol, stored on ice, and analyzed in the laboratory according to the methods of Parsons et al. (1984). Samples for total nitrogen (TN) and total phosphorus (TP) were collected in duplicate on site. Nitrate, TN, orthophosphate, and TP samples were analyzed using *Standard Methods* (APHA 1995).

Chlorophyll *a* concentrations were determined from the filters used for filtering samples for nitrate and orthophosphate analyses. All filters were wrapped individually in aluminum foil, placed in an airtight container with desiccant, and stored in a freezer. During the analytical process, the glass-fiber filters were separately immersed in 10 ml of a 90% acetone solution. The acetone was allowed to extract the chlorophyll from the material for 18–24 h. The extracted material was then analyzed for chlorophyll *a* concentration using a Turner AU-10 fluorometer. This method uses an optimal combination of excitation and emission bandwidths that reduces the errors inherent in the acidification technique (Welschmeyer 1994).

Fecal coliform bacteria samples were collected by lowering preautoclaved 500-ml glass containers about 10 cm below the water surface, facing into the current, if the water was in motion. Samples were kept on ice in coolers until processing at the laboratory, within 6 h of collection. The method used in this study to assess fecal coliform concentrations was the membrane filtration method (mFC), described in *Standard Methods* (APHA 1995). This method uses an elevated temperature incubation to distinguish fecal coliforms from the total coliform group. Five-day BOD (BOD5) analyses were performed according to *Standard Methods* (APHA 1995) and incubated in standard 300-ml BOD bottles in a darkened incubator at 20°C. Twenty-day BOD (BOD20) analyses were also conducted to obtain information on the less-labile component of BOD. Since the samples were from water bodies unaffected by direct sewage loading, BOD samples were not diluted or seeded.

Daily river discharge data were obtained from the U.S. Geological Survey, Raleigh, North Carolina, office for the three main river branches. The flow gauging stations are located at Lock and Dam 1 on the Cape Fear River, 5 km upstream of the Cape Fear River sampling station; near Tomahawk on the Black River, 45 km upstream of the Black River sampling site and 90 km upstream of Wilmington; and near Chinquapin on the Northeast Cape Fear River, 77 km upstream of the Northeast Cape Fear River sampling site and 97 km upstream of Wilmington. For the six rural stream stations flow was measured (in meters per second) midstream from a bridge using a Marsh-McBirney Flo-Mate Model 2000. A lead weight and fin apparatus were used to keep the flow sensor motionless in the water column and pointed into the current. A lead line with 0.5-m gradations was used to measure depth at 3-m intervals across the stream. From these measurements average depth was computed. Average depth was multiplied by stream width to obtain the cross-sectional area of the creek in square meters. Volume of flow was calculated by multiplying flow by cross-section area of the stream to obtain cubic meters per second, subsequently converted to cubic meters per day. Stream discharge data were not available for the urban streams or tidal creeks. Physical, chemical, biological, and hydrological data sets were tested for normality using Proc Univariate in SAS (Schlotzhauer and Littell 1987); where appropriate, data were normalized by log transformation. For each system or set of systems, correlation analyses were run among BOD concentrations and various physical, chemical, and biological parameters using SAS (Schlotzhauer and Littell 1987).

Results

Average BOD5 and BOD20 concentrations were slightly higher in the Cape Fear River than in the Black and the Northeast Cape Fear Rivers (Table 2). Median BOD5 over the 5-yr study was 1.1 mg L^{-1} in the Cape Fear and 0.8 and 0.9 mg L^{-1} in the Black and Northeast Cape Fear Rivers, respectively. The Cape Fear had considerably higher average chlorophyll *a* and turbidity concentrations than the two black water rivers (Table 2). In the Cape Fear River there was a highly significant correlation between BOD5 and BOD20 and chlorophyll *a*, and a weaker correlation between BOD20 and TSS (Table 4). In the two black water rivers BOD was not correlated with chlorophyll *a*. Positive correlations occurred in the Northeast Cape Fear River between BOD5 and BOD20 and fecal coliform bacterial abundance, total phosphorus, and orthophosphate, and between BOD20 and turbidity and TN (Table 4). In the Black River there was a

Table 4. Results of correlation analyses for factors affecting BOD5 and BOD20 in the Cape Fear, Northeast Cape Fear, and Black Rivers, presented as Pearson correlation coefficient (*r*)/probability (*p*). Nonsignificant ($p>0.05$) relationships not shown.

River	LChl *a*	LFC	LTurb	LTSS	LTP	LOP	LTN	LFlow
Cape Fear River								
LBOD5	0.532 0.0001							
LBOD20	0.338 0.0085			0.276 0.0329				
Northeast Cape Fear River								
LBOD5		0.482 0.0001			0.342 0.0075	0.308 0.0176		0.369 0.0037
LBOD20		0.375 0.0034	0.293 0.0258		0.339 0.0087	0.283 0.0315	0.299 0.0214	0.550 0.0001
Black River								
LBOD5						0.293 0.0248		
LBOD20					0.333 0.0100	0.376 0.0037	0.450 0.0003	0.311 0.0164

LBOD5, log BOD5; LBOD20, log BOD20; LChl *a*, log chlorophyll *a*; LFC, log fecal coliform bacteria; LTurb, log turbidity; LTSS, log total suspended solids; LTP, log total phosphorus; LOP, log orthophosphate; LTN, log total nitrogen; LFlow, log river discharge on day sampled.

positive correlation between BOD5 and orthophosphate, and positive correlations between BOD20 and orthophosphate, total phosphorus and total nitrogen concentrations (Table 4). For the Cape Fear and Northeast Cape Fear Rivers there were significant ($p < 0.05$) correlations between fecal coliform bacterial abundance and turbidity as well. Positive correlations occurred between BOD5 and river discharge in the Northeast Cape Fear River and between BOD20 and river discharge in both the Northeast Cape Fear and the Black Rivers (Table 4). Water temperature was not correlated with BOD in any of the rivers.

The six rural streams have differing watershed land use patterns, but had similar average BOD5 and BOD20 concentrations over the study period. Average BOD5 ranged from 1.0 to 1.4 mg L^{-1}, and average BOD20 from 2.7 to 3.6 mg L^{-1} (Table 2). Phytoplankton biomass was significantly correlated with BOD in Browns, Hammond, Great Coharie, and Colly Creeks (Table 5). Turbidity and TSS were correlated with BOD (particularly BOD20) in Browns, Hammond, Six Runs, Great Coharie, and Colly Creeks (Table 5). Fecal coliform bacterial abundance was positively correlated ($p < 0.05$) with BOD20 in Browns, Great Coharie, and Colly Creeks, and with BOD5 in Great Coharie and Colly Creeks. TP, TN, and ammonium were correlated with BOD in most of the creeks as well (Table 5), and orthophosphate was in Six Runs, Great Coharie, and Colly Creeks ($p < 0.05$), but nitrate was only correlated with BOD in Colly Creek ($r = 0.690$, $p = 0.0001$ and $r = 0.604$, $p = 0.0001$ for BOD5 and BOD20, respectively). There was a significant positive correlation between BOD5 and stream discharge in Great Coharie Creek, and between BOD20 and stream discharge in Browns, Hammond, Six Runs, and Great Coharie Creeks (Table 5). Colly Creek showed a negative correlation between BOD5 and BOD20 and stream discharge (Table 5). Water temperature was positively correlated with BOD (mainly BOD20) in most of the creeks (Table 5).

All three of the urban streams showed significant positive correlations between chlorophyll *a* and BOD5, but only in Smith Creek for BOD20 (Table 6). In Barnards Creek BOD5 was also correlated with TSS (Table 6; Fig. 2), and chlorophyll *a* was correlated with TSS ($r = 0.633$, $p = 0.0002$). In Barnards Creek TSS often cycled synchronously with chlorophyll *a*, but on occasion (e.g., August 2001 and April 2003) TSS was apparently mainly allochthonous in origin (Fig. 2). There were significant positive correlations between BOD20 and water temperature in all three creeks (Table 6). Average BOD5 and BOD20 in these urban streams was approximately double that of the rural streams (Table 2).

There were strong and highly significant correlations between BOD5 and chlorophyll *a*, and BOD5 and TSS in the mesohaline tidal creeks (Table 7). There was a significant correlation ($r = 0.576$, $p = 0.0001$) between chlorophyll *a* and TSS as well, indicating that some of the TSS was phytoplankton derived. BOD5 in the tidal creeks was on average more than double that of most of the rural streams (Table 2).

In Greenfield Lake strong positive correlations were found between BOD5 and chlorophyll *a* (Table 8; Fig. 3), turbidity, and TSS (Table 8). Chlorophyll *a* was correlated with turbidity ($r = 0.833$, $p = 0.0015$) and TSS ($r = 0.811$, $p = 0.008$). This indicates that the majority of in-lake turbidity and suspended matter was algal in origin.

In Carolina Beach Lake both BOD5 and BOD20 showed strong correlations with chlorophyll *a*. (Table 8). Both parameters were also positively correlated with turbidity, TN, and TP (Table 8). In contrast to the positive correlations with nutrients seen in several of the other systems, BOD was negatively correlated with ammonia and water temperature in Carolina Beach Lake (Table 8).

Discussion

In the Piedmont-derived Cape Fear River, BOD was primarily driven by the decomposition of phytoplankton bio-

Table 5. Results of correlation analyses for factors affecting BOD5 and BOD20 in six rural streams in the Cape Fear and Black River basins, presented as Pearson correlation coefficient (r)/probability (p). Nonsignificant ($p > 0.05$) relationships not shown.

Stream	LChl *a*	LTurb	LTSS	LTP	LTN	LAmm	LFlow	Temp
Browns Creek								
LBOD5	0.431 0.0069	0.349 0.0319	0.335 0.0376					
LBOD20		0.658 0.0096	0.592 0.0001	0.403 0.0121			0.485 0.0020	0.332 0.0417
Hammond Creek								
LBOD5	0.451 0.0044	0.392 0.0150	0.629 0.0001	0.417 0.0092	0.472 0.0028			
LBOD20	0.372 0.0213	0.543 0.0004	0.687 0.0001	0.489 0.0014	0.379 0.0189		0.348 0.0350	
Six Runs Creek								
LBOD5			0.371 0.0234			0.343 0.0377		
LBOD20		0.574 0.0002	0.632 0.0001	0.492 0.0020	0.541 0.0005	0.616 0.0001	0.423 0.0092	0.387 0.0180
Great Coharie Creek								
LBOD5	0.512 0.0010	0.764 0.0001	0.519 0.0008	0.656 0.0001	0.440 0.0057	0.600 0.0001	0.987 0.0001	0.670 0.0001
LBOD20	0.528 0.0008	0.829 0.0001	0.628 0.0001	0.779 0.0001	0.408 0.0001	0.490 0.0018	0.881 0.0001	0.778 0.0001
Little Coharie Creek								
LBOD5					0.520 0.0008			
LBOD20					0.592 0.0001			0.414 0.0097
Colly Creek								
LBOD5	0.640 0.0001				0.535 0.0005	0.526 0.0007	−0.488 0.0085	0.499 0.0014
LBOD20	0.652 0.0001		0.353 0.0296	0.391 0.0151	0.499 0.0014	0.518 0.0009	−0.447 0.0014	0.615 0.0001

LBOD5, log BOD5; LBOD20, log BOD20; LChl *a*, log chlorophyll *a*; LTurb, log turbidity; LTSS, log total suspended solids; LTP, log total phosphorus; LTN, log total nitrogen; LAmm, log ammonium; LFlow, log stream discharge on day sampled; Temp, water temperature.

mass (as reflected by chlorophyll *a* concentrations) and secondarily by allochthonous inputs of organic suspended matter. The river hosts three lock and dam systems upstream of the sampling site, with the closest (Lock and Dam 1) located about 5 km upstream. This river receives both point and nonpoint nutrient loading, and algal blooms form behind these dams under lentic-like conditions. Downstream, as this phytoplankton biomass dies and decomposes, it serves as an important labile contribution to the BOD load and chronic hypoxia characterizing the lower river and upper estuary. This river also hosts periodic high TSS concentrations from non–point source runoff in the Piedmont and Upper Coastal plains, some of which is likely carbonaceous and contributes to the BOD load. Strong positive correlations between phytoplankton biomass and BOD have also been reported from several Minnesota rivers (Heiskary and Markus 2001). Median BOD5 in the Cape Fear River was in the low range of the Minnesota rivers investigated by Heiskary and Markus (2001), and the mean 2002–2003 chlorophyll *a* concentration in the Cape Fear River (4.3 μg L^{-1}) was comparable to the chlorophyll *a* values in the Minnesota rivers expressing BOD5 in the 1.0–1.2 mg L^{-1} range.

Phytoplankton autotrophy did not significantly influence BOD formation in the Black or Northeast Cape Fear Rivers. These are Coastal Plain–derived black water rivers that are deep (4–11 m) and well mixed, wherein lack of solar irradiance penetration severely constrains phytoplankton growth (Mallin et al. 2004*a*). BOD in these rivers is driven by two factors. One is allochthonous non–point source inputs of organic matter from the watershed, as indicated by the correlation with fecal coliform bacteria and turbidity. The other is autochthonous growth of bacteria and fungi within the rivers, stimulated by inputs of nutrients brought in during periods of elevated river discharge (Padgett et al. 2000; Mallin et al. 2001, 2004*a*). Under experimental conditions water from the Black and Northeast Cape Fear Rivers showed significant phytoplankton biomass (and BOD) increases when provided with nitrogen inputs (ammonium or urea) and sufficient light (Mallin et al. 2001). However, orthophosphate and organic phosphorus inputs did not stimulate phytoplankton but did stimulate adenosine triphosphate production and BOD increases (Mallin et al. 2001). Later experiments using water from two black water creeks in the Black River watershed showed similar phytoplankton and BOD stimulation

Table 6. Results of correlation analyses for factors affecting BOD5 and BOD20 in three urban streams, presented as Pearson correlation coefficient (*r*)/probability (*p*). Nonsignificant ($p > 0.05$) relationships not shown.

Stream	LChl *a*	LTSS	LOP	Temp
Barnards Creek				
LBOD5	0.370 0.0400	0.451 0.0124	0.455 0.0116	
LBOD20		0.429 0.0203	0.384 0.0398	0.568 0.0013
Motts Creek				
LBOD5	0.424 0.0195			
LBOD20				0.482 0.0080
Smith Creek				
LBOD5	0.573 0.0009			
LBOD20	0.566 0.0014			0.545 0.0022

LBOD5, log BOD5; LBOD20, log BOD20; LChl *a*, log chlorophyll *a*; LTSS, log total suspended solids; LOP, log orthophosphate; Temp, water temperature.

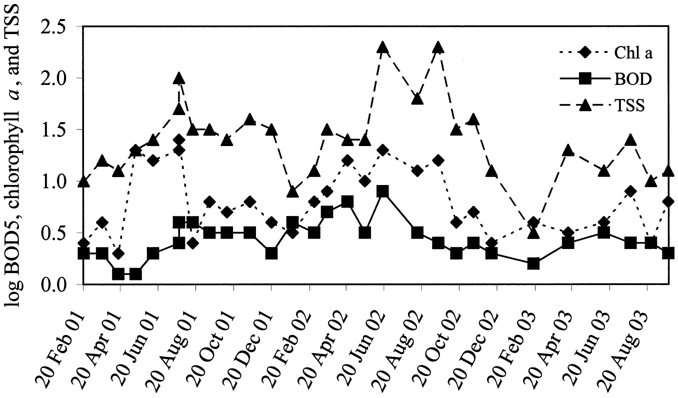

Fig. 2. Seasonal concentration patterns of BOD5 (mg L^{-1}), chlorophyll *a* (μg L^{-1}), and total suspended solids, TSS (mg L^{-1}) in Barnards Creek, North Carolina.

by nitrate and urea inputs, but phosphorus, particularly organic phosphorus, significantly stimulated bacteria abundance and BOD increases (Mallin et al. 2004*a*). A similar pattern of nitrogen stimulating autotrophic production and phosphorus stimulating bacterial production has been found in salt marsh systems (Sundareshwar et al. 2003). Thus, nutrient inputs during the eutrophication process can be reflected by microheterotroph growth, increasing the BOD load and contributing to hypoxia through a nonphotosynthetic pathway. Black water rivers contain high concentrations of bacteria and protozoan grazers (Meyer 1990) that provide a reservoir of heterotrophs whose respiration can significantly contribute to stream hypoxia, especially under nutrient loading conditions.

Finally, we note that river discharge was positively correlated with BOD5 in the Northeast Cape Fear River, and with BOD20 in this river and the Black River as well. Elevated river discharge will bring debris from riparian wetlands into the channels (as reflected by the correlation between BOD and TP and TN), as well as dissolved organic carbon (DOC; Wetzel 2001). Much of the DOC is likely to be refractory (Meyer et al. 1987; Meyer 1990) and thus would not affect BOD5 much. Refractory DOC can be photolysed by ultraviolet radiation from sunlight (Wetzel 2001), but the strong light attenuation and deep mixing in these black water rivers (Mallin et al. 2004*a*) probably do not allow for rapid photolysis to more labile compounds. Since the effect of river discharge was statistically more evident on BOD20 than BOD5 (the BOD20 analysis allows for more of the refractory compounds to be oxidized) we suspect that is the case. During high flow periods, black water rivers also receive inputs of water containing low DO from riparian swamps (Meyer 1992). Inputs of water already low in DO would make the system especially sensitive to any anthropogenic BOD increases (Meyer 1992).

Average chlorophyll *a* concentrations in the rural streams were low, but periodic algal blooms occur in most of these systems (Table 2). Chlorophyll *a* production does play a role in supplying a labile carbon source of BOD in some of these streams (Table 5). In Browns, Hammond, Six Runs, and Great Coharie Creeks, allochthonous inputs (represented as turbidity, TSS, and fecal coliform bacteria) clearly contributed to the BOD load. Browns and Hammond Creeks in particular are much more subject to anthropogenic storm water runoff inputs than the other rural streams, judging by the higher average and maximum concentrations of these pollutant variables (Table 2). Nitrogen inputs may have stimulated chlorophyll *a* concentrations and subsequent BOD in some of these creeks, as demonstrated by nutrient addition experiments with water from Great Coharie and Colly Creeks (Mallin et al. 2004*a*). Additionally, phosphorus inputs likely stimulated bacterial growth and BOD in these same creeks (Mallin et al. 2004*a*). Some creeks showed a BOD and ammonium correlation. Correlations between BOD5 and BOD20 and ammonium were found for Six Runs, Great Coharie, and Colly Creeks (Table 5). While ammonium can stimulate phytoplankton growth in these black water systems (Mallin et al. 2001), it is also a source of nitrogenous BOD in itself (Boyd 2002). Nitrogenous BOD requires longer than 5 d for oxidation but would be at least partially nitrified during the 20-d BOD analysis (Clark et al. 1977; Boyd 2000). Since ammonium discharges from point

Table 7. Results of correlation analyses for factors affecting BOD5 in a set of mesohaline tidal creeks, presented as Pearson correlation coefficient (*r*)/probability (*p*). Nonsignificant ($p > 0.05$) relationships not shown.

System	LChl *a*	LTSS	Temp
Tidal creeks			
LBOD5	0.526 0.0001	0.523 0.0001	0.486 0.0001

LBOD5, log BOD5; LChl *a*, log chlorophyll *a*; LTSS, log total suspended solids; Temp, water temperature.

Table 8. Results of correlation analyses for factors affecting BOD5 and BOD20 in two urban lakes, presented as Pearson correlation coefficient (r)/probability (p). Nonsignificant ($p>0.05$) relationships not shown.

Lake	LChl *a*	LTurb	TSS	Amm	TN	LTP	Temp
Greenfield Lake							
LBDO5	0.757 0.0069	0.832 0.0004	0.819 0.0070				
Carolina Beach Lake							
BOD5	0.613 0.0116	0.696 0.0028	NA	−0.526 0.0370	0.638 0.0078	0.633 0.0085	−0.744 0.0009
BOD20	0.655 0.0059	0.687 0.0033	NA	−0.599 0.0143	0.538 0.0315	0.609 0.0122	−0.764 0.0006

LBOD5, log BOD5; LChl *a*, log chlorophyll *a*; LTurb, log turbidity; TSS, total suspended solids; Amm, ammonium; TN, total nitrogen; LTP, log total phosphorus; Temp, water temperature.

sources are regulated in the Cape Fear watershed, potential sources for stream ammonium are the abundant CAFOs in the Black River basin (Mallin 2000). Airborne ammonium emissions from swine and possibly poultry CAFOs may be deposited in nearby undeveloped stream watersheds, as well as CAFO-rich watersheds. Other sources of ammonium in these systems include autochthonous mineralization and waste secretion by stream biota.

Stream discharge was significantly related to BOD, especially BOD20, in some of the streams (Table 5). These are primarily black water rural streams, which, like the larger rivers, drain riparian wetlands as well as agricultural lands. Flow was strongly correlated with TSS in Hammond ($r = 0.676$, $p = 0.0001$), Browns ($r = 0.962$, $p = 0.0001$), Six Runs ($r = 0.577$, $p = 0.0002$), and Great Coharie Creeks ($r = 0.504$, $p = 0.0013$), and similar correlations occurred between flow and turbidity, fecal coliform bacteria, and some nutrients (mainly TN). Thus, flow-driven non–point source inputs from the watershed clearly contribute to BOD, especially BOD20. DOC serves as a source of nutrition to stream bacteria (Meyer et al. 1987; Meyer 1990; Wetzel 2001), and in black water especially it contributes to high respiration levels (Edwards and Meyer 1987; Meyer 1992). However, as mentioned, DOC is refractive, and its role on stream hypoxia is not yet well defined. Colly Creek showed a strong correlation between BOD and chlorophyll *a*, as well as nutrients (mainly nitrogen). This is a nearly pristine black water stream with low turbidity and fecal coliform counts and little anthropogenic pollutant loading (Mallin et al. 2004*a*). Apparently what little chlorophyll *a* is produced in this stream provides a primary source of BOD. The negative correlation between BOD and stream discharge in this system was unique among this set of streams. Evidently chlorophyll *a* is produced primarily during low-flow conditions in this system, and there are low inputs of labile carbon from the watershed during elevated discharge periods. Stream discharge was negatively correlated with nitrate, chlorophyll *a*, and turbidity in this unaffected system. Little Coharie Creek drains an agricultural watershed, but this analysis was unable to discern the principal factors contributing to BOD in this system (Table 5). We note overall that these rural streams maintained relatively low BOD levels over the course of the study. BOD5 values exceeding 4.0 mg L^{-1} have only been documented in these systems following swine waste lagoon spills (Mallin 2000) and hurricane events (Mallin et al. 2002).

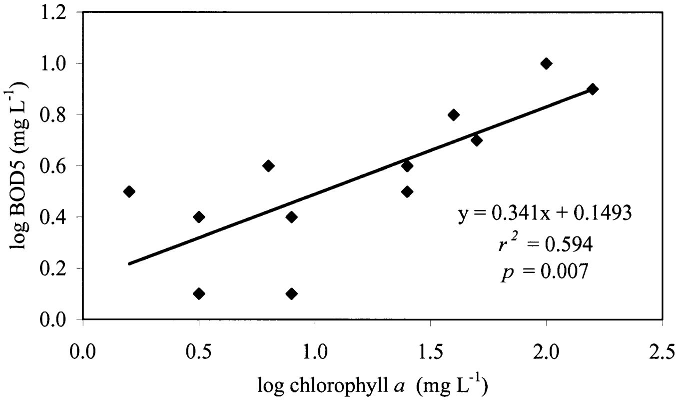

Fig. 3. BOD5 as a function of chlorophyll *a* in Greenfield Lake, Wilmington, North Carolina.

The principal factor that influenced BOD in the three urban streams was chlorophyll *a* (Table 6). All three of these streams host periodic phytoplankton blooms (Table 2) brought about by nutrient pulses from urban and suburban runoff (Table 3). BOD20 was correlated with water temperature, perhaps a result of increased summer chlorophyll *a* and/or a general increase in microbial activity with temperature. These streams are also subject to pulses of TSS (Table 2; Fig. 2), which may be inorganic or organic or both in nature. Since fecal coliform bacteria pulses also occur in these streams (Table 2), some of the anthropogenic loading is likely very labile organic material, such as pet and wild animal manure. Other sources of BOD loading into these urban streams include organic soils disrupted during land clearing operations for developments or roads, vegetative materials, and organic fertilizers used in gardens or on golf courses. There are a plethora of more complex, refractive carbonaceous compounds from automobiles, road maintenance, and other operations that enter urban streams in storm water runoff, mainly associated with suspended particles (Paul and Meyer 2001), some of which may exert a BOD. Thus, in this set of urban streams the eutrophication process plays the major role in driving oxygen demand, but on other occasions storm water runoff carrying the waste of the urbanization process likely affects stream BOD. As mentioned, BOD in these urban streams was about double that of the rural streams. The deforestation and construction of increased impervious surface coverage during urbanization will alter the storm hydrograph (Booth 1991), causing more rapid and intense storm water runoff of nutrients and oxygen-demanding materials than in the more forested rural streams (Paul and Meyer 2001). The higher TSS, turbidity, fecal coliform counts, and nutrient maxima in the urban streams tend to support this premise (Table 2; Table 3).

The tidal creeks showed a strong relationship between BOD5 and phytoplankton abundance. These creeks are primarily nitrogen limited, with periodic phosphorus or light limitation in upper, more eutrophic reaches (Mallin et al. 2004*b*). Blooms exceeding 200 μg L^{-1} of chlorophyll *a* have been recorded in upper reaches of these creeks (Mallin et al. 1999, 2004*b*), and nitrate concentrations in upper creeks can be up to 20 times higher than in lower creeks (Mallin et al. 2004*b*). Hypoxia in these creeks is influenced by sediment oxygen demand (SOD), as well as by BOD. In a comparison of these processes MacPherson (2003) found that BOD played the dominant role in summer and spring, while SOD dominated oxygen demand in winter and fall, with SOD exerting a greater overall annual effect on oxygen demand in these creeks. The correlation between BOD5 and water temperature appears to be a result of the very low winter chlorophyll *a* levels in the tidal creeks (Mallin et al. 2004*b*).

Chlorophyll *a* in Greenfield Lake was strongly correlated with BOD5 (Table 8). While BOD5 was also correlated with turbidity, the TSS and turbidity in this lake were primarily caused by phytoplankton, which can demonstrate significant blooms (Table 2). This lake collects suburban drainage from a watershed consisting of residential areas, commercial districts, and a golf course. The lake is shallow with several inflowing streams but only one outflow; thus, anthropogenic nutrients (and waterfowl droppings) create an environment prone to algal and free-floating macrophyte bloom formation, which upon death and decay become the primary sources of BOD in the lake.

BOD in Carolina Beach Lake was clearly related to phytoplankton abundance as well. The average BOD5 concentration of 2.9 mg L^{-1} was high compared with most of the systems investigated, as was the average lake chlorophyll *a* concentration of 13.3 μg L^{-1} (Table 2). The correlation between BOD and TN and TP may reflect nutrients contained within algal biomass. The low anthropogenic turbidity load in this urban lake (Table 2) suggests that turbidity in this lake is primarily due to phytoplankton. The negative correlation between BOD and water temperature may at first seem counterintuitive, but there are sound reasons for this. First, water temperatures ranged from 9°C to 30°C over the study period, suitable conditions for phytoplankton growth in this shallow, southeastern lake. Lake chlorophyll *a* abundance was actually highest on average (21 μg L^{-1}) during the January sampling, as was BOD5 (3.7 mg L^{-1}). Second, during the warmer periods there was considerably higher macrophyte biomass in the lake, which may have outcompeted the phytoplankton for nutrients (Richard and Small 1984). The effect of decaying macrophyte material may be more reflected in SOD than in BOD. This lake is spring fed but receives new nutrients from urban surface runoff through inflow ditches around the lake and presumably recycles a considerable amount of nutrients from decaying phytoplankton and macrophytes. Hypoxia in this shallow lake appears to be a result of autotrophic processes stimulated by nutrient loading.

Thus, phytoplankton production creates the clearly dominant reservoir of labile carbon driving BOD, and subsequent hypoxia, in several situations. These include a Piedmont river subject to algal blooms above a series of locks and dams, three urban creeks receiving nutrient pulses via storm water runoff, a set of mesohaline tidal creeks receiving anthropogenic nutrient loading, and two urban lakes receiving nutrient loading via storm water runoff. Autotrophic phytoplankton production plays a role in BOD creation in some rural streams. Heterotrophic processes, stimulated primarily by phosphorus and secondarily by nitrogen loading, create the primary BOD in two large black water rivers and some rural black water streams, with inputs of organic materials from riparian wetlands or agricultural areas contributing as well. Watershed non–point source inputs of biochemical oxygen-demanding materials contribute to stream BOD in urban streams and black water rivers. BOD and subsequent hypoxia may have single or multiple factors driving these processes, depending upon the system.

As mentioned earlier, management schemes for the control of hypoxia in a water body often do not fully consider the various ways in which nitrogen and phosphorus loading stimulate BOD. We suggest that reductions of incidents of hypoxia can be better achieved by a system-specific approach that examines an array of factors potentially influencing BOD that encompass both autochthonous and allochthonous variables. In some circumstances targeting the nutrient (or nutrients) stimulating phytoplankton blooms will suffice, but in other situations targeting nutrient(s) limiting bacterial production are necessary. Limiting non–point source inputs of oxygen-demanding materials derived from urbanization, agriculture, or other land disrupting activities is critical in some cases.

For example, our data show different variables stimulating BOD according to the individual system. The Cape Fear River BOD load is primarily associated with chlorophyll *a* production, which comes largely from phytoplankton blooms forming behind the three dams and subsequently transported downstream. Management of BOD for this river should concentrate on upper watershed agricultural nutrient runoff (buffer creation/enhancement) and increased nutrient removal from municipal wastewater discharges. An additional system manipulation approach could include the removal of the three lock and dams (built long ago for barge traffic that no longer exists) to help reduce lentic conditions supporting bloom formation (as well as enhance anadromous fish movement). Reduction of BOD in the two black water rivers would be best accomplished by phosphorus and organic waste loading decreases. Since soils in these CAFO-dominated watersheds are becoming P saturated (Cahoon and Ensign 2004), this would involve mandated streamside buffer zones along animal waste lagoon sprayfields to retard movement of sediment-associated animal waste (and associated phosphorus) into nearby streams. Of course, the most appropriate course would be actual primary and secondary treatment of the animal waste (with chemical P removal) rather than the present lagoon/sprayfield system. Other rural streams would benefit from mandatory streamside buffers, whether CAFOs or traditional agriculture dominates watershed land use.

BOD in the three urban streams, the tidal creeks, and the two urban lakes is strongly associated with chlorophyll *a* (Tables 6, 7, and 8). Entry of nutrients and suspended solids into the set of urban streams appears to be largely storm water driven, aided by deforestation and the urbanization process. Increased usage of wet detention ponds to control sedimentation and cultivation of forested buffer zones to enhance infiltration of runoff are potential options to reduce nutrient inputs and smooth out the hydrograph. Production of phytoplankton in the tidal creeks is primarily limited by nitrogen (Mallin et al. 2004*b*), arguing for nitrogen-reducing best management practices (BMPs). There are a number of storm water outfalls that directly enter these creeks without any kind of pretreatment. Mandatory use of wet detention ponds to reduce suspended sediment inputs (and associated phosphorus), coupled with constructed wetlands (to remove nitrogen through denitrification and plant uptake), would reduce bloom formation.

A number of tributary streams that drain urban and suburban subwatersheds enter Greenfield Lake. Use of water retention devices and enhanced wetland areas on these tributaries prior to entry into the lake would help reduce nitrogen inputs, and subsequent algal blooms. The challenge would be finding appropriate space for such BMPs. Reduction of algal bloom formation in the urban lakes (as well as urban streams and tidal creeks) would also be aided by pet waste ordinances that are enforced by authorities. Dog and cat manure contains nitrogen and phosphorus that is easily washed of the landscape into nearby urban water bodies during rain events. Removal of this nutrient source into landfills, treatment plants, or pet waste digesters prevents a widespread source of nutrients (and BOD) from entering into bloom-prone urban water bodies and the subsequent hypoxia problems that follow.

Finally, an important, but less well-studied contributing factor to hypoxia is SOD. It is infrequently measured because of inherent difficulties with in situ techniques, and methods vary considerably among researchers. However, on an annual basis SOD can exert an overall greater effect on stream DO concentrations than BOD in some situations (MacPherson 2003). In black water systems especially, the high natural bacteria concentrations within the sediments can significantly influence stream heterotrophy (Meyer 1990; Fuss and Smock 1996). The connection between SOD and the eutrophication process should be an important area of further investigation in the study of stream and lake hypoxia.

References

[APHA] American Public Health Association. 1995. Standard methods for the examination of water and wastewater, 19th ed. APHA.

Arbuckle, K. E., and J. A. Downing. 2001 The influence of watershed land use on lake N:P in a predominantly agricultural watershed. Limnol. Oceanogr. **46:** 970–975.

Boesch, D. F., R. B. Brinsfield, and R. E. Magnien. 2001. Chesapeake Bay eutrophication: Scientific understanding, ecosystem restoration, and challenges for agriculture. J. Environ. Qual. **30:** 303–320.

Booth, D. B. 1991. Urbanization and the natural drainage system—impacts, solutions, and prognoses. Northwest Environ. J. **7:** 93–118.

Boyd, C. E. 2000. Water quality: An introduction. Kluwer.

Breitburg, D. L., T. Loher, C. A. Pacey, and A. Gerstein. 1997. Varying effects of low dissolved oxygen on trophic interactions in an estuarine food web. Ecol. Monogr. **67:** 489–507.

Bricker, S. B., C. G. Clement, D. E. Pirhalla, S. P. Orlando, and D. R. G. Farrow. 1999. National Estuarine eutrophication assessment: Effects of nutrient enrichment in the nation's estuaries. NOAA, National Ocean Service, Special Projects Office and the National Centers for Coastal Ocean Science.

Burkholder, J. M. 2001. Eutrophication and oligotrophication, p. 649–670. *In* S. Levin [ed.], Encyclopedia of biodiversity, v2. Academic.

Cahoon, L. B., and S. H. Ensign. 2004. Spatial and temporal variability in excessive soil phosphorus levels in eastern North Carolina. Nutrient Cycling Agroecosyst. **69:** 111–125.

Clark, J. W., W. Viessman, Jr., and M. J. Hammer. 1977. Water supply and pollution control, 3rd ed. IEP, Dun-Donnelley Publ., New York.

Dame, R., and others. 2000. Estuaries of the south Atlantic coast of North America: Their geographical signatures. Estuaries **23:** 793–819.

Diaz, R. J., and R. Rosenberg. 1995. Marine benthic hypoxia: A review of its ecological effects and the behavioural responses of benthic macrofauna. Oceanogr. Mar. Biol. Annu. Rev. **33:** 245–303.

Edwards, R. T., and J. L. Meyer. 1987. Metabolism of a subtropical low gradient blackwater river. Freshw. Biol. **17:** 251–263.

Fuss, C. L., and L. A. Smock. 1996. Spatial and temporal variation of microbial respiration rates in a blackwater stream. Freshw. Biol. **36:** 339–349.

Heiskary, S., and H. Markus. 2001. Establishing relationships among nutrient concentrations, phytoplankton abundance, and biochemical oxygen demand in Minnesota, USA, rivers. J. Lake Reserv. Manage. **17:** 251–267.

Lenihan, H. S., and C. H. Peterson. 1998. How habitat degradation through fishery disturbance enhances impacts of hypoxia on oyster reefs. Ecol. Appl. **8:** 128–140.

MacPherson, T. A. 2003. Sediment oxygen demand and biochemical oxygen demand: Patterns of oxygen depletion in tidal creek study sites. M.S. thesis, Univ. of North Carolina at Wilmington.

Mallin, M. A. 2000. Impacts of industrial-scale swine and poultry production on rivers and estuaries. Am. Sci. **88:** 26–37.

———, L. B. Cahoon, D. C. Parsons, and S. H. Ensign. 2001. Effect of nitrogen and phosphorus loading on plankton in Coastal Plain blackwater streams. J. Freshw. Ecol. **16:** 455–466.

———, E. C. Esham, K. E. Williams, and J. E. Nearhoof. 1999. Tidal stage variability of fecal coliform and chlorophyll *a* concentrations in coastal creeks. Mar. Poll. Bull. **38:** 414–422.

———, M. R. McIver, S. H. Ensign, and L. B. Cahoon. 2004*a*. Photosynthetic and heterotrophic impacts of nutrient loading to blackwater streams. Ecol. Appl. **14:** 823–838.

———, D. C. Parsons, V. L. Johnson, M. R. McIver, and H. A. CoVan. 2004*b*. Nutrient limitation and algal blooms in urbanizing tidal creeks. J. Exp. Mar. Biol. Ecol. **298:** 211–231.

———, M. H. Posey, M. R. McIver, D. C. Parsons, S. H. Ensign, and T. D. Alphin. 2002. Impacts and recovery from multiple hurricanes in a Piedmont-Coastal Plain river system. BioScience **52:** 999–1010.

MEYER, J. L. 1990. A blackwater perspective on riverine ecosystems. BioScience **40:** 643–651.

———. 1992. Seasonal patterns of water quality in blackwater rivers of the coastal plain, southeastern United States. *In* C. D. Becker, and D. A. Neitzel [eds.], Water quality in North American river systems. Batelle.

———, R. T. EDWARDS, AND R. RISLEY. 1987. Bacterial growth on dissolved organic carbon from a blackwater river. Microb. Ecol. **13:** 13–29.

[NCDEHNR] NORTH CAROLINA DEPARTMENT OF ENVIRONMENT, HEALTH, AND NATURAL RESOURCES. 2000. Cape Fear River Basinwide Water Quality Plan. NCDEHNR, Division of Water Quality, Water Quality Section.

[NRC] NATIONAL RESEARCH COUNCIL. 1993. Managing wastewater in coastal urban areas. National Academy Press.

———. 2000. Clean coastal waters: Understanding and reducing the effects of nutrient pollution. National Academy Press.

OFFICER, C. B., R. B. BIGGS, J. L. TAFT, L. E. CRONIN, M. A. TYLER, AND W. R. BOYNTON. 1984. Chesapeake Bay anoxia: Origin, development, and significance. Science **223:** 22–27.

PADGETT, D. E., M. A. MALLIN, AND L. B. CAHOON. 2000. Evaluating the use of ergosterol as a bioindicator for assessing water quality. Environ. Monit. Assess. **65:** 547–558.

PARSONS, T. R., Y. MAITA, AND C. M. LALLI. 1984. A manual of chemical and biological methods for seawater analysis. Pergamon.

PAUL, M. J., AND J. L. MEYER. 2001. Streams in the urban landscape. Annu. Rev. Ecol. Syst. **32:** 333–365.

RABALAIS, N. N., R. E. TURNER, AND W. J. WISEMAN, JR. 2001. Hypoxia in the Gulf of Mexico. J. Environ. Qual. **30:** 320–329.

RICHARD, D. I., AND J. W. SMALL, JR. 1984. Phytoplankton responses to reduction and elimination of submerged vegetation by herbicides and grass carp in four Florida lakes. Aquat. Bot. **20:** 307–319.

SCHLOTZHAUER, S. D., AND R. C. LITTELL. 1987. SAS system for elementary statistical analysis. SAS Institute, Inc.

SUNDARESHWAR, P. V., J. T. MORRIS, E. K. KOEPFLER, AND B. FORWALT. 2003. Phosphorus limitation of coastal ecosystem processes. Science **299:** 563–565.

VAN DOLAH, R. F., AND G. S. ANDERSON. 1991. Effects of Hurricane Hugo on salinity and dissolved oxygen conditions in the Charleston Harbor estuary. J. Coast. Res. **8(SI):** 83–94.

WELSCHMEYER, N. A. 1994. Fluorometric analysis of chlorophyll *a* in the presence of chlorophyll *b* and phaeopigments. Limnol. Oceanogr. **39:** 1985–1993.

WETZEL, R. G. 2001. Limnology: Lake and river ecosystems, 3rd ed. Academic.

Received: 28 March 2004
Accepted: 23 September 2004
Amended: 21 January 2005