Evaluation of the Impacts of Dock Structures on South Carolina Estuarine Environments

Technical Report Number 99

prepared by

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Final Report

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2002

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Submitted to:

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Funding for this project was provided by the Coastal Zone Management Act of 1972, as amended, administered by the Office of Ocean and Coastal Resource Management, National Oceanic and Atmospheric Administration, through Grant-in-Aid award number NA07OZ0133 to the South Carolina Department of Health and Environmental Control's Office of Ocean and Coastal Resource Management.

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I. INTRODUCTION

The estuaries of the Southeastern United States have 1.5-3.0 m tidal ranges (meso-tidal), shallow depth, and expansive intertidal areas (Hackney *et al.* 1976, Nummedal *et al.* 1977, Day 1989). *Spartina alterniflora* salt marshes and the associated networks of tidal creeks that drain them are characteristic geographical features of southeastern estuaries (Teal 1962, Wiegert and Freeman 1990, Field *et al.* 1991). These ecosystems support complex food webs, play major roles in material and nutrient cycles and provide nursery habitat for valued living resources including fish, crabs, and shrimp (Cain and Dean 1976, Hackney *et al.* 1976, Shenker and Dean 1979, Weinstein 1979, Wiegert and Freeman 1990, Hoffmann 1991, Wenner and Beatty 1993, Kneib 1993, Lerberg *et al.* 2000). Estuarine ecosystems also provide critical services to humans by functioning as storm buffers, navigation routes, and repositories for pollutants (Constanza *et al.* 1997, Pederson and Lubchenco 1997, Sanger *et al.* 1999a and 1999b). The conservation and protection of tidal creek-salt marsh habitat is essential to maintaining the ecological integrity of the state's estuarine ecosystems.

The human population of the United States is concentrated near coastal ecosystems with approximately 37% of the 1994 population living within 100 km of a coastline (Cohen *et al.* 1997). Additionally, the natural beauty of South Carolina's tidal creeks and salt marshes and the state's mild climate attracts millions of tourists to the state's coastal zone each year. Land with scenic vistas of creeks and salt marshes is highly valued as sites for homes, tourist resorts, and recreational facilities (Miller 1993). The proportion of the human population that lives in watersheds which drain into South Carolina estuaries has increased substantially over the last decade (Cofer-Shabica *et al.* 1999). For example, the urbanized area of Charleston increased by over 400% between 1973 and 1994 and is projected to increase at an equivalent rate for the next several decades (Vang personal communication).

As the size of the coastal population increased so has the number of requests for dock permits (NOAA 2001). The South Carolina Department of Health and Environmental Control's Office of Ocean and Coastal Resources Management (SCDHEC-OCRM) is the agency responsible for establishing guidelines and policies for when, where, and how docks are constructed. In 1982 SCDHEC-OCRM permitted 80 docks compared to approximately 700 docks per year between 1991 and 2000. Historically the environmental concerns that had to be addressed for each dock permit included: (1) ensuring the structure did not impact navigation or public access to shellfish grounds; (2) developing an engineering design (e.g., height, width, location) that minimized direct impacts on salt marsh vegetation, fauna, and public safety; and (3) ensuring the dock did not represent an "eye sore". In most cases, if navigation and shellfish grounds were not affected and the structure adhered to standard engineering practices, then the permit was granted.

Recently, a number of studies have been conducted that measured dock shading effects on marsh grasses (Kearney *et al.* 1983, McGuire 1990, Colligan and Collins 1995) and submerged aquatic vegetation (Shafer 1999), estimated the leaching rates of wood preservatives from docks (Breslin and Adler-Ivanbrook 1998), assessed the toxicity of dock leachates (chrominated copper arsenate or CCA) to living resources (e.g., Weis and Weis 1992a, Weis *et al.* 1991, Weis *et al.* 1992, Wendt *et al.* 1996), estimated the bioaccumulation of wood preservatives in marine biota (e.g., Weis *et al.* 1993a, Weis and Weis 1999), and evaluated the effects of dock leachates on the kinds and abundances of marine organisms that live in sediments (e.g., Weis and Weis 1996a and 1996b, Weis *et al.* 1998). Models have also been developed for assessing the environmental risks of dock leachates on the marine environment (Brooks 1996) and for minimizing shading impacts from docks on marsh grasses (McGuire 1990). With the exception of Wendt *et al.* (1996) and McGuire (1990), many of the above studies are not directly applicable to the environmental conditions that are characteristic of South Carolina tidal creeks or the type of salt marsh and tidal creek habitats that exist in South Carolina. Some of the environmental concerns about docks that have been identified but remain to be assessed include decreased public access to intertidal environments, particularly shellfish beds; increased erosion of creek banks and salt marsh habitat; increased turbidity; direct and indirect impacts from dock-related recreational boating activity on marine resources including chemical contamination of sediments and biota; and degradation of scenic views.

With each new request for a dock permit, public concerns about the cumulative environmental impacts of dock proliferation on the coastal environment have increased (NOAA 2001). Regulatory agencies responsible for managing docks, including SCDHEC-OCRM, are increasingly being required to defend the guidelines and policies they use to control dock proliferation in court (NOAA 2001). Unfortunately, the science supporting dock policies and regulations have not been compiled and synthesized in a manner which support legal challenges or facilitate the assessment of cumulative impacts.

The objective of this study was to evaluate the cumulative effect of docks on tidal creek and salt marsh ecosystems. The study was composed of three parts: (1) a *Spartina* Shading Study which evaluated the impacts of dock shading on the dominant marsh plant; (2) a Small Tidal Creek Study which evaluated dock impacts on small tidal creek nursery habitats; and (3) a Large Tidal Creek Study which evaluated dock impacts on larger tidal creek nursery habitats. Shading impacts under individual docks were extrapolated to the tidal creek (local), county, and state-wide scales. In addition, wrack accumulation and construction damage were examined as part of the *Spartina* Shading Study. No new data were collected for the small and large tidal creek studies. Rather, existing research and monitoring data collected by the SCDNR were used. A bibliography of the relevant scientific literature and summarization of the science that supports the impacts of dock structures on the marine environment is also provided.

II. METHODS

A. *Spartina* Shading Study

To address the effects of shading from dock structures on the stem density of the dominant salt marsh plant, *Spartina alterniflora*, stem density measurements from two studies were used: (1) a study conducted on the York River, Virginia, by McGuire (1990), and (2) a study conducted in Charleston Harbor, South Carolina in conjunction with a Timberland High School student. Both studies evaluated the effects of shading on stem density under only the walkway of dock structures.

The objective of the McGuire (1990) study was to develop a model that could be used to design dock structures in a manner that minimized their effects on *S. alterniflora*. For her study, McGuire (1990) established a transect perpendicular to 35 docks representing a range of orientations to the sun and amount of shading. Then, a total of 15-17 measurements of stem density were made in 0.1 m^2 areas along each transect under the dock structure and on each side of the dock structure. We obtained McGuire's data and calculated the percent reduction in *S. alterniflora* stem density under dock structures compared to the stem density that occurred greater than 2.5 m away from the dock structure.

The South Carolina *S. alterniflora* Shading Study was conducted on 32 docks located on three tidal creeks in the Charleston area: (1) Long Creek on Wadmalaw Island; (2) Parrot Creek on James Island; and (3) James Island Creek on James Island (Figure 1). The docks surveyed represented a range of lengths, orientations to the sun, and ages. One or two transects were established perpendicular to each dock. A total of 51 transects were sampled. Transects were established using the following criteria:

1) If the dock was greater than 50-m long and both growth forms of *S. alterniflora* (short and tall) occurred under the dock, then two transects were established at:
    a) 10-m from the upland terminus of the dock structure, and
    b) 10-m from the creek bank.
2) If the dock was less than 50-m long and both growth forms of *S. alterniflora* occurred under the dock, then two transects were established at:

a) 5 m from the upland terminus of the dock structure, and
b) 5-m from the creek bank.

3) If only one growth form of *S. alterniflora* occurred under the dock, then one transect was established at the mid-point of the dock.

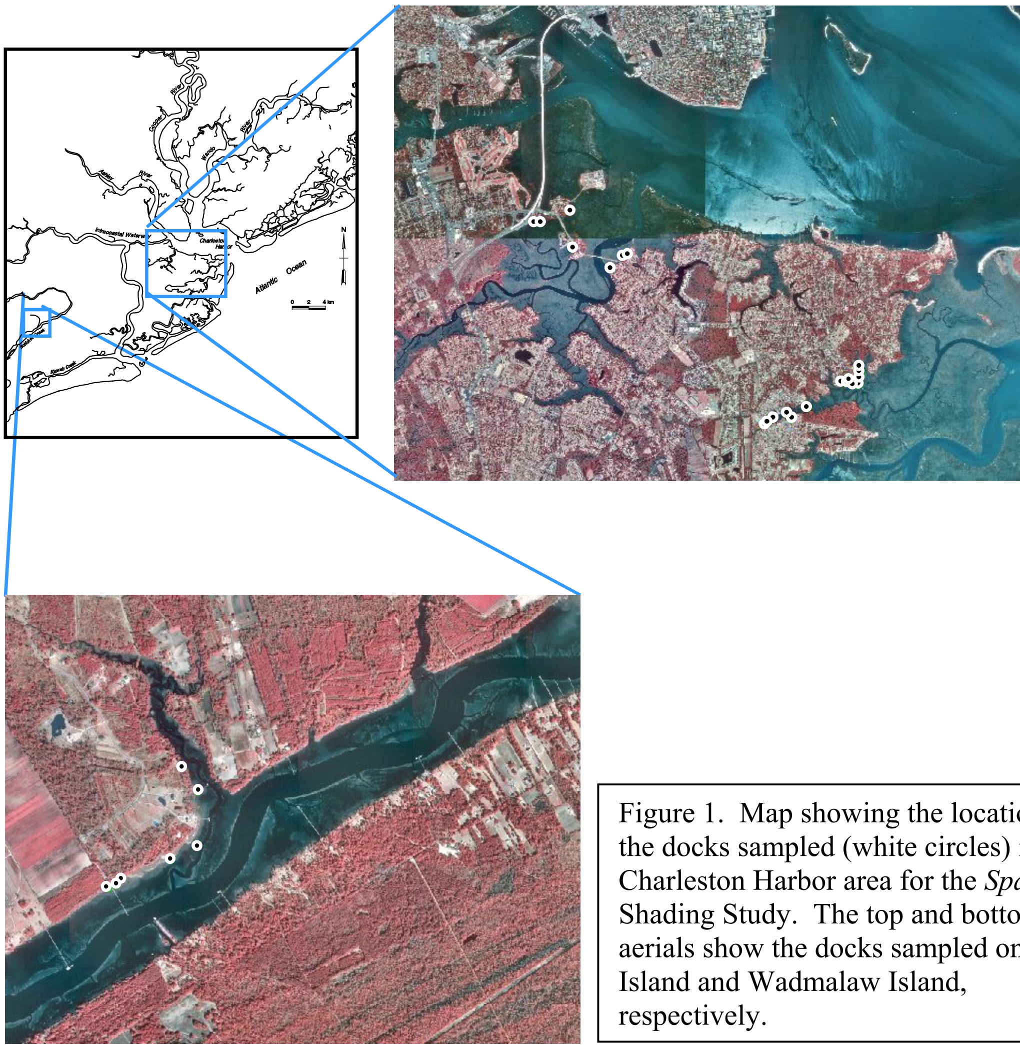


Figure 1. Map showing the location of the docks sampled (white circles) in the Charleston Harbor area for the *Spartina* Shading Study. The top and bottom aerials show the docks sampled on James Island and Wadmalaw Island, respectively.

On each transect a density measurement was made at a site under the center of the dock and at a site 5-m to the left or right of the dock (Figure 2). Stem density was measured using a 0.1 m^2 frame. Accuracy in stem counts was established in the field by having a second person recount the sample plot. The recounts verified that the counting process was 95% accurate. Ancillary measurements made for each dock included dock length, dock height at the location of each transect, dock width, distance between planks, and latitude and longitude. Latitude and longitude measurements were made using a Garmin GPS III nondifferential GPS instrument.

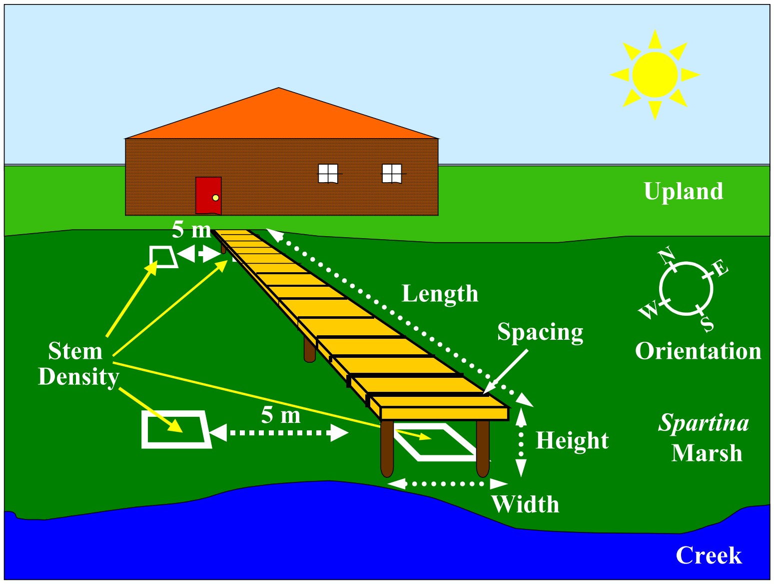


Figure 2. A diagram of the sampling design for the *Spartina alterniflora* shading survey conducted by this study.

Paired t-tests were used to evaluate if the *S. alterniflora* density under docks was different from the density 5 m from the dock for: (1) short-form *S. alterniflora*; (2) tall-form *S. alterniflora*; (3) short- and tall-form *S. alterniflora* combined. A t-test was used to evaluate if magnitude of stem density reductions between the docks oriented in a North-South (315-45° and 135-225°) direction were different from reductions under docks oriented in an East-West (45-135° and 225-315°) direction. Stem densities were log transformed [log 10 (x + 1)] for analyses.

The percent reduction in *S. alterniflora* stem density under each dock was calculated. The average percent reduction for the surveyed docks was also calculated and used to estimate the amount of salt marsh area affected by dock shading at the local or small creek, county, and statewide scales.

Estimates of the shading impacts from dock structures were made at the three spatial scales using the following general equation:

$$SE = (\ell * w * s * n / sma) * 100$$

Where:

SE = a measure of dock shading effects on *Spartina alterniflora* productivity at the specific creek, county, and statewide scales adjusted for the reduction in stem density that was estimated to occur under South Carolina docks.
ℓ = either the actual length of docks in specific creeks, or a range of dock lengths (see Tables 11 and 12 for average lengths used for the county and statewide estimates, respectively).
w = the average width of docks (i.e., 1.22 m).
s = the percent reduction in *Spartina alterniflora* stem density under South Carolina dock structures found in this study (i.e., 71%).
n = the number of docks occurring in specific creeks or the average number of docks permitted by the SCDHEC-OCRM between 1991-2000.
sma = the salt marsh area reported to exist in the drainage area of specific creeks (Lerberg *et al.* 2000), coastal counties or statewide (Tiner 1977).

Projections of the maximum shading effects that could theoretically occur in each specific small tidal creek were based on the existing dock regulations which allow a dock to be constructed every 23 m (75 ft) of marsh front property and the average length of docks in each creek. Projections of shading effects in 2010 were also made for the county and statewide scales. The number of docks projected to occur between 2001 and 2010 was assumed to be equivalent to the number permitted from 1991-2000.

B. Small Tidal Creek Study

1. Classification and Identification of Categories

The investigation of the impacts of docks on small tidal creeks utilized data collected by the Tidal Creek Project (TCP). TCP data was collected, processed, and analyzed by the South Carolina Department of Natural Resources' (SCDNR) Marine Resources Research Institute (MRRI) and the National Oceanic and Atmospheric Administration-National Ocean Service Charleston Laboratory (NOAA-NOS) with funding from the Charleston Harbor Project, South Carolina Marine Recreational Fishery Advisory Board, and SCDNR. The overall goal of TCP was to define linkages between land use and tidal creek environmental quality using a comparative watershed approach at a comprehensive ecosystem scale (Holland *et al.* 1997, Sanger *et al.* 1999a and 1999b, Lerberg *et al.* 2000).

In 1994, 1995, and 2000, the SCDNR Tidal Creek Project (TCP) sampled twenty-four tidal creeks in the Charleston Harbor area. This study utilized data from fifteen of the twenty-four creeks. These fifteen creeks were previously classified into the following land use categories: (1) forested or reference (i.e., <15% of the watershed as urban/suburban land cover and some freshwater inflow), and (2) suburban (i.e., >45% urban/suburban land cover with a human population density >5 but <20 individuals per hectare and some freshwater inflows). The suburban creeks were further classified into the following two groups: (1) creeks containing docks, and (2) creeks without docks (Figure 3). The data for the remaining nine creeks were not used because they did not contain docks and represented habitats (creeks that drained only salt marsh) and land uses (urban, industrial, agricultural) that were not appropriate for inclusion in this study.

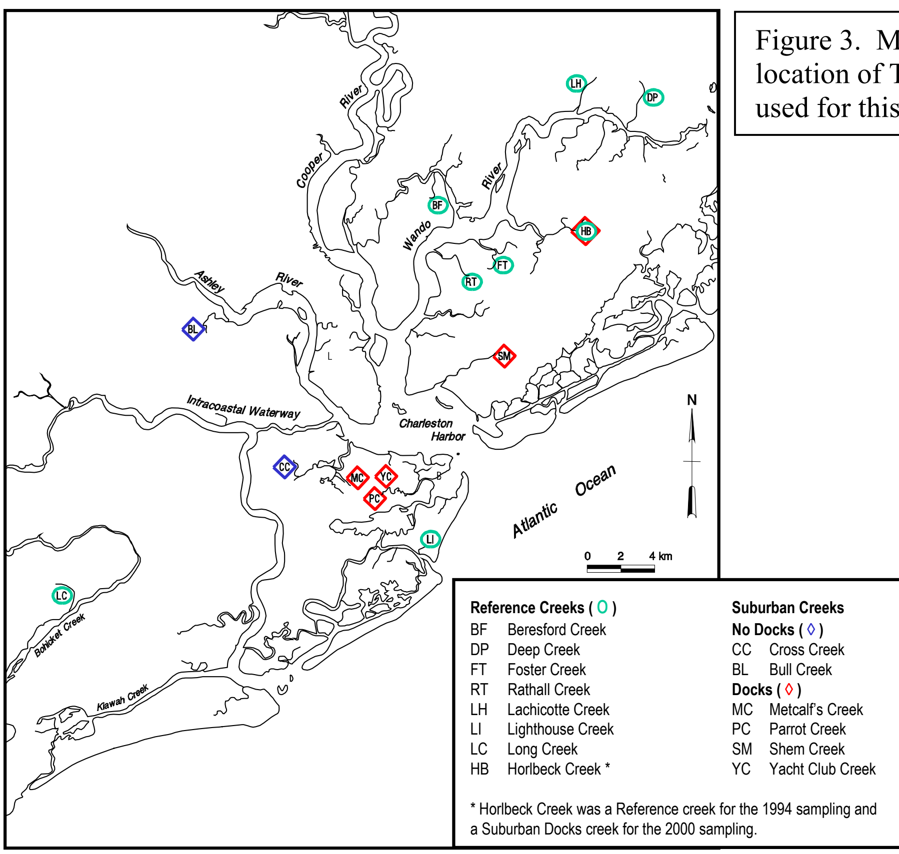


Figure 3. Map showing the location of TCP small tidal creeks used for this study.

The upper boundary of sampling in the small tidal creeks was where the channel depth at mean high tide was about one meter, or where an impassable obstacle (a dike or security fence) was reached. The lower boundary was the point where the creek converged with another water body, or the depth in the channel exceeded three meters at mean high tide. The selected creeks represented the range of salinity distributions, sediment characteristics, creek lengths (231-m to 1491-m), watershed sizes, land cover, and levels of suburban development that occur in Charleston Harbor. Each creek was stratified into 300-m reaches for sampling. Creeks varied in length from 1 to 5 reaches.

2. Sample Collection and Processing

The fifteen creeks were randomly sampled between July and September 1994. In 1994, samples were collected for macrobenthic community characteristics, fish and crustacean community characteristics, sediment grain size, water quality, and physical dimensions. Between July and September 1995, the fifteen tidal creeks were re-sampled to measure sediment chemical contamination in the upper- and lower-most reaches. Between January and February 2000, the upper-most reach of the fifteen tidal creeks were sampled a third time for macrobenthic community characteristics, sediment grain size, water quality, and physical dimensions.

Macrobenthic community samples were collected randomly at approximately 1-m below mean high tide (mid-tide level). Six samples were collected in each reach in 1994 and three samples were collected in the upper-most reaches in 2000. Samples were collected with a 45.6 cm^2 coring device to a depth of 15 cm and sieved through a 500-µm screen. All organisms retained on the screen were identified to the lowest possible taxonomic level and counted. Surface (upper 2 cm) sediment samples were collected at each benthic site and processed to determine percent moisture, percent silts and clays, and percent sand using standard methods modified from Plumb (1981).

Seine (0.6-cm square mesh) samples of fish and crustaceans were collected at a random location in each small tidal creek in 1994. Seines were collected for 25 m in an upstream direction from bank-to-bank on an ebbing tide when the water depth was less than 1-m deep but greater than 0.25-m deep. Efforts were made to minimize disturbance to the seine site prior to sampling. Material collected in the seine was fixed in 10% formalin buffered with seawater for a minimum of one week before processing. Seine samples were processed by two methods depending on the weight of the sample. Samples with a wet weight <2 kg were completely processed. Samples weighing >2 kg were sub-sampled because the time required to completely process these samples would have been excessive for the information obtained. For samples that were subdivided, large (>20-cm) organisms and rare taxa were removed and placed in 50% isopropanol. The remaining sample was divided into ten approximately equal weight sub-samples. Two of the ten sub-samples were randomly selected and processed. Organisms were identified to the lowest possible taxa.

A Hydrolab Datasonde 3 (DS3) water quality monitoring system was placed in the lower-most reach of each creek for two to seven days prior to sampling. The DS3 was positioned approximately 5-10 cm above the creek bottom and measured salinity, water temperature, dissolved oxygen concentration, pH, and water depth at 30-minute intervals.

The physical creek dimensions that were measured included: creek length, creek depth, and creek width at each seine site. Creek width was defined as the distance between the edges of the *Spartina*. Creek depth was defined as the distance from the creek bed to the average high tide mark as indicated by the organic film on *Spartina* stems. For each seine site, depth was measured at 25%, 50%, and 75% of the width of the creek. These depth and width measurements were used to estimate the area and volume of each seine sample.

Surface sediments (upper 2 cm) were collected for chemical analyses at one randomly selected site from the upper- and lower-most reach of each creek in 1995. These samples were homogenized and divided into three aliquots. One aliquot was placed in a pre-cleaned plastic jar for trace metal analyses. The second aliquot was placed in a pre-cleaned glass jar for organic contaminant analyses. The third aliquot was stored in a plastic bag for organic carbon analysis. The samples were immediately placed on ice and stored at – 60°C until processed. Analyses were performed for 14 trace metal (including arsenic, chromium, and copper), 24 polycyclic aromatic hydrocarbon (PAHs), and 20 polychlorinated biphenyl (PCB) analytes by the National Ocean Service Laboratory (NOAA-Charleston, SC) using methods described by Sanger *et al.* (1999a and 1999b).

The boundary of each watershed was defined using elevation contours on a 1:24,000 scale United States Geological Survey (USGS) topographic map and a Geographical Information System (GIS). The area was classified into land use categories based on a modification of the Anderson Land Use Classification System (Anderson *et al.* 1976). The percent of each watershed that was impervious surface was determined by point sampling a minimum of 200 randomly selected sample points within each watershed on 1:4,800 blackline aerial photographs (photographed in 1989) in 1994 and 1:4,800 high-resolution Color Infrared (CIR) National Aerial Photographic Program (NAPP) photographs developed by the United States Department of Agriculture (photographed in 1999) in 2000. The sample points which fell on roads, parking lots, roofs, and other impervious surfaces were counted and converted to a percentage.

C. Large Tidal Creeks Study

1. Classification and Identification of Categories

The investigation of the impacts of docks on large tidal creeks used data collected by the South Carolina Estuarine and Coastal Assessment Program (SCECAP). SCECAP data was collected, processed, and analyzed by the SCDNR-MRRI and the SCDHEC in association with the U.S. Environmental Protection Agencies' Coastal 2000 Program. The overall goal of SCECAP is to assess the environmental condition of the state's estuarine habitats and associated biological resources on an annual basis (Van Dolah unpublished). A secondary goal of SCECAP is to provide environmental data that others can use to address the environmental issues of concern to them (e.g., this study).

SCECAP sample sites were located throughout the state's coastal zone from the coastal ocean to the freshwater boundaries. Sampling was limited to summer when environmental conditions associated with pollution are expected to be stressful to living resources. SCECAP sampling began in the summer of 1999. Sample sites were selected using a probability-based sampling design and a habitat map created from a variety of Geographic Information Systems (GIS) sources. Large tidal creeks, defined as estuarine water bodies that intersect marsh habitat and were between 15 m (50 ft) and 100 m (328 ft) in width from marsh bank-to-marsh bank, were one of the major habitats sampled by SCECAP. The SCECAP sample design provides an unbiased, random array of sample sites that represents conditions for the entire state.

In 1999 and 2000 SCECAP sampled 52 large tidal creeks. Each of the SCECAP large tidal creek sites were mapped on 1994 NAPP digital ortho quarter quad (DOQQ) aerial photography using ArcView© software and a circular buffer (500-m radius, 78.5 ha or 194 ac) was digitally created around each site. Only sites that had approximately >10% uplands in the buffer were included in this study. Twenty six of the 52 sites met this criterion (Figure 4 and Table 1). The buffer for the remaining 26 sites enclosed almost entirely salt marsh vegetation and did not contain any docks.

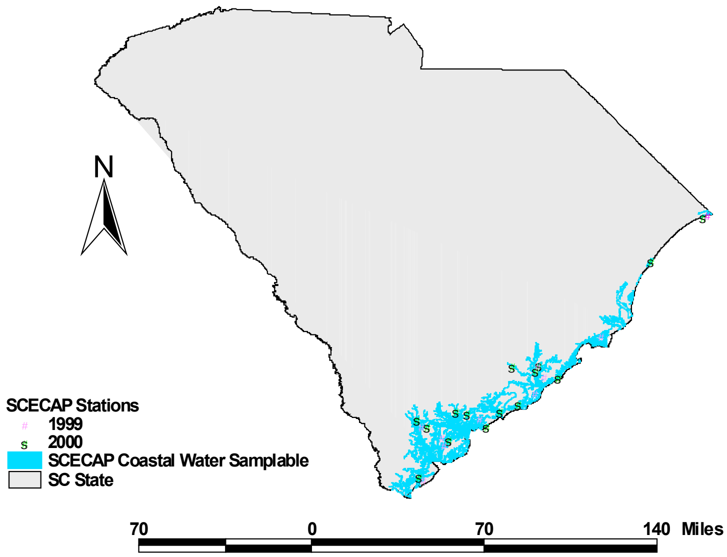


Figure 4. The twenty-six SCECAP stations which were used for this study.

The NAPP DOQQ photographs for the 26 large tidal creek sites that met the uplands criterion were mapped with the 500-m radius buffer and a randomly placed triangular point grid was displayed on the photograph (Figure 5). The dimensions of the grid were such that approximately 700 points occurred within the buffer. Grid points that were located on salt marsh, water, and uplands were counted and used to estimate the percentage of these types of land cover within the buffer. In addition, the upland area was sampled to determine the percentages that were forested and impervious surface. Impervious surface was defined as any grid point that fell on a roof, road, parking lot, or other surfaces that were impermeable to rainfall.

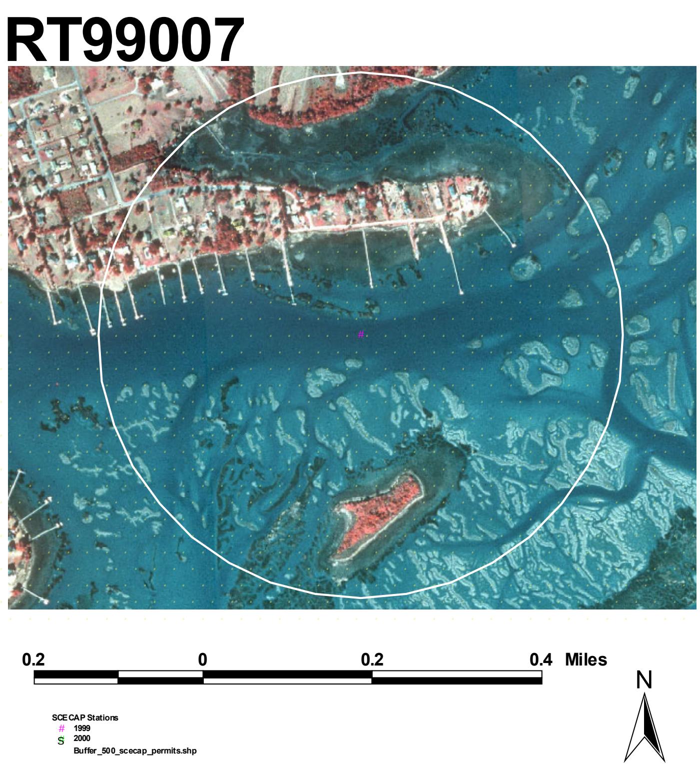
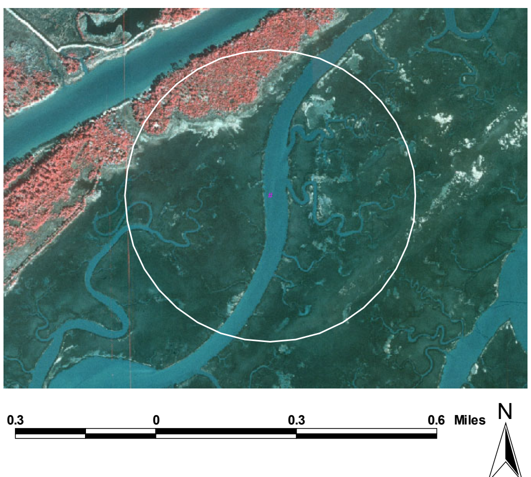


Figure 5. A 1994 NAPP DOQQ aerial photograph of SCECAP station RT99007 with the hexagonal grid and 500-m radius buffer overlaid.

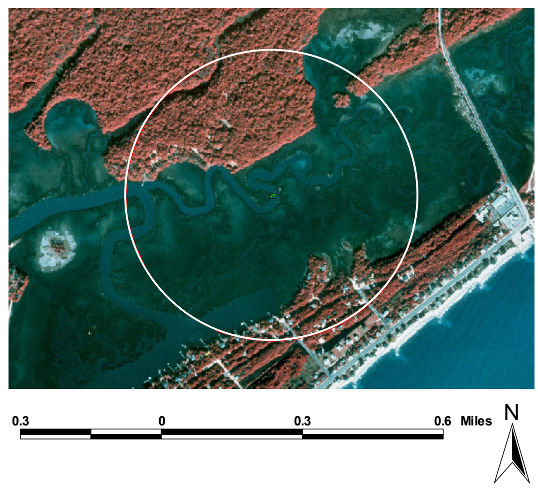
To estimate the number of docks that occurred within the 500-m radius buffer, the number of docks on the aerial photographs was counted and the number of dock permits in an SCDHEC-OCRM ArcView shapefile were identified and counted. The SCDHEC-OCRM shapefile provided locations of permits that had been submitted and approved since 1990. The file, however, did not always contain information about whether the permit was for a dock or another type of structure (e.g., bulkheads) or if the dock had been constructed. Therefore, each permit within the buffer was checked against the SCDHEC-OCRM permit file to determine if the permit was for a dock or another type of structure, if that dock had been inspected by SCDHEC-OCRM, and if a construction placard had been obtained. If an inspection showed the dock was constructed or if the property owner had obtained a construction placard, then the dock was considered to exist. The number of visible docks from the aerial photographs and the number of permits that indicated dock construction had occurred were added to estimate the number of docks existing within the buffer.

SCECAP sites were classified into three categories or treatments based on the number of docks within the buffer: (1) sites with no docks; (2) sites with ≤ 6 docks; and (3) sites with ≥ 7 docks. This resulted in eight samples in the no dock category, eight samples in the low dock category, and 10 samples in the high dock category. Figure 6 presents examples of sample sites for the no dock, low dock, and high dock categories, respectively.

a **RT99026**



b **RT00523**



c **RT99005**

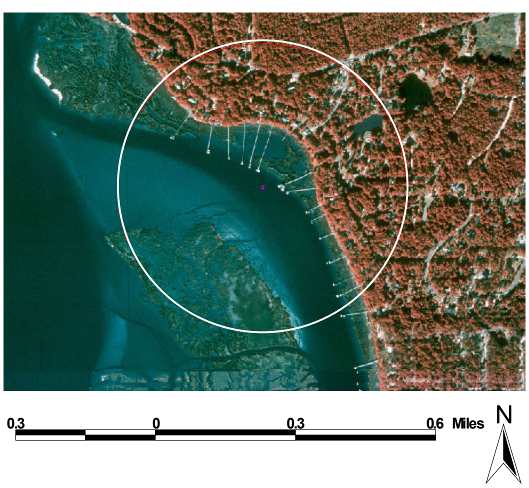


Figure 6. Examples of representative no dock (a), low dock (b), and high dock (c) SCECAP stations on 1994 NAPP DOQQ aerial photographs with the 500-m radius buffer overlaid.

2. Sample Collection and Processing

The sample collection and processing methods provided by Van Dolah *et al.* (in press) for SCECAP are summarized below.

a. Water Quality Samples

Water quality measurements and samples were collected at all stations prior to the deployment of gears to collect sediment and biological samples. Instantaneous measurements included near-surface, mid-depth and near-bottom measurements of dissolved oxygen, salinity, and temperature using a Yellow Springs Instrument (YSI) Inc. Model 85 water quality meter and near-surface measurements of pH were made using a pHep® 3 field microprocessor meter. The near-surface measurements were collected approximately 0.3 meters below the surface and the near-bottom measurements were collected from approximately 0.3 meters above the bottom. Time-profile measurements of the four parameters were obtained from the near-bottom waters of each site using either YSI Model 6920 multiprobe datasonde or a Hydrolab Datasonde 3 (DS3) or 4 (DS4). Datasondes were set to make measurements at 15 min intervals for a minimum of 25 hrs.

Secchi disk readings were collected beginning in 2000. All readings were taken to the nearest 0.1 m using a solid white disk. Measurement protocols were standardized to reduce or eliminate readings that may be affected by glare or surface wave chop.

During 1999, SCDNR staff collected all water quality samples. Dissolved nutrient samples were not collected. Water quality samples were delivered to the SCDHEC Charleston Laboratory for processing and distribution. Beginning in 2000, SCDHEC staff was responsible for collecting water quality samples. These samples were generally collected on the same day as the biological samples, but sampling was not always conducted at the same tidal stage. SCDHEC staff collected additional instantaneous measures of water temperature, salinity, DO and pH at the depth levels of each sample using a YSI 6920 multiprobe and/or a Hydrolab DS4 datasonde.

Water quality samples collected in 2000 for SCECAP included measures of near-surface concentrations of total nitrate/nitrite nitrogen, total Kjeldahl nitrogen, ammonia, total phosphorus, total organic carbon (TOC), total alkalinity, dissolved nutrients (NH_4, NO_3/NO_2, PO_4), dissolved organic nitrogen, dissolved organic phosphorous, dissolved organic and inorganic carbon, dissolved silicon/silica, turbidity, biological oxygen demand (BOD5), and fecal coliform bacteria concentration. All samples were collected by inserting pre-cleaned water bottles to a depth of 0.3 m inverted and then filling the bottle at that depth. The bottles were stored on ice until they were returned to the laboratory for further preservation (nutrient and TOC samples) and processing. Sampling protocols follow standards described by SCDHEC (1997a) and/or SCECAP Quality Assurance Program Plan (Van Dolah unpublished). The dissolved nutrient samples were filtered through 0.45-mm pore cellulose acetate filters prior to preserving and delivering them to the processing laboratory.

Laboratory processing of total nutrient samples, TOC, turbidity, BOD5 and fecal coliform bacteria was completed at SCDHEC laboratories using standardized procedures described by SCDHEC (1997b, 1998). Dissolved nutrients were processed by the University of South Carolina using a Technicon AutoAnalyzer and standardized procedures described by Lewitus *et al.* (1999). Dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC) were measured using a Shimadzu TOC 500, and dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) were calculated by subtracting total inorganic from total dissolved N or P, measured using the persulfate oxidation technique (D'Elia *et al.* 1977).

b. ***Biological and Sediment Quality Samples***

Estimates of phytoplankton biomass were based on chlorophyll measurements. In 1999, two 50-ml samples of water were collected approximately 0.3 m below the surface. Following agitation to homogenize the sample, 50 ml were removed using a syringe and filtered through a Whatman GFC filter to concentrate the sample. The filter was immediately placed in a labeled centrifuge tube with 25 ml of acetone with $MgCO_3$ and stored on ice in the dark for transfer to the laboratory. Upon return to the lab, the tubes with filters were stored in a freezer until processed. Chlorophyll-a sample processing occurred within 48 hours of collection. After this extraction period, the samples were centrifuged and the supernatant quantified using a Turner Model 10-AU fluorometer.

Eight to ten grab samples were collected at each site from an anchored boat using a stainless steel 0.04 m^2 Young grab sampler to evaluate sediment characteristics, sediment contaminant levels, and benthic community composition. The boat was repositioned between each sample to ensure that the same bottom area was not sampled twice and to spread the samples over 10-20 m^2 of bottom area. The grab was cleaned prior to field sampling and rinsed with isopropyl alcohol and seawater between stations.

Three of the grab samples were used for analysis of benthic community composition. These samples were washed through a 0.5-mm sieve to collect the benthic fauna and preserved in a 10% buffered formalin-seawater solution containing rose bengal stain. The remaining grab samples were combined to form a composite sediment sample for analysis of sediment composition, chemical contaminants, and sediment toxicity. Only the surficial sediments (upper 5 cm) were collected from these grabs. The composite sample was thoroughly stirred and subdivided into separate containers for use in sediment bioassays (amphipod, seed clam, microtox tests), sediment characterization analyses (particle size, total organic carbon), porewater analysis (ammonia), and analyses of sediment contaminants (metals, organics). The composite samples were kept on ice until returned to the laboratory, and then stored either at 4^oC (toxicity, porewater) or frozen (contaminants, sediment composition, TOC) until they were processed.

Particle size analyses were performed using a modification of the pipette method described by Plumb (1981). The percentage of sand was determined by separation through a 63-µm sieve. Silt/clay content was determined through timed pipette extractions. Pore water ammonia was measured using a Hach Model 700 colorimeter, and TOC was measured on a Perkin Elmer Model 2400 CHNS Analyzer.

The NOAA National Ocean Service (NOS) Charleston Laboratory processed the sediment contaminant samples using the following protocols. Sample extraction and preparation for organic contaminants used methods described by Krahn *et al.* (1988) and Fortner *et al.* (1996). Samples were extracted with CH_2Cl_2 using accelerated solvent extraction (ASE), concentrated by nitrogen blow-down, and cleaned by gel permeation chromotography where necessary. PAHs were quantified by capillary gas chromatography - ion trap mass spectrophotometry (ITMS). PCBs were analyzed using dual column gas chromatography with electron capture detection (GC-ECD) using methods described by Kucklick *et al.* (1997). Trace metals were analyzed using methods described by Long *et al.* (1997) using inductively coupled plasma spectroscopy (ICP) for aluminum, arsenic, cadmium, chromium, copper, iron, lead, manganese nickel, tin, and zinc, and by graphite furnace atomic absorption for silver, arsenic, cadmium, lead, and selenium. Mercury was analyzed by cold-vapor atomic absorption.

Sediment toxicity was determined using multiple bioassays. In 1999, these included the Microtox assay and a 7-day whole sediment seed clam assay. In 2000, a 10-day whole-sediment amphipod assay was added. The Microtox assay utilizes the photoluminescent bacterium, *Vibrio fischeri*, to provide a sublethal toxicity measure. This assay is based on the attenuation of light production by the bacterial cells following exposure to sediments. Solid-Phase Microtox assays follow the protocols described by the Microbics Corporation (1992). The seed clam assay involved exposing juvenile clams, *Mercenaria mercenaria*, to sediments for seven days using protocols described by Ringwood and Keppler (1998). Seed clam toxicity was measured using both sublethal (growth rate) and lethal end points. The amphipod assay involved exposing *Ampelisca abdita* to sediments over a 10-day period using standard methods described by ASTM (1993).

Benthic samples were sorted in the laboratory. All organisms removed from samples were identified to the lowest taxonomic level possible. Two benthic indices were used, including one defined by Van Dolah *et al.* (1999) for the Carolinian Province and one developed (Van Dolah unpublished) specifically for South Carolina waters.

Fish and large crustaceans (primarily penaeid shrimp and blue crabs) were collected at each site following the benthic sampling to evaluate community composition. Two replicate tows were made at each site using a 4-seam trawl (18-ft foot rope, 15-ft head rope and 0.75-in bar mesh throughout). Trawl tow lengths were standardized to 0.25 km. Tows were made during daylight hours with the current when the marsh was not flooded. Catches were sorted to the lowest practical taxonomic level and counted. All organisms were measured to the nearest 0.1 cm and weighed to the nearest 0.1 kg. When more than 25 individuals of a species were collected, the species was subsampled for length and weight measurements.

D. Statistical Analyses for the Tidal Creek Studies

The small and large tidal creek data were analyzed using a range of parametric and nonparametric statistical tests and graphical summaries. The statistical analyses included analysis of variance/analysis of covariance (ANOVA/ANCOVAs), Shapiro-Wilks normality tests, linear regressions, the Kruskal-Wallis rank sum test, and the Friedman's test. All tests were conducted using SAS Windows Version 8.

Parametric statistics assume the samples were collected from normally distributed populations. Normality was evaluated using a Shapiro-Wilks test. The evaluation of normality, however, was frequently problematic for the environmental data collected for this study. For example, to evaluate normality when using an ANOVA model, the appropriate test should be conducted for each of the defined categories. The categories from small and large tidal creeks were, however, composed of too few samples to conduct a reliable test for normality. Therefore, the normality assumption was tested on untransformed data from the entire data set for each parameter. Normality was rarely achieved using the untransformed data for any of the parameters. Therefore, the data were transformed using a log base 10 or an arcsine square root transformation. Tables 2-3 and Tables 4-6 provide a summary of the transformations made and the Shapiro-Wilks Normality Tests results for the water quality, sediment quality and biological quality data for the Small Tidal Creek Study and Large Tidal Creek Study, respectively. To interpret these tables, the closer the W statistic is to 1 the closer the data approximate normality (SAS 1987). The p-value in the tables is an estimate of the probability that the data were normally distributed. P-values close to zero indicate the data were not normally distributed.

Most of the data, especially the PAH, benthic community, and fish/crustacean community data, were not normally distributed. Because nonparametric statistical methods generally cannot account for the effects of covariates and require large sample numbers to conduct reliable tests, we analyzed these data using both parametric and nonparametric methods. The results of the parametric analyses (i.e., ANOVA/ANCOVAs) are presented first followed by the results of the equivalent nonparametric analysis (i.e., Kruskal-Wallis rank sum test). The results of the parametric and nonparametric statistical tests were generally similar demonstrating the robustness of the ANOVA/ANCOVA to overcome a failure to meet the normality assumption.

ANOVA/ANCOVAs and the nonparametric Kruskal-Wallis statistical tests were used to test whether the environmental conditions represented by the measured parameters were affected by the category. The decision on whether an ANOVA or ANCOVA was conducted was based on the degree to which site-to-site differences in natural environmental conditions influenced measured parameters. Environmental conditions that were evaluated as potential covariates included: (1) sediment composition and salinity for benthic community measures; (2) the clay content of the sediment for sediment metal concentrations; (3) the total organic carbon (TOC) content of the sediment for sediment PAH and PCB concentrations, and (4) the sediment composition and salinity for fish and crustacean community characteristics. If the covariates identified were not significant ($p < 0.15$), then they were not included in the analysis. The Kruskal-Wallis test was also used to determine if using nonparametric statistics would provide similar results.

Least squares regressions were used to investigate if the dock density in small and large tidal creeks was correlated with parameter values. Contaminant concentrations at each site were also compared to a level at which biological effects are reported to occur as defined by Long *et al.* (1995).

In order to determine if sediment concentrations for dock-associated contaminants including arsenic, chromium, copper, and the 24 PAH analytes were cumulatively different across categories, a Friedman's test was performed. The goal of this test is to determine if the ranking (e.g., highest concentrations) of each contaminant occured in one category (e.g., suburban – dock) more than another category. This test ranks each parameter among the categories from the highest to the lowest value. These values were then summed down the rows and a mean score was calculated for each category. If the row scores were different than what would be predicted when no association occurs, then the model was determined to be significant ($p < 0.05$) and a significant difference in the cumulative contaminant exposure was defined to occur.

III. RESULTS

A. *Spartina* Shading Study

The McGuire (1990) Virginia Study examined docks along a fringe marsh (i.e., short expanses of marsh). A range of dock lengths (approximately 15 m to 20 m), widths (0.6 m to 2.4 m), heights (0.1 m to 1.6 m), and geographical orientations (4^o to 358^o) were sampled. Calculations made from the data collected by McGuire (1990) found *Spartina alterniflora* stem densities under docks were reduced by 65% compared to the stem densities adjacent to docks.

The *S. alterniflora* Shading Study we conducted also examined a range of docks representing varying lengths, widths, heights, spaces between planks, and geographical orientations (Table 7). Twenty-one docks were oriented in a North-South direction and eleven were oriented in an East-West direction. The short growth form of *S. alterniflora* averaged approximately 100 stems per 0.1 m^2. The tall growth form density averaged approximately 20 stems per 0.1 m^2. The density of *Spartina alterniflora* under docks was significantly lower than that which occurred in the marsh 5 m from docks for the short-form ($p < 0.0001$), tall-form ($p < 0.0001$), and both forms combined ($p < 0.0001$) based upon paired t-tests. The average difference in short, tall, and both forms combined was 49.8, 21.0, and 33.4 stems per 0.1 m^2, respectively. The reduction in *S. alterniflora* density due to shading was not significantly different between docks orientated in the North-South and East-West directions ($p = 0.6928$). The tall- and short-form data were combined to estimate the average percent reduction in *S. alterniflora* density. The average reduction in stem densities under South Carolina docks was estimated to be 71%.

Several visual observations about the effects of docks on *S. alterniflora* were made during the SC *Spartina* Shading Study.

- The *S. alterniflora* under docks was often taller than that adjacent to docks, especially near the uplands (Figure 7). This observation was often associated with large amounts of bird feces on the dock platform (Figure 8). In rural areas, wading birds apparently used some docks as resting sites.

Figure 7. A photograph of tall-form *Spartina alterniflora* growing under a dock. The *S. alterniflora* adjacent to the dock was entirely short-form. This dock was located on Long Creek.

Figure 8. A photograph of bird fecal matter on the dock shown in Figure 7.

- One dock was not included in the statistical analysis for the SC *Spartina* Shading Study because plywood boards had been placed on the marsh surface for most of the length of the dock (Figure 9). Boats were also frequently observed to be moored on the marsh surface adjacent to docks, especially for docks located on narrow creeks (Figure 10). These observations suggest that shading may not be the only dock-related activities that may be affecting *Spartina alterniflora* productivity.

Figure 9. A photograph of plywood on the marsh surface under the length of the dock.

Figure 10. A photograph of a boat moored on the salt marsh which has resulted in a reduction in *S. alterniflora* densities adjacent to the dock structure.

- Wrack and trash accumulated under two of the thirty-two docks sampled (Figure 11). Wrack and trash were not found at any sites sampled 5 m from the dock; however, they were frequently observed to accumulate on the marsh surface in other areas (Figure 12).

Figure 11. A photograph of wrack and trash accumulation taken under a dock sampled on Long Creek.

Figure 12. A photograph of wrack and trash accumulation on the marsh surface near James Island Creek.

- Construction damage associated with the building of the dock structure was examined at one dock on James Island. This dock was constructed in the summer and fall of 2000. Photographs were taken on August 7, 2000, which documented almost complete elimination of *Spartina alterniflora* along the sides of the dock (Figures 13a and 14a). This dock was re-photographed on November 27, 2001. The November 2001 photograph indicated *S. alterniflora* had recolonized the area adjacent to the dock, suggesting substantial recovery had occurred in 15 months (Figures 13b and 14b). Similar observations were made for other newly constructed docks in Long Creek.

Figure 13. Photographs of the area adjacent to a dock during the construction phase (a) and approximately one year after the construction (b). The photograph during construction was taken on August 7, 2000 and is to the right of the dock walkway. The photograph post-construction was taken on November 27, 2001 and is to the left of the dock walkway.

Figure 14. Photographs of construction effects on *Spartina alterniflora* during construction (a) and post-construction (b) were taken on August 7, 2000 and November 27, 2001, respectively.

The shading effect from dock structures on *Spartina alterniflora* was evaluated at the local (i.e., small tidal creek), county, and state scales. The estimated percent reduction in *S. alterniflora* stem density at the local scale ranged from 0.03 to 0.72% at 1999 dock numbers (Table 8). If the study creeks had been developed in a manner that would maximize the number of waterfront properties, and every land owner chose to build a dock, then the projected reduction in *S. alterniflora* stem density would have ranged from 0.18 to 5.45%. This projection assumes the average dock length that currently occurs in each creek is likely to be maintained into the future (Table 9). We recognize that this projection is not realistic for the study creeks as existing covenants, land use plans, and other laws would prevent this level of development from occurring along their banks; however, most new developments are designed to maximize the number of waterfront properties. The projection, therefore, provides insight into the potential magnitude of shading effects that could theoretically occur at the scale of small tidal creeks.

Table 10 provides the number of docks permitted by SCDHEC-OCRM in each county from 1991 to 2000. The reduction in *Spartina alterniflora* as estimated by stem density, when considered at the scale of the eight coastal counties in 2000 ranged from 0.00 to 0.08% for the average dock length observed in small creeks (i.e., 25 m) to 0.01 to 0.98% for the maximum dock length permitted (i.e., 305 m) by current regulations (Table 11). Shading impacts associated with the projected 2010 dock levels ranged from 0.00 to 0.16% for the average dock length observed in small creeks (i.e., 25 m) to 0.03 to 1.98% for the maximum dock length permitted (i.e., 305 m) by current regulations. The largest shading effect occurred in Horry and Dorchester counties. Both of these counties are characterized by relatively small amounts of salt marsh habitat (Table 11). The smallest shading effect occurred in Jasper and Colleton counties. These counties have modest amounts of salt marsh habitat and relatively low numbers of docks (Table 11).

The state of South Carolina has approximately 1,495 million m^2 (369,500 acres) of salt marsh in the eight coastal counties (Tiner 1977). This number includes low and high salt marsh as well as the brackish water marshes. This is a very large number compared to the area under the 7,000 docks that were estimated to exist in the state in 2000 or the estimated 14,000 docks that are projected to exist in the state in 2010 if docks continue to be permitted at the current rate of approximately 700 dock permits per year. A series of dock lengths were used to estimate the amount of reduction in *S. alterniflora* stem density that would occur if the statewide dock length averaged 25 m, 50 m, 100 m, 150 m, 200 m, 250 m, or 300 m. In 2000, the reduction in *S. alterniflora* stem densities was estimated to range from 0.01% at 25 m dock lengths up to 0.12% at 300 m dock lengths (Table 12). In 2010, the projected reduction in *S. alterniflora* ranged from 0.02 to 0.24% of the total salt marsh area in the state (Table 12).

B. Small Tidal Creek Study

1. Land Use and Dock Numbers

The average number of docks was 0, 0, and 16, respectively, for the reference, suburban – no dock and suburban - dock categories in the small tidal creeks (Table 13). The maximum number of docks within a small tidal creek watershed was 32 (Parrot Creek on James Island, SC). The average watershed size or drainage basin for the small tidal creeks studied was 100, 370, and 200 hectares for the reference, suburban – no dock, and suburban – dock categories, respectively (Table 13). The average percentage of the watershed characterized by agriculture, barren, forest, urban, water, and salt marsh land cover were different between the reference and suburban creeks. The suburban creek categories were, however, similar in the land cover characteristics (Table 13). The average amount of impervious surface was similar between the two suburban categories and approximately 30 times higher than that which occurred in the reference category (Table 13). Table 13 and Figure 15 summarize the land cover data for each creek. Table 13 also provides data on dock abundance for each creek. The number of docks within the small tidal creeks studied was positively correlated (r^2 = 0.23, p = 0.0105) with the amount of impervious surface in the watershed (Figure 16). This finding suggests the abundance of docks in small tidal creeks was linked to the degree of human development in the adjacent uplands at the scale of small tidal creeks.

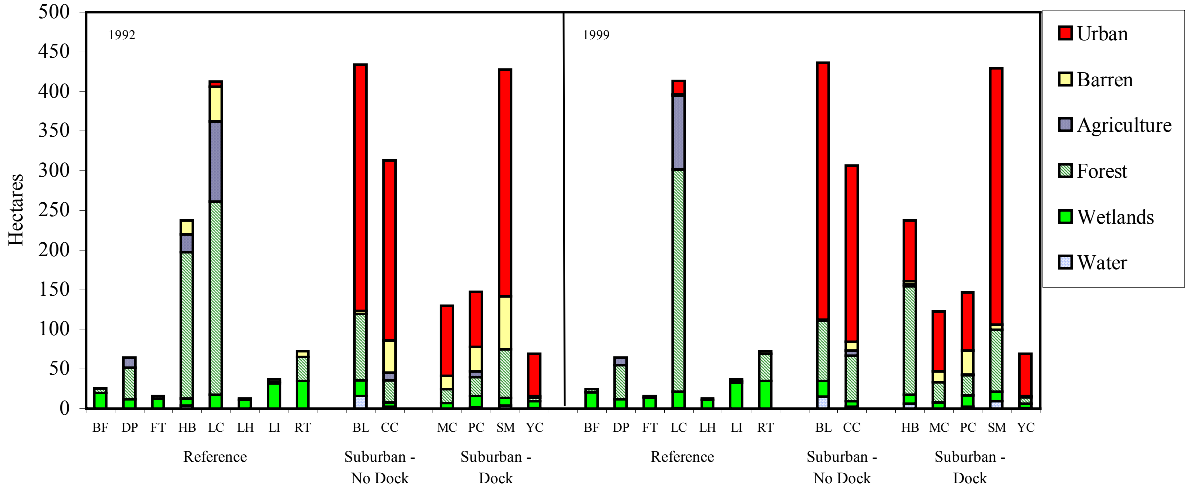


Figure 15. The land cover in each creek watershed for the 1992 and 1999 TCP small tidal creek data set. The creek name is abbreviated along the x-axis and designations can be found in Figure 3.

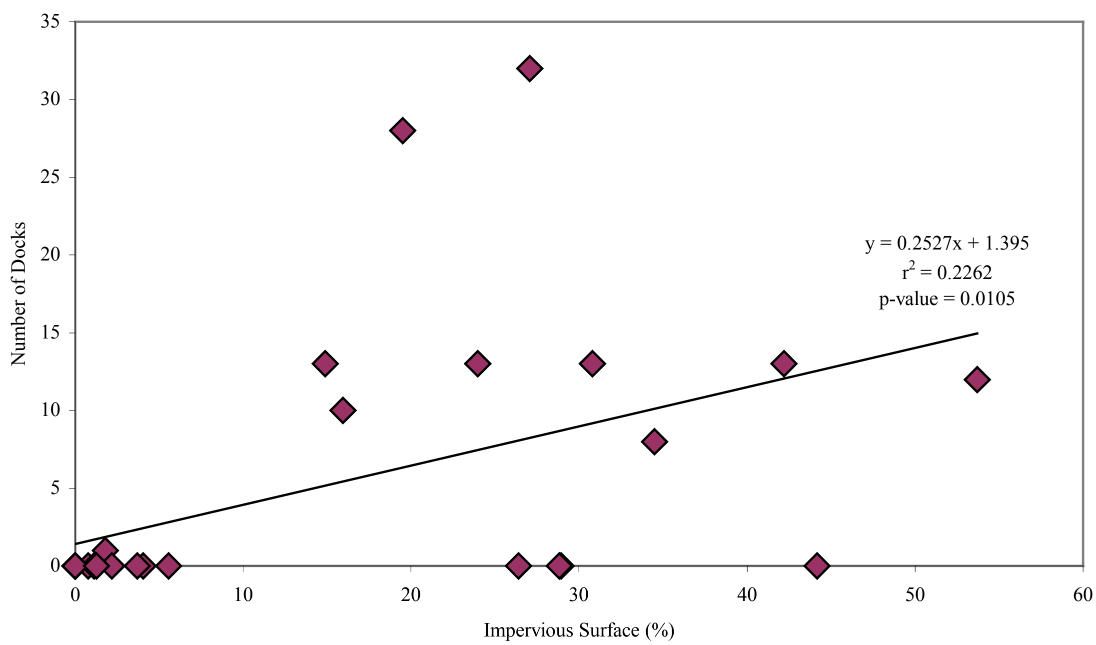


Figure 16. The data points and regression line showing the association between the amount of impervious surface and the number of docks in each creek watershed for the 1992 and 1999 TCP small tidal creek data set.

2. Sediment Quality

The TCP sediment quality data set includes 14 metal analytes, 24 PAH analytes, a low molecular weight PAH (mostly uncombusted hydrocarbons) composite value, a high molecular weight PAH (mostly combustion products) composite value, a total PAH composite value, a total PCB composite value, and five sediment composition parameters. The sediment quality parameters that would be expected to be associated with the docks and their associated uses include chromium, copper, arsenic, the low molecular weight PAH value, the high molecular weight PAH value, and the total PAH value.

a. Trace Metal Analytes

The amount of clay in the sediment may influence sediment trace metal concentrations because clay particles are charged and metals adsorb to them (Olsen *et al.* 1982, Luoma 1989). All of the trace metals except silver had a significant clay composition covariate that accounted for most of the variance among categories (Table 14). The category effects (i.e., reference, suburban – no dock, and suburban – dock) were significant for three (cadmium, mercury, and lead) of the 14 metals. Pairwise contrasts indicated that the abundance of docks did not make a contribution to the category effects for these three metals. Pairwise contrasts also identified significant category differences for chromium, zinc, and silver (Table 14). The highest concentration of arsenic, chromium, and copper consistently occurred in the suburban – dock category and lowest in the suburban – no dock or reference categories (Figure 17). Because the variance within these categories was large, the differences among these categories were not significant individually.

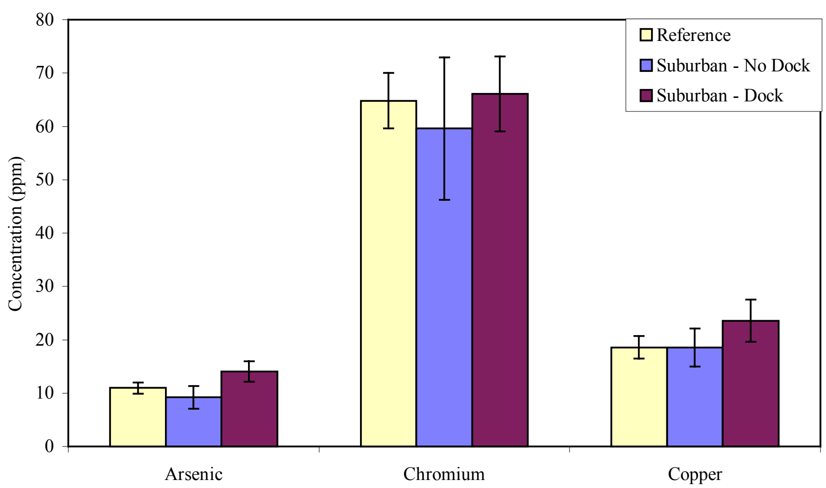


Figure 17. The average concentration and one standard error of arsenic, chromium, and copper for each treatment category evaluated for the TCP data set.

The Kruskal-Wallis rank sum test found no significant differences among categories for any of the fourteen trace metals (Table 15). This result is similar to that reported for the parametric analyses which identified few category effects. None of the metals had a distributional pattern that was correlated with the number of docks (Table 16).

Sediment arsenic concentrations in the southeast are naturally high and frequently exceed levels reported to cause biological harm (Sanger 1998). The concentrations of arsenic exceeded a level where biological effects as defined by Long *et al.* (1995) were likely to occur for 13 of the 16 samples (81%) from the reference category, two of four samples (50%) for the suburban – no dock category, and seven of the eight samples (88%) from the suburban - dock category. The concentrations of chromium exceeded a level where biological effects may occur for five of the 16 samples (31%) from the reference category, one of four samples (25%) from the suburban – no dock category, and one of eight samples (13%) from the suburban – dock category. Sediment concentrations of copper exceeded a level where biological effects may occur for one site in the suburban – dock category. Silver, cadmium, mercury, lead and zinc concentrations did not exceed a level where biological effects were likely to occur (Long *et al.* 1995).

The above findings suggest that the presence of dock structures had little measurable effect on sediment trace metal concentrations at the scale of small tidal creeks. Natural processes and the kinds and degree of development in the watershed appeared to be the major factor associated with sediment metal concentrations. Some of the sediment metal values approached levels that are reported to be associated with biological effects. These threshold values, however, occurred as frequently in reference areas without docks as they did in small tidal creeks with docks.

b. Polycyclic Aromatic Hydrocarbon Analytes

Because many organic contaminants (e.g., PAHs) adsorb to organic carbon particles in the water or sediment, the total organic carbon (TOC) in the sediment may influence sediment PAH concentrations (e.g., Boehm and Farrington 1984, Barrick and Prahl 1987). The TOC covariate was significant ($p < 0.15$) for 17 of the 24 PAH analytes including the composite PAH measures. The ANOVA/ANCOVA models differentiated ($p < 0.05$) among categories (i.e., reference, suburban – no dock, suburban – dock) for all the PAH analytes except 1-methylnapthalene, 2,6-dimethylnapthalene, 2-methylnapthelene, biphenyl, napthalene, 2,3,5-trimethylnapthalene, acenaphthylene, and dibenzoanthracene (Table 14). The highest concentration for 20 of the 27 PAH parameters including the low molecular weight PAH value, the high molecular weight PAH value, and the total PAH value was in the suburban – dock category followed by the suburban – no dock category and the reference category. For 18 of these parameters, the PAH parameter value was significantly higher in the suburban – dock category compared to the reference category (Table 14). In addition, 12 of these PAH parameters were significantly higher in the suburban – no dock category compared to the reference category. Only anthracene and the LMW PAH value was significantly higher in the suburban – dock category compared to the suburban – no dock category (Table 14).

The Kruskal-Wallis rank sum test found 19 of the 27 PAH metrics had significant category effects (Table 15). The reference category always had the lowest mean score and the suburban – no dock category or the suburban – dock category had the highest mean score.

Figure 18 summarizes the concentration data of the 27 PAH parameters by category. The lowest concentration for most of the PAH analytes and combined PAH measures occurred in the reference category and the highest concentrations occurred in the suburban – dock category. Two of the suburban – dock sites had sediment concentrations of phenanthrene and all of the high molecular weight PAH analytes that exceeded values which are reported to cause biological harm. These sites were located in Shem Creek. This creek is reported to have extremely high levels of high molecular weight PAHs in both the creek channel and salt marsh (Sanger *et al.* 1999b).

Only two PAH analytes (anthracene and fluorene) and the cumulative PAH metrics were correlated with the number of docks present ($p < 0.05$) (Table 16). Figure 19a presents the correlation between the total PAH value and the number of docks. This association was not significant ($r^2 = 0.03$, $p = 0.415$). A significant correlation ($r^2 = 0.34$, $p = 0.0012$) was, however, observed between the total PAH value and the amount of impervious surface in the associated watersheds (Figure 19b).

The above findings suggest that dock structures had small effects on sediment PAH concentrations at the scale of small tidal creeks. Sediment PAH concentrations were probably more affected by human development activities that occurred in the watershed than by the abundance of docks. Some PAH analytes that are known to be associated with boating activities, however, consistently attained higher values in the suburban – dock category suggesting boating activities may have contributed to these high concentrations. Few PAH analytes had concentrations (2 of 720) which exceeded levels that are thought to cause biological effects (Long *et al.* 1995).

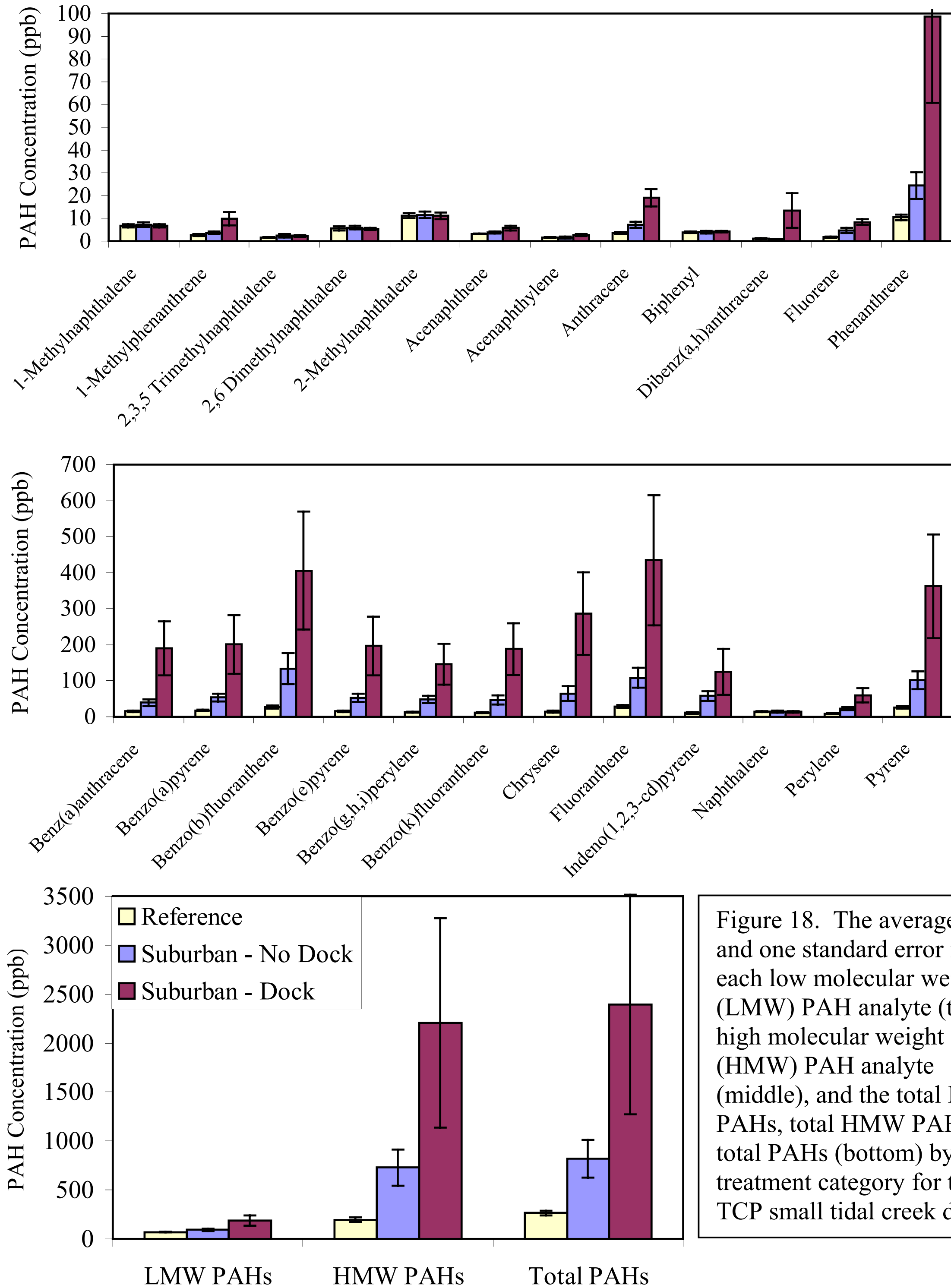


Figure 18. The average value and one standard error for each low molecular weight (LMW) PAH analyte (top), high molecular weight (HMW) PAH analyte (middle), and the total LMW PAHs, total HMW PAHs, and total PAHs (bottom) by treatment category for the TCP small tidal creek data set.

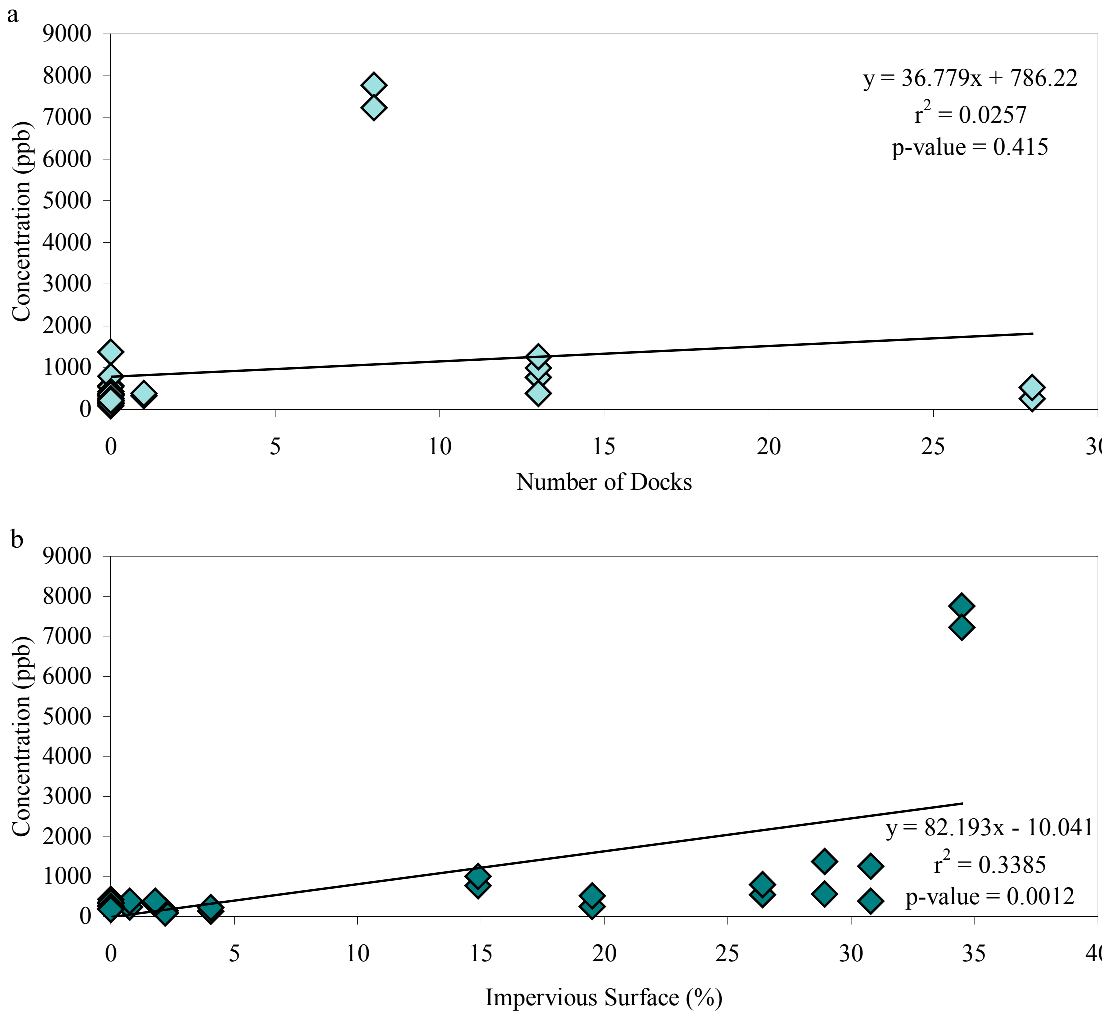


Figure 19. The data points and regression line for the total PAH concentration in the sediment versus the number of docks (a) and percent impervious surface (b) in each watershed in the TCP small tidal creek data set.

c. Trace Metal and Polycyclic Aromatic Hydrocarbon Analytes Combined

A Friedman's test was conducted to evaluate if the cumulative concentrations of arsenic, chromium, copper and the 24 PAH analytes varied among categories in a manner that would suggest docks were a contributing factor. This test found significant ($p<0.0001$) category effects with the highest concentrations occurring in the suburban – dock category followed by the suburban – no dock and the reference categories. This finding suggests the cumulative distribution of dock-associated contaminants was not the result of random processes. The density of docks is a likely contributor to the higher concentrations of dock-associated contaminants in creeks with high numbers of docks at the scale of small tidal creeks.

d. Total Polychlorinated Biphenyls

Because organic contaminants (e.g., PCBs) adsorb to organic carbon particles in the water and sediment, the total organic carbon (TOC) in the sediment may influence sediment PCB concentrations (e.g., Boehm and Farrington 1984, Barrick and Prahl 1987). The TOC covariate was, however, not significant for the total PCB value (Table 14). The ANOVA model was also not significant; however, the pairwise contrasts identified several category differences. PCB concentrations in the suburban – dock category were significantly higher compared to the reference category with the suburban – no dock category having an intermediate value that was similar to both (Table 14). The Kruskal-Wallis rank sum test also found no significant difference among categories for the total PCB measure (Table 15). The total PCB metric was not correlated ($p<0.05$) with the number of docks present but was correlated ($r^2 = 0.21$, $p < 0.0145$) with the amount of impervious surface in the associated watersheds (Table 16). One sample site had sediment concentrations of PCBs that exceeded levels reported to cause biological harm. This site was located in the lower reach of Shem Creek.

There is no evidence that dock structures are a source of PCBs. Therefore, the above findings suggest that sediment PCB concentrations appear to be influenced by kinds and degree of human development that occurs in the associated watersheds (Sanger *et al.* 1999b).

e. Sediment Composition

Several sediment composition metrics were evaluated; however, only the results for the clay and TOC content measures are summarized. Analysis results for the other sediment composition metrics were similar to the findings for the clay and TOC content. The clay and TOC content of the sediment were not significantly different among categories in a one-way ANOVA (Table 14). The Kruskal-Wallis rank sum test also found that clay and TOC content were not different among categories (Table 15). The clay and TOC content of the sediments were also not correlated with the number of docks (Table 16). The above suggest that dock structures had no measurable effect on sediment properties at the scale of small tidal creeks.

3. Biological Integrity

a. Benthic Organisms

Benthic organisms live in bottom sediments and are important food items in the diets of juvenile fish and crustaceans (Chao and Musick 1977, Bell and Coull 1978, Holland *et al.* 1989). As such they represent an important link between primary producers and fish, crabs, and shrimp. Because benthic organisms have limited mobility and generally cannot avoid pollution stress, they are frequently used as indicators of biological integrity and environmental quality (Pearson and Rosenberg 1978, Rhodes *et al.* 1978, Boesch and Rosenberg 1981, Holland *et al.* 1987, Lerberg *et al.* 2000).

The effects of docks on benthic (bottom dwelling) organisms were evaluated using the data collected by the TCP in the summer of 1994 and the winter of 2000. Benthic measures that were examined included abundances of eleven species or taxa, the number of taxa per sample or species richness, the total number of organisms in each sample, the proportion of the sample that was stress sensitive taxa (index of stress sensitive organisms), and the proportion of the sample that was stress tolerant taxa (index of stress tolerant organisms).

The silt-clay content of the sediment and the salinity are important natural factors controlling the kinds and abundances of benthic organisms (e.g., Holland *et al.* 1987, Lerberg *et al.* 2000) and were evaluated as potential covariates in the one-way ANCOVA models for the benthic parameters evaluated except for the indices of stress sensitive and stress tolerant taxa (Table 17). The taxa that comprise the stress sensitive and stress tolerant categories were defined by Lerberg *et al.* (2000).

In the 1994 summer study, all of the benthic metrics except *Neanthes succinea, Paranais litoralis*, and species richness demonstrated significant category effects in ANOVA/ANCOVA models; however, the models accounted for little of the variability in the data (Table 17). In general, the suburban – no dock category had the highest abundances for most species. *Tubificoides heterochaetus*, a stress sensitive species, had significantly higher abundances in the reference category compared to the suburban – dock and – no dock categories. *Tubificoides brownae*, a stress tolerant species, displayed the opposite pattern with significantly higher abundances in the suburban – dock and suburban – no dock categories compared to the reference category. *Monopylephorus rubroniveus*, also a stress tolerant species, and the total abundance (strongly influenced by the high abundances of *M. rubroniveus*) were significantly higher in the suburban – dock category compared to the reference and suburban – no dock categories. *Streblospio benedicti, Capitella capitata, Heteromastus filiformis, Laeonereis culveri, Polydora cornuta*, and Tubificidae did not display a distributional pattern among categories that were related to the abundance of docks (e.g., the suburban – dock category was not different than the reference category). The values for the stress sensitive species index were significantly higher in the suburban – no dock and reference categories than the suburban – dock category. The values for the stress tolerant species index were significantly higher in the suburban – dock and suburban – no dock categories than the reference category (Table 17).

In the 2000 winter study, seven of the 15 benthic parameters metrics demonstrated significant category effects in ANOVA/ANCOVA models (Table 17). In general, the suburban – no dock category had the highest abundances for the individual species. *L. culveri, P. cornuta*, and Tubificidae did not display distributional patterns among the categories that were related to dock abundance (e.g., the suburban – dock category was not different from the reference category). *H. filiformis*, *T. heterochaetus* (a stress sensitive species), and the index of stress sensitive species had significantly higher abundances in the reference category compared to the suburban – dock category with the suburban – no dock category similar to both. The species richness was significantly higher in the suburban – no dock and reference categories compared to the suburban – dock category (Table 17).

The Kruskal-Wallis rank sum test results were consistent with all of the ANOVA (8 analyses) and 14 of the 21 ANCOVAs (Tables 17 and 18). When the nonparametric test results were inconsistent with station contrasts for the corresponding ANCOVAs, a relatively strong salinity covariate was used in the ANCOVA model. This observation suggests that adjustments made in the ANCOVAs for natural site-to-site differences in salinity probably contributed to the inconsistencies observed between the nonparametric and parametric tests. Because the ANCOVA results are adjusted for the effects of natural factors on distributional patterns they probably provide the most useful information for evaluating the responses of benthic organisms to the presence of docks.

Figures 20 and 21 provide the average and one standard error of the benthic community parameters among categories for the 1994 and 2000 studies, respectively.

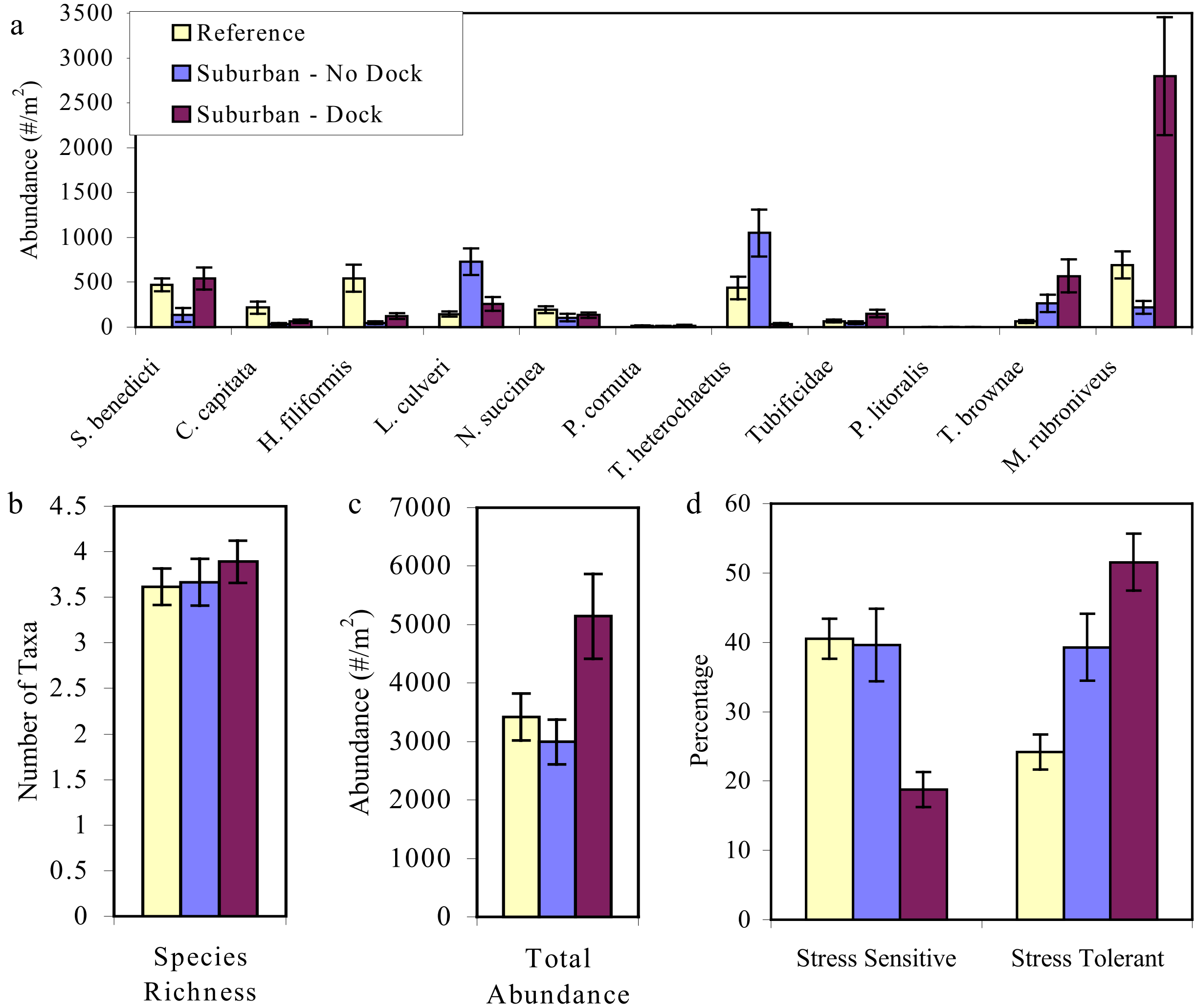


Figure 20. The average value and one standard error for each 1994 TCP benthic taxa abundance (a), species richness (b), total abundance (c), and stress sensitive and tolerant (d) metrics for this study by treatment category.

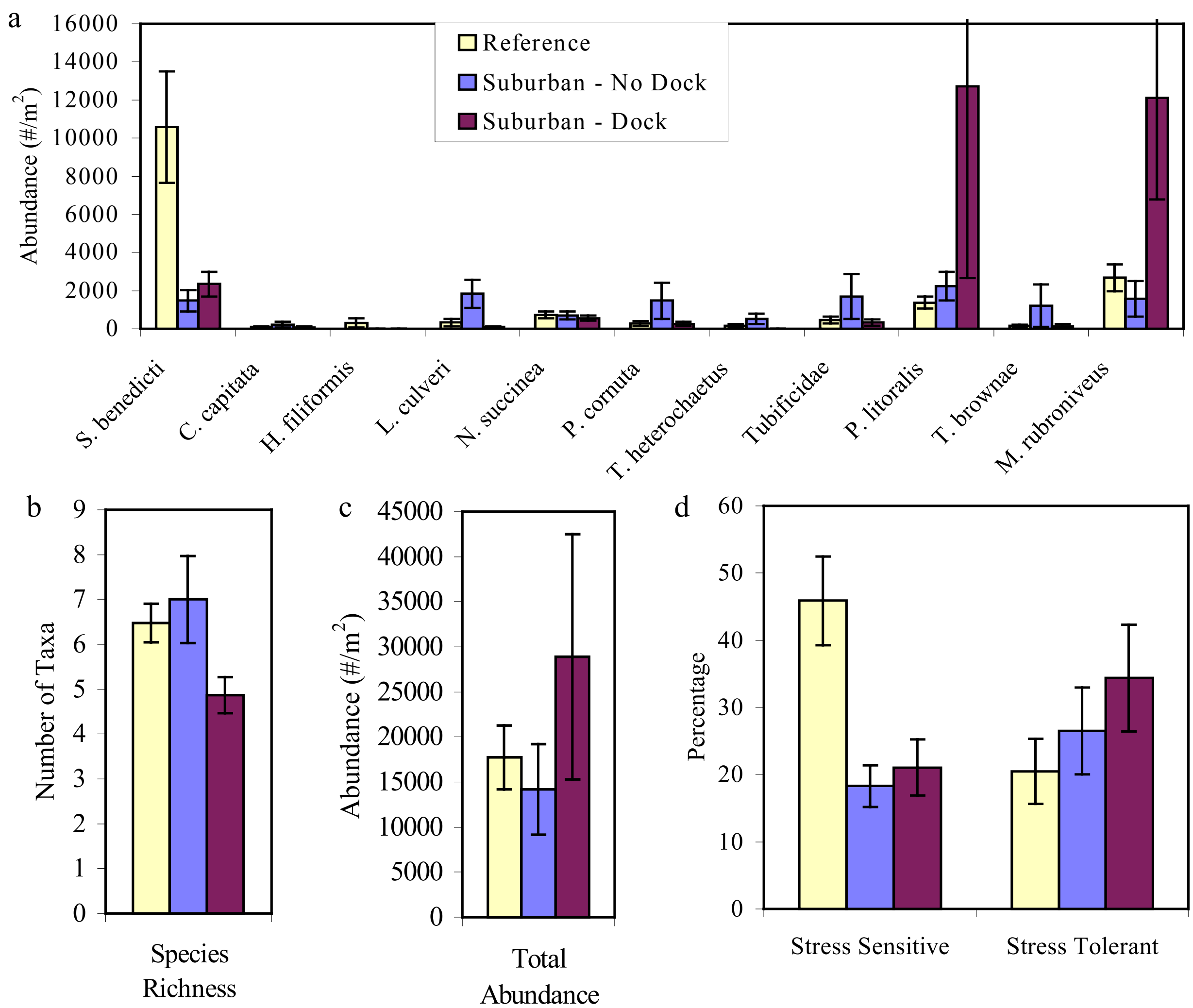


Figure 21. The average value and one standard error for each 2000 TCP benthic taxa abundance (a), species richness (b), total abundance (c), and stress sensitive and tolerant (d) metrics for this study by treatment category.

Ten of the fifteen benthic metrics from the 1994 summer study had significant correlations with the number of docks (Table 19). These regressions have low coefficients of determination (i.e., r^2) indicating they accounted for small amounts (3-8%) of the variability among categories. *Streblospio benedicti*, Tubificidae, *T. brownae, M. rubroniveus*, total abundance, and proportion of stress tolerant taxa had positive slopes indicating that as the number of docks increased, the value of these parameters increased. *Capitella capitata*, *L. culveri, T. heterochaetus*, and the proportion of stress sensitive taxa had negative slopes indicating that as the number of docks increased, the value of these parameters decreased (Table 19). None of the benthic parameters for the winter 2000 TCP study had significant correlations ($p > 0.05$) with the number of docks in each watershed (Table 19).

The above findings suggest that dock structures alone did not adversely affect benthic organisms in small tidal creeks. The indirect effects of human activities associated with suburban development, which includes docks and related changes in environmental conditions, appear to have been the major factors contributing to the observed differences in benthic distributional patterns among categories. The relative abundance of stress sensitive species was adversely affected by suburban development and the relative abundance of stress tolerant species was enhanced by suburban development.

b. Fish and Crustaceans

Fish and crustaceans (i.e., shellfish) have commercial, recreational, and ecological value. Some are harvested; others serve as prey for birds, sport fish, and marine mammals. Unlike benthic organisms, many fish are wide-ranging and have low fidelity for specific sample sites. The value of the kinds and abundances of fish and crustaceans as a measure of environmental quality at a site is therefore questionable.

The small tidal creek fish and crustacean parameters evaluated included abundances and biomass of two fish species (*Anchoa mitchilli* and *Fundulus heteroclitus*), two shrimp taxa (*Paleomonetes* sp. and Penaeidae), the number of taxa per seine, the total number of organisms per seine, and the total biomass per seine (Tables 17, 18, and 19).

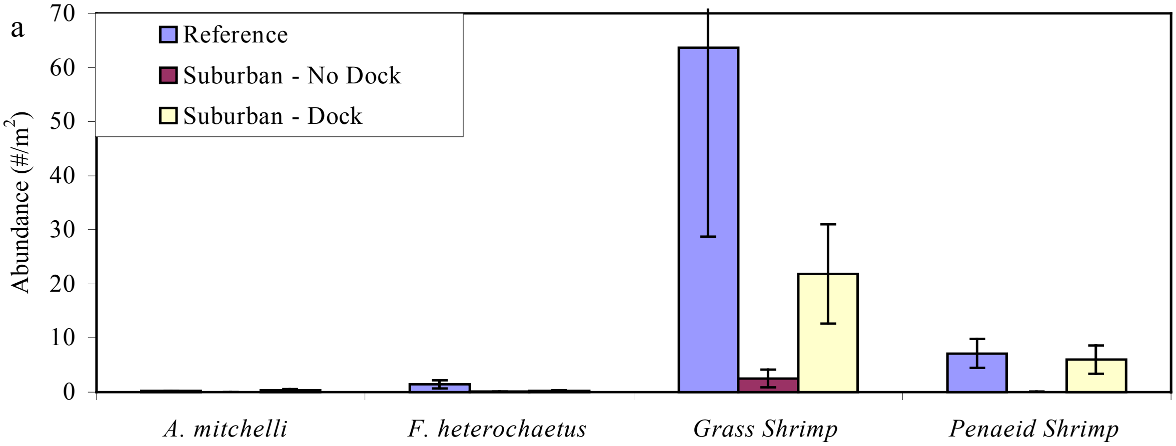
The silt-clay content of the sediment and the salinity are known to be important factors controlling the kinds and abundances of fish and crustaceans (Lippson *et al.* 1979). Salinity was not a significant covariate in any of the one-way ANCOVA models for fish and crustacean parameters. The silt-clay content of the sediments was, however, a significant covariate for three of the fish and crustacean parameters (Table 17). Two of the eleven fish and crustacean metrics evaluated using ANOVA/ANCOVA models had significant category effects (Table 17). The total abundance of fish and crustaceans was significantly higher in the suburban –dock and reference categories than in the suburban – no dock category. The biomass of Penaeid shrimp was significantly higher in the reference category than in the suburban – no dock and suburban – dock category (Table 17).

The Kruskal-Wallis rank sum test found the total abundance and biomass of the Penaeid shrimp had significant category effects with similar patterns to the ANOVA/ANCOVA models (Table 18). In addition, the abundance of the Penaeid shrimp was significantly different among categories. The highest Penaeid abundances were observed in the reference and the suburban – dock categories with the suburban- no dock category characterized by the lowest values.

Figures 22 and 23 show the average and one standard error of the fish and crustacean parameters.

None of the fish and crustacean metrics had distributional patterns that were correlated with the number of docks (Table 19).

The above findings suggest that suburban development may reduce fish and crustacean abundances but the dock structures may potentially mediate the development effect by providing structure that attracts fish and crustaceans.



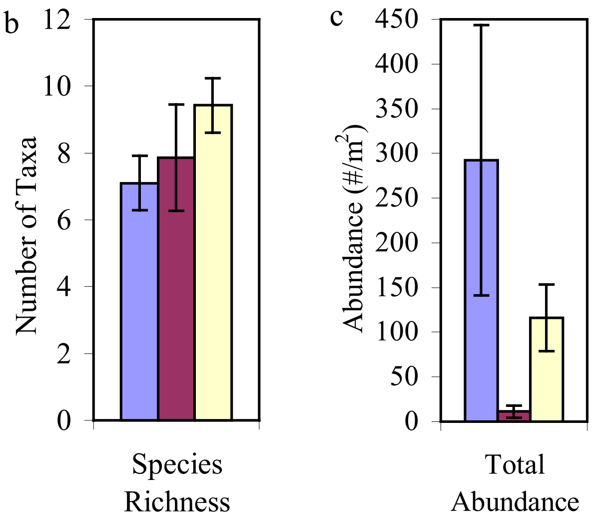


Figure 22. The average value and one one standard error for each 1994 TCP fish and crustacean taxa abundance (a), species richness (b), and total abundance (c) metrics for this study by treatment category.

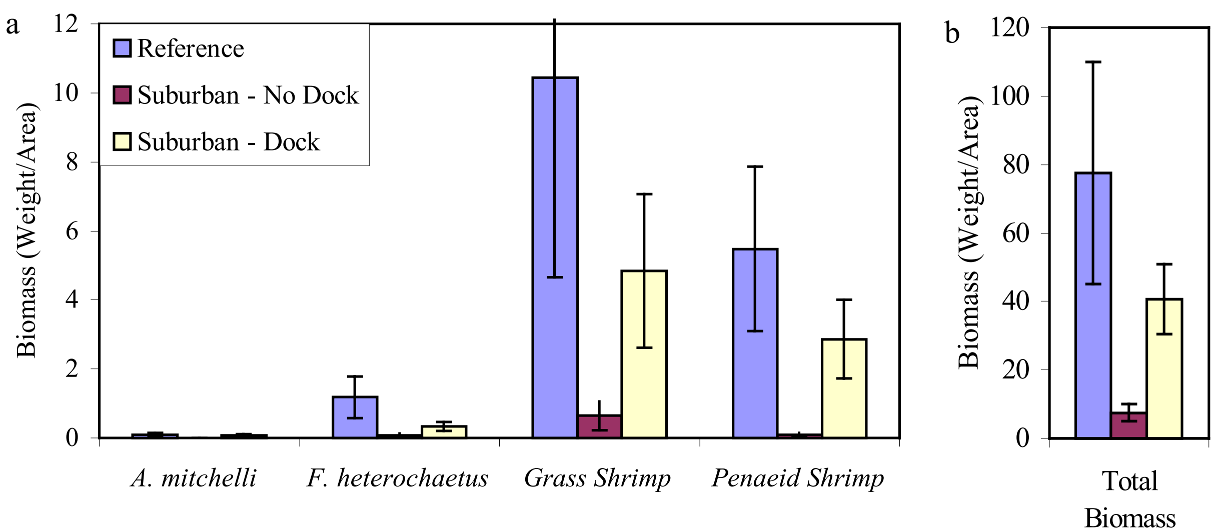


Figure 23. The average value and one standard error for each 1994 TCP fish and crustacean taxa biomass (a) and total biomass (b) metrics for this study by treatment category.

C. Large Tidal Creek Study

1. Land Use and Dock Numbers

The average width of large tidal creeks ranged from 54 m (178 ft) to 76 m (247 ft) with the average creek width increasing from the no dock to the high dock category (Table 20). The average number of docks was 0, 3, and 26, respectively, for the no, low, and high dock categories (Table 20). The maximum number of docks within a 500-m radius buffer was 87 (Table 20). This site was located in the Myrtle Beach area. The average percentage of the buffer characterized by water, salt marsh, and upland land cover was similar among the three categories (Table 20). The average amount of forested upland area was highest in the sites with no docks (92.7%) and lowest in the sites with high numbers of docks (57.3%). The amount of impervious surface was highest in the high dock category (18.3%) and lowest (1.7%) in the no dock category (Figure 24). The low dock category was characterized by intermediate (4.6%) amounts of impervious surface (Table 20, Figure 24). The number of docks within the buffer was positively correlated ($r^2 = 0.82$, $p < 0.0001$) with the amount of impervious surface (Figure 25). This finding suggests that the responses of parameters evaluated with respect to dock structures may be obscured by responses to other human activities associated with suburban development.

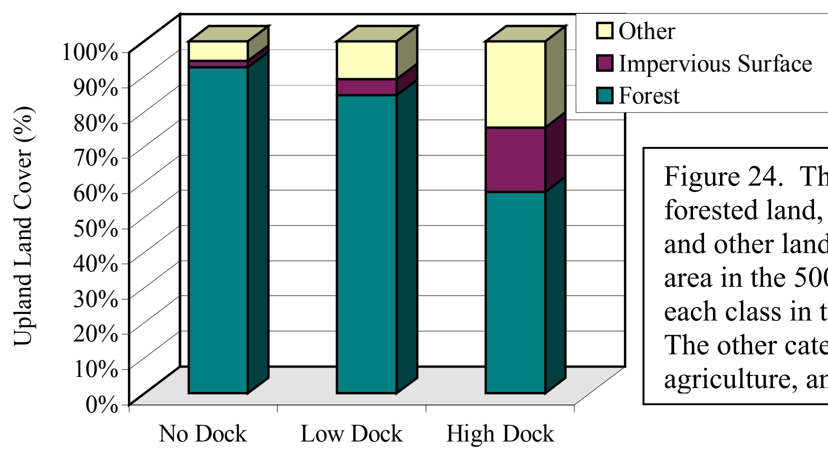


Figure 24. The average amount of forested land, impervious surface, and other land uses for the upland area in the 500-m radius buffer for each class in the SCECAP data set. The other category includes lawns, agriculture, and barren land.

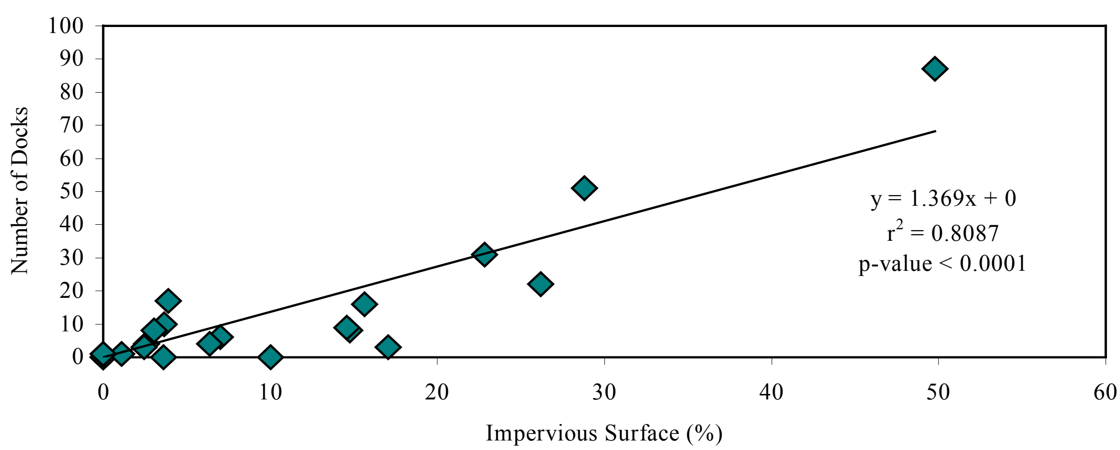


Figure 25. The data points and regression line showing the association between the number of docks and the amount of impervious surface within a 500-m radius buffer for the SCECAP data set.

2. Sediment Quality

The large tidal creek sediment quality data included 14 metal analytes, 24 PAH analytes, a low molecular weight PAH (mostly uncombusted hydrocarbons) composite value, a high molecular weight PAH (mostly combustion products) composite value, a total PAH composite value, a total PCB composite value, five sediment composition values, and two pore water chemistry parameters. The sediment quality parameters that were considered to be associated with dock structures and their related uses include chromium, copper, arsenic, the low molecular weight PAH composite value, the high molecular weight PAH composite value, and the total PAH composite value.

a. Trace Metal Analytes

As previously discussed, the amount of clay in the sediment may affect the distribution of trace metals (Olsen *et al.* 1982, Luoma 1989). All of the trace metal analytes evaluated except mercury had a significant clay covariate that accounted for most of the variance among categories in the ANCOVA models (Table 21). The category effect (i.e., no dock, low dock, and high dock) was significant (p-value < 0.05) for only the cadmium analyte. Cadmium concentrations increased from the no dock category to the high dock category with a significant difference between the no and high dock categories. Cadmium concentrations were, however, below levels that are considered to be biologically harmful (Long *et al.* 1995). Metals that were hypothesized to increase in concentration with the abundance of docks (i.e., copper, chromium and arsenic) did not show a significant difference between the no dock, low dock, or high dock categories. The average concentration of these metals did, however, increase in a systematic manner from the no dock to the high dock categories (Figure 26). The variance within each category was large and possibly prevented the detection of a difference among categories when in fact one may have occurred (a Type II statistical error resulting from low numbers of samples and high variability).

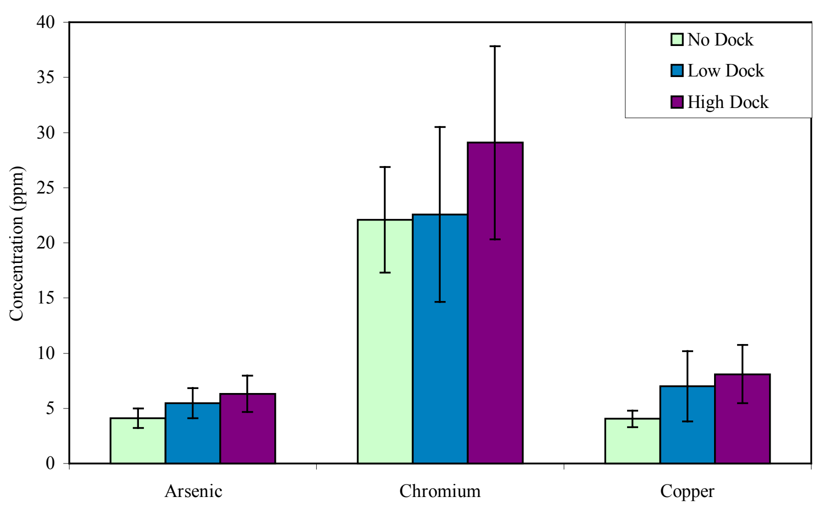


Figure 26. The average concentration and one standard error of arsenic, chromium, and copper for each treatment category evaluted for the SCECAP data set.

The Kruskal-Wallis rank sum test did not detect any significant differences among categories for the fourteen trace metals analytes (Table 22). This is similar to the results of the parametric analyses which found only category effects for cadmium.

Correlations between the number of docks within the buffer and the concentrations of metal analytes were only significant for the cadmium and nickel analytes (Table 23). The concentrations of these analytes in sediments are not likely to be derived from dock structures.

The concentrations of arsenic exceeded a level where biological effects may occur for two sites in the low dock category and four sites in the high dock category (Long *et al.* 1995). Sediment arsenic concentrations in the southeast are naturally high and frequently exceed levels that are reported to cause biological harm (Sanger 1998). Sediment concentrations of chromium and copper did not reach biologically harmful levels at any site (Long *et al.* 1995).

These analyses suggest that dock structures have no adverse effects on sediment metal concentrations at the scale of large tidal creeks. However, a general pattern of increasing sediment concentrations for metal analytes of concern relative to docks (arsenic, chromium, and copper) was observed from the no dock to the high dock category.

b. Polycyclic Aromatic Hydrocarbon Analytes

As previously discussed, the amount of TOC in the sediment may influence the distribution of PAHs (e.g., Boehm and Farrington 1984, Barrick and Prahl 1987). The TOC covariate was significant (p-value < 0.15) for all of the PAH analytes, except acenaphtylene, and the composite PAH measures. The ANOVA/ANCOVA models only differentiated among dock categories for the benzo(b)fluoranthene, benzo(e)pyrene, and benzo(j,k)fluoranthene analytes (Table 21). All had significantly higher concentrations in the high dock category compared to the no dock category.

The variability in the PAH contaminant concentrations within categories was high (Figure 27). Several of the PAH analytes showed an increasing trend from the no dock category to the high dock category including 2,6-dimethylnaphthalene, flourene, benzo(a)anthracene, benzo(j,k)fluoranthene, naphthalene, phenanthrene, anthracene, chrysene+triphenylene, fluoranthene, pyrene, the low molecular weight PAH composite value, the high molecular weight PAH composite value, and the total PAH composite value (Figure 27). No sites had sediment concentrations of PAHs that exceeded values which are reported to cause biological harm (Long *et al.* 1995).

One PAH analyte, benzo(j,k)fluoranthene, had significant differences among categories in the Kruskal-Wallis rank sum test (Table 22). The high dock category had the highest mean score for benzo(j,k)fluoranthene and the no dock category had the lowest mean score.

None of the PAH analytes were correlated with the number of docks (Table 23).

The above findings suggest that dock structures had small effects on sediment PAH concentrations at the scale of large tidal creeks. Sediment PAH concentrations were probably more affected by human development activities that occurred in the watershed. Some PAH analytes that are known to be associated with boating activities, however, consistently attained higher values in the high dock category suggesting boating activities may have contributed to these high concentrations. None of the PAH analytes, however, approached threshold values of concern as defined by Long *et al.* (1995).

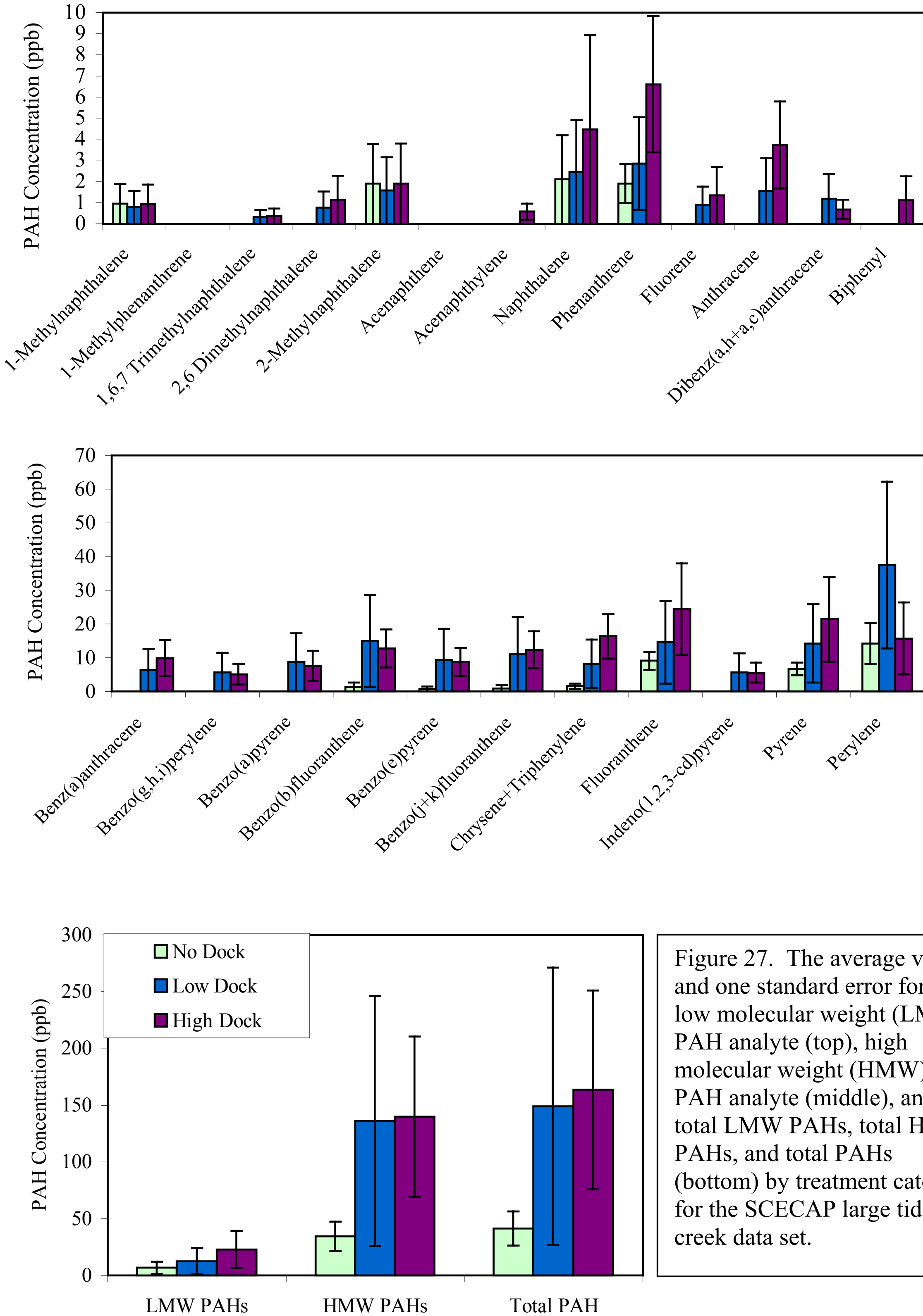


Figure 27. The average value and one standard error for each low molecular weight (LMW) PAH analyte (top), high molecular weight (HMW) PAH analyte (middle), and the total LMW PAHs, total HMW PAHs, and total PAHs (bottom) by treatment category for the SCECAP large tidal creek data set.

c. Trace Metals and Polycyclic Aromatic Hydrocarbon Anlaytes Combined

A Friedman's test was conducted to evaluate if the cumulative concentrations of arsenic, chromium, copper, and the 24 PAH analytes varied among categories in a manner that would suggest docks were a contributing factor. This test found significant (p-value < 0.0001) category effects with the highest concentrations occurring in the high dock category followed by the low dock and no dock categories. This finding suggests the cumulative distribution of dock-associated contaminants was not a random process. The density of docks is a likely contributor to the higher concentrations of dock-associated sediment contaminants in creeks with high numbers of docks at the scale of large tidal creeks.

d. Polychlorinated Biphenyls

As previously discussed the amount of TOC in the sediment may influence the distribution of PCBs (e.g., Boehm and Farrington 1984, Barrick and Prahl 1987). The TOC covariate was significant for the total PCB composite value and accounted for much of the variability in the associated ANCOVA model. The ANCOVA model found no significant difference among categories (Table 21). The Kruskal-Wallis rank sum test also found no significant difference among categories for the total PCB composite value (Table 22). None of the samples sites had sediment concentrations of PCBs that exceeded levels reported to cause biological harm (Long *et al.* 1995). The total PCB value was not correlated (p-value < 0.05) with the abundance of docks (Table 23). The above findings suggest sediment PCB concentrations were not related to the abundance of docks.

c. Sediment Composition

Several sediment composition parameters were evaluated (Table 21). Values for most of these parameters were auto-correlated and only the results for analyses of the clay and TOC content of sediments are summarized. These two parameters represent the patterns observed for the other sediment composition variables. The clay and TOC values were not significantly different among categories in a one-way ANOVA (Table 21) and were not correlated with the number of docks (Table 23). The Kruskal-Wallis rank sum test also found that the clay and TOC content were not different among categories (Table 22). The above findings suggest that dock structures had no measurable effect on sediment properties at the scale of large tidal creeks.

d. Sediment Pore Water

The values of sediment pore water total ammonia (TAN) and sediment pore water unionized ammonia (UAN) parameters were not different among categories in the ANOVA models (Table 21). The Kruskal-Wallis rank sum test also identified no significant category effects for UAN or TAN (Table 22). TAN was, however, negatively correlated (p-value < 0.05) with the number of docks (Table 23). UAN was not correlated with dock abundance (Table 23). The above finding suggests that dock structures had no measurable effect on pore water nutrients at the scale of large tidal creeks.

3. Water Quality

a. Nutrients

The amount of nutrients (concentrations of dissolved and particulate forms of nitrogen and phosphorus) in the water is a measure of the potential for a site to support the growth of algae. Algal blooms are frequently associated with high nutrient concentrations, noxious odors, hypoxic and anoxic conditions, and fish kills. Nutrient concentrations are not likely to be directly affected by dock structures but they were evaluated because they may be associated with the type of land use that docks are associated with.

Of the 17 nutrient parameters evaluated, five (dissolved ammonia, dissolved silicon/silica, orthophosphate, total dissolved phosphate, and total phosphorus) had significant category effects in the ANOVAs (Table 24). Nutrient concentrations for the five parameters were highest in the low dock category. Two (orthophosphate and dissolved silicate) of the 17 nutrient parameters evaluated had significant (p-value < 0.05) category effects in the Kruskal-Wallis rank sum test (Table 25). Highest concentrations for these nutrients occurred in the low dock category.

Nine of the 17 nutrient parameters were significantly ($p < 0.05$) correlated with the number of docks including dissolved organic nitrogen, dissolved silicon/silica, orthophosphate, and total dissolved nitrogen (Table 26). The slopes for all of the regressions were negative indicating that as the number of docks increased, water column nutrient concentrations decreased.

The above findings suggest the amount of nutrients in the water were not related to dock abundance. Higher nutrient concentrations appear to be related to the non-point source runoff associated with human activities and land uses characteristic of low densities of docks (e.g., septic tanks, limited agriculture).

b. Algae Pigments

The concentration of chlorophyll-*a* in the water is a measure of the algal biomass. Chlorophyll-*a* levels were not significantly different among categories (dock treatments) in the ANOVA or Kruskal-Wallis rank sum test (Tables 24 and 25). In addition, the chlorophyll-*a* parameter was not correlated with dock density (Table 26). This finding suggests there was no relationship between dock structures and the biomass of algae in the water at the scale of large tidal creeks.

c. Biological Oxygen Demand

The biological oxygen demand (BOD5) is a measure of the amount of oxygen in the water that is used by the organisms, mainly bacteria, for metabolic processes. At high levels of BOD5 insufficient oxygen may be available to support valued living resources, including fish, crabs, and shrimp. BOD5 is also an environmental quality indicator that is not expected to be directly affected by dock structures, but may be affected by the land uses and human activities associated with higher abundances of docks (i.e., categories). The BOD5 in the water column was not significantly different among categories in either the ANOVA or Kruskal-Wallis rank sum tests (Table 24 and 25). It also was not significantly correlated with the abundance of docks (Table 26). The above findings suggest that dock structures had no measurable effect, direct or indirect, on BOD5 at the scale of large tidal creeks.

d. Conventional Water Quality Measures

A broad range of conventional water quality measures were collected for each large tidal creek sample site including: dissolved oxygen (DO) concentration, pH, salinity, water temperature, turbidity, and secchi disk depth. Category effects in the ANOVA models were significant for only the DO parameter (Table 24). DO was significantly higher in the no dock and high dock categories compared to the low dock category. Only DO had a significant category effect in the Kruskal-Wallis rank sum test with the highest mean score in the no dock category and the lowest in the low dock category (Table 25). The pH, temperature, secchi disc depth, and turbidity values were correlated ($p < 0.05$) with dock abundance (Table 26). The value of the pH and secchi disc depth parameters increased as the number of docks increased. The temperature and turbidity parameter values decreased as the number of docks increased (Table 26). These findings suggest that dock structures have no measurable effect, direct or indirect, on conventional water quality measures at the scale of large tidal creeks. Conventional water quality metrics were probably more related to non-point source runoff associated with land uses and human activities occurring in the watershed.

e. Fecal Coliform Bacteria

The fecal coliform parameter is an indicator of the level of bacterial contamination associated with a site. Fecal coliform bacteria concentrations were not significantly different among categories in the ANOVA or Kruskal-Wallis rank sum test (Tables 24 and 25). In addition, fecal coliform levels were not correlated with dock density (Table 26). This finding suggests that dock structures had no effect on the level of bacterial contamination in the water at the scale of large tidal creeks.

4. Biological Integrity

a. Toxicity

The following assays of sediment toxicity were evaluated for large tidal creeks: (1) a clam assay which involved an acute survival measure and a chronic growth measure; (2) an amphipod assay which provided an acute survival measure; and (3) a bacterial luminescence assay (Solid-phase Microtox) which involved measuring the change in light production after the bacteria were exposed to the sediments from each sample site. Results for none of the toxicity assays were different among categories in the ANOVA or Kruskal-Wallis rank sum tests (Table 27 and 28). The bacterial bioluminescence assay results were the only toxicity measure that was significantly correlated ($p < 0.05$) with the number of docks (Table 29). The bacteria produced more light for the high dock category than other categories suggesting the lowest toxicity occurred at these sites. This is the opposite pattern than what would be expected. Because the bioluminescence assay is influenced by sediment characteristics (i.e., the greater the amount of fine sediment the lower the luminescence value - Ringwood *et al.* 1997), the results of this test were probably more influenced by sediment type differences among categories and sites that by dock density. The above findings suggest that dock structures had no adverse effects on sediment toxicity at the scale of large tidal creeks.

b. Benthic Organisms

As previously discussed, benthic organisms have limited mobility and generally cannot avoid pollution stress and are frequently used as indicators of biological integrity and environmental quality (Rosenberg 1978, Rhodes *et al.* 1978, Boesch and Rosenberg 1981, Holland *et al.* 1987, Lerberg *et al.* 2000). The benthic parameters evaluated for large tidal creeks included abundance of 13 species or taxa, the number of taxa per sample or species richness, the total number of benthic organisms in each sample, the score for the South Carolina benthic index of biotic integrity (IBI-SC), and the score for the Carolinian Province benthic index of biotic integrity (IBI-CP). Figure 28 provides the average value and one standard error of benthic parameters for each category.

The silt-clay content of the sediment and the salinity are natural factors that affect the kinds and abundance of benthic organisms (e.g., Holland *et al.* 1987, Lerberg *et al.* 2000) and both parameters were evaluated as covariates in the one-way ANCOVA models for benthic parameters except for the IBI-SC and IBI-CP (Table 27). Seven of the 17 benthic parameters had significant category effects in ANOVA/ANCOVA models (Table 27). The highest values of the Cirratulidae*, Mediomastus ambiseta, Polydora cornuta*, *Streblospio benedicti, Scoletoma tenuis, Tharyx acutus, Tubificoides brownae*, and total benthic abundance occurred in the low dock category (Table 27). Most of the time lowest values of benthic parameters occurred in the high dock category. Values of the benthic environmental quality indices (IBI-SC, IBI-CP) were not different among categories in the ANOVAs. The value for these indices was insensitive to the environmental conditions represented by the sample sites in the large tidal creek sites.

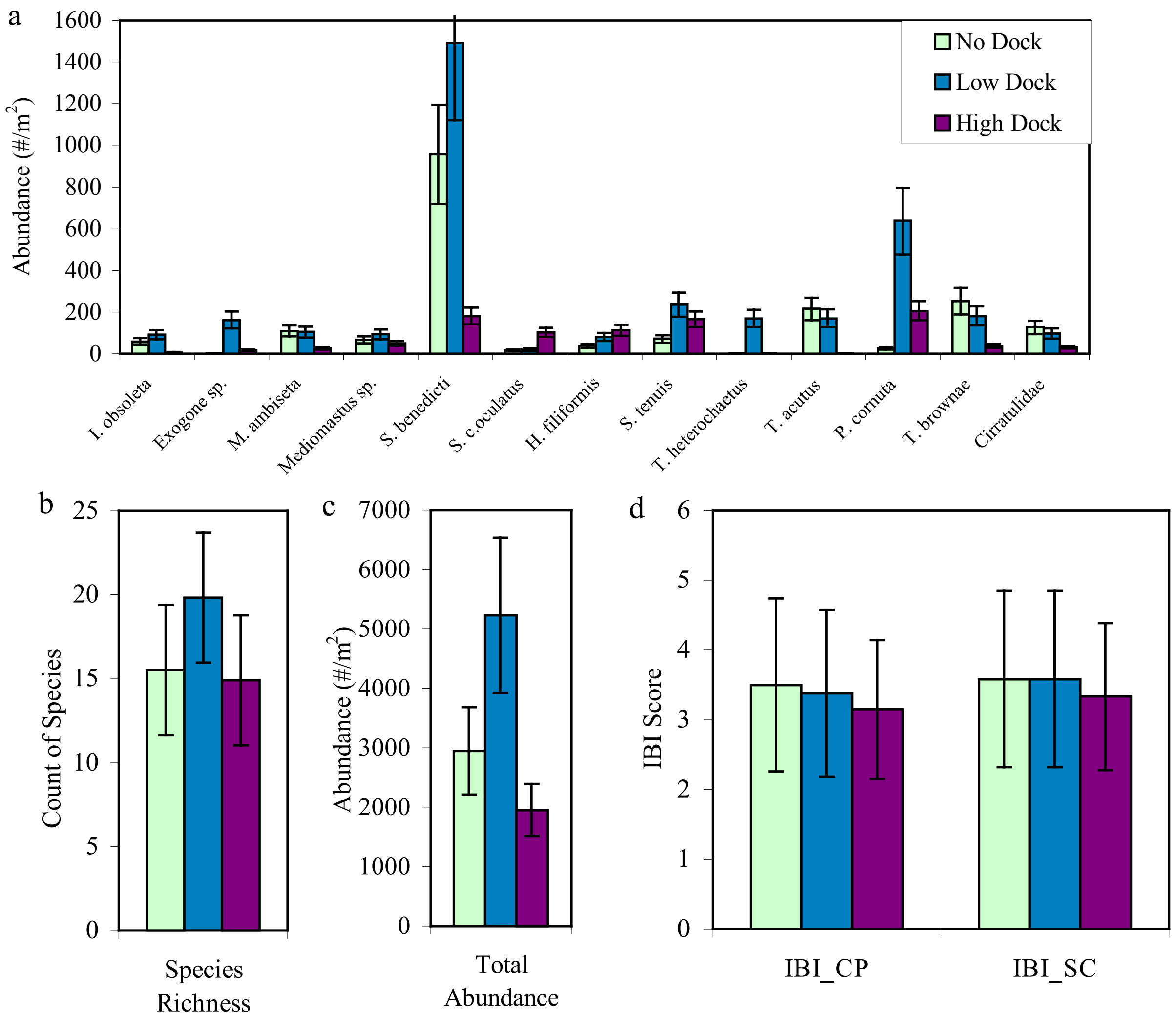


Figure 28. The average value and one standard error for each SCECAP benthic taxa abundance (a), species richness (b), total abundance (c), and indices of biotic integrity score (d) metrics for this study by treatment category.

The Kruskal-Wallis rank sum test results were similar to the ANOVA/ANCOVA results (Table 28). *S. benedicti, T. acutus, P. cornuta*, and the total number of organisms had a significant category effect with the highest abundances in the low dock category. *M. ambiseta* had a significant category effect with the highest abundance in the no dock category.

Three benthic parameters, *M. ambiseta*, *T. acutus*, and total abundance of organisms, had significant correlations with the number of docks near the sampling sites (Table 29). These regressions have low coefficients of determination indicating the correlations were weak. All had negative slopes indicating that as the number of docks increased, the abundance of organisms decreased.

The above findings suggest that dock structures had little to no adverse effects on benthic communities for large tidal creeks. The indirect effects of land uses and changes in environmental conditions associated with the low dock category (e.g., intermediate levels of non-point source runoff, high nutrient loadings), appeared to be associated with an increase in benthic abundances for a suite of species (e.g*., S. benedicti, P. cornuta*) that are generally considered to be tolerant to moderate levels of pollution in large tidal creeks.

c. Fish and Crustaceans

As previously discussed fish and crustaceans (i.e., shellfish) have commercial, recreational, and ecological value. Some are harvested; others serve as prey for birds, sport fish, and marine mammals. Unlike benthic organisms, many fish are wide-ranging and have low fidelity for specific sample sites. The value of the kinds and abundances of fish and crustaceans as a measure of environmental quality at a site is therefore questionable.

The large tidal creek fish and crustacean parameters evaluated included abundances and biomass of three fish species (*Anchoa mitchilli, Bairdiella chrysoura*, and *Leiostomus xanthurus*), two shrimp species (*Penaeus aztecus and Penaeus setiferus*), one squid (*Lolliguncula brevis*), the number of fish and crustacean taxa or species richness per trawl, the total abundance of fish and crustaceans per trawl, and the total fish and crustacean biomass per trawl. Figures 29 and 30 provide the average value and one standard error of fish and crustacean parameters for each category.

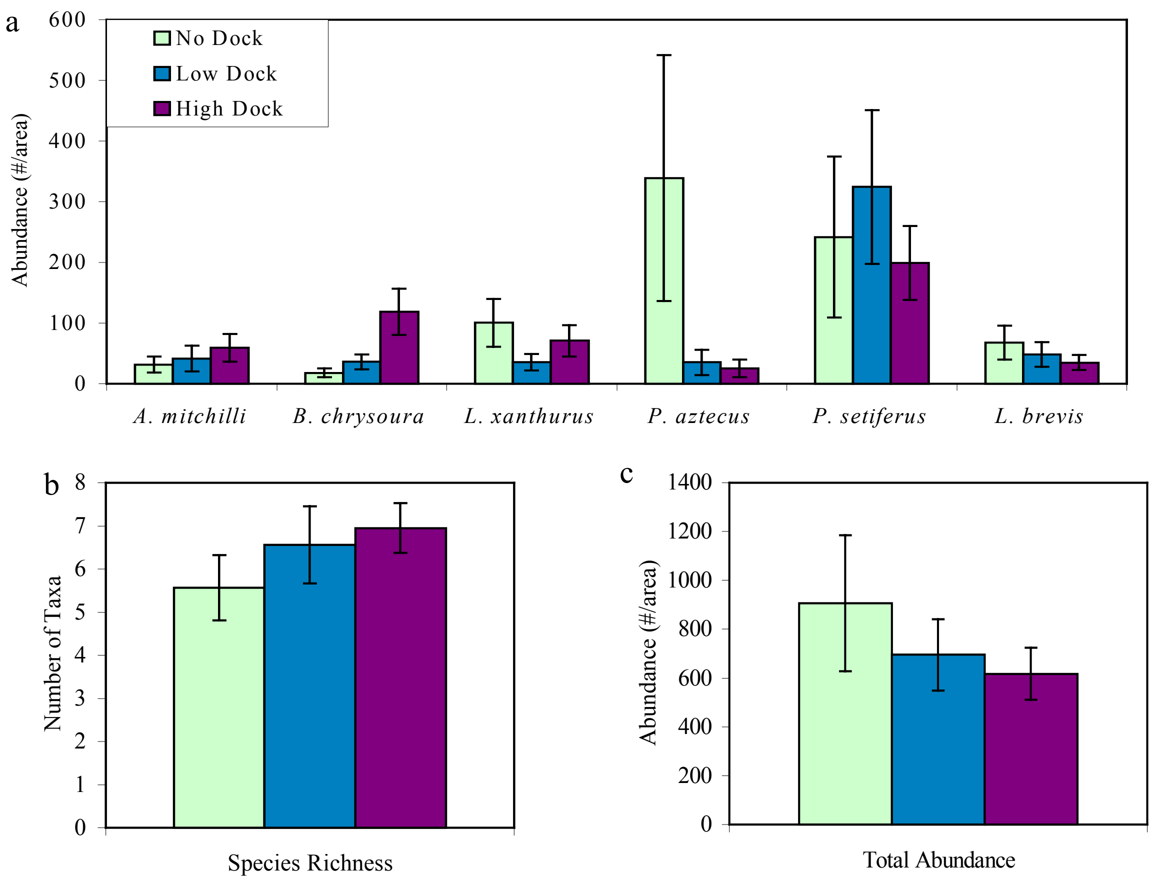


Figure 29. The average value and one standard error for each SCECAP fish and crustacean taxa abundance (a), species richness (b), and total abundance (c) metrics for this study by treatment category.

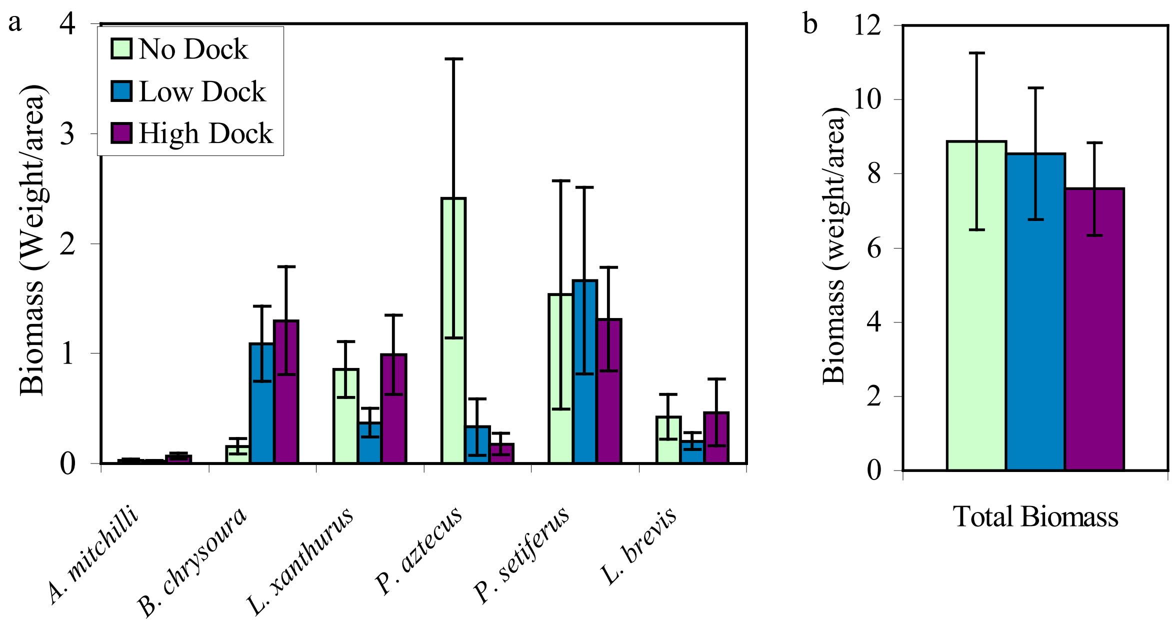


Figure 30. The average value and one standard error for each SCECAP fish and crustacean taxa biomass (a) and total biomass (b) metrics for this study by treatment category.

The silt-clay content of the sediment and salinity are important factors controlling the kinds and abundances of fish (Lippson *et al.* 1979) and both salinity and silt-clay content were evaluated as covariates in ANCOVA models (Table 27). The silt-clay content was a significant covariate for seven of the 15 fish and crustacean parameters including *L. xanthurus* abundance and biomass, *P. aztecus* abundance, *P. setiferus* abundance, species richness, total fish and crustacean abundance, and total fish and crustacean biomass. Salinity was a significant covariate for five fish and crustacean parameters including *B. chrysoura* abundance, *L. xanthurus* abundance and biomass, *L. brevis* abundance, and total fish and crustacean biomass. Category effects were significant for four of the 15 fish and crustacean parameters evaluated using ANOVA/ANCOVA models (Table 27). The highest abundance and biomass of *B. chrysoura* occurred in the high dock category. The abundance and biomass of *Penaeus aztecus* showed the inverse pattern with highest abundance in the no dock category (Table 27). None of the fish and crustacean parameters had significant category effects in the Kruskal-Wallis rank sum test (Table 28). None of the fish and crustacean parameters were correlated with the number of docks (Table 29).

The above findings suggest that dock structures were associated with higher abundance of silver perch and lower abundance and biomass of brown shrimp. Many factors could contribute to these distributional patterns. For example, docks may attract silver perch which feed on the invertebrates including juvenile brown shrimp near dock structures. In addition, human access and fishing pressure on brown shrimp may be higher at sites with high dock densities than in creeks with low dock densities.

IV. DISCUSSION

A. Dock Proliferation Concerns

Several workshops were conducted with scientists at the South Carolina Department of Natural Resources (SCDNR) and the National Oceanic and Atmospheric Administration (NOAA), and permitting staff at the SCDHEC-OCRM to identify and discuss the environmental issues associated with dock proliferation as well as make recommendations for how this study could best address these concerns. The major environmental concerns associated with docks that were identified during these workshops included: (1) avoiding impacts on navigation, (2) reduced productivity of salt marsh vegetation from shading by dock structures, (3) damage to salt marsh vegetation and other living resources during dock construction, (4) harm to tidal creek and salt marsh habitats from pierhead and floating dock structures, (5) chemical contamination of the water and sediments from the leaching of wood preservatives used to retard decay, (6) chemical contamination of the water and sediments from dock-related boating activities, (7) adverse harm to the biological integrity of tidal creeks including nursery functions from dock proliferation, (8) reduced access to shellfish beds and other valued estuarine habitats, and (9) impacts of docks and related boating activities on shoreline erosion and stability. This study did not address all of these issues; however, this study did provide new information concerning: (1) the magnitude and relative importance of the shading impacts at local, county and statewide scales; (2) the magnitude of chemical contamination from dock leachates and boating activities on tidal creek sediments; and (3) the association between the dock abundance and human development of the landscape including the cumulative impact of suburban development and docks on the ecological integrity of tidal creek ecosystems.

B. Shading Effects

The shading effects on salt marsh vegetation from dock structures were easily measured as a decrease in *Spartina alterniflora* stem density. We estimated that shading effects from dock structures decreased the stem density (an indicator of vegetation productivity) of the dominant salt marsh plant *S. alterniflora* by 71% in Charleston Harbor estuaries. A similar decrease in stem density was associated with dock structures in other regions of the country (e.g., Kearney *et al.* 1983, McGuire 1990, Colligan and Collins 1995). We believe this estimate of shading effect applies statewide. Additional sampling in other regions of South Carolina (e.g., Myrtle Beach, Hilton Head) to validate the applicability of this estimate is, however, recommended. In addition, we did not attempt to assess the effects of shading levels or lower stem densities on marsh biota which could be assessed in future studies.

The magnitude of shading effects from docks on salt marsh vegetation was not influenced by the geographical orientation. In a similar study conducted in Virginia, McGuire (1990) observed minor differences in the magnitude of shading effects associated with the geographical orientation. This was probably the result of the relatively broad variance in dock heights and widths sampled by McGuire (1990).

South Carolina is characterized by large expanses of salt marsh (149,517 ha or 369,456 ac). Dock shading effects were small when evaluated from the perspective of the amount of marsh that occurs within specific tidal creeks (0.03-0.72%), in coastal counties (0.01-0.98%), or statewide (0.01-0.12%). If dock shading resulted in the total loss of marsh grass productivity, the net loss at the local, county, and statewide scale would still be small. The only scale at which shading from dock structures was estimated to affect greater than 5% of the available marsh productivity was under a maximum development scenario for selected tidal creeks. Therefore, if there are no land use planning restrictions and builders are allowed to develop properties at current SCDHEC-OCRM regulations (i.e., 75 ft of marsh frontage), then the impacts on marsh productivity tidal creeks could be significant (i.e., 5%).

Shading from the average dock (100 m in length and 1.22 m in width) adversely affects approximately 87 m^2 (932 ft^2) of marsh grass. Annually, approximately 700 docks are permitted in South Carolina resulting in reduced stem density and productivity by an amount equivalent to approximately 6 ha (15 ac) of salt marsh. Approximately 7,000 docks have been permitted over the last decade resulting in a loss of salt marsh equivalent to 60 ha (150 ac).

Dock owners are not required to mitigate the loss of salt marsh productivity resulting from shading effects. Other activities that result in equivalent harm to salt marsh habitat (e.g., industrial development, housing developments, dredged material disposal) would be required by regulatory and resource agencies to mitigate the damage they cause. Typical mitigation actions would include restoration of 1-2 times the amount of degraded salt marsh or creation of 1-3 times as much new marsh at a nearby site or in an equivalent habitat. One method to mitigate the damage from shading effects would be to charge dock owners a fee (one time or periodic) amortized over the expected lifespan of the dock equivalent to the amount required to mitigate for the losses in salt marsh productivity.

Erosion under the dock walkway that could be attributed directly to the dock structure was rarely observed. A few docks in Parrot Creek were not sampled due to rivulets running length-wise under portions of the dock walkway. Rivulets are a natural geographic feature of salt marshes which may have been created from the dock structure or may have been natural. It appeared that in most cases, the reduced plant stem density under docks was apparently sufficient to retard erosion of marsh sediments. The docks that appeared to be used as resting areas by wading birds due to large quantities of bird feces had taller stems under the dock compared to the stems next to the dock. The plants under these docks may have been exposed to higher levels of nutrients which is thought to be a factor in the growth of short- or tall-form *S. alterniflora*.

Vegetation adjacent to and under docks was clearly damaged during construction activities. This damage appeared to result from construction workers using the area adjacent to docks as a walkway and/or as a lay down area for lumber and equipment. The damaged habitats recovered within one to two years after construction had been completed. The effects of dock construction activities on salt marsh habitat is short term and probably does not need to be mitigated. SCDHEC-OCRM should, however, consider holding a workshop with dock builders to identify ways to minimize construction damage and speed up the recovery process. They should also inform waterfront property owners that are granted dock permits that construction activities result in damage to salt marsh habitat and request the property owner to encourage the contractor to minimize this damage.

Some dock owners used the salt marsh habitat adjacent to their docks as a mooring area for boats. Waterfront property owners that did not have docks also frequently used the salt marsh habitat adjacent to their property as a haul out area for small boats (e.g., canoes, kayaks, john boats). Because the damage to salt marsh vegetation resulting from these uses may persist for years, SCDHEC-OCRM should initiate an education program to inform dock owners of the damage resulting from these activities. If dock owners persist in this type of use, then regulatory actions may be required.

Dying and decaying marsh grass or wrack accumulation on the marsh surface can adversely affect salt marsh vegetation. When we observed wrack accumulation under and adjacent to docks, the stem density of living vegetation was severely reduced. Wrack accumulation under docks was a rare event and only affected a small (< 5%) proportion of the area under the few docks where it occurred. Wrack accumulation on adjacent salt marsh environments appeared to occur more frequently. The effects of wrack accumulation under and adjacent to docks appears to be a minor concern. A more important issue we observed was dock owners that place wood, rip rap, shells, and other materials under their docks. The purpose of placing these materials under the dock is not clear. These materials, however, result in the exclusion of the natural marsh vegetation from the area under the dock. SCDHEC-OCRM should also undertake an education program to inform dock owners that the area under docks is State property and not a disposal site for discarded material. Dock owners should be encouraged to remove unnatural materials from the area under their docks. If dock owners continue to dispose of material under their docks, then regulatory actions should be considered.

C. Effects of Dock Leachates and Boating Activities on Sediment Contaminant Concentrations

The contaminants of concern to tidal creeks and salt marshes are the trace metals and polycyclic aromatic hydrocarbons (PAHs). Trace metals are naturally occurring elements from the earth's crust; however, concentrations of some metals can be increased by anthropogenic activities (e.g., mercury, copper, chromium). Potential anthropogenic sources of trace metals to the estuarine environment from suburban development include antifouling paints (USEPA 1985, Kennish 1992), sacrificed anodes used on boats, tire wear from roads (Cole *et al.* 1984, USEPA 1985, Maltby *et al.* 1995), and leaching of preservatives from dock structures. PAHs are a major component of lubricating oils and fuels, but are also produced during the combustion of organic matter, including fossil fuels. PAHs derived from lubricating oils and fuels are generally low molecular weight compounds (≤ 3 rings) and are often heavily alkylated (Lee *et al.* 1981). PAHs derived from the combustion of organic materials are generally high molecular weight compounds (≥ 4 rings) with little alkylation (Lee *et al.* 1981). The low molecular weight PAHs degrade by natural processes faster than the high molecular weight PAHs (Bossert *et al.* 1984). Potential sources of PAHs to estuarine systems include runoff from highways and parking lots, vehicle exhaust, street dust, fuel spills, boating activities including spilling of gas and oil as well as fossil fuel combustion, and atmospheric fallout (Weinstein 1996).

Three trace metals, arsenic, chromium, and copper, are the most common contaminants used as a wood preservative for dock structures. Therefore, these three metals have the greatest potential to increase in estuarine environments from docks. Copper and chromium are essential elements or micronutrients for living organisms. At levels slightly above the required micronutrient concentration (2.5 ppb) copper is toxic to marine organisms, particularly early developmental stages of marine invertebrates (e.g., Brooks 1996). Mollusks (e.g., oysters) and some crustaceans concentrate copper in their tissues to levels several orders of magnitude above background in environments contaminated with copper (Pringle *et al.* 1968, Roosenberg 1969, Cunningham 1979, Suedel *et al.* 1994). The toxicity of chromium varies by more than an order of magnitude depending upon species, age, developmental stage, temperature, pH, salinity, duration of exposure, and interactions with other contaminants. Chromium IV is the most toxic form of chromium and is toxic to segmented worms and crabs at concentrations as low as 200 ppb (Eisler 1986). Chromium does not appear to be biomagnified in biological systems (Suedel *et al.* 1994). Arsenic is a common environmental metal that is toxic to a range of marine organisms including algae, mollusks, crustaceans, and fish (Eisler 1986). Arsenic toxicity ranges from about 20 ppb to 35,000 ppb depending upon taxa, chemical form, oxidative state, and route of exposure. Arsenic tends to bioaccumulate in higher-level predators (e.g., predatory fish) but not in primary and secondary consumers. Elevated tissue levels of arsenic, chromium and copper were also reported to be associated with sublethal responses including reduced growth and abnormal behavior (Weis and Weis 1996).

The available science suggests that preservatives (chrominated copper arsenate or CCA) impregnated into the wood used for construction of the majority (> 90%) of dock structures that exist in the coastal zone become fixed to the fibers (Brooks 1994, Weis and Weis 1996). These preservatives leach from the wood into the marine environment at low to modest rates when the wood is exposed to rain or is submersed in seawater (e.g., Fahlstrom *et al.* 1967, Brooks 1996, Weis and Weis 1996). Leaching decreases by about 50% each day after immersion of the wood with most of the leaching occurring in the first five to six days

(Cooper 1990, Brooks 1996). Approximately 99% of the leaching occurs in the 90 days following construction. CCA pilings, however, retard wood decay for years.

CCA leaching rates are highest in low pH and high salinity (> 10 ppt) environments (e.g., Warner and Solomon 1990, Brooks 1994, Weis and Weis 1996). In marine environments, copper leaches faster than chromium or arsenic (Brooks 1996). Brooks (1996) concluded that in the marine environment copper was the metal of greatest concern from dock leaching because of the greater toxicity of copper (relative to arsenic and chromium) and the higher leaching rate of copper from CCA treated lumber. He concluded that if copper leaching was maintained below toxic thresholds that arsenic and chromium are not likely to pose a threat to the marine environment.

Marine organisms collected from dock pilings and areas immediately adjacent to docks, particularly oysters, frequently contain higher concentrations of copper and other CCA metals (Wendt *et al.* 1996, Weis and Weis 1996). Organisms living on new CCA treated wood and in poorly flushed environments generally accumulated the highest concentrations of CCA metals (Weis and Weis 1996). The survival, growth, and bioaccumulation of oysters deployed adjacent to docks in moderately flushed tidal creeks were, however, not affected by newly constructed docks (Wendt *et al.* 1996). The tissue levels of copper in organisms directly adjacent to CCA treated lumber were reported to be sufficient to cause pathological damage and genotoxic responses to oysters and other marine biota (Couch 1984, Weis and Weis 1995, Weis *et al.* 1995). High tissue levels of chromium and arsenic also have been associated with genotoxic biological responses (Tkeshelaskvilli *et al.* 1980, Nakamuro and Sayato 1981). While copper and arsenic are accumulated in lower trophic levels, the transfer of these metals to higher trophic levels and biomagnification to levels of concern in the natural systems is low (Brooks 1996, Weis and Weis 1999). Therefore, it is unlikely that the bioaccumulation of dock leachates by marine biota is having or likely to have an impact living resources in South Carolina estuaries and tidal creeks for the following reasons: (1) the maximum leaching occurs early in the life of a dock; (2) the size of the area affected appears to be small and the number of organisms involved is small; and (3) the large tidal ranges that characterize South Carolina estuaries limit leaching to the immediate area of docks due to high flushing.

In estuarine environments, leachates from dock structures adsorb to fine-grained materials including silts, clays, and carbon particles (Weis *et al* 1993b, Brooks 1996, Wendt *et al.* 1996). The findings of this study were consistent with this statement. Heavily bulk headed, poorly flushed environments are repositories for CCA leachates and accumulated in bottom dwelling organisms (Weis and Weis 1992b). The sediment levels of CCA metals in tidal creeks with docks were, however, not in ranges that were likely to cause biological harm. In addition, CCA leachates from docks that are deposited in sediments tend to react and bind with acid volatile sulfide in the sediment to form a pool that is probably not available to or toxic to living resources (DiToro *et al.* 1992). This finding was further substantiated by the fact that the toxicity tests conducted as a part of SCECAP did not observe any differences in sediment toxicity between tidal creeks with docks and reference areas without docks. A study conducted by Wendt *et al.* (1996) found CCA metal concentrations only exceeded natural background levels in the immediate vicinity (1 m) of dock pilings. Weis *et al.* (1998) reported similar results for a range of Atlantic Coast estuaries including Debadeau Canal in the North Inlet Estuarine Research Reserve.

Boating activities have been associated with the release of PAHs into the marine environment and are reported to account for up to 20% of the PAH loading in North Carolina coastal systems (Hoss and Engle 1996). In addition, Sanger *et al.* (1999b) attributed the patchy distribution of PAHs in South Carolina tidal creeks to boating activity and small fuel spills. PAHs resulting from boating activities probably accumulate in creek and salt marsh sediments (Sanger *et al.* 1999b). PAHs in estuarine sediments are also derived from many other land based human activities including roadway runoff, improper disposal of motor oil and other hydrocarbons, atmospheric deposition, and leaking fuel tanks. It was not possible for this study to distinguish between the PAHs associated with dock-related boating from that resulting from other recreational boating activities or other anthropogenic activities including non-point source runoff from the upland. The similarity in the distributional patterns of PAHs and PCBs potentially indicates that much (if not most) of the PAHs in tidal creek sediments may be from land-based sources and not associated with boating activities. PCBs are organic contaminants that behave in the environment in a similar manner to PAHs (e.g., are associated with TOC in sediments); however, PCBs are not released into the environment from boating activities. PCBs have been banned from general use for about 20 years.

D. Habitat, Landscape, and Cumulative Effects

Tidal creek water quality, including salinity, temperature, and dissolved oxygen concentration as well as nutrients and algal biomass is exceptionally dynamic and varies over broad ranges from day-to-night, stage of the tide, and during and following rain events (Holland *et al.* 1997, Hubertz and Cahoon 1999, Lerberg *et al.* 2000, Holland 2000). Early life stages of living resources that use tidal creeks as nurseries frequently experience environmental conditions that are near the limits of their physiological tolerances (i.e., "*living on the edge*"). As a result, the environmental quality of tidal creeks may be particularly vulnerable to the process changes that accompany watershed development. For example, sediment depositional zones in tidal creeks accumulate chemical contaminants to levels that are frequently an order of magnitude higher than those in adjacent tidal rivers and harbors (Olsen *et al.* 1982, Sanger *et al.* 1999a and 1999b). These higher levels of sediment contaminants may synergistically interact with the broad fluctuations in water quality to cause acute and/or chronic effects to living resources.

Over 70-80% of the total estuarine area in South Carolina and Georgia consists of tidal creeks and salt marshes (Field *et al.* 1991, NOAA 1990, Nummedel *et al.* 1977). These ecosystems are a primary link between upland environments and estuaries (Lerberg *et al.* 2000, Mallin *et al.* 2000). They also serve as refuges and nursery grounds for many fish and shellfish species (e.g., Hackney *et al.* 1976, Weinstein 1979, Wenner and Beatty 1993) and provide feeding habitat for wading birds and large fish (Wenner 1992, Dodd and Murphy 1996). Because of their natural beauty, tidal creeks and salt marshes of the southeastern coastal zone are preferred sites for human habitation and development. Development pressure on estuarine watersheds is likely to continue into the foreseeable future (Edwards 1989, Culliton *et al.* 1990, Cohen 1997).

Residential development of estuarine watersheds is accompanied by a decrease in the amount of forested lands; infrastructure development including bridges, roads, and highways; and an increase in non-point source pollutant loadings including trace metals, polycyclic aromatic hydrocarbons, pesticides, and nutrients (USEPA 1993, Fortner *et al.* 1996, Kucklick *et al.* 1997). These land cover and use changes may adversely affect the productivity, biodiversity, and ecological functioning of coastal ecosystems (Vitousek *et al.* 1997). Because tidal creeks are shallow and have limited ability to dilute pollutants, development of their watersheds has the potential to substantially alter their environmental quality (Sanger *et al.* 1999a,b, Lerberg *et al.* 2000).

The continued development of tidal creek watersheds will undoubtedly be accompanied by an increasing human demand for access to estuaries. The majority of South Carolina coastal residents perceive that docks are a reasonable and acceptable means for obtaining this access as long as ecologically sensitive habitats are not adversely affected (Felts 2001). In a recent survey of SC coastal residents, the general public's perception was that dock structures do not distract from the natural beauty of tidal creeks and salt marshes. Most of the individuals surveyed felt that the ability of waterfront properties to have docks substantially increased property values (Felts 2001). The first question a real estate broker establishing the value of waterfront property generally asks is "does the property have a dock or is it eligible to have one" (J. Settle, personal communication). Most coastal residents surveyed did not believe dock structures were harmful to the environment and felt that the abundance of docks should only be limited in ecologically sensitive habitats (Felts 2001). These survey results suggest that most coastal residents believe that they have the right to construct a dock as a means of accessing the estuarine environment even though the State generally owns the land over which the dock is placed (Felts 2001).

People generally purchase waterfront property to facilitate recreational boating activities and, if allowed, quickly install docks to access adjacent waterways. Waterfront developments are generally associated with a substantial infrastructure including sidewalks, paved driveways, wide roads, and quick access to shopping (i.e., large areas of impervious surface). The proportion of impervious surface in a drainage area or watershed is an accepted measure of the degree of watershed development (Arnold and Gibbons 1996, Lerberg *et al.* 2000). As an indicator of landscape condition, the amount of impervious surface is correlated with many other estuarine environmental quality indicators including human population density (Lerberg *et al.* 2000), salinity range and variability (Lerberg *et al.* 2000), abundance of fecal coliform bacteria (Mallin *et al.* 2000), sediment quality (Lerberg *et al.* 2000, Sanger *et al.*1999a and 1999b), and biological integrity (Lerberg *et al.* 2000).

The amount of impervious surface was strongly associated ($r^2 = 0.23$ and 0.81) with the abundance of docks for sites in the small and large tidal creek studies evaluated for this study. This suggests that docks are intrinsically linked to the suburban development of coastal watersheds. The environmental impacts associated with dock structures are part of the cumulative impact of suburban development on coastal watersheds. As a result, the effects of watershed development and docks cannot be easily separated. The proliferation of docks along the South Carolina coast is therefore a symptom of a much more serious problem -- uncontrolled landscape development and associated changes in environmental quality.

Dock structures appeared to have small effects on the kinds and abundance of bottom dwelling organisms that are prey for juvenile fish and are indicators of the biological integrity of tidal creek ecosystems. The amount of prey for fish and crustaceans was as abundant in creeks with docks as it was in creeks without docks. Weis *et al.* (1998) reported the effects of CCA treated wood on benthic organisms was generally limited to a 1-m perimeter around docks but may extend up to a 10-m perimeter from heavily bulk headed areas. None of the samples used for this study were collected within one meter of a dock and only one of the in the large tidal creeks included heavily bulk headed areas. The only evidence for a dock effect was the abundance of stress tolerant organisms tended to be more abundant and stress sensitive organisms tended to be less abundant in creeks with large numbers of docks compared to reference areas with no docks or creeks with few docks. However, these indicators of ecological integrity were probably more related to the amount of impervious surface in the watershed than they were to the abundance of docks.

Juvenile fish and crustaceans showed no consistent relationship in creeks with high numbers of docks compared to reference creeks or creeks with few or no docks. Dock structures probably attracted some fish functioning in a manner similar to an artificial reef. Recreationally valued taxa may also be subject to greater harvest pressure in creeks with high numbers of docks than creeks with few to no docks.

Individually, the adverse harm to the marine environment resulting from dock shading, CCA leachates, and increased concentrations of PAHs from boating was found to be small at the scale of tidal creeks. However, docks in combination with associated suburban development including increased pollution exposure (e.g., Sanger *et al.* 1999a and 1999b), changes in hydrology (Lerberg *et al.* 2000), and altered ecological processes (Lerberg *et al.* 2000) represent a major source of environmental degradation to tidal creeks and the associated salt marsh habitats. Our inability to separate anthropogenic watershed-scale effects, as indicated by the amount of impervious surface, and dock effects, as indicted by the abundance of docks, was a major limitation of this study.

The number of docks in most of the creeks for which samples were available was relatively low. Evaluations similar to this one may be warranted in the future when the number of docks in the state and some of the study creeks has substantially increased. Data collected in the future may allow better relationships to be developed between the number of docks and ecological indicators of concern. It will, however, always be impossible to separate the surrounding land use effect from the dock effect using field data. To separate the dock and development effects would require conducting a large scale experiment where a number (6-8) of watersheds representing the kinds of tidal creek ecosystems that occur in the state would need to be developed at typical human densities with relatively few docks allowed. The data from these no dock watersheds would then be contrasted to similar watersheds characterized by similar development but with a range of dock densities.

E. Value of SCECAP and TCP Data

The data collected by the SCECAP and TCP studies made this evaluation possible. Without these data, it is unlikely that an independent sampling effort that collected sufficient numbers of samples at the appropriate scales would have been feasible with the available funding. Processing of the sediment chemistry samples alone that were used for this study would have cost approximately $50,000. We estimate the cost of collecting and processing all of the data that were used for this study would have cost approximately $500,000.

The study elements that made it feasible to use the SCECAP and TCP data included: (1) probability based sampling approaches which ensured the samples represented the existing environmental conditions occurring in the state's tidal creeks and estuaries at broad spatial scales and not just the conditions characteristic of a few individual creeks; (2) a broad suite of environmental parameters which included measurement of physical indicators, chemical indicators, and ecological indicators for each sample site; (3) the quality assurance programs which ensured the accuracy, reliability and comparability of the data across years and studies; and (4) the broad spatial scales which were represented by these programs. These characteristics allow the SCECAP and TCP studies to be used as research platforms for addressing monitoring and assessments objectives for multiple programs and agencies.

Because this study relied on data collected by others, the number of samples in many of the categories was frequently not optimal. For example, data from only two suburban tidal creeks with no docks were available for the Small Tidal Creek Study. The small number of samples available from these creeks limited the power of the assessment to detect the effects of dock structures when in fact dock impacts may have occurred (i.e., the relative power of some of the statistical tests was low). This raised the question of whether additional samples would have detected dock impacts not observed in this study? For those indicators where significant differences among categories were detected the answer is that additional sampling would have provided little additional information. Additional samples would have reduced the variance about the mean for categories with high variability making the identification of category effects more likely. It is unlikely that additional sampling would have, however, altered the findings or conclusions of this study. Even with additional samples it is not likely that the effects of docks could be partitioned from the effects of suburban development. They were too strongly linked.

F. Existing Dock Regulations and Management Strategies

The South Carolina Department of Health and Environmental Control's Office of Coastal Resources Management regulates docks using the following criteria: (1) the walkway width must be less than 1.22 m (4 ft); (2) the height of the walkway must be 0.91 m (3 ft) above mean high water; (3) docks must extend to the first navigable creek with a defined channel (i.e., no creek jumping); (4) property frontage must have at least 22.86 m (75 ft) for a personal dock or at least 15.2 m (50 ft) for a shared dock with an adjacent property owner(s); (5) docks must be ≥ 6.1 m (20 ft) from the property line boundary; (6) dock boundary length must be less than 305 m (1000 ft) over critical areas (e.g., salt marsh); and (7) a suite of dock size restrictions and uses based on the creek width. The size restrictions on docks associated with creek width include: (1) creeks less than 6.1 m (20 ft) wide are restricted to 4.6 sq m (50 sq ft) of walkway and other dock structures; (2) creeks 6.1-15.2 m (20-50 ft) wide are restricted to 11.2 sq m (120 sq ft) of fixed pierhead and floating dock structures; (3) creeks 15.5-45.7 m (51-150 ft) wide are restricted to 14.9 sq m (160 sq ft) of dock but special permission is granted up to 24.2 sq m (260 sq ft); (4) creeks > 45.7m (150 ft) are restricted to average size of existing, adjacent docks (OCRM 1999).

SCDHEC-OCRM has proposed legislation to alter the above criteria to the following: (1) dock length would be limited to < 152 m (500 ft) for a single family and < 229 m (750 ft) for community docks over critical areas; and (2) docks would not be allowed in creeks < 6.1 m (20 ft) wide.

SCDHEC-OCRM has managed proliferation of docks by: (1) requiring developers to develop and submit a master dock plan that identified the location and specified the design of all docks associated with new developments; (2) encouraging the installation of community and multiple-owner docks; and (3) restricting the size and types of structures associated with docks (e.g., floating docks and pier heads).

We found little variance in dock characteristics (e.g., height, width, spacing between boards) among the docks surveyed. The existing regulatory criteria for docks and SCDHEC-OCRM policies appear to provide adequate guidance for ensuring docks are constructed in a consistent manner and in accordance with acceptable engineering standards. The existing dock height and width restrictions appear to ensure adequate salt marsh vegetation is maintained under dock walkways to prevent erosion. Historically, a few docks were constructed using soil fill and rip rap materials. These types of docks eliminate the salt marsh that occurred under them and adversely affect the movement of water and exchange of materials across the marsh surface. We recommend that these types of docks not be permitted in the future.

In general, docks should be maintained at the minimum size possible to limit environmental effects and impact on scenic vistas. Longer docks have greater shading effects and leach more contaminants into the environment than short docks. For example, a 100-m long dock will have roughly twice the shading effects and contaminant leaching as a 50-m long dock. Therefore, it is reasonable for SCDHEC-OCRM to limit the length of docks in South Carolina. Based on the dock lengths that were permitted in 2000, the new length restrictions would affect less than 14% of potential property owners (Chinnis, The Post and Courier April 9, 2001). Based on some limited information presented in this study, jumping small creeks, if allowed, would result in environmental impacts on more than one water body and may affect navigation and human access to valued habitats. SCDHEC-OCRM should continue limiting this type of activity.

The small tidal creeks (generally <6 m or 20 ft wide) that form the primary link between uplands and the estuary are repositories for pollutants including those that leach from dock structures (Sanger *et al.* 1999a and 1999b, Lerberg *et al.* 2000) and are critical habitats including nursery grounds for the earliest life stages of many valued resources (e.g., Hackney *et al.* 1976, Weinstein 1979, Wenner and Beatty 1993). These creeks are mostly intertidal environments that support only marginal boating and fishing opportunities. SCDHEC-OCRM's has proposed regulations to limit dock construction in the upper portions of these creeks (i.e., < 20 ft wide). This is a reasonable regulation based on the current science. Large creeks with expansive salt marshes clearly have greater capacity to dilute contaminants than do small creeks with low volume and limited salt marsh habitat. As a result, larger creeks have the capacity to support higher densities of docks with less ecological impact.

A small crabbing dock would have the same shading effect and leachate releases as an equivalent sized "standard" dock. In addition, a property owner with a crabbing dock may be more likely to moor a boat on the marsh adjacent to a crabbing dock, thereby adversely affecting marsh vegetation and/or navigation in small creeks. Therefore, crabbing docks appear to have the same environmental costs as dock constructed for human access. Based on the information obtained by this study, there appears to be little advantage to only allowing crabbing docks in small tidal creeks.

The environmental condition of a creek and thus its carrying capacity for human development and use is influenced by the degree and type of watershed development, creek and watershed size, the natural environmental setting and hydrographic processes. The impact of docks relative to other human uses of tidal creek ecosystems appears to be small and the current density of docks in the state is low. Therefore, estimation of the carrying capacity of a creek for human development should consider all possible factors including but not limited to dock structures.

While the shading effects, the amount of CCA loadings, and the PAH loadings from an average dock and boat can be estimated, we chose not to make these calculations. They would provide little useful information and may mislead mangers and the public into perceiving that the impacts of a single dock are so small that almost any dock should be allowed. The ecological consequences of the average dock should not be evaluated independently of the level of landscape development, creek size and dilution capacity, and environmental setting as well as the other docks that are constructed or can potentially be constructed. In addition, many other factors must be considered when permitting a dock, including impacts on navigation, effects on scenic vistas, and effect on public access to valued habitats. SCDHEC-OCRM has a long history of evaluating dock permits in a particular system as a whole rather than one dock at a time.

V. TABLES

Table 1. Sampling information for the SCECAP large tidal creek stations for this study including category, latitude, longitude, county, and area.

Category	Station Code	Latitude	Longitude	County	Area
No Docks	RT00519	32.5506	-80.8343	Beaufort	Pocotaligo River in Haulover Creek
No Docks	RT00528	32.5884	-80.4494	Colleton	Ashepoo River in Mosquito Creek
No Docks	RT00531	32.8994	-79.9011	Berkeley	Wando River in Nowell Creek
No Docks	RT00546	32.1808	-80.8215	Beaufort	Calibogue Sound in Bryan Creek
No Docks	RT00557	32.5057	-80.7580	Beaufort	Whale Branch in Middle Creek
No Docks	RT99006	33.8526	-78.5840	Horry	Near Little River
No Docks	RT99010	32.5063	-80.8020	Beaufort	Tributary of Broad Creek On Hilton Head Island
No Docks	RT99026	33.0843	-79.4201	Charleston	Dupre Creek near McClellanville
Low Docks	RT00502	32.6066	-80.5369	Colleton	Old Chehaw River below Social Hall Creek
Low Docks	RT00504	32.4153	-80.5978	Beaufort	Warsaw Island in Jenkins Creek
Low Docks	RT00520	32.8143	-79.7547	Charleston	Goat Island
Low Docks	RT00523	32.5042	-80.3058	Colleton	Edisto Island in creek behind island
Low Docks	RT00526	32.8926	-80.1079	Charleston	Ashley River upriver of Magnolia Plantation
Low Docks	RT00542	32.6465	-80.0576	Charleston	Kiawah River in Chapin Creek
Low Docks	RT99022	32.1578	-80.7882	Beaufort	Tributary of Broad Creek On Hilton Head Island
Low Docks	RT99040	32.3929	-80.6413	Beaufort	Cowen Creek in Beaufort River
High Docks	RT00503	32.5996	-80.2028	Charleston	North Edisto River in Adams Creek
High Docks	RT00545	33.8437	-78.6066	Horry	Town of Cherry Grove Beach near mouth of Hog Inlet
High Docks	RT00549	32.8650	-79.9219	Berkeley	Cooper River in Beresford Creek
High Docks	RT00550	33.5658	-79.0210	Georgetown	Murrell's Inlet in upper reach
High Docks	RT99005	32.4404	-80.6522	Beaufort	In Beaufort River near City of Beaufort
High Docks	RT99007	32.7162	-79.9325	Charleston	Creek on James Island in Clark Sound behind Waites Island
High Docks	RT99009	32.5579	-80.3618	Charleston	Bailey Creek in South Edisto River
High Docks	RT99017	32.8247	-79.8667	Charleston	Hobcaw Creek in Wando River
High Docks	RT99027	32.8934	-79.9069	Charleston	Nowell Creek in Wando River
High Docks	RT99030	32.3885	-80.6334	Beaufort	Cowen Creek in Beaufort River

Table 2. Summary of the test results from the Shapiro-Wilks test of the TCP sediment quality data set for transformed data (log base 10 or arcsine square root). The closer the w statistic is to 1, the more normal the data are distributed. The closer the p-value is to zero, the less normal the data are distributed. Empty cells indicate the values were not high enough for a statistical analysis.

Parameter Group	Parameter	n	Normality (w)	Model p-value	Transformation Applied
Metal	Aluminum	28	0.8645	0.0019	log10(x)
	Arsenic	28	0.9062	0.0160	log10(x)
	Cadmium	28	0.9851	0.9502	log10(x)
	Chromium	28	0.8965	0.0095	log10(x)
	Copper	28	0.8961	0.0093	log10(x)
	Iron	28	0.8920	0.0075	log10(x)
	Lead	28	0.9200	0.0347	log10(x)
	Manganese	28	0.9885	0.9856	log10(x)
	Mercury	28	0.9468	0.1644	log10(x)
	Nickel	28	0.8401	0.0006	log10(x)
	Selenium	28	0.8952	0.0089	log10(x)
	Silver	28	0.9033	0.0137	log10(x)
	Tin	28	0.6296	<0.0001	log10(x)
	Zinc	28	0.9224	0.0397	log10(x)
PAH	1-Methylnaphthalene	28	0.9511	0.2105	log10(x)
	1-Methylphenanthrene	28	0.7311	<0.0001	log10(x)
	2,3,5 Trimethylnaphthalene	28	0.7435	<0.0001	log10(x)
	2,6 Dimethylnaphthalene	28	0.9378	0.0970	log10(x)
	2-Methylnaphthalene	28	0.9162	0.0279	log10(x)
	Acenaphthene	28	0.6712	<0.0001	log10(x)
	Acenaphthylene	28	0.7413	<0.0001	log10(x)
	Anthracene	28	0.9029	0.0134	log10(x)
	Benz(a)anthracene	28	0.9176	0.0303	log10(x)
	Benzo(a)pyrene	28	0.8987	0.0106	log10(x)
	Benzo(b)fluoranthene	28	0.9441	0.1402	log10(x)
	Benzo(e)pyrene	28	0.9017	0.0126	log10(x)
	Benzo(g,h,i)perylene	28	0.9438	0.1383	log10(x)
	Benzo(k)fluoranthene	28	0.9333	0.0745	log10(x)
	Biphenyl	28	0.8946	0.0086	log10(x)
	Chrysene	28	0.9597	0.3428	log10(x)
	Dibenz(a,h)anthracene	28	0.3589	<0.0001	log10(x)
	Fluoranthene	28	0.9210	0.0367	log10(x)
	Fluorene	28	0.8400	0.0006	log10(x)
	Indeno(1,2,3-cd)pyrene	28	0.8936	0.0081	log10(x)
	Naphthalene	28	0.9327	0.0720	log10(x)
	Perylene	28	0.9306	0.0637	log10(x)
	Phenanthrene	28	0.8713	0.0026	log10(x)
	Pyrene	28	0.9344	0.0794	log10(x)
	Low Molecular Weight PAHs	28	0.8518	0.0010	log10(x)
	High Molecular Weight PAHs	28	0.9226	0.0430	log10(x)
	Total Polycyclic Aromatic Hydrocarbons	28	0.8861	0.0055	log10(x)
PCB	Total Polychlorinated Biphenyls	28	0.9058	0.0157	log10(x)
Composition	Percent Clay	28	0.9186	0.0321	Arcsine
	Percent Silt/Clay	28	0.8695	0.0024	Arcsine
	Percent Sand	28	0.8697	0.0024	Arcsine
	Percent Silt	28	0.9562	0.2827	Arcsine
	Total Organic Carbon	28	0.9272	0.0524	Arcsine
	Total Organic Nitrogen	28	0.9172	0.0296	Arcsine

Table 3. Summary of the test results from the Shapiro-Wilks test of the TCP biological quality data set for transformed data (log base 10 or arcsine square root). The closer the w statistic is to 1, the more normal the data are distributed. The closer the p-value is to zero, the less normal the data are distributed. Empty cells indicate the values were not high enough for a statistical analysis.

Study	Parameter Group	Parameter	n	Normality (w)	Model p-value	Transformation Applied
1994	Benthic Community	*Streblospio benedicti*	264	0.7393	<0.0001	log10(x+1)
		Capitella capitata	264	0.4912	<0.0001	log10(x+1)
		Heteromastus filiformis	264	0.5624	<0.0001	log10(x+1)
		Laeonereis culveri	264	0.6126	<0.0001	log10(x+1)
		Neanthes succinea	264	0.6218	<0.0001	log10(x+1)
		Polydora cornuta	264	0.1338	<0.0001	log10(x+1)
		Tubificoides heterochaetus	264	0.5981	<0.0001	log10(x+1)
		Tubificidae	264	0.5034	<0.0001	log10(x+1)
		Paranais litoralis	264			
		Tubificoides brownae	264	0.6217	<0.0001	log10(x+1)
		Monopylephorus rubroniveus	264	0.7708	<0.0001	log10(x+1)
		Species Richness	264	0.9244	<0.0001	Arcsine
		Total Abundance	264	0.7557	<0.0001	Arcsine
		Stress Sensitive	264	0.8202	<0.0001	Arcsine
		Stress Tolerant	264	0.8003	<0.0001	Arcsine
2000	Benthic Community	*Streblospio benedicti*	42	0.8229	<0.0001	log10(x+1)
		Capitella capitata	42	0.5995	<0.0001	log10(x+1)
		Heteromastus filiformis	42	0.3986	<0.0001	log10(x+1)
		Laeonereis culveri	42	0.7119	<0.0001	log10(x+1)
		Neanthes succinea	42	0.7161	<0.0001	log10(x+1)
		Polydora cornuta	42	0.6736	<0.0001	log10(x+1)
		Tubificoides heterochaetus	42	0.5055	<0.0001	log10(x+1)
		Tubificidae	42	0.7681	<0.0001	log10(x+1)
		Paranais litoralis	42	0.7904	<0.0001	log10(x+1)
		Tubificoides brownae	42	0.5126	<0.0001	log10(x+1)
		Monopylephorus rubroniveus	42	0.8279	<0.0001	log10(x+1)
		Species Richness	42	0.9112	0.0032	Arcsine
		Total Number	42	0.9895	0.9621	log10(x+1)
		Stress Sensitive	42	0.8292	<0.0001	Arcsine
		Stress Tolerant	42	0.8916	0.0008	Arcsine
1994	Fish/Crustacean Community	*Anchoa mitchelli* (#)	41	0.6188	<0.0001	log10(x+1)
		Fundulus heterochaetus (#)	41	0.5318	<0.0001	log10(x+1)
		Grass Shrimp (#)	41	0.8199	<0.0001	log10(x+1)
		Panaeid Shirmp (#)	41	0.8050	<0.0001	log10(x+1)
		Species Richness	41	0.9479	0.0589	Arcsine
		Total Abundance	41	0.9582	0.1359	log10(x+1)
		Anchoa mitchelli (Biomass)	41	0.5272	<0.0001	log10(x+1)
		Fundulus heterochaetus (Biomass)	41	0.6081	<0.0001	log10(x+1)
		Grass Shrimp (Biomass)	41	0.7238	<0.0001	log10(x+1)
		Panaeid Shirmp (Biomass)	41	0.8158	<0.0001	log10(x+1)
		Total Biomass	41	0.9771	0.5685	Arcsine

Table 4. Summary of the test results from the Shapiro-Wilks test of the SCECAP sediment quality data set for transformed data (log base 10 or arcsine square root). The closer the w statistic is to 1, the more normal the data are distributed. The closer the p-value is to zero, the less normal the data are distributed. Empty cells indicate the values were not high enough for a statistical analysis.

Parameter Group	Parameter	n	Normality (w)	Model p-value	Transformation Applied
Metals	Aluminum	26	0.9683	0.5787	log10(x+1)
	Arsenic	26	0.9338	0.0955	log10(x+1)
	Cadmium	26	0.5544	<0.0001	log10(x+1)
	Chromium	26	0.8540	0.0017	log10(x+1)
	Copper	26	0.9471	0.3373	log10(x+1)
	Iron	26	0.9568	0.3325	log10(x+1)
	Mercury	26	0.7048	<0.0001	log10(x+1)
	Manganese	26	0.9565	0.3270	log10(x+1)
	Nickel	26	0.9748	0.7495	log10(x+1)
	Lead	26	0.9732	0.7063	log10(x+1)
	Selenium	26	0.6771	<0.0001	log10(x+1)
	Silver	26	0.1980	<0.0001	log10(x+1)
	Tin	26	0.4340	<0.0001	log10(x+1)
	Zinc	26	0.8231	0.0004	log10(x+1)
PAH	1,6,7 Trimethylnaphthalene	26	0.3075	<0.0001	log10(x+1)
	1-Methylnaphthalene	26	0.3853	<0.0001	log10(x+1)
	1-Methylphenanthrene	26			
	2,6 Dimethylnaphthalene	26	0.3085	<0.0001	log10(x+1)
	2-Methylnaphthalene	26	0.3843	<0.0001	log10(x+1)
	Acenaphthene	26			
	Acenaphthylene	26	0.3046	<0.0001	log10(x+1)
	Anthracene	26	0.4509	<0.0001	log10(x+1)
	Benzo(a)anthracene	26	0.5135	<0.0001	log10(x+1)
	Benzo(a)pyrene	26	0.4603	<0.0001	log10(x+1)
	Benzo(b)fluoranthene	26	0.6924	<0.0001	log10(x+1)
	Benzo(e)pyrene	26	0.6589	<0.0001	log10(x+1)
	Benzo(g,h,i)perylene	26	0.4604	<0.0001	log10(x+1)
	Biphenyl	26	0.1980	<0.0001	log10(x+1)
	Benzo(j+k)fluoranthene	26	0.6599	<0.0001	log10(x+1)
	Chrysene+Triphenylene	26	0.7713	<0.0001	log10(x+1)
	Dibenz(a,h+a,c)anthracene	26	0.3923	<0.0001	log10(x+1)
	Fluoranthene	26	0.8341	0.0007	log10(x+1)
	Fluorene	26	0.3084	<0.0001	log10(x+1)
	Indeno(1,2,3-cd)pyrene	26	0.4594	<0.0001	log10(x+1)
	Naphthalene	26	0.3909	<0.0001	log10(x+1)
	Perylene	26	0.7515	<0.0001	log10(x+1)
	Phenanthrene	26	0.7509	<0.0001	log10(x+1)
	Pyrene	26	0.8650	<0.0001	log10(x+1)
	Low Molecular Weight PAHs	26	0.7545	<0.0001	log10(x+1)
	High Molecular Weight PAHs	26	0.9053	0.0206	log10(x+1)
	Total Polycyclic Aromatic Hydrocarbons	26	0.8997	0.0154	log10(x+1)
PCBs	Total Polychlorinated Biphenyls	26	0.8249	0.0005	log10(x+1)
Composition	Clay	26	0.8540	0.0017	Arcsine
	Sand	26	0.8575	0.0020	Arcsine
	Silt	26	0.7839	<0.0001	Arcsine
	Silt/Clay	26	0.8575	0.0020	Arcsine
	TOC	26	0.7954	0.0001	Arcsine
Pore Water	Total Ammonia	26	0.9786	0.8419	log10(x+1)
	Unionized Ammonia	26	0.8570	0.0020	log10(x+1)

Table 5. Summary of the test results from the Shapiro-Wilks test of the SCECAP water quality data set for transformed data (log base 10 or arcsine square root). The closer the w statistic is to 1, the more normal the data are distributed. The closer the p-value is to zero, the less normal the data are distributed.

Parameter Group	Parameter	n	Normality (w)	Model p-value	Transformation Applied
Dissolved Nutrients	Dissolved inorganic carbon	15	0.9401	0.3836	log10(x+1)
	Dissolved ammonia	29	0.9568	0.2733	log10(x+1)
	Dissolved nitrate/nitrite	30	0.7150	<0.0001	log10(x+1)
	Dissolved organic carbon	15	0.9249	0.2290	log10(x+1)
	Dissolved organic carbon	15	0.9299	0.2723	log10(x+1)
	Dissolvd organic nitrogen	15	0.8708	0.0347	log10(x+1)
	Dissolved organic phosphate	15	0.8339	0.0104	log10(x+1)
	Dissolved silicon/silica	30	0.7940	<0.0001	log10(x+1)
	Orthophosphate	30	0.9477	0.1463	log10(x+1)
	Total dissolved nitrogen	15	0.8928	0.0738	log10(x+1)
	Total dissolved phosphate	15	0.9767	0.9415	log10(x+1)
	Total nitrogen	13	0.4570	<0.0001	log10(x+1)
	Total phosphate	15	0.7325	0.0006	log10(x+1)
Nutrients	Total Ammonia Nitrogen mg/l as N	21	0.8453	0.0035	log10(x+1)
	Total Kjedahl Nitrogen mg/l as N	25	0.9585	0.3852	log10(x+1)
	Nitrite + Nitrate mg/l as N	24	0.7974	0.0003	log10(x+1)
	Total Phosphorus mg/l as P	25	0.8179	0.0005	log10(x+1)
	Total Organic Carbon mg/l as C	26	0.8489	0.0014	log10(x+1)
Pigments	Chlorophyll a	24	0.9261	0.0799	log10(x+1)
Oyxgen Demand	Biological Oxygen Demand	26	0.8848	0.0073	log10(x+1)
Water Column	Dissolved Oxygen	92	0.9680	0.0232	log10(x+1)
	pH	41	0.9146	0.0046	log10(x+1)
	Salinity	92	0.5689	<0.0001	log10(x+1)
	Temperature	92	0.9668	0.0191	log10(x+1)
	Secchi depth reading	29	0.8364	0.0004	log10(x+1)
	Turbidity	26	0.9250	0.0588	log10(x+1)
Fecal Coliform	Fecal Coliform A-1 procedure	25	0.9476	0.2212	log10(x+1)

Table 6. Summary of the test results from the Shapiro-Wilks test of the SCECAP biological quality data set for transformed data (log base 10 or arcsine square root). The closer the w statistic is to 1, the more normal the data are distributed. The closer the p-value is to zero, the less normal the data are distributed.

Parameter Group	Parameter	n	Normality (w)	Model p-value	Transformation Applied
Toxicity	Clam Growth (ug/day)	26	0.9587	0.3674	Not Transformed
	Amphipod Survival	15	0.9084	0.1280	log10(x+1)
	Microtox EC50 (%)	26	0.6467	<0.0001	Arcsine
	Clam Survival	26	0.6823	<0.0001	log10(x+1)
Benthic Community	*Ilyanassa obsoleta*	52	0.5448	<0.0001	log10(x+1)
	Exogone sp.	52	0.4268	<0.0001	log10(x+1)
	Mediomastus ambiseta	52	0.7222	<0.0001	log10(x+1)
	Mediomastus sp.	52	0.7144	<0.0001	log10(x+1)
	Streblospio benedicti	52	0.8408	<0.0001	log10(x+1)
	Spiochaetopterus costarum oculatus	52	0.7200	<0.0001	log10(x+1)
	Heteromastus filiformis	52	0.7712	<0.0001	log10(x+1)
	Scoletoma tenuis	52	0.7942	<0.0001	log10(x+1)
	Tubificoides heterochaetus	52	0.3353	<0.0001	log10(x+1)
	Tharyx acutus	52	0.6899	<0.0001	log10(x+1)
	Polydora cornuta	52	0.7131	<0.0001	log10(x+1)
	Tubificoides brownae	52	0.8155	<0.0001	log10(x+1)
	Cirratulidae	52	0.6444	<0.0001	log10(x+1)
	Species Richness	52	0.9555	0.0502	log10(x+1)
	Total Abundance	52	0.9720	0.2554	log10(x+1)
	Index of Biotic Integrity - CP	52	0.8409	0.0006	log10(x)
	Index of Biotic Integrity - SC	52	0.7666	<0.0001	log10(x)
Fish/Crustaceans Community	*Anchoa mitchilli* (#)	52	0.7398	<0.0001	log10(x+1)
	Bairdiella chrysoura (#)	52	0.8409	<0.0001	log10(x+1)
	Leiostomus xanthurus (#)	52	0.8739	<0.0001	log10(x+1)
	Penaeus aztecus (#)	52	0.7149	<0.0001	log10(x+1)
	Penaeus setiferus (#)	52	0.8285	<0.0001	log10(x+1)
	Lolliguncula brevis (#)	52	0.8142	<0.0001	log10(x+1)
	Species Richness	52	0.8102	0.2555	log10(x+1)
	Total Abundance	52	0.8476	<0.0001	log10(x+1)
	Anchoa mitchilli (Biomass)	52	0.5791	<0.0001	log10(x+1)
	Bairdiella chrysoura (Biomass)	52	0.7925	<0.0001	log10(x+1)
	Leiostomus xanthurus (Biomass)	52	0.8351	<0.0001	log10(x+1)
	Penaeus aztecus (Biomass)	52	0.5272	<0.0001	log10(x+1)
	Penaeus setiferus (Biomass)	52	0.7291	<0.0001	log10(x+1)
	Lolliguncula brevis (Biomass)	52	0.6177	<0.0001	log10(x+1)
	Total Biomass	52	0.9644	0.1217	log10(x+1)

Table 7. Summarization (average, one standard deviation, minimum, maximum) of the dock characteristics measured during the South Carolina *Spartina* Shading Study.

Parameter	Mean	Standard Deviation	Minimum	Maximum
Length (m)	77.96	57.92	10	191.8
Width (m)	1.22	0.13	0.9	1.54
Height at the upland (m)	1.23	0.27	0.7	2.05
Height at the berm (m)	1.37	0.25	1	1.8
Space between planks (cm)	1.16	0.55	0.4	2.7
Orientation (Degrees)	171.72	72.47	59	334
Density under dock (#/0.1m^2)	10.14	7.98	8	45
Density next to dock (#/0.1m^2)	43.27	33.89	0	159
% Reduction	71.12	24.47	-38 *	100

* One density measurement under the dock was greater than next to the dock.

Table 8. Summarization of dock information for the six small tidal creeks from the TCP used for projecting the effects of shading by dock structures on *Spartina* stem density.

Creek Name	Marsh Area (ha)	Number of Docks in 1999	Average Dock Length (m)	Percent of Available Marsh Impacted (%)
Horlbeck Creek	11.4	14	53.9	0.72
Parrot Creek	14.5	32	27.1	0.52
Metcalfs Creek	7.3	13	4.6	0.03
Long Creek	20.7	4	57.6	0.11
Shem Creek	12.2	12	23.4	0.26
Yacht Club Creek	5.3	13	20.5	0.28
Average				0.38

Table 9. Summarization of Scenario 1 for six example tidal creeks evaluated to assess the effects of dock shading on *Spartina alterniflora* for the maximum number of docks that could theoretically be constructed along each creek.

Creek Example	Marsh Area (ha)	Maximum Number of Docks Possible	Average Dock Length (m)	Percent of Available Marsh Impacted (%)
Creek 1	11.4	77	53.9	3.96
Creek 2	14.5	144	27.1	2.32
Creek 3	7.3	73	4.6	0.18
Creek 4	20.7	192	57.6	5.45
Creek 5	12.2	92	23.4	1.97
Creek 6	5.3	83	20.5	1.75
Average				2.61

Table 10. The number of docks (private and community) permitted by SC Ocean and Coastal Resource Management in the last ten years. Blank cells indicate the data was not available.

Year	Charleston	Dorchester	Berkeley	Beaufort	Colleton	Jasper	Georgetown	Horry	Totals
2000									717
1999									
1998	351	1	12	248	4	10	30	13	669
1997	405	1	7	168	23	4	25	19	652
1996	442	8	11	180	22	17	43	42	765
1995	366	2	8	144	34	9	68	70	701
1994	316	8	11	256	31	10	44	34	710
1993	281	ND	9	167	11	4	39	24	535
1992	310	ND	9	163	22	3	38	36	581
1991	381	ND	2	148	10	6	67	40	654
Sum	2852	20	69	1474	157	63	354	278	5984
Average	356.5	4	8.625	184.25	19.625	7.875	44.25	34.75	665
Average*10yrs	3565	40	86.25	1842.5	196.25	78.75	442.5	347.5	6649
Shapefile 1991-2000	3952	35	78	1550	182	70	1053	327	7247

Table 11. Summarization of the amount of salt marsh that is potentially affected in 2000 and 2010 for each county and the state at average dock lengths of 25 m and 305 m. Salt marsh area includes brackish water, and low and high salt marshes reported by Tiner (1977).

				25 m length		305 m length	
County	Total Marsh (m^2)	Estimation of Docks in 2000	Estimation of Docks in 2010	Percent of Available Marsh Impacted in 2000	Percent of Available Marsh Impacted in 2010	Percent of Available Marsh Impacted in 2000	Percent of Available Marsh Impacted in 2010
Horry	8,745,293	327	654	0.08	0.16	0.98	1.98
Georgetown	83,122,771	1052	2104	0.03	0.05	0.33	0.67
Charleston	576,278,762	3952	7904	0.01	0.03	0.18	0.36
Berkeley	29,347,923	78	156	0.01	0.01	0.07	0.14
Dorchester	1,776,577	35	70	0.04	0.09	0.52	1.04
Colleton	124,000,236	182	364	0.00	0.01	0.04	0.08
Beaufort	526,154,193	1550	3100	0.01	0.01	0.08	0.16
Jasper	145,744,084	70	140	0.00	0.00	0.01	0.03
Total	1,495,169,839	7246	14492	0.01	0.02	0.13	0.26

Table 12. Summarization of the amount of salt marsh that is affected for different dock lengths for the state of SC. See Table 11 for total salt marsh area in the state.

Average Dock Length (m)	Percent of Available Marsh Impacted in 2000	Percent of Available Marsh Impacted in 2010
25	0.01	0.02
50	0.02	0.04
100	0.04	0.08
150	0.06	0.12
200	0.08	0.16
250	0.10	0.20
300	0.12	0.24

Table 13. The number of docks and land cover in each creek watershed and the average for each category from the TCP small tidal creek data set by year.

Year	Category	Creek Name	Number of Docks	Watershed Size (hectares)	Agriculture (%)	Barren (%)	Forest (%)	Urban (%)	Water (%)	Wetlands (%)	Impervious Surface (%)
1992	Reference	Beresford Creek	0	25.1	0.0	0.0	20.7	0.0	0.0	79.3	0.0
		Deep Creek	0	64.7	19.6	0.0	61.8	0.0	0.0	18.5	4.1
		Fosters Creek	0	16.0	0.0	0.0	21.3	0.0	0.0	78.8	0.0
		Horlbeck Creek	1	237.5	9.4	7.5	77.7	0.0	1.5	3.8	1.8
		Long Creek	0	412.4	24.7	10.6	59.0	1.5	0.0	4.3	2.2
		Lachicotte Creek	0	13.1	0.0	0.0	14.5	0.0	0.0	85.5	0.0
		Lighthouse Creek	0	37.3	0.0	8.0	7.2	0.0	0.0	84.7	0.0
		Rathall Creek	0	72.1	0.0	9.0	42.0	0.0	0.0	49.0	0.8
		Average	0.125	109.8	6.7	4.4	38.0	0.2	0.2	50.5	1.1
	Suburban - No Dock	Bull Creek	0	434.3	0.0	0.8	19.2	71.7	3.6	4.7	28.9
		Cross Creek	0	312.9	2.8	13.1	9.1	72.5	0.8	1.7	26.4
		Average	0.0	373.6	1.4	7.0	14.2	72.1	2.2	3.2	27.7
	Suburban - Dock	Metcalfs Creek	13	129.7	0.0	13.2	13.4	68.0	0.0	5.4	30.8
		Parrot Creek	28	147.2	5.3	20.8	16.0	47.1	1.0	9.9	19.5
		Shem Creek	8	427.8	0.0	15.6	14.4	66.9	1.0	2.2	34.5
		Yacht Club Creek	13	69.1	0.0	2.6	6.7	77.4	1.4	11.9	14.9
		Average	15.5	193.5	1.3	13.0	12.6	64.8	0.8	7.3	24.9
1999	Reference	Beresford Creek	0	24.9	0.0	0.0	16.4	0.0	0.0	83.6	0.0
		Deep Creek	0	64.7	14.8	0.0	67.3	0.0	0.0	17.9	1.1
		Fosters Creek	0	16.1	0.0	0.0	6.4	8.2	0.0	85.4	5.6
		Long Creek	0	413.1	22.7	0.4	67.8	4.0	0.2	5.0	3.7
		Lachicotte Creek	0	13.0	0.0	0.0	12.5	0.0	0.0	87.5	0.0
		Lighthouse Creek	0	37.3	0.0	6.5	5.9	0.0	0.0	87.6	0.0
		Rathall Creek	0	72.1	0.0	3.7	47.8	0.0	0.0	48.4	1.3
		Average	0.0	91.6	5.4	1.5	32.0	1.7	0.0	59.3	1.7
	Suburban - No Dock	Bull Creek	0	436.2	0.0	0.3	17.4	74.3	3.5	4.5	44.1
		Cross Creek	0	306.4	2.1	3.8	18.6	72.3	0.7	2.2	28.8
		Average	0.0	371.3	1.1	2.0	18.0	73.3	2.1	3.3	36.5
	Suburban - Dock	Horlbeck Creek	10	237.1	1.0	1.7	57.8	32.0	2.6	4.8	16.0
		Metcalfs Creek	13	122.3	0.0	11.0	21.4	61.4	0.2	6.0	42.2
		Parrot Creek	32	146.1	0.3	21.0	17.4	49.7	1.6	9.9	27.1
		Shem Creek	12	429.0	0.0	1.4	18.2	75.4	2.2	2.8	53.7
		Yacht Club Creek	13	69.1	0.0	1.8	11.9	77.2	1.4	7.6	24.0
		Average	16.0	200.7	0.3	7.4	25.4	59.2	1.6	6.2	32.6

Table 14. Summary of the ANOVA/ANCOVA analyses for the TCP sediment quality data set. The r^2 indicates the amount of variation explained by the model. The smaller the p-value, the stronger the evidence against the lack of an effect. A blank cell indicates the covariate was not significant at $\alpha = 0.15$. The treatment categories are listed from highest level to lowest level and are abbreviated as follows: (1) Ref = Reference, (2) SN = Suburban – No Dock, and (3) SD = Suburban – Dock. The categories are different at $\alpha = 0.05$ if they do not share a common line.

Parameter Type	Parameter Name	Covariate	Model r^2	Model p-value	Category p-value	Covariate p-value	Category Differences
Metals	Aluminum	Clay	0.84	<0.0001	0.1725	<0.0001	Ref SN SD
	Arsenic	Clay	0.46	0.0017	0.7473	0.0004	SD Ref SN
	Cadmium	Clay	0.39	0.0070	0.0360	0.0058	SN SD Ref
	Chromium	Clay	0.88	<0.0001	0.1205	<0.0001	Ref SN SD
	Copper	Clay	0.64	<0.0001	0.5747	<0.0001	SN SD Ref
	Iron	Clay	0.78	<0.0001	0.4427	<0.0001	Ref SN SD
	Lead	Clay	0.60	<0.0001	0.0049	<0.0001	SN SD Ref
	Manganese	Clay	0.44	0.0026	0.3063	0.0008	SN SD Ref
	Mercury	Clay	0.53	0.0004	0.0421	0.0001	Ref SD SN
	Nickel	Clay	0.81	<0.0001	0.6242	<0.0001	Ref SN SD
	Selenium	Clay	0.19	0.1564	0.5241	0.0649	Ref SD SN
	Silver		0.20	0.0624	0.0624		SD SN Ref
	Tin	Clay	0.27	0.0537	0.8524	0.0084	SN SD Ref
	Zinc	Clay	0.74	<0.0001	0.0978	<0.0001	SN SD Ref
PAHs	1-Methylnaphthalene	TOC	0.40	0.0059	0.5291	0.0006	SN SD Ref
	1-Methylphenanthrene		0.30	0.0126	0.0126		SD SN Ref
	2,3,5 Trimethylnaphthalene		0.09	0.2921	0.2921		SN SD Ref
	2,6 Dimethylnaphthalene	TOC	0.11	0.4289	0.7252	0.1205	SN SD Ref
	2-Methylnaphthalene	TOC	0.29	0.0395	0.7789	0.0047	SN SD Ref
	Acenaphthene		0.39	0.0020	0.0020		SD SN Ref
	Acenaphthylene		0.13	0.1676	0.1676		SD SN Ref
	Anthracene		0.62	<0.0001	<0.0001		SD SN Ref
	Benz(a)anthracene	TOC	0.47	0.0015	0.0006	0.1372	SD SN Ref
	Benzo(a)pyrene	TOC	0.49	0.0010	0.0004	0.1276	SD SN Ref
	Benzo(b)fluoranthene	TOC	0.56	0.0002	<0.0001	0.0491	SD SN Ref
	Benzo(e)pyrene	TOC	0.52	0.0005	0.0002	0.0418	SD SN Ref
	Benzo(g,h,i)perylene	TOC	0.51	0.0006	0.0003	0.0376	SD SN Ref
	Benzo(k)fluoranthene	TOC	0.56	0.0001	<0.0001	0.1079	SD SN Ref
	Biphenyl	TOC	0.34	0.0174	0.5434	0.0021	SD SN Ref
	Chrysene	TOC	0.56	0.0002	<0.0001	0.0994	SD SN Ref
	Dibenz(a,h)anthracene		0.13	0.1709	0.1709		SD Ref SN

Table 14. Continued.

Parameter Type	Parameter Name	Covariate	Model r^2	Model p-value	Category p-value	Covariate p-value	Category Differences
	Fluoranthene	TOC	0.53	0.0004	0.0001	0.1267	SD SN Ref
	Fluorene		0.64	<0.0001	<0.0001		SD SN Ref
	Indeno(1,2,3-cd)pyrene		0.46	0.0005	0.0005		SN SD Ref
	Naphthalene	TOC	0.13	0.3410	0.9551	0.0757	SN SD Ref
	Perylene	TOC	0.43	0.0031	0.0013	0.1407	SD SN Ref
	Phenanthrene		0.45	0.0006	0.0006		SD SN Ref
	Pyrene	TOC	0.55	0.0002	<0.0001	0.0868	SD SN Ref
	Low Molecular Weight PAHs		0.42	0.0011	0.0011		SD SN Ref
	High Molecular Weight PAHs	TOC	0.55	0.0002	<0.0001	0.0884	SD SN Ref
	Total Polycyclic Aromatic Hydrocarbons	TOC	0.53	0.0003	0.0001	0.1191	SD SN Ref
PCBs	Total Polychlorinated Biphenyls		0.17	0.0934	0.0934		SD SN Ref
Composition	Percent Clay		0.06	0.4805	0.4805		SD Ref SN
	Percent Silt/Clay		0.04	0.6245	0.6245		SD Ref SN
	Percent Sand		0.04	0.6244	0.6244		SN Ref SD
	Percent Silt		0.21	0.0515	0.0515		Ref SN SD
	Total Organic Carbon		0.04	0.6316	0.6316		Ref SD SN
	Total Organic Nitrogen		0.02	0.7448	0.7448		SD Ref SN

Table 15. Summary of the test results for the TCP sediment quality data set from the nonparametric Kruskal-Wallis test. The Chi-Square is the test statistic. The smaller the p-value, the stronger the evidence against the lack of an effect. The mean score values are the sum of the rankings for each category.

				Mean Score		
Parameter Type	Parameter	Chi-Square	Model p-value	Reference	Suburban No Dock	Suburban Dock
Metal	Aluminum	0.57	0.7504	14.69	11.75	15.50
	Arsenic	3.89	0.1430	13.38	10.00	19.00
	Cadmium	4.59	0.1008	12.34	22.13	15.00
	Chromium	0.17	0.9202	14.63	13.00	15.00
	Copper	0.36	0.8334	14.06	13.38	15.94
	Iron	1.51	0.4690	14.84	10.00	16.06
	Lead	5.59	0.0611	11.44	20.75	17.50
	Manganese	3.17	0.2047	12.19	16.00	18.38
	Mercury	3.30	0.1923	12.50	20.63	15.44
	Nickel	1.60	0.4490	14.50	10.25	16.63
	Selenium	1.57	0.4572	15.28	9.75	15.31
	Silver	6.43	0.0402	11.44	15.25	20.25
	Tin	0.37	0.8325	13.69	15.75	15.50
	Zinc	2.93	0.2309	12.25	16.25	18.13
PAH	1-Methylnaphthalene	0.16	0.9253	14.25	16.00	14.25
	1-Methylphenanthrene	9.73	0.0077	10.47	19.75	19.94
	2,3,5 Trimethylnaphthalene	1.88	0.3897	13.00	18.50	15.50
	2,6 Dimethylnaphthalene	0.31	0.8549	14.09	16.63	14.25
	2-Methylnaphthalene	0.04	0.9826	14.25	14.75	14.88
	Acenaphthene	10.45	0.0054	11.00	15.25	21.13
	Acenaphthylene	3.47	0.1763	12.59	14.00	18.56
	Anthracene	15.47	0.0004	9.56	16.75	23.25
	Benz(a)anthracene	11.78	0.0028	9.94	18.75	21.50
	Benzo(a)pyrene	13.57	0.0011	9.56	22.25	20.50
	Benzo(b)fluoranthene	14.23	0.0008	9.44	22.25	20.75
	Benzo(e)pyrene	13.31	0.0013	9.63	22.50	20.25
	Benzo(g,h,i)perylene	11.44	0.0033	10.00	22.25	19.63
	Benzo(k)fluoranthene	14.86	0.0006	9.31	21.75	21.25
	Biphenyl	0.21	0.8995	14.03	14.13	15.63
	Chrysene	17.07	0.0002	8.94	21.75	22.00
	Dibenz(a,h)anthracene	2.70	0.2593	13.81	13.00	16.63
	Fluoranthene	15.97	0.0003	9.13	22.25	21.38
	Fluorene	16.20	0.0003	9.34	18.38	22.88
	Indeno(1,2,3-cd)pyrene	13.15	0.0014	9.72	22.50	20.06
	Naphthalene	0.10	0.9513	14.88	14.50	13.75
	Perylene	10.99	0.0041	10.06	21.25	20.00
	Phenanthrene	16.45	0.0003	9.06	20.50	22.38
	Pyrene	16.33	0.0003	9.06	22.25	21.50
	Low MW PAH	11.04	0.0040	10.44	15.25	22.25
	High MW PAH	14.95	0.0006	9.31	22.50	20.88
	Total PAH	15.65	0.0004	9.19	22.50	21.13
PCB	Total PCB	3.94	0.1394	11.88	16.75	18.63
Sediment	Percent Clay	2.68	0.2619	13.06	12.25	18.50
	Percent Mud	1.02	0.6006	14.25	11.50	16.50
	Percent Sand	1.02	0.6006	14.75	17.50	12.50
	Percent Silt	6.33	0.0421	17.88	9.25	10.38
	Total Organic Carbon	1.82	0.4021	16.31	11.75	12.25
	Total Organic Nitrogen	0.94	0.6238	14.41	11.38	16.25

Table 16. Summary of the regression analyses for the TCP sediment quality data set. The r^2 indicates the amount of variation explained by the model. The smaller the p-value, the stronger the evidence against the lack of a slope different than zero. The slope indicates the direction and magnitude of the line.

Parameter Type	Parameter Name	Model r^2	Model p-value	Intercept	Slope
Metal	Aluminum	0.02	0.4641	0.6673	0.0031
	Arsenic	0.07	0.1590	0.9865	0.0072
	Cadmium	0.05	0.2487	0.0561	-0.0007
	Chromium	0.01	0.6403	1.7750	0.0018
	Copper	0.02	0.4391	1.2262	0.0045
	Iron	0.01	0.5556	0.3956	0.0023
	Lead	0.03	0.3674	1.3035	0.0036
	Manganese	0.11	0.0819	2.2244	0.0071
	Mercury	0.00	0.7524	-1.2026	0.0018
	Nickel	0.02	0.4327	1.1723	0.0039
	Selenium	0.01	0.6184	0.1752	0.0017
	Silver	0.11	0.0815	0.0111	0.0003
	Tin	0.02	0.4741	1.4177	0.0058
	Zinc	0.04	0.3262	1.8317	0.0041
PAH	1-Methylnaphthalene	0.00	0.7444	0.8140	0.0012
	1-Methylphenanthrene	0.08	0.1494	0.4732	0.0105
	2,3,5 Trimethylnaphthalene	0.01	0.7176	0.1906	0.0023
	2,6 Dimethylnaphthalene	0.00	0.9337	0.7132	0.0004
	2-Methylnaphthalene	0.01	0.5933	1.0274	0.0022
	Acenaphthene	0.09	0.1268	0.5448	0.0055
	Acenaphthylene	0.07	0.1724	0.1600	0.0080
	Anthracene	0.21	0.0144	0.6443	0.0221
	Benz(a)anthracene	0.06	0.1998	1.3139	0.0159
	Benzo(a)pyrene	0.07	0.1779	1.3924	0.0151
	Benzo(b)fluoranthene	0.07	0.1746	1.6150	0.0173
	Benzo(e)pyrene	0.07	0.1833	1.3458	0.0155
	Benzo(g,h,i)perylene	0.04	0.2801	1.2748	0.0127
	Benzo(k)fluoranthene	0.10	0.1059	1.2291	0.0206
	Biphenyl	0.00	0.9939	0.5920	0.0000
	Chrysene	0.12	0.0733	1.2876	0.0265
	Dibenz(a,h)anthracene	0.01	0.7028	0.1033	0.0036
	Fluoranthene	0.07	0.1616	1.6211	0.0176
	Fluorene	0.23	0.0095	0.3147	0.0246
	Indeno(1,2,3-cd)pyrene	0.04	0.3354	1.2295	0.0130
	Naphthalene	0.00	0.8399	1.1249	0.0009
	Perylene	0.04	0.3128	1.0295	0.0114
	Phenanthrene	0.08	0.1383	1.1609	0.0154
	Pyrene	0.08	0.1447	1.5849	0.0182
	Low MW PAH	0.09	0.1211	1.9075	0.0087
	High MW PAH	0.08	0.1570	2.4480	0.0172
	Total PAH	0.07	0.1588	2.5781	0.0151
PCB	Total PCB	0.01	0.6062	0.8090	0.0025
Sediment	Percent Clay	0.11	0.0821	0.8083	0.0081
	Percent Mud	0.04	0.3368	1.0818	0.0065
	Percent Sand	0.04	0.3368	0.4889	-0.0065
	Percent Silt	0.08	0.1499	0.4824	-0.0034
	Total Organic Carbon	0.01	0.6914	0.2016	-0.0004
	Total Organic Nitrogen	0.01	0.5460	0.0542	0.0002

Table 17. Summary of the ANOVA/ANCOVA analyses for the TCP biological quality data set by year study was conducted. The r^2 indicates the amount of variation explained by the model. The smaller the p-value, the stronger the evidence against the lack of an effect. A blank cell indicates the covariate was not significant at $\alpha = 0.15$. The treatment categories are listed from highest level to lowest level and are abbreviated as follows: (1) Ref = Reference, (2) SN = Suburban – No Dock, and (3) SD = Suburban – Dock. The categories are different at $\alpha = 0.05$ if they do not share a common line.

Study	Parameter Type	Scientific name	Model r^2	Model p-value	Category p-value	Silt/Clay Covariate p-value	Salinity Covariate p-value	Category Differences
1994	Benthic Community	*Streblospio benedicti*	0.15	<0.0001	0.0456		<0.0001	SN Ref SD
		Capitella capitata	0.07	0.0002	0.0014	0.0002		Ref SD SN
		Heteromastus filiformis	0.08	<0.0001	0.0066		<0.0001	SN Ref SD
		Laeonereis culveri	0.44	<0.0001	<0.0001	<0.0001	<0.0001	SD Ref SN
		Neanthes succinea	0.04	0.0296	0.3805	0.0859	0.0243	SN Ref SD
		Polydora cornuta	0.03	0.0488	0.0724		0.0052	SN Ref SD
		Tubificoides heterochaetus	0.32	<0.0001	0.0007	0.0088	<0.0001	Ref SD SN
		Tubificidae	0.04	0.0075	0.0247		0.0039	SN SD Ref
		Paranais litoralis						
		Tubificoides brownae	0.12	<0.0001	<0.0001		0.0570	SN SD Ref
		Monopylephorus rubroniveus	0.07	<0.0001	<0.0001			SD Ref SN
		Species Richness	0.63	0.0055	0.2707	0.0081	0.0545	SN SD Ref
		Total Abundance	0.12	<0.0001	0.0020	<0.0001		SD Ref SN
		Stress Sensitive	0.07	<0.0001	<0.0001			SN Ref SD
		Stress Tolerant	0.10	<0.0001	<0.0001			SD SN Ref
2000	Benthic Community	*Streblospio benedicti*	0.26	0.0228	0.3257	0.0201	0.1251	SN Ref SD
		Capitella capitata	0.09	0.2804	0.2914		0.0872	SN Ref SD
		Heteromastus filiformis	0.18	0.0512	0.1079		0.1277	Ref SN SD
		Laeonereis culveri	0.32	0.0006	0.0006			SN Ref SD
		Neanthes succinea	0.09	0.2927	0.4372	0.0666		SN SD Ref
		Polydora cornuta	0.29	0.0041	0.0098		0.0007	SN SD Ref
		Tubificoides heterochaetus	0.54	<0.0001	0.0582	0.0175	<0.0001	Ref SN SD
		Tubificidae	0.11	0.1003	0.1003			SN Ref SD
		Paranais litoralis	0.25	0.0268	0.3703	0.0637	0.0190	Ref SD SN
		Tubificoides brownae	0.09	0.2928	0.1854		0.1323	SN SD Ref
		Monopylephorus rubroniveus	0.20	0.0726	0.7082	0.0315	0.0426	SD Ref SN
		Species Richness	0.22	0.0203	0.0152	0.0788		SN Ref SD
		Total Abundance	0.00	0.9232	0.9232			Ref SN SD
		Stress Sensitive	0.11	0.1097	0.1097			Ref SN SD
		Stress Tolerant	0.04	0.4499	0.4499			SN SD Ref

Table 17. Continued.

Study	Parameter Type	Scientific name	Model r^2	Model p-value	Category p-value	Silt/Clay Covariate p-value	Salinity Covariate p-value	Category Differences
1994	Fish and Crustacean	*Anchoa mitchelli* (#)	0.09	0.1614	0.1614			SD Ref SN
		Fundulus heterochaetus (#)	0.07	0.2306	0.2306			Ref SD SN
		Grass Shrimp (#)	0.05	0.3952	0.3952			SD Ref SN
		Panaeid Shirmp (#)	0.20	0.0398	0.3817	0.1008		Ref SD SN
		Species Richness	0.09	0.1526	0.1526			SD SN Ref
		Total Abundance	0.15	0.0487	0.0487			SD Ref SN
		Anchoa mitchelli (Biomass)	0.10	0.2506	0.9055	0.1223		Ref SD SN
		Fundulus heterochaetus (Biomass)	0.07	0.2687	0.2687			Ref SD SN
		Grass Shrimp (Biomass)	0.04	0.4940	0.4940			SD Ref SN
		Panaeid Shirmp (Biomass)	0.13	0.0658	0.0658			Ref SD SN
		Total Biomass	0.18	0.0572	0.5581	0.0799		SD Ref SN

Table 18. Summary of the test results for the TCP biological quality data set by year study was conducted from the nonparametric Kruskal-Wallis test. The Chi-Square is the test statistic. The smaller the p-value, the stronger the evidence against the lack of an effect. The mean score values are the sum of the rankings for each category.

					Mean Score		
Study	Parameter Group	Parameter	Chi-Square	Model p-value	Reference	Suburban No Dock	Suburban Dock
1994	Benthic Community	*Streblospio benedicti*	15.30	0.0005	142.4	94.3	135.4
		Capitella capitata	5.44	0.0659	139.2	120.8	127.4
		Heteromastus filiformis	0.86	0.6509	133.3	125.3	134.8
		Laeonereis culveri	27.65	<0.0001	122.6	177.5	126.2
		Neanthes succinea	3.31	0.1914	136.8	117.0	133.2
		Polydora cornuta	0.01	0.9940	132.4	132.6	132.7
		Tubificoides heterochaetus	47.63	<0.0001	131.6	184.7	107.9
		Tubificidae	3.34	0.1886	128.4	128.8	141.1
		Paranais litoralis	0.00	1.0000	132.5	132.5	132.5
		Tubificoides brownae	28.78	<0.0001	114.3	139.1	159.1
		Monopylephorus rubroniveus	21.06	<0.0001	122.5	108.6	160.9
		Species Richness	1.61	0.4469	125.7	132.6	138.8
		Total Number	11.22	0.0037	116.5	139.0	150.8
		Stress Sensitive	23.76	<0.0001	148.3	147.0	99.3
		Stress Tolerant	29.54	<0.0001	109.0	146.8	163.9
2000	Benthic Community	*Streblospio benedicti*	4.92	0.0856	25.6	15.7	18.1
		Capitella capitata	0.97	0.6168	22.1	23.8	19.7
		Heteromastus filiformis	5.51	0.0635	24.0	19.0	19.0
		Laeonereis culveri	13.63	0.0011	19.9	36.5	17.8
		Neanthes succinea	0.31	0.8551	21.8	23.5	20.3
		Polydora cornuta	1.75	0.4173	20.6	26.8	20.6
		Tubificoides heterochaetus	8.04	0.0179	22.3	28.8	17.5
		Tubificidae	3.73	0.1548	21.9	28.8	18.1
		Paranais litoralis	0.98	0.6114	20.0	25.4	22.0
		Tubificoides brownae	1.20	0.5488	20.6	24.8	21.5
		Monopylephorus rubroniveus	1.72	0.4228	20.6	17.3	24.4
		Species Richness	6.79	0.0335	24.8	26.3	15.0
		Total Number	0.45	0.7992	22.8	20.5	20.1
		Stress Sensitive	7.77	0.0205	26.7	14.5	17.0
		Stress Tolerant	1.59	0.4505	19.1	23.5	24.0
1994	Fish/Crustacean Community	*Anchoa mitchelli* (#)	3.70	0.1575	21.8	14.0	23.4
		Fundulus heterochaetus (#)	4.33	0.1148	23.2	12.6	22.0
		Grass Shrimp (#)	2.87	0.2377	20.0	16.0	24.9
		Penaeid Shirmp (#)	8.00	0.0184	23.5	9.4	23.2
		Species Richness	4.26	0.1188	17.8	19.7	26.3
		Total Abundance	7.40	0.0247	21.3	10.7	25.8
		Anchoa mitchelli (Biomass)	3.58	0.1673	22.4	13.9	22.5
		Fundulus heterochaetus (Biomass)	4.39	0.1113	22.7	12.5	22.8
		Grass Shrimp (Biomass)	2.51	0.2844	19.8	16.9	24.9
		Penaeid Shirmp (Biomass)	7.30	0.0261	24.1	10.1	22.0
		Total Biomass	4.80	0.0907	21.8	12.3	24.2

Table 19. Summary of the regression analyses for the TCP biological quality data set by year study was conducted. The r^2 indicates the amount of variation explained by the model. The smaller the p-value, the stronger the evidence that the slope is different than zero. The slope indicates the direction and magnitude of the line.

Study	Parameter Type	Parameter	Model r^2	Model p-value	Intercept	Slope
1994	Benthic Community	*Streblospio benedicti*	0.01	0.0478	1.191	0.019
		Capitella capitata	0.03	0.0056	0.585	-0.019
		Heteromastus filiformis	0.00	0.3188	0.585	0.008
		Laeonereis culveri	0.04	0.0022	0.943	-0.026
		Neanthes succinea	0.00	0.6434	0.757	0.004
		Polydora cornuta	0.00	0.6069	0.052	0.001
		Tubificoides heterochaetus	0.08	<.0001	0.972	-0.039
		Tubificidae	0.04	0.0006	0.358	0.023
		Paranais litoralis				
		Tubificoides brownae	0.04	0.0014	0.626	0.026
		Monopylephorus rubroniveus	0.08	<.0001	1.171	0.048
		Species Richness	0.01	0.1699	0.615	0.002
		Total Abundance	0.02	0.0285	2.983	0.015
		Stress Sensitive	0.02	0.0245	1.198	-0.011
		Stress Tolerant	0.03	0.0035	1.032	0.015
2000	Benthic Community	*Streblospio benedicti*	0.00	0.9630	3.024	-0.001
		Capitella capitata	0.00	0.6642	0.766	-0.009
		Heteromastus filiformis	0.05	0.1468	0.460	-0.023
		Laeonereis culveri	0.03	0.2388	1.291	-0.029
		Neanthes succinea	0.01	0.6008	2.035	0.012
		Polydora cornuta	0.00	0.7725	0.989	0.007
		Tubificoides heterochaetus	0.09	0.0507	0.738	-0.037
		Tubificidae	0.06	0.1334	1.660	-0.037
		Paranais litoralis	0.04	0.2187	2.504	-0.034
		Tubificoides brownae	0.00	0.9056	0.540	0.002
		Monopylephorus rubroniveus	0.08	0.0704	2.218	0.050
		Species Richness	0.05	0.1368	0.844	-0.004
		Total Abundance	0.01	0.6415	4.025	0.004
		Stress Sensitive	0.03	0.2924	1.367	-0.010
		Stress Tolerant	0.07	0.1032	1.042	0.017
1994	Fish/Crustacean Community	*Anchoa mitchelli* (#)	0.03	0.2743	0.0510	0.002
		Fundulus heterochaetus (#)	0.02	0.3410	0.1460	-0.004
		Grass Shrimp (#)	0.01	0.5883	0.6513	0.007
		Penaeid Shirmp (#)	0.08	0.0788	0.3542	0.015
		Species Richness	0.04	0.1852	0.8901	0.004
		Total Abundance	0.05	0.1670	1.3143	0.020
		Anchoa mitchelli (Biomass)	0.00	0.8416	0.0261	0.000
		Fundulus heterochaetus (Biomass)	0.02	0.3756	0.1510	-0.004
		Grass Shrimp (Biomass)	0.00	0.8278	0.3723	0.002
		Penaeid Shirmp (Biomass)	0.04	0.2414	0.3199	0.009
		Total Biomass	0.02	0.3338	1.2003	0.010

Table 20. The creek width at the sample site, number of docks in the 500-m radius buffer, and land cover in the 500-m buffer for each creek and the average by category from the SCECAP large tidal creek data set.

				500-m Buffer			Upland in 500-m Buffer	
Category	Station Code	Creek Width (m)	Number of Docks	Water (%)	Salt Marsh (%)	Upland (%)	Forest (%)	Impervious Surface (%)
No Dock	RT00519	35	0	53.8	42.5	3.7	100.0	0.0
	RT00528	41	0	12.4	66.2	21.4	88.7	10.0
	RT00531	92	0	21.4	44.9	33.7	100.0	0.0
	RT00546	56	0	17.7	49.6	32.7	83.4	0.0
	RT00557	54	0	11.0	53.4	35.7	69.2	3.6
	RT99006	56	0	11.6	62.3	26.1	100.0	0.0
	RT99010	37	0	9.4	56.2	34.4	100.0	0.0
	RT99026	61	0	12.1	77.2	10.7	100.0	0.0
	Average	54	0	18.7	56.5	24.8	92.7	1.7
Low Dock	RT00502	116	1	27.2	46.8	26.0	97.3	1.1
	RT00504	113	3	29.1	41.8	29.1	69.1	2.5
	RT00520	78	1	31.8	62.8	5.4	94.7	0.0
	RT00523	37	6	12.3	55.2	32.5	93.0	7.0
	RT00526	87	1	19.7	74.3	6.0	100.0	0.0
	RT00542	33	4	4.9	54.9	40.2	50.0	6.4
	RT99022	26	3	13.6	67.2	19.3	76.3	17.0
	RT99040	35	4	37.9	39.9	22.1	97.4	2.6
	Average	66	2.875	22.1	55.4	22.6	84.7	4.6
High Dock	RT00503	81	8	19.5	71.8	8.7	78.7	14.8
	RT00545	77	87	30.1	38.1	31.8	8.1	49.8
	RT00549	23	17	6.7	34.7	58.6	56.4	3.9
	RT00550	66	31	16.4	59.2	24.4	49.7	22.8
	RT99005	121	16	32.5	28.2	39.2	70.5	15.6
	RT99007	82	22	29.1	55.6	15.3	39.3	26.2
	RT99009	85	10	27.2	41.5	31.2	53.0	3.7
	RT99017	47	51	13.3	32.2	54.5	53.9	28.8
	RT99027	67	9	24.8	60.5	14.7	66.0	14.6
	RT99030	113	8	24.8	42.5	32.7	96.9	3.1
	Average	76	25.9	22.5	46.4	31.1	57.3	18.3

Table 21. Summary of the ANOVA/ANCOVA analyses for the SCECAP sediment quality data set. The r^2 indicates the amount of variation explained by the model. The smaller the p-value, the stronger the evidence against the lack of an effect. A blank cell indicates the covariate was not significant at $\alpha = 0.15$ or that the parameter values were not high enough for a statistical test. The treatment categories are listed from highest level to lowest level and are abbreviated as follows: (1) ND = No Dock, (2) LD = Low Dock, and (3) HD = High Dock. The categories are different at $\alpha = 0.05$ if they do not share a common line.

Parameter Type	Parameter Name	Covariate	Model r^2	Model p-value	Category p-value	Covariate p-value	Category Differences
Metals	Aluminum	clay	0.93	<0.0001	0.9436	<0.0001	LD ND HD
	Arsenic	clay	0.68	<0.0001	0.5981	<0.0001	LD HD ND
	Cadmium	clay	0.31	0.0411	0.1128	0.0237	ND LD HD
	Chromium	clay	0.66	<0.0001	0.8775	<0.0001	ND LD HD
	Copper	clay	0.76	<0.0001	0.5696	<0.0001	HD LD ND
	Iron	clay	0.89	<0.0001	0.6438	<0.0001	LD HD ND
	Mercury		0.12	0.2409	0.2409		HD ND LD
	Manganese	clay	0.65	<0.0001	0.2633	<0.0001	LD HD ND
	Nickel	clay	0.85	<0.0001	0.4003	<0.0001	LD ND HD
	Lead	clay	0.75	<0.0001	0.5474	<0.0001	HD ND LD
	Selenium	clay	0.21	0.1573	0.4385	0.0674	LD HD ND
	Silver	clay	0.21	0.1519	0.2886	0.0817	LD ND HD
	Tin	clay	0.43	0.0058	0.6594	0.0008	LD HD ND
	Zinc	clay	0.27	0.0720	0.7485	0.0151	HD LD ND
PAH	1,6,7 Trimethylnaphthalene	TOC	0.34	0.0256	0.6546	0.0044	LD HD ND
	1-Methylnaphthalene	TOC	0.26	0.0799	0.9525	0.0110	ND LD HD
	1-Methylphenanthrene						
	2,6 Dimethylnaphthalene	TOC	0.33	0.0276	0.6635	0.0048	LD HD ND
	2-Methylnaphthalene	TOC	0.26	0.0782	0.9502	0.0107	ND LD HD
	Acenaphthene						
	Acenaphthylene		0.21	0.1936	0.1936		HD ND LD
	Anthracene	TOC	0.24	0.1095	0.2580	0.0779	HD LD ND
	Benzo(a)anthracene	TOC	0.31	0.0402	0.1459	0.0372	HD LD ND
	Benzo(a)pyrene	TOC	0.32	0.0356	0.3134	0.0140	HD LD ND
	Benzo(b)fluoranthene	TOC	0.36	0.0193	0.1175	0.0174	HD LD ND
	Benzo(e)pyrene	TOC	0.37	0.0168	0.0938	0.0182	HD LD ND
	Benzo(g,h,i)perylene	TOC	0.32	0.0351	0.3188	0.0135	HD LD ND
	Biphenyl	TOC	0.18	0.2181	0.5013	0.0930	HD ND LD
	Benzo(j+k)fluoranthene	TOC	0.37	0.0167	0.0717	0.0242	HD LD ND
	Chrysene+Triphenylene	TOC	0.32	0.0332	0.1639	0.0250	HD LD ND
	Dibenz(a,h+a,c)anthracene	TOC	0.34	0.0264	0.5194	0.0057	LD HD ND
	Fluoranthene	TOC	0.39	0.0119	0.2030	0.0043	HD ND LD
	Fluorene	TOC	0.33	0.0273	0.6625	0.0048	LD HD ND
	Indeno(1,2,3-cd)pyrene	TOC	0.30	0.0434	0.3030	0.0187	HD LD ND
	Naphthalene	TOC	0.27	0.0659	0.9872	0.0087	ND LD HD

Table 21. Continued.

Parameter Type	Parameter Name	Covariate	Model r^2	Model p-value	Category p-value	Covariate p-value	Category Differences
	Perylene	TOC	0.46	0.0028	0.7696	0.0003	ND HD LD
	Phenanthrene	TOC	0.34	0.0255	0.2970	0.0093	HD ND LD
	Pyrene	TOC	0.33	0.0281	0.4020	0.0071	HD ND LD
	Low Molecular Weight PAHs	TOC	0.32	0.0321	0.4354	0.0086	HD ND LD
	High Molecular Weight PAHs	TOC	0.40	0.0088	0.4298	0.0018	HD ND LD
	Total Polycyclic Aromatic Hydrocarbons	TOC	0.40	0.0089	0.4318	0.0018	HD ND LD
PCBs	Total Polychlorinated Biphenyls	TOC	0.55	0.0005	0.2609	<0.0001	HD ND LD
Composition	Percent Clay		0.01	0.8831	0.8831		HD LD ND
	Percent Sand		0.03	0.7341	0.7341		ND LD HD
	Percent Silt		0.05	0.5304	0.5304		HD LD ND
	Percent Silt/Clay		0.03	0.7341	0.7341		HD LD ND
	Percent TOC		0.01	0.9430	0.9430		HD LD ND
Pore Water	Total Ammonia		0.13	0.1900	0.1900		ND LD HD
	Unionized Ammonia		0.16	0.1323	0.1323		ND HD LD

Table 22. Summary of the test results for the SCECAP sediment quality data set from the nonparametric Kruskal-Wallis test. The Chi-Square is the test statistic. The smaller the p-value, the stronger the evidence against the lack of an effect. The mean score values are the sum of the rankings for each category.

				Mean Score		
Parameter Type	Parameter	Chi-Square	Model p-value	No Dock	Low Dock	High Dock
Metals	Aluminum	0.104	0.9492	13.00	13.25	14.10
	Arsenic	0.659	0.7195	11.75	13.81	14.65
	Cadmium	2.375	0.3050	15.63	14.50	11.00
	Chromium	0.411	0.8141	12.94	12.56	14.70
	Copper	1.313	0.5186	11.25	13.38	15.40
	Iron	0.052	0.9745	13.00	13.63	13.80
	Mercury	2.475	0.2901	13.19	10.88	15.85
	Manganese	1.799	0.4068	11.00	16.13	13.40
	Nickel	0.274	0.8722	12.50	14.50	13.50
	Lead	1.548	0.4611	11.81	12.25	15.85
	Selenium	1.764	0.4140	11.00	15.00	14.30
	Silver	2.250	0.3247	13.00	14.63	13.00
	Tin	0.623	0.7324	13.88	14.38	12.50
	Zinc	0.947	0.6228	11.38	14.00	14.80
PAHs	1,6,7 Trimethylnaphthalene	0.941	0.6248	12.50	14.06	13.85
	1-Methylnaphthalene	0.013	0.9938	13.63	13.50	13.40
	1-Methylphenanthrene	0.000	1.0000	13.50	13.50	13.50
	2,6 Dimethylnaphthalene	0.941	0.6248	12.50	14.06	13.85
	2-Methylnaphthalene	0.013	0.9938	13.63	13.50	13.40
	Acenaphthene	0.000	1.0000	13.50	13.50	13.50
	Acenaphthylene	3.328	0.1894	12.50	12.50	15.10
	Anthracene	3.100	0.2122	11.50	13.06	15.45
	Benzo(a)anthracene	4.408	0.1104	11.00	12.75	16.10
	Benzo(a)pyrene	2.726	0.2559	11.50	13.31	15.25
	Benzo(b)fluoranthene	5.053	0.0799	10.38	12.25	17.00
	Benzo(e)pyrene	5.744	0.0566	10.81	11.56	17.20
	Benzo(g,h,i)perylene	2.726	0.2559	11.50	13.31	15.25
	Biphenyl	1.600	0.4493	13.00	13.00	14.30
	Benzo(j+k)fluoranthene	6.077	0.0479	10.69	11.56	17.30
	Chrysene+Triphenylene	3.023	0.2206	11.19	12.06	16.50
	Dibenz(a,h+a,c)anthracene	1.582	0.4535	12.00	13.75	14.50
	Fluoranthene	2.431	0.2965	13.75	10.38	15.80
	Fluorene	0.941	0.6248	12.50	14.06	13.85
	Indeno(1,2,3-cd)pyrene	2.726	0.2559	11.50	13.31	15.25
	Naphthalene	0.013	0.9938	13.50	13.63	13.40
	Perylene	0.480	0.7865	14.75	12.38	13.40
	Phenanthrene	2.180	0.3362	12.25	11.63	16.00
	Pyrene	2.157	0.3401	13.50	10.63	15.80
	Low Molecular Weight PAHs	1.795	0.4076	12.75	11.50	15.70
	High Molecular Weight PAHs	1.797	0.4071	12.25	11.63	16.00
	Total Polycyclic Aromatic Hydrocarbons	1.586	0.4524	12.63	11.50	15.80
PCBs	Total Polychlorinated Biphenyls	2.071	0.3550	12.63	11.13	16.10
Composition	Clay	0.282	0.8685	13.00	12.75	14.50
	Sand	0.650	0.7225	15.13	13.50	12.20
	Silt	1.096	0.5781	11.25	13.88	15.00
	SiltClay	0.650	0.7225	11.88	13.50	14.80
	TOC	0.071	0.9654	13.13	13.25	14.00
Pore Water	Total Ammonia	3.001	0.2231	17.38	11.44	12.05
	Unionized Ammonia	4.462	0.1074	18.25	11.25	11.50

Table 23. Summary of the regression analyses for the SCECAP sediment quality data set. The r^2 indicates the amount of variation explained by the model. The smaller the p-value, the stronger the evidence that the slope is different than zero. The slope indicates the direction and magnitude of the line. A blank cell indicated the parameter values were not high enough for a statistical test.

Parameter Type	Parameter Name	Model r^2	Model p-value	Intercept	Slope
Metals	Aluminum	0.07	0.1862	0.4743	-0.0035
	Arsenic	0.08	0.1611	0.7408	-0.0049
	Cadmium	0.05	0.2796	0.0349	-0.0007
	Chromium	0.11	0.0915	1.2654	-0.0100
	Copper	0.04	0.3357	0.7545	-0.0039
	Iron	0.07	0.2074	0.3642	-0.0022
	Mercury	0.01	0.6594	0.0060	0.0000
	Manganese	0.07	0.2089	1.8778	-0.0045
	Nickel	0.11	0.0913	0.9247	-0.0055
	Lead	0.05	0.2724	0.9159	-0.0041
	Selenium	0.02	0.4459	0.0149	-0.0002
	Silver	0.01	0.6165	0.0023	0.0000
	Tin	0.00	0.8348	0.0202	-0.0001
	Zinc	0.04	0.3101	1.2108	-0.0067
PAH	1,6,7 Trimethylnaphthalene	0.00	0.9363	0.0483	-0.0001
	1-Methylnaphthalene	0.01	0.6867	0.1219	-0.0013
	1-Methylphenanthrene				
	2,6 Dimethylnaphthalene	0.00	0.9513	0.0766	-0.0002
	2-Methylnaphthalene	0.01	0.6828	0.1586	-0.0017
	Acenaphthene				
	Acenaphthylene	0.01	0.6775	0.0375	0.0007
	Anthracene	0.02	0.5032	0.1389	0.0029
	Benzo(a)anthracene	0.02	0.5500	0.2307	0.0037
	Benzo(a)pyrene	0.00	0.8368	0.2152	0.0012
	Benzo(b)fluoranthene	0.03	0.3986	0.3853	0.0059
	Benzo(e)pyrene	0.04	0.3520	0.2940	0.0057
	Benzo(g,h,i)perylene	0.00	0.8403	0.1923	0.0011
	Biphenyl	0.00	0.7547	0.0342	0.0007
	Benzo(j+k)fluoranthene	0.04	0.3441	0.3265	0.0063
	Chrysene+Triphenylene	0.02	0.5043	0.4593	0.0045
	Dibenz(a,h+a,c)anthracene	0.00	0.9130	0.0910	-0.0003
	Fluoranthene	0.00	0.9153	0.7242	0.0008
	Fluorene	0.00	0.9495	0.0815	-0.0002
	Indeno(1,2,3-cd)pyrene	0.00	0.8125	0.1969	0.0013
	Naphthalene	0.01	0.7261	0.1811	-0.0017
	Perylene	0.01	0.6219	0.6806	-0.0042
	Phenanthrene	0.01	0.5877	0.3626	0.0028
	Pyrene	0.00	0.9154	0.6982	0.0007
	Low Molecular Weight PAHs	0.01	0.7122	0.4854	0.0027
	High Molecular Weight PAHs	0.00	0.9850	1.2497	0.1924
	Total Polycyclic Aromatic Hydrocarbons	0.00	0.9754	1.2791	-0.0003
PCBs	Total Polychlorinated Biphenyls	0.00	0.8869	0.2279	0.0004
Composition	Clay	0.06	0.2244	0.4717	-0.0028
	Sand	0.04	0.3037	1.0047	0.0032
	Silt	0.02	0.4434	0.2539	-0.0012
	SiltClay	0.04	0.3037	0.5661	-0.0032
	TOC	0.04	0.3012	0.0976	-0.0006
Pore Water	Total Ammonia	0.17	0.0378	0.5776	-0.0055
	Unionized Ammonia	0.09	0.1434	1.5502	-0.0083

Table 24. Summary of the ANOVA analyses for the SCECAP water quality data set. The r^2 indicates the amount of variation explained by the model. The smaller the p-value, the stronger the evidence against the lack of an effect. The treatment categories are listed from highest level to lowest level and are abbreviated as follows: (1) ND = No Dock, (2) LD = Low Dock, and (3) HD = High Dock. The categories are different at $\alpha = 0.05$ if they do not share a common line.

Parameter Type	Parameter Name	Model r^2	Model p-value	Category p-value	Category Differences
Dissolved Nutrients	Dissolved inorganic carbon	0.11	0.5046	0.5046	HD LD ND
	Dissolved ammonia	0.25	0.0249	0.0249	LD ND HD
	Dissolved nitrate/nitrite	0.17	0.0760	0.0760	LD HD ND
	Dissolved organic carbon	0.23	0.2088	0.2088	LD ND HD
	Dissolved organic nitrogen	0.24	0.1969	0.1969	ND LD HD
	Dissolved organic phosphate	0.18	0.3021	0.3021	ND LD HD
	Dissolved silicon/silica	0.37	0.0021	0.0021	LD ND HD
	Orthophosphate	0.37	0.0020	0.0020	LD ND HD
	Total dissolved nitrogen	0.28	0.1362	0.1362	ND LD HD
	Total dissolved phosphate	0.38	0.0582	0.0582	LD ND HD
	Total nitrogen	0.06	0.7389	0.7389	ND HD LD
	Total phosphate	0.04	0.7670	0.7670	ND LD HD
Nutrients	Total Ammonia Nitrogen	0.02	0.8040	0.8040	ND HD LD
	Total Kjedahl Nitrogen	0.06	0.5157	0.5157	ND LD HD
	Nitrite + Nitrate	0.06	0.5212	0.5212	LD ND HD
	Total Phosphorus	0.19	0.0969	0.0969	LD ND HD
	Total Organic Carbon	0.06	0.4756	0.4756	ND LD HD
Pigments	Chlorophyll a	0.01	0.8850	0.8850	LD HD ND
Oyxgen Demand	Biological Oxygen Demand	0.13	0.1900	0.1900	HD ND LD
Water Column	Dissolved Oxygen	0.08	0.0247	0.0247	ND HD LD
	pH	0.04	0.4454	0.4454	HD LD ND
	Salinity	0.04	0.1984	0.1984	ND HD LD
	Temperature	0.02	0.3735	0.3735	ND LD HD
	Secchi depth reading	0.06	0.4333	0.4333	HD LD ND
	Turbidity	0.14	0.1853	0.1853	ND LD HD
Fecal Coliform	Fecal Coliform A-1 proceedure	0.10	0.3276	0.3276	LD HD ND

Table 25. Summary of the test results for the SCECAP water quality data set from the nonparametric Kruskal-Wallis test. The Chi-Square is the test statistic. The smaller the p-value, the stronger the evidence against the lack of an effect. The mean score values are the sum of the rankings for each category.

				Mean Score		
Parameter Type	Parameter	Chi-Square	Model p-value	No Dock	Low Dock	High Dock
Dissolved Nutrients	Dissolved inorganic carbon	1.583	0.4531	6.00	8.67	9.50
	Dissolved ammonia	5.924	0.0517	13.90	19.55	10.13
	Dissolved nitrate/nitrite	4.043	0.1324	11.15	18.63	16.25
	Dissolved organic carbon	3.627	0.1631	9.60	9.08	4.38
	Dissolved organic nitrogen	5.381	0.0679	11.20	7.83	4.25
	Dissolved organic phosphate	2.733	0.2550	9.20	9.00	5.00
	Dissolved silicon/silica	13.406	0.0012	12.80	22.42	8.50
	Orthophosphate	10.507	0.0052	15.90	20.50	7.50
	Total dissolved nitrogen	4.418	0.1098	9.80	9.17	4.00
	Total dissolved phosphate	5.336	0.0694	8.60	10.33	3.75
	Total nitrogen	0.360	0.8351	6.60	7.67	6.00
	Total phosphate	1.708	0.4256	9.00	8.83	5.50
Nutrients	Total Ammonia Nitrogen	0.458	0.7954	11.86	9.50	11.17
	Total Kjedahl Nitrogen	0.543	0.7623	14.50	12.79	11.95
	Nitrite + Nitrate	1.462	0.4815	12.44	14.79	10.78
	Total Phosphorus	4.211	0.1218	13.75	16.93	9.65
	Total Organic Carbon	1.178	0.5548	14.69	14.88	11.45
Pigments	Chlorophyll a	0.321	0.8516	11.50	13.57	12.56
Oyxgen Demand	Biological Oxygen Demand	3.254	0.1965	13.56	9.88	16.35
Water Column	Dissolved Oxygen	6.906	0.0316	53.25	36.15	50.07
	pH	1.583	0.4532	18.27	20.54	24.00
	Salinity	0.946	0.6230	42.45	48.75	47.85
	Temperature	1.567	0.4568	50.29	47.93	42.12
	Secchi depth reading	2.745	0.2534	12.85	13.95	19.13
	Turbidity	3.279	0.1941	16.50	14.63	10.20
Fecal Coliform	Fecal Coliform	1.486	0.4758	10.44	14.63	13.83

Table 26. Summary of the regression analyses for the SCECAP water quality data set. The r^2 indicates the amount of variation explained by the model. The smaller the p-value, the stronger the evidence against the lack of a slope different than zero. The slope indicates the direction and magnitude of the line.

Parameter Type	Parameter Name	Model r^2	Model p-value	Intercept	Slope
Dissolved Nutrients	Dissolved inorganic carbon	0.16	0.1340	1.2788	0.0022
	Dissolved ammonia	0.06	0.2030	0.6381	-0.0030
	Dissolved nitrate/nitrite	0.02	0.4328	0.2941	-0.0024
	Dissolved organic carbon	0.22	0.0802	2.7580	-0.0043
	Dissolved organic nitrogen	0.29	0.0384	1.6166	-0.0035
	Dissolved organic phosphate	0.10	0.2578	0.0653	-0.0009
	Dissolved silicon/silica	0.65	<0.0001	1.9382	-0.0122
	Orthophosphate	0.23	0.0079	0.4112	-0.0049
	Total dissolved nitrogen	0.45	0.0060	1.6819	-0.0038
	Total dissolved phosphate	0.32	0.0267	0.4454	-0.0058
	Total nitrogen	0.04	0.4997	1.7477	-0.0287
	Total phosphate	0.04	0.4979	0.5873	-0.0027
Nutrients	Total Ammonia Nitrogen	0.02	0.5318	0.0757	0.0009
	Total Kjedahl Nitrogen	0.15	0.0553	0.2369	-0.0012
	Nitrite + Nitrate	0.01	0.6703	0.0101	0.0000
	Total Phosphorus	0.06	0.2329	0.0504	-0.0003
	Total Organic Carbon	0.12	0.0785	0.6929	-0.0062
Pigments	Chlorophyll a	0.00	0.9227	1.0322	-0.0002
Oyxgen Demand	Biological Oxygen Demand	0.10	0.1119	0.3751	0.0049
Water Column	Dissolved Oxygen	0.02	0.1400	0.6592	0.0008
	pH	0.12	0.0249	0.9208	0.0003
	Salinity	0.02	0.2411	1.4243	0.0013
	Temperature	0.11	0.0011	1.4872	-0.0003
	Secchi depth reading	0.22	0.0100	0.1871	0.0017
	Turbidity	0.33	0.0020	1.2767	-0.0069
Fecal Coliform	Fecal Coliform A-1 proceedure	0.00	0.7443	1.3554	-0.0027

Table 27. Summary of the ANOVA/ANCOVA analyses for the SCECAP biological quality data set. The r^2 indicates the amount of variation explained by the model. The smaller the p-value, the stronger the evidence against the lack of an effect. A blank cell indicates the covariate was not significant at $\alpha = 0.15$ or that the parameter values were not high enough for a statistical test. The treatment categories are listed from highest level to lowest level and are abbreviated as follows: (1) ND = No Dock, (2) LD = Low Dock, and (3) HD = High Dock. The categories are different at $\alpha = 0.05$ if they do not share a common line.

Parameter Type	Parameter Name	Model r^2	Model p-value	Category p-value	Silt/Clay Covariate p-value	Salinity Covariate p-value	Category Differences
Toxicity	Clam Growth (ug/day)	0.01	0.9158	0.9158			ND LD HD
	Amphipod Survival	0.19	0.2767	0.2767			HD ND LD
	Microtox EC50 (%)	0.01	0.9360	0.9360			LD HD ND
	Clam Survival	0.01	0.8426	0.8426			LD HD ND
Benthic Community	*Ilyanassa obsoleta* (#)	0.03	0.4689	0.4689			LD ND HD
	Exogone sp. (#)	0.08	0.1406	0.1406			LD HD ND
	Mediomastus ambiseta (#)	0.26	0.0019	0.0084		0.0084	LD ND HD
	Mediomastus sp. (#)	0.01	0.8008	0.8008			LD HD ND
	Streblospio benedicti (#)	0.24	0.0045	0.0346		0.0031	LD ND HD
	Spiochaetopterus costarum oculatus (#)	0.08	0.2474	0.5714		0.0865	HD LD ND
	Heteromastus filiformis (#)	0.01	0.6994	0.6994			HD LD ND
	Scoletoma tenuis (#)	0.28	0.0013	0.0231		0.0006	LD HD ND
	Tubificoides heterochaetus (#)	0.76	<0.0001	0.2040	0.0492	<.0001	ND LD HD
	Tharyx acutus (#)	0.33	0.0002	0.0007		0.0040	LD ND HD
	Polydora cornuta (#)	0.22	0.0075	0.0079		0.0310	LD HD ND
	Tubificoides brownae (#)	0.09	0.1986	0.3029		0.1107	LD ND HD
	Cirratulidae (#)	0.24	0.0111	0.0897	0.0768	0.0572	LD ND HD
	Species Richness	0.48	<0.0001	0.3133	0.0683	<0.0001	LD ND HD
	Total Abundance	0.30	0.0021	0.0043	0.1444	0.0781	LD ND HD
	Index of Biotic Integrity - CP	0.03	0.7334	0.7334			ND LD HD
	Index of Biotic Integrity - SC	0.01	0.8733	0.8733			ND LD HD
Fish/Crustaceans Community	*Anchoa mitchilli* (#)	0.01	0.8181	0.8181			HD ND LD
	Bairdiella chrysoura (#)	0.13	0.0792	0.1167		0.0891	HD LD ND
	Leiostomus xanthurus (#)	0.25	0.0070	0.2313	0.0029	0.0267	ND HD LD
	Penaeus aztecus (#)	0.27	0.0016	0.0261	0.0009		ND LD HD
	Penaeus setiferus (#)	0.27	0.0017	0.5607	0.0003		LD HD ND
	Lolliguncula brevis (#)	0.08	0.2791	0.6877		0.0793	ND LD HD
	Species Richness	0.18	0.0244	0.5517	0.0072		HD LD ND
	Total Abundance	0.23	0.0047	0.5867	0.0006		ND HD LD
	Anchoa mitchilli (Biomass)	0.06	0.1963	0.1963			HD ND LD
	Bairdiella chrysoura (Biomass)	0.09	0.0872	0.0872			HD LD ND
	Leiostomus xanthurus (Biomass)	0.12	0.1912	0.4683	0.1114	0.1376	HD ND LD
	Penaeus aztecus (Biomass)	0.11	0.0523	0.0523			ND LD HD
	Penaeus setiferus (Biomass)	0.00	0.9498	0.9498			LD ND HD
	Lolliguncula brevis (Biomass)	0.01	0.7065	0.7065			HD ND LD
	Total Biomass	0.20	0.0285	0.9989	0.0020	0.0638	ND LD HD

Table 28. Summary of the test results for the SCECAP biological quality data set from the nonparametric Kruskal-Wallis test. The Chi-Square is the test statistic. The smaller the p-value, the stronger the evidence against the lack of an effect. The mean score values are the sum of the rankings for each category.

				Mean Score		
Parameter Type	Parameter	Chi-Square	Model p-value	No Dock	Low Dock	High Dock
Toxicity	Clam Growth (ug/day)	0.385	0.8251	13.50	14.75	12.50
	Amphipod Survival	2.612	0.2710	8.20	6.08	0.63
	Microtox EC50 (%)	1.007	0.6045	15.25	14.00	11.70
	Clam Survival	0.380	0.8270	12.25	14.25	13.90
Benthic Community	*Ilyanassa obsoleta (#)*	1.065	0.5870	27.34	28.06	24.58
	Exogone sp. (#)	3.053	0.2173	24.44	29.69	25.60
	Mediomastus ambiseta (#)	7.232	0.0269	31.34	29.50	20.23
	Mediomastus sp. (#)	0.488	0.7835	25.00	28.25	26.30
	Streblospio benedicti (#)	7.663	0.0217	27.16	33.91	20.05
	Spiochaetopterus costarum oculatus (#)	0.964	0.6177	24.94	25.22	28.78
	Heteromastus filiformis (#)	0.527	0.7685	24.41	27.31	27.53
	Scoletoma tenuis (#)	3.807	0.1490	21.94	31.84	25.88
	Tubificoides heterochaetus (#)	0.859	0.6510	27.13	27.44	25.25
	Tharyx acutus (#)	10.767	0.0046	28.84	33.03	19.40
	Polydora cornuta (#)	7.139	0.0282	23.19	33.88	23.25
	Tubificoides brownae (#)	2.636	0.2677	29.47	28.66	22.40
	Cirratulidae (#)	3.423	0.1806	25.81	31.03	23.43
	Species Richness	1.316	0.5178	27.38	29.25	23.60
	Total Abundance	8.125	0.0172	26.88	34.38	19.90
	Index of Biotic Integrity - CP	0.561	0.7553	14.81	13.81	12.20
	Index of Biotic Integrity - SC	0.424	0.8088	14.25	14.25	12.30
Fish/Crustacean Community	*Anchoa mitchilli (#)*	0.434	0.8050	26.28	24.94	27.93
	Bairdiella chrysoura (#)	4.775	0.0919	20.94	25.72	31.58
	Leiostomus xanthurus (#)	3.390	0.1836	30.56	21.09	27.58
	Penaeus aztecus (#)	3.358	0.1866	31.09	26.19	23.08
	Penaeus setiferus (#)	1.674	0.4330	22.56	28.38	28.15
	Lolliguncula brevis (#)	0.676	0.7134	28.63	26.69	24.65
	Species Richness	1.980	0.3716	22.69	26.22	29.78
	Total Abundance	0.486	0.7842	26.50	24.53	28.08
	Anchoa mitchilli (Biomass)	0.887	0.6417	26.47	24.13	28.43
	Bairdiella chrysoura (Biomass)	5.185	0.0748	19.56	29.56	29.60
	Leiostomus xanthurus (Biomass)	3.138	0.2083	29.94	21.09	28.08
	Penaeus aztecus (Biomass)	3.485	0.1751	31.31	25.81	23.20
	Penaeus setiferus (Biomass)	1.350	0.5092	22.69	26.66	28.24
	Lolliguncula brevis (Biomass)	0.391	0.8223	28.34	26.03	25.40
	Total Biomass	0.435	0.8047	25.50	25.31	28.25

Table 29. Summary of the regression analyses for the SCECAP biological quality data set. The r^2 indicates the amount of variation explained by the model. The smaller the p-value, the stronger the evidence against the lack of a slope different than zero. The slope indicates the direction and magnitude of the line.

Parameter Type	Parameter Name	Model r^2	Model p-value	Intercept	Slope
Toxicity	Clam Growth (ug/day)	0.03	0.4022	26.0584	0.1460
	Amphipod Survival	0.10	0.2476	1.2625	0.0006
	Microtox EC50 (%)	0.21	0.0200	0.0651	0.0024
	Clam Survival	0.00	0.7331	1.3929	0.0001
Benthic Community	Ilyanassa obsoleta (#)	0.02	0.3828	0.4734	-0.0054
	Exogone sp. (#)	0.01	0.4134	0.3527	-0.0048
	Mediomastus ambiseta (#)	0.10	0.0246	1.0172	-0.0171
	Mediomastus sp. (#)	0.02	0.2820	0.8631	-0.0081
	Streblospio benedicti (#)	0.05	0.1112	2.0855	-0.0157
	Spiochaetopterus costarum oculatus (#)	0.03	0.2545	0.7975	-0.0080
	Heteromastus filiformis (#)	0.00	0.8857	0.8879	0.0011
	Scoletoma tenuis (#)	0.04	0.1547	1.3744	-0.0122
	Tubificoides heterochaetus (#)	0.02	0.3771	0.2472	-0.0044
	Tharyx acutus (#)	0.09	0.0288	0.9144	-0.0170
	Polydora cornuta (#)	0.03	0.2490	1.0435	-0.0106
	Tubificoides brownae (#)	0.02	0.2760	1.2806	-0.0091
	Cirratulidae (#)	0.05	0.1126	0.8055	-0.0122
	Species Richness	0.02	0.3740	1.1308	-0.0023
	Total Number	0.08	0.0377	3.3655	-0.0075
	Index of Biotic Integrity - CP	0.01	0.6191	3.3797	-0.0049
	Index of Biotic Integrity - SC	0.01	0.5856	3.5580	-0.0066
Fish/Crustaceans Community	Anchoa mitchilli (#)	0.01	0.5072	0.8248	-0.0047
	Bairdiella chrysoura (#)	0.00	0.6287	1.0861	-0.0034
	Leiostomus xanthurus (#)	0.06	0.0748	1.3781	-0.0111
	Penaeus aztecus (#)	0.06	0.0861	0.8678	-0.0130
	Penaeus setiferus (#)	0.02	0.3858	1.4221	-0.0078
	Lolliguncula brevis (#)	0.03	0.2312	1.0635	-0.0082
	Species Richness	0.00	0.7292	0.8172	-0.0007
	Total Number	0.04	0.1832	2.5990	-0.0071
	Anchoa mitchilli (Biomass)	0.01	0.6165	0.0424	-0.0003
	Bairdiella chrysoura (Biomass)	0.00	0.7711	0.9188	-0.0034
	Leiostomus xanthurus (Biomass)	0.02	0.3101	0.8524	-0.0088
	Penaeus aztecus (Biomass)	0.02	0.3028	1.1578	-0.0226
	Penaeus setiferus (Biomass)	0.01	0.4031	1.7046	-0.0196
	Lolliguncula brevis (Biomass)	0.00	0.6913	0.4008	-0.0028
	Total Biomass	0.02	0.2774	0.8296	-0.0032

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