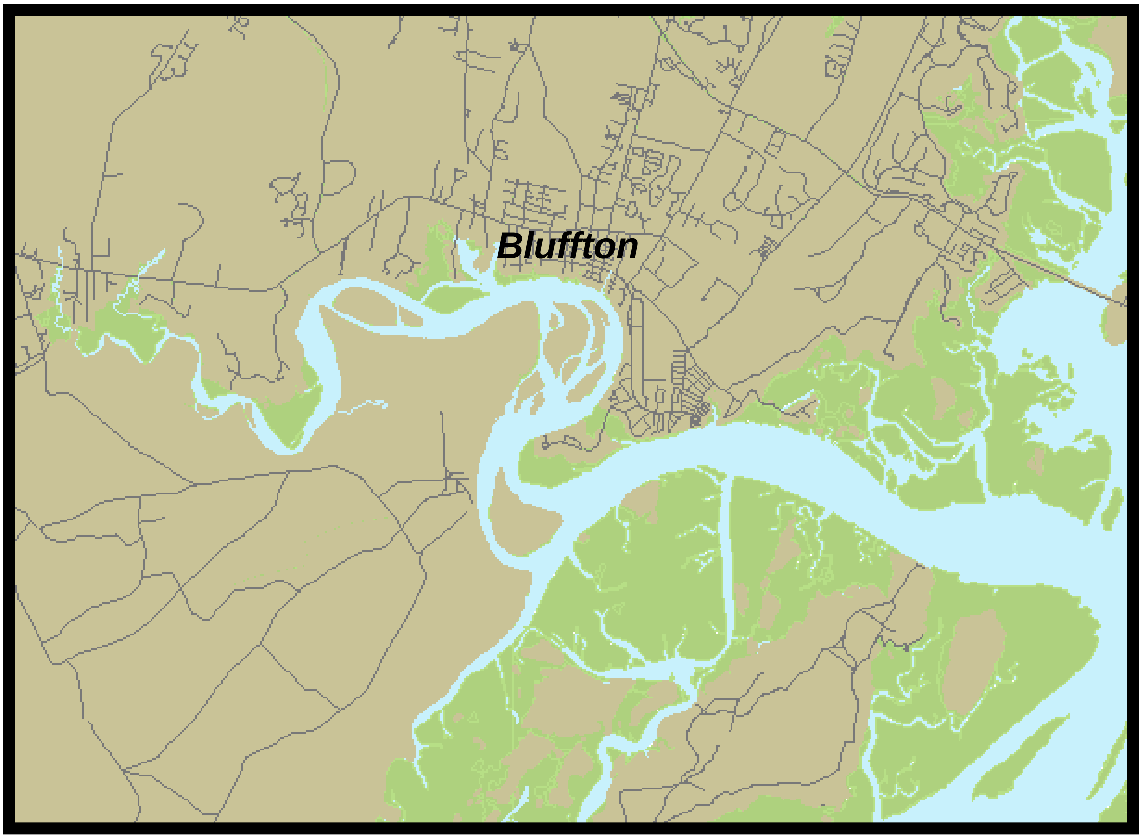
A Baseline Assessment of Environmental and Biological Conditions in the May River, Beaufort County, South Carolina



Final Report Submitted To:
The Town of Bluffton
2004

Submitted by:

South Carolina Department of Natural Resources
USGS science for a changing world
NOAA National Oceanic and Atmospheric Administration U.S. Department of Commerce

Final Report

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Submitted To:

The Town of Bluffton

Submitted By:

Marine Resources Division
South Carolina Department of Natural Resources
217 Ft. Johnson Rd.
Charleston, SC 29412

in cooperation with the

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Center for Coastal Environmental Health
and Biomolecular Research
National Ocean Service
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219 Ft. Johnson Rd.
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A Baseline Assessment of Environmental and Biological Conditions in the May River, Beaufort County, South Carolina

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I. INTRODUCTION

A. Background

The May River represents a significant estuary in Beaufort County that may be adversely affected by planned developments that will result in major land-use changes. These developments will significantly increase the population of the Town of Bluffton, which currently has a population of approximately 1,275 (U.S. Census Bureau, 2002). The May River is designated as an Outstanding Resource Water (ORW) by the South Carolina Department of Health and Environmental Control (SCDHEC, 2001) and is a valued resource of this small coastal community, particularly for its oyster production, which has historically been a significant economic resource for the area. Because the May River is considered a water resource of exceptional importance and value, residents of Bluffton and the surrounding area are concerned that the health of the May River and its oyster beds might be compromised by development of the area. The Town of Bluffton therefore commissioned the Marine Resources Research Institute of the South Carolina Department of Natural Resources (SCDNR), the U.S. Geological Survey, South Carolina District (USGS), and the National Oceanic and Atmospheric Administration's Center for Coastal Environmental Health and Biomolecular Research (NOAA-CCEHBR) to undertake a multidisciplinary study of the May River. The study was conducted to assess the water, sediment, and biological quality of the entire riverine system in 2002-03 and provide a comprehensive database of these conditions prior to any major development activities in the watershed. With the exception of a few stations sampled by SCDHEC and SCDNR as part of existing monitoring programs, limited data were available on the current condition of this estuarine river.

Over the next several decades, the coastal watersheds in the southeastern United States are projected to experience a high rate of human development (Culliton and others, 1990; Cohen and others, 1997). The construction of infrastructure (roadway systems, commercial development, residential housing, and industrial facilities) that accompanies human development will alter the rate and volume of freshwater inflow as well as the type and amount of pollutants introduced into estuaries (Fulton and others, 1993; Lerberg and others, 2000; Mallin and others, 2000). Estuaries, particularly tidal creeks, provide nursery habitat for many species of fish, shrimp, and crabs as well as feeding grounds for wading birds (Wenner and Beatty, 1993; Dodd and Murphy, 1996). Approximately 85% of commercially-harvested fish depend on estuarine habitats for at least part of their life cycle and contribute to an estimated 31% of the gross national product (National Research Council, 1997). Estuaries are also valuable resources for recreational activities. It is estimated that 180 million people use coastal resources each year for swimming and boating (National Research Council, 1997).

A fine line exists between the usage of estuarine resources and the exploitation of an important ecosystem. Coastal areas possess many desirable qualities for industries, residents, and tourists. Consequently, they are the most heavily developed areas of the United States. Coastal areas constitute only 17% of the nation's area, and yet are home to over half of the national population (National Research Council, 2000). On global, regional, and local scales, coastal populations are ever increasing. It is estimated that by the year 2020, 75% of the world's population (approximately 6 billion people) will live within 60 km of the coast (Kennish, 2002).

Population growth and uncontrolled development in coastal regions are reported to have deleterious impacts on estuarine function and overall quality (Nixon, 1995). On a gross ecological scale, degraded areas are associated with lower species diversity, less trophic complexity, altered food webs, altered community composition, and reduced habitat diversity (Nixon, 1995). These land cover changes are projected to adversely affect the productivity, biodiversity, and ecological functioning of coastal ecosystems, particularly the tidal creeks which are the first order connections between uplands and estuaries (Olsen and others, 1982; Arnold and Gibbons, 1996; Vitousek and others, 1997; Sanger and others, 1999a, 1999b; Lerberg and others, 2000).

Studies to evaluate the impact of development in a given area can be designed in two general ways: (1) before- and after-impact studies, and (2) comparison of an impacted area to a control area. In general, before- and after-impact studies are rare. In fact, little effort has been devoted toward gathering baseline data to characterize estuarine systems before the on-set of major pollution problems (Kennish, 2002). Such data are critical to detect future changes in biological responses and quantify lost resources (Clark and Greene, 1988).

Furthermore, most successful environmental monitoring programs are based upon a weight of evidence approach, which provides a comprehensive ecosystem-level assessment (National Research Council, 1990). Rather than relying on sampling a single parameter or making a single measure, successful approaches characterize environmental conditions utilizing a suite of metrics that include water, sediment, and biological quality (Chapman, 1989). The suite of parameters that have proven to be important for evaluating the quality of tidal creeks and their surrounding watershed include the percentage of impervious cover, the concentration of nutrients, the concentration of fecal coliform bacteria, the presence of sediment contaminants, and the abundance of macrobenthic invertebrates (Holland and others, 2004).

The Town of Bluffton has been extremely proactive in seeking funding to perform a comprehensive study of the May River. The current report represents the status of the May River before significant land use changes occur and will enable the Town of Bluffton to continue monitoring the system in the hopes of limiting the impacts of major-development on the May River Estuary.

B. Scope of Study

In order to adequately sample the entire May River Estuary, the river was divided into three general zones (i.e., Upper or Zone 1, Middle or Zone 2, Lower or Zone 3). One USGS continuous gauging station was located in each zone (Figure I-1). These stations measured a suite of water quality and quantity metrics including dissolved oxygen, temperature, conductivity, water level, and flow. Data obtained from these stations is summarized in Section II of this report.

The May River Project team determined that four habitats would be sampled. These habitats provide a comprehensive assessment of the May River estuarine system and are comparable to data collected from similar habitats around the state. The four habitats include headwater tidal creeks, larger tidal creeks, open water, and oyster reefs.

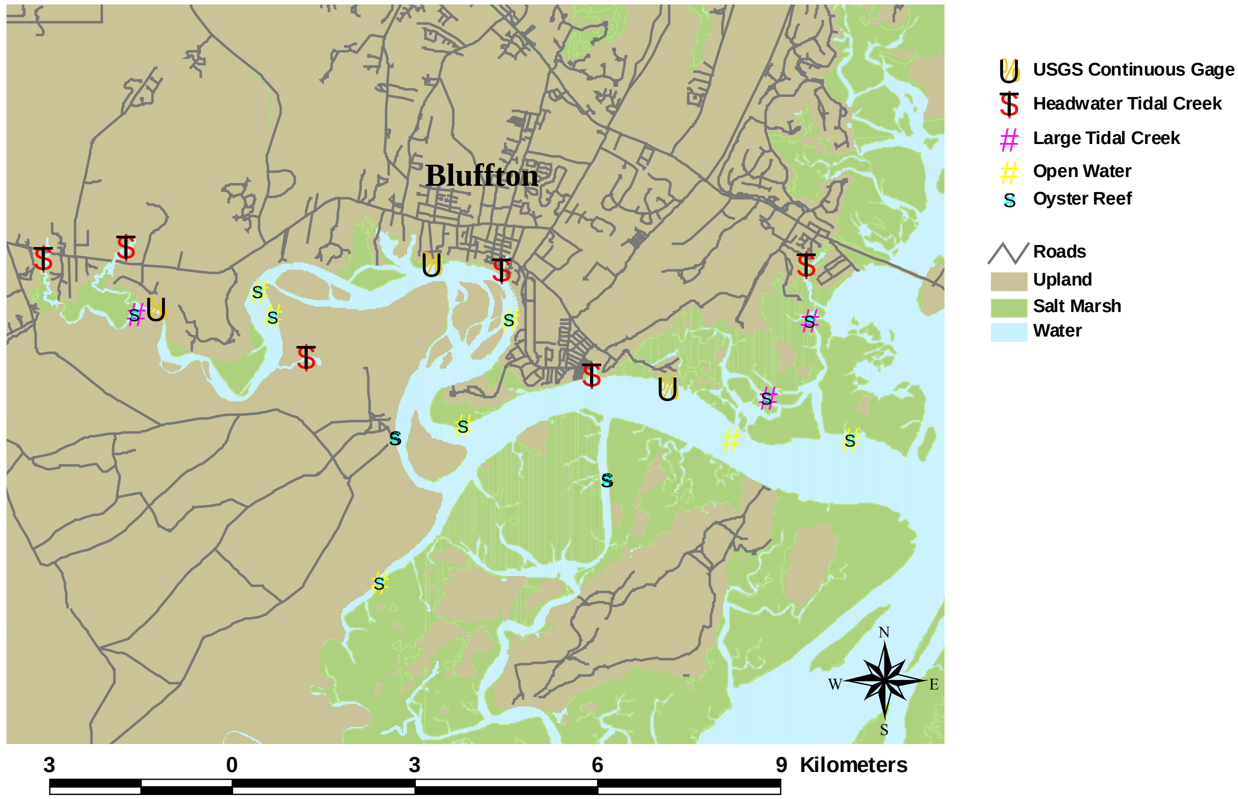


Figure I-1. The sampling sites for the May River estuarine system.

Headwater tidal creeks form the primary linkage between the estuarine area and the upland area. A headwater tidal creek was defined as a 600 m section of the creek starting at the point where water depth in the channel was approximately 1 m deep at mean high tide. Six headwater tidal creeks were sampled for this study and sampling was predominantly conducted intertidally using methods similar to those used in the SCDNR Tidal Creek Project (TCP; Holland and others, 1996, 2004). Three creeks were located in the Upper Zone, two creeks were located in the Middle Zone, and one creek was located in the Lower Zone (Figure I-1). A series of water quality, phytoplankton, and fecal coliform parameters were sampled during four seasons for a one-year period starting in the spring of 2002 and ending in the winter of 2003. In addition, the benthic community, nektonic community, sediment chemistry and toxicity, and additional water quality measures including bacterial typing, were assessed in the summer of 2002. Two headwater creeks were also sampled for wastewater indicators. The headwater tidal creek data were compared to similar habitats previously sampled as part of the TCP and are summarized in Section III of this report.

Three large tidal creeks and seven open water sites were sampled subtidally in the May River Estuary (Figure I-1) using methods similar to the South Carolina Estuarine and Coastal Assessment Program (SCECAP) described by Van Dolah and others (2002). Three sites were sampled in the Upper Zone, three sites were sampled in the Middle Zone, and four sites were sampled in the Lower Zone. A large tidal creek represents estuarine water bodies less than 100 m in width from marsh bank to marsh bank and at least 1 m deep at mean low water. An open water site represents estuarine water bodies greater than 100 m in width. A series of water quality and phytoplankton community parameters were sampled during each of the four seasons for a one-year period starting in the spring of 2002 and ending in the winter of 2003. In addition, the benthic community, nektonic community, sediment chemistry and toxicity, and additional water quality measures including bacterial typing, were assessed in the summer of 2002. The large tidal creek and open water sites sampled in the May River were compared to a comparable set of relatively pristine SCECAP stations also sampled in 2002 located in Beaufort, Jasper, and Colleton counties. Results obtained from the large tidal creeks and open water habitats are also summarized in Section III of the report.

The oyster reefs sampled in this study were located near the large tidal creek and open water sites whenever possible (Figure I-1). This occurred for nine sites with an additional two sampling sites chosen to ensure the entire May River was spatially represented. The reefs were sampled for oyster size and abundance, disease, health, and tissue contamination in the summer of 2002. Recruitment of oysters to each of the sites was determined by deploying oyster shells at each site for an approximately seven-month period (September 2002 to March 2003). The oyster reef data collected in the May River were compared to similar habitats previously sampled as part of the Oyster Reef Project and Biomarker Programs at the SCDNR. The findings are described in Section IV of this report.

Section V of the report presents the conclusions of the study as well as the recommendations of the researchers to the Town of Bluffton. The USGS fully supports the scientific findings and interpretations of this study; however, as a matter of policy, the USGS does not endorse or make recommendations. In addition to this report, the May River Project will provide the Town of Bluffton with an Access® database that contains all of the data collected from this study. This will allow all the data from this study to be accessed at a single location to facilitate future studies.

C. Description of Study Area

The May River is located in Beaufort County, South Carolina between Hilton Head Island and the mainland. As a part of the larger Broad Sound Estuarine System (Dardeau and others, 1992) that extends up the mouth of the Broad River, the May River is influenced by semi-diurnal tides with nearly equal high and low water periods each day. The tidal range in the May River is approximately 2-3 meters. The May River watershed as defined by the USGS 14 digit hydrologic unit code (HUC) is 10,353 hectares in size (Figure I-2). It is therefore affected by freshwater inflow from a drainage area of approximately 8,870 hectares of upland area.

The physiography of this region includes estuaries, tidal marshes, lagoons, and beaches. Natural vegetation includes species found in salt and brackish marshes, such as cord grass (*Spartina* spp.), and rushes (*Juncus* spp.); trees of the maritime swamp forests, such as tupelo (*Nyssa* spp.), red maple (*Acer rubrum*), sweetgum (*Liquidambar styraciflua*), and bald cypress (*Taxodium distichum*); maritime evergreen forest, with slash pine (*Pinus elliottii*), loblolly pine (*P. taeda*), live oak (*Quercus virginiana*), laurel oak (*Q. laurifolia*); and dune grasses, such as sea oats (*Uniola paniculata*), and panic grasses (*Panicum* spp.) (Radford and others, 1968; Griffith and others, 2002).

The May River and surrounding area lies within the Coastal Plain physiographic province. The geological descriptive is that of surficial material and bedrock, with Holocene saline marsh deposits (silt, sand, clay), Holocene dune and beach sand and Pleistocene beach and

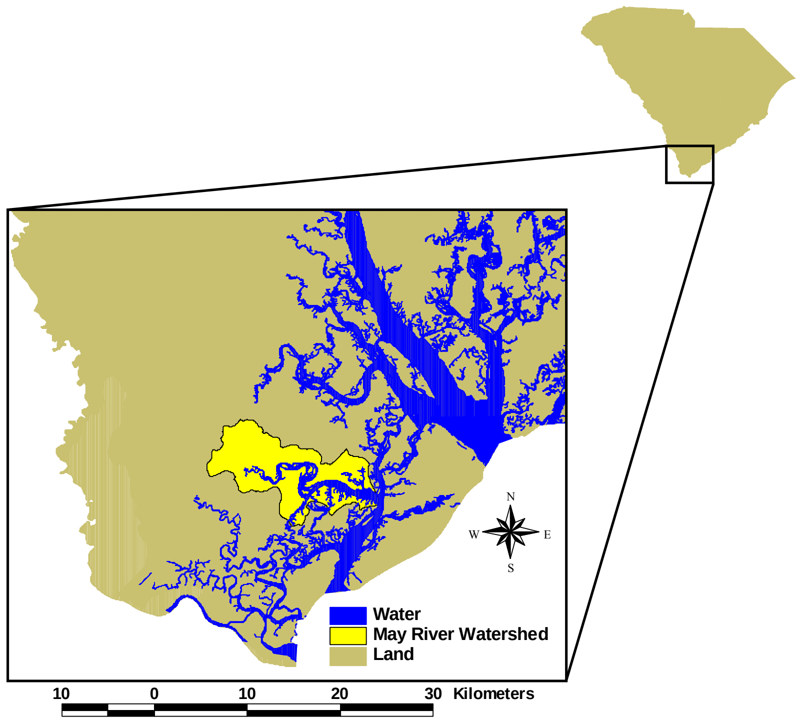


Figure I-2. The USGS 14 digit HUC watershed for the May River.

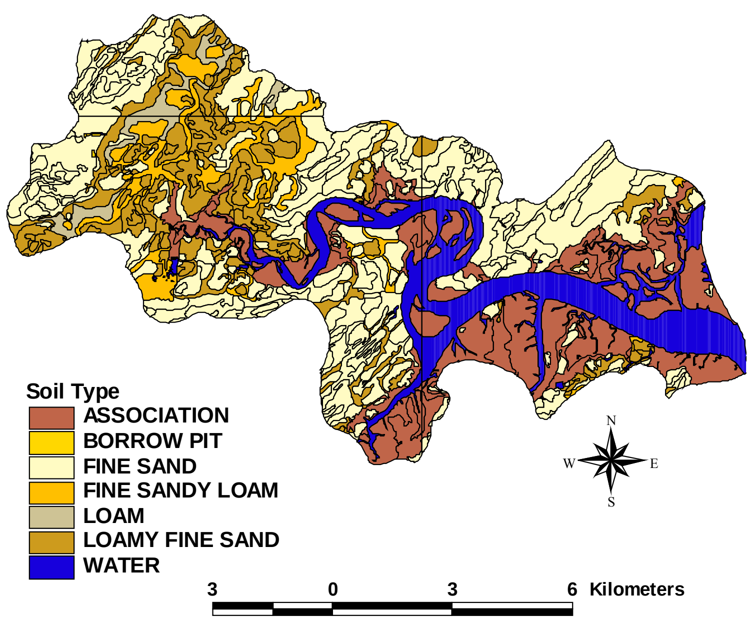


Figure I-3. The soil types of the May River watershed.

near shore marine sand (Griffith and others, 2002). The predominant type of soil on the May River watershed upland is fine sand (4,097 ha, 39.6%) followed by loamy fine sand (1,677 ha, 16.2%) and fine sandy loam (644 ha, 6.2%) (Figure I-3). The marsh soil type is designated as "Association" (U.S. Department of Agriculture, 1997).

Human population density can be indicative of the amount of urbanization in the watershed. The population density in the May River watershed calculated from the 2000 census block data is approximately 5,522 people or 0.5 individuals per hectare (U.S. Census Bureau, 2002). The population in the Town of Bluffton (based on the 2000 census) was 1,275 people (U.S. Census Bureau, 2002).

Aerial photographs from 1994 and 1999 indicate that the land cover in the watershed has not changed dramatically in the 5-year period between flights (Figure I-4). However, beginning in the year 2002, the Town of Bluffton grew from an area of about 2.6 square kilometers to approximately 130 square kilometers by the annexation of 12,950 hectares. Development plans over the next 30 years include construction of about 19,000 residences on lots ranging from 0.2 to about 1.2 hectares, and 324 hectares of commercial establishments, parks and managed forests. The 2002 National Wetlands Inventory classifies the May River watershed as 14.3% bay/estuarine water, 22.5% non-forested wetland, 0.1% commercial, 8.3% residential, 4.5% cropland/pasture, 14.7% planted pine, and 33.3% other forested types (Figure I-5) (U.S. Fish & Wildlife Service, 2002).

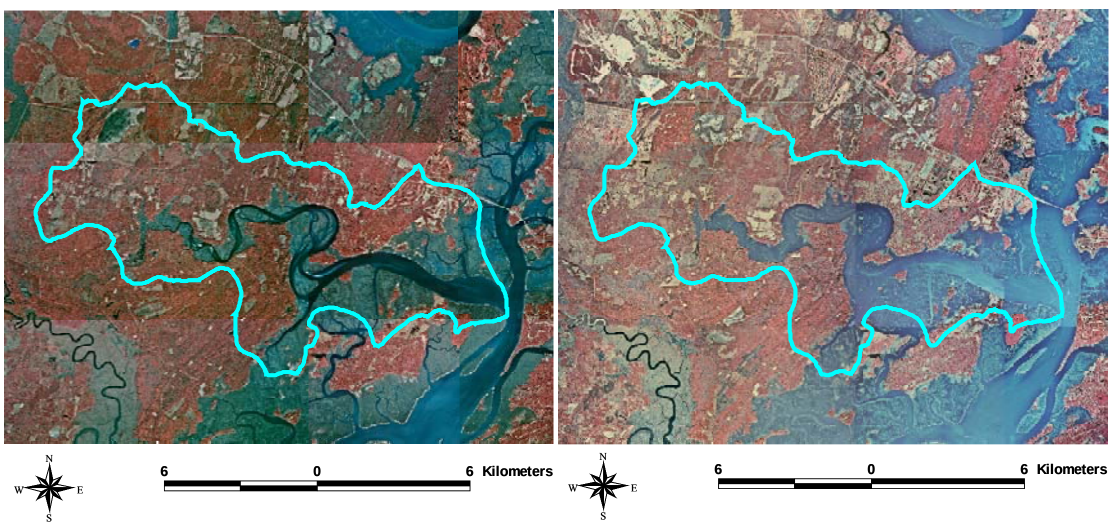
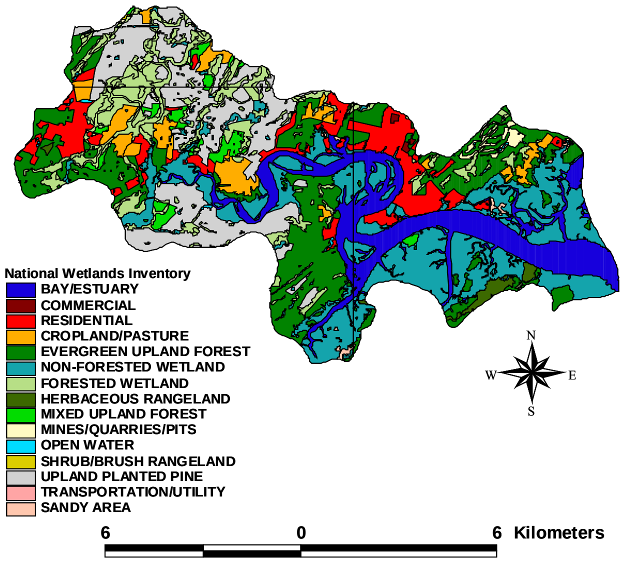


Figure I-4. The 1994 (left) and 1999 (right) National Aerial Photographic Program (NAPP) aerial photographs of the May River watershed.



Land Use (NWI)	Area (hec)	% Area
BAY/ESTUARY	1483	14.33
NON-FORESTED WETLAND	2324	22.45
COMMERCIAL	14	0.13
RESIDENTIAL	860	8.31
CROPLAND/PASTURE	472	4.56
TRANSPORTATION/UTILITY	5	0.05
UPLAND PLANTED PINE	1520	14.69
EVERGREEN UPLAND FOREST	2022	19.53
FORESTED WETLAND	1208	11.66
HERBACEOUS RANGELAND	90	0.87
MINES/QUARRIES/PITS	68	0.65
MIXED UPLAND FOREST	213	2.06
OPEN WATER	32	0.31
SANDY AREA	39	0.38
SHRUB/BRUSH RANGELAND	2	0.02
TOTAL	10353	

Figure I-5. The 2002 National Wetlands Inventory for the May River watershed.

II. CONTINUOUS WATER QUALITY/QUANTITY

A. Introduction

Estuarine rivers, such as the May River, are highly complex systems that result from interaction between various physical forces including semi-diurnal (twice a day) tidal variations, streamflows, rainfall, and changing meteorological conditions. Some of the forces, such as the semi-diurnal tide, are periodic with pronounced hourly, daily, fortnightly, and annual cycles. Other forces, such as rainfall and meteorological conditions, are highly variable and are often characterized as non-periodic or chaotic. Chaotic systems are difficult to predict, as seen in long range weather forecasts that typically are more accurate for 1- or 2-day forecasts than for weekly or monthly forecasts. The May River is continuously integrating tidal dynamics with rainfall and runoff from the watershed.

Continuous monitoring of rivers and estuaries provides a record of how river and estuaries respond to changing hydrologic and tidal conditions. In addition to a large State and national network of primarily freshwater river gauges, the USGS maintains a network of coastal gauge monitoring systems such as the May River. In South Carolina, continuous monitoring networks are maintained on the Beaufort, Okatee, Ashley, Cooper, Wando, Santee, and Waccamaw Rivers along with larger systems such as Savannah and Charleston Harbor and the Intracoastal Waterway in the Grand Strand. These networks provide municipal, State, and Federal scientists and regulators necessary data for effective water resource management.

B. Methods

To gain a better understanding of how the May River responds to these various forces, USGS established a network of three continuous monitoring gauging stations in the Bluffton area (Figure II-1). The gauging stations use satellite telemetry to transmit the data on a “near” real-time basis (4-hour interval) to the USGS Office in Columbia. This network consists of a station in the upper zone of the May River near Pritchardville (station number 02176711), one located in the middle zone of the river near Bluffton (station number 02176720), and one located on the lower zone of the river below Brighton Beach (station number 02172035), which is upstream of the confluence of the May River with Calibogue Sound. Each station records water level, velocity, water temperature, specific conductance, and dissolved oxygen concentration. Streamflow at each station is computed by multiplying the cross-sectional area at the site (obtained from a water level versus area curve) by the mean velocity in the cross-section (obtained from a path velocity versus mean velocity curve). A precipitation gauge was located at the Brighton Beach gauge station. Water level, velocity, and precipitation parameters were recorded on a 15-minute interval and specific conductance, water temperature, and dissolved oxygen are recorded on a 30-minute interval. Continuous monitoring of these parameters began in May 2002 at all three gauging stations. The data from the network are published in the Annual Report Series – U.S. Geological Survey Water Resources Data – South Carolina. The establishment and maintenance of the gauging stations and the processing of the time series data follow the USGS protocol described in Wagner and others (2000).

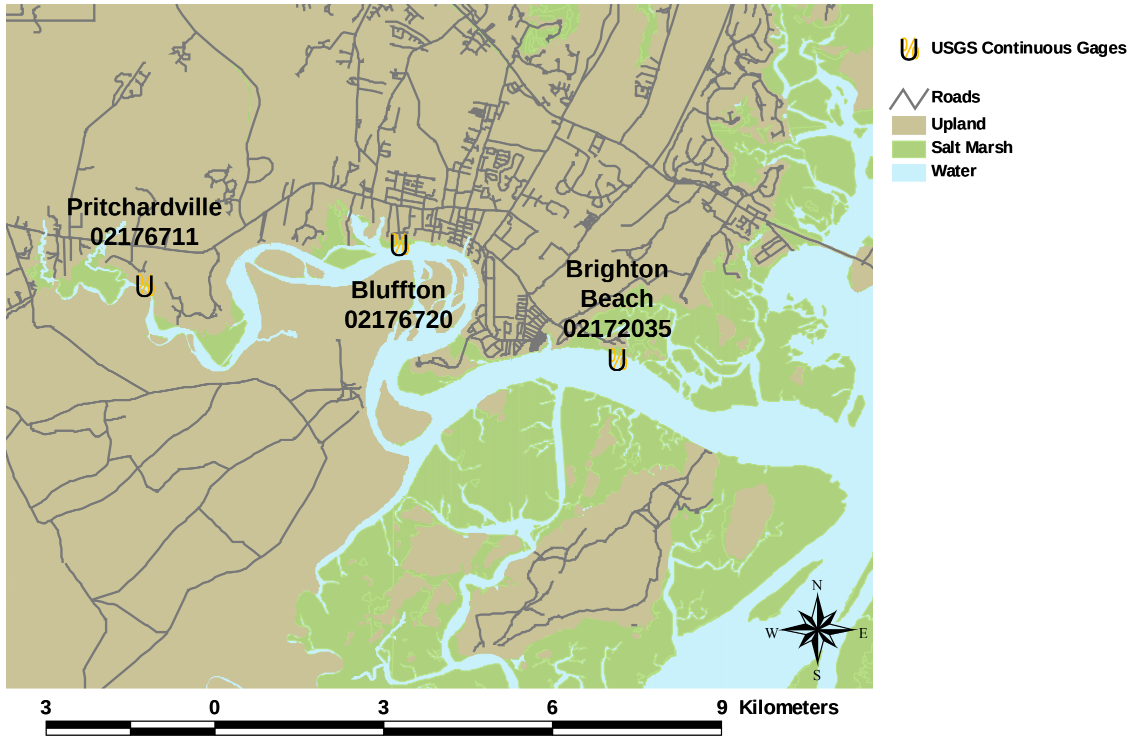


Figure II-1. The sampling sites for the May River estuarine system.

Each gauge was instrumented with a Sutron® 8210 data collection platform to store and transmit data, a Sutron® Accububble pressure sensor to measure water level, a SonTek® Argonaut SL current meter to measure velocity, and a Hydrolab multi-parameter water quality monitor to measure water temperature, specific conductance, and dissolved oxygen concentration. Hydrolab® H_2O units were initially installed and replaced with Hydrolab® Minisondes in May 2003. The precipitation gauge at the Brighton Beach station was a Sutron® tipping bucket (The use of firm, trade, and branch names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government). The water level and velocity instrumentation was serviced approximately every 4-6 weeks and the water quality instruments were serviced weekly during warmer months (May through October) and bi-weekly during the cooler months (November through April). All field notes for the servicing of the gauges and documentation for the processing of the time series data are archived at the USGS District Office in Columbia.

The continuous monitoring network on the May River provides a tremendous amount of data. Annually, the network records over 500,000 data points. The behaviors of the parameters, as recorded in the time series, result from interactions between multiple physical forces. For example, the water temperature at a fixed location is subject to daily, seasonal, and annual air temperature changes and tidal mixing of warmer and cooler waters. To understand how parameters behave over time, it is necessary to analyze changes on various time scales such as hourly, daily, and seasonally.

Some numerical methods must be applied to compute a daily value for a time series that is tidally affected. The semi-diurnal tide is dominated by the lunar cycle that is greater than the 24-hour solar cycle; thus, a 24-hour average is inappropriate to use to compute a mean daily net flow. For the analysis in this report, hourly data were digitally filtered to remove semi-diurnal and diurnal variability. The filtering method of choice is frequency domain filtering. This allows a signal, or time series, component that lies within a window of frequencies (for example, the 12.4-hour tidal cycle lies between periods of 12.0 to 13.0 hours) to be excised, analyzed, and modeled independently of other components (Press and others, 1993). The filter for removing the high frequency tidal cycle is often referred as a "low-pass" filter. Digital filtering also can diminish the effect of noise in a time series to improve the amount of useful information that it contains.

Two variables were computed from the field measurements of the physical parameters: tidal range and dissolved oxygen deficit. Tidal dynamics are a dominant force for estuarine systems and the tidal range is a significant variable for determining the lunar phase of the tide. Tidal range is calculated from water level and is defined as the water level at high tide minus the water level at low tide for each semi-diurnal tidal cycle. Periods of larger tidal ranges (spring tides) occur during full and new moons and periods of small tidal ranges (neap tides) occur during waxing and waning moons. Dissolved oxygen and water temperature are inversely related and highly correlated. Dissolved oxygen deficit is defined as the difference between the actual dissolved oxygen concentration and the saturated dissolved oxygen concentration. The computed variable, dissolved oxygen deficit is derived using an algorithm that assumes a constant barometric pressure (USGS, 1981).

C. Results and Discussion

For a complex tidal system with conditions that change on an hourly, daily, seasonal, and annual basis, it can be difficult to generalize system behavior. The response of the system to events, such a rainfall, is often dependent on the phase of the tide, time of year, and the magnitude of the event. The following discussion is a preliminary analysis of the data from the May River continuous monitoring network to begin to understand the interaction between the tidal dynamics and rainfall-runoff watershed dynamics. The data used for the discussion is the 16-month period from June 2002 to September 2003. The water level and streamflow data for the Brighton Beach and Bluffton stations are still in the review process and are considered provisional and subject to change.

Water Level and Streamflow Data

Tides enter the May River through the confluence with Calibogue Sound and the Intracoastal Waterway and through Bull Creek, a tidal creek connected to the New River and Calibogue Sound. Generally, the time series of the physical properties measured at the gauging station (water level, water temperature, specific conductance, and dissolved oxygen concentration) show the propagation of the tide as it progresses from Calibogue Sound to the upper zones of the river. Figure II-2 shows the water level at the three stations on the May River for January 2003 (water level is shown as gauge height, which is the elevation to an arbitrary datum, not necessarily mean sea-level). There is about an hour lag in the tide from the Brighton Beach gauge (lower gauge, station 02176735) to the Pritchardville gauge (upper gauge, station 02176711). The 14-day semi-diurnal tidal cycle is also apparent in Figure II-2. The neap tidal period, characterized by relatively small amplitude in tidal range, occurs around January 11 and

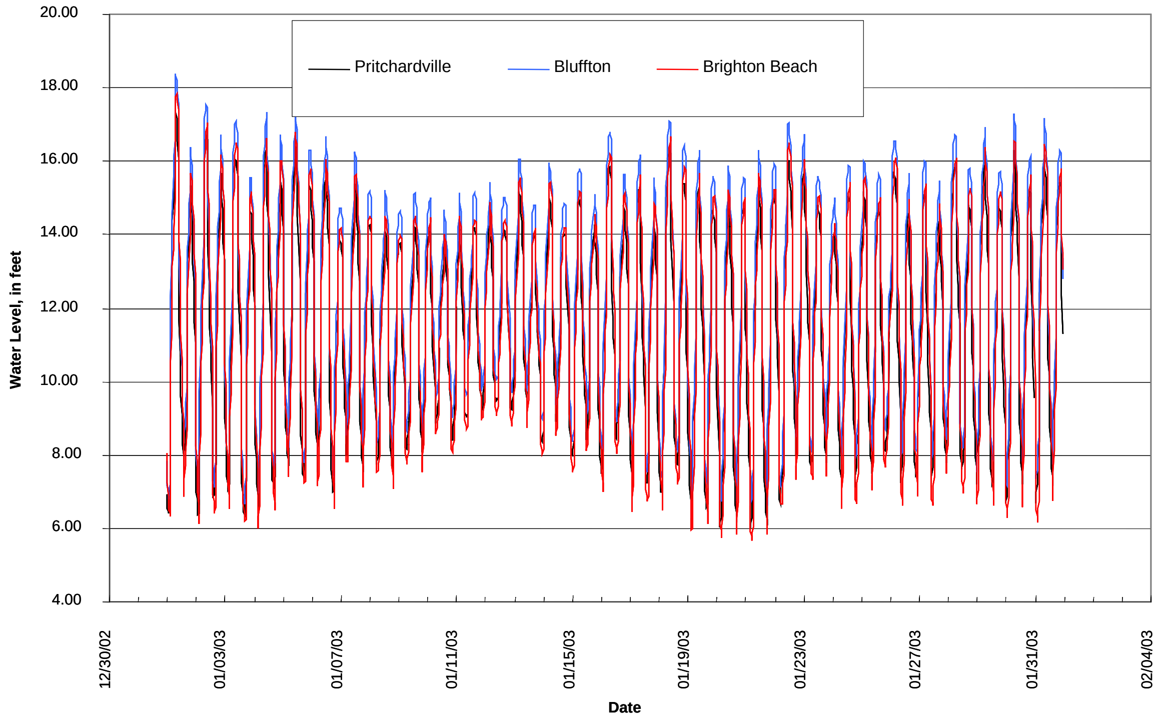


Figure II-2. Hourly water levels for three stations on the May River for January 2003.

25, and the spring tidal period, characterized by larger amplitude in tidal range, occurs around January 4 and 20.

In tidal sloughs like the May River where there is little freshwater inflow, there is often a small increase in the tidal range as the tidal wave propagates upstream. The maximum tidal range for the period shown in Figure II-2 is 11.17 feet for the Brighton Beach gauge, 11.61 feet for the Bluffton gauge, and 11.74 feet for the Pritchardville gauge. The tidal range for the Pritchardville gauge is shown in Figure II-3 for the period June 2002 through September 2003 and clearly shows the longer-term cyclic patterns in the tidal ranges. For example, a high spring tide range (greater than 9 feet) is followed by a low spring tide range (less than 9 feet). A similar pattern is apparent in the neap tides where a low neap tide range (less than 7 feet) is followed by a higher tidal range (greater than 7 feet). Also apparent are semi-annual cycles of minimum and maximum tidal ranges. Tidal ranges greater than 11 ft occur in the fall (October and November) and spring (April) and the smaller tidal ranges, less than 10 feet, occur in the winter (February).

With an 11-foot tidal range and wide channel geometry towards the confluence with Calibogue Sound, the May River experiences large tidal streamflows of greater than 50,000 cubic feet per second (ft^3/s). Figure II-4 shows the hourly streamflows for the Brighton Beach gauge for the first 9 months of 2003. The average positive streamflows (ebb flows or out-going tides) and negative streamflows (flood flows or in-coming tides) are 54,900 ft^3/s and -48,600 ft^3/s, respectively. The maximum ebb and flood tides are 151,000 ft^3/s and -128,000 ft^3/s, respectively.

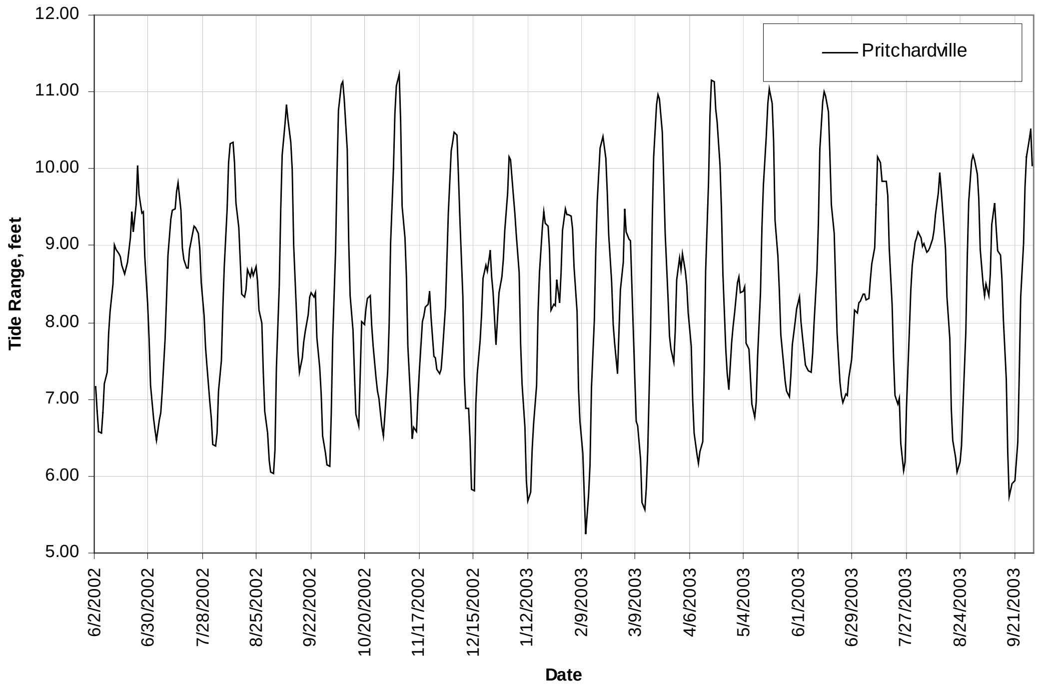


Figure II-3. Daily tidal range for the May River near Pritchardville for the period from June 2002 to September 2003.

As the channel geometry decreases upstream, the tidal streamflows at the Bluffton gauge are approximately a fourth of the streamflows at Brighton Beach. Figure II-4 shows the hourly streamflows for the Bluffton station for the period January 2003 through September 2003. The average ebb and flood tides decreased to 14,600 ft^3/s and -12,300 ft^3/s, respectively. The maximum ebb and flood tides are 40,300 ft^3/s and -34,800 ft^3/s, respectively.

The channel geometry at Pritchardville is significantly smaller than at Brighton Beach and the streamflows are approximately a twenty-fifth of the streamflow at Brighton Beach. Hourly streamflow for Pritchardville from October 2002 to September 2003 (USGS 2003 water year) is shown in Figure II-4. The average ebb and flood tides are 1,950 ft^3/s and -1,840 ft^3/s, respectively. The maximum ebb and flood tides are 5,760 and -7,050 ft^3/s, respectively. Filtering the tidal streamflow data and removing the tidal variability results in the net streamflow of the system and is shown in Figure II-4. The net flow in the system is very low (near zero) with periods of small net upstream flows (negative flows) and net downstream flows (positive flows) towards Brighton Beach. For the 2003 water year, the annual net streamflow is 20 ft^3/s.

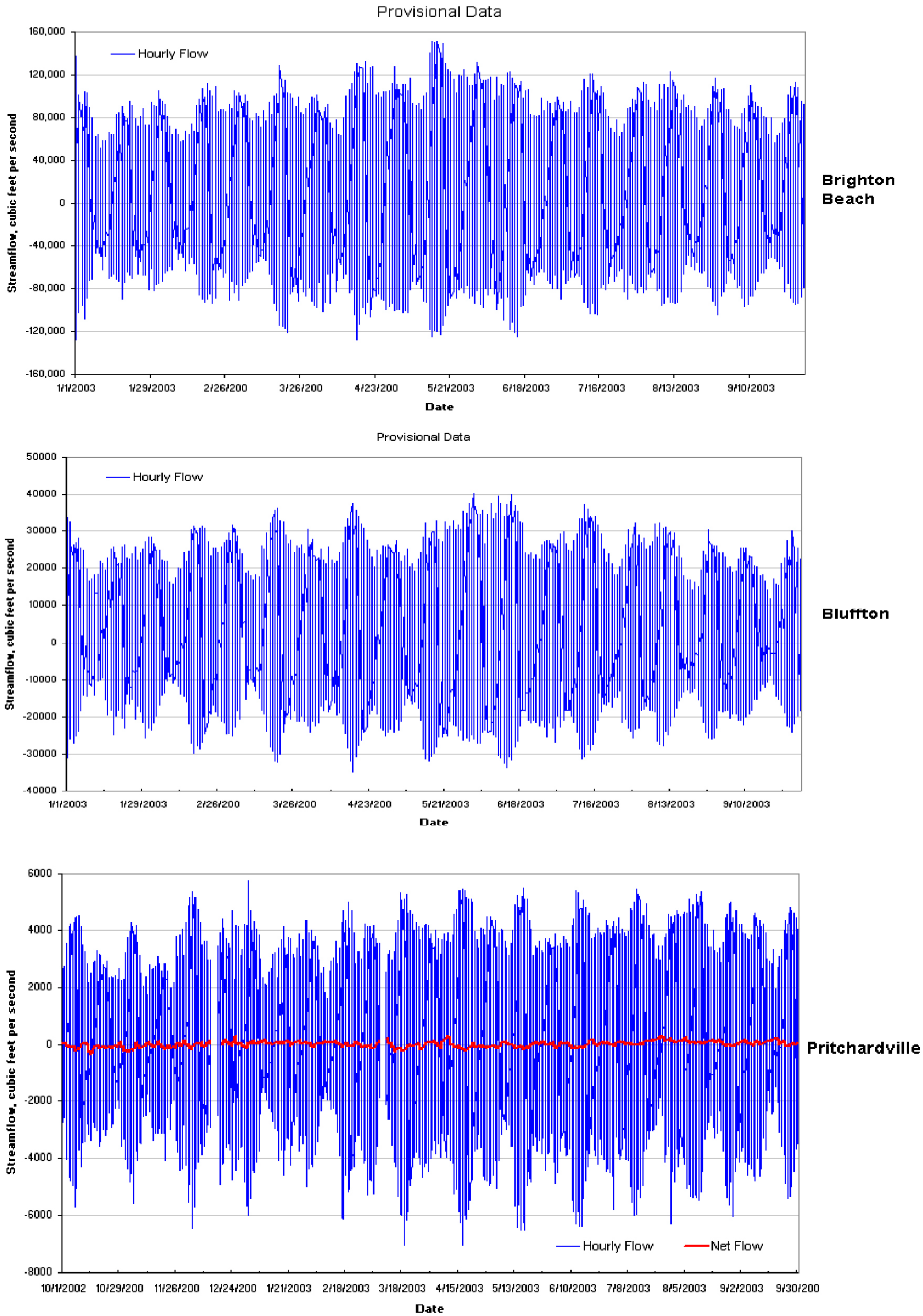


Figure II-4. Streamflows for the May River for the period October 2002 to September 2003.

Precipitation and Water Quality Data

The National Weather Service (NWS) maintained a precipitation gauge on Hilton Head Island from 1953 to 1998 (SCDNR, 2004). The mean annual rainfall for the period of record is 52 inches per year with the greatest monthly rainfall occurring in June, July, August and September. The total monthly precipitation for the gauge at the Brighton Beach station for the period June 2002 through September 2003 is shown in Figure II-5, along with the average monthly total for the NWS Hilton Head gauge and departures from normal (the differences between the monthly sums and the average monthly sums). Positive departures from normal indicate greater than normal rainfall whereas negative departures indicate less than normal rainfall.

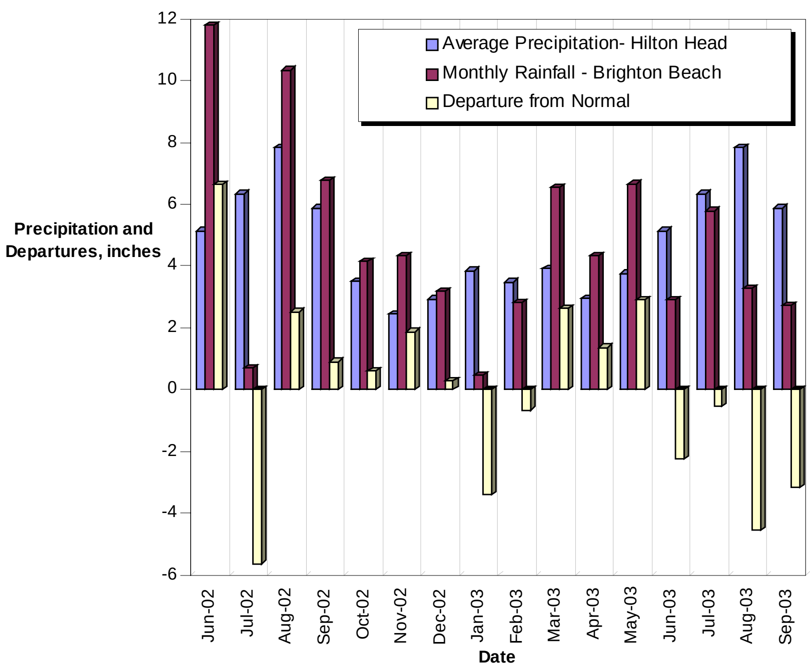


Figure II-5. Total monthly precipitation for Brighton Beach for the period June 2002 to September 2003 and average monthly precipitation at the Hilton Head gauge for the period 1953 to 1998, and departures from normal.

The continuous monitoring of the May River began at the end of an extreme drought in South Carolina that occurred from the period of the last El Niño event in the spring of 1998 until the increased rainfall during the late summer of 2002. Of the sixteen months shown in Figure II-5, seven months had below average rainfall conditions. The total rainfall for the period October 2002 to September 2003 was 44.5 inches, or 7.5 inches below the mean annual precipitation (based on 45 years of record) measured at the NWS Hilton Head gauge.

Watersheds are often evaluated by determining the total quantity of water that is discharged, or “runs off” a drainage basin in a year. Typically, annual runoff is reported as streamflow per unit area of drainage area, usually as cubic feet per second per square mile ($ft^3/s/mi^2$). During years of above normal rainfall, the annual runoff from a drainage basin, or watershed yield, will be greater than years of below normal rainfall. The drainage area for the Pritchardville gauge is 14.1 square miles. With an annual net streamflow of 20 ft^3/s, the annual runoff, or watershed yield, is 1.4 $ft^3/s/mi^2$. As compared to inland watersheds, particularly those that are not tidally influenced, the annual runoff for the May River at Pritchardville is high. Watershed yield values for inland freshwater drainage basins in South Carolina are typically in the 0.3 to 0.8-$ft^3/s/mi^2$ range. The higher annual runoff number indicates that more rainfall runs off the May River watershed than most other gauged freshwater watersheds.

There could be many reasons for the high annual runoff numbers for the May River watershed. The long and narrow shape of the drainage basin minimizes distances water must travel to reach the receiving stream, which would lessen potential losses to evapotranspiration and to groundwater. The sandy soils of the Bluffton area may allow for more efficient transport of water through shallow groundwater to the May River. Depending on the geology of the area, water losses to deeper aquifers may be limited and more water flows to the receiving stream. The continuous measurement of streamflow in small tidal sloughs, like the May River at Pritchardville, has only been possible with recent developments in acoustic velocity-meter technology. As more data are collected and analyzed at similar tidal systems, there will be a better understanding of the watershed dynamics for tidal sloughs and how they compare to dynamics of inland watersheds.

As mentioned previously, meteorological conditions, such as rainfall, do not have the periodicity that is exhibited in the semi-diurnal tidal time-series of the water level and streamflow and are often characterized as chaotic systems. Rainfall plays a significant role in the dynamics of an estuarine system. Rainfall and the subsequent overland flow from the watershed is a principle transport mechanism for moving sediment and other materials to the receiving stream. The effect of rainfall events is often seen in the dilution of the salinity of the system (as measured by specific conductance) and a decrease in dissolved oxygen (or an increase in dissolved oxygen deficit). The specific conductance values for the lower zones of the May River, with its close proximity to the ocean through Calibogue Sound, are similar to ocean values during periods of limited rainfall. Figure II-6 shows hourly specific conductance values for the three May River stations for the period of January 2003 through March 2003. The majority of Brighton Beach values are greater than 40,000 us/cm. Specific conductance values decreased upstream with the dilution due to freshwater inflows.

The time series of hourly specific conductance values clearly show tidal periodicity of specific conductance (Figure II-6). Removing the tidal variability and plotting the daily mean specific conductance values with rainfall clearly shows the non-tidal impact of rainfall on the salinity dynamics of the system (Figure II-7). During periods of limited rainfall, the high specific conductance values of the lower zones of the May River propagate upstream and the specific conductance values upstream increase. This also indicates the net movement of water was moving upstream towards Pritchardville. After rainfall events, specific conductance values, especially at the Pritchardville gauge, show a dynamic response to the freshwater inflow. Prior to the 4.8-inch rainfall of June 19, 2002, the specific conductance values for the three stations were greater than 50,000 µs/cm (Figure II-7). After the rain event, specific conductance values for the Pritchardville gauge decreased to 35,000 µs/cm and did not recover to the pre-event

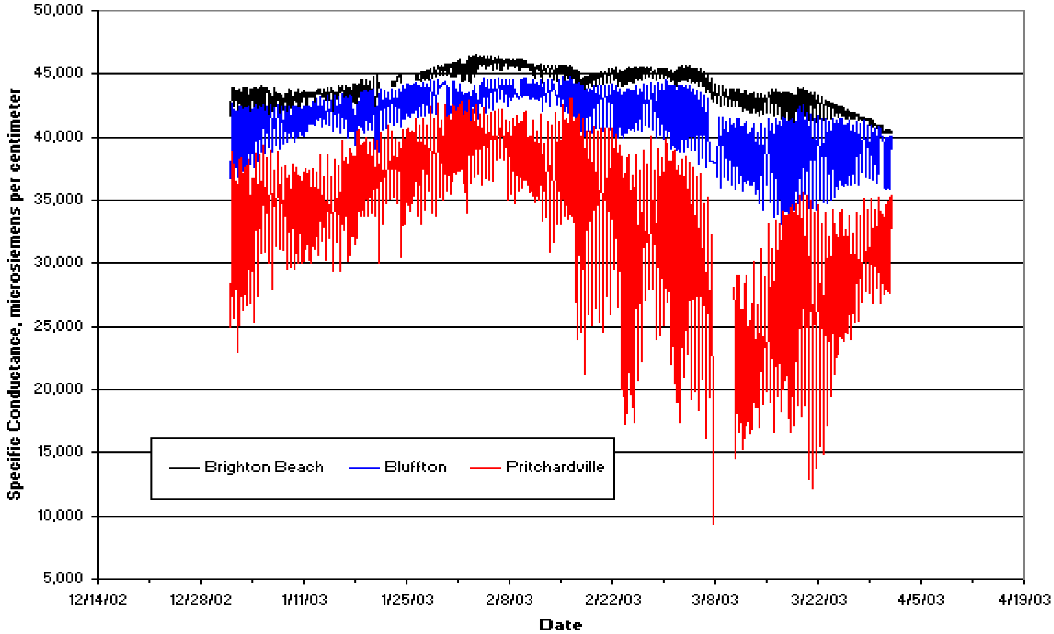


Figure II-6. Hourly specific conductance for three stations on the May River for the period January to March 2003.

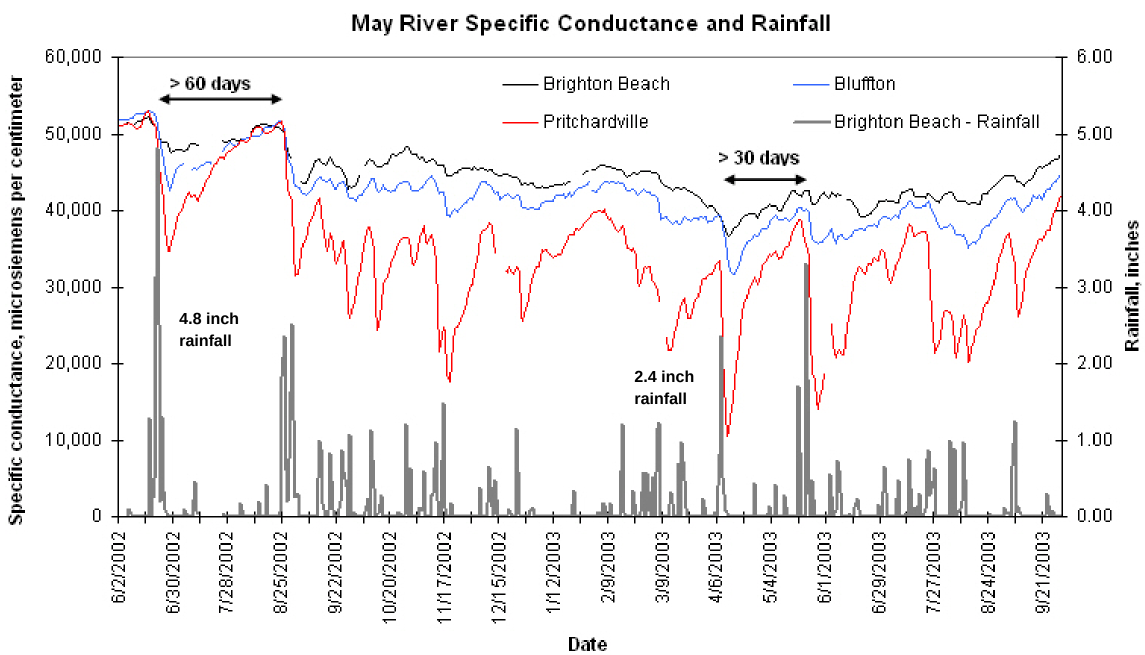


Figure II-7. Daily mean specific conductance for three stations and rainfall for one station on the May River for the period June 2002 to September 2003.

specific conductance levels until August 25, indicating a long retention time in the system (greater than 60 days) and limited flushing. The 2.4 inch rain event on April 7, 2003 decreased the specific conductance values at Pritchardville by greater than 20,000 µs/cm and the system took approximate 30 days to recover to the pre-event specific conductance levels.

The dynamic behavior of water temperature at all of the stations is similar to specific conductance, as shown in Figure II-8. The upstream-most gauge, Pritchardville, recorded the highest and lowest temperatures in the summer and winter, respectively. The Pritchardville gauge is in a small channel and responds faster to changes in air temperature than the downstream gauges in larger channels that are buffered by the thermal mass of larger volumes of water, such as Calibogue Sound and the Atlantic Ocean. The temperatures in the river reaches 20 ^{o}C in late March and early April, increases to 30 ^{o}C by June, and do not fall below 20 ^{o}C until late October.

Similar to many coastal systems, the May River is naturally low in dissolved oxygen. The State water quality standard is a minimum daily mean of 5.0 mg/L or an instantaneous minimum of 4.0 mg/L. Figure II-8 shows the daily dissolved oxygen concentrations for the three stations on the May River. During the summer months, the minimum dissolved oxygen concentration was less than 4.0 mg/L for extended periods, and was generally higher in the lower zones of the river than in the upper zones of the system. Water temperature and dissolved oxygen are inversely related and highly correlated. As water temperature increased to greater than 25 ^{o}C, dissolved oxygen concentrations decreased below the State water quality standard of 5 mg/L.

Time series of dissolved oxygen deficit show the dynamic behavior of dissolved oxygen without the temperature effect. Dissolved oxygen deficit is a measure of the difference between the actual dissolved oxygen concentrations and dissolved oxygen concentration for saturated conditions, and effectively “normalizes” dissolved oxygen to water temperature. Lower values of dissolved oxygen deficit indicate water of higher percent saturation, whereas higher dissolved oxygen deficit values indicate water of lower percent saturation. Figure II-8 shows that dissolved oxygen deficit concentrations are generally higher for the upstream gauge near Pritchardville than the downstream gauge at Brighton Beach.

For the three stations on the May River, cumulative percentages of dissolved oxygen deficit were computed from the time series (Figure II-9). The figure shows that the Brighton Beach gauge and the Bluffton gauge have similar frequencies of dissolved oxygen deficit values and the Pritchardville gauge has higher dissolved oxygen deficit values for the period June 2002 through September 2003. Higher dissolved oxygen deficit values indicate greater stress on dissolved oxygen from point or non-point sources.

Two contributors to dissolved oxygen deficit are (1) non-point source loading of oxygen consuming constituents from the landscape after rainfall events and (2) the flooding and draining of tidal marshes and the export of organic material to the receiving stream. Loadings from the tidal marshes are considered natural non-point source loading. Loading from rainfall on the watershed is a combination of natural loading from unaltered land use and manmade or anthropogenic loading from altered land use.

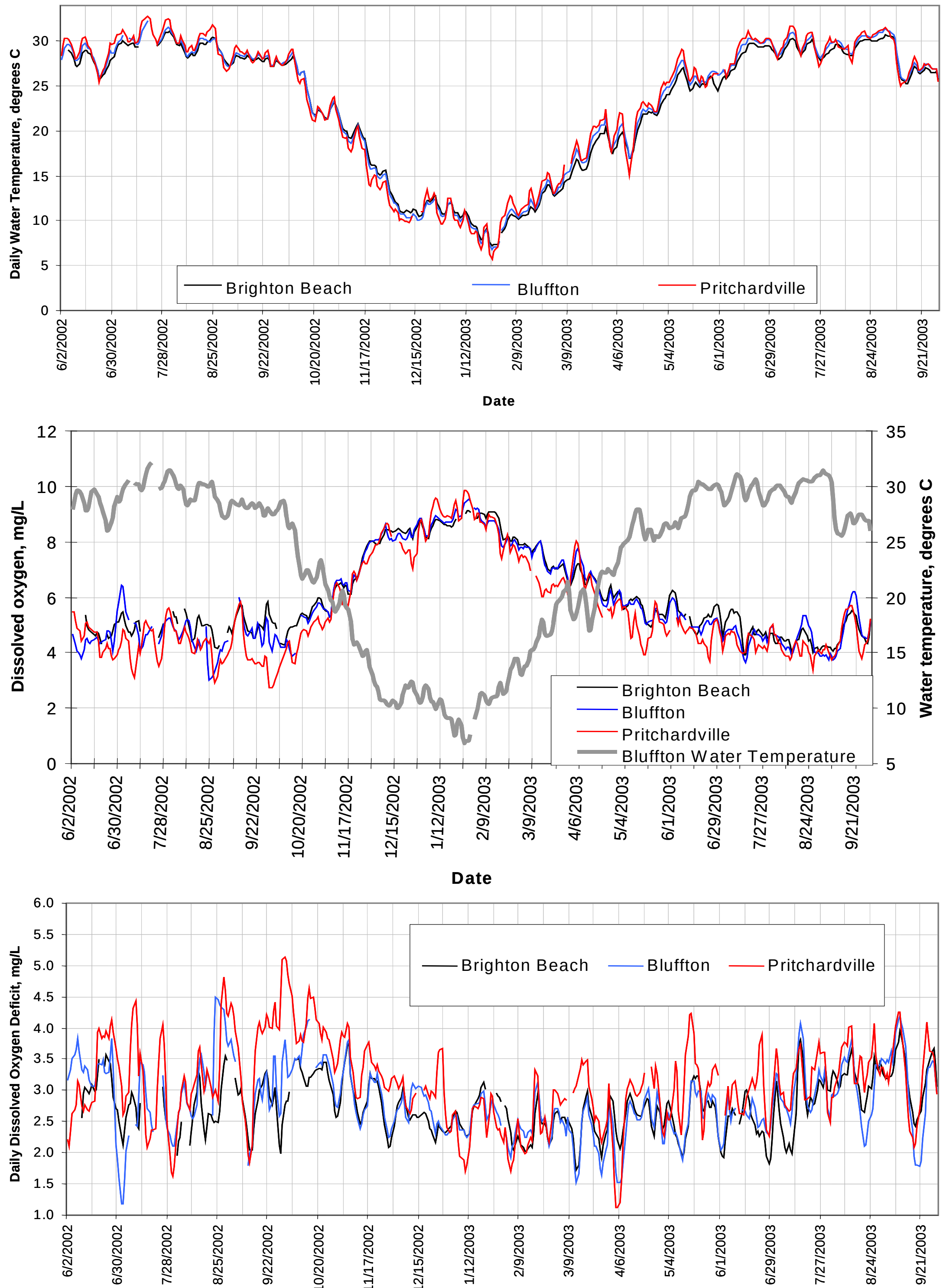


Figure II-8. Daily water temperature, dissolved oxygen, and dissolved oxygen deficit for three stations on the May River for the period from June 2002 to September 2003.

The daily dissolved oxygen deficit and tidal range data from the Pritchardville gauge and rainfall data from the Brighton Beach gauge are shown in Figure II-10. There appears to be a periodic response of the dissolved oxygen deficit due to tide range. During spring tide, with the greatest inundation of the tidal marshes and subsequent transport of material, there was an increase in dissolved oxygen deficit. During the low rainfall period, from June 22, 2002 to August 30, 2002, the spikes in dissolved oxygen deficit clearly occurred during the spring tides. During a wetter period from August 30 to December 24, 2002, there was an overall increase in dissolved oxygen deficit and the spikes in dissolved oxygen deficit do not necessarily coincide with spring tides. During the dry period of January and February 2003, the spike in dissolved oxygen deficit coincides with the spring tides. Spikes in dissolved oxygen deficit appear to be attributable to both non-point source loading of tidal marshes and rainfall/runoff impact.

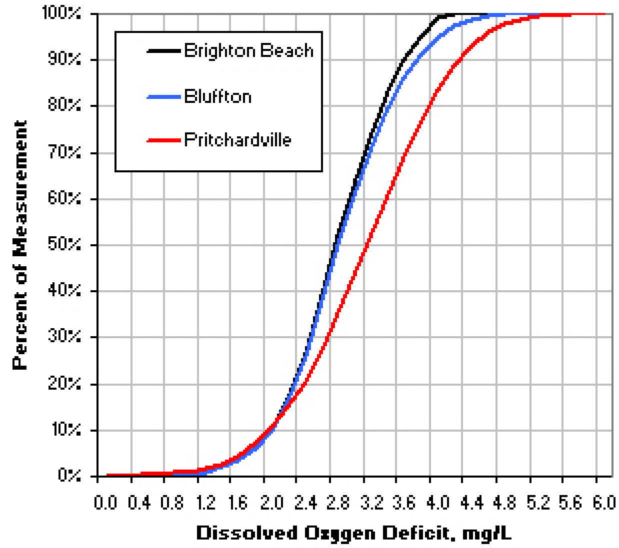


Figure II-9. Cumulative percent of dissolved oxygen deficit measurement for three stations on the May River for the period January to March 2003.

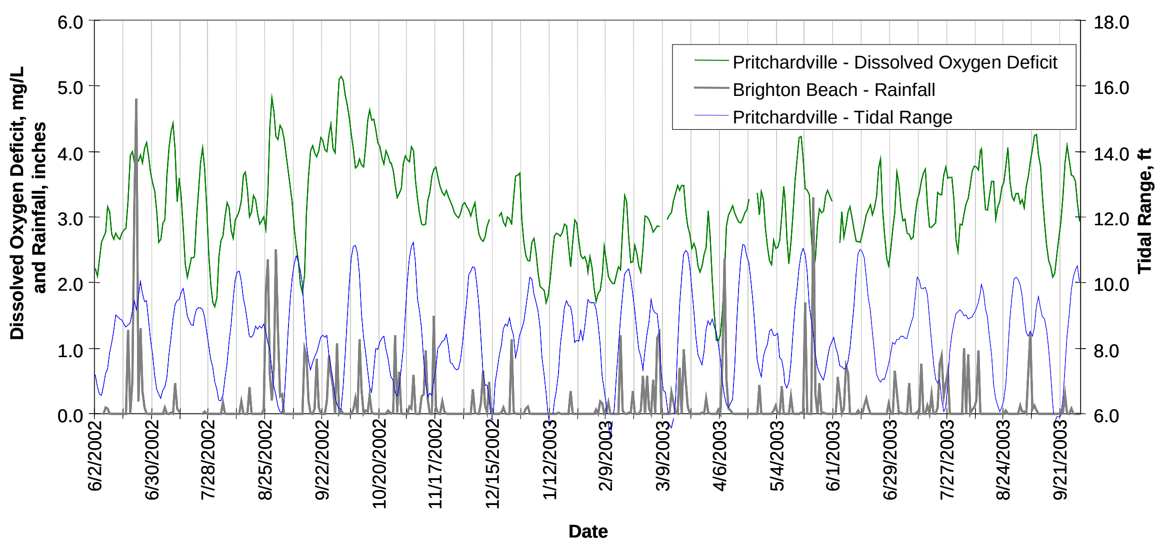


Figure II-10. Dissolved oxygen deficit and tidal range for Pritchardville and rainfall for Brighton Beach for the period June 2002 to September 2003.

III. TIDAL CREEK AND OPEN WATER HABITATS

A. Introduction

Estuarine areas, especially tidal creeks, serve as critical nursery habitat for many fish and crustacean species. Because of their low dilution capacity, variable water quality, and high abundance of valued biota, tidal creeks are vulnerable to impacts from watershed development. Holland and others (2004) found that the density of the human population and impervious cover within headwater tidal creek watersheds were related to the environmental quality. They found that an impervious cover of 10-20 percent or greater in the surrounding watershed alters the physical and chemical environment of tidal creeks. Increased freshwater flow into the creeks causes changes in salinity, more input of contaminants, altered sediment characteristics, and increased bacterial loadings. At 20 to 30 percent impervious cover, Holland and others (2004) detected reduced shrimp abundance, fewer stress-sensitive benthic species, altered food webs, and closures of shellfish beds. These results are consistent with research performed in the headwaters of freshwater streams (Schueler 1994; Arnold and Gibbons 1996).

Larger tidal creek drainage areas are much more extensive than the headwater portions of these creeks. Large tidal creeks are also vulnerable to adverse effects from upland development because they receive runoff from both the headwater portions of the creek, as well as runoff from upland and marsh habitats adjacent to the creeks. Because of their proximity to these sources of runoff, they are considered to be more vulnerable than the larger open water bodies that they drain into but less vulnerable than the headwaters. Large tidal creeks often exhibit greater levels of natural stress compared to the larger open water bodies, which have greater dilution properties (Van Dolah and others, 2000; 2002).

Open water habitats located in the mainstem of the May River represent the majority of aquatic estuarine habitat in this drainage system. This area receives drainage from both tidal creeks and direct runoff from adjacent upland habitats. Because these areas include large water volumes and good tidal flow, they tend to exhibit fewer adverse effects from upland development compared to the peripheral creeks that drain into the May River. However, these areas are also important habitat for marine and estuarine species and generally tend to support later life stages of these organisms.

B. Methods

Site Locations

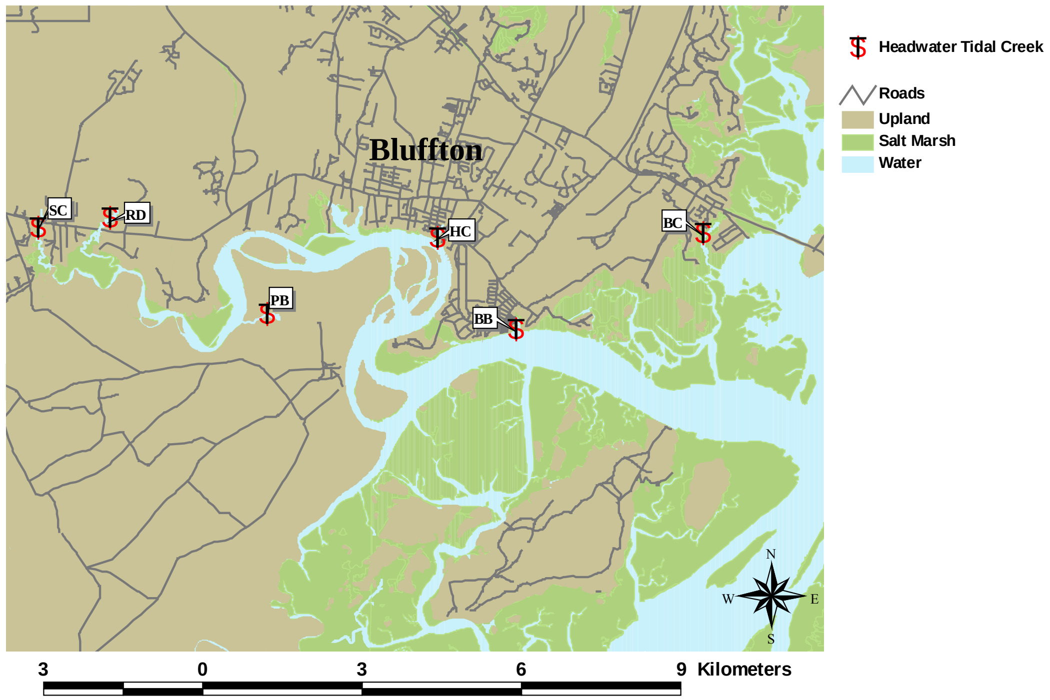
The six headwater creeks selected for this study represented the length of the May River, from near its headwaters to the mouth, where it adjoins Calibogue Sound. Three creeks were sampled from the Upper Zone (Stony, Rose Dhu, and Palmetto Bluff creeks); two from the Middle Zone (unnamed creeks near Heyward Cove and Brighton Beach referred to as Heyward Cove and Brighton Beach creeks in this report), and one from the Lower Zone (Bass Creek) (Figure III-1; Appendix III-1). For the purposes of this study, each creek was sampled within 600 m of the upper boundary. In cases where the creek was less than 600 m in length, the creek was sampled within 100 m of its mouth. The upper boundary of each creek was defined as the point where the water depth was ~1 m at mean high tide. Sites were predominately sampled on falling tide within 3 hours of low tide.

When applicable, comparisons of the data obtained from these headwater tidal creeks were compared to data collected from tidal creeks throughout South Carolina for studies conducted by the Tidal Creek Project (TCP). The comparative data were primarily obtained from the Charleston Harbor Estuary, the majority of which were from the 1994-1995 time period (Holland and others, 1996; Lerberg and others, 2000; Holland and others, 2004). A few parameters sampled for this study were not collected during the 1994 study, in which case other TCP data were used for comparison or no comparison was made and such cases are noted in the text.

The ten sites representing larger tidal creek and open water habitats were located throughout the length of the May River (Figure III-1; Appendix III-1). Samples were collected within 3 hours of low tide. One station in the upper portion of the May River (U-01) was designated as large tidal creek habitat because the river was less than 100 m from marsh bank to marsh bank. The other two large tidal creek stations (L-03, L-04) were located in the Lower Zone of the May River in different branches of Bass Creek. Two of the open water sites were located in the Upper Zone (U-02, U-03), three in the Middle Zone (M-01, M-02, M-03), and two in the Lower Zone (L-01, L02). All large tidal creek and open water sites were selected at random within each zone from a larger array of possible sites using the USEPA Fields tool in ArcView®.

Three large tidal creek and seven open water sites sampled by the South Carolina Estuarine and Coastal Assessment Program (SCECAP) were used for comparison with the large tidal creek and open water sites sampled in the May River (Figure III-2). All SCECAP stations were sampled in the summer of 2002 and were located in the southern portion of the state's waters (Beaufort, Jasper, and Colleton counties) to avoid any differences that might occur due to natural variation in condition with latitude. These sites had little or no upland development nearby and bottom sediments had no elevated levels of contaminants. These sites will be referred to as 2002 SCECAP sites. All SCECAP sites were also selected using a random probability based sampling design required for that program to avoid any bias in station location (Van Dolah and others, 2002).

A.



B.

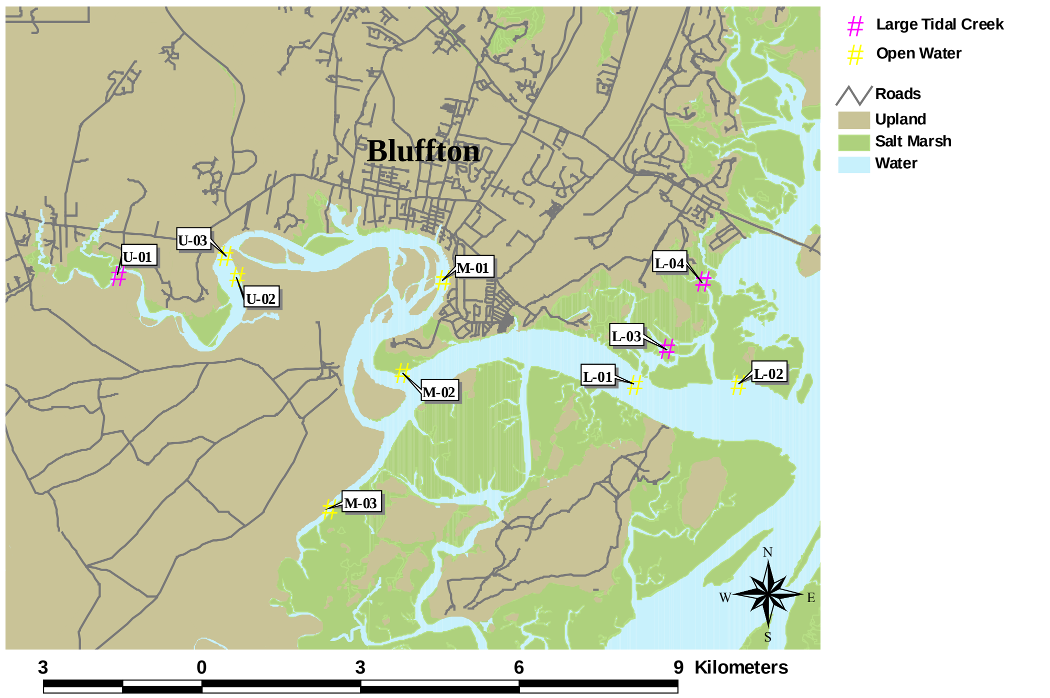


Figure III-1. Location of the six headwater tidal creek sites (A.) and ten large tidal creek and open water sites (B.) sampled in the May River. Location of continuous record gauging stations are shown in Section II of this report.

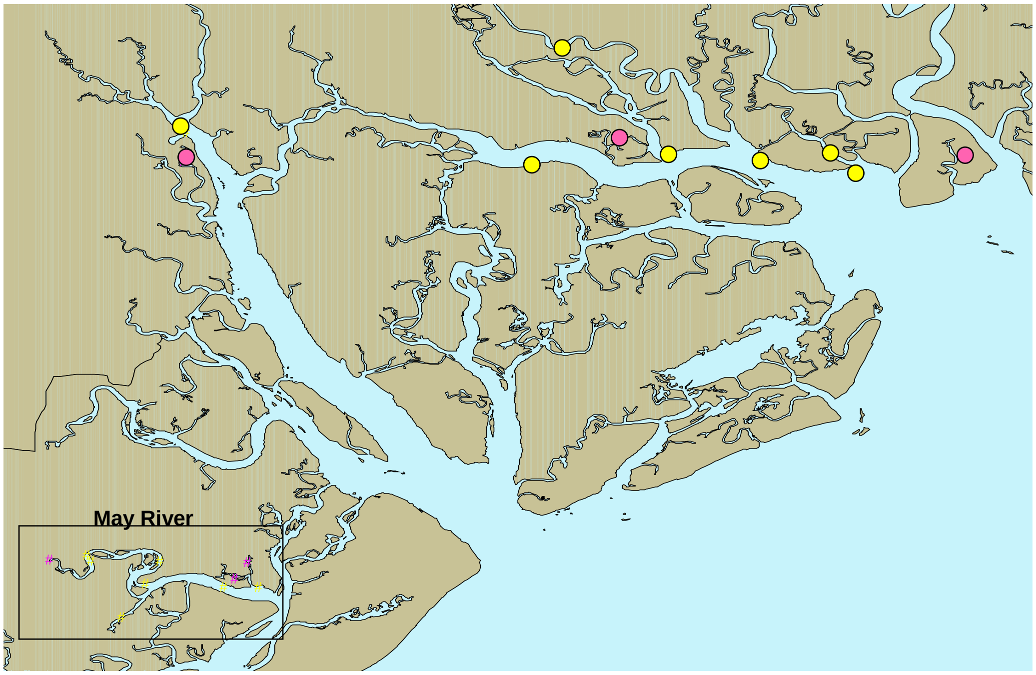


Figure III-2. Location of the 2002 SCECAP sites sampled during the summer of 2002 for comparison with the May River sites (also shown in lower left portion of figure). All SCECAP sites were located in relatively pristine areas of Beaufort, Colleton, and Jasper counties.

Land Use Patterns

The watersheds drained by each headwater tidal creek were delineated on 1:24,000 United States Geological Survey topographic maps using elevation contour lines. The outline of each watershed was digitized in ArcView® (Version 3.2) on digital topographic maps. Using 1999 high-resolution 1:4,800 Digital Ortho-Quarter Quad (DOQQ) National Aerial Photographic Program (NAPP) photographs, the area within each drainage basin was classified into categories based on a modified Anderson Land Use Classification system (Anderson and others, 1976). These categories included agricultural, barren, forest, golf course, suburban/urban, freshwater ponds, and creek/salt marsh. The total area of each land use type was calculated.

In addition, ArcView® was used to determine the percentage of impervious cover by point sampling using a triangular point grid. The number of grid points that fell on roadways, parking lots, roofs, or other impervious surfaces were divided by the total number of points in the watershed, excluding the points that fell in the creek or on the marsh, times 100.

The estimates of land use and impervious cover in the May River headwater tidal creek watersheds are conservative approximations. Aerial photographs from 1999 were used to calculate these values and some additional developments in the three years between 1999 and this study are likely. In addition, the dense tree cover of the Bluffton area may have reduced the ability to determine impervious surfaces from aerial photographs.

Finally, human population density within each watershed was calculated in ArcView® using 2000 United States Census population data at the block level provided by TigerLine (U.S. Census Bureau, 2002). Population blocks did not coincide with watershed boundary lines, so in cases where partial blocks fell within a drainage basin, only the appropriate proportion of the population contained within those blocks was included in the watershed population total. To determine population density (i.e., number of individuals/hectare), the total number of people within a watershed was divided by the size of the watershed.

The six headwater tidal creek drainage basins were classified as either forested or suburban based upon the (a) degree of urban land use, (b) percent impervious cover, and (c) human population density in the watershed (Lerberg and others, 2000; Gawle, 2002). The specific criteria used are outlined in Table III-1. All land use analyses were restricted to the headwater watershed area.

These land use data were integrated with ecological data to produce an ecosystem-level assessment of the current status of headwater tidal creek habitats of the May River and the influence of existing land use on that status.

Table III-1. Land use criteria developed by Lerberg and others (2000) and Gawle (2002) that were used to classify headwater tidal creek watersheds.

Forested	Land use	< 30% suburban/urban land cover
	Impervious cover	< 10% impervious cover
	Population density	< 5 individuals/hectare
Suburban	Land use	> 30% but < 70% suburban/urban land cover
	Impervious cover	> 10% but < 50% impervious cover
	Population density	> 5 but < 20 individuals/hectare
Urban	Land use	> 70% suburban/urban land cover
	Impervious cover	> 50% impervious cover
	Population density	> 20 individuals/hectare
Industrial	Land use	> 45% suburban/urban land cover with industrial facilities
	Impervious cover	> 50% impervious cover

Overall Sampling Design

Three locations (or groups of locations) were selected for sampling within each headwater tidal creek and are outlined below.

1. 500 m (or 5/6th of the creek's length) downstream from the upper extent of the creek: All water quality and water chemistry samples were collected at this site including point water samples collected by USGS and the semi-continuous water quality instruments deployed by SCDNR. In addition, this point was the starting location for the fish and crustacean sampling.
2. Six randomly selected sites along the length of the creek: Prior to field collections, six sites throughout each headwater tidal creek were randomly chosen. These six sites within each creek were sampled for benthic community and sediment characteristics.
3. Primary site: One of the six randomly selected sites was designated as the primary site where samples were collected for sediment contaminant analysis and

toxicity assays, as well as an analysis of grain size that differentiated between silts and clays.

All sites were mapped in ArcView® to obtain latitude and longitude coordinates which were navigated to for sampling using a hand-held GPS unit. All point water quality and sediment samples were collected during ebbing tides.

The large tidal creeks and open water sites were sampled using methods similar to those described for the SCECAP study (Van Dolah and others, 2002). These sites were sampled by boat within 3 hours of low tide and all stations were located using GPS. Water quality, sediment quality, and benthic community samples were collected within 50 m of the designated site location. Finfish and crustacean trawl samples were collected within 250 m of the designated tidal creek sites and within 500 m of the open water sites location since open water trawls covered a greater distance of bottom habitat (see later section).

Water Quality and Chemistry

Seasonal Point Sampling

Field personnel with the USGS collected samples during the spring (late May, early June), summer (late July, August) and fall (late October) of 2002, and during the winter (early March) of 2003. The summer 2002 sample collections were spread over two months to obtain data contemporaneous with ongoing SCDNR investigations. All headwater tidal creek samples were collected on the falling tide, within the time period of three hours before low tide. Large tidal creek and open water samples were also generally collected during this same time period. At the time of sample collection, dissolved oxygen (in mg/L and percent saturation), pH, conductivity, salinity, and temperature were measured at a depth of 0.3 m at all sites, using a HydroLab® water quality datasonde or minisonde. The sonde was calibrated prior to use according to the manufacturer's and USGS protocols (Wagner and others, 2000).

Four field parameters (temperature, pH, salinity, and dissolved oxygen) were evaluated in this study using two methods: semi-continuous sampling over a five-day period during the summer months and seasonal point sampling from May 2002 to March 2003. The seasonal sampling also included determination of nutrients, biochemical oxygen demand, total organic carbon, fecal coliform concentrations, and phytoplankton concentrations.

Water samples from 5 of the headwater tidal creek sites were collected at the 5/6th point of each reach by wading into the creek. Samples from Palmetto Bluff Creek were collected by boat because of water depth and lack of land access. Large tidal creek and open water samples were collected by boat, which was anchored at each station. Sample collection and processing were modified from the USGS protocols (Wilde and others 1999; Wilde and others 2002) to ensure consistency with prior and ongoing SCDNR and SCDHEC sampling for SCECAP and TCP studies.

Grab samples were collected for the analysis of total (unfiltered or raw sample that includes particulate and dissolved forms) constituents in the water column at the centroid of flow by inverting dedicated sample bottles, immersing them to a depth of 0.3 m then turning them upright to fill. They were capped while still immersed. For the analysis of dissolved constituents, a grab sample was collected in a pre-cleaned 3-liter polypropylene bottle, placed on ice, and transported to a field laboratory for filtering.

Fecal coliform bacteria samples were collected in pre-sterilized Whirlpaks® using sterile techniques as a grab sample. Fecal coliform samples were placed on ice and held at 4˚C for transport to the field laboratory for processing within 6 hours of collection.

Alkalinity and fecal coliform concentrations were analyzed in the field by USGS district personnel. Alkalinity was measured using the Gran Function Plot method on a filtered sample (Rounds and Wilde 2001). The Gran Function method was applied because the samples were influenced by sea water and/or relatively high organic acid concentrations. Alkalinity was measured as acid neutralizing capacity. Sample processing and analysis for fecal coliform bacteria concentrations employed the membrane filtration procedure (Myers and Wilde 2003). For each fecal coliform sample, five to six sample volumes (and dilutions) were used to inoculate the culture media (m-FC) in sterile Petri dishes to ensure ideal colony counts. The inoculated dishes were placed in a preheated incubator at 44.5 ˚C for 22 to 24 hours.

Samples for laboratory analyses were processed and preserved in designated bottles in accordance with USGS and USEPA procedures (Wilde and others 2002) in the field. Sample processing sites included the Waddell Mariculture Center field laboratory near Bluffton, South Carolina. The bottles were double-bagged, placed on ice, and maintained at 4 ^{o}C during shipment. The samples were shipped for laboratory analyses to USGS laboratories: Ocala Water Quality and Research Laboratory (OWQRL) in Ocala, Florida, and the National Water Quality Laboratory (NWQL), in Denver, Colorado. Descriptions of laboratory methods, laboratory reporting levels (LRLs), and parameters are provided in Appendix III-2. Specific analytes determined by OWQRL included:

1. salinity-related measures of salinity, chloride, and specific conductance;
2. sediment-related measures of turbidity, total suspended solids (TSS), and total volatile solids (TVS – an estimate of the organic content of the solid residue);
3. nutrient-related measures of dissolved ammonia (NH_3), total organic nitrogen plus ammonia (or Total Kjeldahl Nitrogen, TKN), dissolved organic nitrogen plus ammonia (DKN), dissolved nitrate plus nitrite nitrogen (NO_x), dissolved nitrite (NO_2), total phosphorus (TP), dissolved phosphorus (DP), ortho-phosphate (PO_4), dissolved silica (SiO_2);
4. carbon-related measures of total inorganic carbon (TIC), dissolved inorganic carbon (DIC), total organic carbon (TOC), dissolved organic carbon (DOC), and 5-day biochemical oxygen demand (BOD_5).

Specific analytes determined by NWQL included total particulate carbon (TPC), particulate organic carbon (POC), particulate inorganic carbon (PIC), and particulate nitrogen (TPN). From the analytical results, values for total nitrogen (TN = TKN plus NO_x), total organic nitrogen (TON = TKN minus NH_3); nitrate (NO_3 = NO_x minus NO_2); total fixed solids (an estimate of the inorganic content of the solid residue) (TFS = TSS minus TVS); and particulate phosphorus (PP = TP minus DP) were computed.

The chemical analytical results were reported as values above laboratory reporting levels (LRLs). The LRLs of the analytes measured were set at a quantitation limit that was greater than the actual method detection level of the analyses. These LRLs ensured the true concentration was detected and reported with a 99 percent confidence (Childress and others, 1999). Parameter concentrations above the LRL were reported as a measured value. If a parameter was not detected during the analysis, the concentration was reported as less than the laboratory reporting level (< LRL). For data analysis purposes, these "less than" values were set to zero. If the parameter was detected at a level below the LRL but above the actual method detection level, the concentration was considered to be semi-quantitative because of increased measurement uncertainty (< 99 percent confidence) and was reported as estimated value (E). However, for data analysis purposes, these estimated values were set to the detected, semi-quantitative value.

Semi-Continuous Summer Sampling

Since large and rapid changes in some water quality parameters can be stressful to estuarine biota, especially with respect to dissolved oxygen in the summer, semi-continuous water quality measures were collected by SCDNR during the summer 2002 sampling events. Multiprobe data logger instruments (Hydrolab® DataSonde 3 or DataSonde 4, or YSI Model 6920 water quality meters) were deployed to measure temperature, pH, salinity, conductivity, dissolved oxygen, and depth. In headwater tidal creeks, the loggers were housed in a PVC tube attached to a metal pole that was located at the 5/6th location in each creek. The loggers were positioned approximately 0.3 m above the bottom and recorded measurements every 30 minutes for five days prior to the sampling event for each creek. Large tidal creek and open water stations were monitored over a 25-hr period, with measurements collected every 15 minutes to assess changes in condition over two complete tidal cycles. These meters were deployed approximately 0.5 m above the bottom using an anchor and buoy system at the site where the other water quality measures were collected.

Each data logger was calibrated prior to deployment following the manufacturer's recommendations. Upon return to the laboratory, post-deployment QA/QC checks were performed to ensure that the machine functioned over the deployment period and to verify the accuracy and validity of the data. QA/QC standards for dissolved oxygen were not met in Rose Dhu Creek and a logger was redeployed two weeks later. The salinity values that were recorded from Bass Creek were suspect, even though the logger passed the QA/QC check upon return to the laboratory. Therefore, salinity is not reported for Bass Creek and it was excluded from all analyses. Water quality data were carefully reviewed and readings taken before or after deployment, or when the instrument was air exposed during low tide were removed. Rather than losing important records during low tide exposure periods, records were conservatively extrapolated to reflect estimated water quality conditions.

Phytoplankton Sampling

Phytoplankton samples were collected in triplicate by USGS from all sites at 0.3 m below the surface using acid-cleaned bottles, kept at ambient temperature, and transported to the Algal Ecology Laboratory at SCDNR for phytoplankton biomass and community composition analyses. Phytoplankton biomass was estimated from chlorophyll-*a* concentration. Aliquots were filtered onto glass-fiber filters (GF/F), and chlorophyll was extracted in 90% acetone using a freeze-thaw method and measured fluorometrically. Phytoplankton community composition was determined in two ways; microscopy and HPLC. Identification of harmful algal species was accomplished by microscopic inspection of fresh, unpreserved samples (i.e., screening). Aliquots also were fixed with 3% acid-Lugols solution or 10% hexamethylenetetramine-buffered formaldehyde for sample preservation for enumeration of harmful algae if sample screening revealed potentially high numbers (> 100 cell ml^{-1}) and/or these species made up a relatively high proportion of total community abundance. In no case was enumeration of harmful algal species needed based on these criteria. Phytoplankton community composition also was estimated by High Performance Liquid Chromatographic (HPLC) pigment analysis (Kempton and others, 2002). Pigment profiles were obtained by filtering samples onto GF/Fs, flash-freezing in liquid nitrogen, and extracting in 1 ml of 100% HPLC-grade acetone for 1 hr. Pigments were separated on an Agilent 1100 High Performance Liquid Chromatograph (HPLC) system via a reverse-phase, C_8 column. Individual pigment spectra were analyzed (HPChemstation software, Agilent Technologies) and calibrated pigments were quantified. This method determines the concentration of 20 pigments of known chemotaxonomic importance to algal identification.

Data are presented on seven marker pigments related to taxonomic type. Marker pigment concentrations were normalized to chlorophyll-*a* concentration and therefore the values represent the relative contribution of marker pigment biomass to overall community biomass. The marker pigments included peridinin, fucoxanthin, prasinoxanthin, alloxanthin, zeaxanthin, lutein, and chlorophyll-*b*. Peridinin is found in some species of dinoflagellates but no other phytoplankton and is therefore a specific marker for a subset of dinoflagellates. Fucoxanthin is widely used as a marker for diatoms, a group in which it is universally present in high relative amounts. However, it is also found in some species of chrysophytes and prymnesiophytes (aka. haptophytes), as well as a subset of dinoflagellates. The latter includes *Kryptoperidinium foliaceum*, a "harmful" species that was found in this study (see below). Therefore, caution is warranted in extrapolating fucoxanthin values to diatom biomass. Prasinoxanthin is a specific marker for a subset of prasinophytes. Alloxanthin is a specific marker for cryptophytes. Zeaxanthin has been used as a marker pigment for cyanobacteria (blue-green algae) but is also found in chlorophytes, prasinophytes, raphidophytes, and euglenophytes. Lutein and chlorophyll-*b* are indicative of green algae.

Bacterial Composition

Surface water samples were collected for the analysis of fecal coliform bacteria on a seasonal basis in headwater tidal creeks only by USGS and were also collected at all tidal creek and open water sites during the summer of 2002 by SCDNR. The SCDNR-collected samples were analyzed by the NOAA Laboratory for fecal coliform density and Multiple Antibiotic Resistance (MAR). In headwater tidal creeks, water samples were collected at the most upstream site. Samples for MAR analyses were collected by the SCDNR using sterile 500 mL polypropylene bottles and placed on ice for transport to the laboratory. The holding time for these samples was less than 6 hours. Fecal coliform bacterial concentrations for each sample were determined using the membrane filter technique according to Standard Methods for the Examination of Water and Wastewater (APHA, 1995). Six or seven volumes for each sample were filtered in order to obtain a countable plate (20-60 colony forming units). Filters were placed on m-FC-medium plates, which were placed in water-tight bags and submerged in a 44.5 °C water bath for 24±2 hours. Following incubation, blue to gray colonies on each plate were counted and fecal coliform bacteria counts per 100 mL for each sample was determined.

A Multiple Antibiotic Resistance (MAR) assay was performed on the plated samples from above. Isolates of *Escherichia coli* were confirmed using Nutrient Agar with 4-methylumbelliferyl-β-D glucuronide (MUG) from each sample and were further tested for MAR following the method of Parveen and others (1997). Efforts were made to obtain ten confirmed *E. coli* isolates from each sample; however, in some instances when a water sample had a low fecal coliform count, fewer isolates were obtained. If less than eight confirmed *E. coli* strains could be isolated from a sample, the MAR site index was not calculated. Isolates were inoculated into a 96-well plate containing tryptic soy broth (TSB) and incubated for 4-6 h at 37 °C. The broth cultures were then transferred to Mueller-Hinton agar plates, each containing one of ten antibiotics: ampicillin (10 μg/mL), chlortetracycline (25 μg/mL), kanamycin (25 μg/mL), nalidixic acid (25 μg/mL), neomycin (50 μg/mL), oxytetracycline (50 μg/mL), penicillin G (75 U/mL), streptomycin (12.5 μg/mL), sulfathiazole (500 μg/mL), and tetracycline (25 μg/mL), and control plates without antibiotics. Plates were incubated 18-24 hours at 37 °C. Each plate was then imaged with a digital camera and analyzed using the Sigma Scan Pro (SPSS, Inc.) program to measure the size of bacterial colonies on the agar plates. Colony size on antibiotic plates was measured and compared to the colony size for the same isolate on the control plate.

In addition to MAR, USGS collected additional surface water samples at two headwater tidal creeks to identify potential non-point sources based on the presence of human wastewater indicators. Recently developed analytical techniques (Zaugg and others, 2002) have improved the ability to detect compounds that can be associated with a human wastewater source (Appendix III-3). These wastewater indicators are common components of urban runoff and sanitary wastewater and can be placed into groups that include (1) pharmaceuticals and food by-products, such as cholesterol, caffeine and medications that pass through human systems, (2) fragrances commonly found in personal care products, (3) detergent agents commonly found in cleaning solutions and soaps, (4) pesticides commonly applied to lawns and gardens, and (5) organic compounds common in urban runoff such as fuels and solvents. Detections of multiple compounds and/or groups of compounds in groups 1, 2, or 3 (above) were considered a good indication that human wastewater was the source of fecal coliform bacteria in the watershed. In addition, some of these compounds may have deleterious environmental impacts, such as disrupting endocrine systems of aquatic biota.

The two headwater tidal creeks selected for this analysis were Palmetto Bluff and Heyward Cove creeks. Palmetto Bluff Creek drains a forested watershed with little or no human impact while Heyward Cove Creek drains a suburban watershed with the greatest human population density of the headwater tidal creeks sampled in the May River. Samples were collected during the Fall 2002 and Winter 2003 sampling trips. The samples were collected in 1-liter pre-baked, amber glass bottles as a grab sample, placed on ice, and shipped overnight to the NWQL.

Sediment Quality

Sediments from all sites were evaluated for composition, contaminant concentrations, and toxicity. At the headwater tidal creek sites, all surficial (upper 2-3 cm) sediment samples were collected intertidally using various hand cores at a standard height of 1 m below mean high water. Sediment composition was evaluated at six randomly selected locations within each creek and sediment contaminants and toxicity were assessed at the primary site.

At the large tidal creek and open water sites, 5-7 replicate grab samples were collected using a stainless steel 0.04 square meter (m^2) Young grab sampler from an anchored boat. Only the surficial sediments (upper 2-3 cm) were collected from these grabs and combined to produce a composite sediment sample for analysis of sediment composition, contaminants, and sediment toxicity. The boat was repositioned after each sample to ensure that the same bottom was not sampled twice and to spread the samples over a 10-20 m^2 bottom area. All grab samplers were thoroughly cleaned prior to field sampling and rinsed with isopropyl alcohol and seawater between stations.

Sediment composition analyses included measures of benthic chlorophyll-*a*, porewater chemistry (i.e., pH, salinity, and ammonia), grain size, and total organic carbon (TOC). Benthic chlorophyll-*a* concentration was measured only at the headwater tidal creek stations from sediment samples collected with a 33 mm^2 plastic core to a depth of ~1 cm. Samples were placed on ice and remained in the dark during transport back to the laboratory. An acetone extraction method developed from a modified Strickland and Parsons (1972) spectrophotometric method was used to measure chlorophyll-*a* concentration.

Porewater pH, salinity, and ammonia were measured at all large tidal creek and open water sites while only porewater ammonia was measured in headwater tidal creeks. In headwater tidal creeks, sediments for this analysis were collected with a 12.5 cm^2 PVC core to a depth of ~3 cm. All samples were placed on ice during transport back to the laboratory where they were then homogenized and centrifuged. The supernatant was analyzed for pH and salinity (large tidal creek and open water only) as well as ammonia concentration using the salicylate-cyanurate method (Hach Company, 1994).

Grain size was analyzed using a modified Plumb (1981) method in which collected sediments were separated into their constituent silt, clay, and sand fractions. In large tidal creek and open water sites and at the primary site in each headwater tidal creek % silt and % clay were distinctly analyzed so as to coincide with contaminant analyses. However, at all other stations within headwater tidal creeks they were combined and were measured as % silt/clay. TOC was analyzed at all sites using the combustion method on a CHNS Analyzer (Perkin-Elmer, 1994).

Sediment samples for contaminant and toxicity analyses were collected from the primary site in each headwater tidal creek and from all large tidal creek and open water sites. A broad suite of chemical analytes were evaluated (Appendix III-4) and two toxicity assays were conducted. Sediments were homogenized on site and distributed into the appropriate pre-cleaned plastic or glass jars (i.e., metals in plastic and organics in glass). Samples were placed on ice during transport and upon return to the laboratory, contaminant samples were stored at -60°C and toxicity samples were stored at 4°C until analysis.

The NOAA Laboratory analyzed sediments for four main classes of contaminants; trace metals, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and organochlorine pesticides. Trace metals were analyzed using methods described by Long and others (1998) using inductively coupled plasma spectroscopy (ICP) for aluminum, chromium, copper, iron, manganese, nickel, tin, and zinc, and using graphite furnace atomic absorption for arsenic, cadmium, lead, selenium, and silver.

Mercury concentrations were analyzed by direct mercury analysis with atomic absorption detection. The extraction and sample preparation methods that were followed for organics were similar to those described by Krahn and others (1988) and Fortner and others (1996). Briefly, samples were extracted with CH_2Cl_2 using accelerated solvent extraction (ASE), concentrated by nitrogen blow-down, and cleaned by gel permeation chromatography where necessary. PAHs were quantified by capillary GC-ion trap mass spectrophotometry (ITMS) and HPLC. PCBs and organochlorine pesticides were analyzed using dual column gas chromatography with electron capture detection (GC-ECD) using methods described by Kucklick and others (1997). Concentrations of contaminants that were below the level of detection were set to 0.

To summarize sediment contaminant data, the concentrations of trace metals, PAHs, PCBs, and pesticides from the May River creek and open water sites were compared to sediment quality guidelines established by Long and others (1995). Long and others (1995) summarized published literature on the effects of a suite of sediment contaminants on a wide range of marine biota and derived two threshold values, an effects range-low (ERL) and an effects range-median (ERM) for individual analytes. An ERL was defined as the sediment concentration of a given contaminant where 10% of all published studies have reported an adverse effect and an ERM was defined as the sediment concentration where 50% of all published studies have reported an adverse effect. Values below the ERL would rarely be expected to be associated with measurable biological effects. Values between the ERL and ERM represent a range in which there are possible biological effects for a wide range of organisms. Values above the ERM represent a range above which there are probable biological effects.

An effects range-median quotient (ERMQ) was calculated for each major class of contaminant (i.e., trace metals, PAHs, PCBs, and pesticides) only in headwater tidal creeks using the 24 analytes outlined by Long and others (1995). Calculations were made by: (1) dividing the concentration of each analyte by the ERM value published by Long and others (1995); (2) summing the ratios of analytes within each contaminant class (e.g., trace metals); and (3) dividing by the number of contaminants in that class. In addition, grand total ERMQ values, which encompassed all four classes of contaminants, were calculated for each habitat (i.e., headwater tidal creeks, large tidal creeks, and open water sites). These values were calculated in the same fashion except that analytes were not combined within contaminant classes, instead the ratios of all 24 analytes were summed, and the total was divided by 24 (Long and others, 1998).

Sediment toxicity was measured using multiple assays. The Microtox® assay utilized the photoluminescent bacterium, *Vibrio fischeri*, to provide a sublethal toxicity measure, which was based on the attenuation of light production by the bacterial cells due to toxicant exposure. Solid-Phase Microtox® assays followed the protocols described by the Microbics Corporation (1992). Toxicity was based on criteria described by Ringwood and others (1997), in which variations in response due to sediment composition were accounted for. The seed clam assay involved exposing juvenile clams, *Mercenaria mercenaria*, to site-collected sediments for a 7-day period using protocols described by Ringwood and Keppler (1998). Toxicity was measured using both sublethal (growth rate) and lethal end points compared to exposure in control sediments.

Biological Quality

Benthic Community

Benthic organisms are important components of the diets of juvenile fish and crustaceans and represent an important link in the food web between primary producers and fish, crabs, and shrimp (Cummins and Wuycheck, 1971; Middleditch and others, 1979). Benthic community structure is also an integrative measure of habitat quality over time. Because benthic organisms have limited mobility and generally cannot escape pollution stress, they have been used as indicators of biological quality and environmental integrity (Pearson and Rosenberg, 1978; Dauer and others, 1992, 2000; Weisberg and others, 1997; Lerberg and others, 2000; Van Dolah and others, 2000).

In the May River, the macrobenthic communities were sampled at all sites, but different protocols were used at the intertidal headwater creek sites compared to the subtidal large tidal creek and open water sites. In the headwater areas, six randomly selected sites were sampled within each creek during low tide at ~1 m below mean high water using a 46 cm^2 diameter hand core to a depth of ~15 cm. Samples were placed on ice during transport back to the laboratory where they were sieved on a 500 µm mesh screen. In large tidal creek and open water sites benthic communities were evaluated from three replicate 0.04 m^2 Young grab samples collected at each site in conjunction with the other sediment samples. These samples were processed in the field by washing the sediments through a 500 µm sieve to collect the benthic fauna. All fauna and sediment retained on the sieve from all stations was preserved in a 10% buffered formalin-seawater solution containing rose bengal stain.

All intertidal and subtidal benthic samples were sorted in the laboratory to remove the organisms from sediments remaining in the sample and the organisms were then counted and identified to the species level, or the lowest practical level possible if the specimen was damaged or incomplete. A reference collection of all specimens is maintained at the SCDNR Marine Resources Research Institute. One out of every 10 samples was resorted by a qualified scientist to ensure that a 90% sorting efficiency was maintained. Additionally, one out of every 10 samples was re-identified by an experienced taxonomist to ensure a 90% identification accuracy.

Nektonic Community

The fish and crustacean communities present in the May River system were sampled quantitatively at each station. As noted for the benthic community sampling, different protocols were used at the intertidal headwater creeks compared to the subtidal large tidal creek and open water sites. Sampling in the headwater creeks was conducted at the site located 5/6th of the creek's length downstream from the upper extent of the creek using a seine that spanned from creek bank to creek bank with a square mesh and 6-cm bar with 16-kg weight webbing. The seine was pulled for 25 m upstream against an ebbing tide when water was less than 1 m but greater than 0.25 m deep. Organisms collected in the seine were stored on ice during transport to the laboratory where they were preserved in 10% formalin for at least one week before being processed.

Samples with a total wet weight of <2 kg were completely identified, whereas those that exceeded 2 kg were subsampled, as the time required to completely process these samples would have been excessive compared to the information obtained. The subsampling method outlined by Holland and others (1996) was followed and is briefly described here. Large organisms and rare taxa were first removed from the seine sample and placed in 50% isopropanol for identification. The remaining sample was then divided into 10 approximately equal weight subsamples. Two of the ten subsamples were randomly selected for identification and enumeration of taxa, and the remaining subsamples were stored. The abundances of organisms identified from the two subsamples that were processed were extrapolated to estimate the abundances of taxa in the whole sample. One of the six seine samples collected was re-identified by an experienced taxonomist to ensure a 90% identification accuracy.

At the large creek and open water sites, the finfish and crustacean communities were sampled using a small trawl. Two replicate tows were made at each site using a 4-seam trawl (18-ft rope, 15 ft head rope and 0.75-in bar mesh throughout). Trawl tow lengths were standardized to 0.5 km for open water sites and 0.25 km for creek sites. Tows were made only during daylight hours with the current and speeds standardized as much as possible. Tows made in tidal creeks were limited to periods when the marsh was not flooded (approx. 3 hrs $\pm$ mean low water). This limitation also was generally applied to open water sites. Catches were sorted to lowest practical taxonomic level, counted, and checked for gross pathologies, deformities or external parasites. All organisms were measured to the nearest 0.1 cm. When more than 25 individuals of a species were collected, the species was subsampled for measurements.

Data Analysis and Review

All statistical analyses were considered significant when $\alpha \leq 0.05$. Physical, chemical, and biological parameters were evaluated among stations within each habitat type (i.e., headwater tidal creeks, large tidal creeks, or open water sites) with one-way analysis of variance (ANOVA) models using SAS® for Windows or Sigmastat® software. Where applicable, results were also evaluated between watershed classes (i.e., forested vs. suburban), or between station types (i.e., large tidal creek vs. open water). In headwater tidal creeks, data for certain parameters were only collected from one location in each creek (i.e., semi-continuous water quality measures, sediment contaminant concentrations) and were only analyzed for differences between watershed types, and not among creeks.

Data were first tested for normality with a Shapiro-Wilk test to ensure that the assumptions of ANOVA were met. These assumptions were addressed differently by SCDNR and USGS. Although ANOVA is a robust analysis and can tolerate deviations from normality, caution should be employed when evaluating datasets with small sample sizes, like the May River (USEPA, 2000). SCDNR transformed data when necessary and allowed slight departures from the assumptions of ANOVA. In cases of extreme departures, non-parametric Kruskal-Wallis tests were used and post-hoc pairwise comparisons were made using the Tukey test. USGS did not allow for slight departures from the assumption of normality because of the relatively small sample size. For non-normal data, ANOVAs were conducted on ranked data and when significant, post-hoc multiple pairwise comparisons tests were performed. The Tukey test was used when sample sizes were equal and the Scheffe test was used when sample sizes were unequal. If a parameter was not detected during the analyses, the concentration was reported less than the laboratory reporting level (<LRL). For data analyses purposes, these "less than" values were set to zero. If the parameter was detected at a level below the LRL, the

concentration was reported as an estimated value "E" (Childress and others 1999). For data analyses purposes these estimated values were set to the detected value.

Spearman's Rho correlation analysis was used to evaluate associations between the point water quality data from the different habitats (headwater tidal creeks, large tidal creeks, and open water sites). Correlation measures the observed co-variation between two variables and is quantified with a coefficient. The coefficient for this analysis is called rho, which ranges from 0 to 1; the closer the rho is to 1, the stronger the correlation. Caution should be used in interpreting the results of these analyses as correlation proves only co-variation and does not imply causality.

Data from similar studies of headwater creeks, large tidal creeks, and open water sites in South Carolina were compared to these baseline data, in an effort to place the May River in a larger scope. For headwater creeks, comparisons were made to data collected by the TCP from tidal creeks in the Charleston Harbor Estuary (CHE) during the summers of 1994 or 1995. Two-way ANOVA models were used to evaluate the effects of location (May River or CHE), watershed class, and their interaction on various parameters. Additionally, using linear regression analysis, the relationship between key ecological parameters (identified by Holland and others, 2004) and the percentage of impervious cover were evaluated. For large tidal creeks and open water sites, comparisons were made to data collected by SCECAP from pristine sites sampled in the southern region of the state during the summer of 2002. Two-way ANOVA models were used to evaluate the effects of location (May River or SCECAP) and station type (large tidal creek or open water).

C. Results and Discussion

Headwater Tidal Creek Habitats

Watersheds/Land Use

The watersheds of the six headwater tidal creeks selected for this study cover an area of 3,853 hectares of the larger May River watershed. Watersheds ranged in size from 83 to 2,147 hectares (Table III-2). Watershed size decreased along a gradient that coincided with location in the May River, the largest watershed was situated in the upper zone (Stony Creek) while the smallest watershed was located in the lower zone near the mouth of the system (Bass Creek). The size of the Stony Creek watershed is larger than the size of most watersheds previously studied in South Carolina. It is similar in size, however, to the Okatee Creek watershed located on the Okatee River that is a part of the Land Use-Coastal Ecosystem Study (LU-CES) (Gillett, 2003).

The six headwater tidal creek watersheds were classified into watershed types based upon established criteria related to land use, impervious cover, and population density. Stony, Rose Dhu, Palmetto Bluff, and Brighton Beach creeks were classified as forested or reference creeks, while Heyward Cove and Bass creeks were classified as suburban creeks (Table III-2). None of the creeks in this study met the criteria for urban or industrial watersheds.

Table III-2. Watershed characteristics of the headwater tidal creeks sampled in the May River. Note: population density is in persons per hectare.

Creek Name	Watershed Size (hectares)	% Impervious Cover	1990 Population Density	2000 Population Density	% Change (1990-2000)
Forested					
Stony (U-10)	2147	3.27	0.248	0.561	+ 56
Rose Dhu (U-11)	813	0.97	0.068	0.108	+ 38
Palmetto Bluff (U-12)	447	1.34	0.011	0.000	- 100
Brighton Beach (M-11)	239	2.27	1.065	2.282	+ 53
Suburban					
Heyward Cove (M-10)	124	19.26	1.540	5.431	+ 72
Bass (L-10)	83	23.28	2.590	1.133	- 56

Table III-3. Land use of each headwater tidal creek watershed, expressed as a % of the total area.

	% of Total Area						
Creek Name	Agricultural	Barren	Forest	Golf	Suburban/ Urban	Water	Wetland
Forested							
Stony	0.00	16.76	76.00	0.02	6.01	1.00	0.20
Rose Dhu	8.48	4.89	84.91	0.00	1.15	0.06	0.52
Palmetto Bluff	0.00	9.31	88.53	0.00	0.00	0.00	2.16
Brighton Beach	0.00	1.01	63.51	0.00	30.46	1.18	3.84
Suburban							
Heyward Cove	0.00	10.38	53.70	0.00	34.14	1.21	0.57
Bass	0.00	5.56	13.68	3.19	59.91	0.82	16.85

Reference creeks were dominated by forested land cover (mean = 78.2%) with little suburban/urban land cover (mean = 9.4%). Suburban creeks had a mean of 47.0% suburban/urban land cover but were also comprised of a mean of 33.7% forested land cover (Table III-3). Unlike the other watersheds, approximately 69 hectares of the Rose Dhu Creek watershed was agricultural and 2.7 hectares of the Bass Creek watershed was a golf course. Brighton Beach Creek had an urban land cover of 30.5%, which is typically indicative of a suburban creek; however, the population density (23 persons/hectare) and impervious cover (2.3%) were lower than the established criteria for suburban watersheds (Table III-3). Therefore, Brighton Beach Creek was classified as a forested creek but should be considered a creek in the process of transitioning to a suburban creek.

Bass Creek had the highest percentage of impervious cover in its watershed (23.2%) (Table III-2). Suburban creeks (mean = 21.2%) had higher levels of impervious cover than forested creeks (mean = 1.9%). The range of impervious cover in the May River watersheds (1 to 23%) was at the lower end of the range of impervious cover in TCP watersheds (0 to 85%) (Holland and others, 2004). None of the watersheds in the May River exceeded the 30% impervious cover threshold that has been associated with the degradation of biological resources (Arnold and Gibbons, 1996; Holland and others, 2004).

Heyward Cove Creek was the most populated of the six watersheds assessed in this study and had a population density of 5.43 persons/hectare (Table III-2). Heyward Cove Creek is located near the center of the original Town of Bluffton. Conversely the watershed of Palmetto Bluff Creek had a population density of 0 persons/hectare (U.S. Census Bureau, 2000). In general, suburban creeks had a higher population density (mean = 3.28 persons/hectare) compared to forested creeks (mean = 0.723 persons/hectare). The Bass Creek watershed had an unexpectedly low population density of 1.13 persons/hectare considering it had the highest degree of urban land use (60.0%) and impervious cover (23.2%) of all May River headwater creeks. In spite of its low population density, Bass Creek was classified as a suburban creek.

The total population density for all six headwater creek watersheds analyzed in this study has tripled since the 1990 United States Census. The most extensive population growth occurred in the watersheds of Heyward Cove, Stony, and Brighton Beach creeks, in which population increased by 72, 56, and 53%, respectively (Table III-2).

Water Quality and Chemistry

Field measurements of dissolved oxygen (DO), pH, specific conductance, turbidity, and salinity, and laboratory analytical results for suspended solids, nutrient, organic carbon, silica, and biochemical oxygen demand data are discussed separately in this report; however, it is important to understand that the interaction of these parameters through physical, biological, and chemical processes effects the actual water quality conditions in a creek or estuary and impacts the biota. For example, nutrients such as phosphorus, nitrogen, and silica, are assimilated by algae and other aquatic plants from the water for growth. The greater the amount of available nutrients present in the water, the greater the potential exists for plant growth. During the growth process of aquatic plants, photosynthetic activity produces daytime increases in the DO levels in the water.

Photosynthetic activity also removes carbon dioxide in the water during the daytime causing an increase in the pH of the water. At night, when photosynthetic activity ceases, respiration processes that degrade organic carbon dominate and consume oxygen, reducing the DO levels in the water. Organic carbon can be derived naturally from terrestrial or aquatic plants and it can be introduced by human activities, such as agricultural runoff or wastewater inputs. If the water is enriched in organic carbon, the water can become depleted in DO to the point that is stressful or even lethal to the aquatic biota. The biochemical oxygen demand (BOD) is an estimate of the potential for the depletion of dissolved oxygen by microbial processes.

Results of the analyses are summarized by parameter in the following sections. Comparisons are made with other studies for some of the parameters (e.g., semi-continuous salinity, pH, DO), but not for others because the parameters were not measured in those studies or did not include seasonal measures. Comparisons also were made between forested and suburban creeks in the May River.

Temperature

Seasonal Point-Sampling Data

Average water temperatures during the seasonal sampling ranged from 23.1^{o}C at Heyward Cove and Rose Dhu to 23.6^{o}C at Palmetto Bluff and were very similar among stations within a season and between land-use classes (Appendix III-5). Trends in seasonal water temperatures in the May River are better described by the continuous USGS monitoring stations established for this study (see Section II).

Summer Semi-Continuous Data

Summer mean temperature during the 5-day semi-continuous monitoring ranged from 28.5 to 36.6°C (mean = 30.0°C) across all creeks, which is common for southeastern tidal creek water temperatures during summer months. Mean temperature was not statistically different between watershed types (p-value = 0.0688) however, the May River creeks had higher mean temperatures than the TCP creeks (p-value = 0.0043).

Salinity

Salinity fluctuation, especially in an estuarine environment, is an important factor in maintaining a diverse estuarine ecosystem. Salinity directly affects the distribution, abundance and composition of aquatic biota at different stages of their life cycle. Estuaries experience significant salinity fluctuations both daily and seasonally as a result of freshwater inflow from tributaries and storm runoff, tidal flux, and wind activity. In the TCP study, salinities were generally more variable in developed creeks than in reference creeks, probably due to the increased runoff associated with the increased amount of impervious cover (Holland and others, 2004).

Seasonal Point-Sampling Data

Statistical analysis of the seasonal point salinity data did not identify a significant difference among the headwater tidal creeks (p-value = 0.2828) or between forested and suburban land use (p-value = 0.4873; Appendix III-5). The mean salinity of headwater creeks varied between 9.9 parts per thousand (ppt) at Heyward Cove Creek to 29.2 ppt at Bass Creek (Figure III-3). The suburban sites had salinity values that were half that of the forested sites; however, Heyward Cove had low salinity and Bass Creek had high salinity levels probably resulting from the close proximity to the mouth of the May River. Differences in salinity with season were determined to be significant among the headwater tidal creeks (p-value = 0.0004), such that salinities in the spring and summer were significantly higher than salinities in the fall and winter (Figure III-3).

Correlation analysis of the seasonal salinity data identified several parameters significantly associated with salinity in the headwater tidal creeks. The strongest associations included water temperature, alkalinity, total inorganic carbon (TIC), total organic carbon (TOC), chloride, and fecal coliform (Appendix III-5). These associations suggest alkalinity and TIC increases with increases in salinity (such as incoming tides) and fecal coliform and TOC levels tend to increase with decreases in salinity (such as outgoing tides or increases in freshwater inflow during storm events) in the headwater tidal creeks.

Summer Semi-Continuous Data

Headwater tidal creek systems experience wide fluctuations in salinity that are tidally driven and are also influenced by stormwater run-off from the upland (Figure III-4). These fluctuations can be physiologically stressful or intolerable to many organisms. The 5-day monitoring effort was designed to look at short-term changes in salinity patterns over several tidal cycles.

The salinity data for Bass Creek was suspect and is not presented and thus statistical comparisons were not made between forested and suburban watersheds. Among the remaining creeks of the May River, Brighton Beach Creek had the highest mean salinity (mean = 33.9 ppt) and Stony Creek had the lowest mean salinity (mean = 24.8 ppt). Mean salinity was significantly higher in May River creeks (mean = 29.2 ppt) compared to Charleston Harbor creeks sampled in the summer of 1994 (mean = 17.9 ppt; p-value = 0.0023) (Figure III-4). This is not unexpected given the difference in freshwater riverine size and inputs between the May River Estuary and the Charleston Harbor Estuary. In addition, the salinity may have been higher in the May River during 2002 as a result of drought conditions.

Heyward Cove Creek had the largest salinity range, defined as maximum minus minimum in a 5-day period (range = 28.5 ppt), while Brighton Beach Creek had the smallest salinity range (range = 12.7 ppt). In general, May River creeks had larger salinity ranges than TCP creeks (p-value = 0.0055). The mean salinity range in the May River was 19.4 ppt, while the mean salinity range of the TCP creeks from 1994 was 10.4 ppt (Figure III-4). This difference may be attributable to the higher mean salinity and a higher tidal range in the May River (2-3 m) compared to Charleston Harbor (1-2 m).

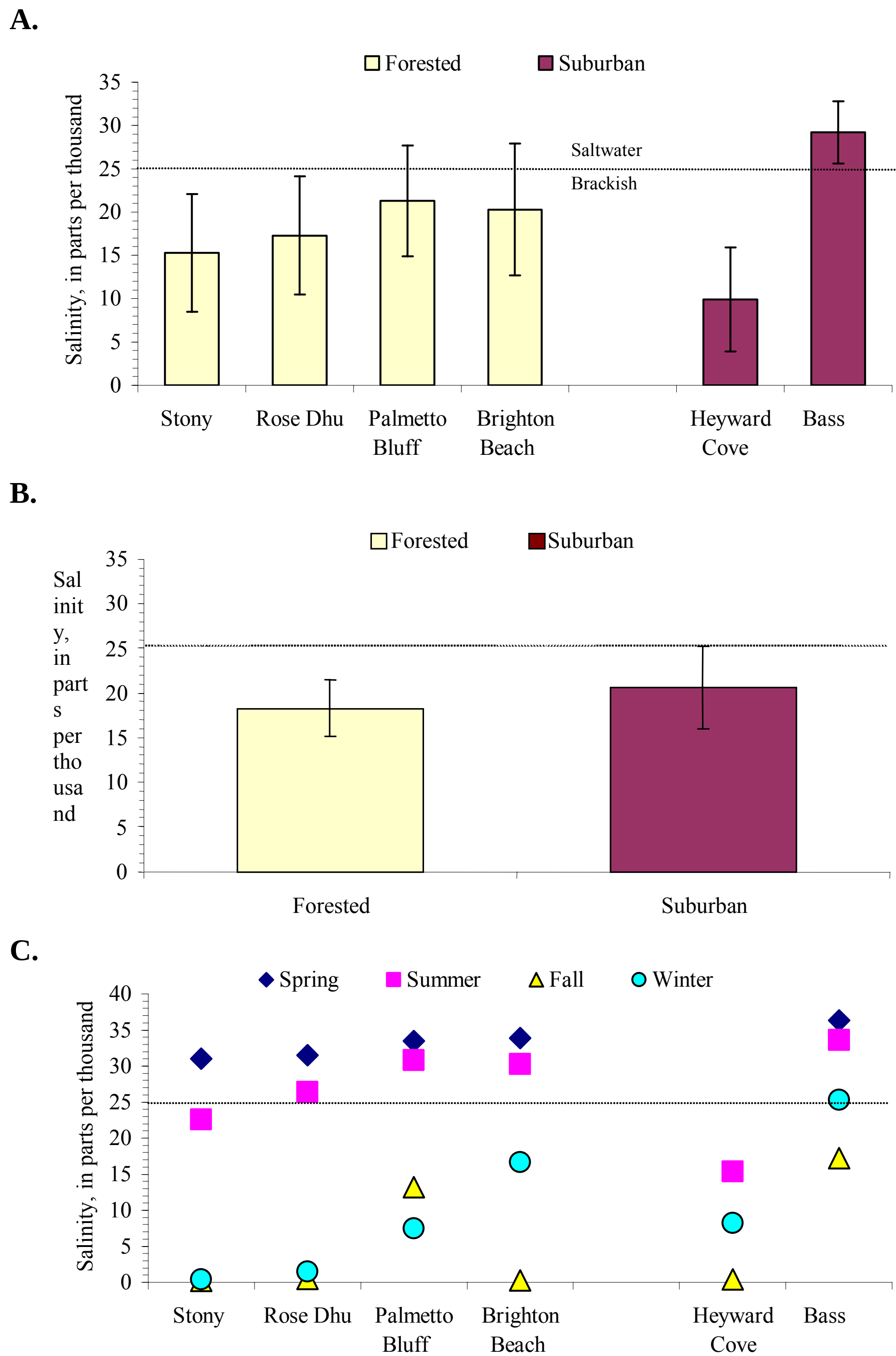


Figure III-3. Mean seasonal salinity among sites (A.) and between forested and suburban land use (B.) and seasonal variation in salinity levels (C.) at 6 headwater tidal creek sites in the May River, 2002 - 2003. Error bars represent 1 standard error.

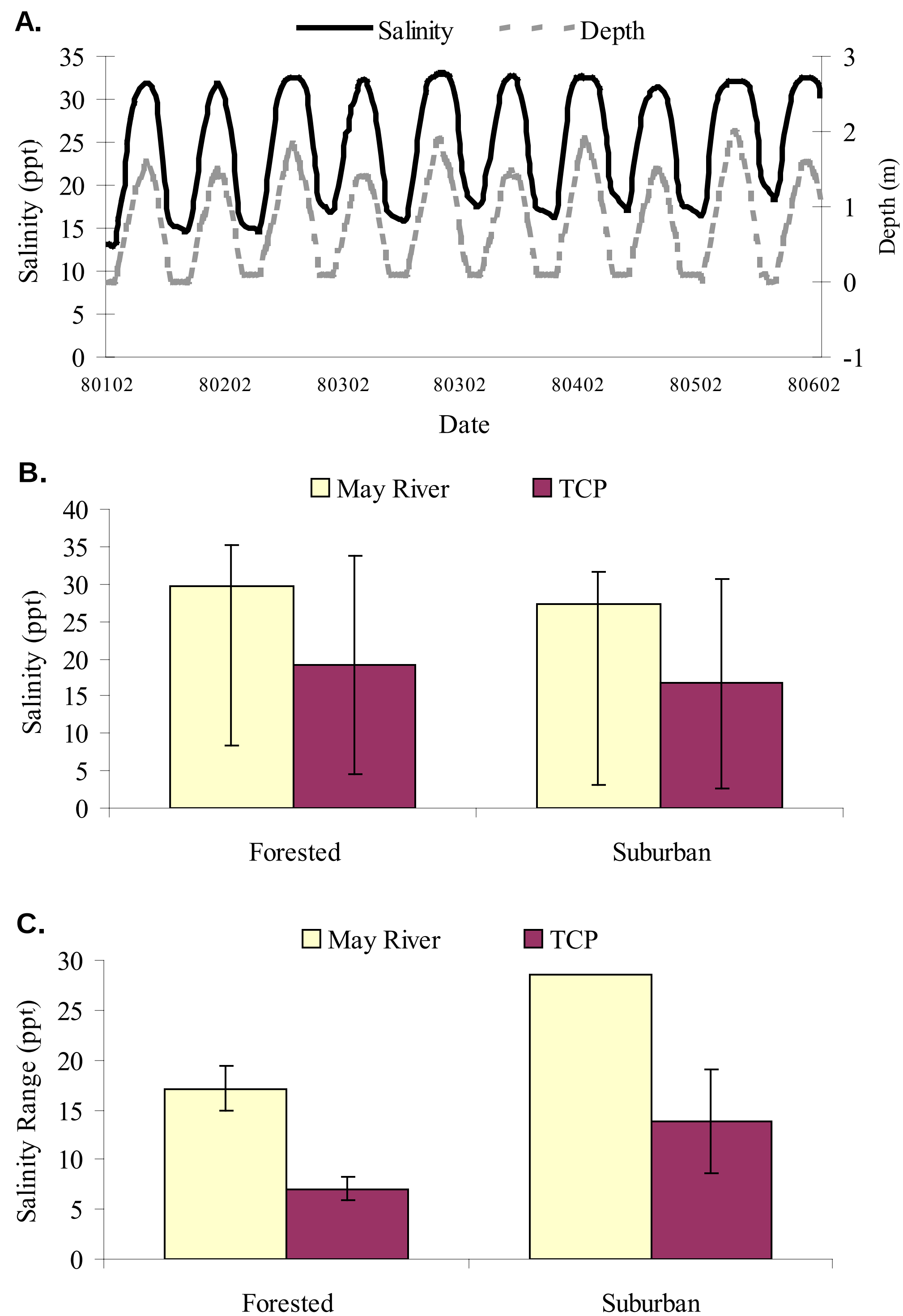


Figure III-4. Semi-continuous salinity measurements from summer 2002. Salinity in Stony Creek, like all May River headwater creeks, fluctuated widely during the 5-day deployment period (08/01/2002-08/06/2002) and was influenced by tidal stage (A.). Mean salinity (B.) and salinity range (C.) recorded from May River forested and suburban headwater tidal creeks were compared to TCP creeks sampled during the summer of 1994. Bars in (B.) represent maximum and minimum values and error bars in (C.) represent 1 standard error.

Dissolved Oxygen

All biota require dissolved oxygen (DO) to survive. If DO concentrations are at the lower end of an organism's tolerance range for extended periods of time, estuarine community may become limited or may die. The amount of oxygen dissolved in water is controlled by several factors, including water temperature and biological processes. In general, the summer season is the time period when water is warmest and the DO concentrations are expected to be at their lowest. Warmer temperatures decrease DO saturation and increase oxygen-consuming biological activity (i.e., growth, respiration). SCDHEC has designated the May River as shellfish harvesting waters. State criteria for such waters require that mean instantaneous, day-time samples not less than 5 mg/L, and minimum DO concentration not less than 4 mg/L.

<u>Seasonal Point-Sampling Data</u>

Among the headwater tidal creeks, mean DO concentrations ranged from 3.74 mg/L at Brighton Beach to 6.30 mg/L at Heyward Cove (Figure III-5). No significant differences in DO concentrations were identified among these sites for the study period (p-value = 0.5134; Appendix III-5). Statistical comparison between headwater sites classified as forested and suburban land use did not indicate significant differences in DO (p-value = 0.5664; Figure III-5). This similarity in DO concentrations between land use classes suggested that natural causes, not suburban development, had the greatest impact on DO levels in the headwater tidal creeks at the time of sampling.

Seasonal effects accounted for much of the statistical variability in the DO concentrations (p-value = 0.0097, Appendix III-5). Although Brighton Beach was the only site with the mean DO concentration below the minimum criteria level of 4 mg/L, all headwater tidal creeks had at least one instantaneous measurement below the minimum criteria during the summer season (Figure III-5). Except for Bass Creek and Brighton Beach the highest DO concentrations were observed in the winter. Summer DO concentrations were significantly lower than winter and spring concentrations, with fall DO concentrations being similar to summer, winter, and spring concentrations.

Correlation analysis of the headwater tidal creek data determined that DO concentration in mg/L and percent saturation were negatively correlated with ammonia, dissolved phosphorus, alkalinity, and TIC (Appendix III-5). This suggests that increased nutrients (ammonia and phosphorus) may reduce DO levels in headwater tidal creeks.

<u>Summer Semi-Continuous Data</u>

Summer data obtained from the 5-day deployment period were evaluated based on the mean values and the percent of time that DO was less than 28% saturation. Conditions of low DO (<28%) or hypoxia naturally occur in headwater tidal creek habitats (Holland and others, 2004), but may also be related to increased eutrophication of these near shore estuarine environments.

DO concentration was highly variable in headwater tidal creeks and was found to fluctuate on both tidal and diurnal (day/night) cycles (Figure III-6). Stony Creek had the lowest mean DO saturation over the deployment period (mean = 32.1% saturation) and experienced the most frequent occurrence of hypoxia (~45% of all records). Heyward Cove Creek had the highest mean DO saturation (mean = 59.9% saturation) and had no records that reflected hypoxic conditions. In general, Stony and Rose Dhu creeks had low DO conditions (mean = 34%) and high frequencies of hypoxia (mean = 43%) compared to the suburban creeks in the May River and also compared to the TCP creeks. The position of Stony and Rose Dhu creeks in the headwater region of the May River and the fact that the location of the 600 meter reach of these two creeks that was sampled in this study was far removed from the mainstem of the river may have contributed to a reduced flushing rate in these systems which resulted in low dissolved oxygen levels.

Forested creeks had lower mean DO saturations and experienced hypoxia more frequently than suburban creeks (Figure III-6); however, these differences were not significant (mean DO, p-value = 0.2226; hypoxia, p-value = 0.0784). Across all creeks, DO averaged 50% saturation and ranged from 1 to 139% saturation, which although extreme, is typical of the DO regime observed in headwater tidal creeks in South Carolina. For example, in Charleston Harbor, mean DO in all creeks was 57% and ranged from 0 to 189%.

pH

pH, in standard units, is used to estimate the degree of acidity of the water by measuring the hydrogen ion concentration. pH can range from 0 to 14, with 7 considered to be neutral; less than 7, acidic; and more than 7, basic. Surface waters generally have a pH range from 6 to 8. The SCDHEC standard for pH in the May River is not lower than 6.5 or greater than 8.5 to ensure the health of aquatic life.

Seasonal Point-Sampling Data

Except for Stony Creek, all seasonal samples had pH values within the state standard range of 6.5 to 8.5. Stony Creek had pH below 6.5 in 2 of 4 seasonal samples and had the greatest range in pH values. The mean pH ranged from 6.70 at Stony Creek to 7.28 at Brighton Beach creek (Figure III-7). Even with Stony Creek's relatively low pH, no significant difference in pH among the headwater tidal creeks was detected (p-value = 0.5685). Mean pH in suburban creeks (7.13) was similar to forested creeks (7.01) in the May River (Figure III-7) (p-value = 0.8178; Appendix III-5). No seasonal trend in pH was identified among the headwater tidal creeks (p-value = 0.1900; Figure III-8). Several parameters were significantly associated with pH based on correlation analysis including salinity, total suspended solids, and silica.

Summer Semi-Continuous Data

Mean pH during the 5-day semi-continuous monitoring in the summer ranged from 6.9 to 7.7, and values were not significantly different between forested and suburban creeks (p-value = 0.1801). The values were typical of southeastern headwater

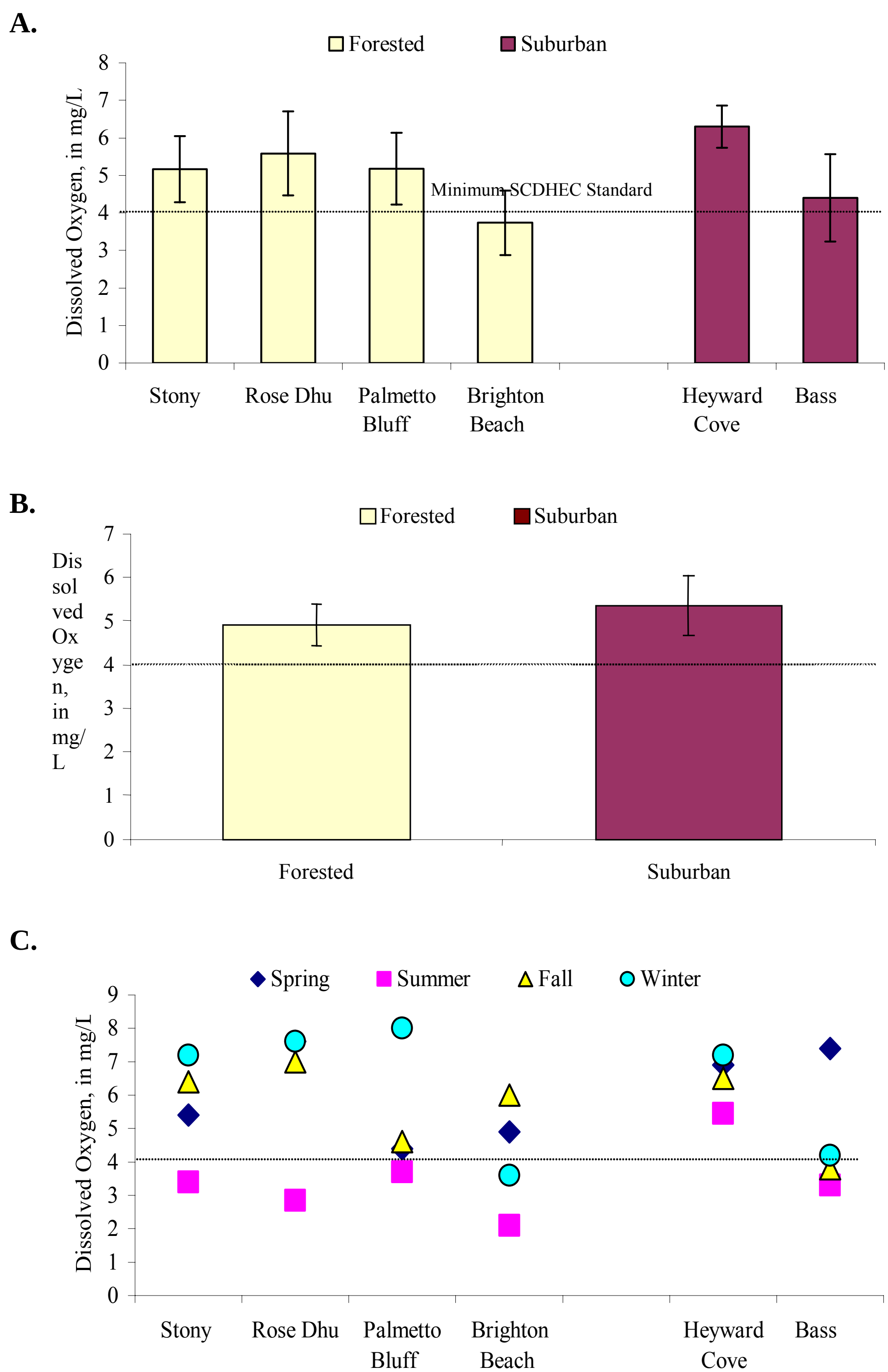


Figure III-5. Mean seasonal dissolved oxygen concentrations among sites (A.) and between forested and suburban land-use (B.) and seasonal variation in dissolved oxygen concentrations (C.) at 6 headwater tidal creek sites in the May River, 2002 – 2003. Error bars represent 1 standard error.

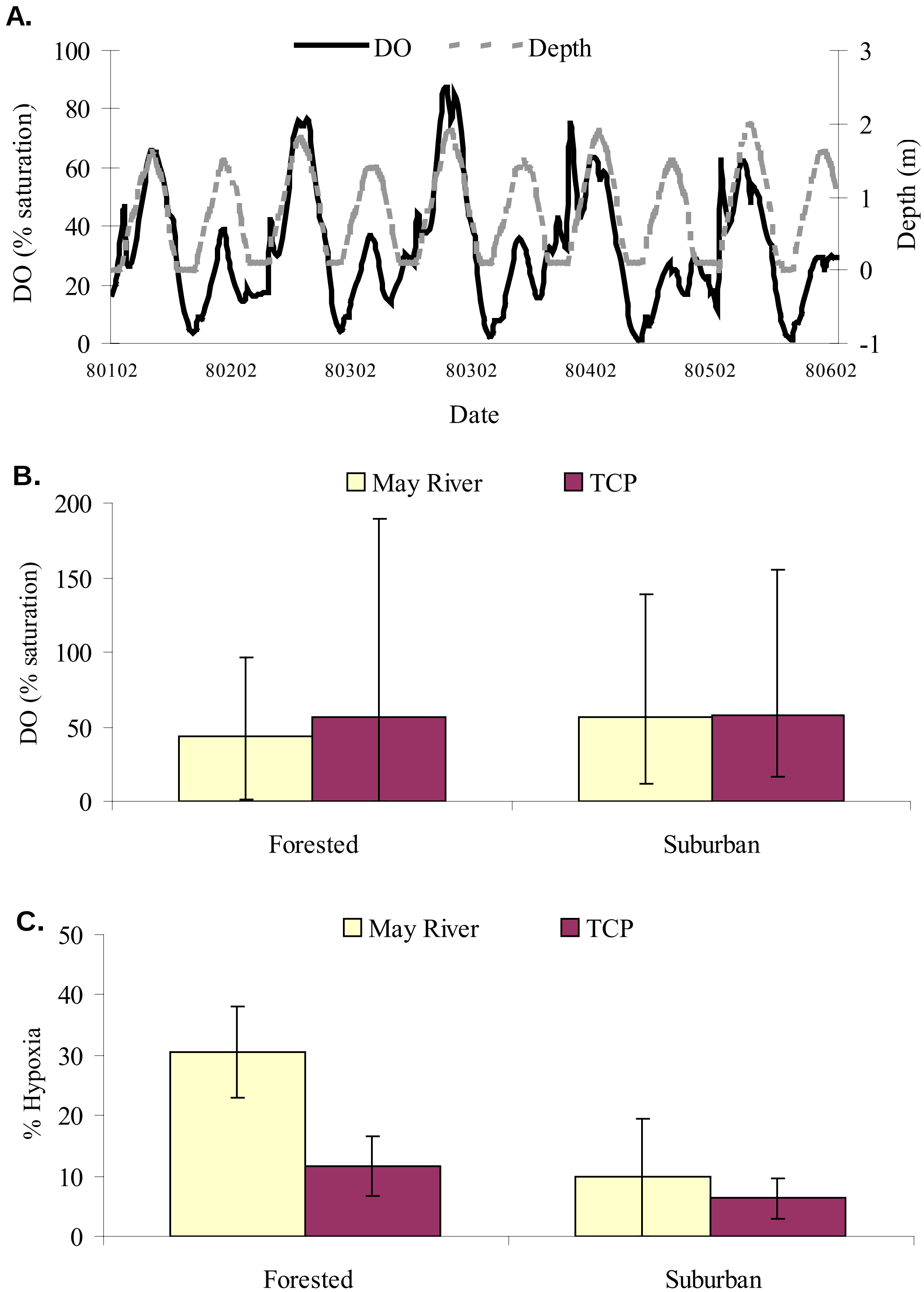


Figure III-6. Semi-continuous dissolved oxygen (DO) measurements (in % saturation) from summer 2002. Stony Creek, like all May River headwater creeks, experienced large fluctuations in DO during the 5-day deployment (08/01/2002-08/06/2002) that coincided with tidal and diel cycles (A.). Mean DO (B.) and mean % hypoxia (time <28% DO) (C.) recorded from May River forested and suburban headwater creeks were compared to TCP creeks sampled during the summer of 1994. Error bars in (B.) represent maximum and minimum values and bars in (C.) represent 1 standard error.

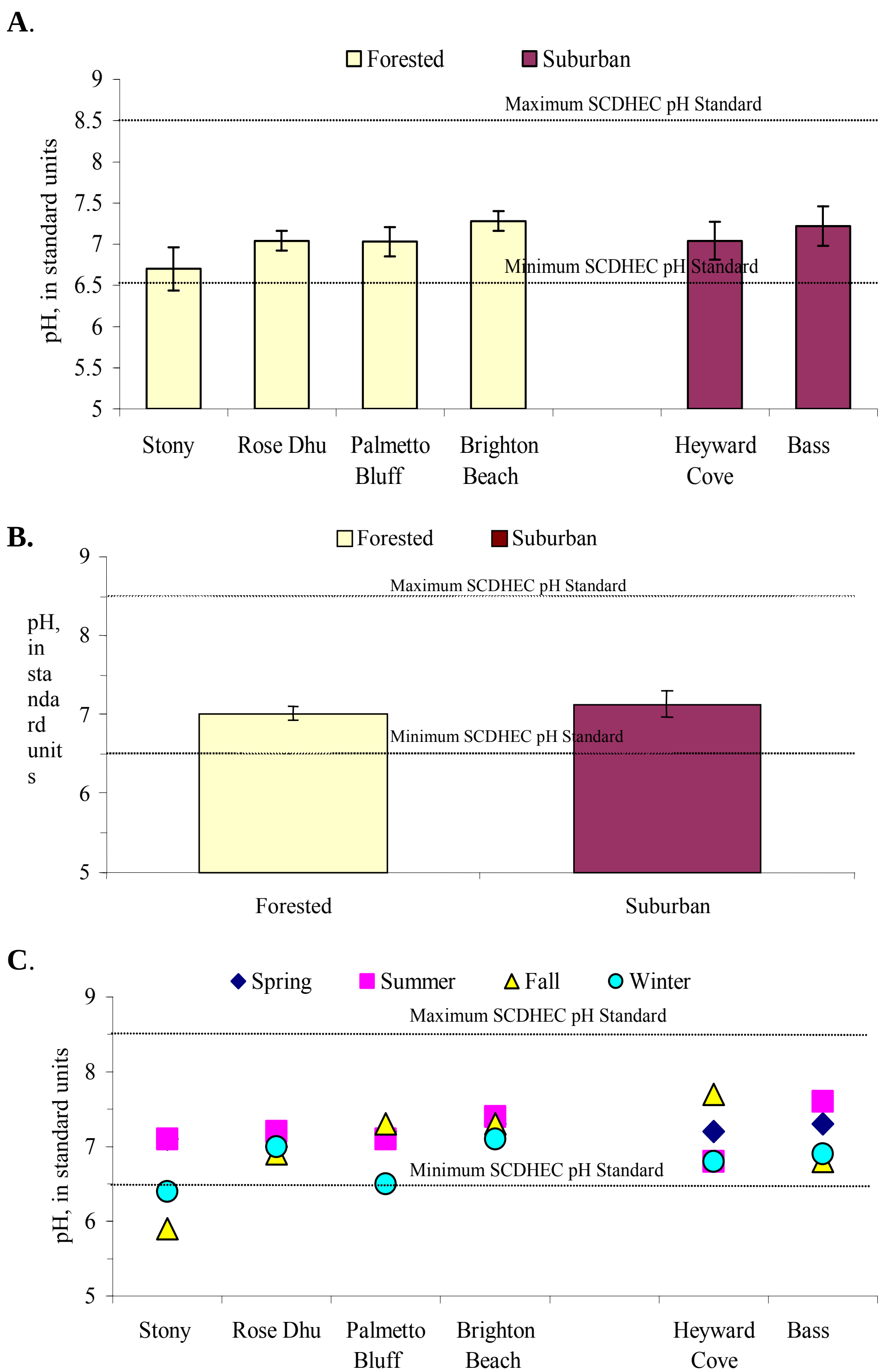


Figure III-7. Mean pH levels among sites (A.) and between forested and suburban land use (B.) and seasonal variation in pH levels (C.) at 6 headwater tidal creek sites in the May River 2002 – 2003. Error bars represent 1 standard error.

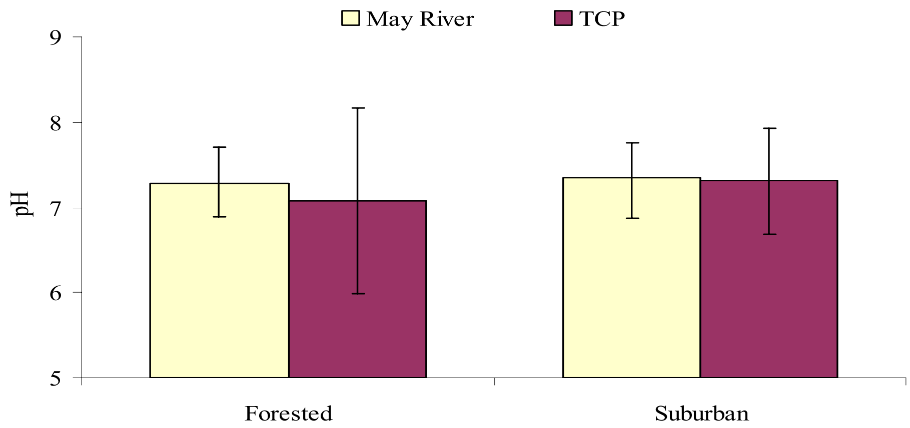


Figure III-8. Semi-continuous mean pH concentrations between forested and suburban creeks in the May River 2002 and TCP creeks sampled during the summer of 1994. Error bars represent 1 standard error.

tidal creeks and were not significantly different from those recorded in Charleston Harbor tidal creeks during 1994 (p-value = 0.2997, Figure III-8).

Biochemical Oxygen Demand

DO concentrations can be influenced by the amount of organic or reduced-nitrogen (ammonia, nitrite) compounds in the water. Biochemical oxygen demand (BOD) is an estimate of the potential depletion of DO from these compounds. These compounds can have both natural and man-made sources. BOD is reported as the amount of DO, in mg/L, depleted from a sample when incubated at 20 ^{o}C over a 5-day period.

Mean BOD ranged from 1.9 mg/L at Palmetto Bluff Creek to 3.2 mg/L at Brighton Beach Creek (Figure III-9) over the one-year seasonal sampling effort. Brighton Beach Creek was also identified as having the lowest mean DO concentrations. No statistical difference in BOD among headwater tidal creeks was identified. Although the lowest mean BOD was observed at Palmetto Bluff Creek (undeveloped, reference creek), no statistical difference in BOD existed between forested and suburban watersheds (p-value = 0.5812; Appendix III-5; Figure III-9).

Similar to DO concentrations, BOD also exhibited significant seasonal variation (p-value = 0.0360; Appendix III-5). BOD in the spring was significantly higher than winter. Spring BOD ranged from 1.1 mg/L at Palmetto Bluff Creek to 7.1 mg/L at Heyward Cove Creek (Figure III-9). Winter BOD were below 2 mg/L at all sites.

Increased concentrations of particulate and total forms of nutrients and carbon were associated with an increased BOD in headwater creeks. The strongest associations with BOD were with particulate forms of nitrogen, TSS, total nitrogen and phosphorus, and turbidity (Appendix III-5). Specific conductance (but not salinity) was also significantly correlated with BOD. The significant correlation between BOD and turbidity suggests that the more readily measured parameter, turbidity, may provide for a surrogate indicator for BOD changes in the headwater tidal creeks in the May River.

Turbidity

Turbidity is a measurement that quantifies the degree to which light traveling through a water sample is scattered by the suspended organic (including algae) and inorganic particles. Turbidity is measured in Nephelometric Turbidity Units (NTU). As the suspended, particulate load increases, light scatter increases such that clear water has a low turbidity (low light scattering ability) and cloudy or murky water has a high turbidity (high light scattering ability).

This relationship between turbidity and suspended particles makes turbidity a potentially useful surrogate parameter for estimating levels of total suspended solids. SCDHEC has recently developed a maximum saltwater State standard for turbidity of 25 NTU. This value corresponds to the 90th percentile of all estuarine turbidity levels in the SCDHEC saltwater database. This database includes data predominately from larger estuarine systems and does not include turbidity data from headwater tidal creeks, which are known to be more turbid, in general.

The suburban creek, Heyward Cove Creek, was the only headwater tidal creek site that consistently met the state saltwater turbidity SCDHEC standard of 25 NTU. Mean turbidity ranged from 16.7 NTU at Heyward Cove to 355 NTU at Brighton Beach with statistically significant differences among sites (Figure III-10).

Although a significant difference in turbidity levels between forested and suburban sites was determined with the ANOVA test (p-value = 0.0400; Appendix III-5), the difference between the mean turbidity at the forested sites (168 NTU) and suburban sites (42 NTU) was less than the minimum significant difference level for the Scheffe test. Small sample sizes (n = 4, 2) and large ranges in the data can be a cause of this problem. No seasonal differences in turbidity were identified for headwater tidal creeks (Figure III-10).

Several strong, statistically significant positive correlations were identified between turbidity and selected water quality parameters in the headwater tidal creeks. The strongest correlation was between turbidity and total suspended solids. These parameters are both measures of the concentration of suspended particles in the water at the time of sampling. The other parameters strongly associated with turbidity included particulate forms of nitrogen and carbon, total nitrogen, total Kjeldahl nitrogen (TKN), ammonia, total phosphorus, and BOD. This strong association of turbidity and particulate-related parameters listed above implies that turbidity has a strong potential to be a surrogate for concentrations of particulate-related parameters. This implication is important for two reasons: (1) turbidity is quickly and easily measured; and (2) turbidity can be measured continuously. As a surrogate, turbidity could provide data on event-driven, seasonal, and long-term changes in the concentrations of particulate-related parameters.

Nutrients

The amount of biologically available nutrients, mainly phosphorus and nitrogen compounds, is an important factor in ecosystem health, including estuaries. Nutrient enrichment in a creek or estuary accelerates the process of eutrophication, which results in excessive growth of algae (algal blooms) and other aquatic plants. Nuisance algal production causes a variety of associated water quality problems, including increased BOD concentrations, extreme fluctuations in DO concentrations (from oversaturated to depleted), high turbidity, and unpleasant odors. Some algal blooms can produce toxins that are harmful to aquatic biota and humans.

Nitrogen naturally occurs in several different forms, including dissolved molecular nitrogen (N_2), organic compounds (amino acids, amines, proteins, refractory humic compounds, and dissolved and total organic carbon), and inorganic compounds (ammonia, nitrate, and nitrite). TKN is the sum of total organic nitrogen and total or dissolved ammonia forms.

Like nitrogen, phosphorus exists in the aquatic environment in several forms, including inorganic and organic species. The most important form of inorganic phosphorus is orthophosphate (PO_4^{-3}), which is the form of phosphorus that is available for use by biota

(Wetzel, 1983). Phosphorus tends to adsorb strongly to particles in soils, suspended solids, and bed sediments.

Mean total nitrogen concentrations in the headwater tidal creeks in the May River ranged from 0.78 mg/L at Heyward Cove Creek to 3.47 mg/L at Brighton Beach Creek (Figure III-11); but were not significantly different among creeks for the 2002 to 2003 sampling period (p-value = 0.1223; Appendix III-5). Statistical analysis of the total nitrogen concentrations identified forested creek concentrations (2.26 mg/L) to be significantly higher than suburban creek concentrations (1.04 mg/L) (p-value = 0.0155) (Figure III-11). Brighton Beach Creek had the greatest seasonal range in total nitrogen concentrations; while Stony, Rose Dhu, and Heyward Cove creeks had the least seasonal range (Figure III-11). No statistically significant seasonal differences were observed for total nitrogen (p-value = 0.2634).

In addition to the strong correlations with turbidity and BOD described previously, total nitrogen concentrations were strongly associated with TSS, TKN, total particulate nitrogen; total phosphorus; and total particulate carbon (Appendix III-5). Of all the water quality parameters included in the correlation analysis, total nitrogen was most closely associated with TKN resulting in a one-to-one ratio, which implies that TKN accounts for the majority of the total nitrogen in these systems. Total nitrogen is calculated from the sum from TKN and NO_x.

The mean percentage of dissolved nitrogen fractions ranged from 45% at Brighton Beach Creek to 82% at Heyward Cove Creek. Mean dissolved nitrogen concentrations ranged from 0.58 mg/L at Heyward Cove to 1.25 mg/L at Stony Creek (Figure III-12). Creeks that drained forested watersheds had significantly higher (p-value = 0.0317, Appendix III-5; Figure III-12) dissolved nitrogen than suburban creeks (1.00 and 0.74 mg/L, respectively). Summer dissolved nitrogen concentrations were generally higher than or equal to the concentrations in the other seasons for the headwater tidal creeks (Figure III-12); however, no seasonal differences in dissolved nitrogen were identified (p-value = 0.1325).

Although nutrient criteria for estuarine systems are not established by SCDHEC, NOAA established ranges of dissolved nitrogen concentrations that represented high, medium, and low levels based on larger water bodies: High > 1 mg/L, Medium 0.1 to 1 mg/L, Low < 0.1 mg/L (Bricker and others 1999). In comparison to these criteria, Stony and Rose Dhu had high levels of dissolved nitrogen in 75% of their samples and Palmetto Bluff, Brighton Beach, and Bass had high levels in 25% of their samples (Figure III-12). All remaining samples had dissolved nitrogen concentrations in the medium range.

The headwater tidal creeks had mean TKN concentrations that were similar in magnitude, distribution, and statistical analyses to total nitrogen concentrations among sites, land use classes, and seasons, respectively (Figure III-13), indicating that TKN was the predominate form of nitrogen in the headwater tidal creeks. As with total nitrogen concentration, TKN concentration was significantly higher (p-value = 0.0120; Appendix III-5) in forested creeks (2.26 mg/L) than suburban creeks (1.00 mg/L).

Significant correlation results between total nitrogen and total organic nitrogen of 0.9362 and between total nitrogen and ammonia of 0.4780 suggested that organic forms of nitrogen, rather than ammonia, was the predominant form were making up total nitrogen concentration (Appendix III-5). These findings are similar to the findings of recent and ongoing research at the larger SCECAP sites (Van Dolah and others, 2000).

Mean ammonia concentrations in the headwater tidal creeks ranged from 0.08 mg/L at Heyward Cove Creek to 0.50 mg/L at Rose Dhu Creek (Figure III-14). No statistical difference was identified among creeks (p-value = 0.2767) or between watersheds (p-value = 0.9029)

(Appendix III-5; Figure III-14). Ammonia concentrations in headwater creeks did not exceed any state criteria for the range in temperature, pH, and salinity measured during the period of sampling (SCDHEC, 2001). Even though the highest ammonia concentrations of 0.99, 1.21, and 1.14 mg/L occurred in Stony, Rose Dhu, and Bass creeks, respectively, during the summer months when temperatures were highest, these maximum levels were below the established criteria (Figure III-14). No seasonal differences in ammonia concentrations were identified (p-value = 0.1925).

In addition to the previously described correlations among ammonia and dissolved oxygen, turbidity, and total nitrogen, ammonia concentrations were associated significantly with TSS, total phosphorus, dissolved phosphorus, total particulate nitrogen, TIC, and alkalinity (Appendix III-5). These findings suggest that ammonia concentrations increased when the amount of particulate material, phosphorus, and inorganic carbon increased in the headwater tidal creeks in the May River during the period of sampling.

Mean nitrate plus nitrite concentrations were extremely low in comparison to the other forms of nitrogen, ranging from 0.010 mg/L at Palmetto Bluff Creek to 0.076 at Heyward Cove Creek (Figure III-15). No statistical difference was identified among creeks (p-value = 0.1147, (Appendix III-5), between watersheds (p-value = 0.3994), or among seasons (p-value = 0.2834) (Figure III-15). No significant correlations existed among nitrate plus nitrite and the other selected water quality parameters.

Mean total phosphorus concentrations at the headwater tidal creeks ranged from 0.15 mg/L at Heyward Cove Creek to 0.82 mg/L at Rose Dhu Creek; however, no significant differences existed among creeks (p-value = 0.2407) or between watersheds (p-value = 0.2979) (Appendix III-5; Figure III-16). However, mean total phosphorus concentrations were higher in forested creeks than suburban creeks. Bass, Rose Dhu, and Stony creeks had the greatest seasonal variability in total phosphorus concentrations, with maximum concentrations occurring in the summer months (Figure III-16). Heyward Cove Creek, a suburban creek, had the lowest range in total phosphorus concentrations. No statistical difference in total phosphorus concentrations among seasons was identified (p-value = 0.2894).

In addition to correlations among total phosphorus concentrations and turbidity, BOD, and all forms of nitrogen (except nitrate plus nitrite), total phosphorus concentrations were also associated with TSS, total particulate nitrogen and carbon, alkalinity, TIC, silica, chloride, and dissolved phosphorus (Appendix III-5). These associations among total phosphorus and particulate-related measures of turbidity, including TSS, particulate nitrogen and carbon suggests that the total phosphorus in headwater creeks tended to be particulate-bound. However, a strong association between dissolved and total phosphorus suggested that the dissolved forms were also important.

The headwater tidal creeks in the May River had mean dissolved phosphorus concentrations that ranged from 0.017 mg/L at Heyward Cove Creek to 0.300 mg/L at Bass Creek (Figure III-17). There was a significant difference among creeks (p-value = 0.0137; Appendix III-5), however, no specific differences were found in the Tukey or Scheffe tests, probably due to low sample size. In contrast to mean total phosphorus concentrations, mean dissolved phosphorus concentrations were slightly higher in suburban creeks than forested creeks (Figure III-17); but these differences were not significant (p-value = 0.0809; Appendix III-5). Consistent with total phosphorus concentrations, Bass, Stony, and Rose Dhu creeks had the greatest range in dissolved phosphorus concentrations with the highest concentrations occurring in the summer months (Figure III-17), although no seasonal difference were identified (p-value =

0.2649). Dissolved phosphorus was strongly correlated with the same parameters as total phosphorus.

Although nutrient criteria for estuarine systems have not been established by SCDHEC, NOAA established ranges of dissolved phosphorus concentrations that represented high, medium, and low levels: High > 0.1 mg/L, Medium 0.01 to 0.1 mg/L, Low < 0.01 mg/L (Bricker and others, 1999). Stony Creek had High levels of dissolved phosphorus in 75 % of its samples and Rose Dhu and Bass creeks had High levels in 50% of its samples (Figure III-17). The remaining samples were in the Medium and Low range.

In summary, TKN accounted for the greatest percentage of the total nitrogen in the majority of the samples at the headwater tidal creeks. Concentrations of all forms of nitrogen and total phosphorus were not statistically different among sites or among seasons for the period of sampling. Total and dissolved nitrogen and TKN concentrations were statistically different between land use classes, such that forested creeks were significantly higher than suburban creeks. Inorganic forms of nitrogen and total and dissolved phosphorus had similar concentrations between land use classes.

Total Organic Carbon

Measurements of total organic carbon (TOC) are used to indicate the amount of organic detritus and its decomposition by-products in the water column. Mean TOC concentrations ranged from 6.7 mg/L at Heyward Cove Creek to 21 mg/L at Stony Creek (Figure III-18). No significant difference was identified among headwater creeks (p-value = 0.4556) or between watershed classes (p-value = 0.2702), although TOC concentrations were higher in forested creeks than suburban creeks (Appendix III-5).

TOC exhibited significant seasonal variations (Figure III-18, Appendix III-5) and was highest in the fall (mean = 20.8 mg/L) and lowest in the spring (mean = 6.1 mg/L) and summer (mean = 6.0 mg/L). TOC in the fall ranged from 7.6 mg/L at Palmetto Bluff Creek to 40 mg/L at Stony Creek (Figure III-18).

As previously described, TOC was negatively correlated with salinity, indicating that fresh water from the upland watershed carries higher TOC than estuarine water. In addition, an almost one-to-one correlation existed between total and dissolved organic carbon.

Fecal Coliform

Fecal coliform bacteria are a group of bacteria used as indicators of fecal pollution. However, problems exist when fecal coliform bacteria are used as indicators of fecal pollution, especially if the source of the pollution needs to be determined. First, fecal coliform bacteria inhabit the gastrointestinal tracts of all warm-blooded and some cold-blooded animals, and therefore, provide no information about the specific source of fecal contamination. Secondly, studies in subtropical coastal areas similar to the South Carolina coast have shown that *Escherichia coli* (part of the fecal coliform bacteria group) and enterococci (another commonly used fecal indicator bacteria) occur and proliferate in soil and natural vegetation in the absence of fecal contamination (Lopez-Torrez and others, 1987; Fujioka and Hardin, 1995; Ashbolt and others, 1997; Byapparahalli and Fujioka, 1998; Solo-Gabriele and others, 2000; Desmarais and others, 2002).

In saltwater environments, SCDHEC uses the number of fecal coliform bacterial colonies for recreational water quality standards to determine the extent of fecal contamination. Recreational standards for primary (whole body) contact, such as swimming, are a geometric mean that does not exceed 200 colony forming units per 100 milliliters (CFU/100 mL) based on 5 consecutive samples in a 30-day period and no more than 10 percent of the samples can exceed 400 CFU/100 mL (SCDHEC 2001). More stringent standards are established for waters related to shellfish harvesting are a geometric mean of 14 CFU/100 mL and less than a 10 percent exceedence level of 43 CFU/100 mL. The sampling frequency for the May River study prevents the application of the fecal coliform data to determine compliance with these standards but the data can be compared to the standards as a screening tool.

For the May River study, only one headwater tidal creek sample was collected for fecal coliform bacteria analysis per season. Mean fecal coliform concentrations ranged from 339 CFU/100 mL at Bass Creek to over 6,100 CFU/100 mL at Brighton Beach Creek for the one-year period of study (Figure III-19). However, no significant differences in fecal coliform concentrations were identified among headwater tidal creeks in the May River (p-value = 0.5048, Appendix III-5). Even Palmetto Bluff Creek, a site that represented undeveloped, forested land use, had a mean fecal coliform concentration (2,100 CFU /100 mL) well above the 400 CFU/100 mL exceedence standard for the period of study. In fact, the mean fecal coliform concentrations for suburban sites were lower than forested sites (Figure III-19), but this difference was not significant (p-value = 0.1774). Stony and Heyward Cove creeks exceeded the 400 CFU/100 mL standard in 25 percent of the samples (1 of 4); Palmetto Bluff and Bass creeks, 50 percent of the samples (2 of 4); Brighton Beach Creek, 75 percent of the samples (3 of 4); and Rose Dhu Creek, 100 percent of the samples (Figure III-19). Brighton Beach and Palmetto Bluff creeks had the highest variation in fecal coliform concentrations; Rose Dhu Creek had the lowest variation among seasons. There was a significant difference among seasons (p-value = 0.0494), however, no specific differences were found in the Tukey and Scheffe tests, probably due to low sample size and large variability (Appendix III-5).

<u>Fecal Coliform/MAR</u>

Fecal coliform densities and Multiple Antibiotic Resistance (MAR) analyses for the six headwater creeks were conducted by NOAA during the summer of 2002 (Table III-4). All of the headwater tidal creek sites exhibited fecal coliform counts greater than 14 CFU/100 mL (the standard for SC waters approved for shellfish harvesting, (SCDHEC, 2001) and 4 of the 6 HWTC sites exhibited fecal coliform counts greater than 200 CFU/100 ml (the standard for SC waters approved for contact recreation, SCDHEC, 2001). Fecal coliform counts ranged from 46 to 14,000 CFU/100 mL. The highest fecal coliform concentrations occurred at Brighton Beach Creek. In general, the forested creeks of the May River had higher fecal coliform concentrations than the forested creeks in the TCP, however the suburban creeks of the May River had lower concentrations than those in the TCP (Figure III-20).

Only one of the six headwater tidal creeks, Stony Creek, showed the presence of any antibiotic resistance (Table III-4). Since the 95% confidence intervals for the MAR indices at Stony Creek included zero, this creek was not considered to exhibit antibiotic resistance. An earlier study conducted in the Broad Creek and Okatee River showed that the undeveloped region of the Okatee River had an overall MAR Index of 1% in 1997 (Van Dolah and others, 2000), which is similar to the present overall MAR Index for May River (0.64%). Based on the interpretation of these MAR results, the coliform levels at the sites sampled did not appear to be related to anthropogenic sources of contamination.

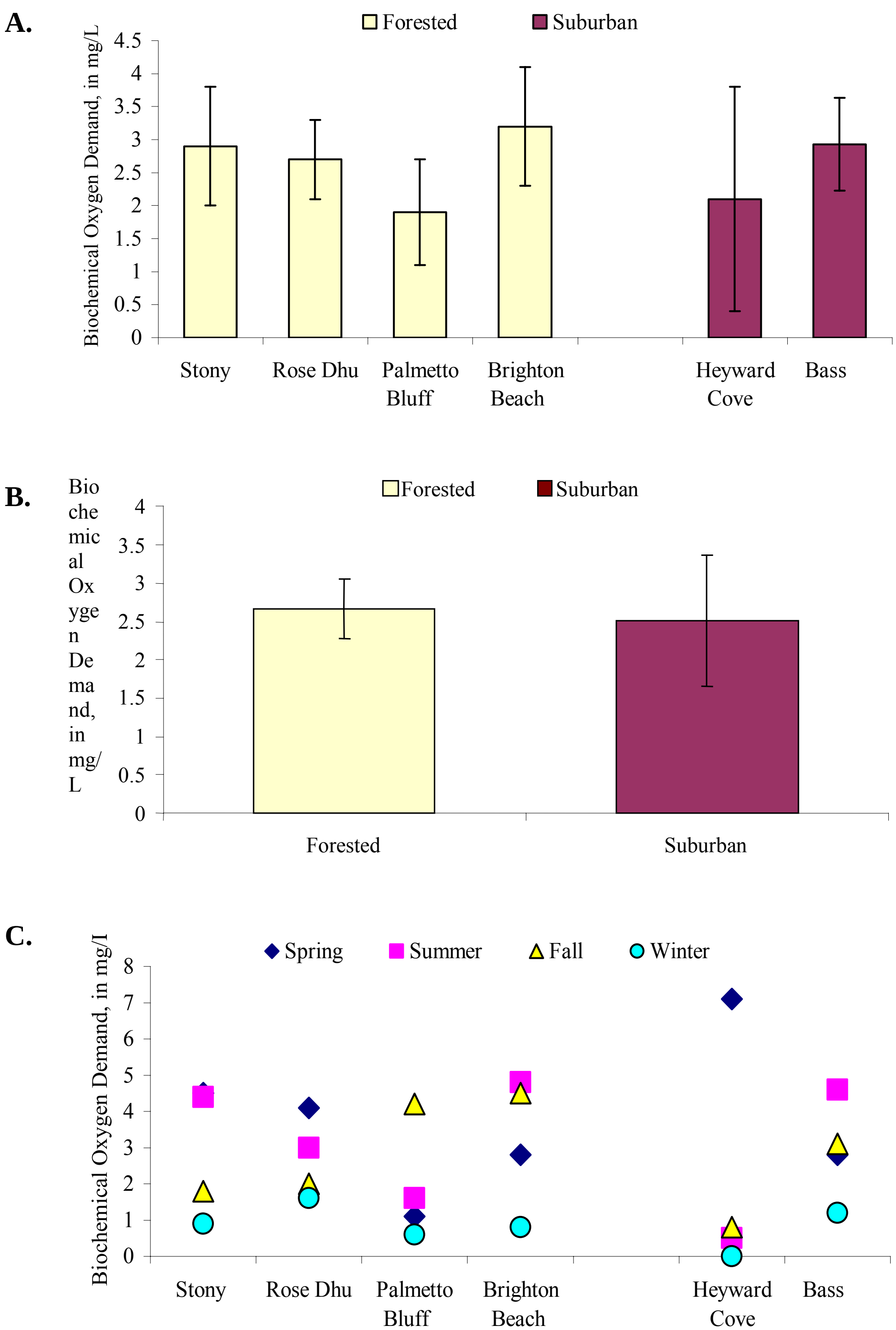


Figure III-9. Mean biochemical oxygen demand concentrations among sites (A.) and between forested and suburban land use (B.) and seasonal variation in biochemical oxygen demand (C.) at 6 headwater tidal creek sites in the May River, 2002 - 2003. Error bars represent 1 standard error.

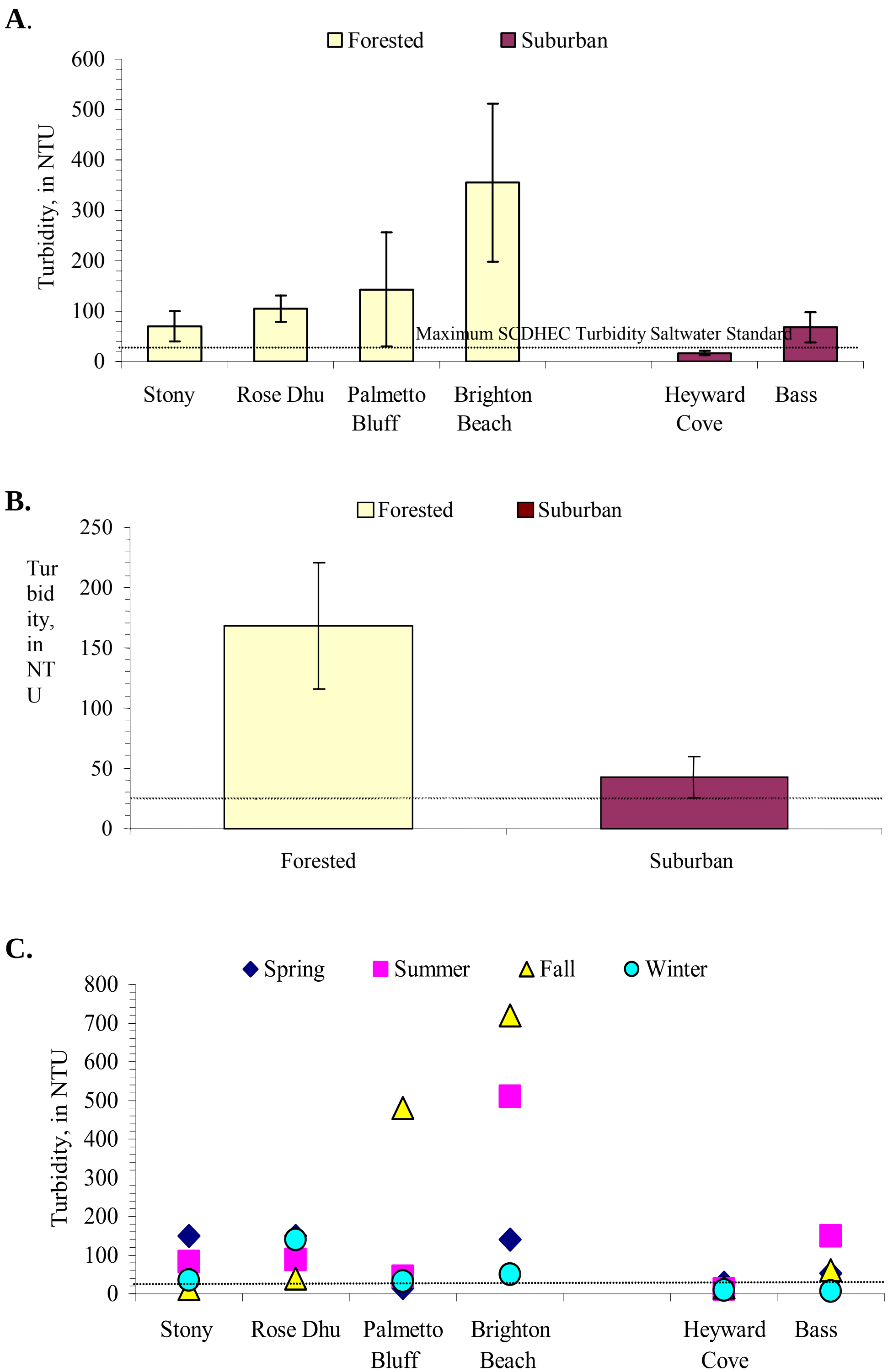


Figure III-10. Mean turbidity levels among sites (A.) and between forested and suburban land use (B.) and seasonal variation in turbidity levels (C.) at 6 headwater tidal creeks in the May River, 2002 – 2003. Error bars represent 1 standard error.

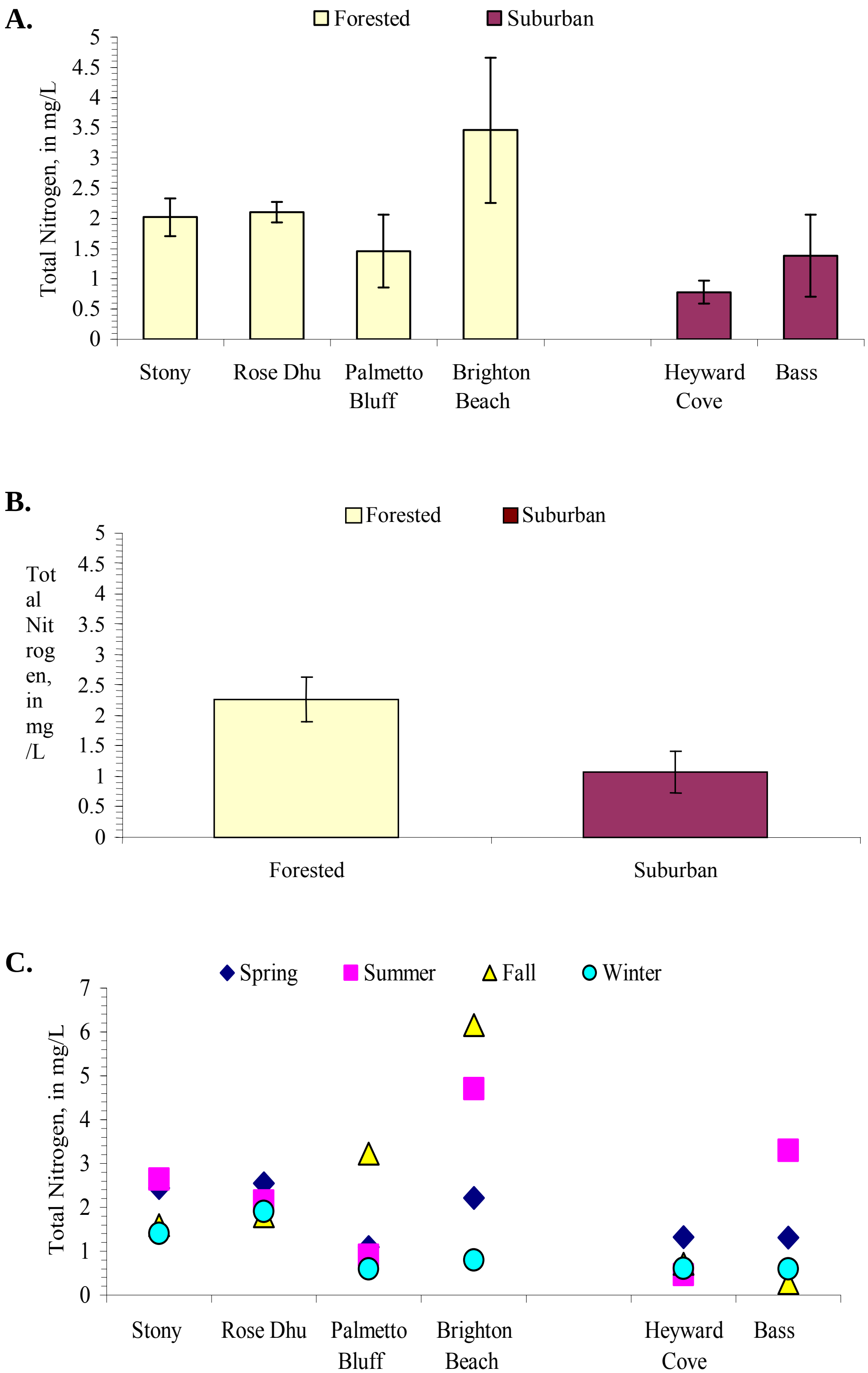


Figure III-11. Mean total nitrogen concentrations among sites (A.) and between forested and suburban land use (B.) and seasonal variation in total nitrogen concentrations (C.) at 6 headwater tidal creeks in the May River, 2002 – 2003. Error bars represent 1 standard error.

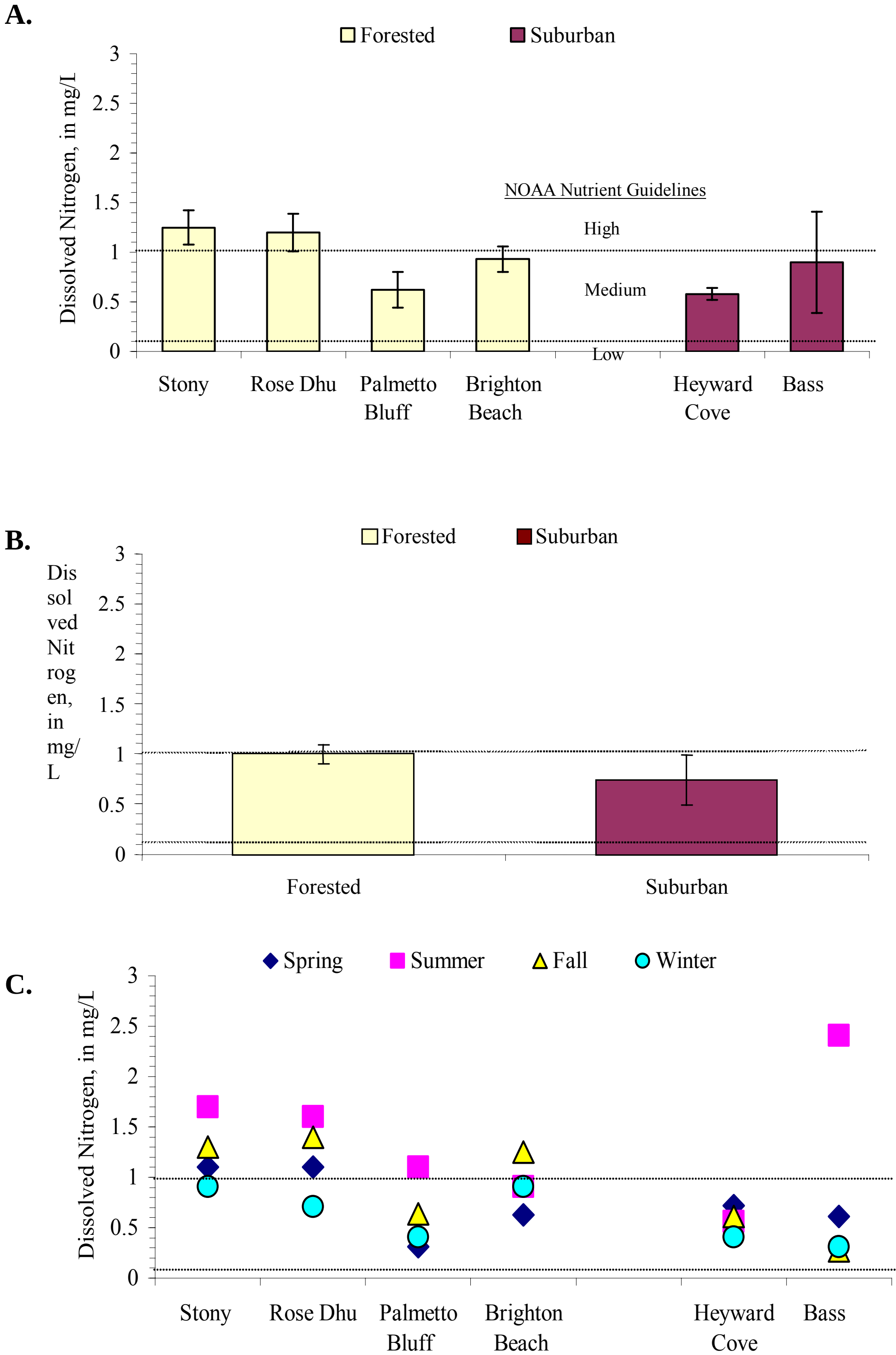


Figure III-12. Mean dissolved nitrogen concentrations among sites (A.) and between forested and suburban land use (B.) and seasonal variation in dissolved nitrogen concentrations (C.) at 6 headwater tidal creeks in the May River, 2002 – 2003. Error bars represent 1 standard error.

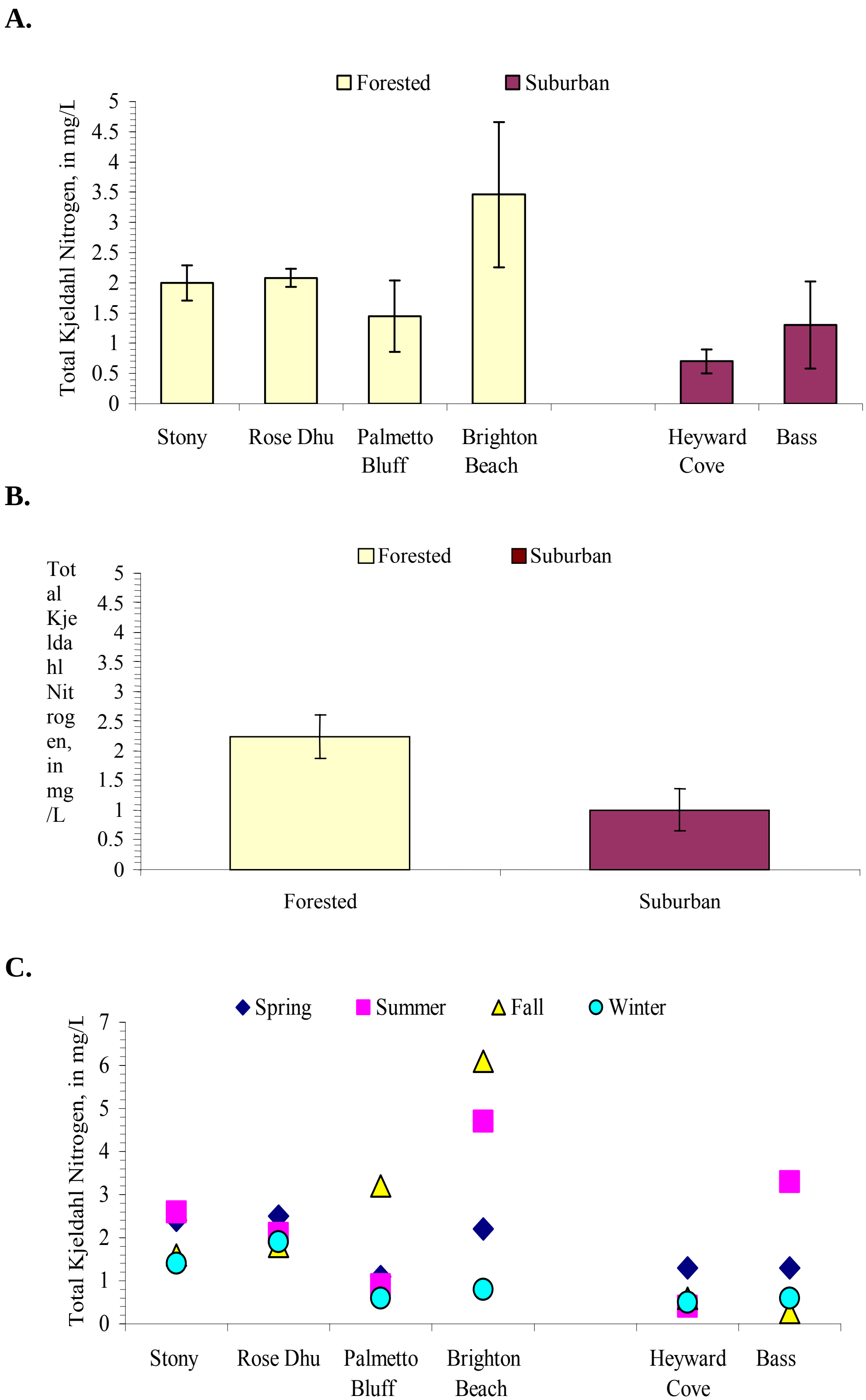


Figure III-13. Mean total Kjeldahl nitrogen concentrations among sites (A.) and between forested and suburban land use (B.) and seasonal variation in total Kjeldahl nitrogen concentrations (C.) at 6 headwater tidal creeks in the May River, 2002 – 2003. Error bars represent 1 standard error.

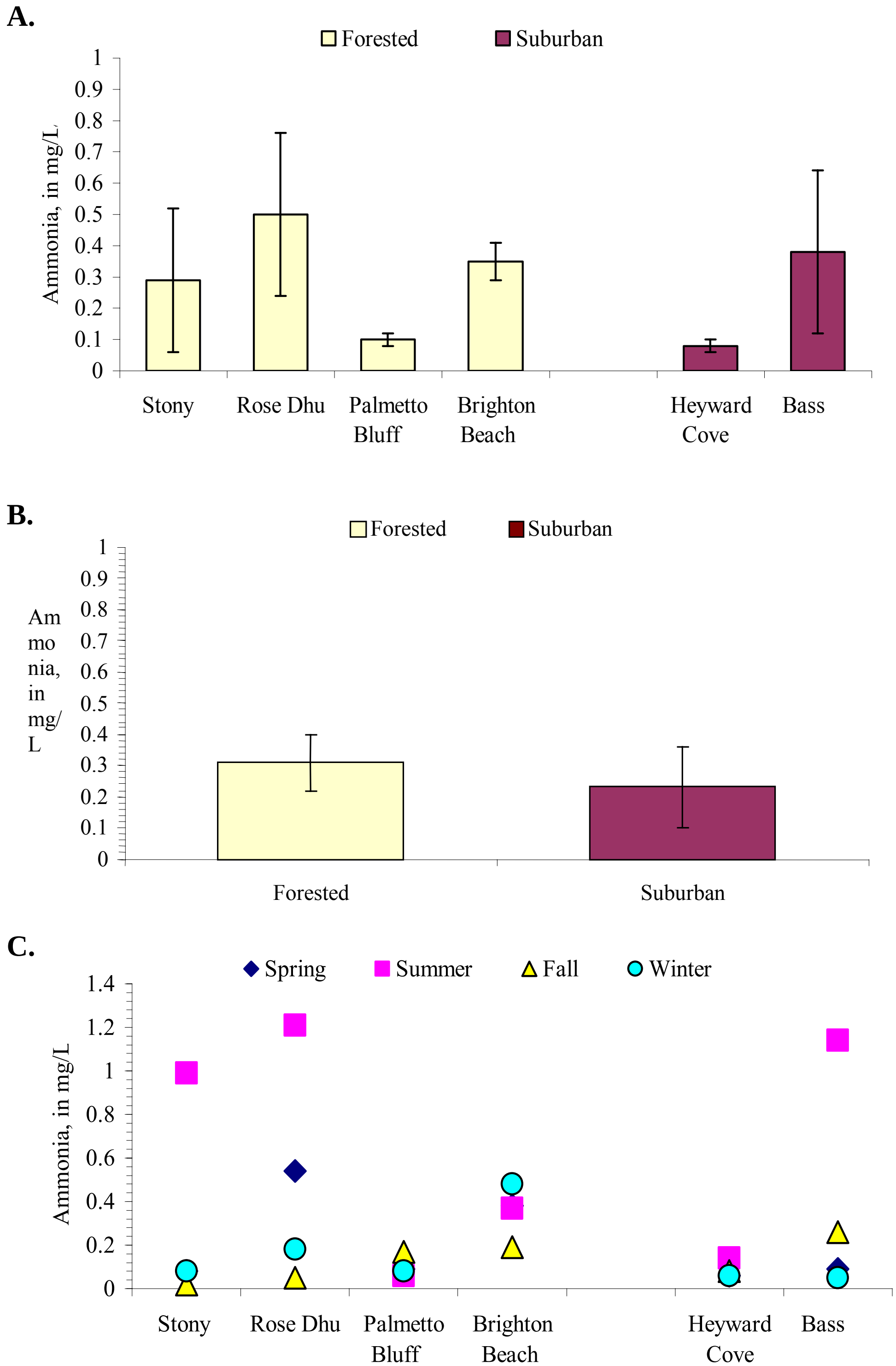


Figure III-14. Mean ammonia concentrations among sites (A.) and between forested and suburban land use (B.) and seasonal variation in ammonia concentrations (C.) at 6 headwater tidal creeks in the May River, 2002 – 2003. Error bars represent 1 standard error.

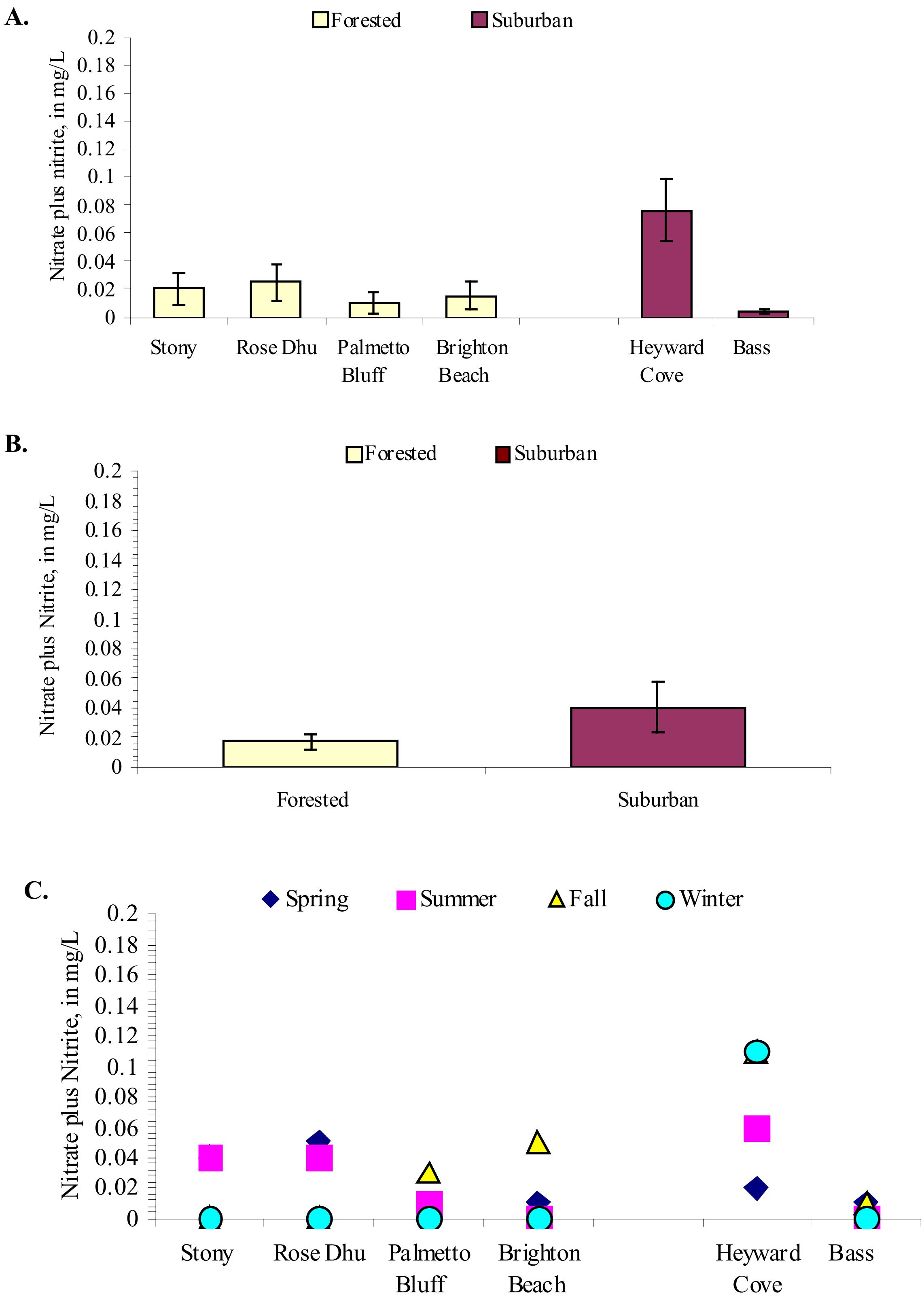


Figure III-15. Mean nitrate plus nitrite concentrations among sites (A.) and between forested and suburban land use (B.) and seasonal variation in nitrate plus nitrite concentrations (C.) at 6 headwater tidal creeks in the May River, 2002 – 2003. Error bars represent 1 standard error.

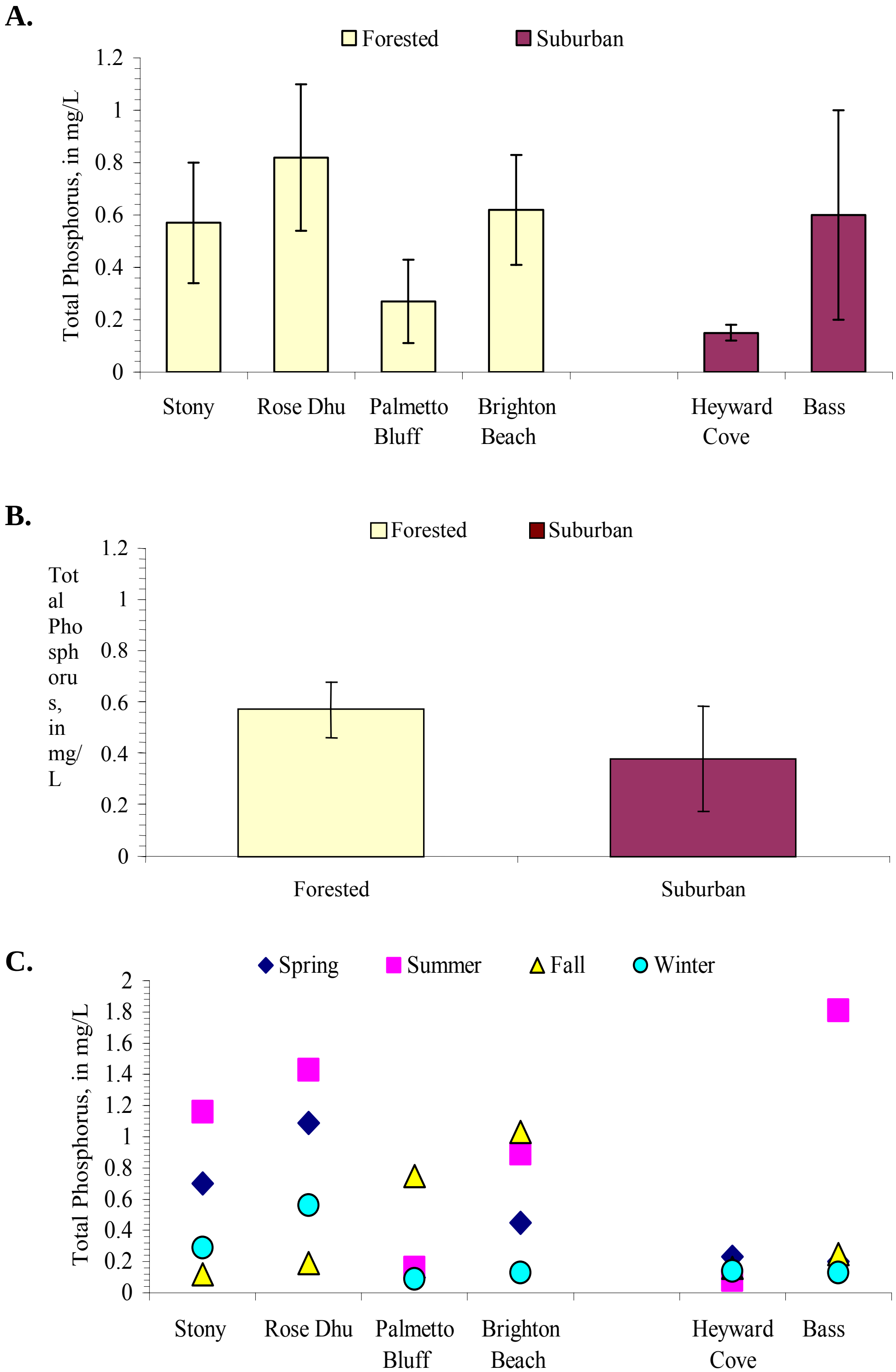


Figure III-16. Mean total phosphorus concentrations among sites (A.) and between forested and suburban land use (B.) and seasonal variation in total phosphorus concentrations (C.) at 6 headwater tidal creeks in the May River, 2002 – 2003. Error bars represent 1 standard error.

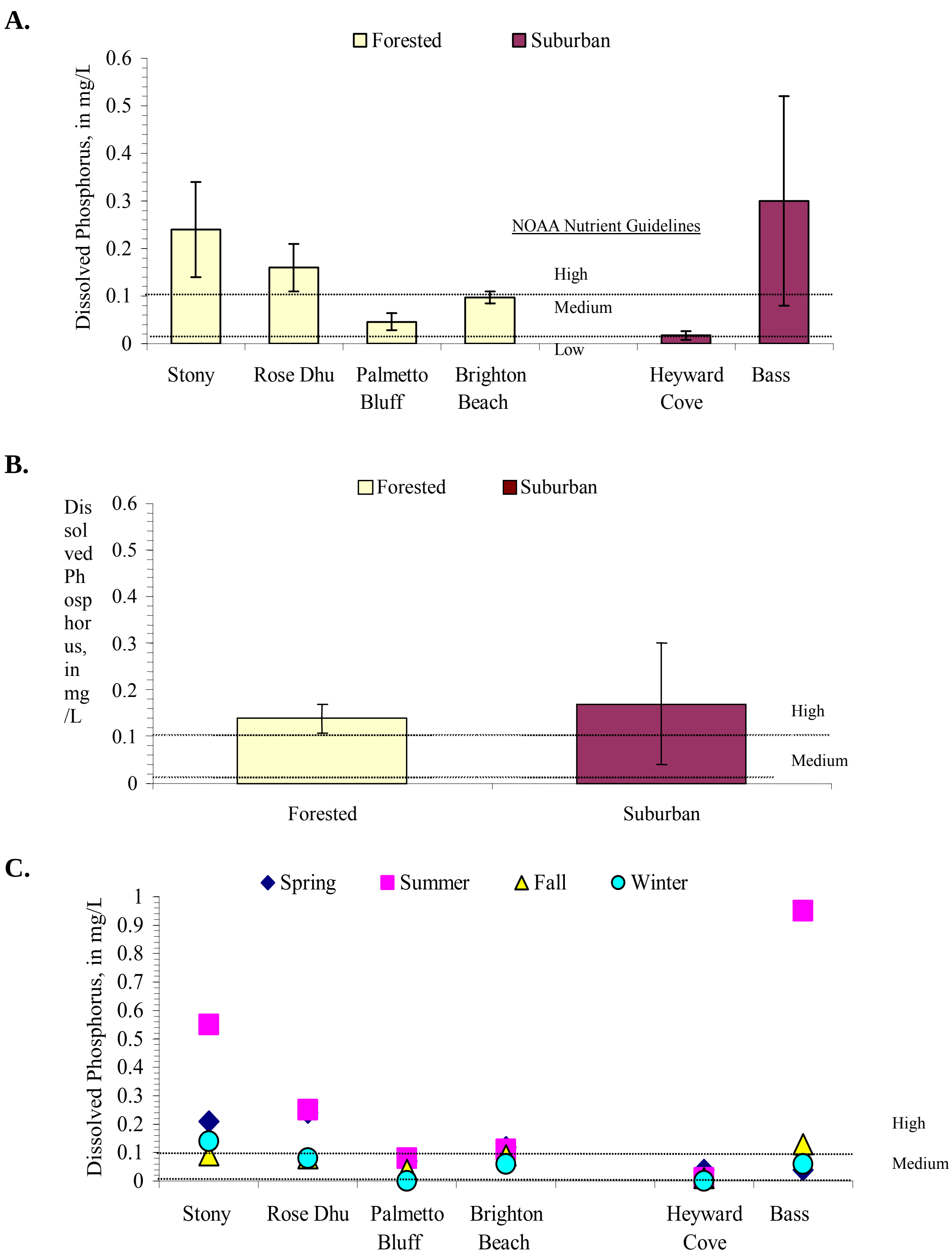


Figure III-17. Mean dissolved phosphorus concentrations among sites (A.) and between forested and suburban land use (B.) and seasonal variation in dissolved phosphorus concentrations (C.) at 6 headwater tidal creeks in the May River, 2002 – 2003. Error bars represent 1 standard error.

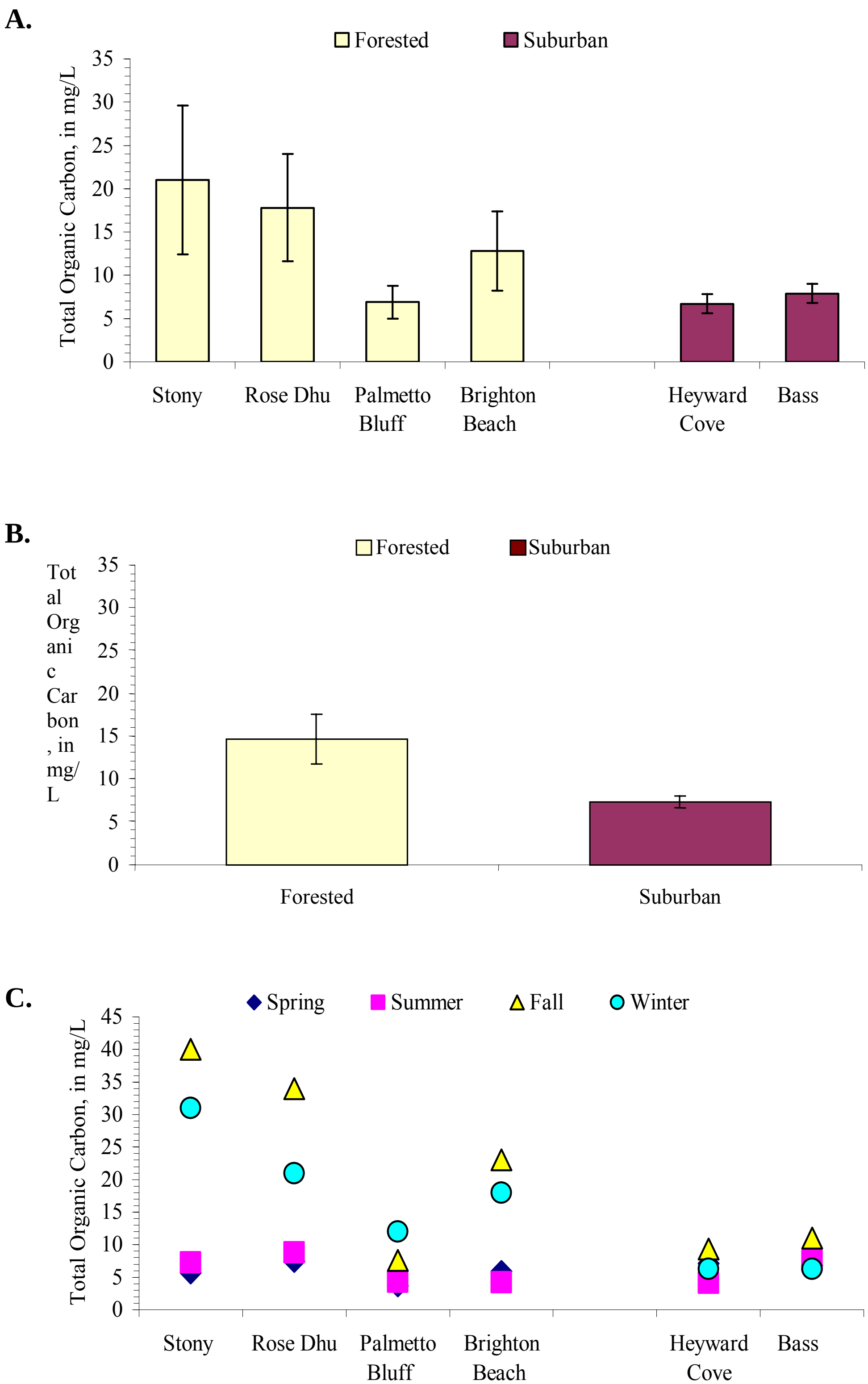


Figure III-18. Mean total organic carbon concentrations among sites (A.) and between forested and suburban land use (B.) and seasonal variation in total organic carbon concentrations (C.) at 6 headwater tidal creeks in the May River, 2002 – 2003. Error bars represent 1 standard error.

Wastewater Indicators

Because of the inherent problems with fecal coliform bacteria, the USGS used an alternative method to identify potential of human fecal contamination. The approach consisted of analyzing samples for the presence of chemical indicators of human wastewater concurrently with fecal coliform bacteria sampling from Palmetto Bluff and Heyward Cove creeks, which drain forested and suburban watersheds, respectively. Wastewater indicators can be placed into groups that include (1) food or digestive by-products, such as cholesterol, caffeine and pharmaceuticals that pass through human systems; (2) fragrances commonly found in personal care products; (3) detergent agents commonly found in cleaning solutions and soaps; (4) pesticides commonly contributed from residential and agricultural areas; and (5) urban runoff compounds contributed from parking lots, roads, and commercial areas. A "cluster approach" was applied, whereby elevated fecal coliform concentrations in conjunction with detections of multiple compounds and/or groups of compounds, especially those directly associated with domestic wastewater (i.e., caffeine, fragrances, detergent agents) were considered a good indication of a human wastewater source to the fecal coliform bacteria in the watershed.

Samples for wastewater analysis were collected in the fall (October 30-31, 2002) and in the winter (March 11 -13, 2003). In the fall, Palmetto Bluff Creek, a forested creek, had a fecal coliform concentration of 4,930 CFU/100 mL, which was well above all SCDHEC recreational and shellfish harvesting standards. Heyward Cove Creek, a suburban creek, had a fecal coliform concentration of 248 CFU/100 mL, which, although lower still exceeds the 200 CFU/100 mL (geometric mean) recreational standard (Figure III-19) (SCDHEC 2001). Concurrently, wastewater indicator analysis in the fall detected two compounds in Palmetto Bluff Creek and Heyward Cove Creek (Appendix III-2). The two wastewater indicators in Palmetto Bluff Creek were beta-sitosterol (a plant sterol derived from either the digestion of vegetable matter or decomposition of organic debris) and bisphenol A (a manufactured polycarbonate resin and antioxidant found in urban runoff and municipal effluent). In Heyward Cove Creek, the two wastewater indicators were the pesticides atrazine (a widely used herbicide) and N,N-diethyltoluamide or DEET (a widely used mosquito repellant). All indicators were detected at levels below the laboratory reporting limit, and therefore were classified as estimated values. These compounds are more indicative of compounds derived in urban runoff (bisphenol A, atrazine, DEET) and marsh drainage (beta-sitosterol) than human wastewater and suggests that human wastewater was not the source of the fecal coliform concentrations in either Palmetto Bluff or Heyward Cove creeks.

In the winter samples, Palmetto Bluff Creek had a fecal coliform concentration of 96 CFU/100 mL and Heyward Cove Creek had a higher fecal coliform concentration of 1,000 CFU/100 mL (Figure III-19). Wastewater indicator analysis detected no compounds in Palmetto Bluff Creek, but detected 34 compounds in Heyward Cove (Appendix III-2). These indicators consisted of ten detergent agents (including nonylphenols, tri-phosphates, triclosan), three fragrances or additives in personal care products (benzophenone, ethyl citrate, galaxolide or HHCB), two food by-products (caffeine, cholesterol) and nine urban runoff compounds, including a variety of polycyclic aromatic hydrocarbons found in diesel and gasoline (benzo(a)pyrene, fluoranthene, naphthalene, pyrene, and phenathrene) (Appendix III-2). All indicators were detected at levels below the laboratory reporting limit and therefore were classified as estimated values.

Table III-4. Fecal coliform densities and MAR indices from headwater tidal creeks. The number in parentheses is the standard error.

Site	Fecal Coliform /100 ml	Fecal Coliform /100 ml (log10)[a]	Site MAR Index
Stony	1080	3.03	0.03 (0.03)
Rose Dhu	909	2.96	0
Palmetto Bluff	1364	3.13	0
Heyward Cove	58	1.76	0
Brighton Beach	14000	4.15	0
Bass	46	1.66	0

Phytoplankton

In developing a basis for classifying the eutrophication status of U.S. estuaries, Bricker and others (1999) identified several categories of chlorophyll-*a* concentrations:

"Low" concentrations are < 5 μg/L.

"Medium" concentrations range between 5 and 20 μg/L.

"High" concentrations range between 20 and 60 μg/L.

"Hypereutrophic" concentrations are > 60 μg/L.

In the six May River headwater tidal creeks, the mean chlorophyll-*a* values ranged between 3.5 and 15.3 μg/L over the study period, which would be considered Low to Medium in terms of eutrophication status. Overall, no consistent pattern was found between forested and suburban creeks. When broken down by season (Figure III- 21), chlorophyll-*a* concentrations from three of the 24 samples were classified as High; Stony and Bass creeks in summer (38.3 ± 5.4 μg/L and 28.5 ± 3.8 μg/L, respectively), and Palmetto Bluff Creek in fall (31.6 ± 2.2 μg/L). The remaining samples were lower than 15 μg/L, and 11 of these were below 5 μg/L (i.e., in the Low class), which included most of the winter samples. With the exception of relatively low winter values, no consistent seasonal trend in chlorophyll-*a* concentration was apparent. The summer mean chlorophyll-*a* values in the May River headwater tidal creeks were similar to the summer mean chlorophyll-*a* values in the Okatee headwater tidal creeks (Figure III-21; Van Dolah and others, 2000).

Harmful algae were rare in the headwater creek samples (Appendix III-6a). The only known harmful species identified from these samples was the dinoglagellate *Kryptoperidinium foliaceum*, which was observed in the spring in Brighton Beach Creek, in both the spring and summer in Stony and Rose Dhu creeks, and in the summer in Bass Creek. *K. foliaceum* is the most common harmful alga in South Carolina tidal creeks, and is regularly found in upper tidal creeks as opposed to larger creeks and open estuaries (Lewitus and others, 2001). Based on temporal distributional patterns in North Inlet Estuary (near Georgetown, SC), Lewitus and

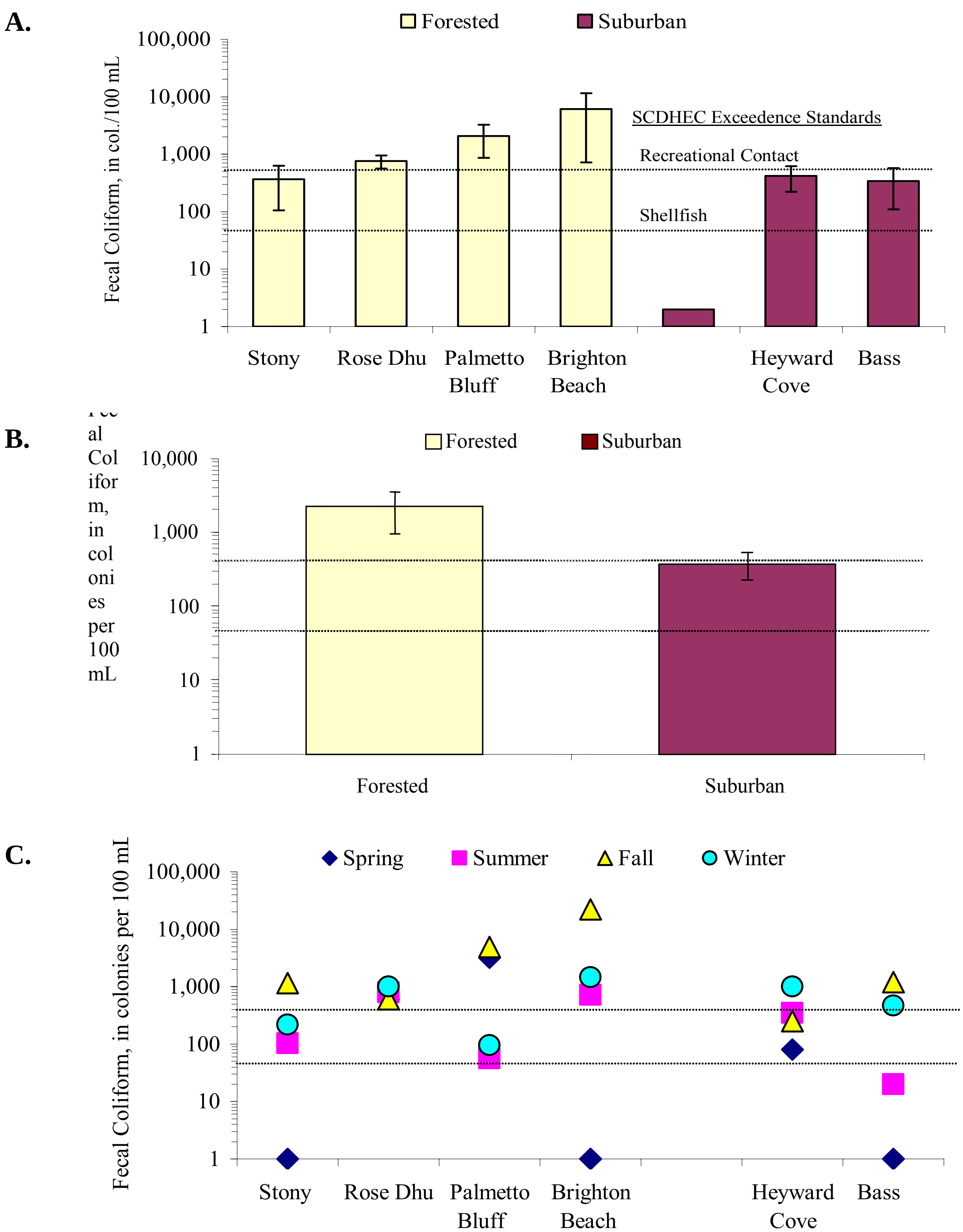


Figure III-19. Mean fecal coliform concentrations among sites (A.) and between forested and suburban land use (B.) and seasonal variation in fecal coliform concentrations (C.) at 6 headwater tidal creeks in the May River, 2002 – 2003. Error bars represent 1 standard error.

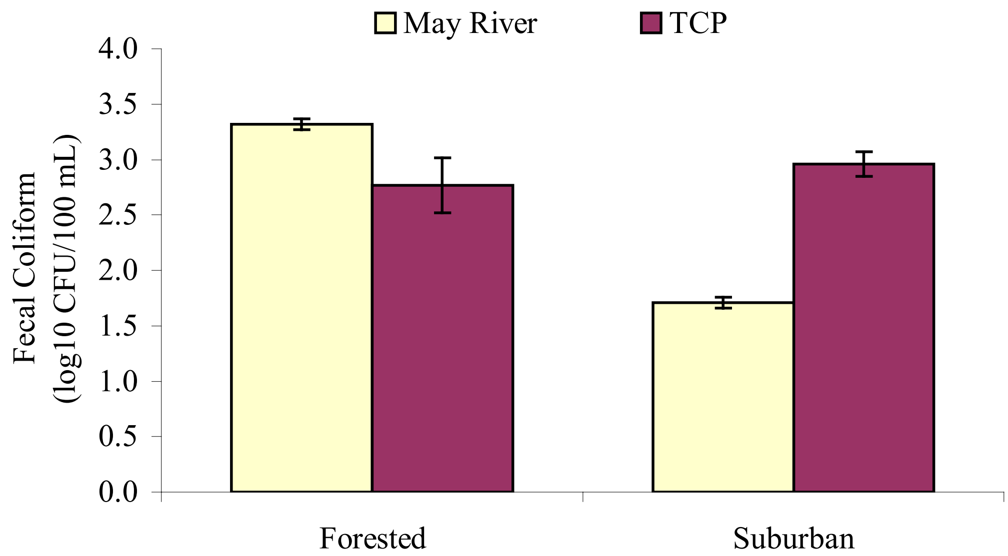


Figure III-20. Comparison of mean fecal coliform bacterial concentrations in forested and suburban creeks of the May River compared to the Tidal Creek Project (TCP).

Hayes (unpublished data) hypothesized that this species has the capability of maintaining populations in upper tidal creeks by aggregating on sediments on the ebbing tide. When it blooms, *K. foliaceum* can cause shellfish stress effects (Lewitus and others, 2003, Ringwood and Keppler, unpublished. data), and therefore the discovery of this species in several May River tidal creeks emphasizes the need to include this species in future monitoring plans.

When comparing the relative contribution of chemotaxonomic marker pigment concentrations to chlorophyll-*a* concentrations, no consistent differences were apparent between forested and suburban sites (Appendix III-6b). Seasonal patterns were not conspicuous, with the possible exceptions of the peridinin:chlorophyll-*a* and prasinoxanthin:chlorophyll-*a* patterns. Although peridinin was consistently low relative to other pigments, five of the six cases where concentrations were detectable occurred in the spring or summer, suggesting that this subset of dinoflagellates was more prevalent in warmer periods. At all sites, the highest relative contribution of prasinoxanthin-containing prasinophytes was highest in spring or summer. This subset of prasinophytes is also an important contributor to late spring-to-summer phytoplankton communities in North Inlet Estuary (Lewitus, unpublished data), a high salinity salt marsh estuary with little anthropogenic influence. The appearance of prasinoxanthin in North Inlet samples during the summer phytoplankton bloom coincides with a microbial food web that is influenced strongly by regenerated nutrients and grazing control. These autochthonous processes are more important than allochthonous effects (nutrient loading, turbidity) in regulating phytoplankton dynamics (Noble and others, 2003). We hypothesize that these prasinophytes may be indicative of warm temperature communities relatively unimpacted by anthropogenic influences.

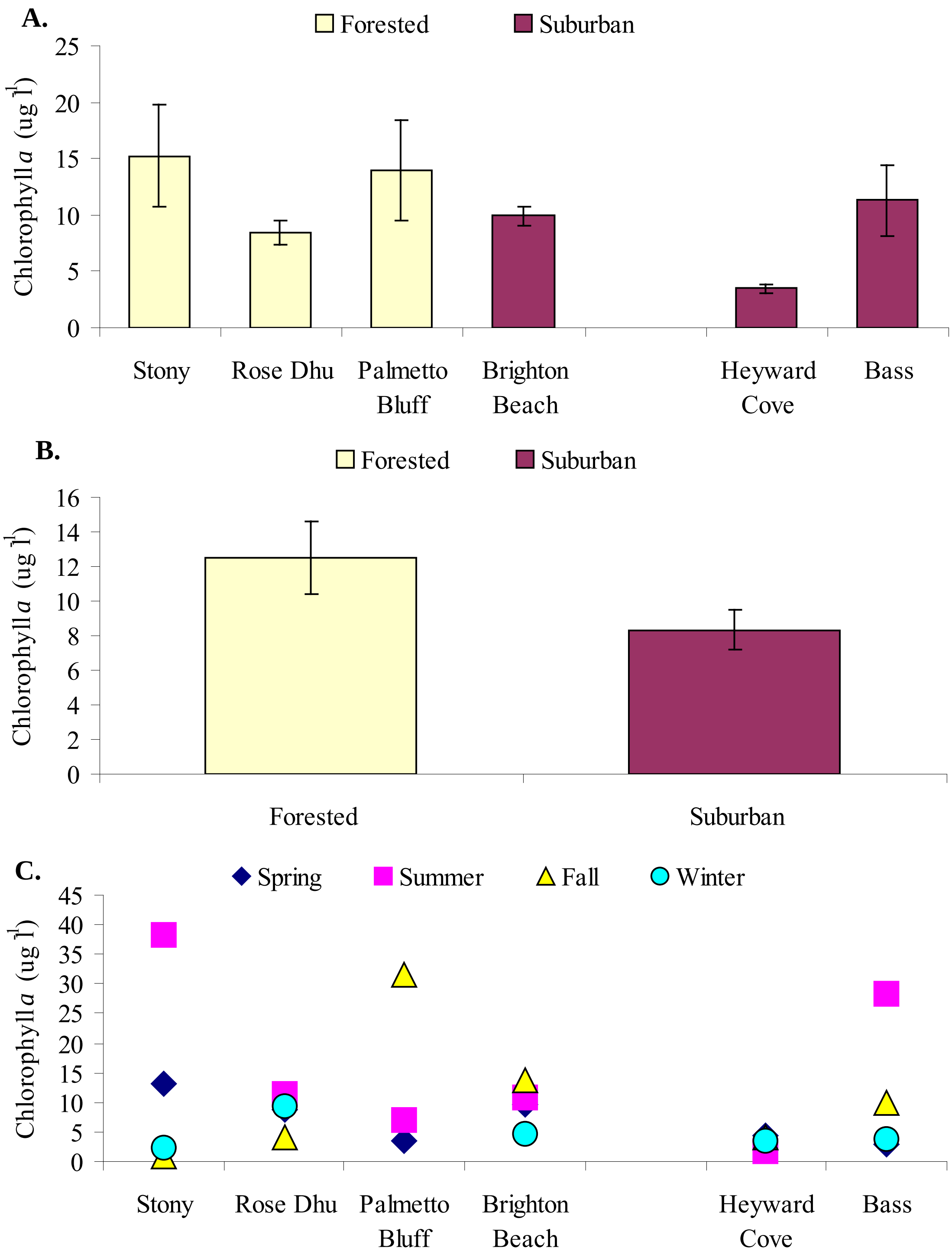


Figure III-21. Summary of chlorophyll-*a* concentrations among sites (A.) and between forested and suburban land use (B.) and seasonal variations of chlorophyll-*a* concentrations (C.) at 6 headwater tidal creeks in the May River, 2002 - 2003. Error bars represent 1 standard error.

Sediment Quality

Composition

Overall sediment quality included measures of porewater ammonia, benthic chlorophyll-*a*, general sediment characteristics (i.e., grain size, TOC), and sediment contaminants, which consisted of analytical measures of chemical concentrations as well as measures of bioeffects obtained through toxicity assays. Sediment characteristics are one of the driving forces that structure benthic communities and are also important to the patterns seen in the distribution of sediment contaminants. High levels of sediment contaminants are known to have adverse effects on benthic fauna as well as commercially and recreationally important species of shrimp, crabs, and fish that are associated with the benthos, or feed on sediment-dwelling organisms.

In environmental monitoring studies, porewater ammonia concentration is often measured because at very high levels it has been found to be toxic to benthic organisms. Porewater ammonia concentration is also indicative of high levels of bacterial production. Gillett (2003) suggested that bacteria may serve as an important food source for benthic organisms, and in particular for the oligochaete *Monopylephorus rubroniveus*. In the May River, porewater ammonia concentration was statistically different among creeks (p-value = 0.0162) but not between watershed classes (p-value = 0.7836). Rose Dhu Creek had the highest mean concentration of porewater ammonia (mean = 7.7 mg/L) while Brighton Beach Creek had the lowest (mean = 2.0 mg/L) (Figure III-22). These porewater ammonia values are consistent with those measured in the upper reaches of Malind and Okatee creeks during the summer of 2001, which averaged 7.0 mg/L.

Benthic chlorophyll-*a* concentration is a gross estimation of the abundance of microphytobenthos. Benthic diatoms form mats on the creek bank and are considered to be an important component of the food web in headwater tidal creek systems. There was a significant difference in chlorophyll-*a* concentration among creeks (p-value = 0.0008) that followed the opposite pattern of porewater ammonia (Figure III-22). Brighton Beach Creek had the highest mean chlorophyll-*a* concentration (mean = 152.9 mg/m^2) while Rose Dhu Creek had the lowest (mean = 63.7 mg/m^2). Chlorophyll-*a* concentration was significantly higher (p-value = 0.0484) in suburban creeks (mean =143.6 mg/m^2) than forested creeks (mean = 112.8 mg/m^2).

May River headwater tidal creeks were dominated by mixed muddy sediments, however grain size was patchy even within a given creek. There was a significant difference in sediment grain size among creeks (p-value = 0.0013). Palmetto Bluff Creek had the highest mean silt/clay content (mean = 77.8%; range = 59 to 96%), while Bass Creek had the lowest (mean = 16.7%; range = 6.8 to 31.5%). Forested creeks had a significantly higher mean silt/clay composition (p-value = 0.0034, Figure III-23). This trend of sandier sediments in developed creek systems has previously been found in TCP creeks. Suburban creeks are surrounded by increased amounts of impervious cover, and as a result experience ‘flashier’ run-off from the upland which carries with it sandy soils that are deposited into creek beds. This increased run-off may also wash away finer grained sediment particles (like silts and clays), leaving behind sediment particles of a larger grain size. A change in sediment type is one of the first in a succession of responses to watershed alteration and results in an altered benthic community structure, which can have ecosystem-level ramifications.

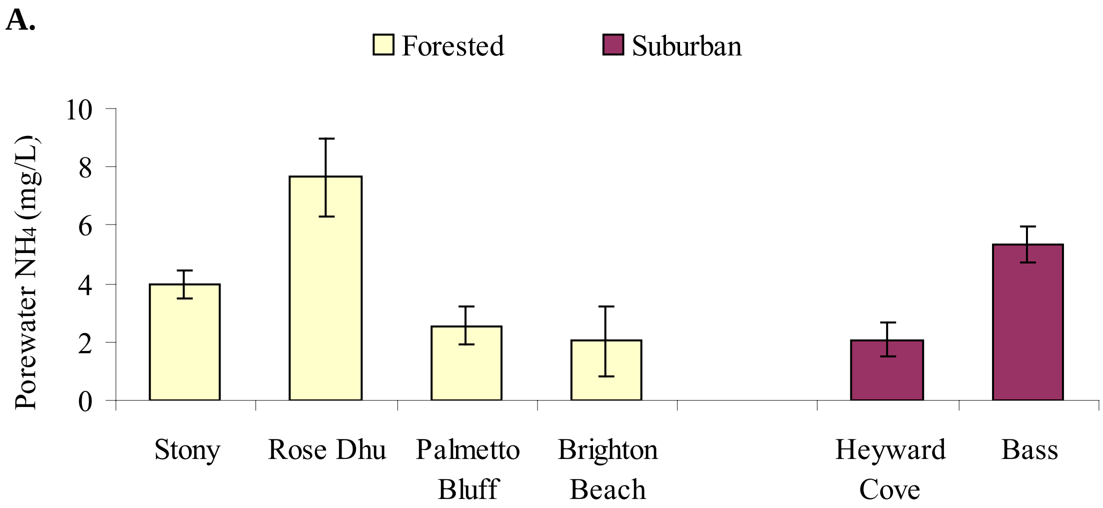
Total organic carbon (TOC) is a measure of the nutritive value of sediment, and is known to be dependent on sediment grain size. Sediments with a high silt/clay fraction typically have a higher organic content. For example, Palmetto Bluff Creek sediments had the highest TOC content (mean = 3.8 %) and highest silt/clay composition (mean = 77.8%) of all May River creeks while Bass Creek had the lowest TOC content (mean = 0.45%) and lowest silt/clay composition (mean = 16.7%). There was a significant difference in mean TOC content between forested and suburban creeks (p-value = 0.0007) in the May River system. The mean TOC composition of forested creek sediments was 2 times higher than that of suburban creek sediments (means = 3.1% and 1.4%, respectively) (Figure III-23). There was no significant difference in percent TOC between the May River and TCP creeks (p-value = 0.1813).

Contamination

All of the sediments collected from the six headwater tidal creeks had relatively low contaminant concentrations. Chemical concentrations did not exceed the effects range-median (ERM) levels for any of the contaminants analyzed and most fell below effects range-low (ERL) levels as well. However, ERL concentrations were exceeded for select analytes in three creeks (Table III-5). Palmetto Bluff and Brighton Beach creeks, which are forested creeks, had concentrations of arsenic that exceeded the ERL level, and Brighton Beach Creek also had concentrations of nickel that were higher than the ERL level. These two elements are naturally occurring and are the result of the weathering of sediments. Elevated concentrations are not unusual in the southeastern region of the United States and have been encountered in pristine locations (Sanger and others, 1999a; Scott and others, 2000).

Heyward Cove, a suburban creek, had concentrations of three PAHs (acenaphthylene, benzo(a)pyrene, and fluoranthene) that exceeded the ERL levels. Heyward Cove Creek also had the highest DDT and total pesticide concentrations of all May River creeks, however those values did not exceed the ERL levels. The other suburban creek, Bass Creek, had very low levels of all types of contaminants. This may have been influenced by the sediment composition in this creek, which was predominately sand. Contaminants do not generally adsorb to sand.

An effects range-median quotient (ERMQ) was calculated for each contaminant class (i.e., metals, PAHs, PCBs, pesticides) in each creek. In addition, a grand ERMQ was calculated for each creek that included a suite of 24 chemical compounds. Bass Creek had the lowest grand ERMQ value (0.003) and the lowest values for each class of contaminants (Table III-6), which are probably related to the high sand content of its sediment. In general, forested creeks had higher levels of trace metals and the suburban creek, Heyward Cove Creek, had higher levels of PAHs. None of the May River headwater tidal creeks had high concentrations of PCBs or pesticides. Overall, the ERMQ values for each of the contaminant classes as well as the grand ERMQ values for each creek were respectively lower than the values from Charleston Harbor creeks that were surveyed in 1995 (Figure III-24). Two creeks (Palmetto Bluff, Brighton Beach) had ERMQ values >0.020 and Heyward Cove had an ERMQ value >0.058. Hyland and others (1999) provide evidence that ERMQ values greater than 0.020 represent a moderate risk of observing a degraded benthic community and values >0.058 represent a high risk of observing a degraded benthic community. However, these estimates were derived from larger, subtidal systems and may not accurately reflect the tolerances of the macrobenthic community in shallow, intertidal headwater creeks to sediment contaminants.



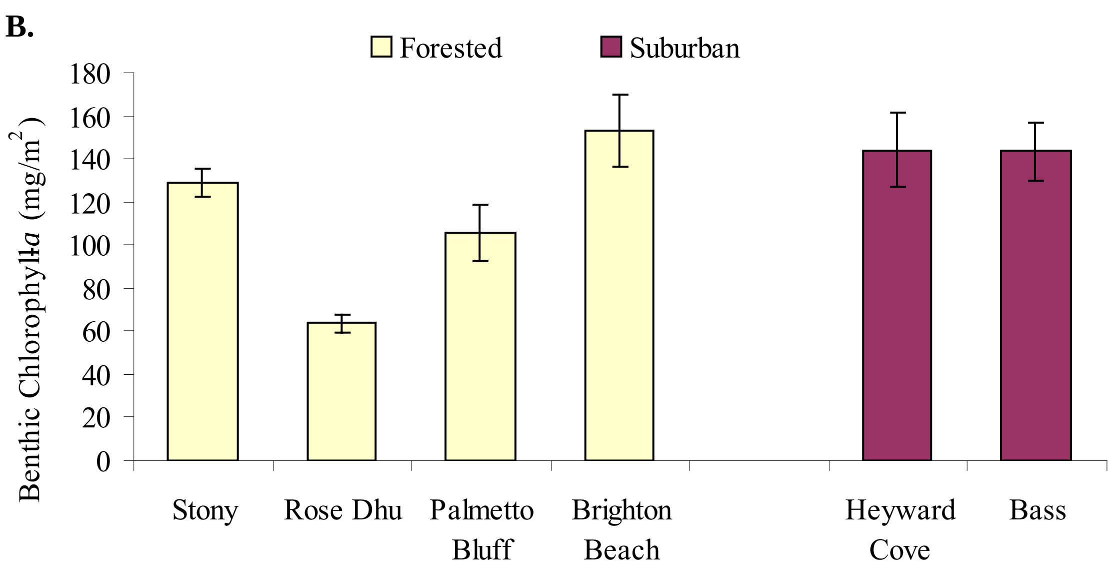


Figure III-22. Mean porewater ammonia (A.) and benthic chlorophyll-*a* (B.) concentrations in headwater tidal creeks. Error bars represent 1 standard error.

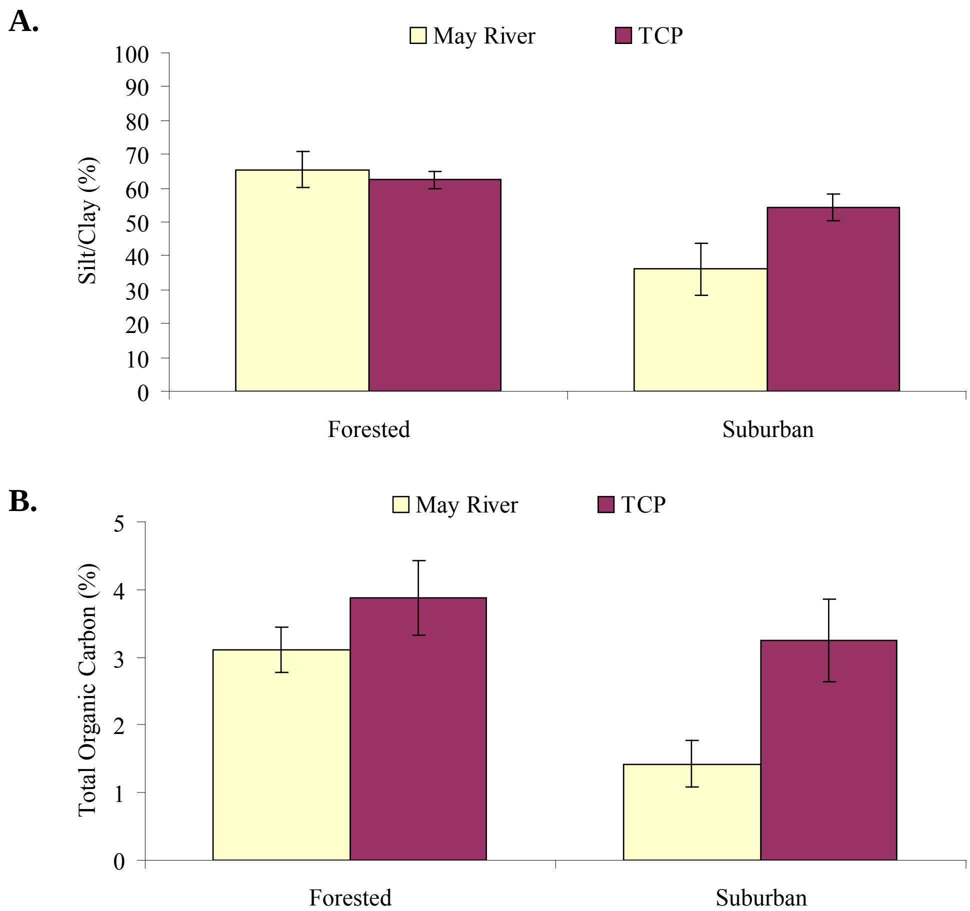


Figure III-23. Mean silt/clay (A.) and total organic carbon concentration (B.) of forested and suburban creeks in the May River compared to TCP creeks sampled during the summer of 1994. Error bars represent 1 standard error.

Table III-5. Summary of contaminants that exceeded effects range-low (ERL) values in headwater tidal creeks of the May River.

Creek Name	# of ERL Excedences	Analytes	Contaminant Class
Forested			
Stony	0		
Rose Dhu	0		
Palmetto	1	Arsenic	Metals
Brighton Beach	2	Arsenic, Nickel	Metals
Suburban			
Heyward Cove	3	Acenaphthylene, Benzo(*a*)anthracene, Fluoranthene	PAHs
Bass	0		

Table III-6. Effects range-median quotient (ERMQ) values for 4 contaminant classes (Trace Metals, PAHs, PCBs, and Pesticides) as well as total contaminants (Grand) and results of toxicology tests from sediment collected from the designated primary sites in headwater tidal creeks. Shaded values indicate sediments that were potentially toxic. The Microtox® assay was corrected for the percent silt/clay and moisture content of the sediment.

Creek Name	% Silt/Clay	ERMQ Values: Trace Metals	PAHs	PCBs	Pesticides	Grand	Toxicity: Microtox® EC_{50}	% Clam Growth
Forested								
Stony	56	0.036	0.003	0.000	0.015	0.020	0.029	31
Rose Dhu	45	0.036	0.003	0.000	0.003	0.014	0.050	40
Palmetto Bluff	71	0.080	0.001	0.000	0.000	0.028	1.233	78
Brighton Beach	74	0.087	0.005	0.000	0.004	0.032	0.093	90
Suburban								
Heyward Cove	38	0.022	0.091	0.000	0.020	0.059	5.545	80
Bass	13	0.003	0.000	0.000	0.000	0.001	1.068	68

Toxicity

Laboratory bioassays are used to determine if estuarine sediments are toxic to organisms. They serve as indicators of contaminant bioavailability and are important in determining if the contaminant concentrations measured have adverse biological effects. Two toxicity bioassays were used for the May River Project, the Microtox® assay and the seed clam assay.

Sediments from five of the six headwater tidal creeks evaluated in this study were considered to have toxic effects in at least one of the assays used (Table III-6). The seed clam assay identified four creeks as having toxic sediments as exemplified by low clam growth rates. Two creeks, Stony and Rose Dhu, had particularly low growth rates (< 40% of the control). These same two creeks, along with Brighton Beach Creek, expressed toxicity in the Microtox® assay as well (Table III-6). Toxicity did not correlate well with the concentration of sediment contaminants, as expressed by the ERMQ values. Sediments from Stony and Rose Dhu creeks expressed toxicity for both of the assays used, however these creeks had relatively low contaminant concentrations (ERMQ = 0.0203 and 0.0140, respectively). Heyward Cove Creek had the highest ERMQ value of all headwater tidal creeks and of any site measured in the entire May River (ERMQ = 0.0586) and yet the sediment was not toxic to the organisms in either of the assays used. It is unclear what caused the potential toxicity, especially considering the overall low contaminant concentrations.

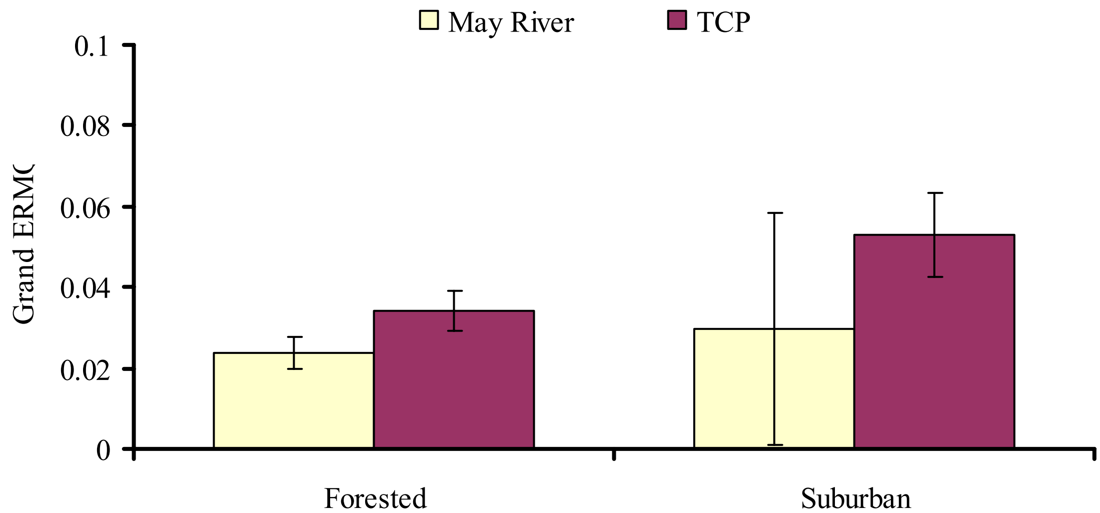


Figure III-24. Comparison of sediment Grand ERMQ values in forested and suburban headwater creeks of the May River compared to TCP creeks sampled during the summer of 1995. Error bars represent 1 standard error.

Biological Quality

Macrobenthic Community

Macrobenthic community structure has been used in a variety of ecosystems as an indicator of habitat condition and quality. Macrobenthic organisms are largely sedentary and are not able to escape environmental or anthropogenic stresses. It is for this reason that the analysis of macrobenthic community structure provides an integrated, long-term assessment of overall ecosystem quality. Parameters such as sediment grain size, salinity regime, and water column dissolved oxygen concentration are important drivers that shape benthic communities, as individual species have varied tolerances to physical and chemical environmental variables.

A few of the benthic samples collected from headwater tidal creeks contained mobile species (i.e., Brachyuran crabs), non-macrobenthic taxa (i.e., insects), or very small organisms (i.e., Platyhelminth worms) that were poorly sampled by the methods used. These organisms were not included in the data summaries or analyses presented in this section.

A total of 1,405 individual organisms representing 30 taxa were identified from 36 benthic samples collected intertidally from the six headwater tidal creeks (Access ® database). Two classes of segmented annelid worms, oligochaetes and polychaetes, were the numerically dominant benthic fauna and comprised 78% and 18% of the total taxa, respectively. The other major taxa identified consisted of molluscs (2.7%), crustaceans (0.85%), and nemerteans (0.35%). Oligochaetes comprised 82% of the fauna in forested creeks and 48% of the fauna in suburban creeks. Polychaetes comprised 14% of all organisms in forested creeks and 46% in suburban creeks (Figure III-25).

Twelve taxa (five oligochaetes, six polychaetes, and one gastropod) comprised 95% of the fauna collected (Table III-7). The oligochaete *Monopylephorus rubroniveus* was the most abundant organism and accounted for 64% of all fauna and 83% of the oligochaetes. *Streblospio benedicti*, a cosmopolitan polychaete species, was the second most abundant taxa and comprised approximately 10% of all fauna and 56% of the polychaetes.

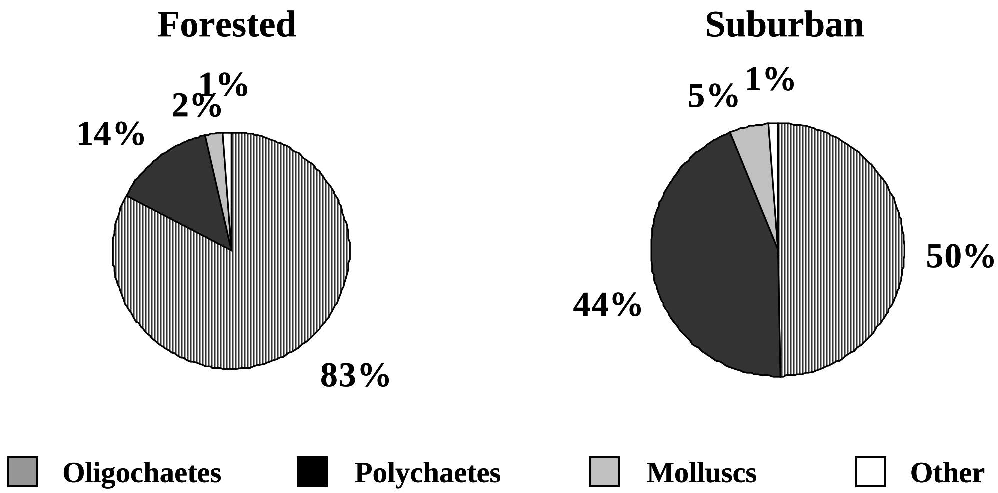


Figure III-25. Composition of major taxonomic groups in forested and suburban classes of headwater creeks of the May River.

Table III-7. Mean absolute abundance of the 12 most dominant taxa collected from headwater tidal creeks as well as the Shannon-Weaver diversity index and species evenness of each creek and of all headwater creeks combined.

		Forested				Suburban		
Species Name	Taxonomic Group	Stony	Rose Dhu	Palmetto Bluff	Brighton Beach	Heyward Cove	Bass	All Headwater Creeks
		Avg. #/m^2						
Monopylephorus rubroniveus	Oligo	25,037	1,535	5,227	402	256	914	5,562
Streblospio benedicti	Poly	110	256	2,632	585	1,096	439	853
Tubificidae sp.	Oligo	110	73	2,047	548	841	146	627
Tubificoides heterochaetus	Oligo	1,170	73	0	0	0	0	207
Illynassa obsoleta	Mol	0	146	0	877	0	110	188
Capitella capitata	Poly	0	73	0	0	439	475	164
Cirratulidae sp.	Poly	0	0	804	0	37	110	158
Monopylephorus irroratus	Oligo	0	0	0	0	731	73	134
Tubificoides brownae	Oligo	0	37	475	37	37	0	97
Leitoscolopolos fragilis	Poly	0	146	183	219	37	0	97
Neanthes succinea	Poly	0	37	329	73	37	0	79
Enchytraeidae	Oligo	0	0	0	0	292	37	54
Percent of total abundance		100	98	98	77	91	85	
Mean H' Diversity		0.126	0.358	0.441	0.553	0.549	0.407	0.406
Mean Evenness		0.344	0.753	0.581	0.818	0.822	0.673	0.665

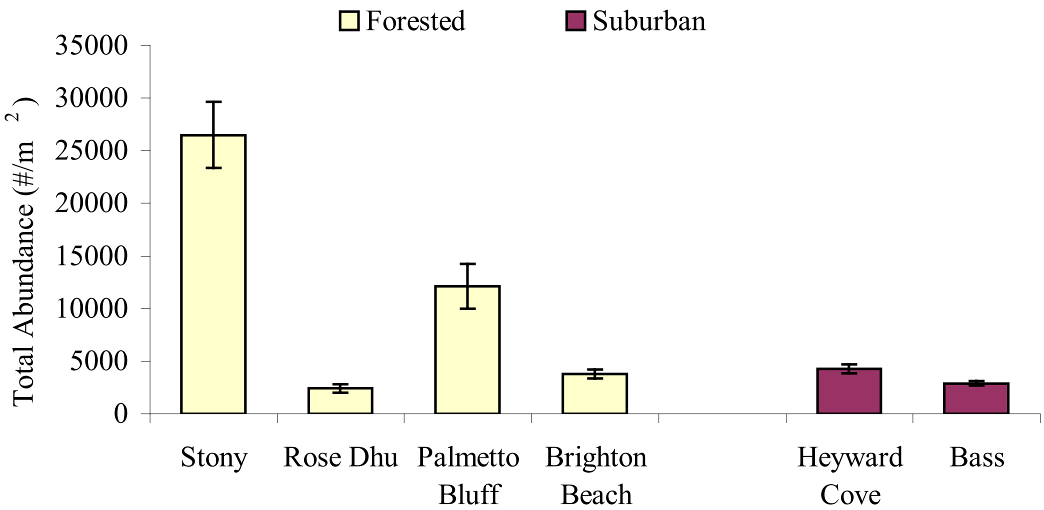


Figure III-26. Total benthic infaunal abundance of May River headwater tidal creeks. Error bars represent 1 standard error.

Stony Creek had the greatest mean overall abundance of 26,498 individuals/m^2, which was an order of magnitude greater than Bass Creek, which exhibited the lowest mean overall abundance (mean = 2,887 individuals/m^2, Figure III-26). The large benthic faunal abundances found in Stony Creek were largely due to the presence of a single species, *M. rubroniveus*. Mean overall abundance (of all taxa) tended to be greater in forested creeks compared to suburban creeks, although this difference was not significant (p-value = 0.1609).

The distribution patterns of some of the most abundant taxa found in headwater tidal creeks were assessed among creeks and between watershed classes. Previous research has demonstrated the importance of analyzing the distribution patterns of individual species, as the information that exists on the physical and chemical tolerances of certain species are helpful in interpreting environmental data. The three most dominant taxa will be discussed in this report.

The Tubificid oligochaete, *Monopylephorus rubroniveus,* was the most abundant organism in May River headwater tidal creeks, it was found in all six creeks and in 66% of the samples. *M. rubroniveus* has also been found to be the numerically dominant species in headwater tidal creeks throughout South Carolina (Sanger, 1998; Lerberg and others, 2000; Gawle, 2002; Gillett, 2003; Holland and others, 2004). This opportunistic species thrives in stressful environments, whether they are naturally occurring or anthropogenically-induced. Field studies have found *M. rubroniveus* to be widely tolerant of long episodes of low DO, fluctuating salinity regimes, and high sediment contaminant concentrations, all of which are conditions associated with headwater tidal creek environments (Sanger, 1998; Lerberg and others, 2000; Holland and others, 2004). Laboratory toxicological studies have confirmed the tolerance of *M. rubroniveus* to low DO, high fluoranthene (a common PAH) concentrations, and high copper concentrations (Calle unpublished).

The abundance of *M. rubroniveus* was significantly different among creeks (p-value = 0.0038). *M. rubroniveus* was fifteen times more abundant in Stony Creek (mean = 25,000 individuals/m^2) compared to all other May River creeks (mean = 1,600 individuals/m^2; Figure III-27). Densities of this species reached levels as high as 49,780 individuals/m^2 in some portions of Stony Creek. These high abundances may be due to the fact that this creek is located in the headwater region of the May River and drains an extremely large watershed (~2150 hectares). A similar pattern was noted in another study in Beaufort County in which the creeks located near the headwaters of Broad Creek and Okatee River had the greatest overall abundances (Van Dolah and others, 2000).

As a result of extremely high abundances in Stony Creek, forested creeks had a higher abundance of *M. rubroniveus* compared to suburban creeks (p-value = 0.0144, Figure III-28). The trend of higher abundances of *M. rubroniveus* in reference creeks compared to developed creeks found in this study is opposite the pattern observed in other studies of headwater tidal creeks throughout South Carolina, including the 1994 TCP study (Figure III-28). This oligochaete has generally been found to be more prevalent in creeks that drain suburban and urban watersheds, as these creeks are typically more chemically and physically stressful than forested creeks. The naturally low DO levels in Stony Creek may have factored into the high densities of *M. rubroniveus* that were found.

The second most abundant organism was the polychaete *Streblospio benedicti*, which was found in all six creeks and in 53% of the samples. The abundances of *S. benedicti* were not statistically different among creeks (p-value = 0.1463), however Palmetto Bluff Creek had the highest mean abundance (mean = 2,631 individuals/m^2, Figure III-27). In a two-way ANOVA model, abundances of *S. benedicti* were similar between the May River and the TCP, however *S. benedicti* was more abundant in forested creeks than suburban creeks (p-value = 0.0209, Figure III-28).

The Tubificidae species (sp.) complex was the third most abundant organism and was also found in all six May River headwater tidal creeks, and in 42% of the samples. This taxon is a grouping of oligochaetes that could not be identified to the species level based upon setal structure alone. The total abundance of Tubificidae sp. was not significantly different among creeks (p-value = 0.3817), however Palmetto Bluff Creek had the highest mean abundance (mean = 2,046 individuals/m^2, Figure III-27). In a two-way ANOVA model, abundances of Tubificidae sp. were similar between forested and suburban creeks, however Tubificidae sp. was more abundant in the May River creeks compared to the TCP creeks (p-value = 0.0061, Figure III-28)

Evaluating the diversity of the benthic community is a common tool used to assess overall habitat quality (Peet, 1974); however, measures of diversity have not been found to be useful indicators in headwater creeks due to the inherently stressful nature of these systems. The theory behind various diversity calculations is that the benthic community is less complex in more stressed (or polluted) habitats, which tend to be dominated by a few opportunistic and highly tolerant species. Three measures of diversity were used in this study including species richness, Shannon-Weaver species diversity (H') and species evenness (E).

Species richness, or the total number of taxa was not statistically different among creeks (p-value = 0.2252). In general, the number of benthic taxa in headwater tidal creeks was low, due to the inherently stressful nature of these systems. Stony Creek had the lowest total number of taxa while Brighton Beach and Heyward Cove creeks had the highest total number of taxa (Figure III-29). The low number of species found in Stony Creek was probably related to the stressful conditions due to low DO, as water quality records from the five-day deployment revealed frequent and prolonged periods of hypoxia. Lerberg and others (2000) reported that species richness was inversely related to hypoxia. There was no significant difference in the total number of species between watershed types (p-value = 0.3688, Figure III-30) and the mean number of taxa across all May River creeks was 13, which is comparable to the mean number of taxa found in TCP creeks (Lerberg and others, 2000).

Species diversity (H') was statistically different among creeks (p-value = 0.0056) and showed a similar pattern to the total number of taxa. Brighton Beach Creek had the highest species diversity and Stony Creek had the lowest (Figure III-30). In general, suburban creeks were slightly more diverse than forested creeks (H' = 0.47 and 0.37, respectively) but there was no significant difference between watershed classes (p-value = 0.1827) or between May River creeks and TCP creeks (p-value = 0.5512, Figure III-30).

The calculation of species evenness represents the distribution of individuals across the number of taxa identified. Evenness was statistically different among the six May River headwater tidal creeks (p-value = 0.0103, Figure III-30). Stony Creek had the lowest value of species evenness due to dominance by one species, *Monopylephorus rubroniveus,* while Heyward Cove Creek exhibited the highest species evenness. Evenness was not statistically different between watershed types (p-value = 0.2044) or between May River creeks and TCP creeks (p-value = 0.2954, Figure III-30).

In an effort to effectively utilize macrobenthic community composition to infer overall habitat quality and make assessments of ecological condition, the TCP developed two metrics based upon the tolerances of individual species to stressful conditions (Lerberg and others, 2000). The stress-sensitive taxa are those species that are not frequently found in high stress habitats including *Streblospio benedicti*, *Tharyx acutus*, *Tubificoides heterochaetus*, *Heteromastus filiformus*, and nemerteans. The stress-tolerant taxa are those species that are often associated with high stress habitats including *Laeonereis culveri*, *Monopylephorus rubroniveus*, *Paranais litoralis*, and *Tubificoides brownae.*

The relative abundance of stress-sensitive taxa was not statistically different among creeks (p-value = 0.1941) or between forested and suburban watersheds (p-value = 0.5214) (Figure III-31). Heyward Cove Creek had the highest proportion of stress-sensitive species while Brighton Beach and Bass creeks had the lowest. Overall, the abundance of stress-sensitive taxa was relatively low in both forested and suburban watersheds of the May River and was similar to that of suburban creeks of the TCP (Figure III-32).

The relative abundance of stress-tolerant taxa was significantly different among creeks (p-value = 0.0017) and between watershed types (p-value = 0.0081) in the May River. Stony Creek had the highest proportion of stress-tolerant taxa (Figure III-31), which was driven primarily by the high densities of *M. rubroniveus*. Forested creeks had a higher relative abundance of stress-tolerant species compared to suburban creeks in the May River (Figure III-32). This pattern is opposite that seen in other creeks throughout South Carolina, where stress-tolerant species are typically more abundant in more developed systems (Lerberg and others, 2000; Holland and others, 2004). In the May River, the macrobenthic community reflects that the forested creeks, especially those located near the headwaters of the system (i.e., Stony and Rose Dhu creeks), are more physiologically stressful to benthic organisms than creeks that drain suburban watersheds.

Nektonic Community

One seine sample was collected from each creek. A total of 35,732 organisms were collected that represented 19 taxa. However, 10 of these taxa comprised more than 99% of all the nekton found at each site (Table III-8). The two most abundant species were crustaceans and they accounted for 99% of all nektonic fauna in headwater tidal creeks. The grass shrimp, *Palaemonetes* sp., and the commercial shrimp, *Penaeus* sp., accounted for 92% and 7% of all nektonic taxa, respectively. The grass shrimp was most abundant in Brighton Beach Creek and least abundant in Bass Creek. There was no significant difference in the abundance of this species between forested and suburban creeks (p-value = 0.8441). *Penaeus* sp. were generally found in greater abundances in forested creeks (6.2 individuals/m^2) compared to suburban creeks (1.1 individuals/m^2), however, this difference was not significant (p-value = 0.1539). Stony Creek had the highest density of *Penaeus* sp. with 9.7 individuals/m^2. Various species of fish, including mummichogs (*Fundulus heteroclitus)* and anchovys (*Anchoa mitchilli*), accounted for the remaining 1% of the fauna.

Brighton Beach and Heyward Cove creeks had the highest abundances of nekton, which was explained by high abundances of grass shrimp in these two creeks (Figure III-33). Bass Creek had the lowest nektonic abundance with fifty times fewer individuals per meter than Brighton Beach and Heyward Cove creeks. Palmetto Bluff Creek had the most diverse nektonic assemblage (11 different species) of all May River headwater tidal creeks (Figure III-33). Stony Creek had the lowest nektonic diversity (6 different species).

Table III-8. The ten most abundant nektonic species collected in seine nets in May River headwater tidal creeks.

			Forested				Suburban	
Species Name	Total Abun	Total Abun	Stony	Rose Dhu	Palmetto	Brighton	Heyward	Bass
	%	# individuals	# individuals /m^2					
Grass Shrimp (*Palaemonetes* sp.)	92.05	32,892	23.85	5.55	17.73	144.67	120.13	1.28
Commercial Shrimp (*Penaeus* sp.)	6.78	2,421	9.71	5.52	8.53	1.06	1.09	1.14
Anchovy (*Anchoa mitchilli*)	0.61	217	0	0.04	0.39	0	1.98	0.01
Mummichog (*Fundulus heteroclitus*)	0.29	102	0.04	0.04	0.12	0.34	0.39	0.06
Silver Perch (*Bairdiella chrysoura)*	0.12	44	0	0	0.08	0	0.41	0
Silverside (*Menidia menidia*)	0.05	17	0	0	0.09	0.05	0.02	0
Striped Mullet (*Mugil cephalus)*	0.03	12	0.02	0.01	0.02	0	0.02	0.05
Killifish (*Fundulus majilis*)	0.02	7	0	0	0.02	0.02	0	0.03
Anchovy (*Anchoa* sp.)	0.01	5	0	0	0	0	0.06	0
Spot (*Leiostomus xanthurus*)	0.01	4	0	0	0.01	0	0.01	0.02
Percent of Total Abundance			99.88	99.82	99.93	99.99	99.98	99.62
Number of Species			6	7	11	7	10	8

A.

B.

C.

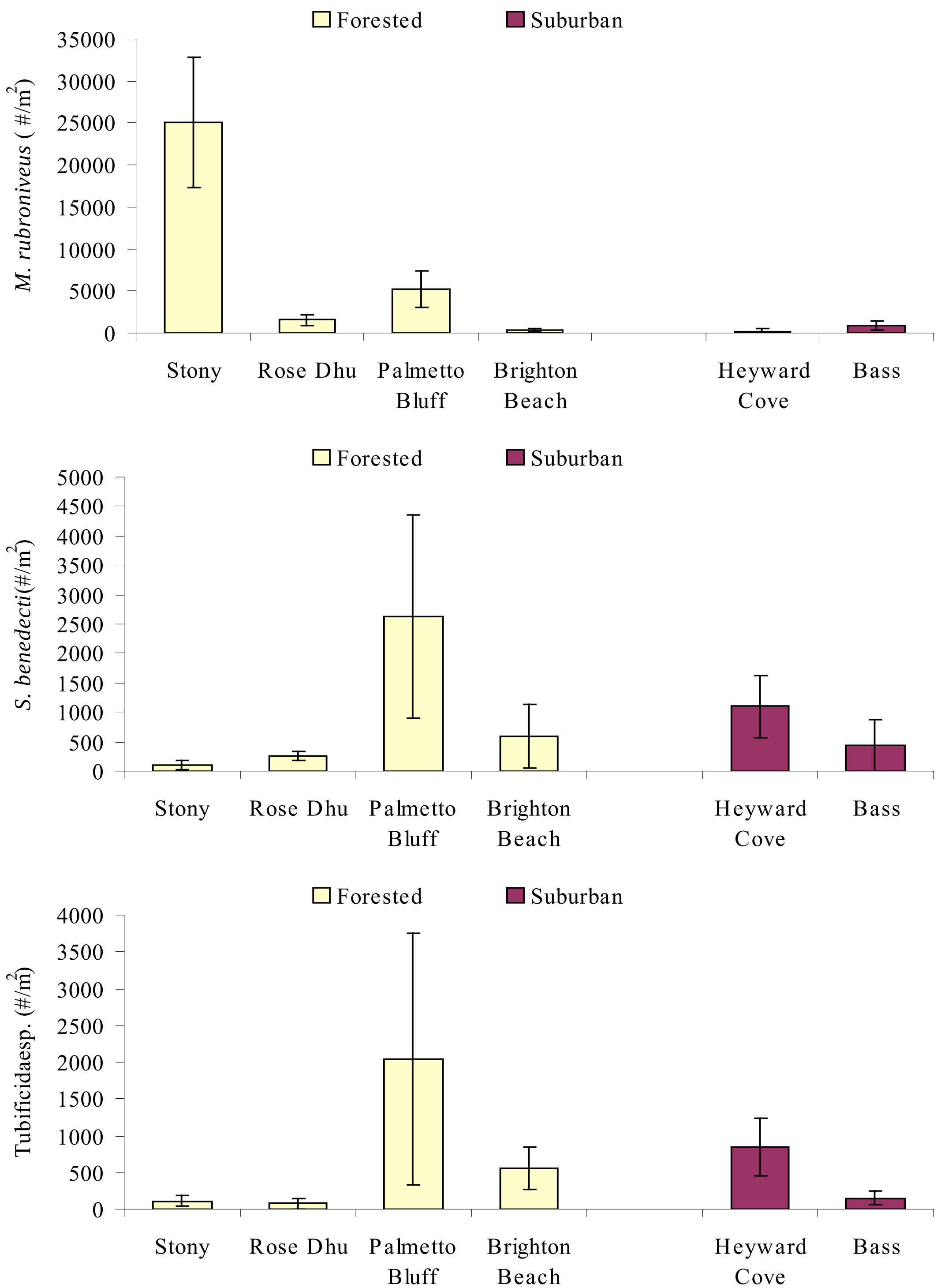


Figure III-27. Mean abundance of the three most dominant benthic species, *Monopylephorus rubroniveus* (A.), *Streblospio benedicit* (B.), and Tubificidae sp. (C.) in forested and suburban headwater tidal creeks of the May River. Error bars represent 1 standard error.

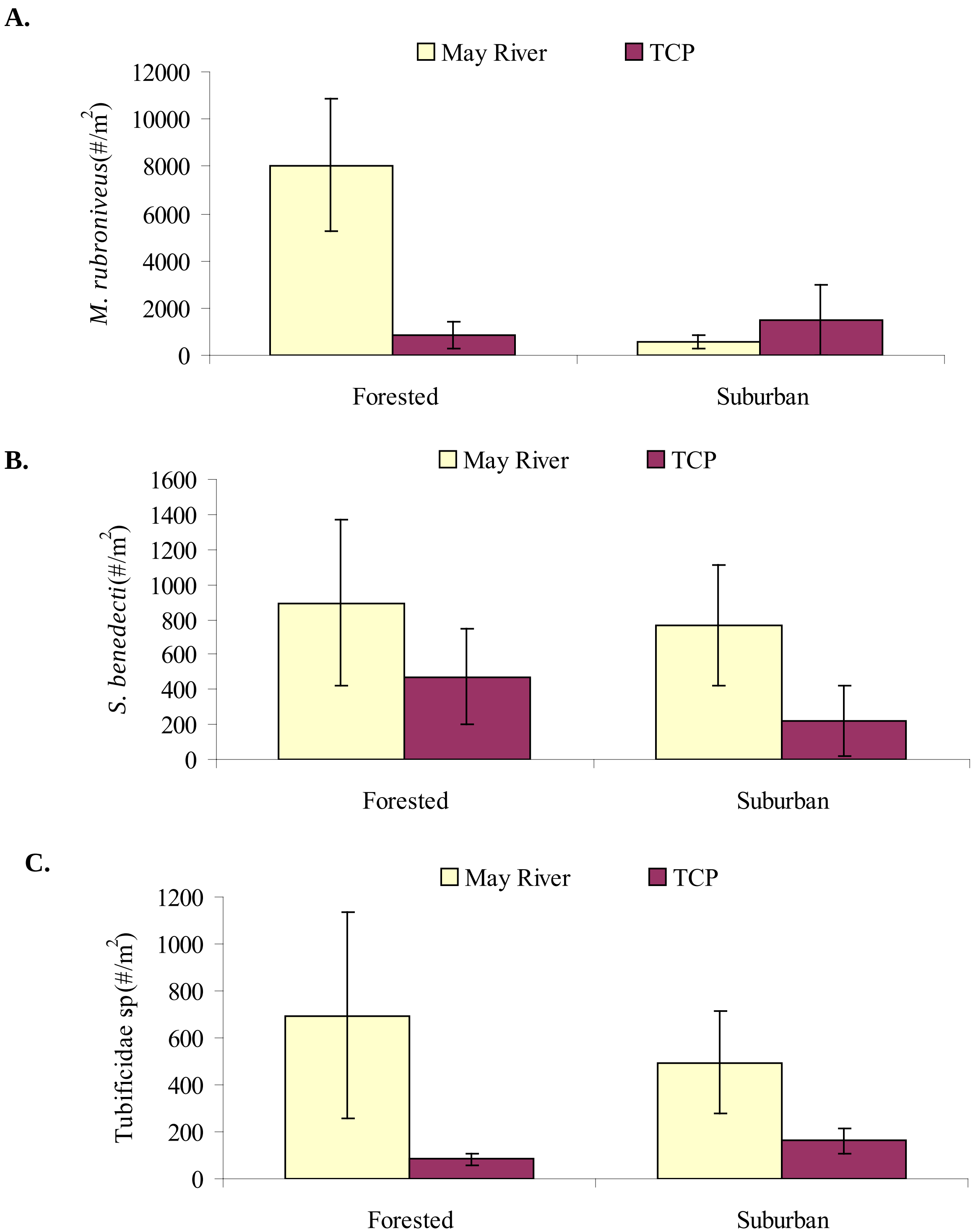


Figure III-28. Mean abundances of the three most dominant benthic taxa; *Monopylephorus rubroniveus* (A.), *Streblospio benedicti* (B.), and Tubificidae sp. (C.) in forested and suburban headwater tidal creeks of the May River compared to TCP creeks sampled during 1994. Error bars represent 1 standard error.

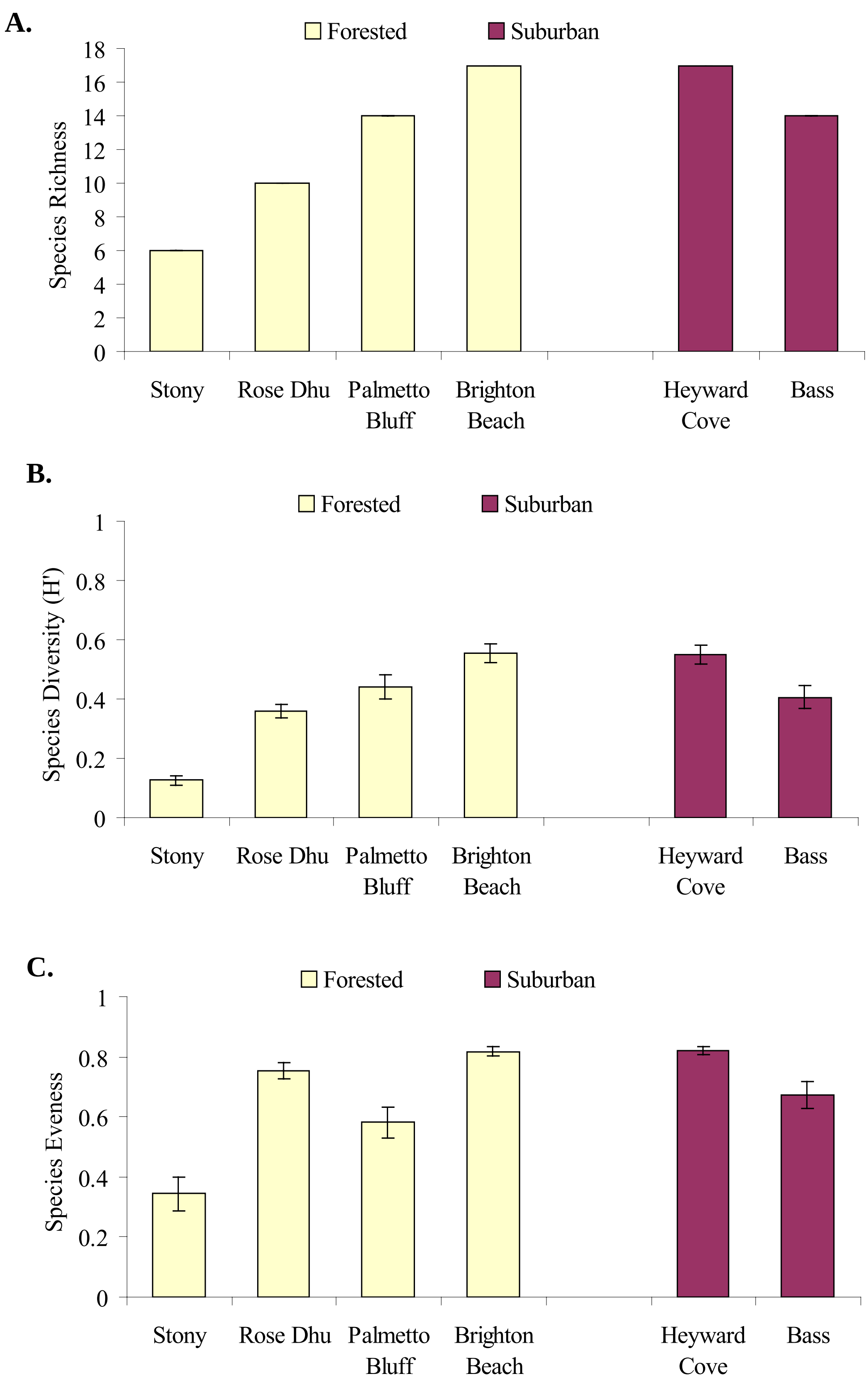


Figure III-29. Measures of macrobenthic species diversity in May River headwater tidal creeks. Mean species richness (A.), Shannon-Weaver species diversity index (H') (B.), and species evenness (C.) were compared among headwater creeks. Error bars represent 1 standard error.

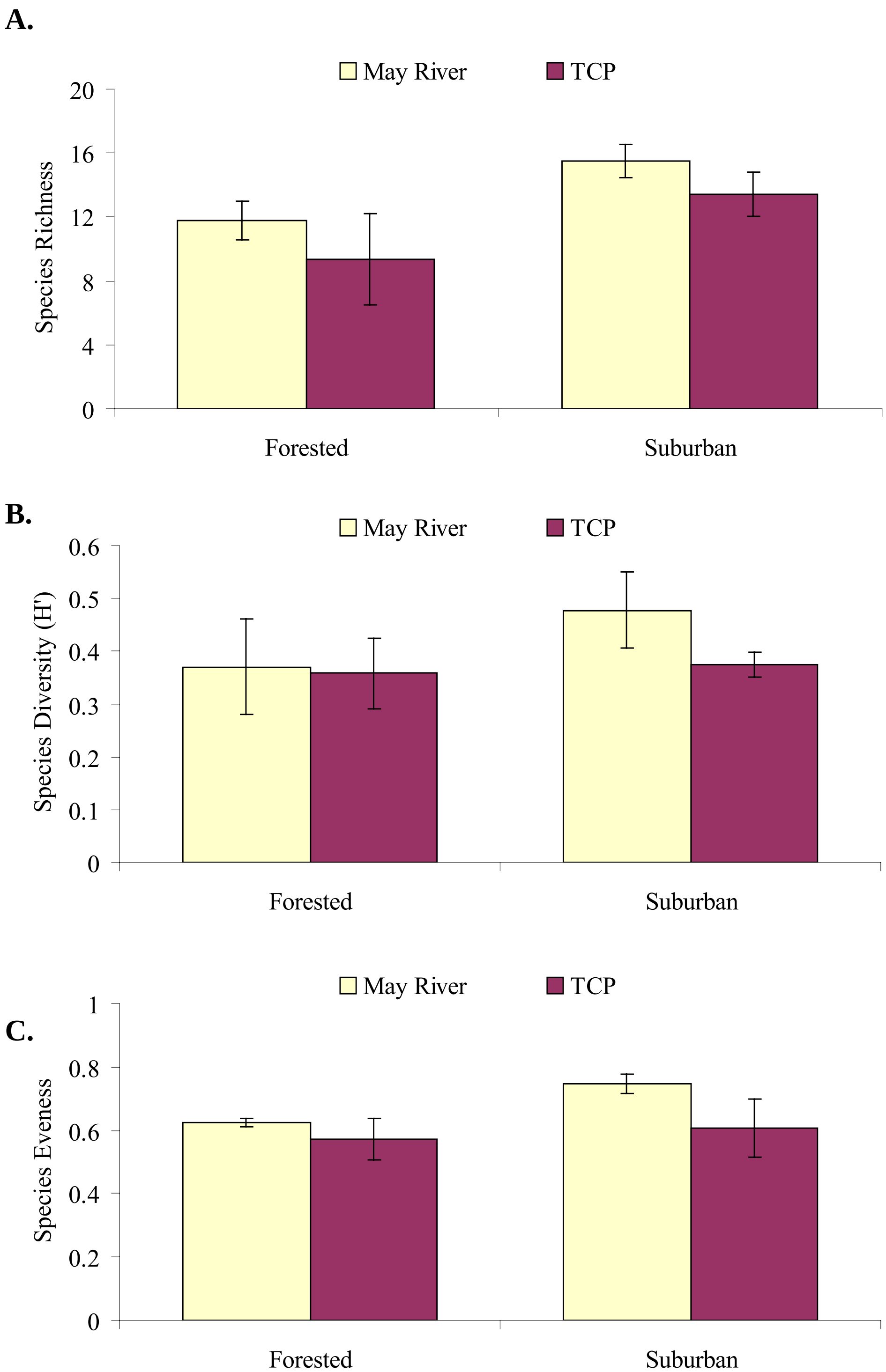


Figure III-30. Mean macrobenthic species richness (A.), diversity (H') (B.), and evenness (C.) in forested and suburban May River creeks compared to TCP creeks sampled during the summer of 1994. Error bars represent 1 standard error.

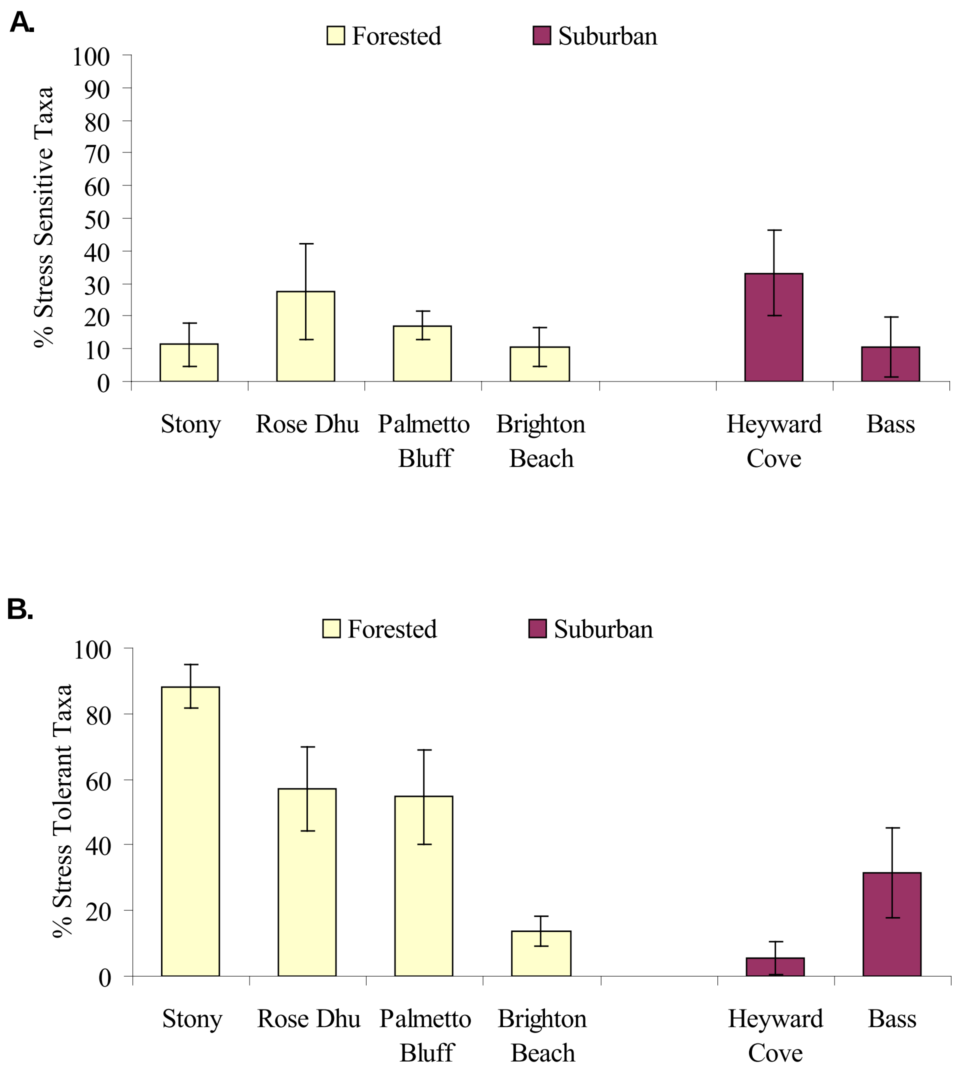


Figure III-31. Mean relative abundance of stress-sensitive (A.) and stress-tolerant (B.) benthic taxa in headwater tidal creeks sampled in the May River. Error bars represent 1 standard error.

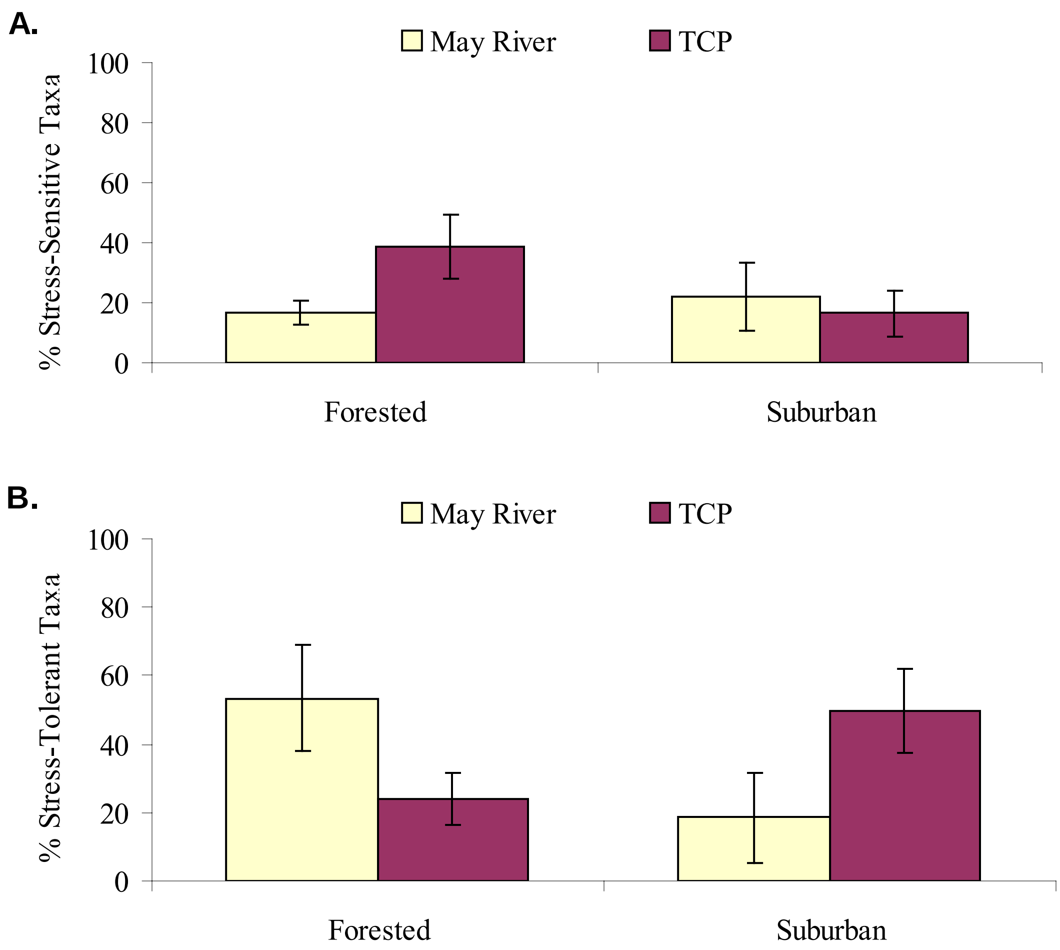


Figure III-32. Mean relative abundance of stress sensitive (A.) and stress tolerant taxa (B.) in forested and suburban headwater creeks of the May River compared to TCP creeks sampled during the summer of 1994. Error bars represent 1 standard error.

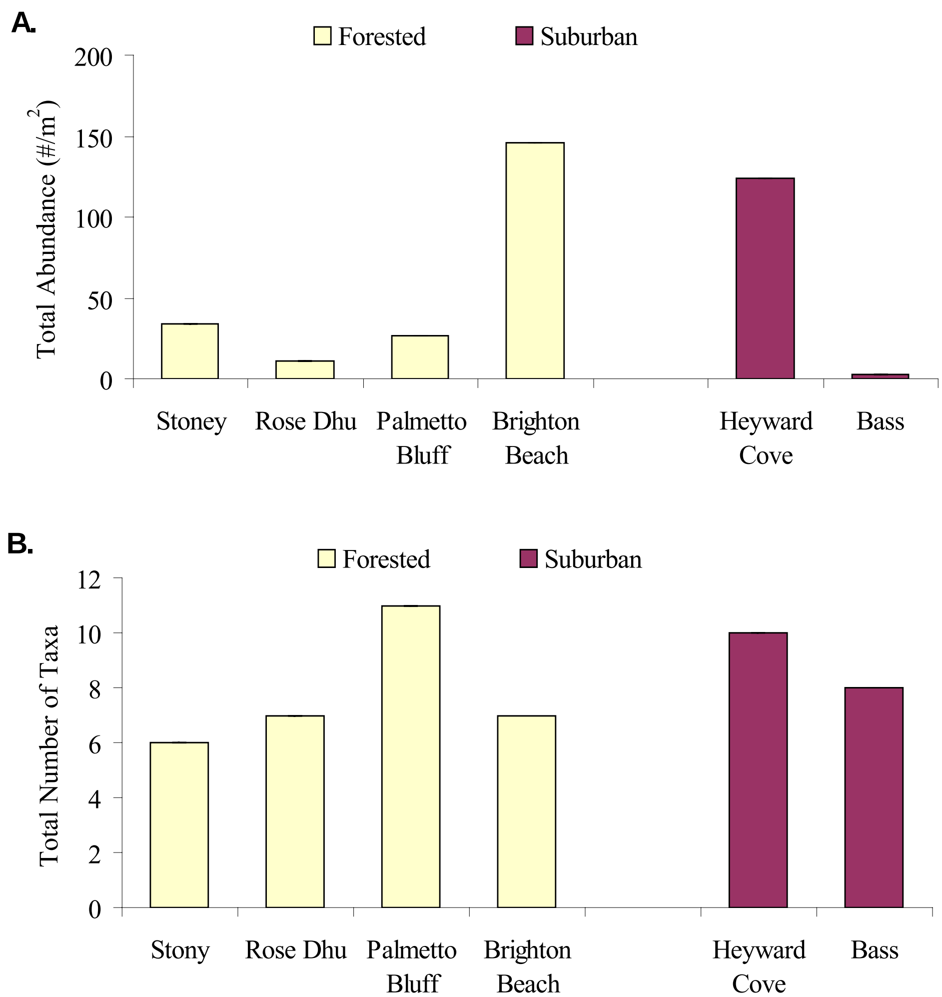


Figure III-33. Total abundance (# individuals/m^2) of all nekton (A.) and the total number of taxa collected (B.) by seine net in headwater tidal creeks of the May River.

Large Tidal Creeks/Open Water Habitats

Water Quality and Chemistry:

As noted for the headwater tidal creeks, sampling periods for the seasonal assessment of water quality at the three larger tidal creek and seven open water sites were the spring (May and June 2002), summer (July and August 2002), fall (October 2003) and winter (March 2003). The summer samples at the May River sites were collected contemporaneously with samples from SCDNR's sampling of the large tidal creek and open water sites. The water quality data (except dissolved oxygen, pH, salinity, and temperature) from 2002 SCECAP sites were compared to the water quality data (except dissolved oxygen, pH, salinity, and temperature) from May River sites to identify whether May River sites were similar to other reference sites in the southern portion of SC. These data are summarized statistically and compared graphically to identify differences among May River sites, between habitat types (i.e., large tidal creek and open water), among seasons, and between May River and 2002 SCECAP sites. The water quality data is provided in the Access® database. Results of the statistical analyses and correlation analyses are summarized in Appendix III-5. Descriptions of the parameters and their related regulations are provided in the previous section on water quality and chemistry of Headwater Tidal Creeks.

The semi-continuous measures of temperature, salinity, pH, and dissolved oxygen were monitored over a 25-hr period during the summer 2002 sampling period to allow comparisons with 2002 SCECAP data obtained from sites outside the May River. These data are also summarized graphically and compared statistically in this report. The Access® database provides the full data set for each site.

Temperature

Seasonal Point Sampling Data

Mean temperature measurements obtained from the seasonal assessment ranged from 21.3^{o}C at L-02 to 25.3^{o}C at U-03. Water temperature at the time of sampling was similar among sites within each season (p-value = 0.0602) and between the two habitats (large tidal creek and open water) (p-value = 0.6684); however, as would be expected, water temperature did vary among seasons (p-value < 0.0001, Appendix III-5c; Figure III-34). As noted previously, data from the USGS continuous monitoring stations provides the best indication of seasonal changes in water temperature (see Section II).

Water temperature in open water and large tidal creek sites was negatively correlated with dissolved oxygen and pH and positively correlated with specific conductance, total suspended solids (TSS), total particulate nitrogen and carbon, organic nitrogen, and all forms of phosphorus (Appendix III-5c).

Semi-Continuous Summer Data

Mean bottom water temperatures measured during the 25-hr semi-continuous monitoring period were very similar both among sites within the May River and when compared with the 2002 SCECAP sites (Figure III-35). In the May River, mean bottom temperatures among all of the sites ranged from 29.0 to 30.3^{o}C with the change in temperature over the tidal periods sampled always ≤ 4°C. Mean bottom temperatures at the ten 2002 SCECAP sites were not different from May River stations and ranged from 27.2 to 30.6 °C (overall mean of 28.5 °C) with a similar range in temperature over the tidal cycles sampled (≤ 5.4°C).

Salinity

Seasonal Point Sampling Data

Mean salinity of open water and large tidal creek sites in the May River ranged from 23.3 ppt at U-02 to 30.3 ppt at L-01 (Figure III-36). Salinities were not considered different among the sites or between the two habitats (open water and large tidal creek). Two of the three sites in the Upper Zone (U-01 and U-02) had the lowest mean salinities that represented brackish conditions (< 25 ppt). Salinities in the spring and summer were higher than the salinities in the winter at all sites (Figure III-36).

Salinity was positively correlated with chloride and specific conductance at an almost one-to-one association and with alkalinity (Appendix III-5c). Ammonia concentrations were negatively, but weakly, correlated with salinity, indicating increases in ammonia concentrations were associated with lower salinity values.

Semi-Continuous Summer Data

Due to the drought conditions present during 2002, all of the stations represented polyhaline conditions (>18 ppt) during the summer semi-continuous sampling period, and only the three uppermost sites in the river had mean bottom salinities slightly below 30 ppt (Figure III-37). Mean bottom salinities among the three large creek sites ranged from 28.9 to 33.5 ppt (mean = 31.9 ppt). Mean bottom salinities at the seven open water stations were similar and ranged from 28.8 to 34.3 ppt (mean = 31.7 ppt; Figure III-37). As expected, bottom salinities were generally higher in the lower zones of the river, which are closer to Calibogue Sound and the ocean and less influenced by upland and groundwater runoff.

The 2002 SCECAP sites sampled for comparative purposes had slightly higher mean salinities in both tidal creeks and open water sites (Figure III-37). Mean bottom salinities at the tidal creek sites ranged from 32.0 to 36.3 ppt (mean = 34.8 ppt) and mean bottom salinities at the open water sites ranged from 32.7 to 38.1 ppt (mean = 34.9 ppt). Because these differences were not very large (3 to 4 ppt) the variation in salinity between the 2002 SCECAP and May River sites probably did not significantly affect the composition of most biota sampled at these sites. Salinity fluctuations over the tidal cycles sampled were less than 4 ppt in the May River, except at the uppermost site (U-01), where the salinities varied as much as 10 ppt. This amount of fluctuation is not likely to represent stressful conditions for the estuarine biota present. Salinity fluctuations over the tidal cycles sampled for the comparison 2002 SCECAP sites were less than 5 ppt.

Dissolved Oxygen

Seasonal Point Sampling Data

Instantaneous dissolved oxygen (DO) concentrations represented daytime DO levels at the time of sampling. Because the measurements do not represent the full daily range in DO, these measurements could not be used to determine compliance with SCDHEC saltwater criteria for DO because the criteria require the daily mean and daily minimum DO concentrations. However, comparison of the instantaneous values with the criteria provided a general screening of the sites.

Mean DO concentrations ranged from 5.27 mg/L at U-01 to 7.67 mg/L at M-02 during the period of study in the May River (Figure III-38). These differences were not significant among the sites or between habitat types. However, DO concentrations were significantly higher during the winter season than during the other seasons at all sites in the May River (p-value = 0.0004, Figure III-38). Seasonal DO concentrations were consistently above the SCDHEC saltwater daily minimum criteria of 4 mg/L at all the sites. In general, the seasonal variability in DO concentrations was lower in sites in the Upper Zone than sites in the Middle or Lower zones. The very high maximum daytime DO indicate high productivity in these creeks.

In addition to water temperature, DO was negatively correlated with all forms of nutrients (nitrogen and phosphorus), TSS, and turbidity (Appendix III-5c).

<u>Semi-Continuous Summer Data</u>

The mean DO conditions in the May River measured over two complete tidal cycles (25-hr) provides a better indication of whether organisms may be experiencing stressful conditions compared to the instantaneous measures. Because the mean DO values obtained using this technique can not be directly related to the state water quality standard, SCDNR and SCDHEC staff have developed guidelines for continuous data sets that have been adopted for the SCECAP program. These criteria identify mean summer-time bottom water DO values < 4.0 mg/L as marginal and mean bottom DO values < 3.0 mg/L as poor and are considered to be potentially stressful to many invertebrate and fish species during this time of the year. Although these criteria should not be used for regulatory purposes, they do allow comparison of DO conditions in the May River with data obtained from other sites in the state using a standardized protocol.

All except one of the stations (U-01) sampled in the May River had mean DO levels > 4 mg/L, which represents non-stressful conditions (Figure III-39). The mean bottom DO at the large tidal creek station U-01 was considered to be marginal (3.8 mg/L), but was close to the 4.0 mg/L criteria. Comparison with the relatively pristine large tidal creek 2002 SCECAP sites indicated that two of the three sites had good DO values (> 4.8 mg/L) and one site had poor DO conditions (2.7 mg/L) (Figure III-39). This suggests that DO concentrations below state water quality criteria occur naturally during summer periods when water temperature is elevated, especially in high salinity waters.

In general, open water stations in the May River had higher DO concentrations than large tidal creeks, but this difference was not statistically significant (p-value = 0.07). However, since tidal creeks generally support a greater diversity and abundances of fish and crustaceans, water quality standards established for larger open water bodies traditionally monitored by the SCDHEC may not be indicative of stressful conditions in these creeks. Van Dolah and others (1999, 2000) also observed that tidal creek habitats had lower DO conditions than open water sites sampled in South Carolina's coastal zone.

pH

<u>Seasonal Point Sampling Data</u>

Mean pH in the large tidal creek and open water sites in the May River were well within the SCDHEC standard range of 6.5 to 8.5 (Figure III-40). Average pH ranged from 7.07 at U-02 to 7.80 at M-02 during the period of sampling. Open water and large tidal creek sites in the May River did not have significantly different pH, however, slight differences were noted among sites (p-value = 0.0358). Except for the summer sample at L-04, seasonal samples at all sites had relatively consistent pH and were within the SCDHEC standard pH range of 6.5 to 8.5. In open water and large tidal creeks, pH levels were negatively correlated with nutrient levels, TOC, and silica (Appendix III-5c).

Semi-Continuous Summer Data

For polyhaline waters (>18 ppt), the SCECAP program has identified pH values below 7.4 as marginal based on SCDHEC's historical database and pH values less than 7.1 as indicative of poor water quality based on state water quality standards (SCDHEC, 2001).

Both mean bottom water pH measurements and the instantaneous surface water quality samples in the May River were $\geq$ 7.4, indicating good water quality conditions (Figure III-41). The lowest pH concentration occurred at station U-01, which was classified as a large tidal creek, and was located near the headwaters of the May River. Mean pH values in the open water stations were significantly higher than values measured in the large tidal creeks (p-value < 0.0010), but this difference (7.7 vs. 7.6) is not likely to be ecologically significant.

pH values collected at 2002 SCECAP stations were similar to the May River sites (Figure III-41). May River open water stations had slightly higher pH values than mean values for 2002 SCECAP open water sites, but these differences were not statistically significant (p-value = 0.53).

Biochemical Oxygen Demand

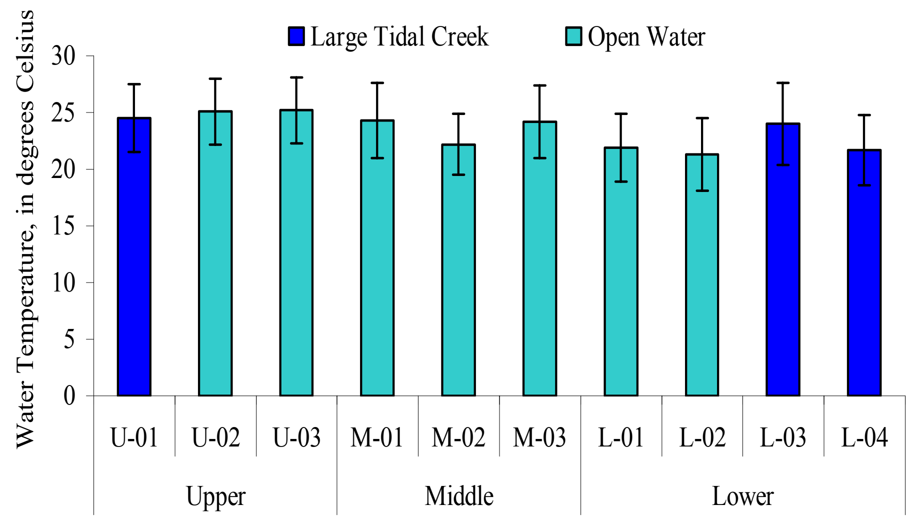
Average BOD ranged from 0.8 mg/L at M-02, an open water site, to 1.6 mg/L at L-03, a large tidal creek site, in the May River during the sampling period (Figure III-42). BOD were similar among the individual sites in the May River. However, large tidal creek sites had higher BOD than the open water sites (p-value = 0.0407). BOD did vary significantly among seasons (p-value = 0.0036, Figure III-42). The seasonal range in BOD was generally less than 1 mg/L at open water and large tidal creek sites, except L-03, which had a seasonal BOD range of 2.9 mg/L. Open water and large tidal creek sites in the May River had higher BOD than 2002 SCECAP sites (p-value < 0.0001, Figure III-42).

Like for headwater tidal creeks, BOD was correlated with turbidity, TSS, and particulate forms of nitrogen and carbon (Appendix III-5).

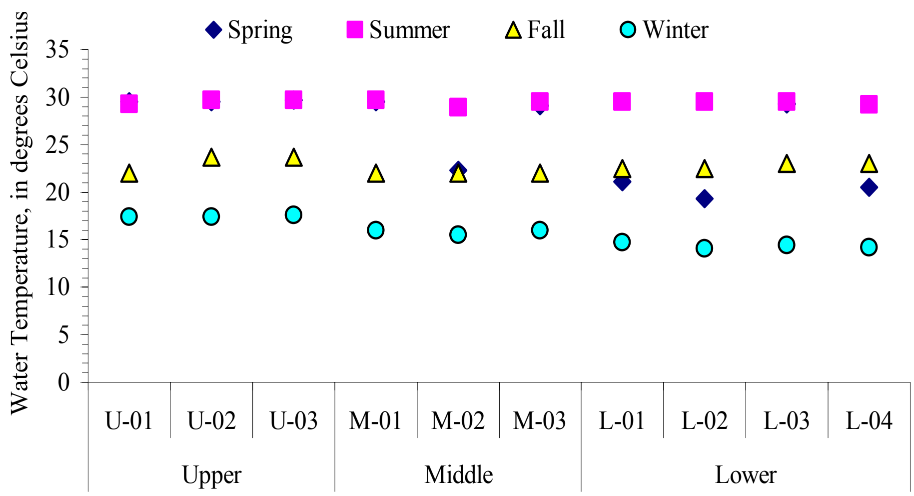
Turbidity

Mean turbidity in open water and large tidal creeks in the May River ranged from 5.5 NTU at M-03 to 110 NTU at L-03 (Figure III-43) and were considered similar among sites and habitat types. The open water site L-03 was the only site with mean turbidity above the SCDHEC maximum criteria level of 25 NTU. Only two samples had turbidity above the maximum criteria level: the summer sample at M-01 (32 NTU) and the spring sample at L-03 (423 NTU). Turbidity during the summer of 2002 were similar between May River and 2002 SCECAP open water and large tidal creek sites (Figure III-43).

A.



B.



C.

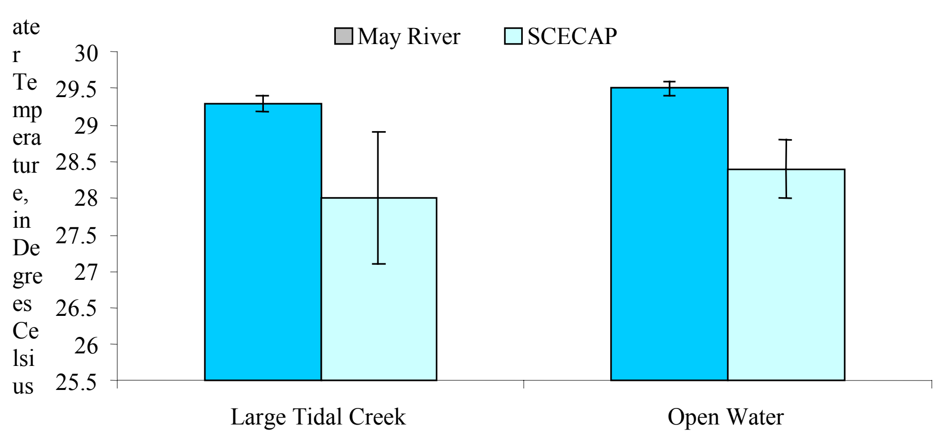
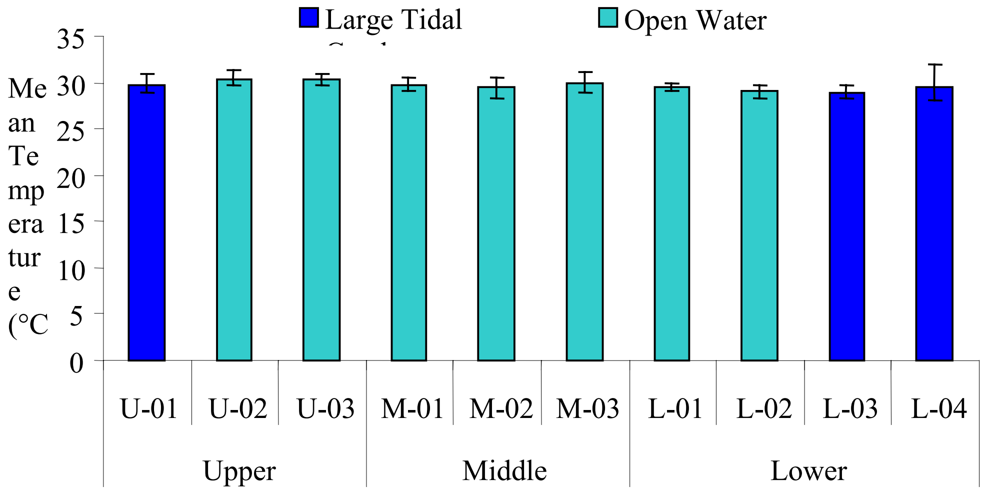


Figure III-34. Mean seasonal water temperature measured from point samples among sites (A.), seasons (B.), and mean summer water temperature in large tidal creek and open water sites in the May River and nearby SCECAP sites (C.), 2002 - 2003. Error bars represent 1 standard error.

A.



B.

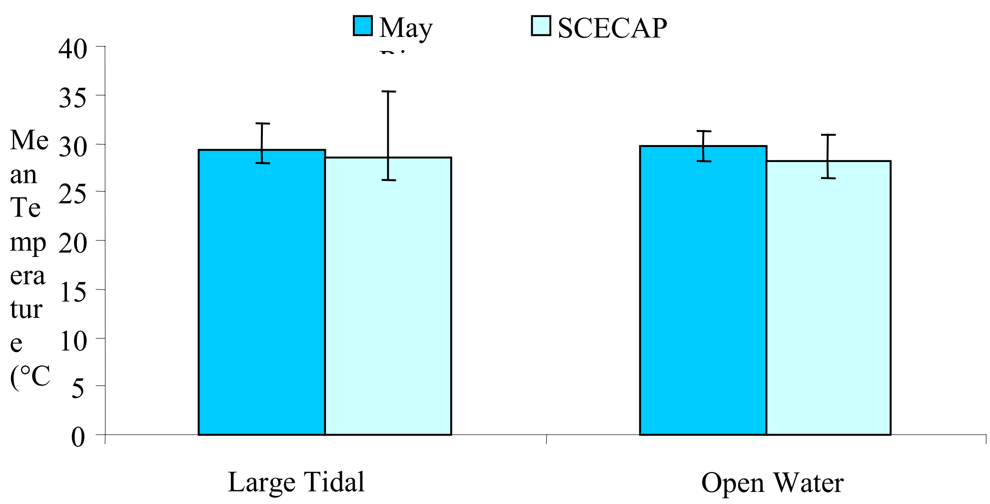


Figure III-35. Mean bottom water temperatures observed during the semi-continuous summer deployment at the May River large tidal creek and open water sites (A.) and compared to 2002 SCECAP stations (B.). Error bars represent 1 standard error.

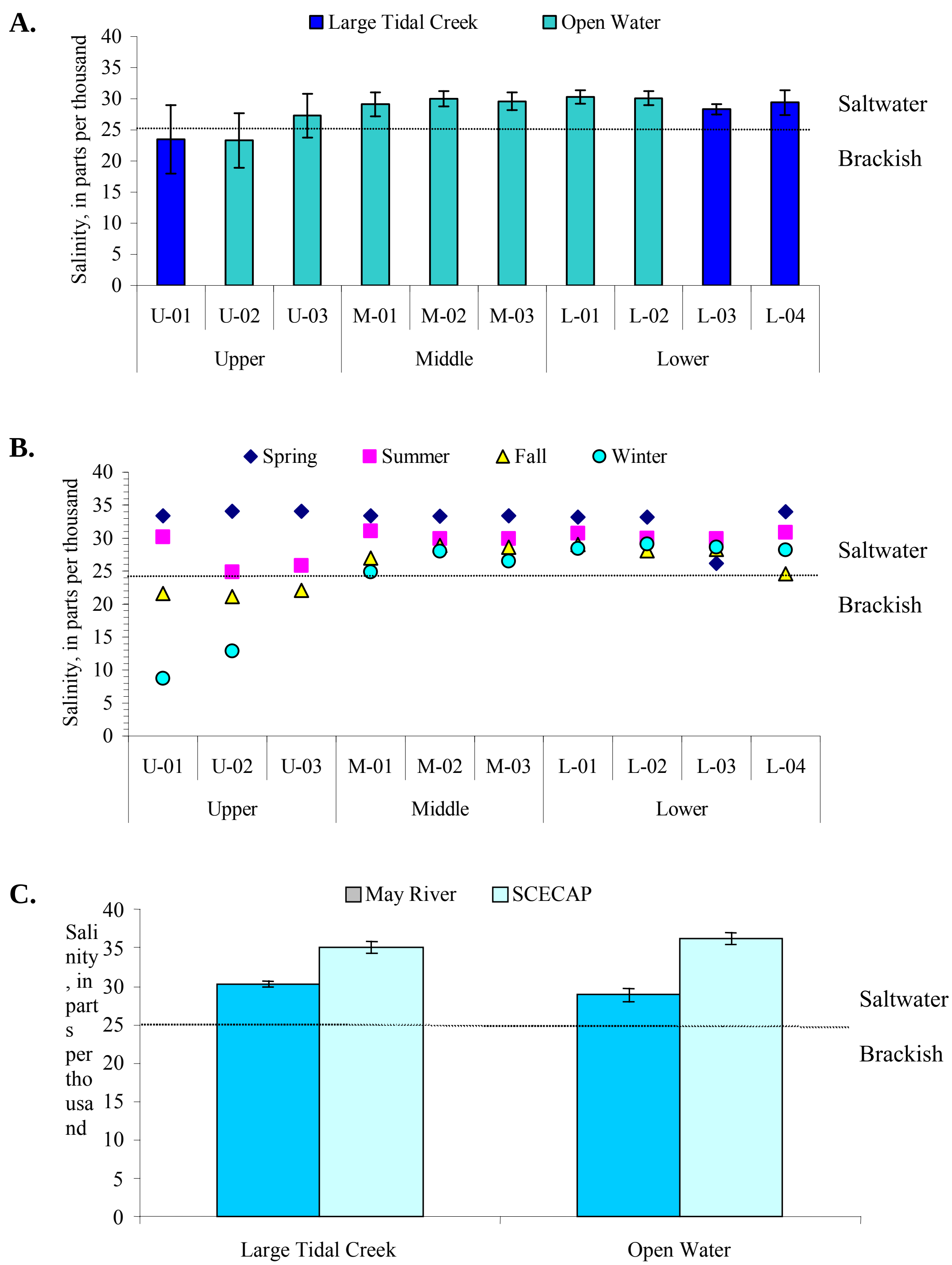
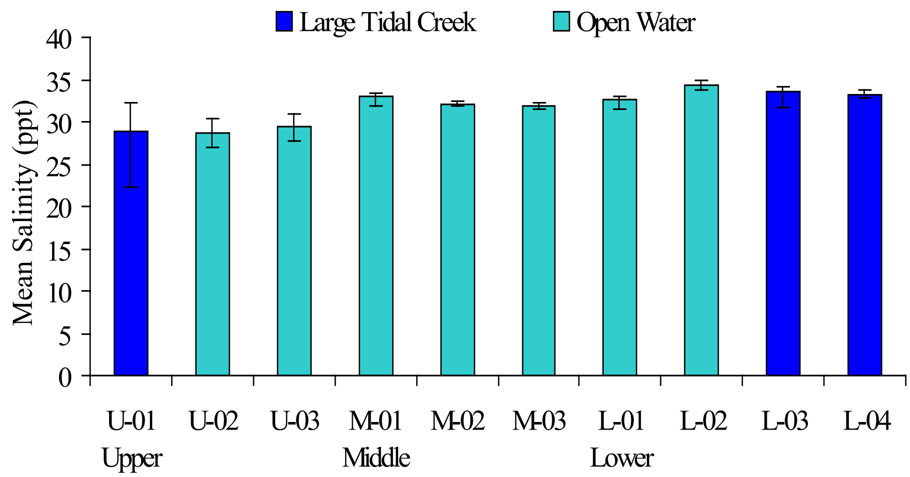


Figure III-36. Mean seasonal point-sampled salinity among sites (A.), by season (B.), and mean summer salinity in May River sites compared to nearby SCECAP sites (C.), 2002-2003. Error bars represent 1 standard error.

A.



B.

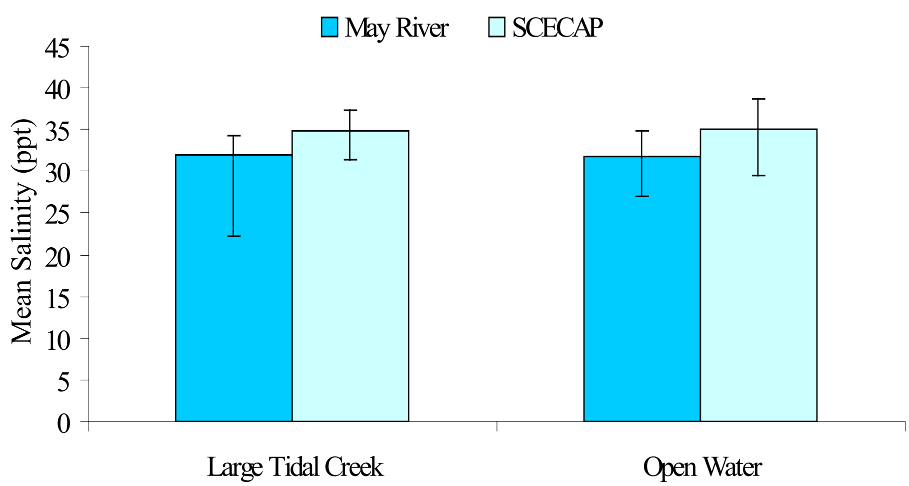


Figure III-37. Mean bottom salinity collected during the semi-continuous summer deployment at the May River large tidal creek and open water sites (A.) compared to 2002 SCECAP sites (B.). Error bars represent 1 standard error.

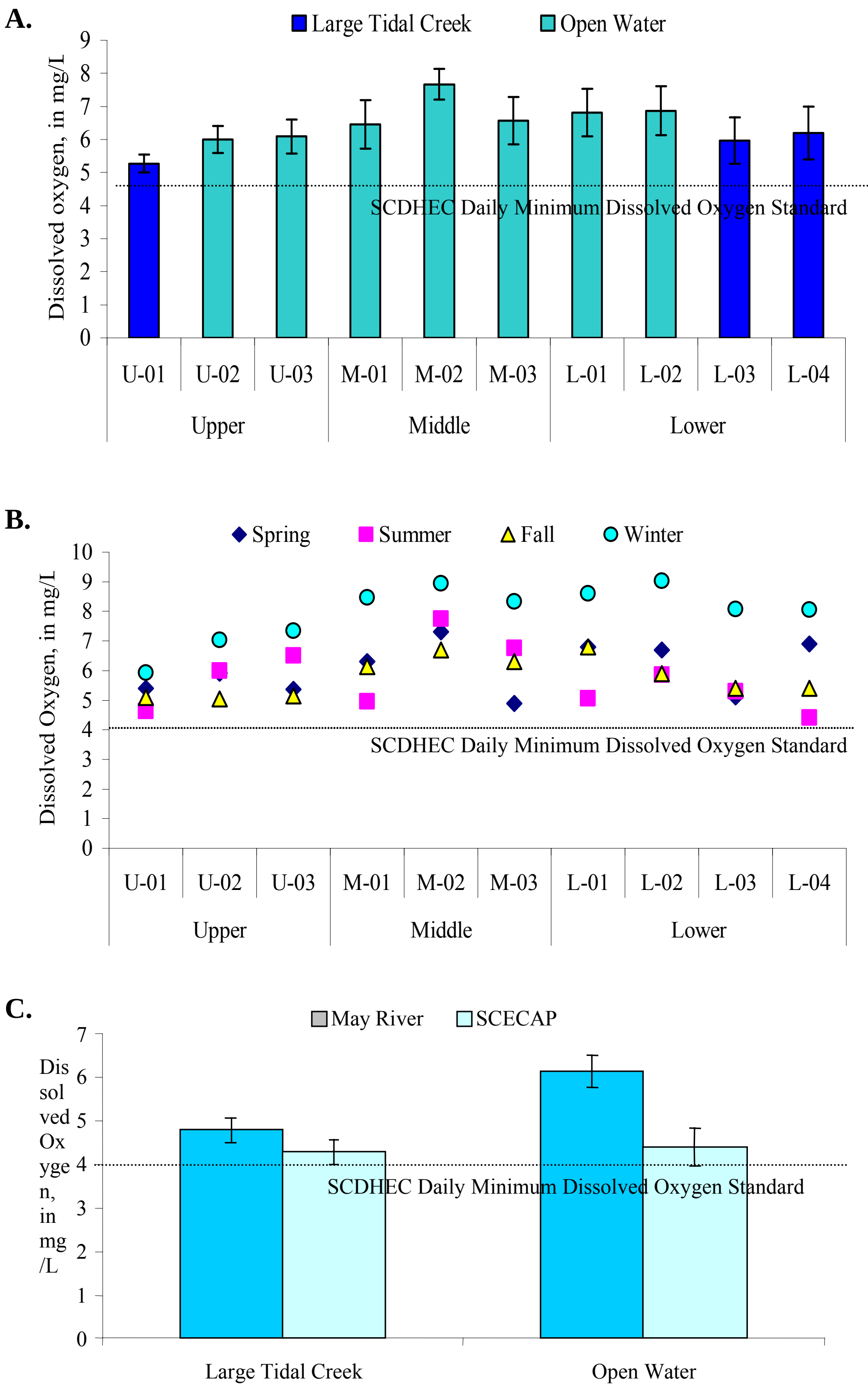
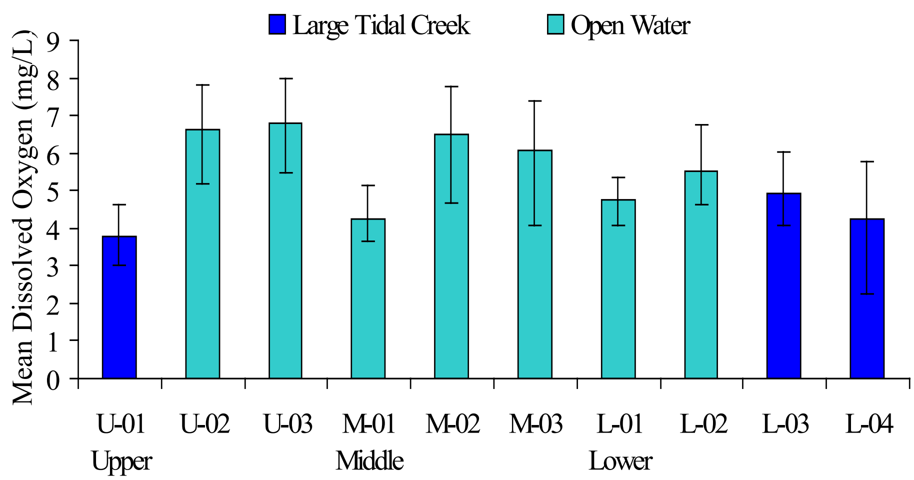


Figure III-38. Mean seasonal point-sampled dissolved oxygen among sites (A.) and seasons (B.), and mean summer dissolved oxygen concentrations in large tidal creek and open water sites in the May River and nearby SCECAP sites (C.), 2002-2003. Error bars represent 1 standard error.

A.



B.

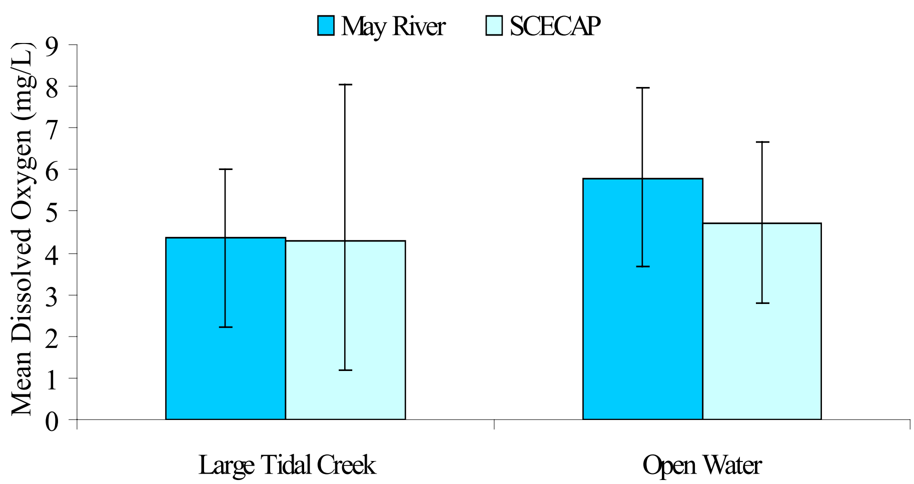


Figure III-39. Mean bottom dissolved oxygen during the semi-continuous summer deployment from the May River large tidal creek and open water sites (A.), and compared to 2002 SCECAP stations (B.). Error bars represent 1 standard error.

A.

B.

C.

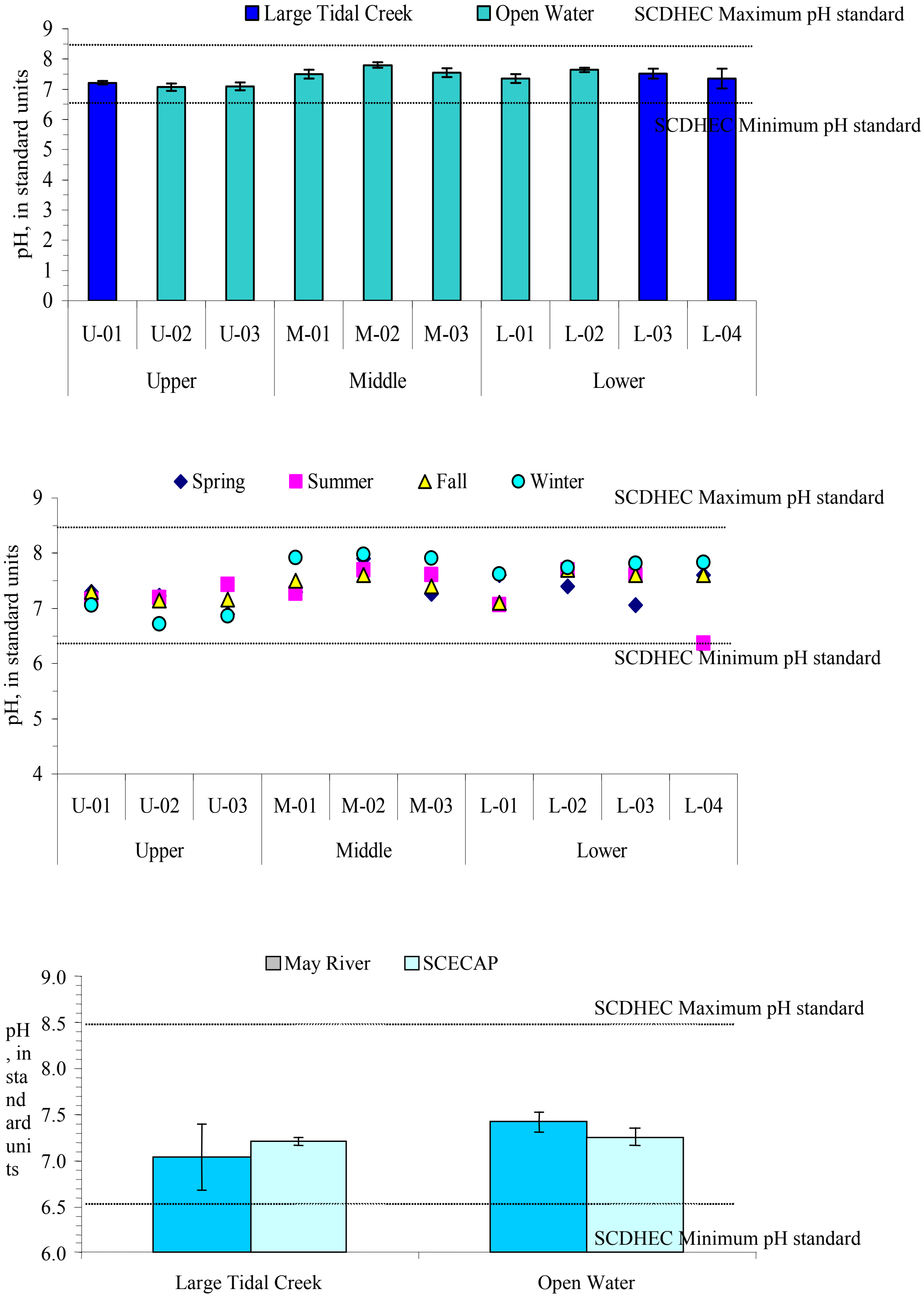
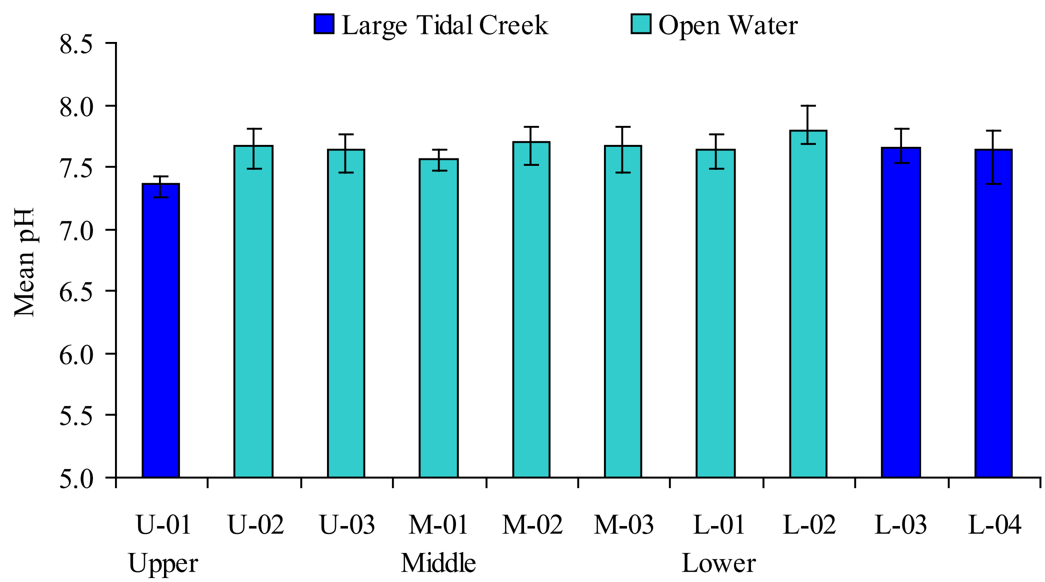


Figure III-40. Mean seasonal point-sampled pH among sites (A.) and among seasons (B.), and mean summer pH in large tidal creek and open water sites in the May River and nearby SCECAP sites (C.), 2002-2003. Error bars represent 1 standard error.

A.



B.

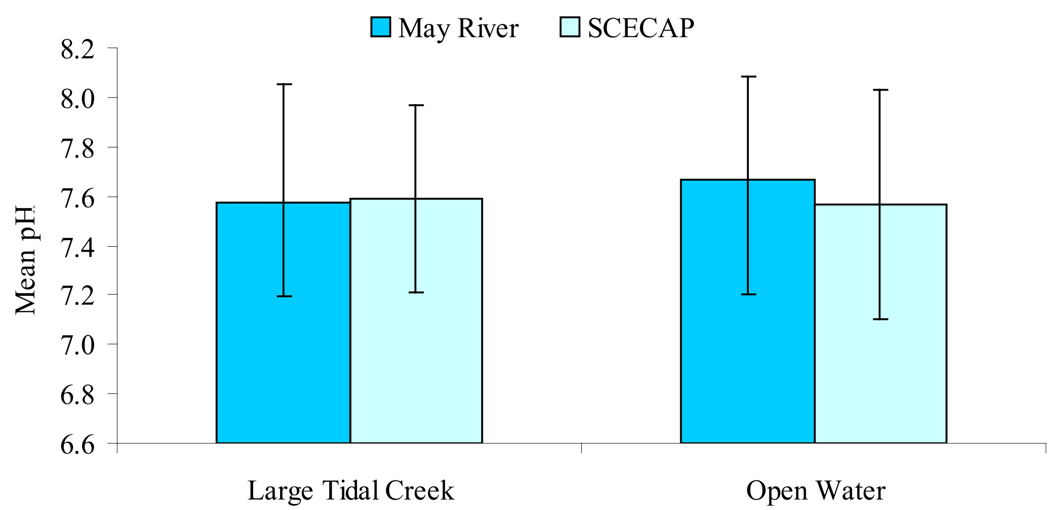


Figure III-41. Mean bottom pH collected during the semi-continuous summer deployment at the May River large tidal creek and open water sites (A.), and compared to 2002 SCECAP sites (B.). Error bars represent 1 standard error.

Nutrients

Mean total nitrogen concentrations ranged from 0.39 mg/L at M-02 to 0.96 mg/L at U-01. Total nitrogen was almost entirely composed of TKN. Average TKN concentrations in open water and large tidal creek sites ranged from 0.38 mg/L at M-02 to 0.95 mg/L at U-01 in the May River (Figure III-44). The large tidal creek site U-01 had significantly higher TKN concentrations than the open water sites L-02, M-02, and M-03 and the large tidal creek site L-04 (p-value = 0.0159). However, open water sites had similar TKN concentrations to large tidal creek sites. During the period of sampling, concentrations of TKN were similar among seasons in the open water and large tidal creek sites in the May River (Figure III-44). The greatest seasonal variability in TKN occurred in sites U-01 and L-03. During the summer of 2002, TKN concentrations were similar between May River and 2002 SCECAP sites (Figure III-44).

In the May River, mean ammonia concentrations ranged from 0.034 mg/L in L-02 to 0.123 mg/L in U-01 (Figure III-45). These mean concentrations were much lower than the average TKN concentrations, indicating that total organic nitrogen comprised the larger fraction of the TKN than inorganic nitrogen. Ammonia concentrations were not different among sites or between habitat types during the period of sampling. However, a significant difference among seasons was identified (p-value = 0.0446, Figure III-45). The summer ammonia concentrations at the May River open water and large tidal creek sites were similar to the 2002 SCECAP sites (Figure III-45).

Mean nitrate plus nitrite concentrations in the open water and large tidal creek sites in the May River were low and ranged from 0.010 mg/L at M-03 to 0.025 mg/L at L-02 (Figure III-46). Nitrate plus nitrite concentrations were similar among sites and between habitat types for the period of sampling, but different among seasons. Seasonally, nitrate plus nitrite concentrations were highest in the fall and lowest in the winter (p-value < 0.0001, Figure III-46). SCECAP and May River sites had similar summer nitrate plus nitrite concentrations in 2002 (Figure III-46).

Dissolved nitrogen concentrations in the open water and large tidal creek sites in the May River included all dissolved forms of nitrogen (ammonia, organic nitrogen, nitrate plus nitrate). Although nutrient criteria for estuarine systems have not been established by SCDHEC, NOAA established ranges of dissolved nitrogen concentrations that represented High, Medium, and Low levels: High > 1 mg/L, Medium 0.1 to 1 mg/L, and Low < 0.1 mg/L (Bricker and others, 1999). Based on these ranges, the majority of the May River open water and large tidal creek sites had seasonal and mean dissolved nitrogen levels in the Medium range of the NOAA guidelines (Figure III-47). Mean dissolved nitrogen levels ranged from 0.26 mg/L at M-02 to 0.61 mg/L at U-01 in the May River.

Mean total phosphorus concentrations ranged from 0.057 mg/L at L-01 to 0.241 at U-02 in the May River during the period of sampling (Figure III-48). Total phosphorus concentrations did not vary among sites, among seasons, or between habitat types. The largest seasonal variability in total phosphorus concentrations occurred at U-02 and L-03 (Figure III-48). Summer phosphorus concentrations in open water and large tidal creek sites in the May River were significantly higher than concentrations at 2002 SCECAP sites (p-value = 0.0142, Figure III-48).

Although nutrient criteria for estuarine systems have not been established by SCDHEC, NOAA established ranges of dissolved phosphorus concentrations that represented High, Medium, and Low levels: High > 0.1 mg/L, Medium 0.01 to 0.1 mg/L, and Low < 0.01 mg/L (Bricker and others, 1999). Based on these ranges, the majority of the May River open water and large tidal creek sites had seasonal and mean dissolved phosphorus levels in the Medium range

(Figure III-49). Mean dissolved phosphorus levels ranged from 0.034 mg/L at L-01 to 0.070 mg/L at M-02 in the May River sites.

Total Organic Carbon

Open water and large tidal creek sites in the May River had mean TOC concentrations that ranged from 2.9 mg/L at L-02 to 8.5 mg/L at U-01 (Figure III-50). U-01 had higher TOC concentrations than L-01, L-02, and L-03 for the period of sampling (p-value = 0.0002). TOC concentrations were similar between open water and large tidal creek habitats and did not vary seasonally, however, greater seasonal variation was observed in open water and large tidal creek habitats in the Upper Zone than those in the Middle and Lower zones (Figure III-50). SCECAP open water and large tidal creek sites had greater summer TOC concentrations than the May River sites (p-value = 0.0042, Figure III-50).

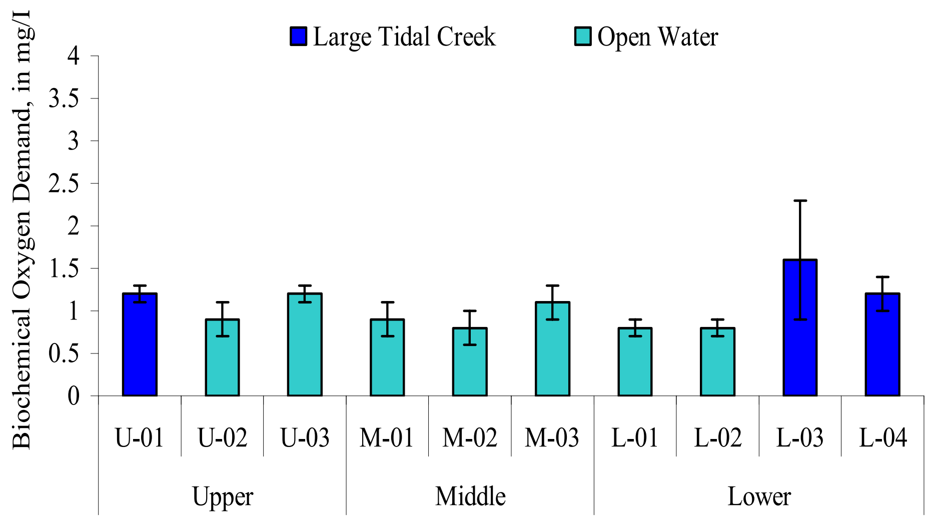
Fecal Coliform/MAR

Fecal coliform densities and Multiple Antibiotic Resistance (MAR) analyses were performed for the ten large tidal creek and open water sites in the May River during the summer of 2002. Nine of the sites exhibited fecal coliform densities less than the SCDHEC standard for shellfish harvesting of 14 CFU/100 mL (SCDHEC, 2001) (Table III-9; Figure III-51). The fecal coliform density for the remaining site (U-01) was only 15 CFU/100 mL. This large tidal creek site was the farthest upstream site sampled. The fecal coliform densities in the May River large tidal creek and open water sites were similar to the fecal coliform densities in the 2002 SCECAP sites (Figure III-51); however, it should be noted that the two methods employed to obtain these results were different.

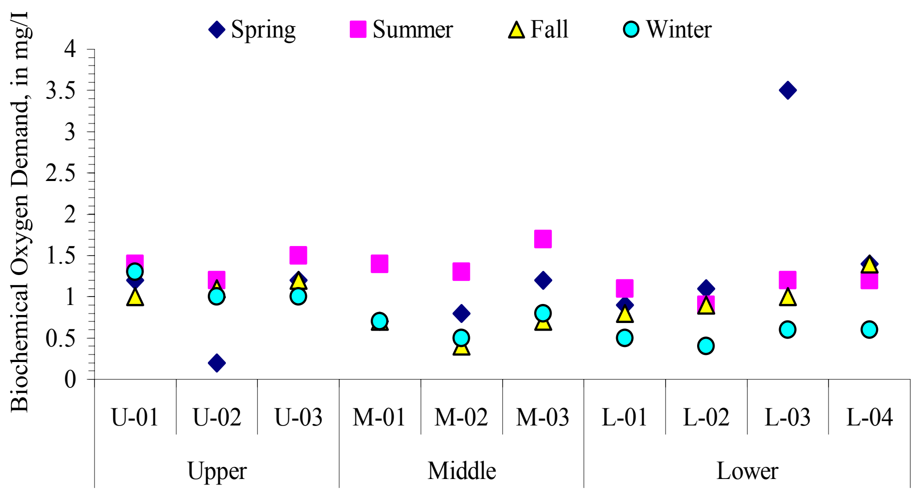
Table III-9. Summer 2002 fecal coliform densities and MAR indices for large tidal creek and open water sites in the May River. Numbers in ( ) are standard error. NC = not calculated due to an insufficient number of *E. coli* isolates.

Site	Fecal Coliform /100 ml	Fecal Coliform /100 ml (log10)[a]	Site MAR Index
U-01	15	1.18	0
U-02	12	1.08	0
U-03	6	0.78	0
M-01	3	0.48	NC
M-02	8	0.9	0.03 (0.02)
M-03	6	0.78	0.01 (0.01)
L-01	1	0	NC
L-02	1	0	NC
L-03	3	0.48	NC
L-04	1	0	NC

A.



B.



C.

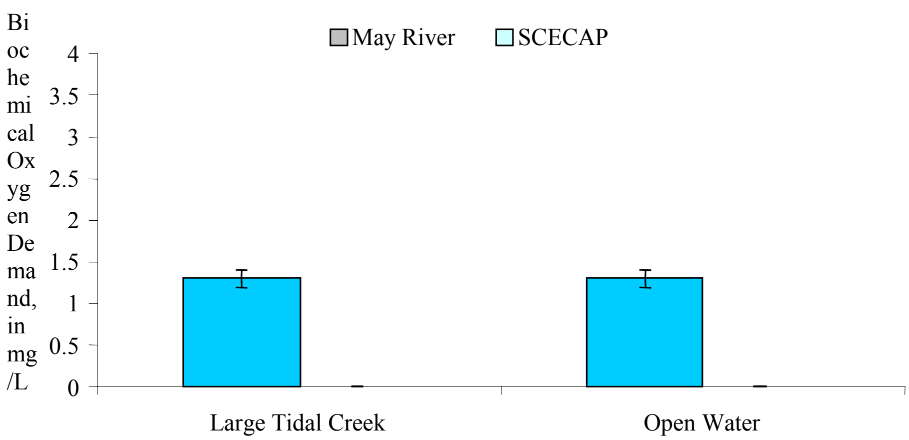


Figure III-42. Mean seasonal point-sampled biochemical oxygen demand among sites (A.) and among seasons (B.), and mean summer biochemical oxygen demand concentrations in large tidal creek and open water sites in the May River (C.), 2002-2003. Error bars represent 1 standard error.

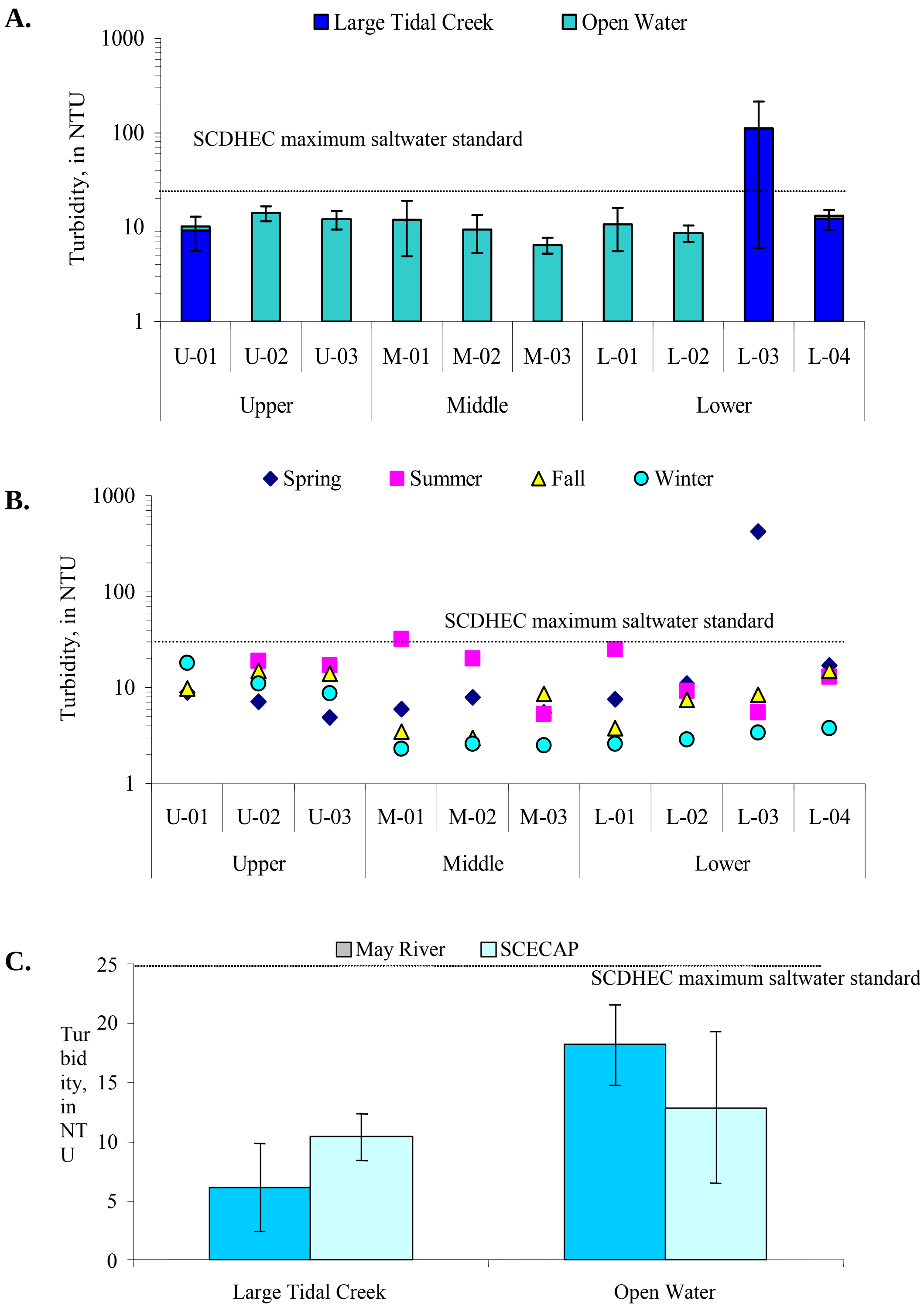


Figure III-43. Mean seasonal point-sampled turbidity among sites (A.) and among seasons (B.), and mean summer turbidity levels in large tidal creek and open water sites in the May River and nearby SCECAP sites (C.), 2002-2003. Error bars represent 1 standard error.

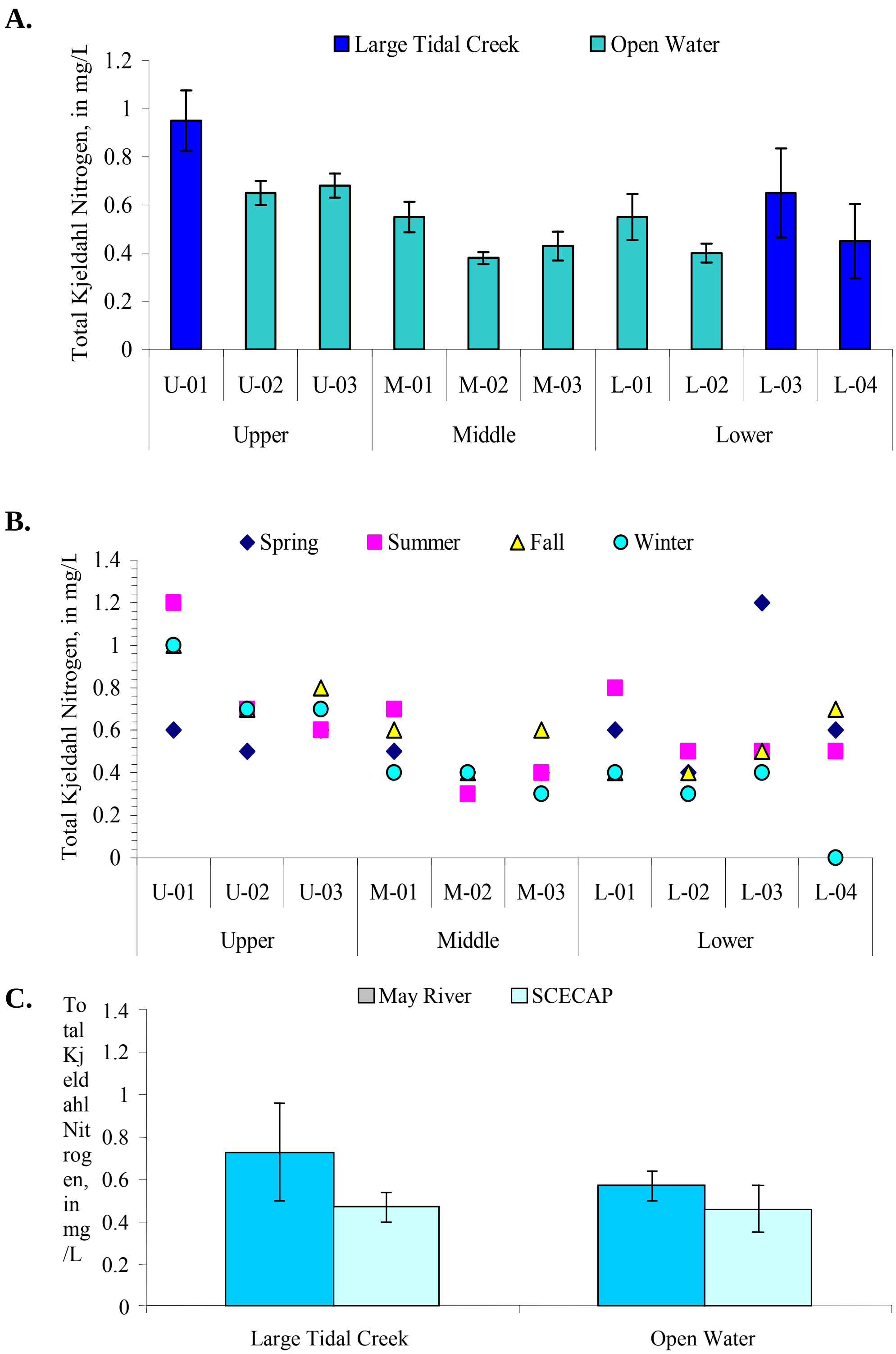


Figure III-44. Mean seasonal point-sampled total Kjeldahl nitrogen among sites (A.) and among seasons (B.), and mean summer total Kjeldahl nitrogen levels in large tidal creek and open water sites in the May River and nearby SCECAP sites (C.), 2002-2003. Error bars represent 1 standard error.

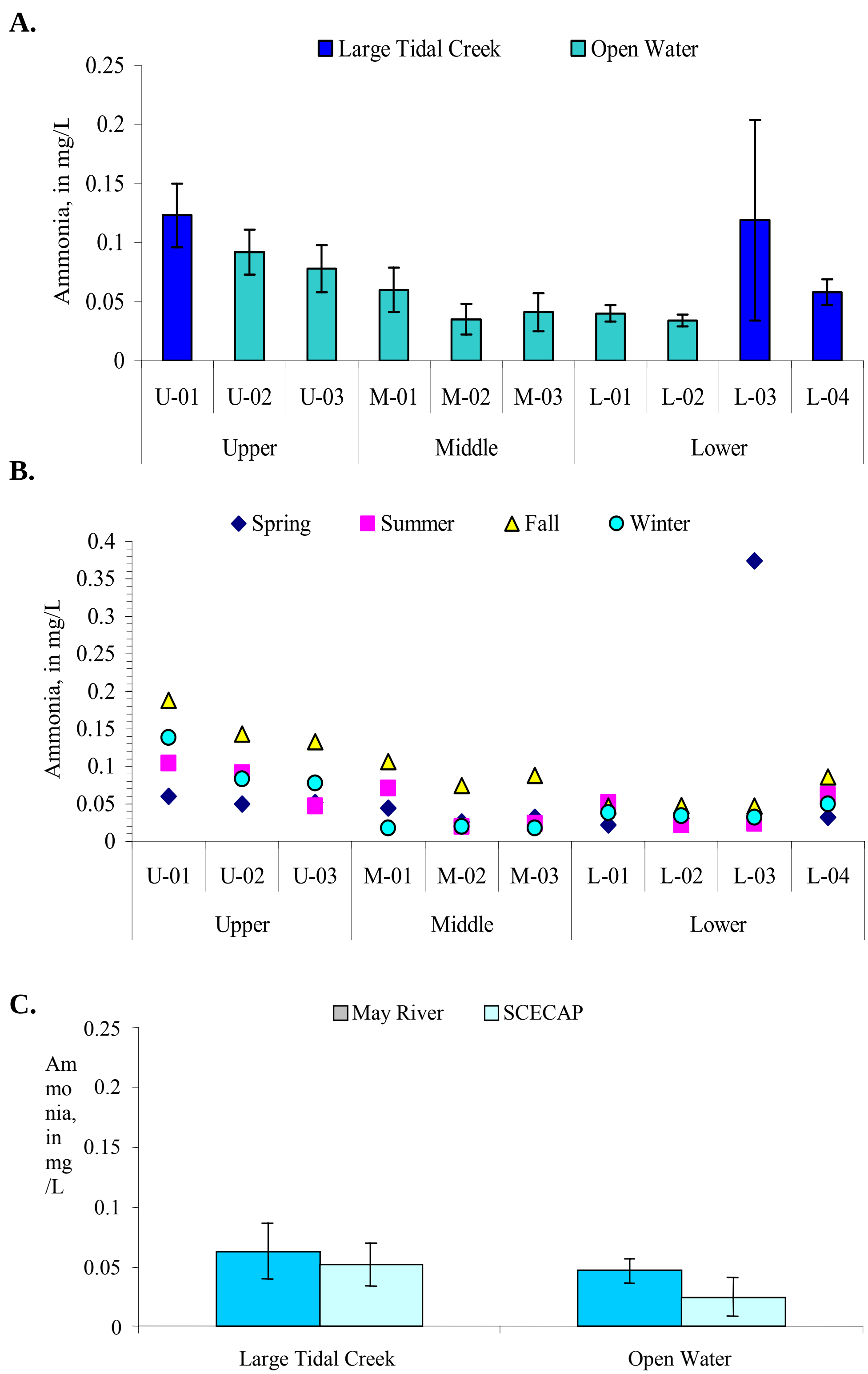


Figure III-45. Mean seasonal point-sampled ammonia concentrations among sites (A.) and among seasons (B.), and mean summer ammonia concentrations in large tidal creek and open water sites in the May River and nearby SCECAP sites (C.), 2002-2003. Error bars represent 1 standard error.

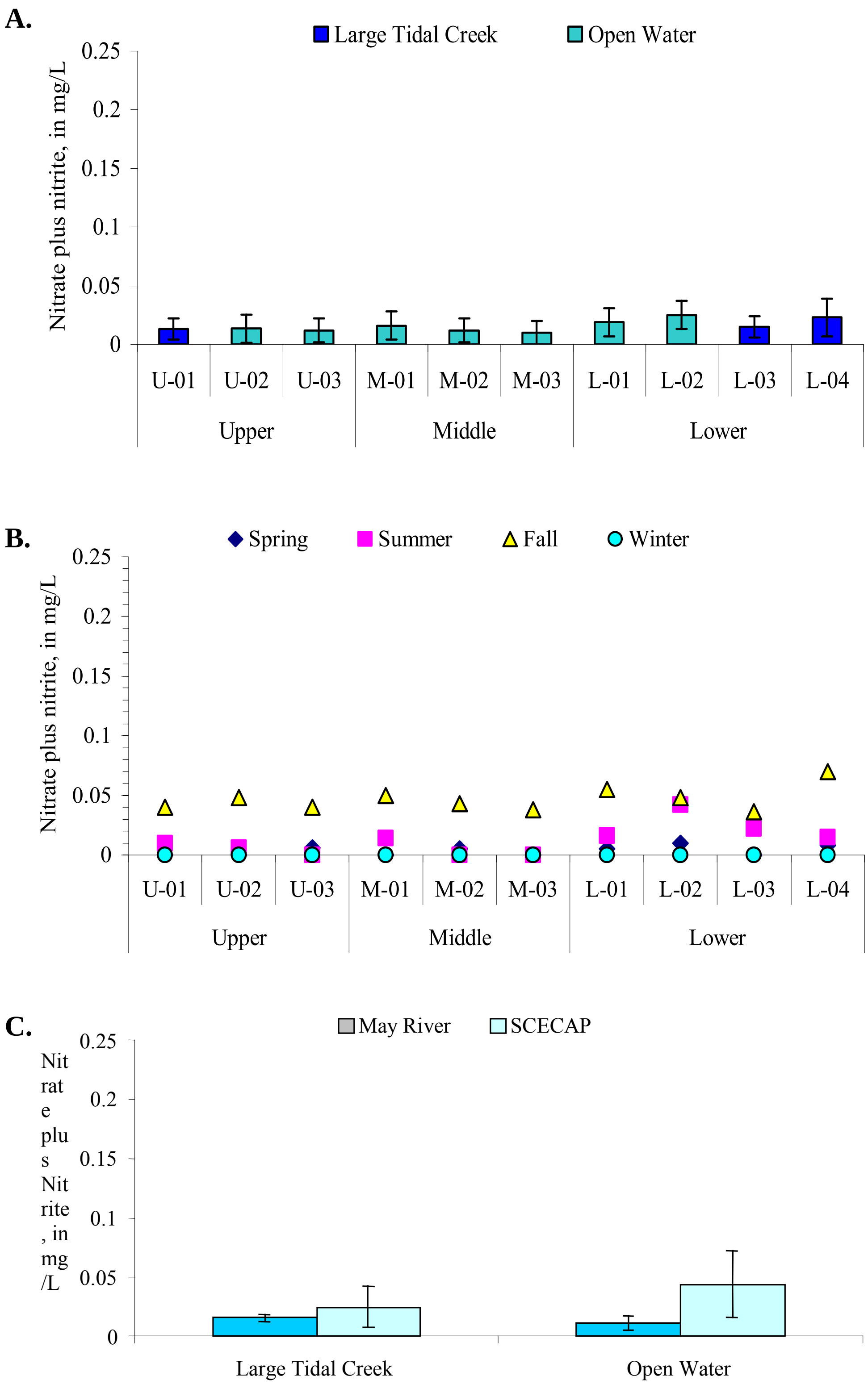


Figure III-46. Mean seasonal point-sampled nitrate plus nitrite among sites (A.) and among seasons (B.), and mean summer nitrate plus nitrite levels in large tidal creek and open water sites in the May River and nearby SCECAP sites (C.), 2002-2003. Error bars represent 1 standard error.

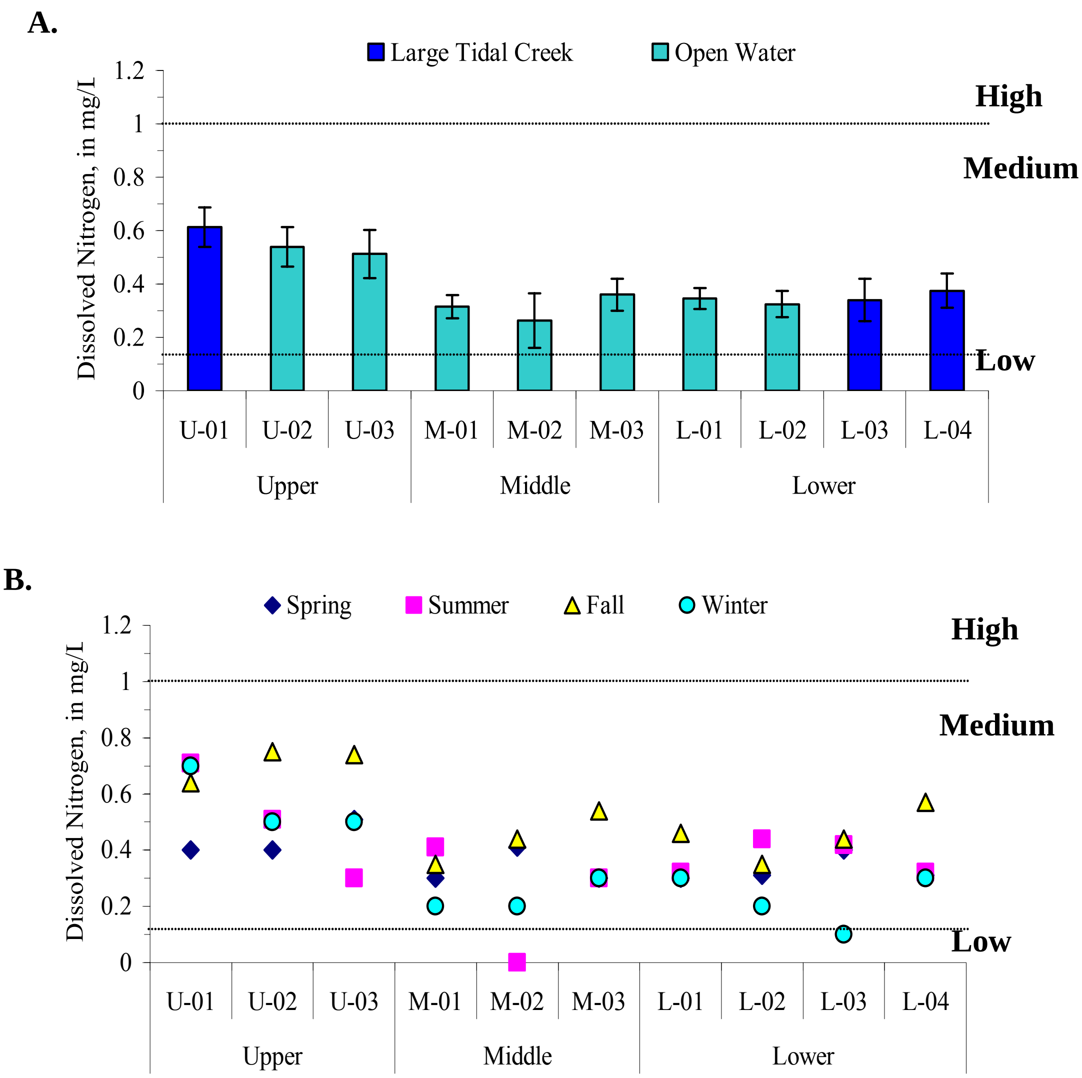


Figure III-47. Mean seasonal point-sampled dissolved nitrogen among sites (A.) and among seasons (B.) in large tidal creek and open water sites in the May River, 2002-2003. Lines represent NOAA guidelines for nutrient enrichment. Error bars represent 1 standard error.

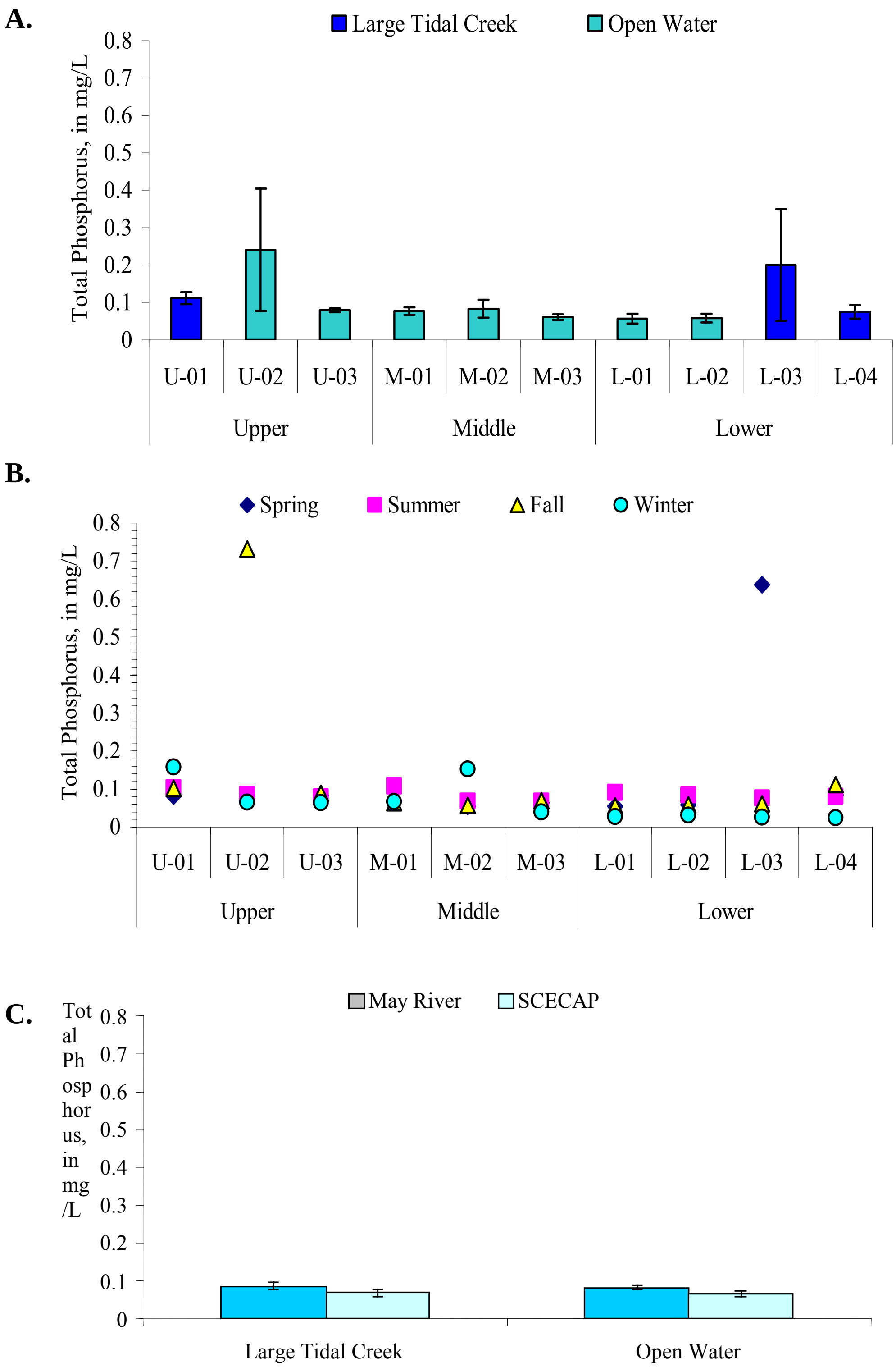


Figure III-48. Mean seasonal point-sampled total phosphorus among sites (A.) and among seasons (B.), and mean summer total phosphorus levels in large tidal creek and open water sites in the May River and nearby SCECAP sites (C.), 2002-2003. Error bars represent 1 standard error.

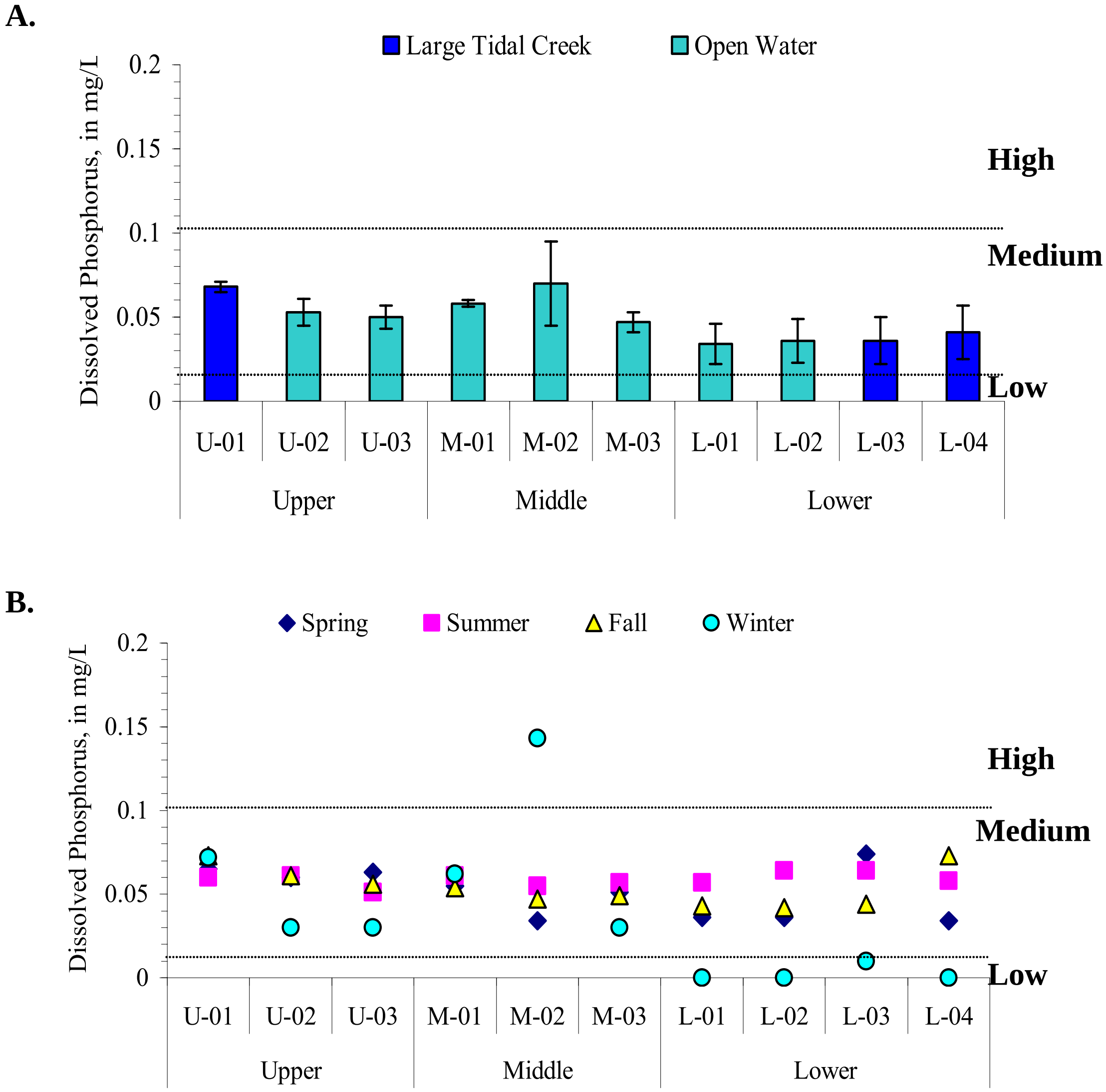


Figure III-49. Mean seasonal point-sampled dissolved phosphorus among sites (A.) and among seasons (B.) in large tidal creek and open water sites in the May River, 2002-2003. Lines represent NOAA guidelines for nutrient enrichment. Error bars represent 1 standard error.

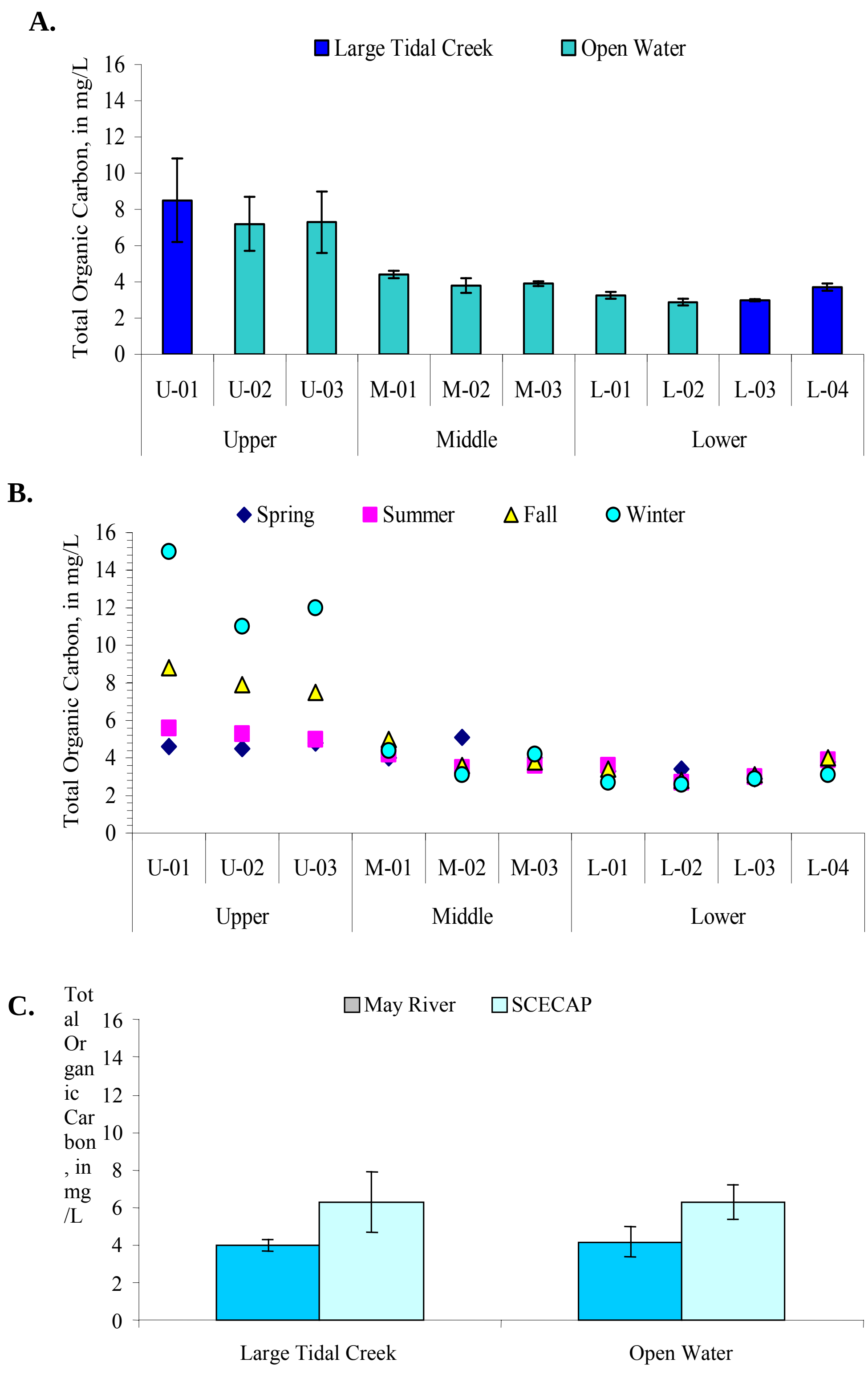
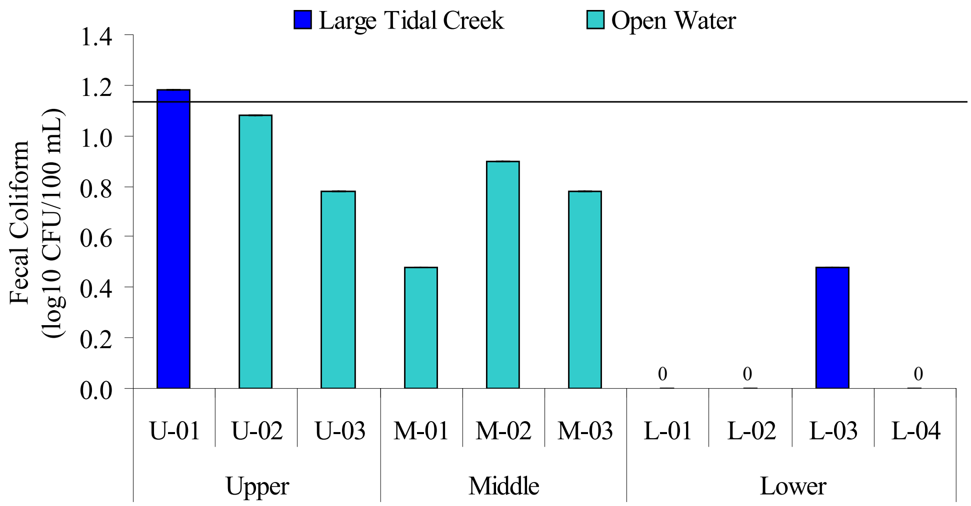
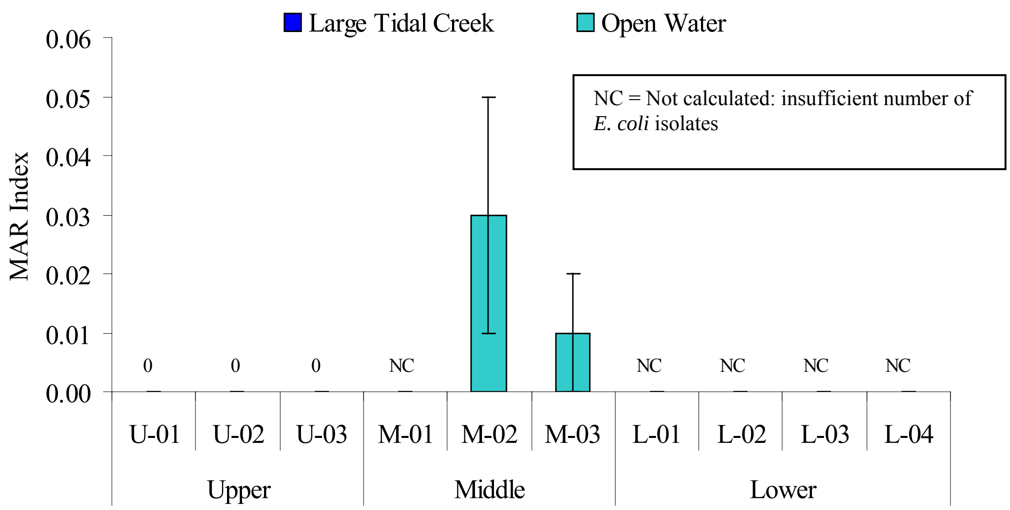


Figure III-50. Mean seasonal point-sampled total organic carbon among sites (A.) and among seasons (B.), and mean summer total organic carbon levels in large tidal creek and open water sites in the May River and nearby SCECAP sites (C.), 2002-2003. Error bars represents 1 standard error.

A.



B.



C.

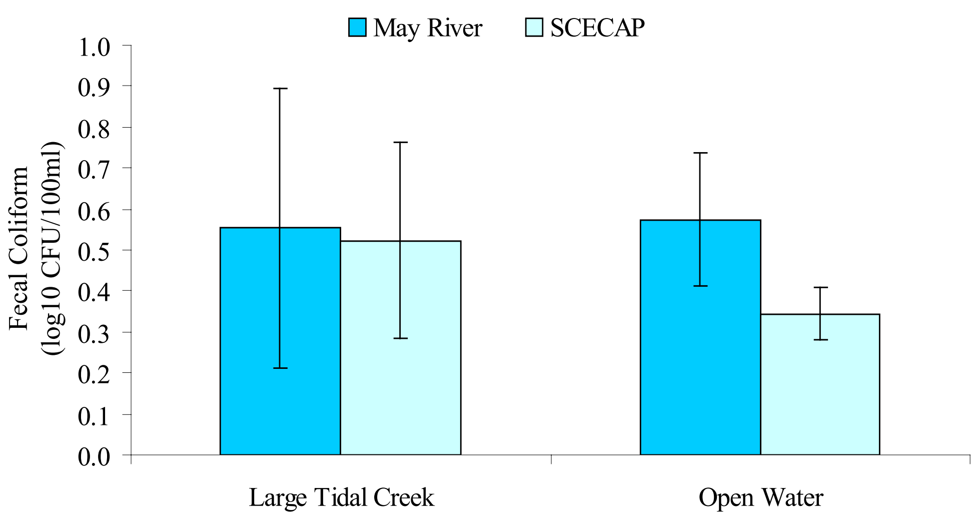


Figure III-51. Summary of fecal coliform densities (A.) and MAR index values in the large tidal creek and open water sites of the May River (B.). May River sites were compared to 2002 SCECAP sites (C.) Error bars represent 1 standard error.

MAR analysis could not be performed for five of the ten sites due to low fecal coliform densities and an insufficient number of confirmed *E. coli*. Three of the sites that could be measured had MAR values of zero (Table III-9). Only two of the ten sites showed the presence of any antibiotic resistance (1 to 3%). However, since the 95% confidence intervals for the MAR indices at these two sites included zero, the sites were not considered to exhibit antibiotic resistance. An earlier study conducted in the Broad Creek and Okatee River showed that the undeveloped region of the Okatee River had an overall MAR Index of 1% in 1997 (Van Dolah and others, 2000), which is similar to the present overall MAR Index for the May River (0.64%). Based on the interpretation of these MAR results, the coliform levels at the sites sampled did not appear to be related to anthropogenic sources of contamination.

Phytoplankton

As with the headwater creeks, phytoplankton samples were collected seasonally in large tidal creek and open water sites. Mean chlorophyll-*a* concentrations in the large tidal creek and open water sites were Low to Medium with respect to the eutrophication classification outlined by Bricker and others (1999) (Figure III-52). No consistent patterns in chlorophyll-*a* concentrations were apparent when comparing large tidal creek to open water sites overall, but the five highest mean values were associated with open water locations.

Harmful algae were never relatively abundant in large tidal creek or open water sites, but several known harmful species were present (Appendix III-6a), as opposed to headwater samples which contained only one known harmful species (*Kryptoperidinium foliaceum*). This species was observed once in a large tidal creek (site Lower-04 in May 2002), but never in the open water samples. However, *Scrippsiella,* a close relative of this dinoflagellate that also forms “red tides” and has harmful effects on oysters, was found in two open water samples. The discovery of “PLO’s” (*Pfiesteria*-like organisms) and *Heterosigma akashiwo* in open water samples is significant but not surprising given that these types commonly occur in South Carolina estuarine waters (Lewitus and others, 2003). It is perhaps more surprising that these species were not found in large or headwater tidal creeks. *H. akashiwo* is one of four raphidophyte species that are common bloom-formers in brackish stormwater detention ponds in the South Carolina coastal zone, and is infamous for causing fish kills around the world.

The relative contribution of peridinin to phytoplankton community biomass did not differ significantly when comparing the means of all large tidal creek and open water sites, but decreased from the Upper to the Lower Zone (Appendix III-6b). Alloxanthin and lutein had contrasting patterns in that the relative concentration of the former was significantly greater in open water sites and the latter in large tidal creek sites. Cryptophytes (indicated by alloxanthin) are an opportunistic group of phytoplankton known to prosper under conditions (e.g., low light and low inorganic nutrients) not amenable to the growth of phytoplankton that have a more strict dependency on photosynthesis. The relatively greater abundance of green algae in the large tidal creek sites is supported by the trend in chlorophyll-*b*, which was also relatively higher in the creeks (Appendix III-6b). In summary, our baseline data on marker pigment distribution suggests that: (a) peridinin-containing dinoflagellates contribute the most to phytoplankton biomass in the upper regions of the May River; (b) fucoxanthin-containing groups (most likely diatoms and some dinoflagellates) are evenly distributed (note also that headwater tidal creek levels of fucoxanthin:chlorophyll-*a* are comparable to large tidal creek and open water levels); (c) the contribution of cryptophytes to phytoplankton biomass was highest at open water sites and lowest at headwater creeks; and (d) lutein-containing green algae contribute most to the phytoplankton communities of headwater and large tidal creeks (Appendix III-6b).

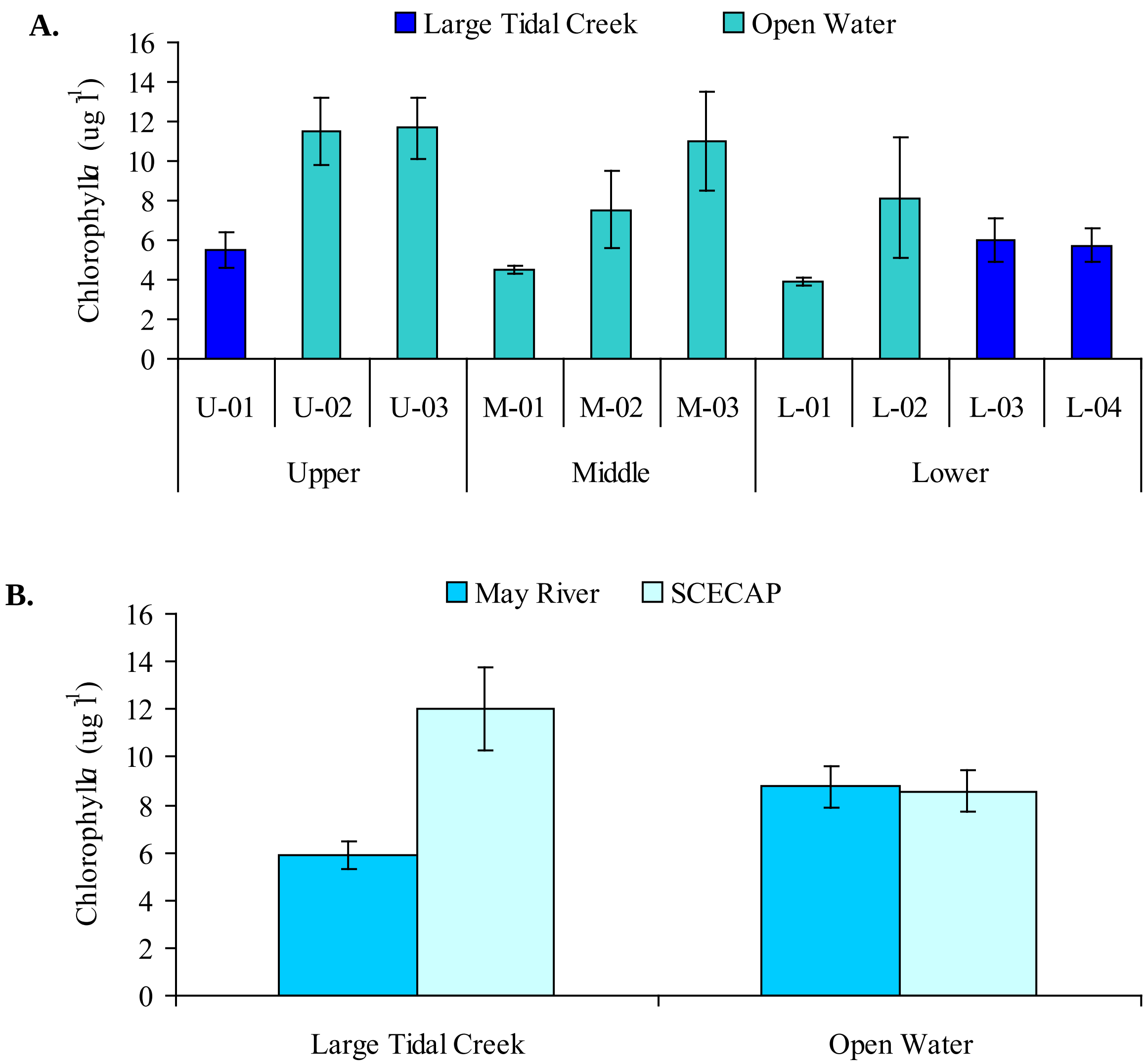


Figure III-52. Mean chlorophyll-*a* concentrations observed at the May River large tidal creek and open water sites (A.), and comparison of chlorophyll-*a* concentrations to 2002 SCECAP sites (B.). Error bars represent 1 standard error.

Sediment Quality

Composition

The percentage of mud (silts and clays) in estuarine sediments can affect both the biological community as well as the bioavailability of certain contaminants to organisms. Sediment composition was similar between large tidal creek and open water sites, and was composed primarily of sand (86%) with only a moderate amount of silts and clays (14%). The highest percentages of silts and clays occurred in the lower portion of the May River, predominantly in the form of clay (Figure III-53).

Sediments collected from the May River large tidal creek stations had significantly more sand content compared to those collected from the 2002 SCECAP tidal creek sites (p-value < 0.05; Figure III-53) when compared statistically, the difference was less than 10 % and both sites had more than 75 % sand. Sediments collected at the May River open water stations had less sand on average than the 2002 SCECAP sites (86 vs. 94 %, p-value = 0.013), but this difference is not likely to significantly impact comparisons of the biota present. In general, the sediment composition for the May River versus 2002 SCECAP sites probably had little influence on the benthic community composition observed at those sites, since all sites represented predominantly sandy sediments.

Contaminants

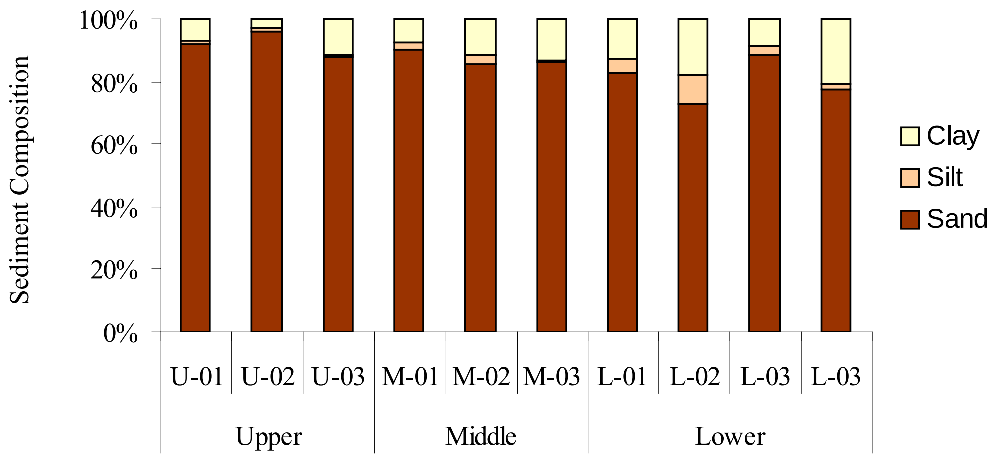
Sediment contaminant levels at the large tidal creek and open water stations were relatively low in the May River. None of the 24 contaminants that are commonly used in evaluating possible bioeffects based on sediment toxicity assays were above the effects range-low (ERL) concentrations (Access® database). Additionally, the integrated measure of these contaminants, or effects range-median quotient (ERMQ), at each of the sites was also well below levels that have been shown to have a moderate to high risk of observing adverse benthic community effects (Hyland and others, 1999). The mean ERMQ value among all sites was 0.0038 with a range of values from 0.0013 to 0.0067 (Table III-10). In comparison, the threshold ERMQ concentration that indicates a moderate risk of observing adverse effects in benthic communities is 0.0200.

Sediment contaminant levels observed at the 10 SCECAP sites used for comparison were also quite low, confirming that these were relatively pristine locations (Table III-10). As in the May River, none of contaminants used for evaluating bioeffects were at concentrations above ERL levels and the mean ERMQ concentration among these sites was 0.0062 (range of 0.0012 to 0.0114). The relatively low contaminant concentrations found at these sites and those in the May River may be largely due to the generally low silt/clay (mud) content observed at these sites, since contaminants tend to bind with these fine sediments compared to quartz sand and shell hash.

Toxicity

Toxicity results based on the whole-sediment Microtox® and seed clam assays identified some toxicity effects for one of the two assays at one tidal creek and four open water sites in the May River (Table III-10). None of the sediments at these sites caused significant toxicity in both of the assays, which would be more indicative of a serious contaminant problem.

A.



B.

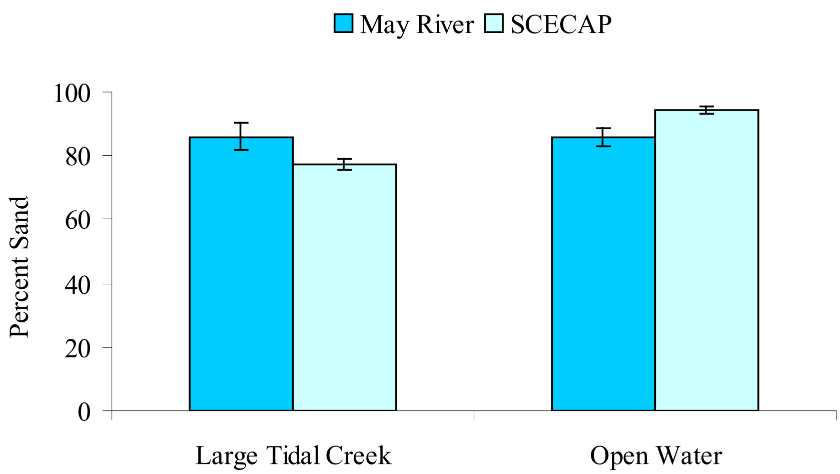


Figure III-53. Sediment composition observed at the May River large tidal creek and open water sites (A.) and compared to the 2002 SCECAP sites (B.). Error bars represent 1 standard error.

Table III-10. Results of sediment bioassay test and effects range-median quotient (ERMQ) values observed in sediments collected from the May River large tidal creek and open water sites. Shaded values indicate toxicity.

Station	Percent SiltClay	Microtox EC50	Clam Growth	ERMQ
U-01	8	1.44	83	0.0013
U-02	4	14.62	3	0.0022
U-03	12	1.87	85	0.0041
M-01	10	0.52	85	0.0016
M-02	14	0.50	75	0.0034
M-03	14	0.25	90	0.0035
L-01	17	0.72	95	0.0064
L-02	27	0.43	100	0.0067
L-03	12	0.96	103	0.0023
L-04	22	0.18	91	0.0067
SCECAP Stations	**Percent SiltClay**	**Microtox EC50**	**Clam Growth**	**ERMQ**
RO026001	4	4.42	120	0.0038
RO026005	9	4.56	26	0.0051
RO026007	7	2.27	59	0.0073
RO026019	1	16.44	80	0.0012
RO026025	7	3.07	129	0.0054
RO026027	8	5.62	127	0.0052
RO026151	2	16.97	84	0.0032
RT022009	27	0.19	89	0.0114
RT022015	18	0.47	115	0.0080
RT022019	23	0.18	144	0.0114

Both the Microtox® and seed clam assays can result in "false positive" results due to their high sensitivity, so SCECAP criteria would only score these sites as marginal for toxicity based on the uncertainty of these assays. Site U-02, which had some of the lowest benthic measures among the May River sites (see next section), had very low clam growth compared to all other sites, but a very high Microtox® EC_{50} (suggesting no toxicity). The low clam growth may be indicative of the presence of an unmeasured contaminant that adversely affects this species, since we also observed some indication of near-marginal conditions in the benthic community.

Toxicity results at the 2002 SCECAP sites showed similar results with six of the ten stations showing toxicity in one of the two assays. As noted for the May River, this may represent "false positive" test results since the contaminant levels at these sites were quite low.

Biological Quality

Benthic Community

Approximately 6,900 benthic organisms representing 163 taxa were collected from May River open water and large tidal creek sites (Access® database). Mean abundance of the benthic organisms ranged from 2,308 to 13,533 individuals/m^2 among the 10 stations sampled (Figure III-54, Table III-11), with a significantly greater mean abundance observed in open water compared to large tidal creek sites (p-value = 0.005). The open water station M-02 had the highest mean abundance of benthos and the tidal creek station L-04 had the lowest mean abundance, but there were no clear trends in faunal abundance among sites within each habitat type (i.e., open water versus large tidal creek) related to station location along the length of the river. When all May River stations were considered collectively, the mean abundance of benthos among sites within each habitat was not significantly different from the mean abundance of fauna collected in comparable habitats from the 2002 SCECAP sites (p-value > 0.05; Figure III-54, Table III-12).

Similar patterns were observed with respect to the number of benthic taxa collected at the May River stations, which provides an indication of faunal diversity. The mean number of taxa at the May River stations ranged from 15 to 42, with significantly more species collected, on average, at the open water sites compared to the large tidal creek sites (p-value = 0.002; Figure III-55, Table III-11). As with the faunal abundance data, there were no clear patterns in the number of species found along the river gradient within each habitat type. Station U-02 had a significantly lower mean number of species compared to the other open water sites except M-03. As noted for the faunal abundance data, this difference may be related to some sediment toxicity effects at that site based upon results from the seed clam assay. There were no significant differences in the mean number of taxa collected between the May River open water or large tidal creek sites and the 2002 SCECAP sites (p-value > 0.05; Figure III-55, Table III-12).

Another measure of species diversity is the Shannon-Weaver Index (H'), which incorporates a measure of the number of species and the relative abundance of those species present at a site. Measures of H' at the May River sites ranged from 2.5 to 4.31 compared to 2.19 to 3.97 at the 2002 SCECAP sites (Table III-11). Among the May River stations, only U-02 and L-04 had H' values less than 3.0, which was probably due to the relatively low number of taxa observed at these sites combined with a high numerical dominance by only a few species.

Polychaete worms were the dominant taxa in both open water and large tidal creek stations, comprising approximately 75% of the total abundance (Figure III-56). Amphipods (small crustaceans) made up 14% of the total abundance at open water stations, and 6% of the overall abundance at large tidal creek stations. Oligochaetes comprised 4% and 7% of the total abundance at open water and large tidal creek stations, respectively. Molluscs (snails and clams) made up approximately 5% of the total abundance in both open water and large tidal creek stations. Organisms falling into the "other taxa" category comprised 3% and 7% of the total abundance in open water and tidal creek stations, respectively.

Ten species comprised more than 60% of the total faunal abundance (Table III-11). Nine of these taxa were polychaete worms, including *Streblospio benedicti*, *Mediomastus* sp., and *Spiochaetopterus costarum*, which are commonly found in high abundances in South Carolina estuaries. One amphipod species, *Ampelisca abdita*, which is considered to be pollution sensitive and is widely used in sediment bioassay tests as a measure of pollution was also among the ten most abundant species. Oligochaete worms, including Tubificidae undetermined, *Tubificoides wasselli*, *T. brownae*, Tubificidae sp. b, and Enchytraeidae were a minor component of the benthic community, contributing approximately 4% to the total abundance. The small clam, *Leptonacea* sp., was the most abundant mollusc (0.8%) and sea anemones were the most abundant organisms in the "other taxa" category. Many of the taxa that were dominant in the May River were also among the most abundant fauna at the 2002 SCECAP sites (Table III-12).

When the dominant benthic infauna inhabiting open water and large tidal creeks stations were evaluated separately, *Streblospio benedicti* remained the most abundant organism at both station types (15% and 22%, respectively). Other dominant taxa at the open water stations followed the overall pattern of abundance described above. The dominant benthic species in the large tidal creeks was characterized by a similar group of polychaetes, with the addition of *Caulleriella* sp. (8%) and *Paraonis fulgens* (5%), and the oligochaete *Tubificoides wasselli* (5%).

Perhaps the best indication of overall benthic community condition is provided by the Benthic Index of Biotic Integrity (B-IBI), which has been developed for the southeastern region to distinguish between degraded and undegraded environments (Van Dolah and others, 1999). This index incorporates a number of benthic metrics (faunal abundance, number of species, relative abundance of pollution sensitive taxa; and a measure of species dominance) and has been demonstrated to show a good correspondence with sediment quality (i.e., contaminant levels). All sites in the May River had B-IBI values > 2.5 which is indicative of undegraded benthic communities (57 III-58). The lowest B-IBI value was found at station U-02, which may reflect some effects of sediment toxicity from some unmeasured contaminant as noted previously. Overall, the mean B-IBI values observed at the open water and tidal creek stations were very similar to the B-IBI means measured at the 2002 SCECAP sites (Figure III-57).

Based on our overall comparisons of faunal abundances, diversity (i.e., number of species and H') species composition, and the B-IBI measures, the May River benthic communities were very consistent with the sites sampled in other areas of southern South Carolina that are considered to be in relatively pristine locations. Minor differences observed between the May River and 2002 SCECAP sites were most likely due to natural differences observed among locations and possible influences of environmental variables, such as salinity. Since the benthic communities are considered to be one of the better indicators of habitat condition, our data indicate that the larger tidal creeks and open water areas of the May River are not showing evidence of degradation due to the existing development level on the river.

Nektonic Community

Approximately 1,100 organisms representing 34 taxa were collected by trawl in the May River open water and large tidal creek stations (Access® database). Mean abundance ranged from 80 to 1,435 individuals/hectare (Figure III-58, Table III-13). There were significantly more organisms collected by trawls at large tidal creek stations (1,005 individuals/hectare) than at the open water stations (358 individuals/hectare) (p-value = 0.005, Figure III-59). Mean abundance values at the large tidal creek station L-04 were the highest observed and significantly greater than values at the open water station U-02, which had the lowest mean abundance of nektonic organisms (p-value = 0.019). A comparison of trawl catches from May River stations to catches collected in the 2002 SCECAP sites did not indicate any significant differences with respect to mean nektonic abundance in either open water areas or large tidal creeks (p-value > 0.05, Figure III-58, Table III-13).

The mean number of taxa collected in trawls ranged from 2 to 13.5 taxa at the May River stations, and was not significantly different between open water and large tidal creeks sites (p-value > 0.05, Figure III-59, Table III-13). It should be noted that differences in trawl tow lengths between open water stations (0.5 km) and large tidal creek stations (0.25 km) cannot be normalized because species numbers are not expected to increase linearly with area swept. However, based on these data, it is very likely that the tidal creeks had a higher number of species per unit area than the open water sites since comparable numbers of species were found in half the distance towed compared to the open water sites. The most species were collected at station L-02, which had significantly more nektonic taxa than station U-02 (p-value = 0.021). No significant differences in the mean number of species within each habitat type were found between May River stations and those collected in 2002 SCECAP comparable habitats (p-value > 0.05, Figure III-59, Table III-13).

Diversity measures, including the Shannon-Weaver Index (H'), evenness (J'), and Margalef's Species Richness were calculated for all stations to examine overall community characteristics. Ranges of these measures were similar for May River sites and 2002 SCECAP sites (Tables III-13).

White shrimp (*Penaeus setiferus*) were the dominant organisms collected at both large tidal creek and open water sites (Table III-13). In large tidal creeks, white shrimp made up 74% of all animals collected, whereas they only comprised 34% of the total abundance in open water areas. Other dominant taxa included squid (*Lolliguncula brevis*), spot (*Leiostomus xanthurus*), pigfish (*Orthopristis chrysoptera*), weakfish (*Cynoscion regalis*), and silver perch (*Bairdiella chrysoura*). Although lesser blue crabs (*Callinectes similis*) were one of the ten most abundant overall organisms collected, contributing to 2% of the total abundance, blue crabs (*Callinectes sapidus*) were noticeably absent from trawl catches. The same pattern in lesser blue crab and blue crab abundances was observed in the 2002 SCECAP sites (Table III-13). The extended drought conditions from 1999-2002 probably resulted in the movement of blue crabs further upstream from the areas sampled in these studies, resulting in the effective elimination of blue crabs from trawl catches.

Differences in species composition were observed between the May River sites and the 2002 SCECAP sites. In large tidal creeks, white shrimp were the dominant organism collected at both May River and 2002 SCECAP sites, but this species made up a much larger percentage of the overall abundance at May River stations (74%) than 2002 SCECAP stations (45%). Two species, pinfish (*Lagodon rhomboides*) and brown shrimp (*Penaeus aztecus*), which were among the ten most abundant taxa collected at 2002 SCECAP sites, were not collected at any May River large tidal creek stations. Other taxa, such as the recreationally important Atlantic Croaker (*Micropogonius undulatus*), contributed less to the overall abundance at the May River stations (0.2%) than at the 2002 SCECAP stations (3%). Differences were also observed with respect to community composition at open water sites sampled in the May River and at 2002 SCECAP sites. Atlantic Croaker, the most abundant organism at pristine open water sites (30% of total abundance), was found in very low abundances at May River stations, contributing only 0.3% of the total abundance. Star drum (*Stellifer lanceolatus*), composing 22% of the total abundance at pristine open water stations, was not found at May River open water stations. Some species, such as spot and silver perch, contributed to more of the overall abundance at May River stations than at 2002 SCECAP sites (14% and 3% at May River stations vs. 3% and 1% at 2002 SCECAP stations).

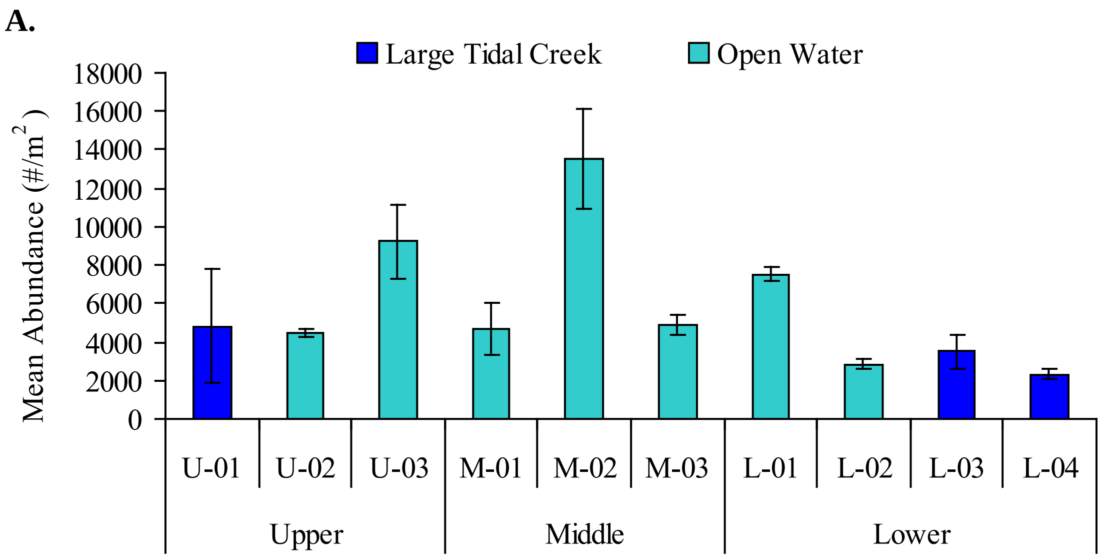
The differences between species composition at May River sites and 2002 SCECAP sites may reflect differences in habitat quality, but it is also likely to be related to other natural differences in the habitats of the areas sampled. Finfish encounter complex natural variations in physical, chemical, and biological factors which strongly influence the accessibility and variety of estuarine habitats, consequently affecting the distribution, diversity, and abundance of these species (Monaco and others, 1992). Differences in community structure may also represent seasonal patterns in the abundance of nektonic organisms. The three May River large tidal creek stations were sampled later in the summer (July 24th, August 7th) than the 2002 SCECAP sites (June 19th and 26th, and July 10th). A trend of higher brown shrimp catch with respect to sampling date was observed at 2002 SCECAP sites; this trend was also observed statewide in all tidal creek stations sampled during that year (n =34). It is also possible that the trawl catch for the May River and 2002 SCECAP sites reported here is not fully representative of the nektonic communities actually present due to the relatively small number of stations sampled, particularly for large tidal creek stations (May River, n = 3; 2002 SCECAP, n = 3).

Table III-11. The 25 numerically dominant taxa collected from May River large tidal creek and open water sites. Abundance values are the mean number per grab (0.04m^2).

Species Name		Total Abundance	U-01	U-02	U-03	M-01	M-02	M-03	L-01	L-02	L-03	L-04
Streblospio benedicti	Poly	379	42	92	23	12	66	44	45	3	53	0
Mediomastus sp.	Poly	255	0	1	41	13	135	14	39	5	5	2
Spiochaetopterus costarum oculatus	Poly	178	1	0	8	6	68	23	22	11	7	32
Scoletoma tenuis	Poly	100	2	0	16	11	23	8	9	5	9	17
Exogone sp.	Poly	99	0	0	37	15	26	3	12	0	5	2
Ampelisca abdita	Amp	95	0	0	63	10	4	3	11	0	3	0
Aricidea bryani	Poly	78	2	11	1	2	29	4	5	6	16	1
Polydora cornuta	Poly	75	1	0	22	40	4	1	6	0	1	0
Scoloplos rubra	Poly	71	4	3	15	6	12	18	0	0	10	2
Mediomastus ambiseta	Poly	71	0	1	17	17	17	0	15	4	0	0
Tharyx acutus	Poly	50	0	8	1	2	13	19	6	0	0	0
Ampelisca verrilli	Amp	43	0	1	6	4	17	11	0	1	3	0
Caulleriella sp.	Poly	40	30	8	0	0	0	0	0	0	0	2
Tubificidae	Oligo	40	2	0	2	2	20	5	6	1	2	0
Cirrophorus sp.	Poly	39	4	20	6	4	0	3	1	0	0	0
Dulichiella appendiculata	Amp	37	0	0	14	0	0	0	22	0	0	0
Tubificoides wasselli	Oligo	30	22	0	0	0	5	0	3	0	0	0
Paraonis fulgens	Poly	26	23	4	0	0	0	0	0	0	0	0
Tubificoides brownae	Oligo	25	2	1	1	1	12	2	4	0	1	2
Cirratulidae	Poly	25	1	1	2	0	7	10	2	0	0	1
Batea catharinensis	Amp	24	0	0	0	0	2	0	22	0	0	0
Scolelepis texana	Poly	21	0	14	3	0	3	1	0	0	0	0
Diopatra cuprea	Poly	19	1	0	7	2	3	1	3	0	1	2
Leptonacea sp.	Moll	18	0	0	0	0	1	0	1	14	1	0
Acteocina canaliculata	Moll	16	0	0	4	2	0	0	0	6	2	2
Percent of total abundance			71	92	78	79	86	88	78	50	85	73
Mean total abundance (#/0.04m^2)			193	181	369	187	541	194	301	114	140	92
Mean density per station (#/m^2)			4833	4517	9217	4683	13533	4858	7525	2842	3492	2308
Mean number of species (#/0.04m^2)			15	19	39	28	42	23	42	27	22	19
Mean H' - Diversity			3.11	2.50	4.20	3.75	4.01	3.59	4.31	4.08	3.08	2.95
Mean J' - Evenness			0.79	0.59	0.80	0.80	0.75	0.79	0.80	0.86	0.70	0.69
Mean Species Richness			2.98	3.41	6.61	5.15	6.78	4.31	7.32	5.65	4.27	3.99

Table III-12. The 25 numerically dominant taxa collected from SCECAP large tidal creek and open water sites. Abundance values are the mean number per grab ($0.04m^2$).

Species Name		Total Mean Abundance	RO026001	RO026005	RO026007	RO026019	RO026025	RO026027	RO026151	RT022009	RT022015	RT022019
Streblospio benedicti	Poly	749	0	229	6	0	58	313	2	21	101	21
Aphelochaeta sp.	Poly	270	0	0	54	0	0	0	0	0	0	216
Exogone sp.	Poly	203	0	8	83	0	13	3	0	1	1	96
Ampelisca abdita	Amp	154	0	5	137	0	1	2	0	1	2	8
Tubificidae	Oligo	143	0	0	76	0	1	49	0	0	2	16
Clymenella torquata	Poly	125	0	115	0	0	3	8	0	0	0	1
Scoletoma tenuis	Poly	113	0	5	34	0	3	3	0	10	18	42
Polycirrus sp.	Poly	96	0	1	2	0	0	0	0	0	0	94
Enchytraeidae	Oligo	95	2	0	94	0	0	0	0	0	0	0
Aricidea wassi	Poly	85	1	2	0	0	25	40	18	0	0	0
Unciola serrata	Amp	84	0	13	69	0	1	1	0	0	1	0
Batea catharinensis	Amp	79	0	2	22	0	48	7	0	0	1	0
Scoloplos rubra	Poly	77	0	0	19	0	0	1	1	1	14	43
Mediomastus sp.	Poly	71	0	8	10	0	14	11	0	0	3	27
Tubificoides brownae	Oligo	66	0	5	18	0	4	8	0	25	7	0
Tharyx acutus	Poly	48	0	4	12	0	9	4	1	0	4	15
Cirratulidae	Poly	47	1	2	22	0	0	1	1	0	1	21
Sabellaria vulgaris	Poly	40	0	18	13	0	2	0	0	0	0	7
Cyathura burbancki	Other	38	0	0	37	0	1	0	0	0	0	0
Listriella clymenellae	Amp	31	0	18	0	0	3	10	0	0	1	0
Scolelepis texana	Poly	27	1	0	0	0	14	1	3	0	9	0
Streptosyllis sp.	Poly	26	1	0	7	14	0	0	4	0	1	1
Protohaustorius deichmannae	Amp	25	2	0	0	0	6	0	18	0	0	0
Tellina texana	Moll	25	0	0	0	0	12	13	0	0	0	0
Turbonilla sp.	Moll	23	0	15	0	0	1	8	0	0	0	0
Percent of total abundance			26	80	81	31	75	82	72	85	86	78
Mean total abundance (#/$0.04m^2$)			22	552	882	43	287	584	63	66	189	782
Mean density per station (#/m^2)			538	13800	22038	1075	7175	14600	1563	1650	4725	19550
Mean number of species (#/$0.04m^2$)			10	45	44	6	44	38	13	11	19	53
Mean H' - Diversity			2.78	3.27	3.95	2.19	3.97	2.79	2.63	2.50	2.56	3.68
Mean J' - Evenness			0.89	0.60	0.72	0.86	0.73	0.53	0.71	0.74	0.62	0.64
Mean Species Richness			2.72	6.99	6.31	1.33	7.58	5.76	2.91	2.56	3.34	7.82



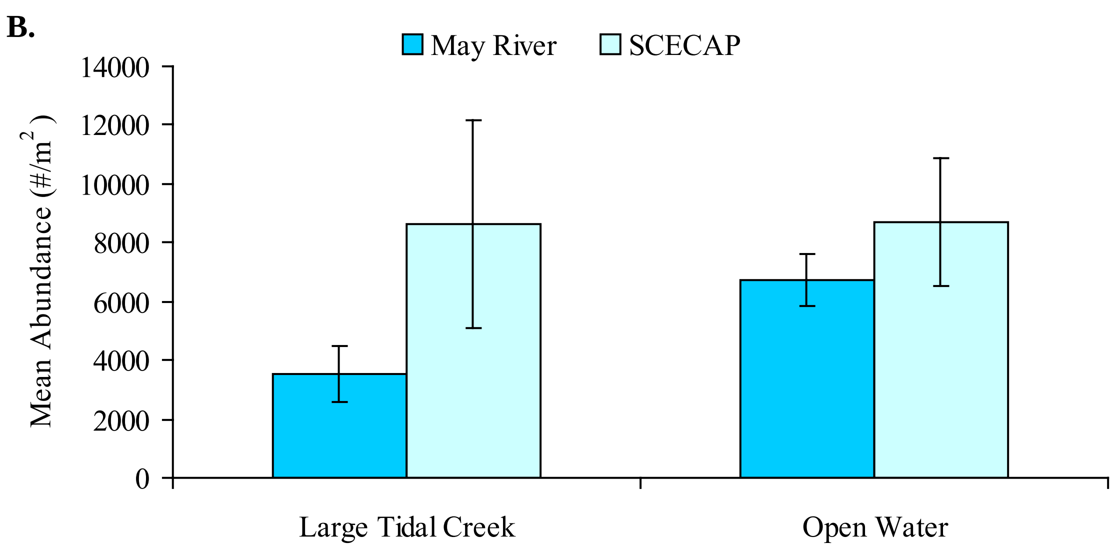


Figure III-54. Mean abundance of benthic fauna at large tidal creek and open water sites sampled in the May River (A.) and compared to SCECAP stations by habitat type (B.). Error bars represent 1 standard error.

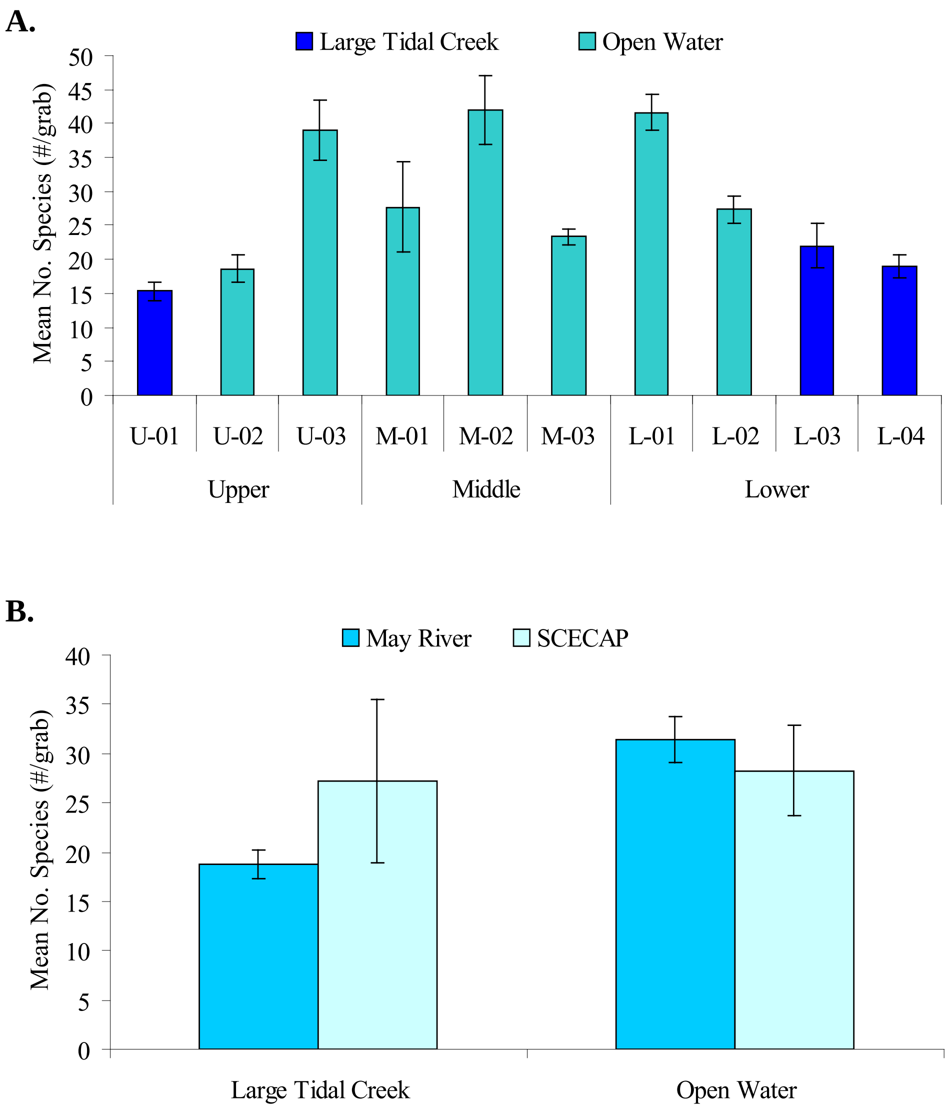
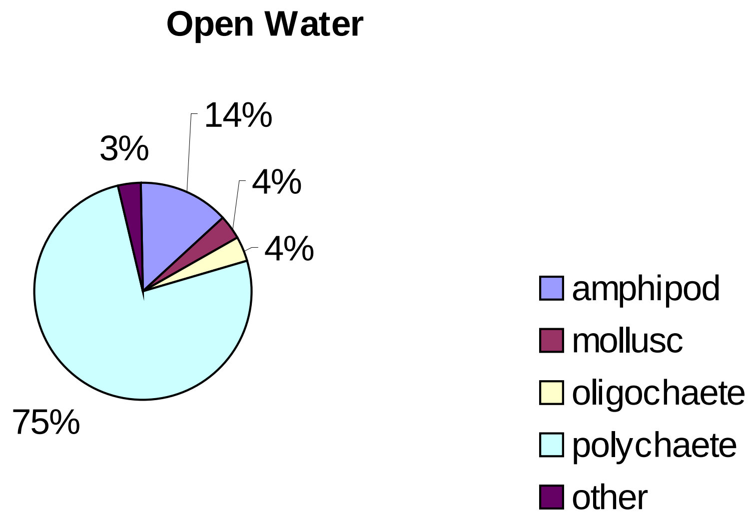


Figure III-55. Mean number of benthic fauna at large tidal creek and open water sites sampled in the May River (A.) and compared to SCECAP stations by habitat type (B.). Error bars represent 1 standard error.

A.



B.

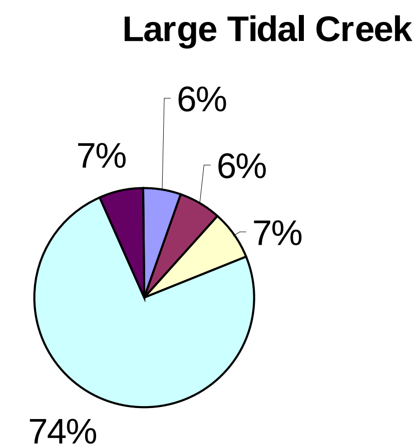
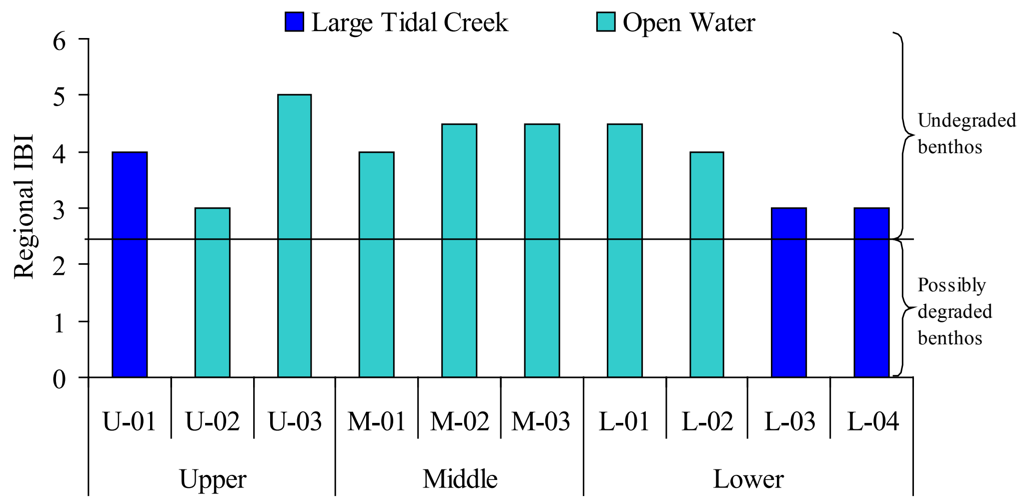


Figure III-56. Composition of benthic fauna collected from the large tidal creek and open water sites sampled in the May River.

A.



B.

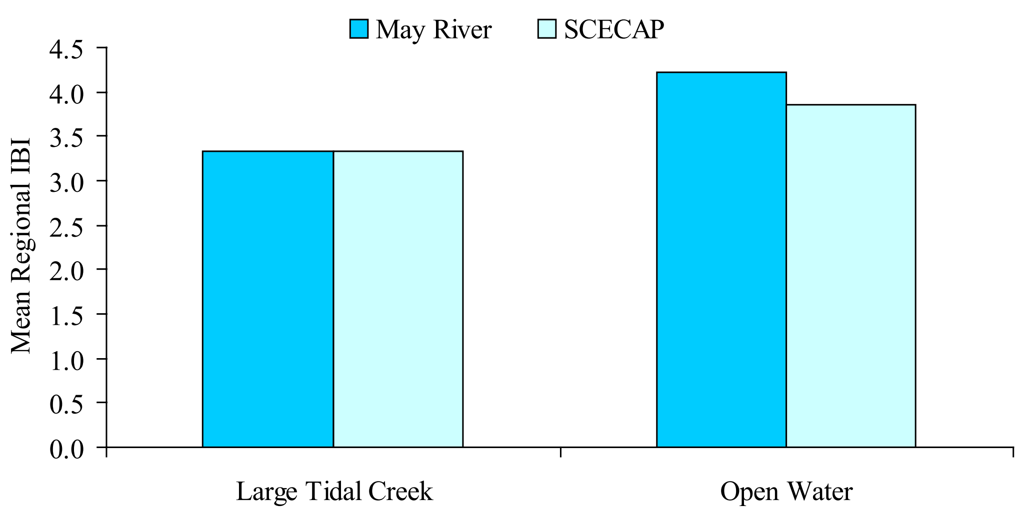


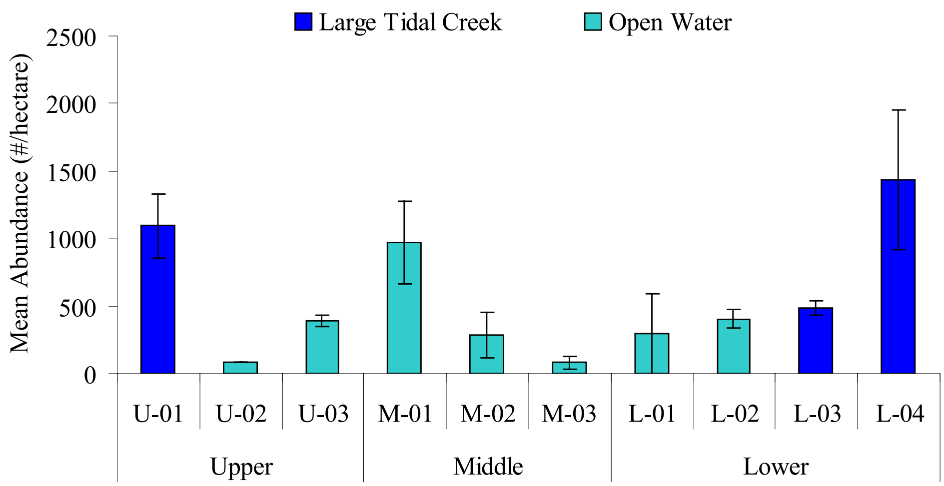
Figure III-57. Benthic Index of Benthic Integrity (B-IBI) measured at the large tidal creek and open water sites sampled in the May River (A.) and compared to SCECAP stations by habitat type (B.).

Table III-13. Abundance of the dominant nektonic fauna collected by trawl from large tidal creek and open water sites in the May River (upper table) and the 10 SCECAP sites sampled during the same period.

Species Name	% of Total Abun	Total Abun	Total Abun Large Tidal Creek	Total Abun Open Water	U-01	U-02	U-03	M-01	M-02	M-03	L-1	L-02	L-03	L-04
White shrimp (*Penaeus setiferus*)	56	3094	2239	855	906	0	98	493	0	0	101	163	123	1210
Squid (*Lolliguncula brevis*)	14	779	319	460	123	58	98	14	210	58	0	22	138	58
Spot (*Leiostomus xanthurus*)	10	533	174	359	0	14	65	91	40	7	134	7	94	80
Pigfish (*Orthopristis chrysoptera*)	4	199	14	185	0	0	4	167	4	0	0	11	7	7
Weakfish (*Cynoscion regalis*)	3	188	14	174	7	0	36	40	0	0	58	40	7	0
Silver Perch (*Bairdiella chrysoura*)	3	185	101	83	0	0	33	14	0	0	0	36	43	58
Lesser Blue Crab (*Callinectes similis*)	2	101	22	80	22	0	0	76	0	0	0	4	0	0
Anchovy (*Anchoa mitchilli*)	1	51	14	36	0	0	4	4	22	7	0	0	0	14
Lookdown (*Selene vomer*)	1	51	7	43	0	4	22	0	0	0	0	18	7	0
Brown shrimp (*Penaeus aztecus*)	1	43	0	43	0	0	22	14	0	7	0	0	0	0
Percent of total abundance					97	91	98	94	97	100	99	74	87	99
Mean density per station (#/m^2)					1094	83	388	971	283	80	297	406	485	1435
Mean number of species					6	3	7	10	4	4	2	14	8	5
Mean H' - Diversity					0.94	0.95	2.19	1.75	1.33	1.46	0.79	2.87	2.08	0.83
Mean J' - Evenness					0.39	0.68	0.79	0.53	0.67	0.73	0.40	0.77	0.71	0.38
Mean Species Richness					0.64	0.34	1.01	1.21	0.56	0.75	0.23	2.09	1.12	0.48

Species Name	% of Total Abun	Total Abun	Total Abun Open Water	Total Abun Large Tidal Creek	RO026001	RO026005	RO026007	RO026019	RO026025	RO026027	RO026151	RT022009	RT022015	RT022019
White shrimp (*Penaeus setiferus*)	26	2315	178	2138	0	0	0	149	0	0	29	2138	0	0
Brown shrimp (*Penaeus aztecus*)	21	1899	906	993	54	112	0	4	225	446	65	58	812	123
undulatus)	16	1411	1280	130	203	141	9	7	47	837	36	0	101	29
Star Drum (*Stellifer lanceolatus*)	10	913	913	0	250	105	0	138	25	395	0	0	0	0
Spot (*Leiostomus xanthurus*)	5	413	130	283	0	0	0	0	7	120	4	58	196	29
Weakfish (*Cynoscion regalis*)	4	377	290	87	65	22	0	0	87	116	0	0	87	0
Squid (*Lolliguncula brevis*)	3	298	52	246	4	0	9	0	33	4	4	43	174	29
Hogchoker (*Trinectes maculatus*)	2	149	141	7	43	0	0	0	0	98	0	7	0	0
Lesser Blue Crab (*Callinectes similis*)	2	141	141	0	22	0	0	11	7	101	0	0	0	0
Silver Perch (*Bairdiella chrysoura*)	1	112	40	72	0	0	0	0	14	22	4	65	7	0
Percent of total abundance					99	98	22	96	95	99	89	98	98	64
Mean density per station (#/m^2)					649	388	78	322	471	2156	159	2420	1399	326
Mean number of species					8	5	6	6	11	11	7	9	7	8
Mean H' - Diversity					2.16	1.47	2.39	1.49	2.40	2.33	2.14	0.81	1.86	2.43
Mean J' - Evenness					0.75	0.67	0.96	0.64	0.71	0.69	0.79	0.27	0.66	0.83
Mean Species Richness					1.01	0.59	1.17	0.85	1.63	1.24	1.11	0.96	0.83	1.12

A.



B.

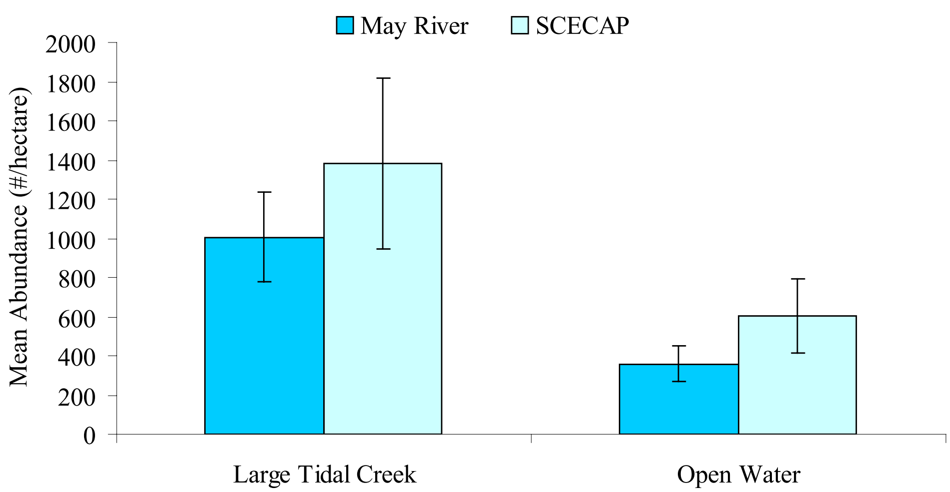
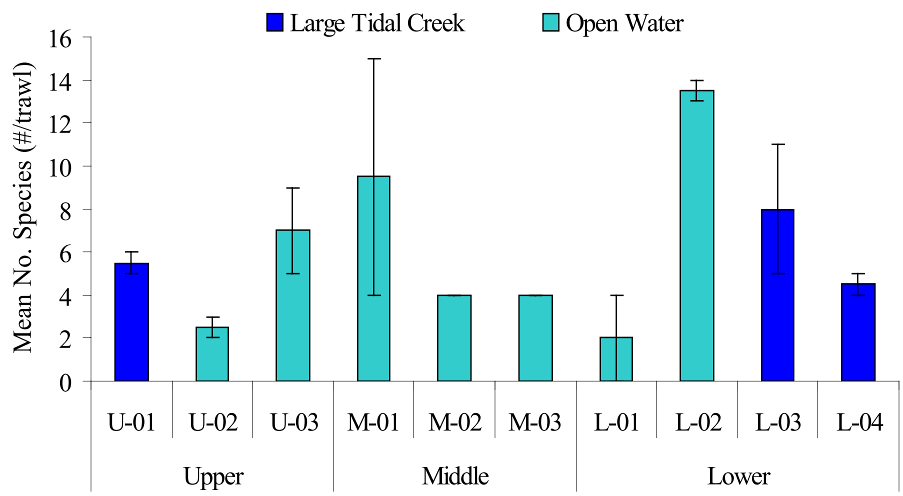


Figure III-58. Mean abundance of nektonic fauna at large tidal creek and open water sites sampled in the May River (A.) and compared to SCECAP stations by habitat type (B.). Error bars represent 1 standard error.

A.



B.

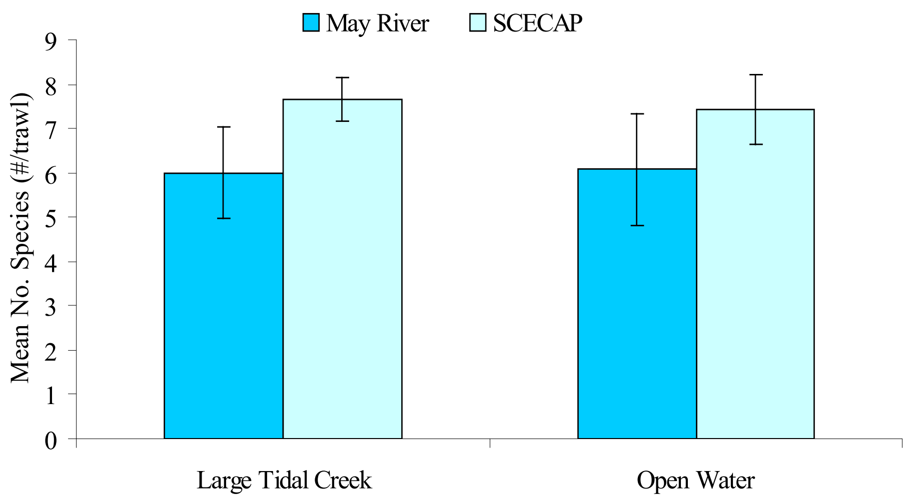


Figure III-59. Mean number of nektonic species collected at the May River stations (A.) and compared to SCECAP stations by habitat type (B.). Error bars represent 1 standard error.

IV. OYSTER BEDS

A. Introduction

Oysters and the habitat they generate are of special concern for South Carolina waters, particularly in the May River, due to their demonstrated value: (1) as a recreational and commercially important species, (2) for their ecological role as a nursery or natural habitat, and (3) the ability of oysters to improve water quality through their filtering capabilities, and protection of marsh lined tidal creeks. The eastern oyster, *Crassostrea virginica* forms living reefs that support a host of associated organisms generally not found in the surrounding sand or mud habitats (Coen and others, 1999a, b; Coen and Luckenbach, 2000). This is especially true in South Carolina where seagrass beds (or submerged aquatic vegetation-SAV) are absent. Many bivalve species have been shown also to be useful indicators of water quality (e.g., Capuzzo, 1996; Anonymous, 1998) as they accumulate and concentrate metals, biotoxins and organic pollutants due to their mode of filter feeding.

The range of the eastern oyster, *Crassostrea virginica,* extends from the Gulf of St. Lawrence in Canada to Brazil and Argentina (Carriker and Gaffney, 1996) and populations have declined tremendously in many east coast states during the past century to less than 1% of their historical highs. This decline has been attributed to over harvesting, dredging, declines in water quality, habitat degradation and disease (Rothschild and others, 1994; Kennedy and others, 1996; Luckenbach and others, 1999). Oyster populations are often assessed due to their proven role as "ecosystem engineers" or keystone habitat species (Coen and others, 1999b), as well as their value as an economically important resource and their demonstrated significance as indicators of estuarine condition (Ringwood and others, 1999).

The Oyster Reef Program (ORP) is a SCDNR study which assessed the health of oyster reefs in South Carolina. ORP has been examining size-frequency relationships, recruitment potential, and disease (MSX and Dermo) levels of native oyster populations as indicators of habitat health, along with estimates of population size of select epibenthic invertebrate species (Coen and others, 1999a; Coen and Luckenbach, 2000; Luckenbach and others, In press) for the past 8 years. These measures appear to provide an excellent indication of habitat health and oyster status and recruitment potential (Van Dolah and others, 2000).

The SCDNR Biomarker Program has evaluated oyster health using three biomarkers: (1) lysosomal destabilization, (2) glutathione concentrations, and (3) lipid peroxidation. The underlying premise of biomarker tools is that the effects at higher levels of organization (populations and communities) represent the net sum of the effects on individuals that resulted from alterations in cellular and molecular responses. Therefore, cellular responses should function as indicators for identifying individuals and populations that are experiencing chronic stress, which if unmitigated, may progress to severe effects at higher levels of organization.

B. Methods

Oyster reefs at 11 sites within the May River were sampled for oyster size and abundance, oyster disease (MSX and Dermo), oyster health (lysosomal integrity, lipid peroxidation, and glutathione concentrations), and oyster tissue contamination in the summer of 2002 to assess overall oyster population condition. The recruitment and post-settlement growth of oysters to each of the sites was also determined by deploying South Carolina oyster shell at each site for an approximately seven-month period. Trays were deployed in April 2002 and retrieved in March 2003.

Oyster reefs near large tidal creeks or open water sites were sampled whenever possible (Figure IV-1). This occurred for nine sites, with an additional two sampling sites chosen to ensure the entire May River was spatially sampled. In the Upper Zone, three sites were sampled, all of which occurred within private Culture Permit areas. In the Middle Zone, five sites were sampled, of which four sites occurred in private Culture Permit areas; the other site (M-05) occurred within State (SSG S-007) and Public Shellfish Grounds (PSGs). The recruitment portion at the M-05 site occurred in the PSG portion (R-008). In the Lower Zone, all three sites occurred in private Culture Permit areas.

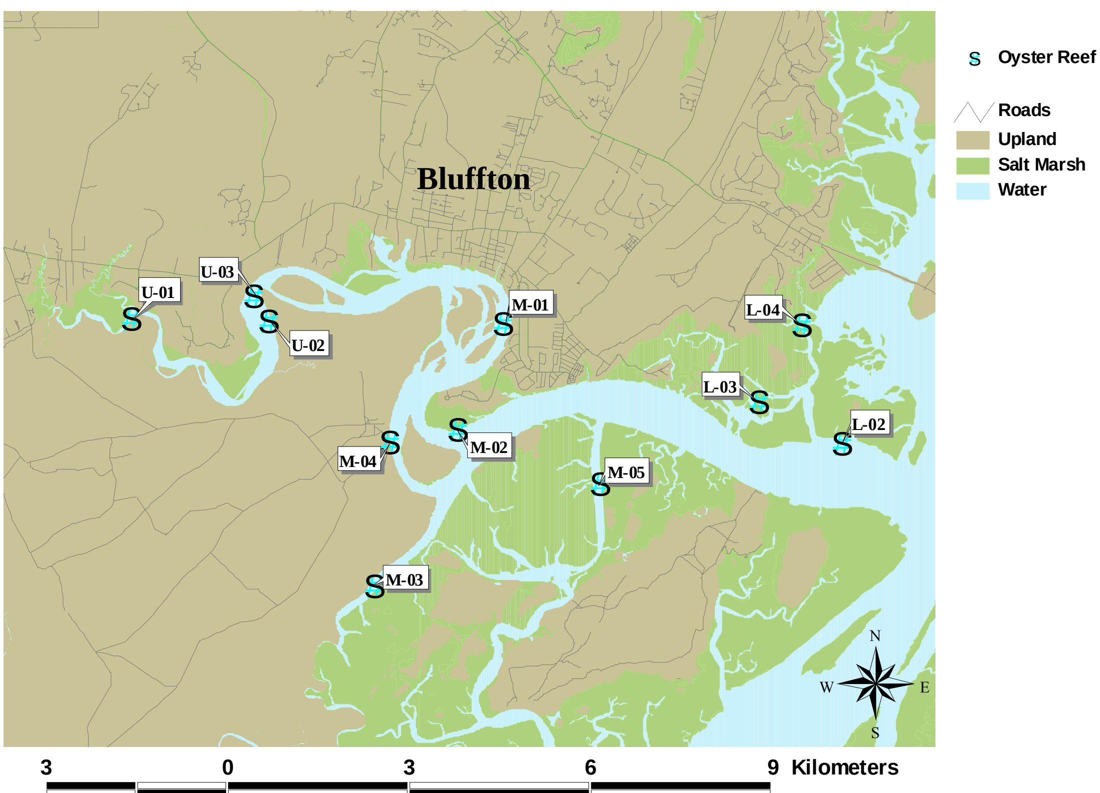


Figure IV-1. Map of the oyster reef locations sampled in the May River.

Oyster Community Assessment

Size and Reef Abundance

At each reef site, a 20-meter transect line was placed along the shoreline at approximately mean sea level or approximately 4.2 m below mean high water (MHW), which is the level with densest oyster populations. Five replicate samples were collected using a 0.143 m^2 quadrant at pre-selected random locations along the transect line. Digital photos were taken prior to sampling for strata identifications. If no oysters were located within a pre-assigned random location, another random position was chosen to ensure that five representative oyster samples were collected from each site. All oysters and sediment were removed to a depth of ~11 cm (standard for prior sampling across the state) and then placed in a labeled bucket. The specific quadrant placement for each sample was adjusted to maximize the percentage of oyster shell (live and dead) cover within the quadrant.

All samples were washed in the laboratory to remove sediment and associated large animals (primarily crabs), and stored in a refrigerator prior to workup. Using a digital caliper system, all live oysters in a sample were measured [shell height (SH) = the distance from the umbo to the outermost edge] to the nearest millimeter, including all oysters removed for disease assessment. All data were directly recorded as measured in Excel® via a program called Software WinWedge® (V 1.2). Resident species abundances (crabs and mussels) were noted using a previously employed qualitative ranking system (see results for explanation of ranking criteria).

Diseases

Two protozoan pathogens of oysters, *Perkinsus marinus* (Dermo) and *Haplosporidium nelsoni* (MSX), are prevalent in East Coast oysters in the United States. Historically, *P. marinus* has been present in oyster samples collected by the SCDNR since 1972, with observed seasonal patterns of the disease being similar to patterns reported for Gulf Coast oyster populations (Bobo and others, 1997). Prevalence (% of oysters in a sample with the disease observed) and intensity levels (log scale of infection from 0-6) are generally greatest in the summer and fall, although Dermo is present year round. *Haplosporidium nelsoni* infections were first observed in South Carolina oysters in 1986 (Bobo and others, 1997). Since then, the parasite has been documented in nearly 50% of the SC sites examined to date.

Oysters (total of 275 individuals) were collected from the 11 May River sites in September 2002 to determine prevalence and infection levels for both Dermo and MSX. At each of the sites, five adult oysters were collected and examined from each of the five random sampling areas utilized in the above population sampling sites (total of 25 oysters per site). *H. nelsoni* infections were rated according to standard categories (Appendix-IV-1a). See Bobo and others (1997) for additional information on methods for examining Dermo and MSX infections.

Health

A subset of oysters, approximately 20 different individuals from each site, were evaluated to assess subcellular level effects using three different assays (lysosomal integrity, lipid peroxidation, and glutathione concentrations). A neutral red (NR) assay was used to evaluate lysosomal integrity in the digestive gland cells of oysters (Lowe and others, 1992; Ringwood and others, 1998a). Briefly, samples were prepared from pieces of digestive gland tissue dissected from individual oysters and disaggregated in a calcium/magnesium-free saline solution. An aliquot of this cell solution was mixed with NR and allowed to incubate in a humidity chamber for 60 minutes. Digestive gland cells were then evaluated under a light microscope; those with NR retained in lysosomes were scored as stable and those with NR leaking into the cytoplasm were scored as destabilized. The leaking of NR reflects the efflux of lysosomal contents into the cell, which can ultimately cause cell death. A minimum of 50 cells were counted for each preparation, and the data were expressed as percent destabilized lysosomes per individual oyster.

In the glutathione (GSH) assay, GSH concentrations of individual oyster digestive gland tissues were determined using the DTNB-GSSG Reductase Recycling Assay (Anderson, 1985). Glutathione is a ubiquitous tripeptide that functions as an overall modulator of cellular homeostasis, and serves essential functions including detoxification of metals and oxy-radicals. This assay follows the rate of 5-thio-nitrobenzoic acid (TBA) formation. Briefly, individual digestive gland tissues were dissected out, homogenized in buffer, and centrifuged. GSSG reductase was added, and the rate of TBA formation was monitored using a spectrophotometer at 15-sec intervals for 90 seconds. Concentrations of GSH were estimated from a standard curve and reported as nM GSH/gram wet weight.

Finally, the lipid peroxidation (LPx) assay is an indicator of damage to cell membranes (Kehrer, 1993). The thiobarbituric acid (TBA) test was used to measure lipid peroxidation (Gutteridge and Halliwell, 1990). Digestive gland tissues were homogenized and centrifuged. A subsample of the supernatant was mixed with tricholoroacetic acid containing TBA and butylated hydroxytoluene, heated for 15 min and centrifuged to remove the precipitate. The resulting malondialdehyde (MDA) was detected using a spectrophotometer. Standards were prepared as described by Csallany and others (1984), and the data were expressed as nM MDA / gram wet weight.

Tissue Contamination

A subset of oysters, approximately 10 individuals from each of the 11 sites, was used for tissue contaminant analyses. Tissues were dissected from the shell and frozen at –80°C until processed. The NOAA Laboratory analyzed oyster tissues for four main classes of contaminants; trace metals, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and pesticides, and followed the methods discussed previously in Section III.

Tissue concentrations for each analyte were compared to tissue guidelines determined by the United States Environmental Protection Agency (1995) and the Food and Drug Administration (FDA) screening values (in NOAA 1998). EPA tissue guidelines were for carcinogens and non-carcinogens and were based on the consumption of 6.5 g/day of fish for a 70 kg adult. EPA tissue guidelines based on wet weight were converted to dry weight guidelines to reflect the contaminant concentration based upon an 80% organismal water content (Galtoff 1964). FDA tissue guidelines were based on the lowest FDA value for contaminants in shellfish based on the 90th percentile consumption rate and several human age groups (in NOAA 1998). Recommended tissue concentration guidelines were not available for many of the analytes in this study. Total DDT was defined as the sum of the 2,4' and 4,4' DDD, DDE, and DDT. Reported total PCBs and PAHs were based on the same compounds used to develop the tissue guidelines. Total PCBs were defined as the sum of 17 measured congeners (8, 18, 28, 44, 52, 66, 77, 101, 105, 118, 126, 128, 138, 153, 170, and 187). Total PAHs were defined as the sum of 16 PAHs (naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo(a)anthracene, benzo(b)fluoranthene, benzo(a)pyrene, indeno(1,2,3-cd)pyrene, benzo(g,h,i)perylene, chrysene + triphenylene, dibenz(a,h+a,c) anthracene, and benzo(j+k)fluoranthene). Total pesticides were defined as the sum of pesticides (total DDT, aldrin, chlorpyrifos, cis-chlordane (alpha-chlordane), dieldrin, endosulfan I and II, gamma-HCH (g-BHC, lindane), heptachlor, heptachlor epoxide, hexachlorobenzene, mirex, trans-nonachlor).

In addition, lipid normalized tissue contaminant levels were divided by the octanol-water partitioning coefficient (K_{ow}) for each contaminant to extrapolate the mean estimated surface water concentration for each contaminant (Scott and others, 1997). Tissues were lipid normalized by dividing the contaminant concentration (per gram dry weight) by the percentage of lipids in the sample. The estimated water concentrations were then compared to the United States Environmental Protection Agency Water Quality Criteria to derive a risk assessment estimate of water quality for each site. Scott and others (1997) have used this approach to estimate environmental risks from pesticide runoff in South Carolina and Florida. This technique was not applied to metal concentrations because K_{ow}s were not available. Water quality criteria used were based upon the saltwater criterion continuous concentration established by the United States Environmental Protection Agency (USEPA 1999). The criterion continuous concentration is an estimate of the highest concentration of a compound in surface water that will elicit no adverse community effects in an indefinite exposure (USEPA 1999). The Endosulfan water quality criterion was based on the saltwater continuous criterion for both parent isomers (Endosulfan I and II). Due to the absence of a saltwater continuous criterion concentration for g-HCH (Lindane), the water quality criterion was based on the saltwater criterion maximum concentration, which is an estimate of the highest concentration of a compound to elicit no adverse effects in a brief exposure (USEPA 1999).

Oyster Recruitment

Recruitment potential is an integrated measure of habitat quality as it assesses initial settlement and long-term growth and post-settlement survival. Three replicate trays (0.434 m^2) each lined with ¼" mesh and contained a standardized quantity (11.5 gallons by volume) of South Carolina oyster shell ('cultch') were deployed at the 11 sites for a period of 323-336 days (April 2002 to March 2003). Each tray was covered with 1 ¼" mesh to ensure shell retention over the duration of development. After recovery and upon return to the laboratory, the shell height (SH) of all live oysters were measured as described above.

Oyster Reef Residents

Along with the oyster population and related recruitment data, we also collected information on densities of other important invertebrate species that reside in oyster beds such as: (1) several species of brachyuran (primarily xanthid) crabs, all predators on small oysters (Grant and McDonald 1979, Meyer, 1994, Grabowski 2002), (2) a recent invader to the South Carolina coast, the green porcelain crab, *Petrolisthes armatus* (an anomuran crab), and (3) several mussels whose occurrence as filter feeders can be as important as the related oysters (Stiven, and Gardner, 1992; Kreeger and Newell, 2000) in that they filter even smaller particles from the water column and reach densities over 1,500/m^2 (Coen and others, 1999a, b; Luckenbach and others, In press). Green Porcelain crabs are an invasive species that has been studied by SCDNR since the 1990s (Knott and others, 2000; Coen and others, in preparation). We have observed these crabs to reach densities of over 20,000/m^2 at some South Carolina sites (Coen and McAlister, 2000). In other areas such as the Gulf Coast of Florida, they are very abundant on oyster reefs (Glancy and others, 2003). Abundance class estimates were made for each replicate (quadrant) per site during both population (0.143 m^2) and recruitment (0.434 m^2) sampling. Abundance classes used here were: 0 (0 observed individuals), 1 (1-9 observed individuals), 2 (10-50 observed individuals), or 3 (> 50 observed individuals), which have also been used in previous studies across South Carolina.

Data Analysis and Review

Cumulative size frequency distributions (5 mm intervals) were computed using Excel® for each of the 55 samples and visualized in SPSS's Sigma Plot® (Version 6.0). The following analyses were performed for both population and recruitment assessments. Mean numbers and mean numbers/m^2 were calculated using PC-SAS® (Version 8.2). A third-degree polynomial was fit to each of the cumulative frequency distributions (n = 55) in SAS's statistical program JMP (Version 3.2.2). Regression line parameters (intercept, slope, and curvature) for each sample were then recorded from this analysis and plots of mean shell height and total number of live oysters were adjusted to a per m^2 basis in Excel®.

To evaluate differences among the three distinct May River zones and among sites within each of these three zones, a one-way analysis of variance (ANOVA) was performed using regression line parameters, mean size, and mean number of oysters as treatments. Analysis of the regression line parameters, slope, and intercept values identified the numbers and relative sizes (smaller versus larger) of the resident oyster population in one zone or site were also performed.

Prevalence (percent) data were arcsine-transformed to evaluate disease prevalence and intensity levels among sites within each zone and among the three zones. If the data were normal and had homogeneous variances a one-way ANOVA was used. If data were not normal or homogeneous, a Kruskal-Wallis non-parametric test was used. When significance was detected among the sites or among the zones, an appropriate Multiple Comparison Test (MCT) was then used (Dunn's Method for Kruskal-Wallis Test; Bonferonni for ANOVAs).

C. Results and Discussion

Oyster Community Assessment

The amount of harvesting and related culture husbandry within each zone and site certainly influenced conditions observed at each of the oyster beds sampled. Most of the sites were within culture permit areas; hence, many of these permit holders have differing husbandry techniques and the beds may have been either recently harvested or planted with shell by the lessee, thus complicating interpretation. One site (M-05) falls within both a State Shellfish Ground (SSG) and a Public Shellfish Ground (R-008) (PSG). It is important to note the DHEC's classification of these sites at the time they were sampled was "Approved" (i.e. open), where direct harvesting was permitted.

Size and Abundance

Oyster size-frequency (or abundance) distributions and related overall densities can be affected by a multitude of factors including, past planting/harvesting history, substrate available for settlement and recruitment, local larval supply, and post-settlement survival and growth. Environmental parameters such as food availability, predation, sedimentation, and associated water quality (i.e., flow, salinity, DO, and temperature) can also play a key role in the condition of oysters. Size frequency information has been collected throughout South Carolina by SCDNR staff for the past ten years to better evaluate oyster population trends. These data make it possible to compare the condition of oyster beds in the May River with sites throughout the state in 2002, as well as historically.

The mean size of oysters was significantly larger in the Upper Zone compared to the Middle and Lower zones (p-value < 0.0001). Less than 30% of the oysters at each site were >75 mm (or 3" SH) and one site had < 20% of the oysters in this size range. Mean size ranged from 31 to 56 mm SH at all sites (Figure IV-2). Previous population studies throughout South Carolina yielded a mean SH of 35 mm. The mean size of oysters in all three May River zones was above the state mean, as the overall mean among all 11 sites was approximately 39 mm. Cumulative frequency distributions detected a greater percentage of smaller, recently recruited oysters or "spat" (<15 mm SH) within the Middle and Lower zones (~15 & 22% respectively), as compared to the Upper Zone (12%) (Figure IV-3). Interestingly, the upper-most site, U-01, had the largest overall oysters of all of the 11 sites examined in the May River, with 29% of oysters greater than 75 mm SH.

The total abundance of oysters (five replicate samples summed per site) ranged from a low of 1,016 individuals/m^2 at site U-02 to 2,657 individuals/m^2 at site L-04. Average oyster densities varied in each of the three zones, with mean numbers ranging from 1,029 - 2,654 individuals/m^2 (Figure IV-4), but there were no significant differences (p-value = 0.2116, ANOVA) among the three zones. The overall mean abundance among all 11 sites was 1,746 individuals/m^2. There was no significant difference in mean oyster abundance between the five sites sampled within the Middle Zone. In the Upper Zone, site U-02 was significantly greater than the other two sites. In the Lower Zone, site L-04 had a significantly greater number of oysters than L-02, but not when compared to L-03. When compared to long-term population studies from across South Carolina (mean of 1,578/m^2 for 13 sites), 9 of the 11 of the May River sites were close to or well above this average.

A statistical analysis of slopes by ANOVA for the site size-frequency data showed significant differences among the three zones in the May River (p-value = 0.0068), with the Lower Zone having a significantly higher slope than the Upper Zone. This indicates that more oysters within the smaller size classes (young oysters) are found in the Lower Zone compared to the Upper Zone. The Middle Zone was not significantly different from either of the other two zones using this approach. Similar intercept analyses showed identical results (p-value = 0.0311), with the Lower Zone having more oysters within the smaller size classes.

Analysis of the slopes and intercepts for the within-zone site size-frequency data indicated no significant differences in either parameter between the Lower and Middle zones; however, analysis of the Upper Zone revealed a significantly higher slope (p-value = 0.0035) for site U-02 as compared to either site U-01 or site U-03, indicating that site U-02 had more oysters in the smaller size classes (new recruits, age 1 or less). There were no significant differences in the calculated intercepts between sites.

A.

B.

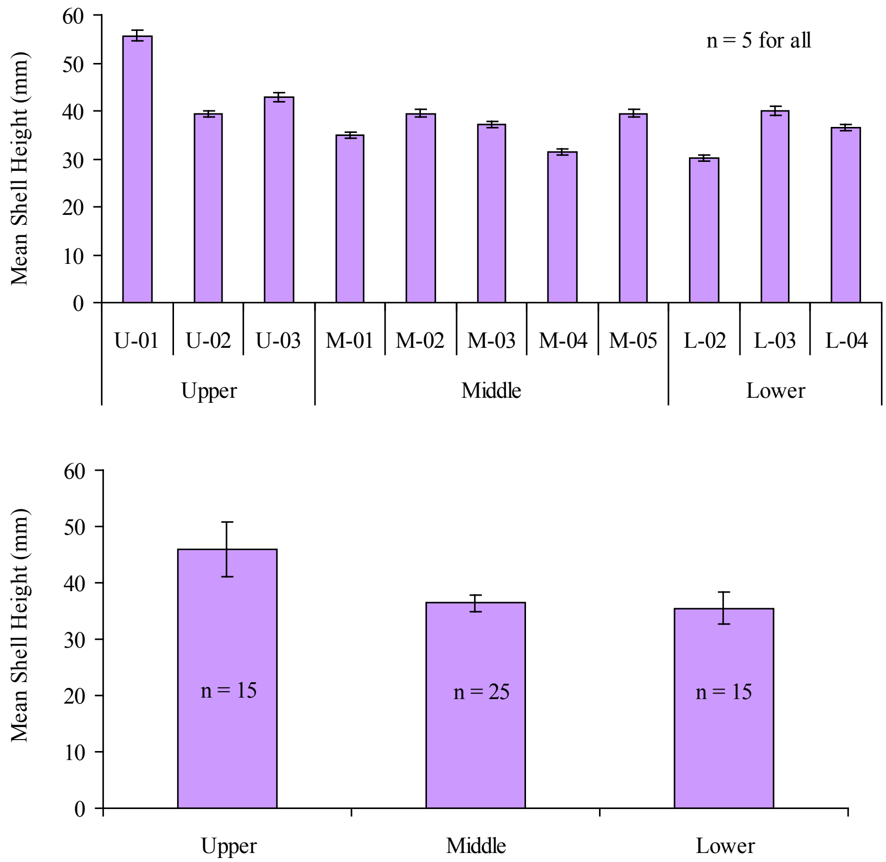


Figure IV-2. Mean shell heights by site (A.) and zone (B.) for 2002 oyster population assessment. Error bars represent 1 standard error.

Oyster Disease

Dermo (*Perkinsus marinus*) infections were detected at all of the eleven sites sampled. Mean prevalence levels (% of the sample infected) ranged from 52% at site M-01 to 88% at U-02, U-03 and M-02 (Figure IV-5; Appendix IV-1b). Mean prevalence levels greater than 80% were observed at four of the 11 (or 36%) stations sampled overall. There were no significant differences in prevalence levels among the sites within each zone or between the three zones (Kruskal-Wallis, p-value = 0.411, Figure IV-5). Mean infection intensity levels ranged from 0.92 (low infection levels for South Carolina) at M-01 to 2.32 (moderate infection levels) at M-02 and M-05. Three of the 11 sites (or 27%) had infection intensities > 2.00 which indicates a moderate level of infection (Figure IV-6). No significant differences were detected for infection

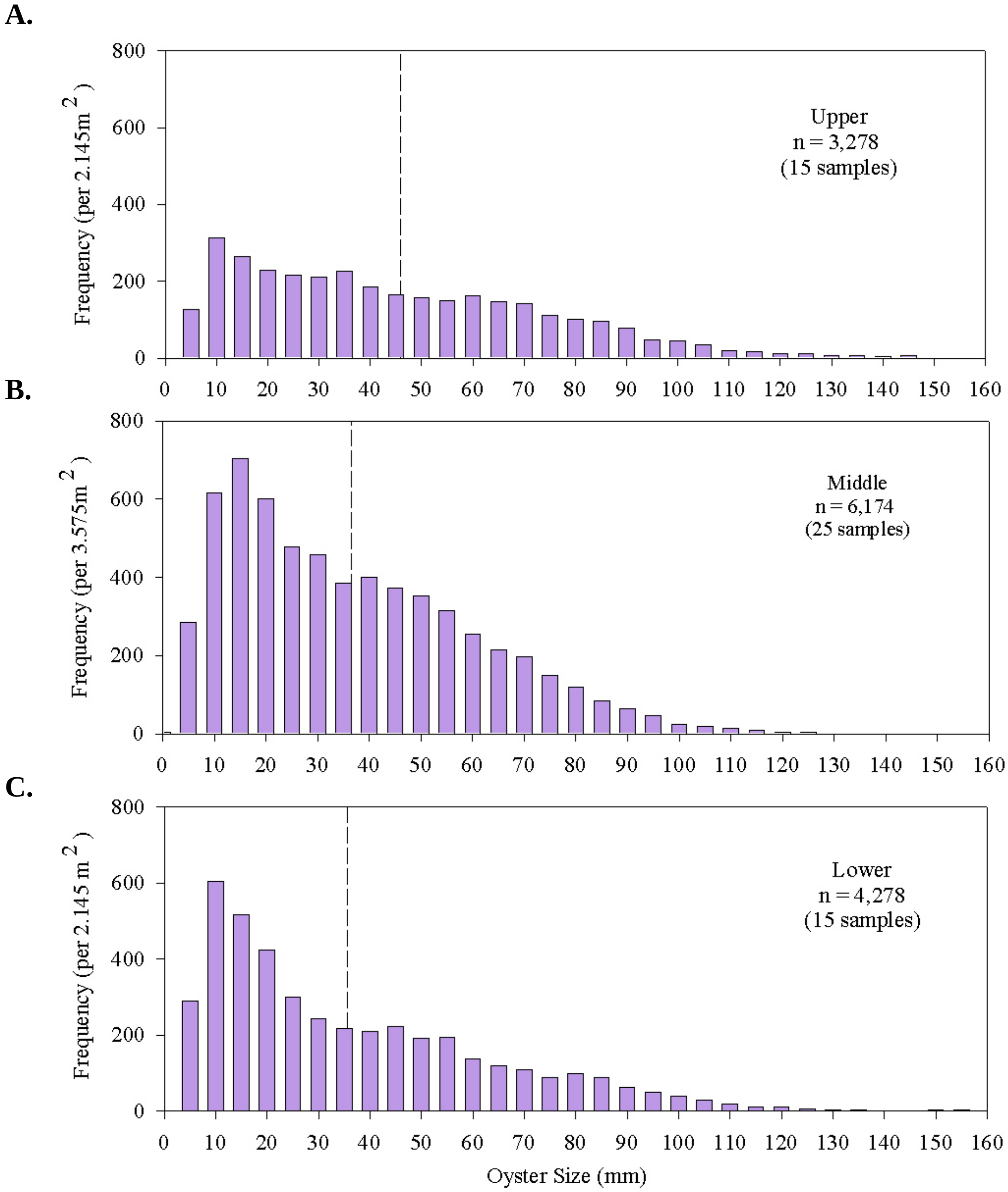


Figure IV-3. Oyster population size-frequency distributions for all May River Zones (A., B. and C.) summed across sites. Dotted vertical line indicates mean shell height and n is the total number of oysters collected for all that zone's samples (# in parentheses).

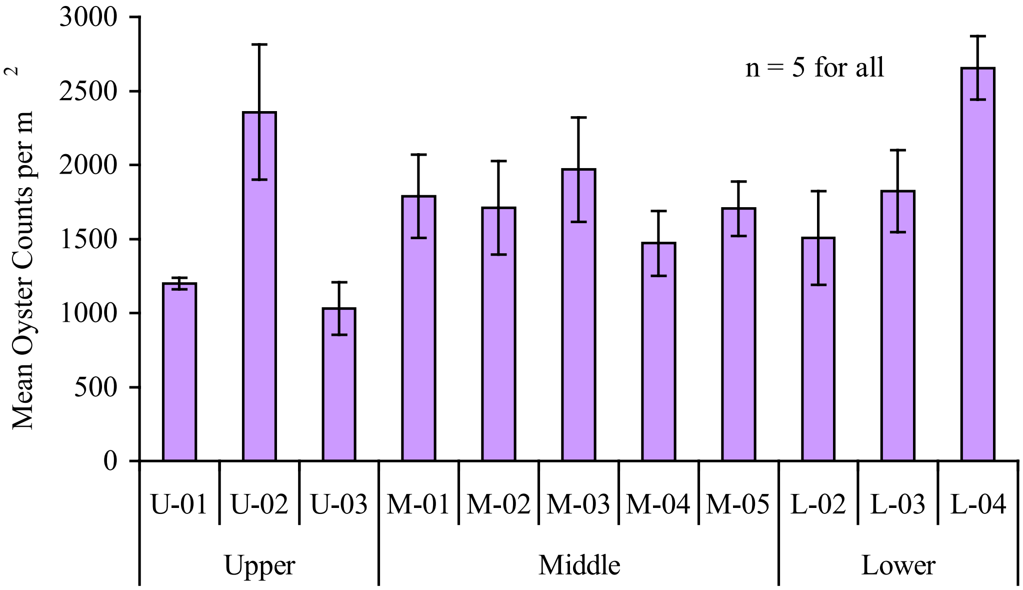


Figure IV-4. Mean number of oysters/m^2 for each May River site by zone for the 2002 population assessment. Error bars represent 1 standard error.

intensity levels among the sites within the Upper (Kruskal-Wallis, p-value = 0.157) and Lower zones (Kruskal-Wallis, p-value = 0.988). Similarly, there were no significant differences detected between the Upper and Lower zones (Kruskal-Wallis, p-value = 0.378). A statistical difference (p-value = 0.008) in infection intensity was, however, detected among the five Middle Zone sites, where site M-01 was significantly lower than sites M-02 and M-05.

Dermo infections (both prevalence and intensity) were similar to those observed previously for other South Carolina sites, where prevalence is typically high (>70- 80%) and mean infection levels rarely exceed 3.00 based on 30 years of sampling by DNR (Bobo and others, 1997; Bobo and others, in prep.). For comparison, Dermo levels at our six long-term sites sampled during the same period (August and September 2002) were comparatively higher (84-100%) than those observed at any of the 11 May River sites (52% to 88%). For infection intensity, levels were slightly higher at the six long-term monitoring sites (from 1.85 to 2.64) versus the May River sites, whose range was from 0.92 to 2.32 (Appendix IV-1c). In the southeast, the warm summer months are generally the time when *P. marinus* disease levels are at their highest (Crosby and Roberts, 1990; Bobo and others, 1997; Bushek and others, 2002). As we observed in our sampling since 1972 (Bobo and others, 1997) and for the Broad-Okatee study (Van Dolah and others, 2000), it is unlikely that any South Carolina oyster populations are free of Dermo and the long-term absence of the parasite from an oyster population is more likely noteworthy than its presence (Craig and others, 1989; Soniat, 1996).

MSX (*Haplosporidium nelsoni*) infections were present at 7 of the 11 sites in the May River (or 64%), with prevalence levels ranging from 0% at 4 of the 11 sites to 8% at L-04 (Figure IV-7; Appendix IV-1b). Infection intensities among individual oysters ranged from rare (few cells present in the entire tissue section) to heavy (cells present in most tissue sections), with most infections confined to the gills. Of the 275 individual oysters examined, eight (or 3%) were positive for MSX; one had a 'rare' infection, five were 'lightly infected', one was ‘moderately infected’, and one had a ‘heavy infection’. Kruskal-Wallis or ANOVA on ranks analyses detected no statistical difference in disease prevalence among the sites within each of the three zones, nor were significant differences found between the three zones.

The low MSX infections at the May River sites were within the lower range of those observed to date at any other South Carolina site (Dougherty and others 1993, Bobo and others, 1997; Bushek and others, 2002). No oysters from the Upper Zone (3 sites) had any MSX infections (Figure IV-7). Prevalence levels were low at both the Middle (5 sites, 0 – 4%) and Lower Zones (4 – 8%) and were within levels observed since 1993 by DNR (Bobo and others, 1997; Bushek and others, 2002). In comparison, MSX infections were observed in 50% of the six concurrent state-wide sites sampled during the same period (August and September 2002), where MSX prevalence levels ranged from 0% to 24% (Appendix IV-1c). Our 2002 observations were lower than the 1996 statewide assessment, where 54% of the oyster populations sampled in the state had MSX. For that statewide study, prevalence levels ranged from 0 to 32%, although prevalence levels at most sites were typically less than 20% (Appendix IV-1d).

Similarly in the Broad-Okatee Study that was conducted in 1997, MSX infected oysters were present at all 12 sites. Infection prevalence ranged from 4% to 33% in Broad Creek and from 4% to 12% in the Okatee River (Appendix IV-1e and Van Dolah and others, 2000).

Oyster Health

Lysosomes are regarded as valuable indicators of pollutant-induced injury (Moore, 1994) and the destabilization of lysosomes can be caused by environmental pollutants such as metals and PAHs (Moore, 1985; Regoli, 1992; Lowe and others, 1995; Ringwood and others, 1998a, b). A lysosomal destabilization level less than 35% is considered to be normal. The rates of lysosomal destabilization in May River oysters, which indicates hepatopancreatic toxicity, were all below 35%, and based on a long-term database, the responses were typical of non-stressed oysters (Figure IV-8). The lysosomal destabilization was similar to that observed in the Okatee River (Van Dolah and others, 2000).

Glutathione (GSH) is regarded as one of the most important “first-line” defense mechanisms in cells. GSH depletion has been hypothesized as a signal of stress (frequently exposure to metals), and a predisposing factor for increased adverse effects (Meister and Angerson, 1983; Viarengo and others, 1991; Regoli 1998). The levels of GSH were generally similar across all sites (Figure IV-8). The site with the lowest level was L-04; however, all levels observed were greater than the levels in the Okatee River (Van Dolah and others, 2000). Glutathione levels, an important anti-oxidant response, were also in the range typical of healthy oysters.

Finally, levels of lipid peroxidation (LPx) have not been shown to correlate well with contaminants but may indicate DO stress. In the May River, elevated LPx levels (~ 500 to 600 nmol MDA/g) in oyster tissue were found at five sites, while oysters from the remaining 6 sites had lower levels (~ 100 nmol MDA/g) (Figure IV-8). If elevated levels indicate exposure to DO stress then the sites with the higher LPx values (U-02, U-03, M-01, L-03, L-04) may have some DO stress; however, in general a consistent pattern was not observed with the summer DO data collected for these sites. It is unclear from the May River results whether the LPx results pose a problem to the oysters. Overall, our assessment of the cellular biomarkers of organismal health was "good", especially with regard to the lysosomal destabilization and glutathione assays.

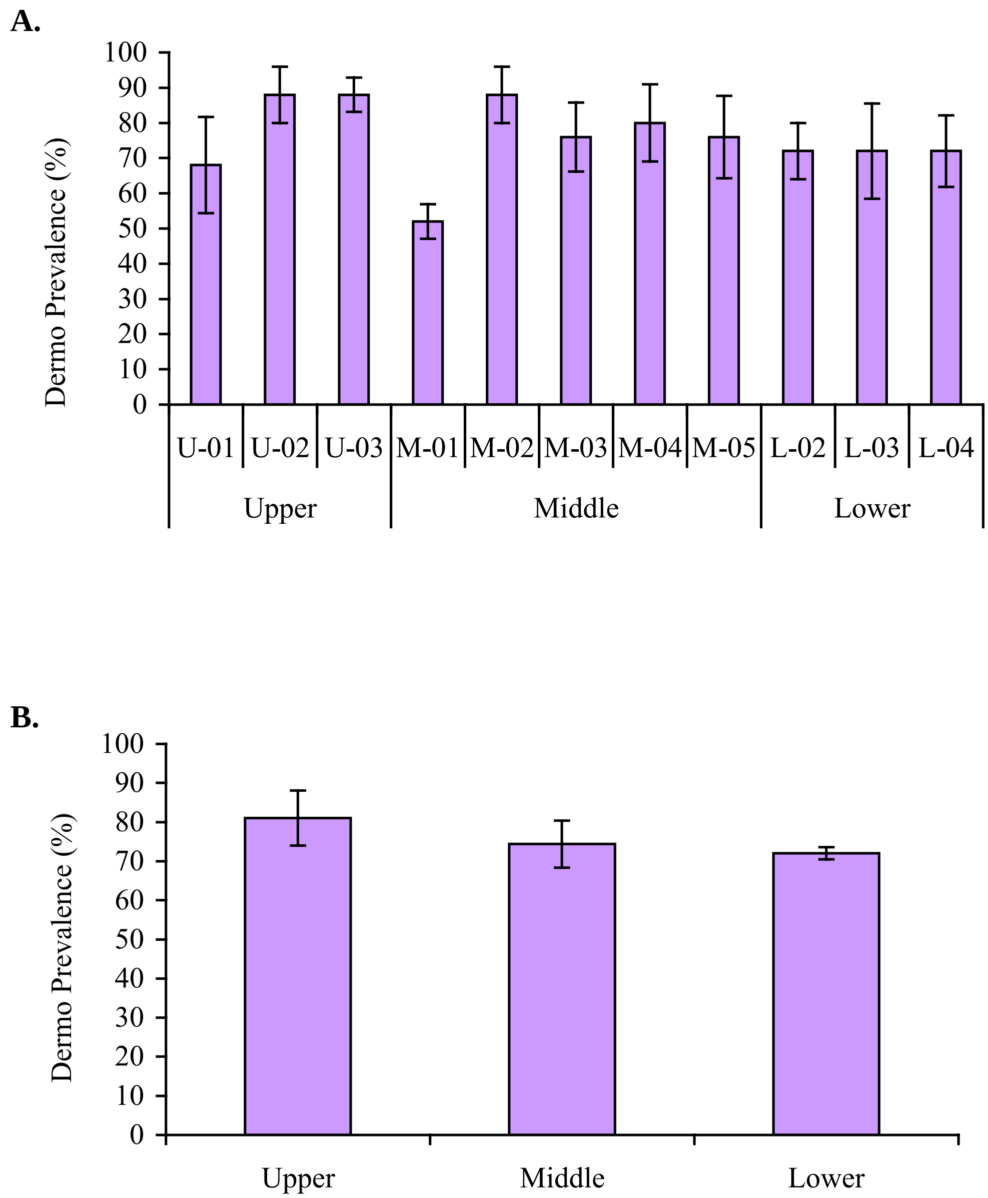


Figure IV-5. Perkinsus marinus (Dermo) prevalence in oysters sampled in 2002 averaged by site (A.) and by zone (B.) for the May River. Error bars represent 1 standard error.

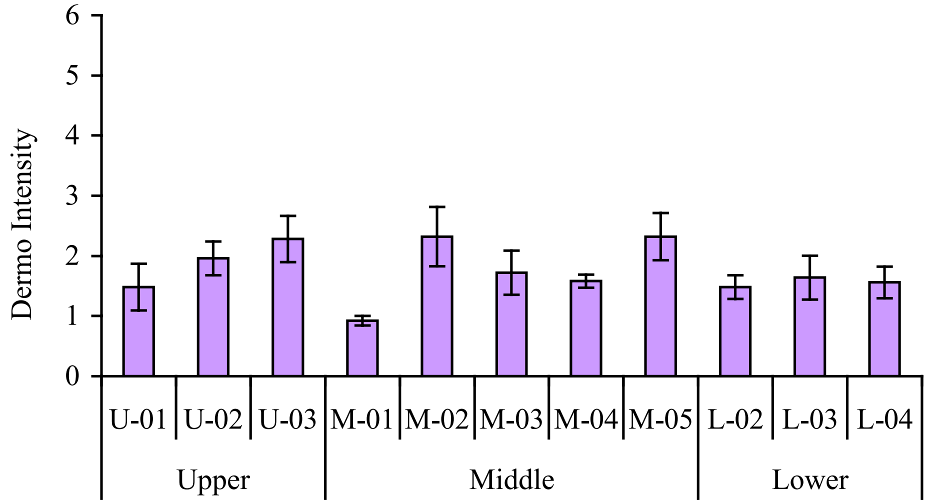


Figure IV-6. ***Perkinsus marinus*** **(Dermo) infection intensity (mean infections) in oysters by site within the zones sampled in the May River. Error bars represent 1 standard error.**

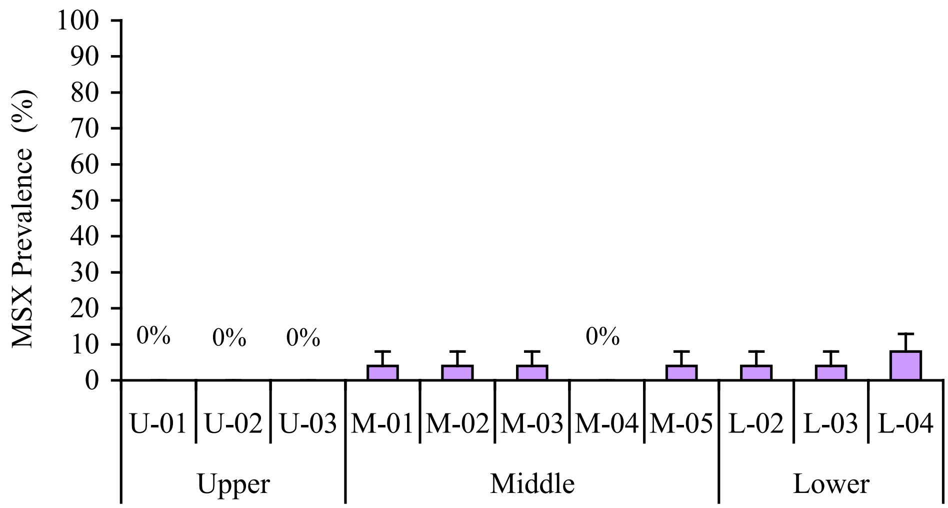


Figure IV-7. ***Haplosporidium nelsoni*** **(MSX) prevalence in oysters sampled by site in 2002 within each zone of the May River. Error bars represent 1 standard error.**

Tissue Contamination

The oyster tissue lipid concentrations were similar and ranged from 5.11 to 6.68%. The total PAH concentrations in the May River oyster tissues ranged from 0 to 155.34 ng/g dry weight (Figure IV-9). The total PAH concentrations were < 10 ng/g at 8 of the sites sampled. The two sites with total PAH concentrations greater than 10 were M-01 (155.43 ng/g) and U-03 (42.6 ng/g). The PAHs that were found above the detection level were chrysene+triphenylene (1 site), dibenzothiophene (7 sites), fluoranthene (2 sites), phenanthrene (1 sites), and pyrene (1 site). Dibenzothiophene is a component of petroleum that is fairly persistent (Irwin and others, 1997). These levels are similar to the ranges observed in oysters collected from North Inlet, a pristine National Estuarine Research Reserve (NERR) (Fortner and others, 1996).

Fourteen of the 15 metal analytes were detected in all of the oysters sampled in the May River. The only analyte that was only detected at one site was thallium. The ten primary metals in the May River oyster tissues were divided by their molar weight in order to sum the metals together. The sum of the ten metals ranged from 25.26 to 57.16 μM/g dry weight in the May River oysters (Figure IV-10). The three Upper Zone sites had the highest values. In general, most of the metal concentrations were similar among the 10 sites analyzed except for copper and zinc concentrations. The copper concentrations ranged from 47 to 129 μg/g dry weight. Oysters from North Inlet were found to range between 22 and 85 μg/g dry weight (Fortner and others, 1996). A molar copper level exceeding 2 μM/g dry weight was found to be associated with an approximately 50% drop in overall oyster condition (Ringwood and others, 2003). Site M-01 had a molar copper concentration of 2.41 μM/g dry weight.

The zinc concentrations ranged from 1,170 to 2,580 μg/g dry weight among the sites. Oysters from North Inlet were found to have zinc concentrations ranging between 420 and 2,500 μg/g dry weight (Fortner and others, 1996). A molar zinc level exceeding 40 μM/g dry weight was found to be associated with an approximately 50% drop in overall oyster condition (Ringwood and others, 2003). None of the sites had molar concentrations greater than 40 μM/g dry weight; however, U-01 and U-02 had molar concentrations close to this level. Based on the analysis of all metal concentrations, May River oyster tissue metal concentrations were similar to pristine areas; however, a few sites had concentrations reaching some levels of potential concern.

Total PCB concentrations were similar among the 10 sites. The values ranged from 3.30 to 6.13 ng/g dry weight. Total pesticide concentrations ranged from 8.39 to 33.39 ng/g dry weight in the May River oyster tissues. Site M-03 had the highest concentration of total pesticides was driven by lindane followed by chlorpyrifos. Lindane was once widely used as a pesticide to treat lice, agricultural products, livestock, and trees (Weinhold, 2001). Chlorpyrifos is an insecticide, which has been used in the south to treat termites.

Comparison of tissue levels in oysters with established Food and Drug Administration guidelines for safe human consumption (see Scott and others, 2002) indicated that no values exceeded these guidelines. Comparison with USEPA tissue guidelines for carcinogens and non-carcinogens for human consumption (see Scott and others, 2002) indicated slight exceedances

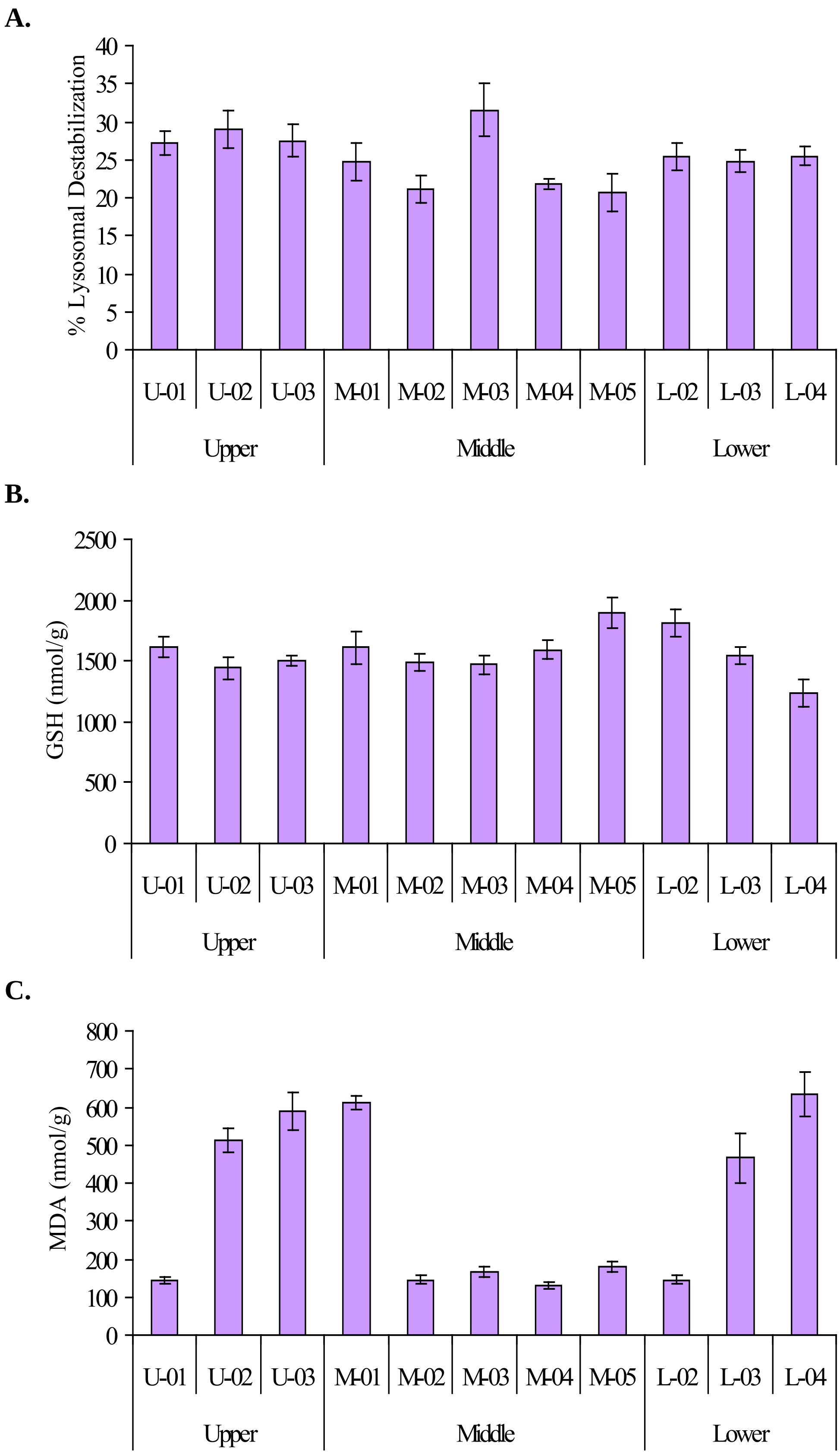


Figure IV-8. Oyster health for lysosomal destabilization (A.), glutathione (B.), and lipid perioxidation (C.) by site. Error bars represent 1 standard error.

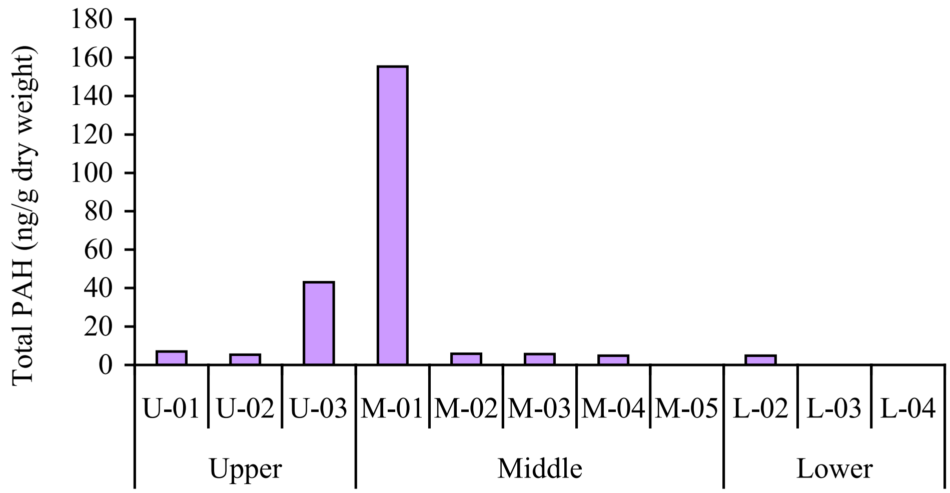


Figure IV-9. Total PAH concentration in the oyster tissues for each site sampled in the May River.

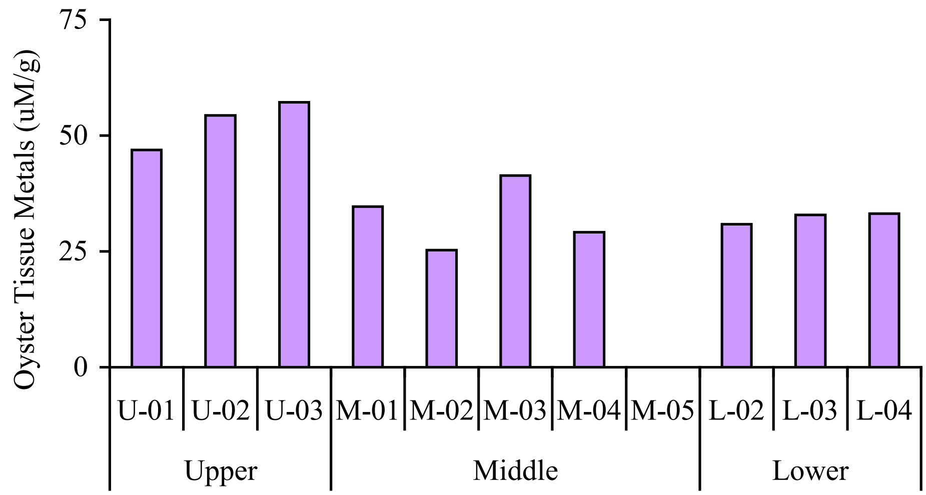


Figure IV-10. Sum of ten trace metals divided by the molecular weight concentration in the oyster tissues for each site sampled in the May River.

for total PAHs; however, similar exceedances have been observed in oysters sampled in the ACE Basin, where there were only atmospheric sources (Scott and others, 2000). The overall tissue concentration of PAHs in the May River (< 156 ppb) were only slight exceedances for PAHs and do not represent evidence of pervasive PAH pollution within the May River. Additionally, there was a general absence of PAHs associated with urban runoff. Therefore, PAH pollution from atmospheric sources may be more likely. The fact that tissue PAH residues exceed EPA guidelines underscores the importance for controlling non-point source runoff from urban developments in the future.

Tissue residue concentrations were also lipid normalized and divided by their respective octanol water partitioning coefficient to extrapolate mean estimated surface water concentrations (tissue residue/K_{ow} = mean surface water concentration). The results indicated that none of the individual contaminants exceeded more that 30% of the established USEPA water quality guidelines (Scott and others, 1997). Using oysters as biofilters establishes long-term measurements of surface water contaminant concentrations and provides an excellent estimate of contaminant exposure within estuarine water bodies. Our results indicate extremely good water quality in the May River.

Oyster Recruitment

Recruitment of oysters (i.e., the status of settled oysters at some arbitrary length of time after settlement) varies not only throughout the natural range of oysters, but also within an estuary (e.g., Roegner, 1991). Factors affecting recruitment include the number of larvae available in the water column to settle, current velocity, suitability of the substrate, competition for space, and the presence of predators (e.g., Olafsson and others, 1994; Wildish and Kristmanson, 1997). Other environmental factors that may potentially affect recruitment to an area include salinity, dissolved oxygen concentration, and temperature. Recruitment studies using techniques similar to the method used in the May River have been conducted throughout South Carolina over a number of years. These studies provide useful reference data to compare recruitment success in the May River with other sites throughout the state.

The mean size of recruited oysters was significantly larger in the Upper Zone than the Middle and Lower zones. This finding is similar to the observed natural oyster population as described above. Specifically, mean oyster SH for the deployed trays (deployed for 323 to 336 days) ranged from 18–28 mm (Figure IV-11), with the range of previous statewide averages around 22 mm (deployed for 281-336 days). Seven of 11 May River sites (or 64%) had a mean SH of newly recruited oysters at or above this average (Figure IV-11, p-value = 0.0047). No significant differences among sites were observed in the Upper and Middle zones; however, newly recruited oysters at site L-04 were significantly larger than those recruiting to trays in L-02. Nearly all of the sites with low oyster recruit numbers had larger than average-sized oysters (Figure IV-12). This may be explained by the fact that oyster densities were much lower at these sites, resulting in lower food depletion, lower competition between individuals for space and food resources within the trays, and therefore greater individual growth. The cumulative frequency analyses (Figure IV-12) detected significantly greater percentages of relatively small, recently recruited oysters or “spat” (<15 mm SH) within the Middle and Lower zones (28% and 26%, respectively), as compared with the Upper Zone (15%), consistent with the natural oyster population study. Four sites had some oysters with SH > 80 mm, but these oysters accounted for <1% of the overall total number.

The total number of recruited oysters (three replicate samples summed per reef site) were not significantly different among sites and ranged from a low of 930 individuals/m^2 at U-03 to 3,768 individuals/m^2 at M-02. Recruitment was significantly different among zones (p-value < 0.0001) and ranged from 928 individuals/m^2 in the Upper Zone to 3,759 individuals/m^2 in the Lower Zone (Figure IV-13). When compared to SCDNR long-term recruitment studies across the state (mean = 3,111 individuals/m^2 for 36 sites), 7 of the 11 May River sites (64%) had recruitment below this average, and two sites (U-01 and U-03) had abundances well below the mean number for the rest of the state.

Analyses of slopes of the size-frequency data showed significant differences (p-value = 0.0006) among the three May Rivers zones, as the Lower Zone had a significantly higher slope (indicative of more oysters in the smallest size classes) than the Upper Zone. Similar to the natural oyster population part of this study, the Middle Zone was not significantly different from either of the other two zones. Similar intercept analyses detected no significant differences among the zones (p-value = 0.1413). Finally, no significant differences in the intercept or slope among sites within a zone were detected.

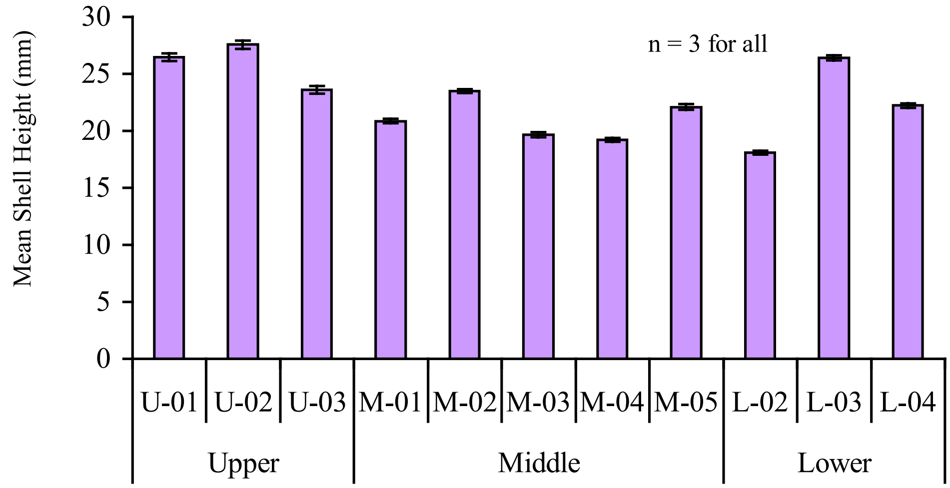
Oyster Reef Residents

Healthy and functioning oyster reefs are an important habitat for many organisms including numerous mobile (i.e., finfish, crabs, and shrimp) and sedentary species (i.e., mussels). Both brachyuran and anomuran crabs are important reef residents as they are significant oyster predators, and are potentially more efficient than blue crabs (*Callinectes sapidus)* in consuming larger oysters up to 50 mm (Reames and Williams 1983; Bisker and Castagna, 1987; White and Wilson, 1996). Resident macrofauna collected from the natural oyster reef habitats and deployed recruitment trays were placed in the following abundance classes: 0 (0 observed individuals), 1 (1-9 observed individuals), 2 (10-50 observed individuals), or 3 (> 50 observed individuals).

Mussel abundance ranged from 0 to 2 individuals/0.143 m^2 (or a total of 11-50 individuals) for natural population samples versus 0 to 1 individuals/0.434 m^2 (or a total of 1-9 individuals) for recruitment samples. The majority (53 out of 55 samples or 96%) of natural oyster community study samples had one or more mussels, 31 of 55 had between 1 and 9 mussels (abundance class 1); 20 of 55 had between 10 and 50 mussels (abundance class 2); and 2 had >50 (abundance class 3). In the recruitment study, only 4 of 33 samples (or 12%) had mussels in abundance class 1. The low numbers in the tray samples are not surprising given that mussels recruit slowly (Luckenbach and others, In press; SCDNR, unpublished.). Although many sites throughout South Carolina (e.g., Inlet Creek and Toler's Cove, Charleston County) exhibit extremely high mussel densities (abundance classes 2 and 3 or some as high as 1,000/m^2), as a whole most sites sampled to date in Beaufort County (e.g., Broad-Okatee, Warsaw) have had mussel abundances less than abundance class 2.

Forty-seven of the 55 (over 85%) natural oyster population samples across the 11 sites had one or more resident brachyuran (i.e., xanthids) or anomuran (i.e., green porcelain crab, *Petrolisthes armatus*) crabs. Twenty-nine of the 55 (or 53%) samples had 1-9 brachyuran crabs (abundance class 1) and 11 of 55 (20%) had between 10 and 50 brachyuran crabs (abundance class 2). Thirty-six of 55 (65%) had between 1 and 9 green porcelain crabs (abundance class 1) and 14 of 55 samples had between 10 and 50 green porcelain crabs (abundance class 2). No samples had 50 or more crabs of either type, despite previous years when *P. armatus* abundances exceeded 20,000/m^2 at many South Carolina sites. Cold winters and associated low tide

A.



B.

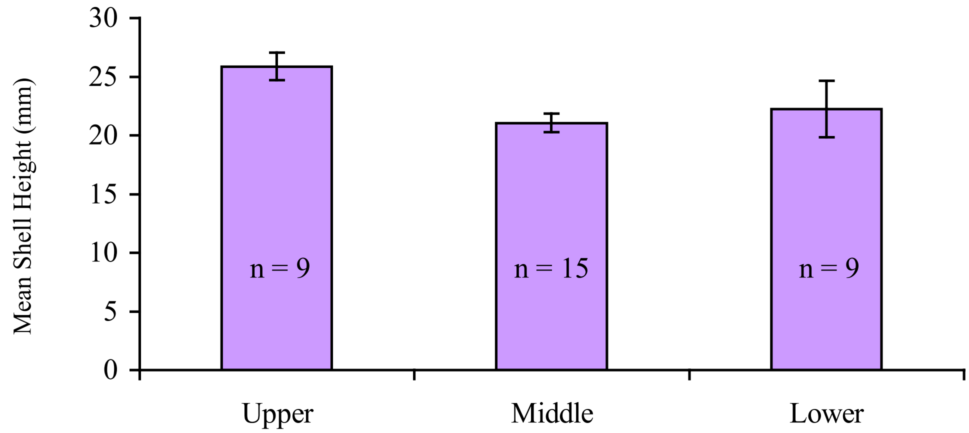


Figure IV-11. Mean shell heights by site (A.) and zone (B.) for 2002 recruitment assessment. Error bars represent 1 standard error.

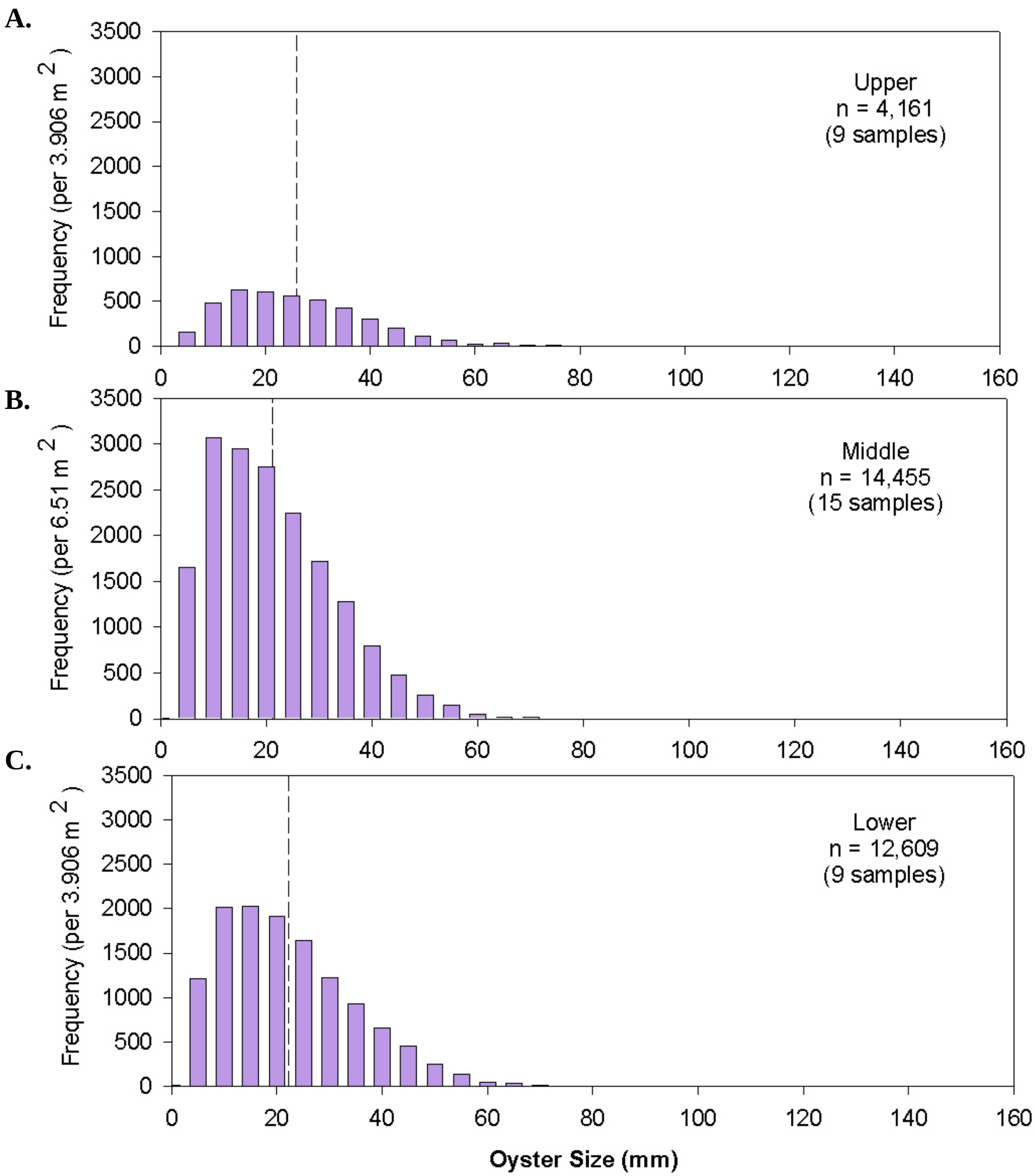


Figure IV-12. Oyster recruitment size-frequency distributions for all three May River Zones (A., B. and C.) summed across sites within a zone. Dotted vertical line indicates mean shell height and n is the total number of individuals per zone. Note that the areas are different with respect to the density of oysters per unit area.

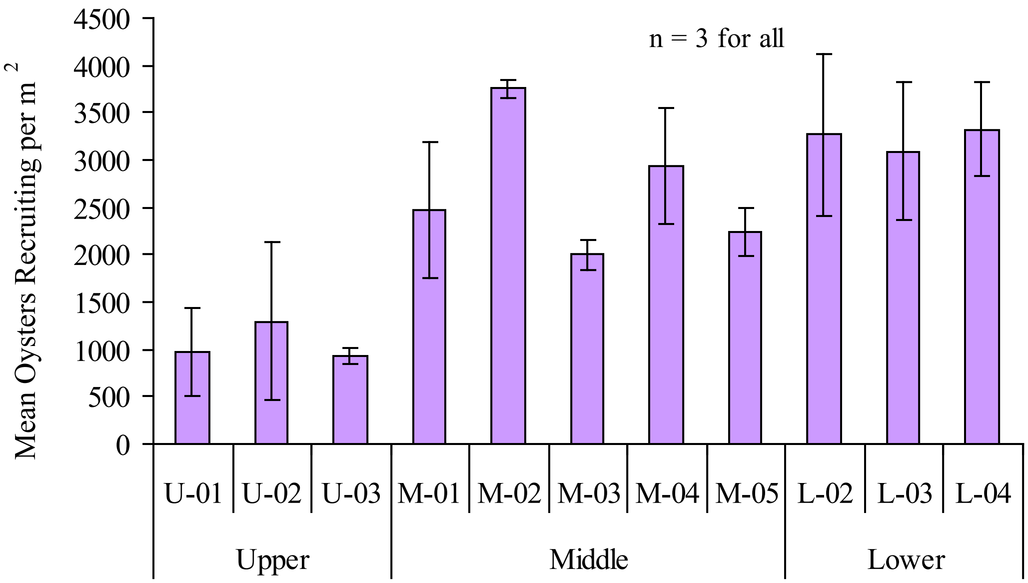


Figure IV-13. Mean number of oysters in recruitment trays (per m^2) by site. Error bars represent 1 standard error.

exposure has drastically reduced their numbers since 1999 (Coen and Knott, pers. obs.; Hartman and Lohrer, pers. comm.).

In recruitment trays, brachyuran crabs were present in all 33 trays, with 30 out of 33 (91%) having between 10-50 individuals (abundance class 2). Two samples had over 50 individuals (abundance class 3) and only 1 sample had less than 10 individuals (abundance class 1). *P. armatus* was found in 29 of 33 (88%) samples of which 9 had between 1 and 9 porcelain crabs (abundance class 1), 15 of 33 (45%) samples had between 10 and 50 (abundance class 2), and 5 of 33 (15%) samples had > 50 crabs (abundance class 3). The lower numbers of crabs in natural oyster community replicate samples was not surprising as the sample area is greater for the recruitment trays.

V. CONCLUSIONS AND RECOMMENDATIONS

The primary purpose of this study was to collect baseline data on the water quality, sediment quality, and biological condition of the May River prior to the construction of major planned unit developments (PUDs) in this drainage system, and to compare current conditions with data obtained from other studies of comparable habitats including the relatively pristine sites sampled during the same time period by the South Carolina Estuarine and Coastal Assessment Program (SCECAP) (Van Dolah and others, 2000, 2002; Holland and others, 2004). Studies to evaluate the impact of development in a given area can be designed in two general ways: (1) before- and after-impact studies, and (2) comparison of impacted areas to "control" or "reference" areas. In general, before- and after-impact studies are rare. In fact, little effort has been devoted toward gathering baseline data to characterize estuarine systems before the on-set of major pollution problems (Kennish, 2002); however, such data are critical to detect future changes in biological responses and quantify lost resources (Clark and Greene, 1988). The Town of Bluffton has been extremely proactive in seeking funding to perform a baseline study of the May River. The baseline data obtained from this study will enable the Town of Bluffton to compare future monitoring data with conditions observed in 2002-2003 in order to detect and correct any adverse changes that may occur with development of the May River watershed.

A triad assessment of water quality, sediment quality, and biotic condition was used in this study to evaluate overall condition in each habitat (i.e., headwater creeks, large tidal creeks, and open water sites) using a weight of evidence. Based on current State criteria and regional guidelines, the results indicated that most of the May River estuarine habitats are in good condition, although several headwater creeks showed some signs of stress (Table V-I). Based on an evaluation of land use patterns, the stressful conditions observed in these creeks were probably not related to anthropogenic inputs, and are likely natural phenomena of this system.

Table V-1. Summary of environmental and ecological condition of headwater creek, large tidal creek and open water habitats in the May River.

	Headwater Tidal Creeks	Large Tidal Creeks	Open Water Habitat
Water Quality	Fair most measures poor in Rose Dhu, Stoney, Brighton Beach Crks.	Good to Fair some measures marginal or poor	Good
Sediment Quality	Good to Fair some sites had higher contaminants and toxicity	Good	Good
Biotic Condition	Good to Poor stressed benthic communities in Rose Dhu, Stoney, Palmetto Crks.	Good	Good

A. Water Quality

Continuous Water Quantity/Quality Assessment at USGS Gauge Sites

The May River has a small drainage area and small freshwater inflow that is dominated by semi-diurnal tides, and experiences large tidal ranges of up to 11 ft. There is a small increase (about 0.5 ft) in the tidal range in the system as the tide wave moves upstream from Brighton Beach to Pritchardville. The larger tidal streamflows at Brighton Beach (greater than 100,000 ft^3/s) decrease significantly by Pritchardville (less than 10,000 ft^3/s) due to the decreased channel geometry. A net movement of water upstream was observed during the study time frame.

The annual runoff values calculated for the Pritchardville site (1.4 $ft^3/s/mi^2$) indicated that a larger percentage of the rainfall in the corresponding watershed reaches the May River system, compared to annual runoff values for freshwater inland drainage basins (0.3-0.8 $ft^3/s/mi^2$). The reason for elevated runoff at this site may be due to: (1) the long narrow shape of the watershed which minimizes distances water must travel to the creek; (2) sandy soils allowing for efficient transport of water through shallow groundwater to the creek; and (3) limited water loss to deep aquifers. It is also possible that the runoff levels are not elevated and are typical of estuarine systems, however comparisons are not possible as few data sets have been collected in similar systems due to technological limitations.

The effects of rainfall and runoff from the watershed were clearly seen in the specific conductance time-series. These data also showed that the May River has limited flushing and long residence times. For example, the effects of a 4.8-inch rainfall was observed for over 60 days. This long residence time clearly underscores the importance of reducing contaminants from entering the May River.

Like the majority of coastal systems in South Carolina, the May River was naturally low in dissolved oxygen. For extended periods of time, dissolved oxygen concentrations were below the State water quality standard of a minimum daily mean concentration of 5.0 mg/L and a minimum instantaneous concentration of 4.0 mg/L. Transport of oxygen consuming constituents from natural loadings associated with the flooding and drying of tidal marshes as well as non-point source loading from the landscape after rainfall events, which are considered to be natural or anthropogenic depending on the land use, are causes for the observed increase in dissolved oxygen deficit. Spikes in dissolved oxygen deficit in the May River appear to be attributable to non-point source loading of tidal marshes and runoff from rainfall events, which appear to be from primarily natural sources given the current land use. Changes in land use which increase the amount of oxygen consuming wastes may cause greater hypoxic conditions in this system.

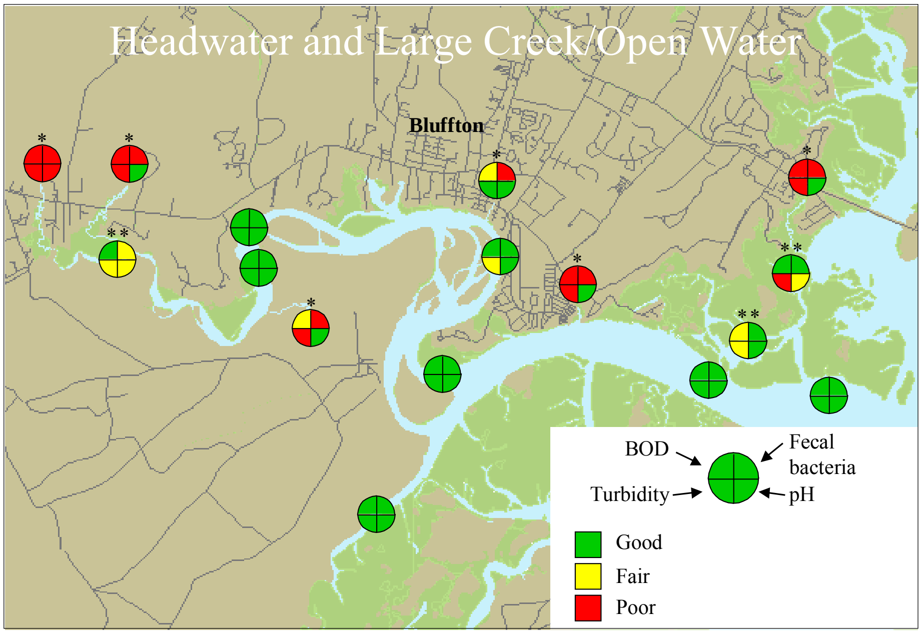
Assessment of Tidal Creek and Open Water Habitats (Seasonal and Summer)

Results obtained from the 2002-2003 seasonal sampling period and the more intensive 2002 summer sampling period for a subset of the water quality parameters were generally good and suggested that current land use activities were not affecting water quality at the majority of the sites sampled in the May River. Seasonal changes were identified for many of the water quality parameters measured in all of the habitats sampled, but most of the changes were typical of conditions expected in South Carolina waters.

In general, most water quality characteristics were as good as or better than those noted for similar systems in other portions of the state in previous studies (Van Dolah and others, 2000; 2002; Holland and others, 2004), and compared to conditions measured concurrently by SCECAP during this study. Sites that exhibited some evidence of potentially stressful water quality were primarily located in the headwater creeks and the uppermost portion of the river. Sites in the upper most portions were considered to be a large tidal creek by the SCECAP program based on width of the water body (Figure V-1). It should be noted that many of the standards used to classify the water quality of the May River were developed for large, deep water systems. The headwater tidal creeks, and to a lesser extent the large tidal creeks, are naturally stressful systems that are expected to have more extreme values. Therefore, the overall water quality was classified as fair or good to fair in these smaller drainage systems.

During one or two seasons, nutrient concentrations were higher in headwater creeks than the established NOAA guidelines, but did not appear to trigger an increase in phytoplankton biomass, with the possible exception of Stony Creek. Nutrient and phytoplankton levels in the large tidal creek and open water habitats were also considered to be low to moderate in concentration. While the presence of harmful algal species were observed occasionally in the large tidal creeks and open water habitats, none of the concentrations were of concern. Although presently in low numbers, blooms of harmful species can be triggered in response to nutrient loading. Adverse effects of *Kryptoperidinium foliaceum*, *Heterosigma akashiwo*, *Pfiesteria*, and *Scrippsiella* on shellfish have been demonstrated in other areas (Springer and others, 2002, Lewitus and others, 2003, Keppler and others, accepted).

Fecal coliform bacteria concentrations, while relatively high in all headwater tidal creeks, were generally not indicative of human sources. Only Stony Creek had some evidence of antibiotic resistance (i.e., human sources), but the evidence was weak. Similar low levels of antibiotic resistance have been detected in extremely pristine watersheds, such as North Inlet, SC. High bacterial counts in the unpopulated Palmetto Bluff Creek and the sparsely populated Stony and Rose Dhu creek watersheds indicate a natural source of fecal coliform bacteria that is probably attributable to wildlife. High concentrations of fecal coliforms have been observed in headwater tidal creeks during previous studies (Vernberg and others, 1996; Holland and others, 2004). Furthermore, samples in the current study were collected during the falling low tide to maximize the signature of the upland influence. Low tide concentrations of fecal coliforms are generally the highest observed during the tidal cycle. Although concentrations during high tide, when people might be using these systems for recreational contact, would be lower, caution should still be exercised in the headwater tidal creek systems since high bacterial counts are still possible. Fecal coliform levels were generally low in the other habitats, and only two of the open water



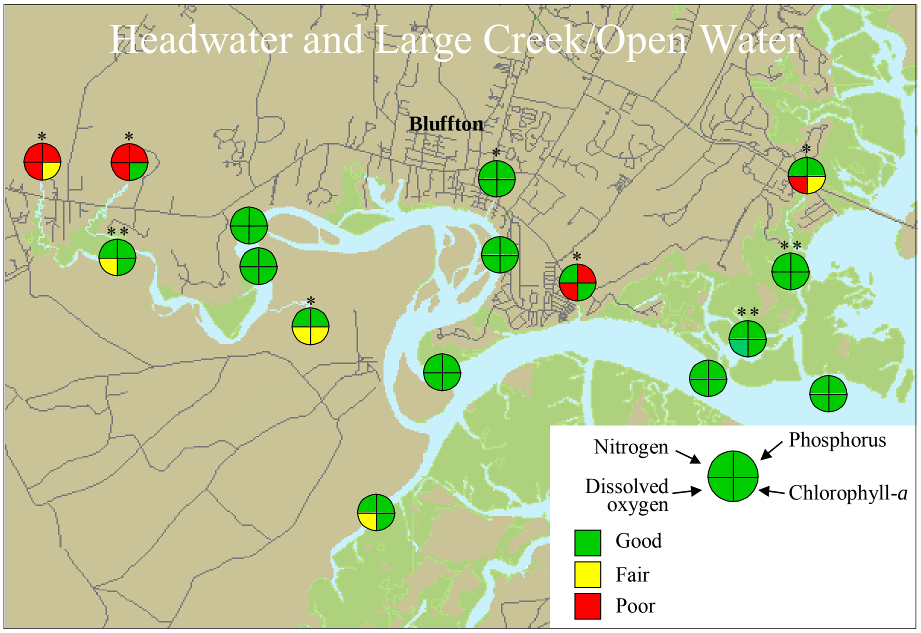


Figure V-1. Summary of key water quality measures collected at the May River sites. Green represents no exceedances of standards established by SCDHEC (2001) or criteria used by the SCECAP program (Van Dolah and others, 2002). Yellow indicates at least one exceedance, and red indicates two or more exceedances. Sites with an * represent headwater tidal creeks. Sites with ** represent large tidal creeks.

sites (M-02, M-03) showed any evidence of antibiotic resistance. These open water sites were located in the middle zone of the river in close proximity to the center of the Town of Bluffton.

Wastewater indicator compounds were only sampled at Heyward Cove and Palmetto Bluff creeks. These compounds were generally reported as estimated concentrations because concentrations were less than the standard reporting limit or analyses had poor recovery for that compound. Several wastewater indicator compounds were detected in Heyward Cove Creek during the winter that may be indicative of human sources of contamination from leaking sewer or septic systems. Heyward Cove Creek also drains a watershed with the highest human population density of any creek studied in the May River.

The sites sampled in the May River were classified for other water quality parameters such as BOD, turbidity, salinity, and pH based on a set of criteria (see Figure V-1). In general, the headwater tidal creeks had potentially degraded water quality but these are probably natural phenomena. Water quality measures within the larger tidal creeks and open water stations were generally very good and similar to comparable sites sampled during the same time period by SCECAP. Among the sites sampled, only station U-01, located in the uppermost portion of the May River, had DO and pH levels that were indicative of marginal conditions during the summer months based on criteria developed for the SCECAP program. Average and instantaneous dissolved oxygen conditions at all other large tidal creek and open water sites were generally good, even during the summer months.

Overall, the water quality of the May River is in good condition and similar to the pristine waters sampled elsewhere in previous studies (Lewitus and others, 1998; Van Dolah and others, 2002) and as part of this study. Where impaired conditions were observed, they were largely confined to headwater tidal creeks, but as noted previously, these conditions are probably naturally occurring rather than representative of land use effects.

B. Sediment Quality

The quality of sediments at the May River sites sampled in this study was generally good and comparable to other studies of relatively undeveloped watersheds with a few exceptions (Figure V-2). Suburban headwater creeks had significantly higher sand content than the forested headwater creeks, which is a trend that has been noted by Holland and others (2004). Suburban creeks are surrounded by increased amounts of impervious cover, and as a result experience ‘flashier’ runoff from the upland, which transports sandy, land-derived soils into creek beds. This increased runoff may also wash away finer grained sediment particles (like silts and clays), leaving behind sediment particles of a larger grain size. A change in sediment type is one of the first in a succession of responses to watershed alteration and results in an altered benthic community structure, which can have ecosystem-level ramifications. The larger tidal creeks sampled in this study also had a significantly higher sand content than the open

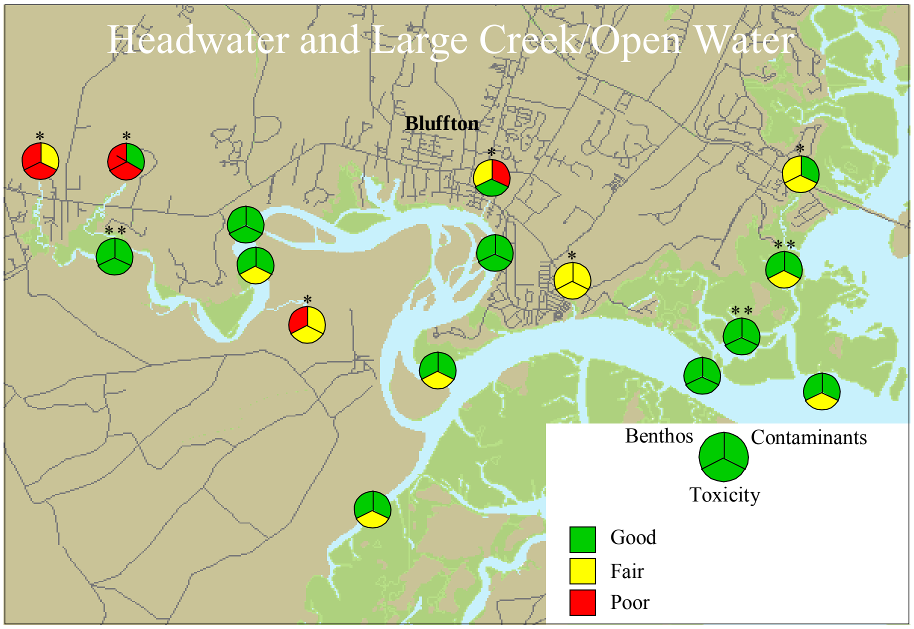


Figure V-2. Summary of sediment quality in the May River based on contaminant levels, sediment toxicity tests, and benthic community condition. Sediment contaminant guidelines were based on Hyland and others (1999). Toxicity condition was based on SCECAP criteria (Van Dolah and others, 2002). Benthic condition was based on B-IBI criteria described by Van Dolah and others (1999) for the large tidal creeks and open water sites and Holland and others (2004) for the headwater tidal creeks. Sites with an * represent headwater tidal creeks. Sites with ** represent large tidal creeks.

water sites, however, only one of these sites was in an area with suburban development nearby.

Contaminant levels were highest in three headwater creeks: Palmetto Bluff, Brighton Beach, and Heyward Cove creeks. Palmetto Bluff and Brighton Beach creeks only had one or two metals above effects range-low (ERL) concentrations, which are based on adverse bioeffects observed in 10% of the studies examined by Long and others (1995). Arsenic was above ERL concentrations at both sites, but this metal is naturally high in South Carolina estuarine sediments (Sanger and others, 1999; Scott and others, 2000; Van Dolah and others, 2000). The other metal found above ERL levels at Brighton Beach Creek was nickel; however, the ERL for nickel has low predictive capabilities. Heyward Cove Creek, which drains a suburban watershed, had three contaminants above ERL levels, all of which were polycyclic aromatic hydrocarbons (fluoranthene, acenaphthylene, benzo(a) anthracene). This site also had the highest integrated contaminant measure [effects range-median quotient (ERMQ) greater than 0.058], which corresponds to a high risk of observing a degraded benthic community in subtidal habitats (Hyland and others, 1999). However, most of the benthic measures collected at this site did not show evidence of significant degradation. None of the May River headwater tidal creeks had high concentrations of PCBs or pesticides. Overall, the ERMQ values for May River creeks were comparatively lower than those in the Charleston Harbor estuary (Sanger, 1998).

The larger tidal creek and open water sites sampled in the May River had low contaminant concentrations, with no sites having ERMQ concentrations that exceeded values of moderate or high risk of observing negative impacts on benthic communities in similar habitats (Hyland and others, 1999). In general, the contaminant concentrations were comparable to or lower than similar habitats located in pristine areas that were sampled during the same time period by the SCECAP program. Sediment characteristics at most sites were predominantly sandy, with relatively little silt and clay compared to the headwater creeks. Sandy sediments generally do not accumulate high concentrations of contaminants compared to muddy sediments. Unless sediment composition were to change due to increase runoff of silts and clays, it is unlikely that contaminants will ever be great enough to cause adverse bioeffects if proper stormwater controls are established for future developments.

The two sediment toxicity tests conducted using sediments from each site were inconclusive, as they did not correlate well with the concentration of sediment contaminants, expressed by the ERMQ values. Heyward Cove Creek had the highest ERMQ value of all headwater tidal creeks, and of any site measured in the entire May River (ERMQ = 0.0586), and yet the sediment was not toxic to the organisms in either of the assays used. In contrast, two of the forested creeks (Stony and Rose Dhu) showed toxicity in both of the assays, even though ERMQ values were relatively low. Toxicity responses in sediments from the larger tidal creek and open water sites were comparable to those observed at the relatively pristine sites sampled by the SCECAP program. Five of the ten sites in the May River showed toxicity in one of the two assays; however, six of the ten SCECAP sites also showed toxicity in one of the assays. The toxicity noted at sites with low ERMQ values could be due to the presence of contaminants not included in the ERMQ or not measured, but it is unlikely. Both assays can have a relatively high "false positive" when contaminant concentrations are not high, so the results obtained in this study should be interpreted with caution. Furthermore, the toxicity end point for both of these assays is indicative of sublethal effects.

C. Biological Condition

Benthic Communities

The assessment of benthic invertebrate communities provides one of the best biological measures of estuarine habitat condition, as these fauna are generally sessile and are often the first biota to be adversely affected by poor water quality and poor sediment quality. Most environmental studies routinely assess the benthos as a measure of ecological health. The benthic communities in the May River generally did not show evidence of stress, except at a few of the headwater sites, and even in those locations, stress was probably attributable to natural conditions (Figure V-2).

Oligochaete worms were the dominant fauna of headwater creeks. According to metrics developed by Holland and others (2004), stress-tolerant species were more abundant in the three headwater creeks near the upper end of the May River than in other creeks of comparable land uses. The benthic communities in these forested creeks (Stony, Rose Dhu, and Palmetto Bluff creeks) were more similar to creeks that drain heavily developed watersheds rather than forested watersheds. We believe that natural stressors (i.e., low DO) associated with creeks located in the headwater region of a river system and those that drain large watersheds resulted in the altered benthic community, rather than stressors associated with current land use activities.

Benthic communities sampled in the larger tidal creeks and open water sites were much more diverse and were dominated by several polychaete worms. The overall abundance of fauna was significantly greater at the open water sites compared to the creeks, which has been observed previously in the SCECAP program (Van Dolah and others, 2002). Various measures of species diversity also showed no clear pattern among sites, although station U-02 had significantly fewer taxa than most of the other open water sites. The clam assay showed significant toxicity at this site even though no elevated levels of contaminants were observed. A benthic index of biotic integrity (B-IBI) developed for the southeastern region (Van Dolah and others, 1999) indicated that all of the large tidal creek and open water sites had undegraded benthic communities. The lowest B-IBI value was found at station U-02, which may reflect some effects of sediment toxicity from unmeasured contaminants as noted previously. Overall, the mean B-IBI values observed at the open water and tidal creek stations were very similar to the B-IBI means measured at the relatively pristine SCECAP sites used for comparison.

Nektonic Communities

Fish and crustacean assemblages collected by seine at the headwater tidal creeks and by trawl at the other sites provide another biological measure of conditions in the May River. Grass shrimp and penaeid shrimp were the predominant taxa in the seine samples, with no significant differences observed among creeks; however, there was a generally decreasing trend in abundances as related to impervious cover. Penaeid shrimp were most abundant in Stony Creek, which in spite of its frequent occurrence of hypoxia and degraded benthic community, seemed to be functioning as a nursery and feeding area for nekton.

As noted in previous studies by Van Dolah and others (2002), the abundance of fish and crustaceans was significantly greater in the large tidal creeks compared to the open water sites. The abundance and diversity of nekton within each of these habitat types was comparable to similar sites sampled by the SCECAP program during the same time period in relatively pristine areas. Shrimp were also the predominant fauna in the larger tidal creeks and, to a lesser extent, at the open water sites. A variety of fish species were also collected at most sites. In general, the faunal assemblages collected in the trawls were similar to those collected at similar SCECAP sites although the relative abundance of the various species was different. The differences between species composition found in the May River samples and the pristine areas may reflect differences in habitat quality, but it is also likely to be related to other natural differences in the creek habitats. Finfish encounter complex natural variations in physical, chemical, and biological factors which strongly influence the accessibility and variety of estuarine habitats, consequently affecting the distribution, diversity, and abundance of these species (Monaco and others, 1992). It is also possible that the trawl catch for the May River and pristine SCECAP stations reported here is not fully representative of the nektonic communities actually present due to the relatively small number of stations sampled, particularly for large tidal creek stations.

Oyster Beds, Related Diseases, and Associated Recruitment

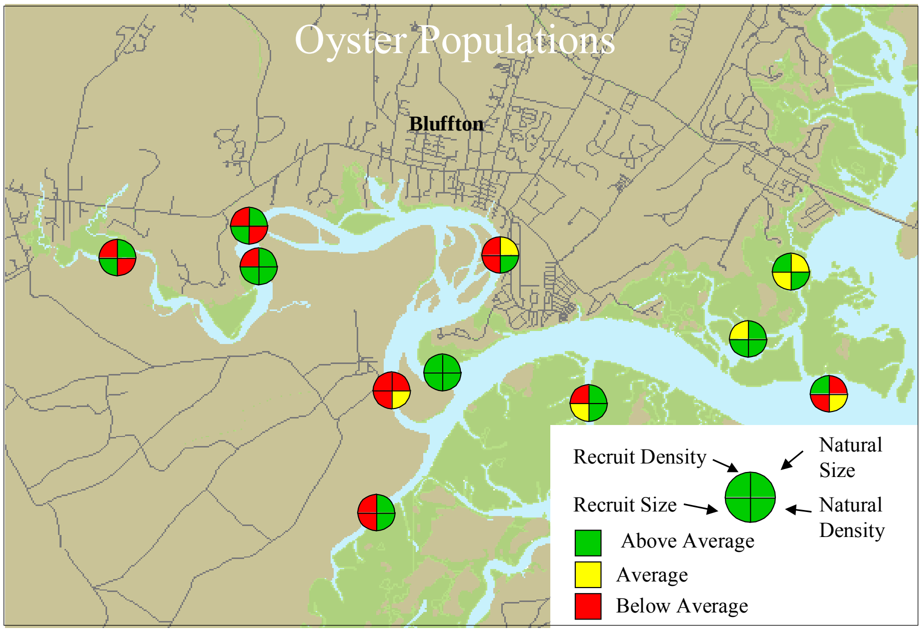
The May River is one of the few remaining relatively pristine systems in the state that includes oyster beds whose DHEC shellfish harvesting classification is still entirely classified as "open to harvesting", with no recent closures. In addition, most of the areas we studied are included within active and productive oyster leases. An integrated summary of overall oyster bed condition for each of the May River sites is shown in Figure V-3. Overall measures of oyster bed status and health, oyster size and number (density), recruitment and growth (= recovery potential), and disease occurrence and infection levels were considered. When the May River oyster beds were compared to recent SCDNR statewide averages generated from numerous sites, the overall condition of the oyster beds was 'average' to 'above average' and the recovery potential was 'average' to 'below average'. The extended drought period (4 years) encompassing the current study and active harvesting makes interpretations of natural bed condition relative to environmental quality difficult. Assessments are futher complicated due to the culture lease status of many sites.

Oysters disease levels (Dermo and MSX) at the May River sites were highly comparable to results observed from other South Carolina oyster population studies since the 1990s and were ranked 'good' (Figure V-3). Dermo infection intensity levels at all sites were below a value of 3 (on a scale of 0-6), which is normal for South Carolina oyster populations based on SCDNR long-term disease data (Bobo and others, 1997; Bushek and others, 2002; SCDNR, unpublished). MSX levels were all below 20% occurrence, which is the maximum annual mean MSX prevalence ever observed for South Carolina oyster populations based on DNR studies since 1994 (Bobo and others, 1997; Bushek and others, 2002; SCDNR, unpublished).

The physiological measures of oyster condition generally indicated good health of the organisms at all sites, especially with regard to the lysosomal destabilization and glutathione assays (Figure V-3). The rates of lysosomal destabilization, which indicates hepatopancreatic toxicity, were all below 35%, which is typical of non-stressed oysters based on the SCDNR database. Glutathione levels, an important anti-oxidant response, were also in the range typical of healthy oysters. The contaminant concentrations in oyster tissues were similar to pristine areas in the North Inlet National Estuarine Research Reserve.

When contaminant concentrations were evaluated in the oyster tissue collected from all sites, only very low concentrations of trace metals, pesticides, PCBs and a few PAHs were identified. Comparison of tissue levels in oysters with established Food and Drug Administration guidelines for safe human consumption indicated that no values exceeded these guidelines. Comparison with USEPA tissue guidelines for carcinogens

A.



B.

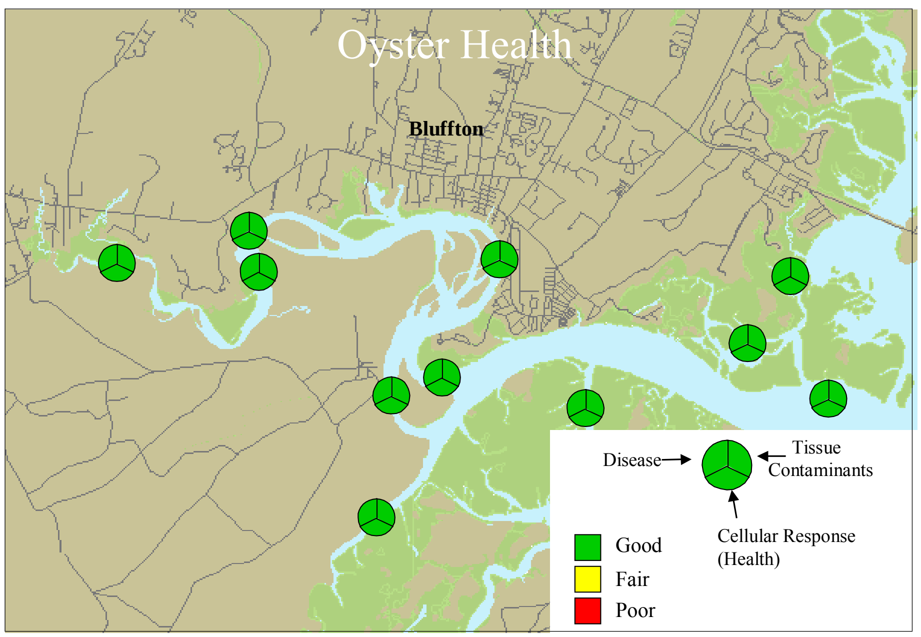


Figure V-3. Summary of oyster bed condition and oyster health assessment in the May River. Figure A. shows condition on the size and density of oysters in the existing beds sampled and the size and density of newly recruited oysters to recruitment trays. Figure B. summarizes oyster conditions with respect to disease relative to statewide conditions, physiological health, and tissue contaminant levels.

and non-carcinogens for human consumption indicated slight exceedances for total PAHs; however, similar exceedances have been observed in oysters sampled in the ACE Basin, where there were only atmospheric sources (Scott and others, 2000). The overall tissue concentration of PAHs in the May River do not represent evidence of pervasive PAH pollution within the May River. Additionally, a likely source of these PAHs is atmospheric since there was a general absence of PAHs associated with urban runoff. The fact that tissue PAH residues exceed EPA guidelines underscores the importance for controlling non-point source runoff from urban development in the future. The results of the lipid-normalized tissue analysis to extrapolate mean estimated surface water concentrations indicated that none of the individual contaminants exceeded more that 30% of the established USEPA water quality guidelines (Scott and others, 1997). Using oysters as biofilters establishes long-term measurements of surface water contaminant concentrations and our results indicate extremely good water quality in the May River.

D. Implications

Urbanization and increased human population density in the May River watershed will likely be accompanied by changes in creek/river hydrology, alteration of the pattern of overland stormwater flow due to the unavoidable increase in impervious cover, and possible declines in approved shellfish growing waters. Increased impervious area will result in increased quantities of contaminants, such as lawn and garden fertilizers and pesticides, additional quantities of suspended solids in runoff, and changes in salinity or nutrient inflow to the estuary. Increased numbers of pets and potential sewer system leaks may result in higher concentrations of bacteria and the closure of shellfish beds. This is presented in the conceptual model developed by Holland and others (2004), which depicts the manner in which human population growth is related to adverse changes in the physicochemical environment and also to the biotic composition of headwater tidal creeks (Figure V-4). A comprehensive stormwater diversion/treatment plan for all planned developments will alleviate this situation, but it will also alter the natural overland flow patterns. As the amount of impervious cover increases, the changes it brings about will likely have an effect on the river's ecosystem. Continuous monitoring and periodic comprehensive evaluations that will allow regulators to respond to potential or realized changes in water quality and the estuarine ecosystem should be undertaken to prevent the degradation of the natural resources of the May River Estuary.

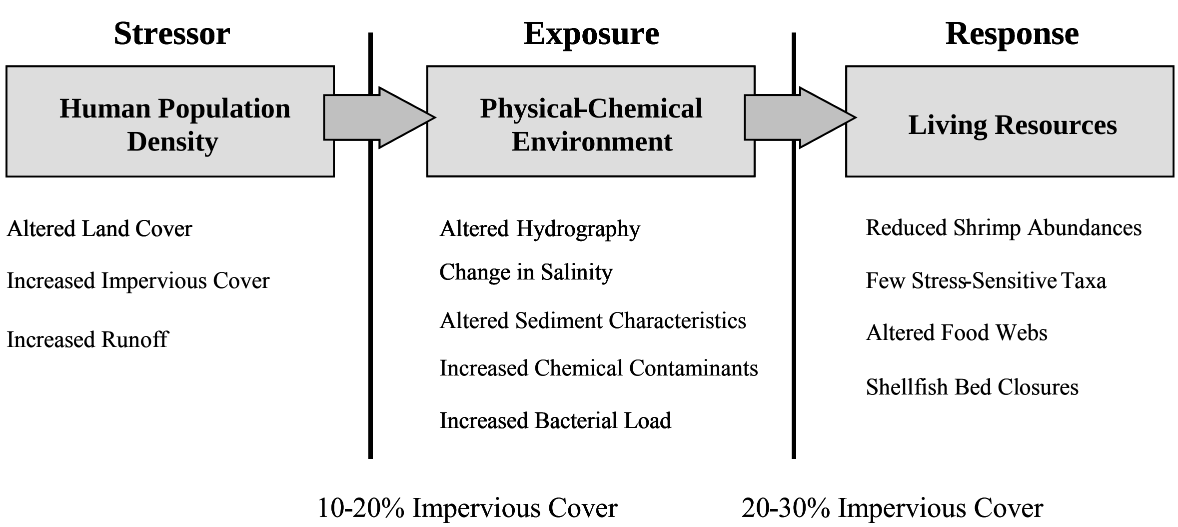
Holland and others (2004) presented several significant regression models that examined the relationship between impervious cover and key ecological factors, including: population density, salinity range, % silt/clay, chemical contaminants, % stress-sensitive taxa, % stress-tolerant taxa, and the abundance of peneaid shrimp. To better understand how May River headwater tidal creeks compared to headwater tidal creeks throughout South Carolina, the regression models developed by Holland and others (2004) were plotted with the May River sites added (Appendix Figure V-1). In general, May River creeks were typical of undeveloped and lesser-developed creeks of South Carolina. For each of the parameters that were evaluated, the May River and TCP data points were in the same range. In some cases, the May River creeks were of higher quality than their TCP counterparts (i.e., chemical contaminants). If only evaluating the May River headwater tidal creeks, no significant relationships were found between impervious cover and any of the parameters evaluated for the May River creeks. This was probably due to the relatively low impervious cover in all May River watersheds, which represent only a narrow range of the values previously investigated.

The lack of degraded water quality conditions in the three larger tidal creeks and open water habitat, with the exception of a few measures, indicates that current land use activities have not had an adverse effect on these important habitats. Conditions may be maintained if new planned unit developments have rigorous controls related to best management practices (BMPs).

Furthermore, the current study has revealed that the tidal creeks (Stony and Rose Dhu creeks) located in the true headwater region of the terminal May River system may function differently than previously sampled tidal creeks that were not located in headwater areas of a major drainage system. The functional differences may be related to their terminus status as well as the observation that the watershed sizes of these types of creeks are 3 to 10 times larger than other creeks sampled. A recent study of the Okatee River supports the finding that these terminal river creeks respond differently and have comparatively large watersheds (Sanger and Holland, unpublished).

At present, the May River Estuary appears to have a good overall ecological condition and habitats within the system were comparable to other areas throughout South Carolina with little to no development. However, in the coming years population size is expected to grow substantially in this watershed, and this growth will be accompanied by an increase in impervious cover. The conceptual model and the regression models provide insight into the continuum of change that can be expected to occur in headwater tidal creeks and potentially the larger system. In order to maintain the good water quality and ecological conditions of the May River, new planned developments may require rigorous controls related to BMPs.

Figure V-4. Conceptual model of the linkages between land use and headwater tidal creek environmental quality (Holland and others, 2004).



E. Recommendations

The following recommendations are provided to assist the Town of Bluffton in managing their watersheds and conducting future monitoring of condition in the May River. The USGS and NOAA fully support the scientific findings and interpretations of this study; however, as a matter of policy, the USGS does not endorse or make recommendations. Also see NOAA disclaimer on page iv.

- The Town of Bluffton should delineate the sub-watersheds of May River for all areas to enable a clear understanding of where upland runoff is flowing.

- An educational campaign related to watershed awareness should be undertaken to instill in the residents that everything they put on their lawns, etc. will end up in their creek, which drains into the May River. Citizens can take an active part in helping to maintain water quality of the May River and other nearby drainage basins using vegetated buffers, pet waste disposal (e.g., pooper scooper programs), hazardous waste disposal (e.g., oil recycling), stormwater pond maintenance, septic system maintenance, proper lawn care (i.e., pesticide and fertilizer use). The Town may want to post signs to indicate what watershed residents are in.

- Continuous monitoring of the May River provides a dynamic record of how the estuary is responding to changing hydrologic conditions. The Town of Bluffton should consult with the USGS regarding the value of continuing operation of one or more of the existing gauges and what parameters would be most useful to monitor.

- The high annual runoff or watershed yield at the Pritchardville gauge site, which indicated that more rainfall runs off the upper May River watershed by either surface flow or shallow groundwater, is an important finding. This indicates that water flowing from on-site wastewater treatment facilities (e.g., septic systems) and stormwater ponds may reach the May River system quicker than other areas. Therefore, on-site wastewater treatment facilities and stormwater ponds should be appropriately engineered to limit this effect, particularly in the Stony and Rose Dhu creek areas. Strict best management practices (BMPs) should be used including minimizing the use of septic systems, maximizing naturally vegetated buffers, and the latest technologies available for stormwater ponds and septic systems.

- Determining the ground water contribution to the system may be important if alterations are observed which cannot be explained by surface flow. Some of the techniques to assess this are isotope tracers (cesium) and piezometers.

- Seasonal fecal coliform sampling of the May River system should be a major component of any sampling plan. SCDHEC currently monitors 8 sites in the larger May River system, which should provide an adequate assessment of the larger water components. Additional sampling in the headwater systems should be included to better target upland sources. In addition, source tracking either by wastewater indicator analysis or MAR should be included to properly assess the source of the bacteria observed. Another useful indicator for surface runoff is the monoclonal antibody kit for atrazine (a commonly used herbicide).

- Large developments should be required or encouraged to monitor their stormwater pond efficiencies and the receiving tidal creeks for a variety of parameters such as nitrogen, phosphorus, fecal coliforms, dissolved oxygen, and salinity. New technologies and sensors are available for measuring some of these parameters (e.g., optical probes to measure chlorophyll and nitrate) as well as SCDHEC approved measures.

- Future studies of water quality, sediment quality, and biotic condition should concentrate on sampling tidal creeks since they represent a direct connection with the upland environment. The headwater and large tidal creek sites studied in this baseline assessment should be resampled at appropriate intervals to identify whether conditions degrade as development of the basin proceeds. Sampling in additional tidal creeks may also be warranted. A lower priority should be placed on sampling the larger open water sites, with the possible exception of including one or two stations in the upper or middle portion of the system near the primary residential development on the river. The SCDHEC station routinely sampled for ambient surface water quality should provide representative data for the lower portion of the drainage system. Additional open water sites should only be sampled if significant degradation is detected in the larger creek sites, or where no sites exist, in the shallow water habitats near the mouth of the smaller creeks.

- Water quality monitoring should be conducted more frequently than sediment and biotic condition sampling. The frequency of water quality monitoring should be dependent on the completion of planned developments, and in conjunction with monitoring conducted for the retention pond effluents.

- The water quality parameters that appear to be the most important for consideration in monitoring based on the results of this study and other studies include DO, salinity, turbidity, chlorophyll-*a* (with HAB typing if problems occur), pH, nutrients, fecal coliforms (with typing if problems occur), and potentially total organic carbon and/or dissolved organic carbon.

- Water quality parameters of a lower priority include BOD, groundwater tracing, and nutrient particulates and speciation.

- It is recommended that sediment and biotic condition assessments be conducted at five-year increments, or less frequently if there have not been major changes in land use patterns.

    - Sediment quality sampling should not include sediment toxicity assays unless ERMQ levels are detected at concentrations significantly above those observed in this study.
    - Biotic condition measures are important to incorporate into any future study, as they provide a measure of whether degradation in water or sediment quality results in a biotic response. Biological sampling should place priority on assessing benthic community condition, re-evaluating some of the oyster populations sampled in this study and related condition measures (e.g., recruitment and juvenile growth), analysis of oyster tissue contaminants in the upper portion of the estuary as an additional sentinel measure, and disease monitoring if the local commercial oyster harvesters begin detecting a problem with apparent die-off in the shell beds. The local shellfish permit holders have a vested interest in maintaining good water quality and should be enlisted to work with the Town to make observations and report unusual oyster dieoffs or decreased bed quality.
    - Sampling of the nektonic community can also provide useful information if conducted using a comparative station approach.

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VII. APPENDICES

Appendix III-1.

Summary of station information for sites sampled in the May River estuary during 2002-2003.

Appendix III-1a. Summary of the zone, abbreviation, GPS coordinates, and length of the headwater tidal creeks sampled during the summer of 2002.

Zone	Location	Abbreviation	Latitude	Longitude	Length (m)
Headwater Creeks					
Upper	Stony Creek	ST	32.23553	80.94109	600
Upper	Rose Dhu Creek	RD	32.23513	80.92524	600
Upper	Palmetto Bluff	PB	32.21653	80.89303	600
Middle	Heyward Cove	HC	32.23256	80.86030	350
Middle	Brighton Beach	BB	32.21669	80.84550	350
Lower	Bass Creek	BC	32.23300	80.80533	600
Large Tidal Creeks					
Upper	May River Headwater	U-01	32.22426	80.92580	NA
Lower	Bass Creek	L-03	32.21136	80.81494	NA
Lower	Bass Creek	L-04	32.22287	80.80774	NA
					NA
Open Water					NA
Upper		U-02	32.22358	80.90115	NA
Upper		U-03	32.22715	80.90360	NA
Middle		M-01	32.22315	80.85990	NA
Middle		M-02	32.20773	80.86840	NA
Middle		M-03	32.18400	80.88271	NA
Lower		L-01	32.20527	80.82195	NA
Lower		L-02	32.20509	80.80098	NA

Appendix III-1b. Cross reference of sites for this report and USGS NWIS database.

USGS STATION NUMBER	TYPE OF SITE	USGS STATION NAME	SITE CODE	SITE NAME
02176704	HEADWATER	Stony Creek at Pritchardville	U-10	Stony
02176706	HEADWATER	Rose Dhu Creek at Pritchardville	U-11	Rose Dhu
02176713	HEADWATER	Palmetto Bluff Creek nr Palmetto Bluff	U-12	Palmetto Bluff
02176723	HEADWATER	Unnamed Creek near Heyward Cove	M-10	Heyward Cove
02176742	HEADWATER	Bass Creek nr Goat Island	L-10	Bass
02176734	HEADWATER	Unnamed creek to May River at Brighton Beach	M-11	Brighton Beach
02176715	OPEN WATER	May River below Pritchardville	U-02	
02176718	OPEN WATER	May River above Bluffton	U-03	
02176725	OPEN WATER	May River below Bluffton	M-01	
02176730	OPEN WATER	May River above Brighton Beach	M-02	
02176732	OPEN WATER	May River near Hoophole Island	M-03	
02176740	OPEN WATER	May River near Racoon Island	L-01	
02176745	OPEN WATER	May River near Baratana Island	L-02	
02176744	LARGE TIDAL	May River below Jess Island	L-03	
02176743	LARGE TIDAL	Bass Creek below Goat Island	L-04	
02176708	LARGE TIDAL	May River at Pritchardville	U-01	

Appendix III-2.

Description of the analytical methods and parameters determined by the U. S. Geological Survey Ocala Water Quality and Research Laboratory (OWQRL) and National Water Quality Laboratory (NWQL). [SM, standard methods for the determination of water and wastewater; EPA, Environmental Protection Agency; I-XXXX-XX, USGS-approved methods; NTU, nephelometric turbidity units; mg/L milligrams per liter; mS/cm, microsiemens per centimeter; $CaCO_3$, calcium carbonate; as N, concentration reported as nitrogen; as P, concentration reported as phosphorus].

Appendix III-2. Description of the analytical methods and parameters determined by the U. S. Geological Survey Ocala Water Quality and Research Laboratory (OWQRL) and National Water Quality Laboratory (NWQL). [SM, standard methods for the determination of water and wastewater; EPA, Environmental Protection Agency; I-XXXX-XX, USGS-approved methods; NTU, nephelometric turbidity units; mg/L milligrams per liter; mS/cm, microsiemens per centimeter; $CaCO_3$, calcium carbonate; as N, concentration reported as nitrogen; as P, concentration reported as phosphorus].

Lab Code	*USGS Parameter Code*	*Parameter Name*	*Method*	*Method Description*	*Laboratory Report Level*	*Unit*	*Analytical Laboratory*	*Abbreviated Name*
68	00403	PH	I-1586-85	Electrometry, glass-electrode	0.1	pH units	OWQRL	pH
70	90410	ALKALINITY	I-2030-85	Electrometry, titration	1	mg/L as $CaCO_3$	OWQRL	Alk
69	90095	SPECIFIC CONDUCTANCE AT 25 DEGREES CELSIUS	I-1780-85	Electrometry, Wheastone bridge	1	µS/cm	OWQRL	SC
292	00940	CHLORIDE, DISSOLVED	I-2057-85	Ion Chromatograph	0.1	mg/L	OWQRL	Cl
169	00530	TOTAL SOLIDS (AS TOTAL RESIDUE AT 105 DEGREES CELSIUS)	I-3765-85	Gravimetry	1	mg/L	OWQRL	TS
49	00535	TOTAL VOLATILE SOLIDS (AS VOLATILIE RESIDUE)	I-3767-85	Gravimetry	1	mg/L	OWQRL	TVS
50	00076	TURBIDITY	I-3860-85	Nephelometry	0.05	NTU	OWQRL	Turbidity
1977	00613	NITRITE, DISSOLVED LOW-LEVEL (AS N)	I-2540-85 [EPA 353.2]	Colorimetry, diazotization	0.001	mg/L	OWQRL	NO_2
1978	00671	ORTHO-PHOSPHATE, DISSOLVED LOW-LEVEL (AS P)	I-2601-85 [EPA 365.1]	Colorimetry, phosphomolybdate	0.001	mg/L	OWQRL	PO_4
1979	00631	NITRATE PLUS NITRITE, DISSOLVED LOW-LEVEL (AS N)	I-2545-85 [EPA 353.2]	Colorimetry, cadmium reduction-diazotization	0.002	mg/L	OWQRL	NOx
1980	00608	AMMONIA, DISSOLVED LOW-LEVEL (AS N)	I-2522-85 [EPA 350.1]	Colorimetry, salicylate-hypochlorite	0.002	mg/L	OWQRL	NH3
1981	00666	PHOSPHORUS, DISSOLVED LOW-LEVEL (AS P)	I-2600-85 [EPA 365.1]	Colorimetry, phosphomolybdate	0.002	mg/L	OWQRL	NOX
1982	00665	PHOSPHORUS, TOTAL LOW-LEVEL (AS P)	I-4600-85 [EPA 365.1]	Colorimetry, phosphomolybdate	0.002	mg/L	OWQRL	TP
1985	00623	AMMONIA PLUS ORGANIC NITROGEN, DISSOLVED (AS N)	[EPA 365.1]	Colorimetry, block digester, salicylate-hypochlorite	0.2	mg/L	OWQRL	DKN
1986	00625	AMMONIA PLUS ORGANIC NITROGEN, TOTAL (AS N)	I-4552-85 [EPA 351.2]	Colorimetry, block digester, salicylate-hypochlorite	0.2	mg/L	OWQRL	TKN
3078	00955	SILICA, DISSOLVED	EPA 200.7	Inductively coupled plasma	0.01	mg/L	OWQRL	Si
3094	00310	5-DAY BIOCHEMICAL OXYGEN DEMAND AT 20 DEGREES CELSIUS	SM 507	Dissolved-Oxygen Probe/Incubation	0.1	mg/L	OWQRL	BOD
3273	00681	DISSOLVED ORGANIC CARBON	SM 5310B	Combustion-Infrared	0.1	mg/L	OWQRL	DOC
114	00680	TOTAL ORGANIC CARBON	SM 5310B	Combustion-Infrared	0.1	mg/L	OWQRL	TOC
19	00685	TOTAL INORGANIC CARBON	SM-5310B	Combustion-Infrared	0.1			TIC
306	00691	DISSOLVED INORGANIC CARBON	SM 5310B	Combustion-Infrared	0.1	mg/L	OWQRL	DIC
2606	00694	TOTAL (INORGANIC PLUS ORGANIC) PARTICULATE CARBON	EPA 440.0	Combustion-thermal conductivty detector	0.12	mg/L	NWQL	TPC
2607	49570	TOTAL PARTICULATE NITROGEN	EPA 440.0	Combustion-thermal conductivty detector	0.022	mg/L	NWQL	TPN
2608	00688	PARTICULATE INORGANIC CARBON	EPA 440.0	Combustion-thermal conductivty detector	0.12	mg/L	NWQL	PIC
2611	00689	PARTICULATE ORGANIC CARBON	EPA 440.0	Combustion-thermal conductivty detector	0.12	mg/L	NWQL	POC

Appendix III-3.

Parameter description and detection limits of the U.S. Geological Survey National Water Quality Laboratory analysis for human wastewater indicators in unfiltered water (Zaugg and others, 2002).

Appendix III-3. Parameter description and detection limits of the U.S. Geological Survey National Water Quality Laboratory analysis for human wastewater indicators in unfiltered water (Zaugg and others, 2002).

Parameter	**Group**	**Description**	**Detection Limit**
4-cumylphenol	Detergent Agent	Nonionic detergent metabolite	< 1.000
4-n-octylphenol	Detergent Agent	Nonionic detergent metabolite	< 1.000
4-tert-octylphenol	Detergent Agent	Nonionic detergent metabolite	< 1.000
diethyl phthalate	Detergent Agent	Flame retardant	< 0.500
diethylhexyl phthalate	Detergent Agent	Plasticizer	< 0.500
ethanol,2-butoxy-,phosphate	Detergent Agent	Flame retardant	< 0.500
Nonylphenol NPEO1-total	Detergent Agent	Nonionic detergent metabolite	< 5.000
Nonlyphenol NPEO2-total	Detergent Agent	Nonionic detergent metabolite	< 5.000
Nonylphenol degradate OPEO1	Detergent Agent	Nonionic detergent metabolite	< 1.000
Nonylphenol degradate OPEO2	Detergent Agent	Nonionic detergent metabolite	< 1.000
para-nonylphenol-total	Detergent Agent	Nonionic detergent metabolite	< 5.000
tri(2-chloroethyl)phosphate	Detergent Agent	Flame retardant	< 0.500
tri(dichlorisopropyl)phosphate	Detergent Agent	Flame retardant	< 0.500
tributylphosphate	Detergent Agent	Flame retardant; antifoam agent	< 0.500
triclosan	Detergent Agent	Disinfectant, antimicrobial (concern for acquired microbial resistance)	< 1.000
triphenyl phosphate	Detergent Agent	Flame retardant	< 0.500
benzophenone	Fragrance/Additive	Fixative for soaps and perfumes	< 0.500
acetophenone	Fragrance/Additive	Fragrance in detergent and tobacco, flavor in beverages	< 0.500
BHA	Fragrance/Additive	Antioxidant, general preservative	< 5.000
camphor	Fragrance/Additive	Flavor, odorant, ointments	< 0.500
d-limonene	Fragrance/Additive	Fumigant, antimicrobial, antiviral, fragrance in aerosols	< 0.500
ethyl citrate	Fragrance/Additive	Cosmetics, pharmaceuticals	< 0.500
galaxolide (HHCB)	Fragrance/Additive	Musk fragrance (widespread usage) persistent in ground water	< 0.500
isoborneol	Fragrance/Additive	Fragrance in perfumery, in disinfectants	< 0.500
isoquinoline	Fragrance/Additive	Flavors and fragrances	< 0.500
menthol	Fragrance/Additive	Cigarettes, cough drops, liniment, mouthwash	< 0.500
skatol	Fragrance/Additive	Fragrance, stench in feces and coal tar	< 1.000
tonalide (AHTN)	Fragrance/Additive	Musk fragrance (widespread usage) persistent in ground water	< 0.500
methyl salicylate	Fragrance/Additive	Liniment, food, beverage, UV-absorbing lotion	< 0.500
17-alpha-ethynyl esterdiol	Pharmaceutical/Food By-product	Oral contraceptive	< 5.000
17B-estradiol	Pharmaceutical/Food By-product	Oral contraceptive	< 5.000
3-beta-coprostanol	Pharmaceutical/Food By-product	Carnivore fecal indicator	< 2.000
beta-sitosterol	Pharmaceutical/Food By-product	Plant sterol	< 2.000
caffeine	Pharmaceutical/Food By-product	Beverages, diuretic, very mobile/biodegradable	< 0.500

Appendix III-3. (Continued**)**

Parameter	Group	Description	Detection Limit
cholesterol	Pharmaceutical/Food By-product	Often a fecal indicator, also a plant sterol	< 2.000
cotinine	Pharmaceutical/Food By-product	Primary nicotine metabolite	< 0.500
equilenin	Pharmaceutical/Food By-product	Hormone replacement therapy drug	< 5.000
estrone	Pharmaceutical/Food By-product	Biogenic hormone	< 5.000
stigmastanol	Pharmaceutical/Food By-product	Plant sterol	< 2.000
carbazole	Pesticide	Insecticide, Manuf. dyes, explosives, and lubricants; Coal tar and cigarette smoke	< 0.500
3,4-dichlorophenyl isocyanate	Pesticide	Herbicide degradate - diuron (aquatic veg)	< 0.500
atrazine	Pesticide	Herbicide; widely used on broadleaf and grassy leaf plants	< 0.500
bromacil	Pesticide	Herbicide, >80% noncrop usage on grass/brush	< 0.500
carbaryl	Pesticide	Insecticide, crop and garden uses, low persistence	< 1.000
chlorpyrifos	Pesticide	Insecticide, domestic pest and termite control (domestic use restricted as of 2001)	< 0.500
diazinon	Pesticide	Insecticide, > 40% nonagricultural usage, ants, flies	< 0.500
dichlorvos	Pesticide	Insecticides, pet collars, flies, also a degradate of naled or trichlofon	< 1.000
indole	Pesticide	Pesticide inert ingredient, fragrance in coffee	< 0.500
metalaxyl	Pesticide	Herbicide, Fumigant, mildew, blight, pathogens, golf/turf	< 0.500
metolachlor	Pesticide	Herbicide, indicator of agricultural drainage	< 0.500
N,N-diethyltoluamide (DEET)	Pesticide	Insecticides, urban uses, mosquito repellent	< 0.500
prometon	Pesticide	Herbicide (noncrop only), applied prior to blacktop	< 0.500
isophorone	Urban Runoff	Solvent for lacquer, plastic, oil, silicon, resin	< 0.500
1,4-dichlorobenzene	Urban Runoff	Volatile organic compound	< 0.500
1-methylnapthalene	Urban Runoff	Polycyclic aromatic hydrocarbon; -5% of gasoline, diesel fuel, or crude oil	< 0.500
2,6-dimethylnapthalene	Urban Runoff	Polycyclic aromatic hydrocarbon; Present in diesel/kerosene (trace in gasoline)	< 0.500
2-methylnapthalene	Urban Runoff	Polycyclic aromatic hydrocarbon:2-5% of gasoline, diesel fuel, or crude oil	< 0.500
5-methyl-1H-benzotriazle	Urban Runoff	Antioxidant in antifreeze and deicers	< 2.000
Anthracene	Urban Runoff	Polycyclic aromatic hydrocarbon; Wood preservative, component of tar, diesel, or crude oil, CP	< 0.500
anthraquinone	Urban Runoff	Manuf. dye/textiles, seed treatment, bird repellant	< 0.500
benzo(a)pyrene	Urban Runoff	Polycyclic aromatic hydrocarbon: Regulated PAH, used in cancer research, CP	< 0.500
bisphenol A	Urban Runoff	Manufactured polycarbonate resins, antioxidant, FR	< 1.000
bromoform	Urban Runoff	Trihalomethane; disinfection by-product	< 0.500
cumene	Urban Runoff	Manuf. phenol/acetone, fuels and paint thinner	< 0.500
fluoranthene	Urban Runoff	Polycyclic aromatic hydrocarbon; Component of coal tar and asphalt (only traces in gasoline or diesel fuel), CP	< 0.500

Appendix III-3. (Continued)

Parameter	Group	Description	Detection Limit
naphthalene	Urban Runoff	Polycyclic aromatic hydrocarbon; Fumigant, moth repellent, major component (about 10%) of gasoline	< 0.500
para-cresol	Urban Runoff	Wood preservative	< 1.000
PBDE4-1	Urban Runoff	Polybrominated diphenyl ethers	< 10.000
PBDE4-2	Urban Runoff	Polybrominated diphenyl ethers	< 10.000
PBDE4-3	Urban Runoff	Polybrominated diphenyl ethers	< 10.000
PBDE5-1	Urban Runoff	Polybrominated diphenyl ethers	< 10.000
PBDE5-2	Urban Runoff	Polybrominated diphenyl ethers	< 10.000
PBDE5-3	Urban Runoff	Polybrominated diphenyl ethers	< 10.000
PBDE6-1	Urban Runoff	Polybrominated diphenyl ethers	< 10.000
PBDE6-2	Urban Runoff	Polybrominated diphenyl ethers	< 10.000
pentachlorophenol	Urban Runoff	H, F, wood preservative, termite control	< 2.000
phenanthrene	Urban Runoff	Polycyclic aromatic hydrocarbon; Manuf. explosives, component of tar, diesel fuel, or crude oil, CP	< 0.500
phenol	Urban Runoff	Disinfectant, manuf. several products, leachate	< 0.500
pyrene	Urban Runoff	Polycyclic aromatic hydrocarbon; Component of coal tar and asphalt (only traces in gasoline or diesel fuel), CP	< 0.500
tetrachloroethylene	Urban Runoff	Chlorinate solvent	< 0.500

Appendix III-4.

Summary of contaminants measured in sediments of the May River estuary, with information on analytical method and detection limits.

Appendix III-4. Summary of contaminants measured in sediments of the May River estuary, with information on analytical method and detection limits.

Chemical Class	Analyte	CAS Number	Typical Detection Limit	AA	ICP	Hg Cold Vapor	GC\ECD	GC\MS	HPLC\FD
Metal	Aluminum	7429-90-5	0.088 %		X				
Metal	Arsenic	7440-38-2	0.036 ug/g dry wt	X	X				
Metal	Cadmium	7440-43-9	0.035 ug/g dry wt	X	X				
Metal	Chromium	7440-47-3	0.03 ug/g dry wt	X	X				
Metal	Copper	7440-50-8	0.3 ug/g dry wt		X				
Metal	Iron	7439-89-6	0.0012 %		X				
Metal	Lead	7439-92-1	0.16 ug/g dry wt	X	X				
Metal	Manganese	7439-96-5	0.1 ug/g dry wt		X				
Metal	Mercury	7439-97-6	0.04 ug/g dry wt	X		X			
Metal	Nickel	7440-02-0	1.9 ug/g dry wt		X				
Metal	Selenium	7782-49-2	0.034 ug/g dry wt	X					
Metal	Silver	7440-22-4	0.02 ug/g dry wt	X					
Metal	Tin	7440-31-5	7.8 ug/g dry wt		X				
Metal	Zinc	7440-66-6	0.2 ug/g dry wt		X				
PAH	1-Methylnaphthalene	90-12-0	26.2 ng/g dry wt					X	
PAH	1-Methylphenanthrene	832-69-9	24.2 ng/g dry wt					X	
PAH	1,6,7 Trimethylnaphthalene	2245-38-7	12.2 ng/g dry wt					X	
PAH	2-Methylnaphthalene	91-57-6	36 ng/g dry wt					X	
PAH	2,6 Dimethylnaphthalene	581-42-0	24.4 ng/g dry wt					X	

Appendix III-4. (Continued)

Chemical Class	Analyte	CAS Number	Typical Detection Limit	AA	ICP	Hg Cold Vapor	GC\ECD	GC\MS	HPLC\FD
PAH	Acenaphthene	83-32-9	42.2 ng/g dry wt					X	
PAH	Acenaphthylene	208-96-8	11 ng/g dry wt					X	
PAH	Anthracene	120-12-7	22.6 ng/g dry wt					X	X
PAH	Benzo(a)anthracene	56-55-3	49.8 ng/g dry wt					X	X
PAH	Benzo(a)pyrene	50-32-8	63.2 ng/g dry wt					X	X
PAH	Benzo(b+j)fluoranthene	205-99-2 & ??	38.6 ng/g dry wt					X	
PAH	Benzo(e)pyrene	192-97-2	29.2 ng/g dry wt					X	X
PAH	Benzo(g,h,i)perylene	191-24-2	39.6 ng/g dry wt					X	X
PAH	Benzo(k)fluoranthene	207-08-9	33 ng/g dry wt					X	X
PAH	Biphenyl	92-52-4	41.2 ng/g dry wt					X	
PAH	Chrysene+Triphenylene	218-01-9 & ??	14.2 ng/g dry wt					X	
PAH	Dibenz(a,h+a,c)anthracene	53-70-3 & ??	10.6 ng/g dry wt					X	
PAH	Fluoranthene	206-44-0	27.8 ng/g dry wt					X	X
PAH	Fluorene	86-73-7	18.2 ng/g dry wt					X	X
PAH	Indeno(1,2,3-cd)pyrene	193-39-5	61.4 ng/g dry wt					X	X
PAH	Naphthalene	91-20-3	65.6 ng/g dry wt					X	
PAH	Perylene	198-55-0	36.8 ng/g dry wt					X	X
PAH	Phenanthrene	85-01-8	21.8 ng/g dry wt					X	X
PAH	Pyrene	129-00-0	20.4 ng/g dry wt					X	X
PCB	PCB 101	37680-73-2	1.3 ng/g dry wt				X		

Appendix III-4. (Continued)

Chemical Class	Analyte	CAS Number	Typical Detection Limit	AA	ICP	Hg Cold Vapor	GC\ECD	GC\MS	HPLC\FD
PCB	PCB 104	56558-16-8	1.3 ng/g dry wt				X		
PCB	PCB 105	32598-14-4	1.59 ng/g dry wt				X		
PCB	PCB 118	31508-00-6	0.87 ng/g dry wt				X		
PCB	PCB 126	57465-28-8	1.69 ng/g dry wt				X		
PCB	PCB 128	38380-07-3	0.91 ng/g dry wt				X		
PCB	PCB 138	35065-28-2	2.31 ng/g dry wt				X		
PCB	PCB 153	35065-27-1	1.33 ng/g dry wt				X		
PCB	PCB 154	60145-22-4	1.3 ng/g dry wt				X		
PCB	PCB 170	35065-30-6	2.04 ng/g dry wt				X		
PCB	PCB 18	37680-65-2	1.95 ng/g dry wt				X		
PCB	PCB 180	35065-29-3	1.39 ng/g dry wt				X		
PCB	PCB 187	52663-68-0	0.62 ng/g dry wt				X		
PCB	PCB 188	74487-85-7	1.3 ng/g dry wt				X		
PCB	PCB 195	52663-78-2	1.56 ng/g dry wt				X		
PCB	PCB 201	52663-75-9	1.3 ng/g dry wt				X		
PCB	PCB 206	40186-72-9	1.27 ng/g dry wt				X		
PCB	PCB 209	2051-24-3	1.3 ng/g dry wt				X		
PCB	PCB 28	7012-37-5	2.54 ng/g dry wt				X		
PCB	PCB 29	15862-07-4	1.3 ng/g dry wt				X		
PCB	PCB 44	41464-39-5	0.68 ng/g dry wt				X		

Appendix III-4. (Continued)

Chemical Class	Analyte	CAS Number	Typical Detection Limit	AA	ICP	Hg Cold Vapor	GC\ECD	GC\MS	HPLC\FD
PCB	PCB 50	62796-65-0	1.3 ng/g dry wt				X		
PCB	PCB 52	35693-99-3	0.88 ng/g dry wt				X		
PCB	PCB 66	32598-10-0	0.79 ng/g dry wt				X		
PCB	PCB 77	32598-13-3	19.5 ng/g dry wt				X		
PCB	PCB 8	34883-43-7	1.66 ng/g dry wt				X		
PCB	PCB 87	38380-02-8	1.3 ng/g dry wt				X		
Pesticide	2,4'-DDD	53-19-0	0.79 ng/g dry wt				X		
Pesticide	2,4'-DDE	3424-82-6	0.75 ng/g dry wt				X		
Pesticide	2,4'-DDT	789-02-6	1.87 ng/g dry wt				X		
Pesticide	4,4'-DDD	72-54-8	3.16 ng/g dry wt				X		
Pesticide	4,4'-DDE	72-55-9	0.43 ng/g dry wt				X		
Pesticide	4,4'-DDT	50-29-3	0.21 ng/g dry wt				X		
Pesticide	Aldrin	309-00-2	0.17 ng/g dry wt				X		
Pesticide	Chlorpyrifos	2921-88-2	1.3 ng/g dry wt				X		
Pesticide	Cis-chlordane (alpha-chlordane)	5103-71-9	1.07 ng/g dry wt				X		
Pesticide	Dieldrin	60-57-1	2.36 ng/g dry wt				X		
Pesticide	Endosulfan ether		1.3 ng/g dry wt				X		
Pesticide	Endosulfan I	959-98-8	1.3 ng/g dry wt				X		
Pesticide	Endosulfan II	33213-65-9	1.3 ng/g dry wt				X		
Pesticide	Endosulfan Lactone		1.3 ng/g dry wt				X		

Appendix III-4. (Continued)

Chemical Class	Analyte	CAS Number	Typical Detection Limit	AA	ICP	Hg Cold Vapor	GC\ECD	GC\MS	HPLC\FD
Pesticide	Endosulfan Sulfate	1031-07-8	1.3 ng/g dry wt				X		
Pesticide	Gamma-HCH (g-BHC, lindane)	58-89-9	0.99 ng/g dry wt				X		
Pesticide	Heptachlor	76-44-8	0.52 ng/g dry wt				X		
Pesticide	Heptachlor epoxide	1024-57-3	1.32 ng/g dry wt				X		
Pesticide	Hexachlorobenzene	118-74-1	0.81 ng/g dry wt				X		
Pesticide	Mirex	2385-85-5	2.03 ng/g dry wt				X		
Pesticide	Trans-nonachlor	39765-80-5	1.22 ng/g dry wt				X		

Appendix III-5.

Statistical comparisons of water quality parameters sampled in the May River estuary.

Appendix III-5a. Spearman Rho Correlation Coefficients on surface-water parameters from headwater tidal creek sites in May River 2002-2003. [Highlighted cells indicate significant correlations; Alpha level was set at 0.05, such that p-values < 0.05 indicated significant correlations with 95 % confidence; p-values of < 0.01 indicated significant correlations with 99 % confidence].

Italics = p-value < 0.05 **bold = p-value < 0.01**	Dissolved Oxygen	Dissolved Oxygen Saturation	Turbidity	pH	Salinity	Specific Conductance	Water Temperature	Alkalinity as ANC	Total Kjeldahl Nitrogen	Ammonia
Dissolved Oxygen		**0.88071**	-0.27548	-0.15908	-0.32392	-0.29494	**-0.53567**	**-0.57323**	-0.26519	*-0.44764*
Dissolved Oxygen Saturation			-0.24695	0.13918	-0.08111	-0.03820	-0.21562	-0.40261	-0.26257	*-0.41433*
Turbidity				0.37511	0.26308	0.24015	0.35358	**0.55260**	**0.81543**	**0.68119**
pH					*0.44813*	*0.43600*	**0.49660**	0.22772	0.26156	0.10515
Salinity						**0.99969**	**0.59490**	**0.80187**	0.13175	0.38518
Specific Conductance							**0.57231**	**0.76545**	0.15346	0.31737
Water Temperature								**0.64034**	0.40901	0.37484
Alkalinity as ANC									*0.43946*	**0.68140**
Total Kjeldahl Nitrogen										*0.47233*
Ammonia										
Total Organic Nitrogen										
Total Particulate Nitrogen										
Total Nitrogen										
Dissolved Nitrogen										
Nitrate plus Nitrite										
Dissolved Phosphorus										
Ortho-phosphate										
Total Phosphorus										
Total Particulate Carbon										
Total Organic Carbon										
Biochemical Oxygen Demand										
Fecal Coliform										
Silica										
Chloride										
Total Fixed Solids										
Total Suspended Solids										
Total Volatile Solids										

Appendix III-5a. (Continued)

bold = p-value < 0.01 *Italics = p-value < 0.05*	Total Organic Nitrogen	Total Particulate Nitrogen	Total Nitrogen	Dissolved Nitrogen	Nitrate plus Nitrite	Dissolved Phosphorus	Ortho-phosphate	Total Phosphorus	Total Particulate Carbon	Total Organic Carbon
Dissolved Oxygen	-0.16663	-0.18269	-0.25278	-0.29889	0.05320	*-0.48503*	*-0.41266*	-0.32188	-0.09972	0.27518
Dissolved Oxygen Saturation	-0.16687	-0.12614	-0.24428	-0.28690	0.28773	*-0.46171*	*-0.46639*	-0.29230	-0.06445	0.01022
Turbidity	**0.80574**	**0.87114**	**0.81935**	*0.46913*	0.16906	**0.59163**	**0.52362**	**0.82630**	**0.81216**	0.06488
pH	0.27911	0.38600	0.28149	-0.09454	0.18422	-0.16316	-0.11928	0.19125	0.34283	-0.22084
Salinity	0.11079	0.14064	0.12387	-0.08718	-0.07499	0.31452	0.26045	0.23327	-0.00848	**-0.67514**
Specific Conductance	0.13544	0.20048	0.14889	-0.07367	-0.06117	0.25480	0.23619	0.24310	0.07858	**-0.66116**
Water Temperature	0.31480	0.29485	*0.41576*	0.33718	0.37476	*0.42429*	*0.41192*	0.39052	0.24902	**-0.64158**
Alkalinity as ANC	0.36539	*0.49467*	*0.43815*	0.24815	0.16873	**0.67590**	**0.61423**	**0.68523**	0.36150	*-0.45692*
Total Kjeldahl Nitrogen	**0.93766**	**0.71382**	**0.99782**	**0.70441**	0.08562	**0.60836**	**0.60514**	**0.80914**	**0.69020**	0.09186
Ammonia	0.30975	*0.44091*	*0.47800*	0.28730	0.20252	**0.57393**	**0.55899**	**0.65550**	0.38126	-0.07640
Total Organic Nitrogen		**0.88708**	**0.93617**	*0.50942*	0.03488	**0.58902**	**0.60158**	**0.72992**	**0.38126**	-0.07640
Total Particulate Nitrogen			**0.72122**	0.34082	0.20473	*0.52584*	*0.49880*	**0.79130**	**0.94886**	0.09306
Total Nitrogen				**0.70419**	0.11911	**0.60921**	**0.60035**	**0.81523**	**0.70349**	0.09012
Dissolved Nitrogen					0.04420	**0.66893**	**0.57498**	**0.60057**	0.34642	0.31953
Nitrate plus Nitrite						-0.08990	-0.10682	0.27475	0.23934	-0.29188
Dissolved Phosphorus							**0.92234**	**0.77662**	*0.47568*	0.09432
Ortho-phosphate								**0.70885**	*0.45886*	0.14572
Total Phosphorus									**0.76039**	0.09437
Total Particulate Carbon										0.18067
Total Organic Carbon										
Biochemical Oxygen Demand										
Fecal Coliform										
Silica										
Chloride										
Total Fixed Solids										
Total Suspended Solids										
Total Volatile Solids										

Appendix III-5a. (Continued)

bold = p-value < 0.01 *Italics = p-value < 0.05*	**Biochemical Oxygen Demand**	**Fecal Coliform**	**Silica**	**Chloride**	**Total Fixed Solids**	**Total Suspended Solids**	**Total Volatile Solids**
Dissolved Oxygen	-0.38289	-0.15046	-0.04397	-0.34102	-0.24886	-0.22667	-0.14831
Dissolved Oxygen Saturation	-0.23226	*-0.39651*	-0.16848	-0.16224	-0.18969	-0.16620	-0.08419
Turbidity	**0.71983**	0.03747	0.27538	*0.47671*	**0.92989**	**0.91770**	**0.88256**
pH	0.29742	-0.21372	**-0.53439**	0.26565	*0.43840*	*0.44496*	0.37942
Salinity	0.33107	**-0.53428**	-0.20696	**0.77492**	*0.41638*	0.40565	*0.47081*
Specific Conductance	*0.43765*	**-0.57607**	-0.27291	**0.79669**	*0.43062*	*0.42845*	*0.49848*
Water Temperature	**0.51904**	-0.21941	0.04875	*0.44140*	*0.44019*	*0.43280*	*0.43306*
Alkalinity as ANC	**0.62552**	-0.25855	0.14239	0.76963	**0.61401**	**0.59225**	**0.63460**
Total Kjeldahl Nitrogen	**0.75343**	0.01629	*0.43781*	0.30511	**0.81149**	**0.81628**	**0.80402**
Ammonia	0.37290	0.19974	0.38379	0.39347	**0.59708**	**0.57183**	**0.58533**
Total Organic Nitrogen	**0.81004**	-0.01131	0.25526	0.27329	**0.81315**	**0.81611**	**0.79128**
Total Particulate Nitrogen	**0.82158**	0.02342	0.07833	0.40609	**0.83801**	**0.83018**	**0.76085**
Total Nitrogen	**0.74319**	0.02085	*0.43912*	0.29684	**0.80758**	**0.81193**	**0.80162**
Dissolved Nitrogen	*0.51211*	-0.02577	**0.65562**	0.18523	0.35680	0.35942	0.38505
Nitrate plus Nitrite	0.12951	0.04853	0.05677	0.10015	0.13186	0.12123	0.08510
Dissolved Phosphorus	**0.58564**	-0.11082	**0.60621**	0.37616	*0.50551*	*0.49082*	**0.55716**
Ortho-phosphate	**0.57410**	0.00098	0.51534	0.32050	*0.48412*	*0.47020*	*0.51047*
Total Phosphorus	**0.77241**	-0.04229	*0.45953*	*0.46609*	**0.79800**	**0.78930**	**0.79181**
Total Particulate Carbon	**0.77652**	0.10218	0.13635	0.26681	**0.75664**	**0.75229**	**0.69664**
Total Organic Carbon	-0.05876	0.38952	0.34472	*-0.44618*	-0.15376	-0.15376	-0.17444
Biochemical Oxygen Demand		-0.10130	0.11930	**0.56353**	**0.77887**	**0.77780**	**0.74144**
Fecal Coliform			-0.03583	**-0.56669**	-0.10345	-0.12817	-0.30518
Silica				-0.06179	0.10229	0.09358	0.23042
Chloride					**0.58447**	**0.57752**	**0.61260**
Total Fixed Solids						**0.99826**	**0.95639**
Total Suspended Solids							**0.96075**
Total Volatile Solids							

Appendix III-5b. Summary of the results of the multiple comparison tests to identify differences among sites, land use, and seasons in the headwater tidal creek sites in the May River watershed, South Carolina, 2002 - 2003. Statistical tests included Analysis of Variance test on ranked data (Kruskal-Wallis) and the Tukey (equal sample sizes) or Scheffe (unequal sample sizes) tests.

Parameter	STATISTICAL COMPARISON: KRUSKAL-WALLIS ONE-WAY ANALYSIS OF VARIANCE OF HEADWATER TIDAL CREEKS								
	Sites in the May River Watershed			Land-Use Classes [Suburban and Forested]			Seasons		
	Degrees of Freedom	Chi Squared	p-value	Degrees of Freedom	Chi Squared	p- value	Degrees of Freedom	Chi Squared	p-value
Water Temperature	4	0.3692	0.9849	1	0.1235	0.7253	3	18.0538	0.0004
Dissolved Oxygen	5	4.2545	0.5134	1	0.3291	0.5662	3	**11.413**	**0.0097**
Dissolved Oxygen in Percent Saturation	5	5.5657	0.3508	1	1.2646	0.2608	3	**8.207**	**0.0419**
Biochemical Oxygen Demand	5	3.8843	0.5662	1	0.3043	0.5812	3	**8.5432**	**0.0360**
Turbidity	5	9.5399	0.0894	1	**4.2194**	**0.0400**	3	3.0379	0.3858
Salinity	5	6.2487	0.2828	1	0.4824	0.4873	3	**18.1018**	**0.0004**
pH	5	3.8684	0.5685	1	0.0531	0.8178	3	4.7635	0.1900
Silica	5	**13.0828**	**0.0226**	1	3.6100	0.0574	3	3.5862	0.3097
Dissolved Phosphorus	5	**14.3287**	**0.0137**	1	3.0473	0.0809	3	3.9684	0.2649
Total Phosphorus	5	6.7400	0.2407	1	1.0838	0.2979	3	3.7533	0.2894
Total Nitrogen	5	8.6863	0.1223	1	**5.8535**	**0.0155**	3	3.9817	0.2634
Dissolved Nitrogen	5	9.7165	0.0837	1	**4.6158**	**0.0317**	3	5.6051	0.1325
Total Kjeldahl Nitrogen	5	9.3077	0.0974	1	**6.3175**	**0.0120**	3	3.7749	0.2868
Nitrate plus Nitrite	5	8.8613	0.1147	1	0.7100	0.3994	3	3.8039	0.2834
Ammonia	5	6.3162	0.2767	1	0.3420	0.9029	3	4.732	0.1925
Total Organic Carbon	5	4.6845	0.4556	1	1.2155	0.2702	3	**12.3854**	**0.0062**
Dissolved Organic Carbon	5	4.9093	0.4271	1	0.8445	0.3581	3	**12.2356**	**0.0066**
Total Suspended Solids	5	6.3002	0.2781	1	2.3448	0.1257	3	4.4136	0.2201
Fecal Coliform	5	4.3168	0.5048	1	1.8193	0.1774	3	**7.8402**	**0.0494**

Appendix III-5c. Spearman Rho Correlation Coefficients for selected surface-water parameters from open-water and large tidal creek sites in May River, 2002-2003. [Highlighted cells indicate significant correlations; Alpha level was set at 0.05, such that p-values < 0.05 indicated significant correlations with 95 % confidence; p-values of < 0.01 indicated significant correlations with 99 % confidence].

bold = p-value < 0.01 *Italics = p-value < 0.05*	Dissolved Oxygen	Dissolved Oxygen Saturation	Turbidity	pH	Salinity	Specific Conductance	Water Temperature	Alkalinity as ANC	Total Kjeldahl Nitrogen	Ammonia
Dissolved Oxygen		**0.83643**	**-0.43995**	**0.61795**	-0.10369	-0.14499	**-0.67143**	*-0.31512*	**-0.62332**	**-0.55435**
Dissolved Oxygen Saturation			-0.26409	**0.61786**	0.22066	0.19740	-0.30726	0.01503	**-0.61082**	**-0.73694**
Turbidity				**-0.47696**	-0.10470	-0.11289	0.39834	0.06181	**0.56942**	*0.39817*
pH					-0.01771	0.05152	**-0.42275**	-0.05990	**-0.70917**	**-0.69680**
Salinity						**0.99863**	0.34830	**0.84020**	-0.24017	*-0.38735*
Specific Conductance							*0.37768*	**0.84551**	-0.27463	**-0.40817**
Water Temperature								**0.40622**	*0.36173*	0.17827
Alkalinity as ANC									-0.08443	-0.22408
Total Kjeldahl Nitrogen										**0.75684**
Ammonia										
Total Organic Nitrogen										
Total Particulate Nitrogen										
Total Nitrogen										
Nitrate plus Nitrite										
Dissolved Nitrogen										
Dissolved Phosphorus										
Ortho-phosphate										
Total Phosphorus										
Total Particulate Carbon										
Total Organic Carbon										
Biochemical Oxygen Demand										
Silica										
Chloride										
Total Fixed Solids										
Total Suspended Solids										
Total Volatile Solids										

Appendix III-5c. (Continued)

bold = p-value < 0.01 *Italics = p-value < 0.05*	Total Organic Nitrogen	Total Particulate Nitrogen	Total Nitrogen	Nitrate plus Nitrite	Dissolved Nitrogen	Dissolved Phosphorus	Ortho-phosphate	Total Phosphorus	Total Particulate Carbon	Total Organic Carbon
Dissolved Oxygen	**-0.61363**	**-0.50534**	**-0.65077**	**-0.48486**	**-0.56832**	**-0.57385**	**-0.76989**	**-0.62495**	**-0.53460**	-0.27251
Dissolved Oxygen Saturation	**-0.58069**	-0.28641	**-0.63502**	*-0.38260*	**-0.61999**	**-0.47692**	**-0.69480**	**-0.53211**	-0.23291	-0.29246
Turbidity	**0.53902**	**0.56436**	**0.57531**	0.15918	*0.32297*	0.30873	0.18722	**0.56428**	**0.63086**	0.25136
pH	**-0.68223**	**-0.63477**	**-0.69941**	-0.15001	**-0.60053**	-0.26357	*-0.38870*	**-0.47176**	**-0.49447**	**-0.52484**
Salinity	-0.23770	0.09319	-0.25345	-0.11962	-0.27708	-0.13446	0.05715	-0.17886	0.15082	-0.23938
Specific Conductance	-0.26966	0.03438	-0.28237	-0.06500	-0.30828	-0.07090	0.13195	-0.14547	0.10645	-0.29440
Water Temperature	*0.33811*	**0.51469**	*0.35779*	0.18478	0.27522	**0.52838**	**0.55447**	**0.49660**	**0.63463**	0.19799
Alkalinity as ANC	-0.07156	0.18864	-0.07045	0.07936	-0.16074	0.04658	0.29171	-0.02006	0.21848	*-0.38038*
Total Kjeldahl Nitrogen	**0.97909**	**0.58818**	**0.98557**	0.26857	**0.70186**	**0.49636**	**0.46700**	**0.74015**	**0.59951**	**0.57065**
Ammonia	**0.66615**	*0.32803*	**0.78929**	*0.35004*	**0.74195**	0.28944	**0.49024**	**0.49082**	0.29041	**0.51992**
Total Organic Nitrogen		**0.66352**	**0.94050**	0.07905	**0.59966**	**0.47590**	*0.34965*	**0.72667**	**0.61400**	**0.56432**
Total Particulate Nitrogen			**0.56330**	-0.04210	*0.34326*	0.28004	0.17005	**0.49030**	**0.87274**	**0.43329**
Total Nitrogen				**0.40664**	**0.76930**	**0.48374**	**0.52029**	**0.72609**	**0.59554**	**0.54378**
Nitrate plus Nitrite					**0.57979**	0.18800	**0.55910**	0.21005	0.11343	-0.00305
Dissolved Nitrogen						*0.36026*	**0.51049**	**0.50246**	*0.38074*	**0.54691**
Dissolved Phosphorus							**0.64249**	**0.84764**	**0.41552**	0.23306
Ortho-phosphate								**0.58158**	0.24447	0.19039
Total Phosphorus									0.60674	*0.39064*
Total Particulate Carbon										0.31641
Total Organic Carbon										
Biochemical Oxygen Demand										
Silica										
Chloride										
Total Fixed Solids										
Total Suspended Solids										
Total Volatile Solids										

Appendix III-5c. (Continued)

bold = p-value < 0.01 *Italics = p-value < 0.05*	**Biochemical Oxygen Demand**	**Silica**	**Chloride**	**Total Fixed Solids**	**Total Suspended Solids**	**Total Volatile Solids**
Dissolved Oxygen	**-0.54486**	**-0.58145**	-0.28752	*-0.34022*	**-0.51308**	**-0.54738**
Dissolved Oxygen Saturation	-0.29985	**-0.62451**	0.04016	-0.11453	-0.22111	-0.23565
Turbidity	**0.63229**	**0.40712**	-0.02505	**0.49759**	**0.57504**	**0.57554**
pH	**-0.42627**	**-0.70347**	-0.05396	-0.15393	-0.36313	**-0.41464**
Salinity	-0.01175	-0.22078	**0.91371**	0.24719	0.28107	*0.36172*
Specific Conductance	0.00377	-0.26651	**0.91599**	0.27354	0.28907	*0.37050*
Water Temperature	**0.54147**	**0.41003**	**0.46418**	**0.42218**	**0.57889**	**0.59351**
Alkalinity as ANC	0.09342	*-0.07749*	**0.96267**	0.32603	*0.38629*	**0.46627**
Total Kjeldahl Nitrogen	**0.52709**	**0.72204**	-0.14005	*0.41176*	0.01550	**0.48719**
Ammonia	0.25316	**0.71745**	-0.27842	0.06605	0.18017	0.17758
Total Organic Nitrogen	**0.57658**	**0.70316**	-0.12084	*0.41680*	**0.56660**	**0.54314**
Total Particulate Nitrogen	**0.68856**	**0.66026**	0.16585	**0.54527**	**0.65492**	**0.65438**
Total Nitrogen	**0.51354**	**0.71853**	-0.14188	*0.38717*	**0.48346**	**0.46462**
Nitrate plus Nitrite	0.05701	0.13149	-0.01506	-0.10137	-0.01650	-0.00282
Dissolved Nitrogen	0.27712	**0.72280**	-0.21453	0.01666	0.16940	0.16629
Dissolved Phosphorus	**0.42693**	**0.43027**	0.06075	0.20168	*0.34173*	*0.33480*
Ortho-phosphate	0.28045	**0.46963**	0.27820	0.22046	0.30210	0.29200
Total Phosphorus	**0.58664**	**0.55150**	-0.05856	**0.46399**	**0.52454**	**0.48501**
Total Particulate Carbon	**0.75036**	**0.55463**	0.21029	**0.63753**	**0.71184**	**0.69756**
Total Organic Carbon	0.28878	**0.65411**	*-0.32382*	0.08321	0.26003	0.23836
Biochemical Oxygen Demand		**0.50463**	0.06221	**0.42677**	**0.56345**	**0.56887**
Silica			-0.09764	0.07817	0.30740	0.29605
Chloride				0.29570	*0.36958*	**0.44673**
Total Fixed Solids					**0.98989**	**0.88955**
Total Suspended Solids						**0.94449**
Total Volatile Solids						

Appendix III-5d. Summary of the results of the multiple comparison test to identify differences among sites, habitat types, and seasons in the large tidal creek and open-water sites in the May River watershed, 2002 - 2003, and between summer samples in the May River and SCECAP sites that included Analysis of Variance test on ranked data (Kruskal-Wallis) and the Tukey (equal sample sizes) or Scheffe (unequal sample sizes) tests. [Tukey Studentized Range (HSD) test].

	STATISTICAL COMPARISON: KRUSKAL-WALLIS ONE-WAY ANALYSIS OF VARIANCE OF HEADWATER TIDAL CREEKS											
	SCECAP and May River Sites			Habitat Types [Open Water and Large Tidal Creeks]			Sites in May River Watershed			Seasons		
Parameter	Degrees of Freedom	Chi Squared	P-value	Degrees of Freedom	Chi Squared	P- value	Degrees of Freedom	Chi Squared	P- value	Degrees of Freedom	Chi Squared	P-value
Water Temperature	1	3.5329	0.0602	1	0.1835	0.6684	9	5.4824	0.7904	3	**28.4445**	**<0.0001**
Dissolved Oxygen	1	**6.8267**	**0.009**	1	3.4596	0.0629	9	8.7442	0.4612	3	**18.3373**	**0.0004**
Dissolved Oxygen in Percent Saturation				1	**6.6736**	**0.0098**	9	14.0662	0.12	3	6.5469	0.0878
Biochemical Oxygen Demand	1	**15.1626**	**< 0.0001**	1	**4.1873**	**0.0407**	9	10.0141	0.3493	3	**13.5303**	**0.0036**
Turbidity	1	0.8824	0.3475	1	0.6833	0.4084	9	5.7389	0.7657	3	7.7799	0.0508
Salinity	1	**13.5475**	**0.0002**	1	0.2839	0.5942	9	3.74493	0.9271		**22.2105**	**<0.0001**
pH	1	0.9608	0.327	1	0.2382	0.6255	9	**19.9503**	**0.0358**	3	3.3404	0.3421
Dissolved Phosphorus				1	2.4054	0.1209	9	9.98801	0.3603	3	6.0146	0.1109
Total Phosphorus	1	**6.0105**	**0.0142**	1	2.1362	0.1439	9	10.9016	0.2825	3	6.3152	0.0972
Total Nitrogen	1	1.5013	0.2205	1	2.8432	0.0918	9	**17.6981**	**0.0388**	3	6.3603	0.0953
Dissolved Nitrogen				1	1.1716	0.2791	9	14.1342	0.1176	3	**10.9569**	**0.012**
Total Kjeldahl Nitrogen	1	2.4237	0.1195	1	2.8253	0.0928	9	**20.3514**	**0.0159**	3	3.6838	0.2965
Nitrate plus Nitrite	1	1.4461	0.2292	1	0.0539	0.8164	9	3.5151	0.9403	3	**29.0924**	**<0.0001**
Ammonia	1	0.8145	0.3668	1	2.8339	0.0923	9	16.2827	0.0612	3	**8.0671**	**0.0446**
Dissolved Nitrogen	1	2.6667	0.1025	1	2.5885	0.1076	9	10.2258	0.3325	3	**17.1304**	**0.0007**
Total Organic Carbon	1	**8.181**	**0.0042**	1	0.0962	0.7564	9	**32.6041**	**0.0002**	3	0.3027	0.9595

Appendix III-6

List of algae identified from the May River estuary and ratio of the concentrations of several phytoplankton pigments relative to chlorophyll-*a* concentrations.

Appendix III-6a. Algae identified in May River Large Tidal Creek and Open Water samples. Potential harmful algae are shown in bold.

Site	Characterization	Date	Algae Present
Stoney	Headwater	5/22/2002	*Melosira*
	Tidal Creek	5/22/2002	*Amphiprora*
		5/22/2002	*Thalassiosira*
		5/22/2002	*Navicula*
		5/22/2002	***Kryptoperidinium foliaceum***
		5/22/2002	*Plankothrix*
		5/22/2002	*Cryptomonas*
		5/22/2002	*Pleurosigma*
		7/31/2002	*Thalassiosira*
		7/31/2002	*Melosira*
		7/31/2002	***Kryptoperidinium foliaceum***
		7/31/2002	*Protoperidinium*
		7/31/2002	*Gyrodinium*
		3/13/2003	*Navicula*
		3/13/2003	*Rhodomonas*
		3/13/2003	*Nitzschia*
		10/31/2002	*Cocconeis*
		10/31/2002	*Navicula*
		10/31/2002	*Synedra*
Rose Dhu	Headwater	5/22/2002	*Cyclotella*
	Tidal Creek	5/22/2002	*Navicula*
		5/22/2002	*Bacillaria paxillifera*
		5/22/2002	***Kryptoperidinium foliaceum***
		5/22/2002	*Melosira*
		5/22/2002	*Heliotheca tamerence*
		7/31/2002	*Amphiprora*
		7/31/2002	*Nitzchia closterium*
		7/31/2002	***Kryptoperidinium foliaceum***
		3/13/2003	*Pleurosigma*
		3/13/2003	*Suriella*
		3/13/2003	*Navicula*
		3/13/2003	*Guinardia*
		10/31/2002	*Euglena*
		10/31/2002	*Navicula*
Palmetto Bluff	Headwater	3/11/2003	*Cocconeis*
	Tidal Creek	3/11/2003	*Pinnularia*
		3/11/2003	*Cyclotella*
		3/11/2003	*Amphora*
		3/11/2003	*Pleurosigma*
		6/6/2002	*Gyrodinium pinque*
		6/6/2002	*Pleurosigma*

Appendix III-6a. (Continued)

Site	Characterization	Date	Algae Present
		6/6/2002	*Nitzchia Longissima*
		8/1/2002	*Gyrosigma*
		8/1/2002	*Pleurosigma*
		8/1/2002	*Melosira*
		8/1/2002	*Amphiprora*
		8/1/2002	*Gyrodinium*
		8/1/2002	*Chaetoceros*
		8/1/2002	*Nitzchia closterium*
		8/1/2002	*Navicula*
		10/30/2002	*Pleurosigma*
		10/30/2002	*Cyclotella*
Brighton Beach	Headwater Tidal Creek	5/21/2002	*Navicula*
		5/21/2002	*Cyclotella*
		5/21/2002	*Amphiprora*
		5/21/2002	*Pleurosigma*
		5/21/2002	*Gyrosigma*
		5/21/2002	*Gyrodinium*
		5/21/2002	***Kryptoperidinium foliaceum***
		5/21/2002	*Plankothrix*
		8/1/2002	*Cyclotella*
		8/1/2002	*Navicula*
		8/1/2002	*Plankothrix*
		3/13/2003	*Meuniera*
		3/13/2003	*Navicula*
		3/13/2003	*Gyrosigma*
		3/13/2003	*Amphiprora*
		10/31/2002	*Pandorina*
		10/31/2002	*Pleurosigma*
Heyward Cove	Headwater Tidal Creek	5/22/2002	*Cyclotella*
		5/22/2002	*Navicula*
		5/22/2002	*Melosira*
		5/22/2002	*Gyrodinium*
		5/22/2002	*Pleurosigma*
		5/22/2002	*Skeletonema costatum*
		5/22/2002	*Synedra*
		8/1/2002	*Skeletonema costatum*
		8/1/2002	*Thalassiosira*
		8/1/2002	*Melosira*
		8/1/2002	*Amphiprora*
		8/1/2002	*Thalassionema*
		3/13/2003	*Navicula*
		3/13/2003	*Rhizosolenia*

Appendix III-6a. (Continued)

Site	Characterization	Date	Algae Present
		3/13/2003	*Thalassiosira*
		3/13/2003	*Synedra*
		10/30/2002	*Melosira*
		10/30/2002	*Achnanthes*
		10/30/2002	*Synedra*
		10/30/2002	*Cyclotella*
		10/30/2002	*Merismopedia*
		10/30/2002	*Plankothrix*
Bass	Headwater	7/31/2002	*Nitzchia closterium*
	Tidal Creek	7/31/2002	*Skeletonema costatum*
		7/31/2002	*Pleurosigma*
		7/31/2002	***Kryptoperidinium foliaceum***
		5/22/2002	*Navicula*
		5/22/2002	*Protoperidinium*
		5/22/2002	*Halteria*
		3/13/2003	*Euglena*
		3/13/2003	*Amphiprora*
		3/13/2003	*Cyclotella*
		3/13/2003	*Pleurosigma*
		3/13/2003	*Gymnodinium*
		10/31/2002	*Pleurosigma*
		10/31/2002	*Amphiprora*
		10/31/2002	*Navicula*
U-01	Large Tidal Creek	6/6/2002	*Nitzchia Longissima*
		6/6/2002	*Chaetoceros*
		6/6/2002	*Rhizosolenia*
		6/6/2002	*Protoperidinium*
		6/6/2002	*Gyrodinium pinque*
		3/12/2003	***Prymnesium***
		3/12/2003	*Euglena*
		3/12/2003	*Skeletonema costatum*
		3/12/2003	*Cyclotella*
		3/12/2003	*Gymnodinium*
		3/12/2003	*Rhodomonas*
		10/28/2002	*Skeletonema costatum*
		10/28/2002	*Corethron criophilum*
		10/28/2002	*Cyclotella*
		10/28/2002	*Thalassionema*
		10/28/2002	***Akashiwo sanguineum***
U-02	Open Water	3/12/2003	*Euglena*
		3/12/2003	*Chaetoceros*
		3/12/2003	*Nitzchia closterium*

Appendix III-6a. (Continued)

Site	Characterization	Date	Algae Present
		3/12/2003	***Prorocentrum minimum***
		3/12/2003	*Gyrodinium pinque*
		3/12/2003	*Heterocapsa triquetra*
		3/12/2003	*Katodinium rotundatum*
		3/12/2003	*Protoperidinium*
		3/12/2003	*Rhodomonas*
		6/5/2002	*Nitzchia Longissima*
		6/5/2002	*Centrodinium*
		6/5/2002	*Gyrodinium*
		6/5/2002	**PLO**
		10/31/2002	*Amphiprora .*
		10/31/2002	*Melosira*
		10/31/2002	*Cocconeis*
		10/31/2002	***Akashiwo sanguineum***
U-03	Open Water	3/12/2003	*Euglena*
		3/12/2003	*Thalassiosira*
		3/12/2003	*Heterocapsa triquetra*
		3/12/2003	*Katodinium rotundatum*
		3/12/2003	***Prorocentrum minimum***
		3/12/2003	*Rhodomonas*
		6/5/2002	*Nitzchia Longissima*
		6/5/2002	*Centrodinium*
		6/5/2002	*Chaetoceros*
		6/5/2002	*Pleurosigma*
		6/5/2002	*Rhizosolenia*
		6/5/2002	*Gyrodinium pinque*
		10/30/2002	*Cyclotella*
		10/30/2002	*Nitzchia closterium*
M-01	Open Water	3/11/2003	*Cyclotella*
		3/11/2003	*Chaetoceros*
		3/11/2003	*Gymnodinium*
		3/11/2003	*Rhodomonas*
		6/5/2002	*Nitzchia Longissima*
		6/5/2002	*Rhizosolenia*
		6/5/2002	*Chaetoceros*
		6/5/2002	*Skeletonema*
		6/5/2002	***Scrippsiella***
		10/28/2002	*Thalassionema*
		10/28/2002	*Cyclotella*
M-02	Open Water	3/11/2003	*Pleurosigma*
		3/11/2003	*Thalassiosira*
		3/11/2003	*Chaetoceros*

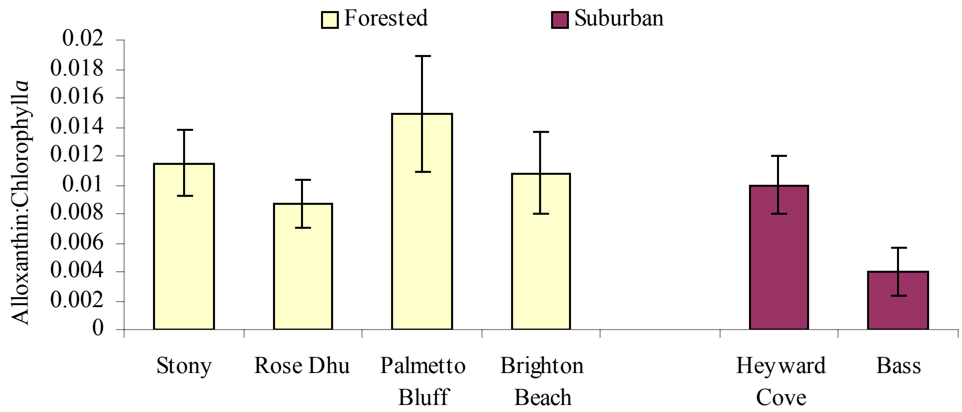
Appendix III-6a. (Continued)

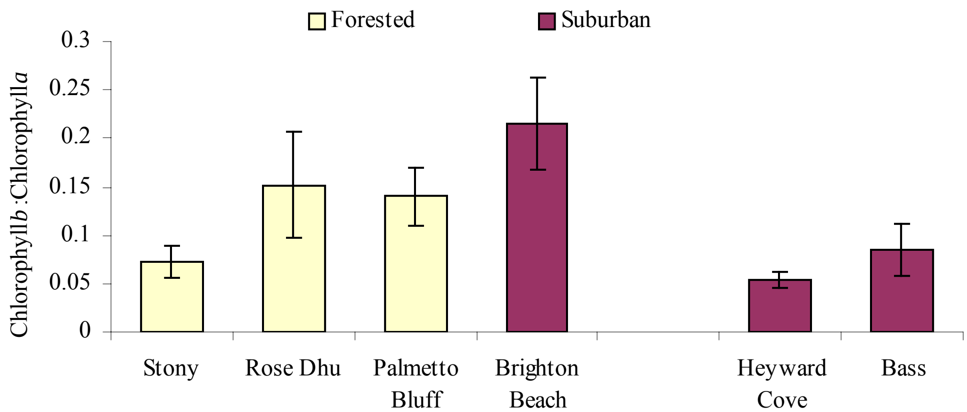
Site	Characterization	Date	Algae Present
		3/11/2003	*Gyrodinium pinque*
		3/11/2003	*Rhodomonas*
		5/23/2002	*Chaetoceros*
		5/23/2002	*Nitzchia closterium*
		5/23/2002	*Gyrodinium*
		5/23/2002	*Gyrodinium pinque*
		10/28/2002	*Cyclotella*
M-03	Open Water	3/11/2003	*Carteria*
		3/11/2003	*Euglena*
		3/11/2003	*Cyclotella*
		3/11/2003	*Thalassiosira*
		6/5/2002	*Skeletonema costatum*
		6/5/2002	*Chaetoceros*
		6/5/2002	*Rhizosolenia*
		6/5/2002	*Gyrodinium pinque*
		6/5/2002	*Gyrodinium*
		6/5/2002	*Nitzchia Longissima*
		6/5/2002	***Scrippsiella***
		6/5/2002	**PLO**
		10/28/2002	*Melosira*
		10/28/2002	*Navicula .*
		10/28/2002	*Cyclotella*
		10/28/2002	*Ditylum brightwellii*
L-01	Open Water	5/23/2002	*Thalassionema*
		5/23/2002	*Nitzchia closterium*
		5/23/2002	*Skeletonema costatum*
		5/23/2002	*Leptocylindricus*
		5/23/2002	*Katodinium rotundatum*
		5/23/2002	*Protoperidinium*
		5/23/2002	*Gyrodinium*
		5/23/2002	***Heterosigma akashiwo***
		10/29/2002	*Cyclotella*
		10/29/2002	*Thalassionema*
		10/29/2002	*Thalassiosira*
		10/29/2002	*Corethron criophilum*
L-02	Open Water	5/23/2002	*Tinntinnid*
		5/23/2002	*Melosira*
		5/23/2002	*Nitzchia closterium*
		5/23/2002	*Cyclotella*
		5/23/2002	*Corethron criophilum*
		5/23/2002	*Thalassiosira*
		5/23/2002	*Chaetoceros*

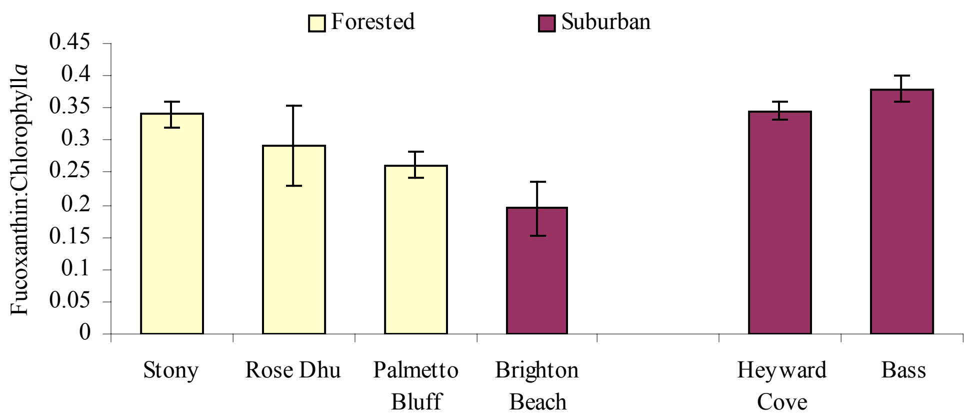
Appendix III-6a. (Continued)

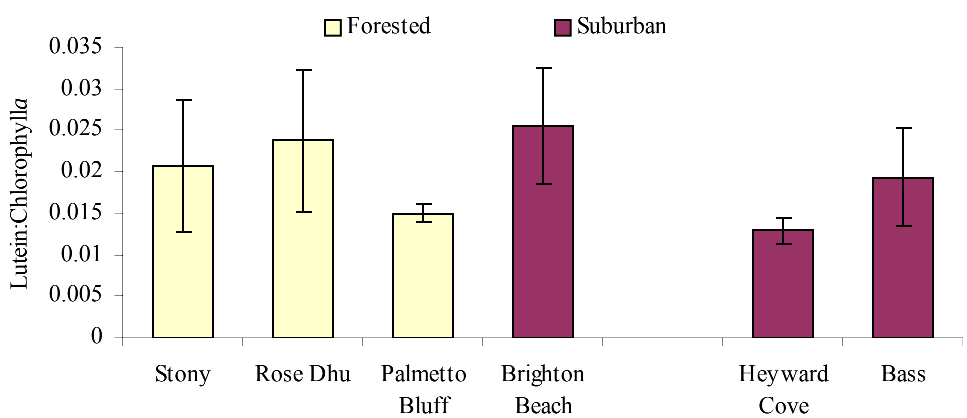
Site	Characterization	Date	Algae Present
		5/23/2002	*Gyrodinium*
		5/23/2002	*Rhodomonas*
		10/29/2002	*Navicula .*
		10/29/2002	*Oxyphysis oxytoxoides*
		10/29/2002	*Prorocentrum micans*
L-03	Large Tidal Creek	6/6/2002	*Centrodinium*
		6/6/2002	*Pleurosigma*
		6/6/2002	*Rhizosolenia*
		10/29/2002	*Cyclotella*
		10/29/2002	*Pleurosigma*
		10/29/2002	*Nitzchia closterium*
L-04	Large Tidal Creek	5/23/2002	*Cyclotella*
		5/23/2002	*Pleurosigma*
		5/23/2002	*Bacillaria paxillifera*
		5/23/2002	*Thalassionema*
		5/23/2002	***Kryptoperidinium foliaceum***
		5/23/2002	*Rhodomonas*
		10/29/2002	*Amphiprora .*
		10/29/2002	*Navicula .*
		10/29/2002	*Gyrodinium*

Appendix III-6b.

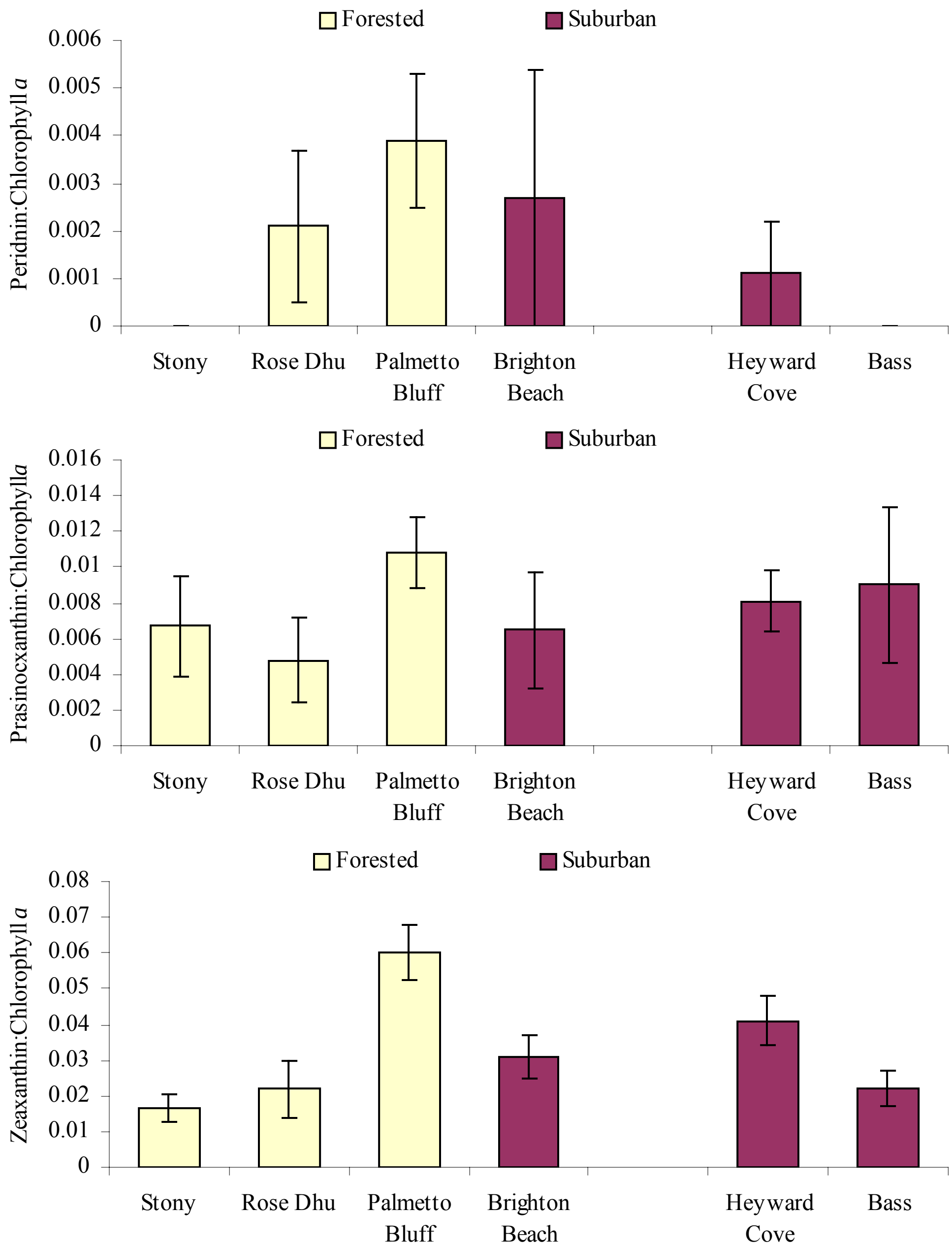




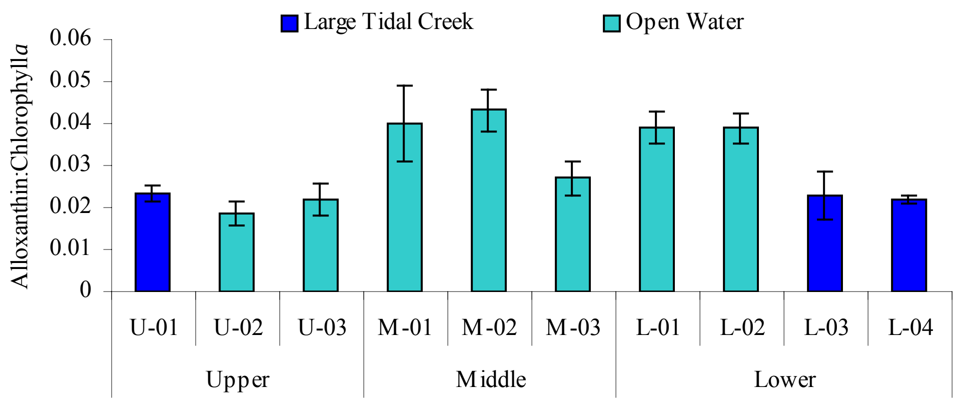


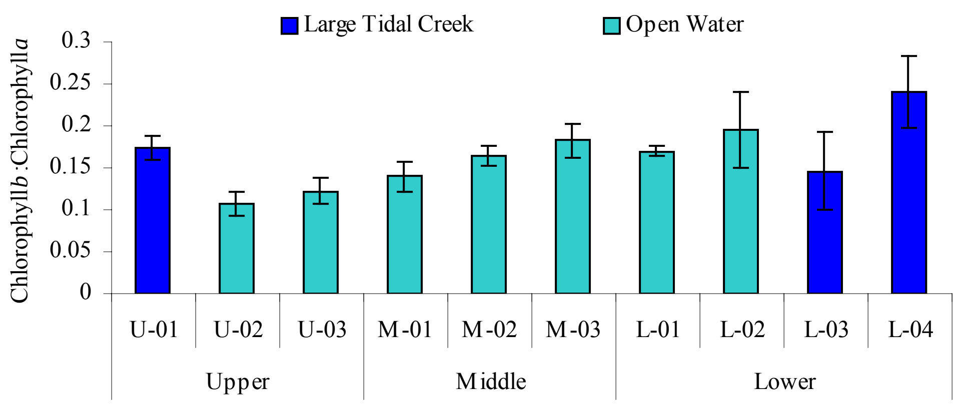


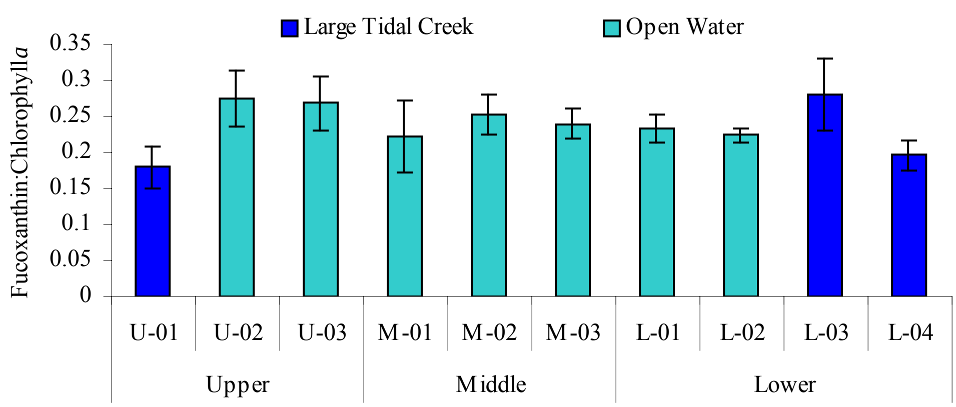
Appendix III-6b. (Continued)

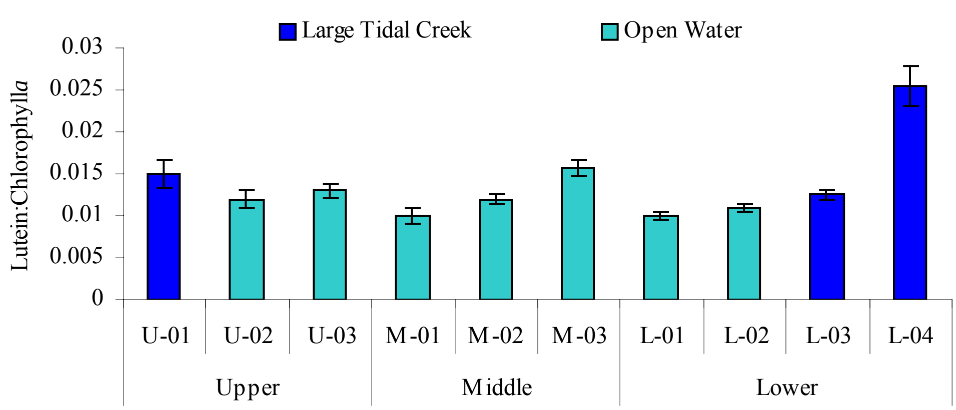


Appendix III-6b. (Continued)

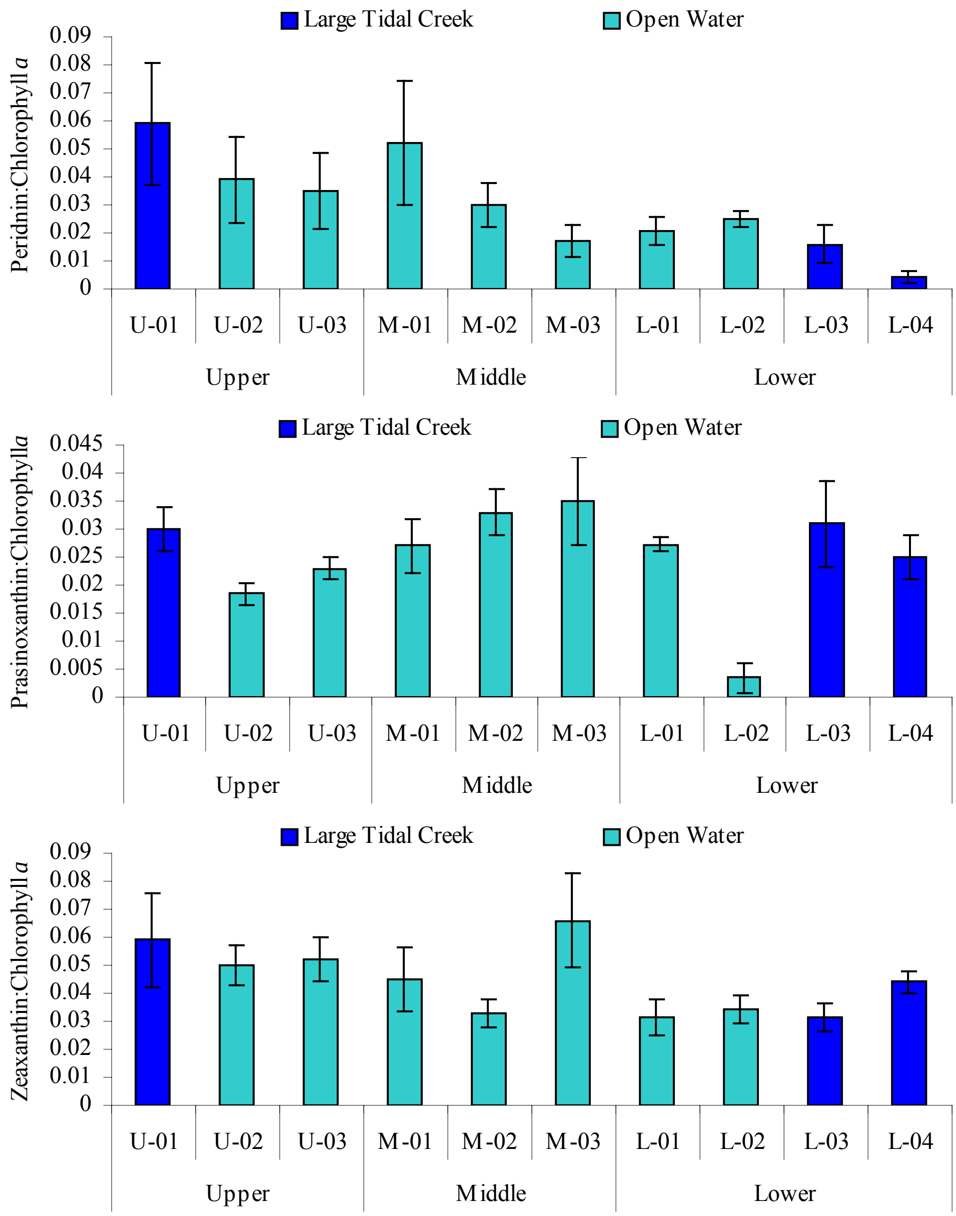








Appendix III-6b. (Continued)



Appendix IV-1.

Data on diseases in oysters sampled in the May River and elsewhere in South Carolina.

Appendix IV-1a. Description of ratings assigned for *Haplosporidium nelsoni* (MSX) infection intensity.

Intensity	
H	Number of Heavy Infections (>5 plasmodia/ 400x field)
M	Number of Moderate Infections (2-5 plasmodia/ 400x field)
L	Number of Light Infections (<2 plasmodia/ 400x field)
R	Number of Rare Infections (1-10 plasmodia/ entire section)
Category	
HS	Heavy Systemic Infections
LS	Rare to Light Systemic Infections
GB	Plasmodia present in Gill with Breakthrough to Digestive Tissue
G	Plasmodia Confined to Gill Epithelial

Appendix IV-1b. *Perkinsus marinus* and *Haplosporidium nelsoni* results from the May River (n=25 oysters/station).

Stations	Dermo Mean Prevalence	Dermo Mean Intensity	Mean Shell Height (mm)	MSX Mean Prevalence	MSX Intensity (H-M-L-R)	MSX Category (HS-LS-GB-G)
Upper Zone						
U-01	68%	1.48	90	0%	0-0-0-0	0-0-0-0
U-02	88%	1.96	84	0%	0-0-0-0	0-0-0-0
U-03	88%	2.28	83	0%	0-0-0-0	0-0-0-0
Grand Mean	**81%**	**1.90**	**86**	**0%**	**0-0-0-0**	**0-0-0-0**
Middle Zone						
M-01	52%	0.92	78	4%	0-0-1-0	0-1-0-0
M-02	88%	2.32	91	4%	0-1-0-0	0-0-0-1
M-03	76%	1.72	85	4%	0-0-1-0	0-0-0-1
M-04	80%	1.58	78	0%	0-0-0-0	0-0-0-0
M-05	76%	2.32	94	4%	0-0-1-0	0-0-0-1
Grand Mean	**74%**	**1.77**	**85**	**3%**	**0-1-3-0**	**0-1-0-3**
Lower Zone						
L-02	72%	1.48	79	4%	1-0-0-0	0-0-0-1
L-03	72%	1.64	100	4%	0-0-1-0	0-0-0-1
L-04	72%	1.56	92	8%	0-0-1-1	0-0-0-2
Grand Mean	**72%**	**1.56**	**90**	**5%**	**1-0-2-1**	**0-0-0-4**

Appendix IV-1c. Prevalence and Intensity of *Perkinsus marinus* (Dermo) and *Haplosporidium nelsoni* (MSX) at monthly monitoring sites sampled in 2002. Stations are listed by county, north to south.

Station Name	County	Date	Dermo Mean Prevalence	Dermo Mean Infection Intensity	MSX Mean Prevalence	MSX Intesnity (H-M-L-R)	MSX Category (HS-LS-GB-G)
Oyster Landing	Georgetown	8-Aug	87%	2.38	0%	0-0-0-0	0-0-0-0
Murrell's Inlet	Georgetown	8-Aug	87%	1.85	0%	0-0-0-0	0-0-0-0
Toler's Cove	Charleston	5-Sep	100%	2.56	8%	2-0-0-0	0-0-0-2
Inlet Creek	Charleston	5-Sep	100%	2.64	8%	0-2-0-0	0-0-0-2
Price Creek	Charleston	5-Sep	84%	2.16	24%	2-0-2-2	1-0-0-5
Warsaw Flats	Beaufort	22-Aug	84%	1.92	0%	0-0-0-0	0-0-0-0

Appendix IV-1d. Mean infection intensity and prevalence of the 52 stations sampled for *Perkinsus marinus* (Dermo) and *Haplosporidium nelsoni* (MSX) during August through September 1996. Stations are listed by counties.

Station Name	County	Date	Dermo Mean Prevalence	Dermo Mean Intensity	MSX Mean Prevalence	MSX Intensity (H-M-L-R)	MSX Category (HS-LS-GB-G)
Main Creek	Horry	27-Aug	88%	2.48	0%	0-0-0-0	0-0-0-0
Weston Creek	Georgetown	27-Aug	76%	2.31	0%	0-0-0-0	0-0-0-0
Clambank Creek	Georgetown	27-Aug	96%	2.88	0%	0-0-0-0	0-0-0-0
North Jones Creek	Georgetown	26-Sep	28%	0.52	0%	0-0-0-0	0-0-0-0
Casino Creek	Charleston	28-Aug	92%	2.16	12%	1-0-0-2	0-1-0-2
Mathew=s Cut	Charleston	28-Aug	64%	1.36	32%	2-2-4-0	0-3-5-0

Appendix IV-1d. (Continued)

Station Name	County	Date	Dermo Mean Prevalence	Dermo Mean Intensity	MSX Mean Prevalence	MSX Intensity (H-M-L-R)	MSX Category (HS-LS-GB-G)
Horsehead Creek	Charleston	19-Sep	68%	2.16	8%	1-1-0-0	1-1-0-0
Nellie Creek	Charleston	19-Sep	76%	1.48	4%	0-1-0-0	0-0-0-1
Sandy Point Creek	Charleston	1-Oct	72%	2.24	4%	1-0-0-0	1-0-0-0
Five Fathom Creek	Charleston	28-Aug	76%	2.04	16%	1-2-1-0	0-1-3-0
Key Inlet	Charleston	19-Sep	68%	1.88	12%	1-0-0-2	0-0-0-3
Graham Creek	Charleston	1-Oct	80%	1.92	0%	0-0-0-0	0-0-0-0
Vanderhorst Creek	Charleston	3-Oct	76%	1.52	0%	0-0-0-0	0-0-0-0
Bull Creek	Charleston	3-Oct	32%	0.60	4%	0-0-0-1	0-0-0-1
Skipper Munn=s	Charleston	15-Aug	80%	1.92	8%	0-0-1-1	0-0-0-2
Sewee Bay	Charleston	23-Sep	100%	2.88	0%	0-0-0-0	0-0-0-0
Clausen Creek	Charleston	23-Sep	60%	1.32	0%	0-0-0-0	0-0-0-0
Copahee Sound	Charleston	23-Sep	68%	1.40	0%	0-0-0-0	0-0-0-0
Capers Creek	Charleston	23-Sep	76%	1.60	8%	2-0-0-0	0-0-2-0
Swinton Creek	Charleston	16-Oct	100%	1.56	4%	0-0-0-1	0-0-0-1
Conch Creek	Charleston	16-Oct	84%	0.92	8%	2-0-0-0	0-0-2-0
Church Creek	Charleston	3-Sep	88%	2.20	0%	0-0-0-0	0-0-0-0
Clark Sound	Charleston	16-Oct	100%	2.00	4%	0-1-0-0	0-0-1-0
Folly River, North	Charleston	20-Aug	76%	1.84	12%	1-0-2-0	0-0-0-3
Cut Off Reach	Charleston	20-Aug	100%	2.88	12%	1-1-1-0	1-0-0-2
Folly Creek, Crosby’s	Charleston	20-Aug	92%	2.64	8%	1-1-0-0	0-0-0-2
Kiawah River	Charleston	21-Aug	56%	1.60	0%	0-0-0-0	0-0-0-0
Bass Creek	Charleston	21-Aug	80%	1.80	4%	0-1-0-0	0-1-0-0

Appendix IV-1d. (Continued)

Station Name	County	Date	Dermo Mean Prevalence	Dermo Mean Intensity	MSX Mean Prevalence	MSX Intensity (H-M-L-R)	MSX Category (HS-LS-GB-G)
Capt. Sam=s Inlet	Charleston	21-Aug	84%	2.16	4%	0-0-0-1	0-1-0-0
Russel Creek	Charleston	18-Oct	64%	1.32	8%	0-0-2-1	0-1-0-1
Leadenwah Creek	Charleston	3-Sep	64%	1.31	0%	0-0-0-0	0-0-0-0
Bohicket Creek	Charleston	3-Sep	76%	2.00	8%	0-2-0-0	0-0-1-1
Ocella Creek	Charleston	18-Oct	44%	0.68	0%	0-0-0-0	0-0-0-0
Big Bay Creek	Charleston	2-Oct	68%	2.00	12%	1-1-1-0	1-0-1-1
Two Sisters Creek	Colleton	21-Oct	92%	1.48	12%	1-2-0-0	0-1-0-2
North Fish Creek	Colleton	2-Oct	68%	1.56	12%	0-0-3-0	0-1-0-2
South Fish Creek	Colleton	2-Oct	72%	1.72	4%	0-0-1-0	0-0-0-1
Lucy Point Creek	Beaufort	13-Aug	20%	0.28	0%	0-0-0-0	0-0-0-0
Dataw Island	Beaufort	14-Aug	56%	1.20	0%	0-0-0-0	0-0-0-0
Johnson Creek	Beaufort	9-Oct	84%	1.04	0%	0-0-0-0	0-0-0-0
Capers Creek/Distant Is.	Beaufort	7-Oct	48%	1.16	0%	0-0-0-0	0-0-0-0
Battery Creek	Beaufort	7-Oct	44%	0.72	0%	0-0-0-0	0-0-0-0
Chechessee River	Beaufort	24-Sep	28%	0.60	0%	0-0-0-0	0-0-0-0
Colleton River	Beaufort	24-Sep	52%	1.00	4%	0-0-0-1	0-0-0-1
Fripp Inlet	Beaufort	9-Oct	100%	1.88	0%	0-0-0-0	0-0-0-0
Club Bridge Creek	Beaufort	8-Oct	92%	1.68	0%	0-0-0-0	0-0-0-0
Station Creek	Beaufort	8-Oct	64%	1.40	0%	0-0-0-0	0-0-0-0
Mackay Creek	Beaufort	10-Sep	96%	2.48	0%	0-0-0-0	0-0-0-0
Jarvis Creek	Beaufort	10-Sep	92%	2.40	8%	0-1-0-1	0-1-0-1
Bull Creek	Beaufort	10-Sep	96%	2.12	8%	0-0-1-1	0-0-0-2

Appendix IV-1d. (Continued)

Station Name	County	Date	Dermo Mean Prevalence	Dermo Mean Intensity	MSX Mean Prevalence	MSX Intensity (H-M-L-R)	MSX Category (HS-LS-GB-G)
Old House Creek	Beaufort	10-Sep	100%	2.56	4%	0-0-1-0	0-0-1-0
Broad Creek	Beaufort	10-Sep	84%	2.12	0%	0-0-0-0	0-0-0-0

100% positive for Dermo

54 % (or 28) positive for MSX

Appendix IV-1e. *Perkinsus marinus* and *Haplosporidium nelsoni* results from Broad Creek (BOY) and Okatee River (OOY), Beaufort County (Sampled during 1997).

Station Name	Date	Dermo Mean Prevalence	Dermo Mean Intensity	MSX Mean Prevalence	MSX Intesnity (H-M-L-R)	MSX Category (HS-LS-GB-G)
BOY1	26-Aug	100%	1.84	12%	0-1-1-1	0-0-0-3
BOY2	26-Aug	92%	1.24	12%	1-1-0-1	0-1-0-2
BOY3	26-Aug	100%	2.04	4%	0-0-0-1	0-0-0-1
BOY4	27-Aug	100%	2.00	33%	2-1-4-1	2-3-0-3
BOY5	27-Aug	36%	0.76	21%	0-2-1-2	0-2-0-3
BOY6	27-Aug	84%	2.08	8%	0-0-2-0	0-0-0-2
Station Grand Mean		**85%**	**1.66**	**15%**	**3-5-8-6**	**2-6-0-14**
OOY1	10-Sep	92%	1.84	4%	0-1-0-0	0-0-0-1
OOY2	10-Sep	96%	2.28	4%	0-1-0-0	0-0-0-1
OOY3	11-Sep	84%	1.56	12%	1-0-2-0	1-0-0-2
OOY4	11-Sep	88%	1.40	4%	0-0-1-0	0-1-0-0
OOY5	11-Sep	96%	1.72	4%	0-1-0-0	0-1-0-0
OOY6	11-Sep	92%	1.48	8%	0-0-0-2	0-0-0-2
Station Grand Mean		**91%**	**1.71**	**6%**	**1-3-3-2**	**1-2-0-6**

Appendix Figure V-1.

Comparison of the relationships (black line) between the eight parameters evaluated in the Tidal Creek Project (TCP) for the 1994/1995/2002 data (diamonds) and impervious cover (%) from Holland and others (2004). The black circles represent the six tidal creeks sampled during the May River Project.

Appendix Figure V-1