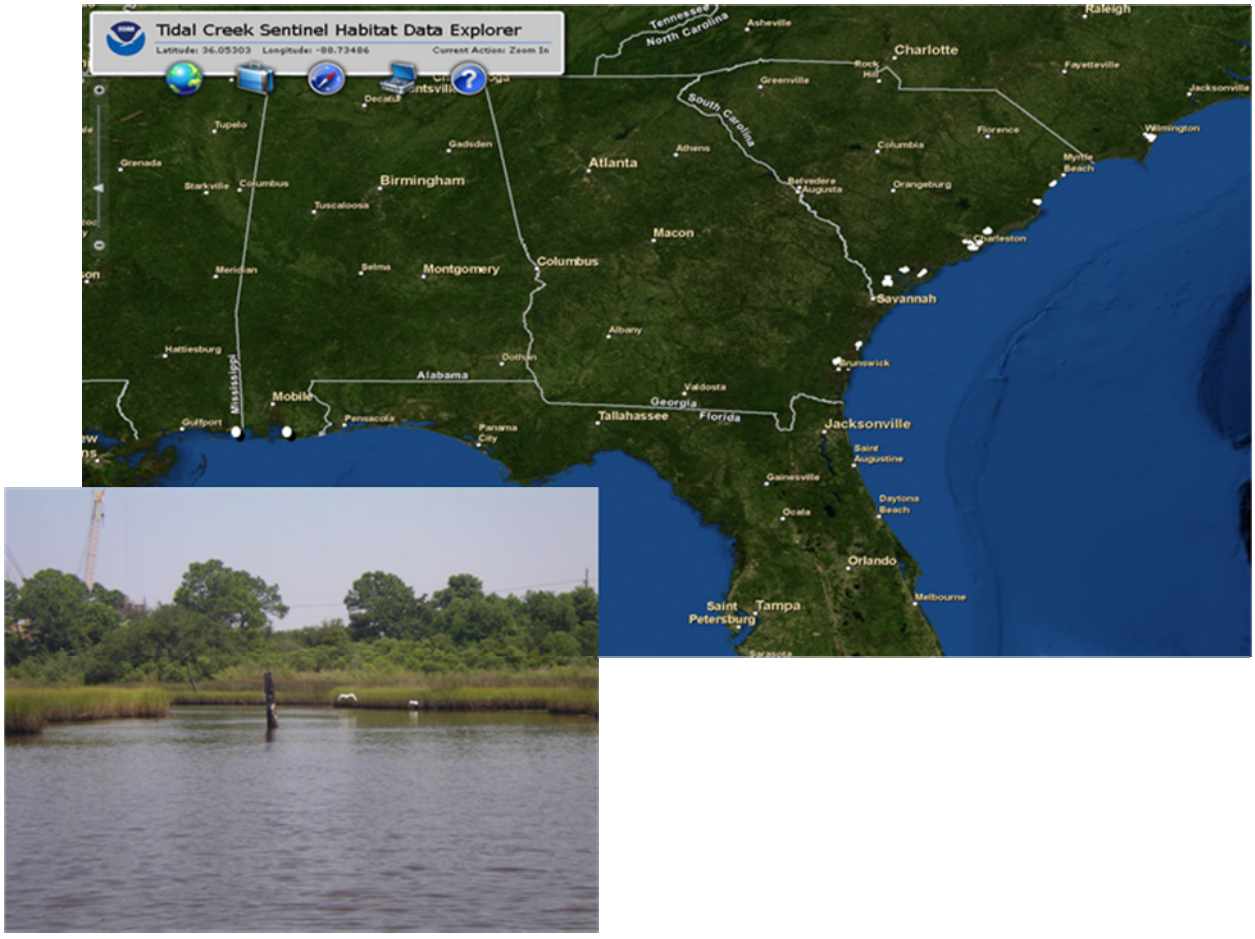
Gulf of Mexico Tidal Creeks Serve as Sentinel Habitats for Assessing the Impact of Coastal Development on Ecosystem Health



NOAA Technical Memorandum NOS NCCOS 136

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Gulf of Mexico Tidal Creeks Serve as Sentinel Habitats for Assessing the Impact of Coastal Development on Ecosystem Health

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Abstract

A study was conducted, in association with the Alabama and Mississippi National Estuarine Research Reserves (NERRs) in the Gulf of Mexico (GoM) as well as the Georgia, South Carolina, and North Carolina NERRs in the Southeast (SE), to evaluate the impacts of coastal development on tidal creek sentinel habitats, including potential impacts to human health and well-being. Uplands associated with Southeast and Gulf of Mexico tidal creeks, and the salt marshes they drain, are popular locations for building homes, resorts, and recreational facilities because of the high quality of life and mild climate associated with these environments. Tidal creeks form part of the estuarine ecosystem characterized by high biological productivity, great ecological value, complex environmental gradients, and numerous interconnected processes. This research combined a watershed-level study integrating ecological, public health and human dimension attributes with watershed-level land cover data. The approach used for this research was based upon a comparative watershed and ecosystem approach that sampled tidal creek networks draining developed watersheds (e.g., suburban, urban, and industrial) as well as undeveloped sites (Holland et al. 2004, Sanger et al. 2008). The primary objective of this work was to define the relationships between coastal development with its concomitant land cover changes, and non-point source pollution loading and the ecological and human health and well-being status of tidal creek ecosystems.

Nineteen tidal creek systems, located along the Southeastern United States coast from southern North Carolina to southern Georgia, and five Gulf of Mexico systems from Alabama and Mississippi were sampled during summer (June-August) 2005, 2006 (SE) and 2008 (GoM). Within each system, creeks were divided into two primary segments based upon tidal zoning: intertidal (i.e., shallow, narrow headwater sections) and subtidal (i.e., deeper and wider sections), and watersheds were delineated for each segment. In total, we report findings on 29 intertidal and 24 subtidal creeks. Indicators sampled throughout each creek included water quality (e.g., dissolved oxygen, salinity, nutrients, chlorophyll-*a* levels), sediment quality (e.g., characteristics, contaminant levels including emerging contaminants), pathogen and viral indicators (e.g., fecal coliform, enterococci, F+ coliphages, F- coliphages), and abundance and tissue contamination of biological resources (e.g., macrobenthic and nektonic communities, shellfish tissue contaminants).

Tidal creeks have been identified as a sentinel habitat to assess the impacts of coastal development on estuarine areas in the southeastern US. A conceptual model for tidal creeks in the southeastern US identifies that human alterations (stressors) of upland in a watershed such as increased impervious cover will lead to changes in the physical and chemical environment such as microbial and nutrient pollution (exposures), of a receiving water body which then lead to changes in the living resources (responses). The overall objective of this study is to evaluate the applicability of the current tidal creek classification framework and conceptual model linking tidal creek ecological condition to potential impacts of development and urban growth on ecosystem value and function in the Gulf of Mexico US in collaboration with Gulf of Mexico NERR sites. The conceptual model was validated for the Gulf of Mexico US tidal creeks. The tidal creek classification system developed for the southeastern US could be applied to the Gulf of Mexico tidal creeks; however, some differences were found that warrant further examination. In particular, pollutants appeared to translate further downstream in the Gulf of Mexico US

compared to the southeastern US. These differences are likely the result of the morphological and oceanographic differences between the two regions. Tidal creeks appear to serve as sentinel habitats to provide an early warning of the ensuing harm to the larger ecosystem in both the Southeastern and Gulf of Mexico US tidal creeks.

Key Words: National Estuarine Research Reserve System (NERRS), North Carolina NERR, Sapelo Island NERR, Weeks Bay NERR, Grand Bay NERR, tidal creek, sentinel habitat, conceptual model, impervious cover, land cover, urbanization, sediment and tissue contaminants, water quality, pathogens, nekton, oysters, macrobenthos, physical and chemical environment.

1. Introduction

The coastal United States hosts abundant natural resources that contribute hundreds of billions of dollars to the US economy annually (Colgan 2003). In addition, these resources provide many free ecological services, including waste processing, clean air and water, and scenic vistas worth untold billions of dollars (Costanza et al. 1997). Approximately 17% of the US land area (excluding Alaska) and >50% of the population are located along the US coasts (Crossett et al. 2004). Coastal ocean-based tourism is the fastest growing component of the coastal economy, with hundreds of millions of Americans and international guests visiting our coasts annually (Colgan 2003). Not surprisingly, coastal population densities are 2-5 times higher than in the rest of the nation (Beach 2002). Along our coasts, land is being consumed for urban development 3-6 times faster than the rate of population growth (Beach 2002), resulting in major and permanent alterations to coastal ecosystems. These trends appear to be accelerating, with potentially serious impacts on the long term health of coastal ecosystems and the quality of life of the people who live, work, and recreate there (Cohen et al. 1997, Vitousek et al. 1997).

Recent reports have noted the measurably diminished condition of our nation's coastal natural resources (USEPA 2001a, NMFS 2002, Pew 2003). Major sources of impairment across all regions and habitats include chemical and microbial contamination, increased "flashiness" in freshwater inflows, nutrient over-enrichment, hypoxia, increased frequency of harmful algal blooms, habitat modification and degradation, wetland loss, increased abundance of non-native species, over-harvesting of fisheries, and impaired biological communities. Most of these reports conclude that the major environmental threats to coastal resources are from diffuse or non-point sources of pollution. In addition, the cumulative effects of multiple stressors, including the interactions among them, have been identified as the major contributor to diminished resources.

New approaches and collaborations are required to understand and resolve the complex, regional-scale environmental issues including cumulative stress from multiple sources facing our coasts. Existing observational systems do not provide early warning and have failed to link degraded ecosystem condition and human health and quality of life. Public health and well-being and ecosystem science can no longer be viewed as separate domains but are interconnected and linked disciplines. It is obvious that the health of people, wildlife, and ecosystems are linked in the web of life. A paradigm of one health (human, wildlife, and ecosystem) is crucial to sustain the critical ecological services and quality of life that currently exists in the coastal zone.

One of the earliest symptoms of broad scale coastal ecosystem impairment has historically been declines in the amount and condition of rare and critical habitats that are sensitive to localized and relatively small changes in environmental conditions. Notable examples include sea grass beds, oyster reefs, kelp forests, coral reefs, and wetlands. These "sentinel habitats" or "first responders" generally decline in extent and condition years-to-decades before system wide impairment is documented by routine environmental quality monitoring activities. For example, the extent of sea grass beds in the Chesapeake Bay exhibited declines as early as the 1960s. The greatest declines occurred in the headwaters of major tributaries of the Bay well before non-point source nutrient and sediment inputs were recognized as a Bay-wide problem (Bayley et al. 1978). Coral reefs in Florida Bay also exhibited symptoms of disease, bleaching, and impairment years

before the impacts of alterations in pollution loading and changes in freshwater flow were identified as major issues for the region (Dustan and Halas 1987). Unfortunately, the scientific knowledge needed to understand the warning signals provided by sentinel habitats has only recently become available (Kemp et al. 1983, Hoegh-Guldberg 1999, Porter and Tougas 2001, Turgeon et al. 2002).

In estuarine environments of the southeastern US (SE), salt marsh habitats and their associated tidal creek networks serve a range of important functions including nursery habitats, storm buffers, and pollutant filters. Tidal creeks and salt marshes also serve as the interface between the upland landscape and estuaries where freshwater from the land mixes with saline water from the ocean, resulting in dynamic environments that are renowned for their ecological complexity, biological productivity, and seafood production (Kneib 1997, Sanger et al. 1999a, b, Lerberg et al. 2000, Mallin et al. 2000b, Holland et al. 2004). Salt marshes are bisected by tidal creeks which facilitate water moving into and out of the system. In the SE, the watersheds associated with headwater tidal creeks are among the most rapidly developing in the nation. Because tidal creek networks form the primary hydrologic link between estuaries and land-based activities, small intertidal portions of tidal creek networks represent the first zone of impact for non-point source pollution runoff from upland areas. The potential for the microbial and chemical contamination in these tidal creek habitats is great. Consequently, tidal creeks provide a potential sentinel habitat for impacts from human landscape alterations in coastal areas.

Holland et al. (2004) developed a conceptual model for SC, further refined by Sanger et al. (2008) for the SE, of the source-receptor links between the origin of an environmental problem (e.g., human activity, extreme natural event, linkages between ocean processes) and anticipated impacts on these tidal creek ecosystems. The model mirrored the US Environmental Protection Agency (USEPA) Ecological Risk Assessment model with stressors leading to changes in the physical-chemical environment (i.e., exposures) which in turn leads to biological responses. The conceptual model for tidal creeks describes that human alterations of upland in a watershed such as increased impervious cover (stressors) will lead to changes in the physical and chemical environment such as microbial and nutrient pollution (exposures) of a receiving water body which will then lead to changes in the living resources (responses) (Figure 1-1). Holland et al. (2004) and Sanger et al. (2008) predicted that adverse changes in the physical and chemical environment (e.g., water quality indicators such as indicator bacteria for sewage pollution or sediment chemical contamination) generally occurred when impervious cover levels in the watershed exceed 10-20% and that ecological processes responded and were generally impaired when impervious cover levels exceeded 20-30% in suburban and urban watersheds (Figure 1-1).

The original conceptual model did not include the linkages between the ecosystem condition and human health and well-being. There is an emerging consensus that current patterns of coastal development are associated with increasing fecal pollution in tidal creeks, estuaries, and bathing beaches (Mallin et al. 2000a, Karn and Harada 2001, Holland et al. 2004, Mallin 2006). From a human health perspective, the accumulation of pathogens in the water, sediments, and organisms may render seafood products unsafe to eat and water unsafe for primary contact recreation. Current patterns of coastal development may also affect flooding vulnerability, public health risk, and the economic impacts. In order to begin addressing impacts on human health and welfare, the conceptual model has been updated to include societal responses, including human

health and well-being (Figure 1-1; Sanger et al. 2008). Impervious cover levels defining where human uses are impaired are currently being determined, but it generally appears that shellfish bed closures increase and the flooding vulnerability of headwater regions become a concern when impervious cover values exceed 10-30%.

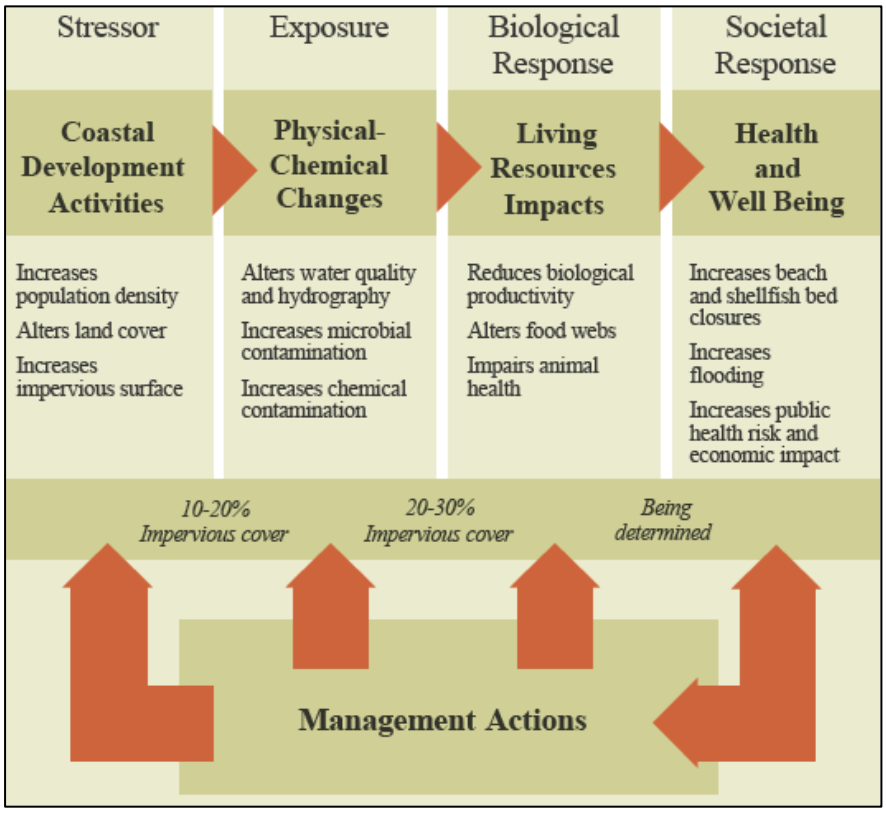

Figure 1-1. Conceptual model identifying linkages between development of the upland and the ecological responses of South Carolina tidal creeks, expanded to include societal responses and associated feedback loops.

To assist in understanding the complexity and variability associated with freshwater and wetland ecosystems, classification frameworks have been developed that integrate the ecological attributes of these systems within their biogeography, hydrology, and short- and long-term ecological history (e.g., Horton 1945, Cowardin et al. 1979, Frissell et al. 1986). Classification approaches have made, however, only limited contributions to the understanding of spatial and temporal variability and scale issues for estuarine ecosystems (e.g., Anderson et al. 1976, Odum 1984). To partition and account for the variability and complexity of SE tidal creek networks, we developed a geographically-independent, hierarchical classification framework for tidal creeks that is patterned after the freshwater stream classification system (Horton 1945, Strahler 1957). This preliminary tidal creek classification framework enhanced the characterization and understanding of natural and human induced environmental gradients and spatial variability for SE tidal creeks (Sanger et al. 2008) and is being evaluated for GoM tidal creeks as a part of this effort.

Although the conceptual model and tidal creeks classification scheme were developed initially for South Carolina, they have been validated as applicable to the SE (South Carolina, North Carolina, and Georgia; Sanger et al. 2008). It is currently unknown if similar tidal creek networks can also be used as early warning systems for regions outside the SE. The overall objective of the current study is to evaluate the applicability of the current tidal creek classification framework and conceptual model linking tidal creek ecological condition to potential impacts of development and urban growth on ecosystem value and function in the GoM through collaborations with GoM NERR sites. The study also provides the two GoM NERRs with data to assess the health of their NERR and surrounding sites.

2. Methods

2.1 Study Sites

A total of twenty-three tidal creek networks were sampled along the southeastern (SE) and Gulf of Mexico (GoM) coasts during the summer (June-August) (Table 2-1, Figure 2-1). In the SE, nineteen tidal creek networks from New Hanover County, NC, to Glynn County, GA, were sampled in 2005 and 2006. Twelve tidal creek networks were sampled in SC in 2005. Four of these networks were re-sampled in 2006. In 2006, four creek networks were sampled in GA, and two were sampled in NC. In the GoM, five tidal creek networks from Mobile Bay, AL (n=2) to Pascagoula, MS (n=3), were sampled in 2008. The GA, NC, AL, and MS sites were sampled in association with the staff of the adjacent National Estuarine Research Reserves.

Table 2-1. Creek order sampled, date sampled, latitude, and longitude for each tidal creek network sampled by state. Latitude and longitude are average values for each creek. NIWB = North Inlet-Winyah Bay NERR, ACE = Ashepoo, Combahee, and Edisto NERR, SAP = Sapelo Island NERR.

System	Orders Sampled	Dates Sampled	Latitude	Longitude
Alabama				
Weeks Bay	1, 2	7/16/2008	30.415	-87.830
Bear Creek	1, 2	7/15/2008	30.273	-87.716
Mississippi				
Bayou Heron	1, 2, 3	7/28/2008	30.410	-88.406
Bayou Chico	1, 2, 3	7/29/2008	30.345	-88.517
Bayou Pattassa	1, 2	7/30/2008	30.377	-88.546
North Carolina				
Hewlitts	1, 1, 2	8/7/2006	34.189	-77.857
Whiskey Creek	1, 2	8/9/2006	34.161	-77.865
South Carolina				
Albergottie	1, 2, 3	8/17/2005	32.448	-80.720
Bulls	1, 2, 3	6/29/2005	32.825	-80.027
Guerin	1, 2, 3	7/5/2005, 6/20/2006	32.944	-79.766
James Island	1, 1, 2, 2, 3	8/1/2005, 8/17/2006	32.744	-79.974
Murrells Inlet	2, 3	6/22/2005	33.564	-79.025
New Market	1	8/8/2005, 7/24/2006	32.806	-79.940
NIWB-Town	1, 2, 3	6/20/2005	33.340	-79.177
Okatee	1, 2, 3	7/18/2005	32.287	-80.929
Orangegrove	1, 2, 3	7/20/2005	32.812	-79.978
Parrot	1, 2, 3	7/7/2005	32.733	-79.910
Shem	1, 2	8/29/2005	32.801	-79.869
ACE-Village	1, 2, 3	8/3/2005, 7/5/2006	32.419	-80.522
Georgia				
Burnett	1, 2	7/19/2006	31.234	-81.538
SAP-Duplin	1, 2, 3	7/19/2006	31.147	-81.375
SAP-Oakdale	1	7/11/2006	31.481	-81.272
Postell	1	7/11/2006	31.415	-81.285

A longitudinal gradient was defined for each tidal creek network by applying a freshwater stream classification model (Horton 1945, Strahler 1957). The first order, or headwater, of each creek was defined as directly draining coastal uplands or salt marsh habitat and was characterized by narrow (<10 m) width with predominately intertidal habitat. These first order sections will be referred to as intertidal sections in the remaining text. The second order of each creek was formed by the confluence of two or more first order creeks. Second order creeks were wider (usually >10 m but <30 m) and had abundant subtidal habitats. The third order of each creek was formed by the confluence of two or more second order creeks. Third order systems were large, wide (>30 m)

creeks composed mainly of subtidal habitat and a small amount of intertidal habitat in the creek. For simplicity, the second and third order systems will be collectively referred to as subtidal sections throughout the remaining text. While combining second and third orders results in the loss of some information about the tidal creek longitudinal gradient, two points support their pooling in this study: (1) in SC, the differences in a wide range of parameters between second and third orders were small (Sanger et al. 2008), and (2) only a limited number of third order creeks were sampled outside of SC, making a regional evaluation of that order impossible.

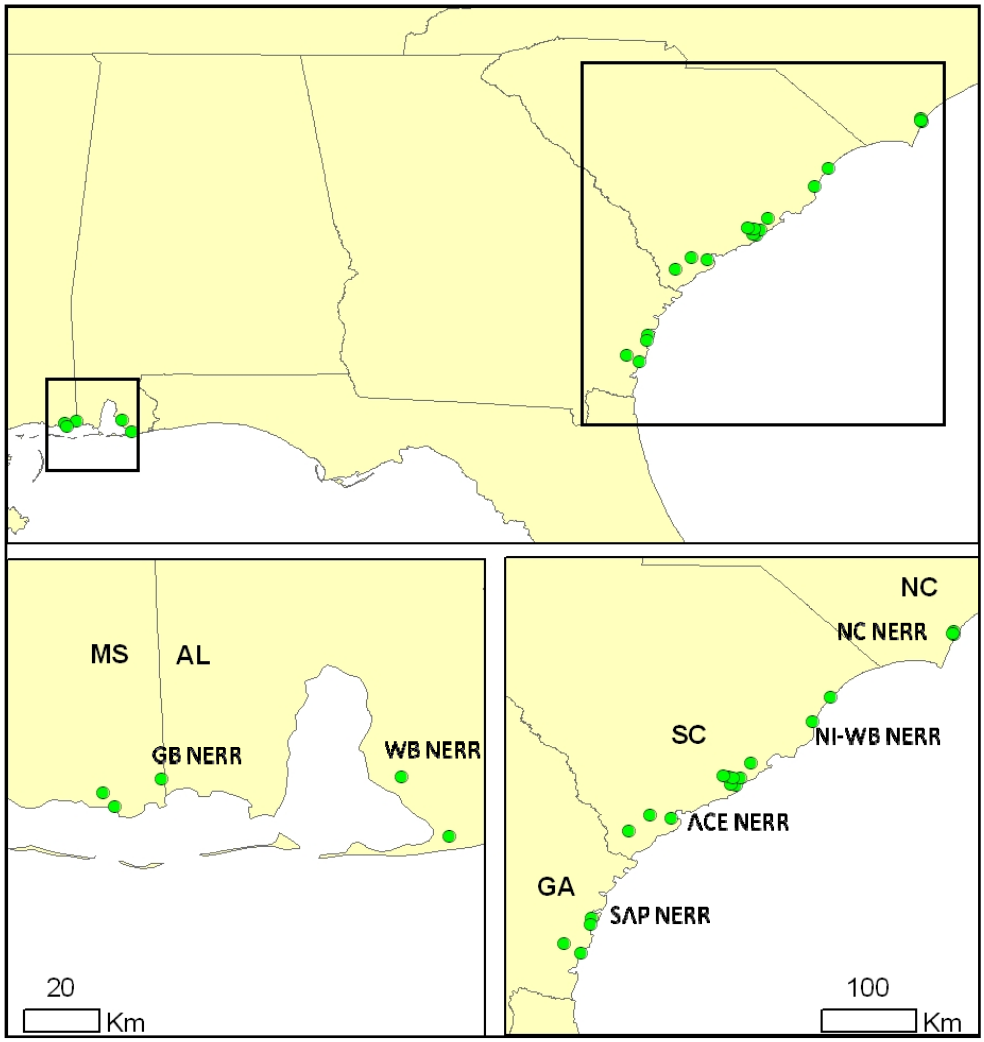

Figure 2-1. Map of the Gulf of Mexico and southeastern sampling sites with insets for the two regions. NERRs sampled were NC = North Carolina, NI-WB = North Inlet-Winyah Bay, ACE = Ashepoo, Combahee, and Edisto, SAP = Sapelo Island, GB = Grand Bay, WB = Weeks Bay.

Each creek order was divided into three equidistant reaches using ArcGIS 9 (ESRI, Redlands, CA); by convention, the first reach within an order was the furthest upstream, while the second reach was the middle, and the third reach was the furthest downstream section. Within any reach of any creek order, stations were randomly located for sample collection. The specific sampling activities that occurred within each reach are detailed below.

2.2 Watershed Determinations and Land Cover Characterizations

Watersheds and sub-watersheds were identified using ArcGIS 9 to evaluate the land use and impervious cover of each creek and order. Watersheds and their sub-watersheds were delineated based on elevation contours. Elevation Derivatives for National Applications (EDNA) data were downloaded from the United States Geological Survey (USGS, http://edna.usgs.gov/). The EDNA watersheds have a resolution of 30 meters that corresponds to a scale of 1:24,000. Single watershed boundaries were identified for the respective research area and overlaid with digital elevation model (DEM) and USGS topographic data. Visual confirmation of the delineation of the EDNA watersheds using the topographic maps was conducted. Sub-watersheds were delineated for each creek order. In general, the EDNA data were found to represent the expected watershed boundaries with only slight modifications needed to reflect specific elevation gradients or other attributes such as roads that might impede surface runoff. Modifications to EDNA watershed boundaries were made by hand digitization.

National Land Cover Data (NLCD) (Homer et al. 2004) were downloaded for selected regions in the five states using the MRLC web tool (http://gisdata.usgs.net/website/MRLC/viewer.php). Land cover data were derived from the 2001 NLCD dataset. The land cover and impervious cover data were matched to the watershed and sub-watershed boundary data. Land cover estimates were determined from this layer and summed to obtain simplified categories of land cover. The impervious cover data were further modified by removing data for marsh and open water categories. All editing functions were performed in ArcView. Impervious cover levels were then calculated from the NLCD for all watersheds and sub-watersheds. These NLCD-derived impervious cover estimates were compared to values published in Holland et al. (2004). Both of these data sets used aerial photography from similar time frames (1999-2001) to estimate impervious cover, but the NLCD impervious cover values underestimated the ground-truthed data reported in Holland et al. (2004). A similar underestimation of NLCD-derived impervious cover data has also been reported by Jarnagin et al. (2006). A quadratic relationship was developed ($y = 2.9301 + 2.16789x - 0.01611x^2$ where y is the adjusted impervious cover percent and x is NLCD-derived impervious cover; White et al. in prep) and used to adjust the NLCD-derived impervious cover percentage.

Creek watersheds were classified at the highest order level into the following land use categories based on impervious cover as modified from Holland et al. (2004): (1) forested (<10% impervious cover); (2) suburban (≥10% but <35% impervious cover); and (3) urban (≥35% impervious cover). There were two exceptions to this classification. The Orangegrove Creek (SC) watershed was estimated to have 37.3% impervious cover; however, since this was primarily light residential development on a small amount of upland (127 ha) relative to the total watershed size (322 ha), we categorized this site as a suburban watershed. The Burnett Creek (GA) watershed was estimated to have 11.8% impervious cover; however, since this was a superfund site designated by the USEPA, it was categorized as an urban watershed.

2.3 Sample Design

Each creek was sampled during the ebbing tide over a two or three day period. Southeast creeks (semi-diurnal tides) were sampled approximately 2-3 hours prior to low tide and GoM creeks (diurnal tides) were sampled approximately 4-5 hours prior to low tide. First order was sampled by walking to designated sample sites, and second and third order creeks were sampled from a

small boat. Sampling was generally conducted by moving in an upstream direction to minimize effects of physical disturbance on subsequently sampled stations. Within each creek order, samples were collected to characterize water quality, water column nutrients, pathogen indicators, macrobenthic infauna, resident nekton, sediment contaminants, and oyster pathogen and contaminant body burdens. Sampling stations were selected using a stratified random method with each order divided into 3 equal reaches. The first reach was furthest upstream, the second reach was in the middle, and the third reach was located furthest downstream. The number of samples collected in each order varied by sample type.

Water quality data (temperature, dissolved oxygen, salinity, pH, turbidity, chlorophyll-*a*) were collected in bottom waters (0.3 m above bottom) using Yellow Springs Instrument (YSI) 6600 data loggers. A logger was deployed in the second reach of each creek order and collected data at 15 minute intervals for at least 2 full tidal cycles (25 hrs in the SE and 50 hrs in the GoM). One water sample for pathogen indicators was collected in the second reach of each creek order in sterile 2 L polypropylene bottles. In addition, one water sample was collected in an acid-washed 500 mL polyethylene bottle within each reach of each creek order for nutrient quantifications. All water samples were collected approximately 0.3 m below the surface of the water, or mid-water column in water less than 0.3 m in depth. In addition, water samples were collected in an upstream direction to prevent contamination. Bottle caps were removed immediately prior to sampling, and bottles were inverted at the appropriate sample depth.

Macrobenthic infauna were collected using two field methods. In intertidal creeks, the benthos was sampled approximately 1 m below mean high water (MHW), primarily when sediment was exposed, using a 0.0044 m^2 core sampler to a depth of 15 cm. A total of 9 cores (3 from each reach) were collected at randomly located stations to ensure sampling of the benthic fauna along the entire intertidal habitat. A small scoop of mud (upper 2 cm) was collected next to each core sample for sediment analysis (% sand, % silt, % clay, total organic carbon [TOC]). Additionally, 2 small cores (upper 2 cm, 0.0009 m^2) were collected at each site and composited across all sites within each reach to quantify porewater ammonium (NH_4^+). In the subtidal reaches, the infauna were sampled using a 0.04 m^2 modified Van Veen grab sampler in the SE and a 0.023 m^2 Petit Ponar grab sampler in the GoM. One grab sample was collected for benthos in each reach. Sediment samples for grain size analysis and porewater ammonia determination were taken from the top 2 cm of a second intact grab from each site.

Sediments were sampled for chemical contaminants once in each creek order. At a randomly selected station in the second reach of intertidal creeks, the top 2 cm of sediment were carefully scraped off the surface of mud exposed at low tide and homogenized in a stainless steel bowl. In the second reach of subtidal creeks, the top 2 cm of a minimum of 3 successful grab samples were homogenized for chemical analysis. A stainless steel spatula was used to remove sediment samples from the sediment surface and place them in the appropriate container. The homogenate was apportioned to appropriate pre-cleaned sample jars (i.e., metals in plastic and organics in glass) and placed on ice as soon as possible.

Nekton, predominantly fish and crustaceans, were sampled in first order creeks. Nekton was sampled using a 0.635 mm mesh seine net. One seine was pulled in each reach in an upstream direction over a distance of 25 m. Every effort was made to stretch the net from bank to bank.

When this was not possible the seined width was estimated to the nearest meter. Water width and depth were measured at both the starting and end points in the seine to calculate the area and volume of the creek swept.

In 2006, oysters (*Crassostrea virginica*) were hand-collected in every creek order when present. In 2008, oysters (*Crassostrea virginica*) in MS and ribbed mussels (*Geukensia demissa*) in AL were hand-collected in every creek order when present. When possible oysters or mussels were collected in the second reach near the site where the data logger was deployed and sediment samples were collected. After collection, oysters and mussels were separated for determination of pathogen loads (~ 20 individuals, oysters only), and chemical contaminant body burdens (~12 individuals).

2.4 Laboratory Processing Methods

2.4.1 Basic Water Quality

Basic water quality data (i.e., temperature, pH, DO, salinity, turbidity, depth, chlorophyll-*a*) were downloaded from the data loggers and examined to remove data resulting from exposure at low tide (common in first order creeks). In the SE, data loggers were calibrated prior to deployment and a post-calibration check was conducted after retrieval to ensure the logger was functioning properly. Summary data (i.e., mean, maximum, minimum, range) for the measured parameters were calculated. In the GoM, data loggers were calibrated prior to deployment, but a post-calibration check was not conducted after retrieval. Data were reviewed extensively to determine if each logger was functioning properly. Any data that did not meet pre-established criteria were removed.

2.4.2 Nutrients and Phytoplankton

Both whole and filtered water samples were used for nutrient analyses. Whole water samples were analyzed for total nitrogen (TN) and total phosphorus (TP) using the persulfate digestion method (D'Elia et al. 1977). Additional samples were filtered through a 47 mm GF/F (Whatman) to quantify dissolved constituents (i.e., ammonium [NH_4^+], nitrite+nitrate [$NO_{2/3}$], total dissolved nitrogen [TDN], ortho-phosphate [PO_4^{3-}], total dissolved phosphorus [TDP], and silicate [DSi]. Ammonium was analyzed via the Berthelot Reaction using a Technicon AutoAnalyzer (Technicon Industrial Systems), and silicate was measured using the "molybdenum blue" method on the same AutoAnalyzer (Technicon Industrial Systems). Both ortho-phosphate and nitrate+nitrite were analyzed using standard methods (EPA methods 365.1 and 365.2, respectively, in USEPA 1979). The material remaining on the filter paper was extracted in acetone and analyzed for chlorophyll-*a* (Chl-*a*) and phaeophytin (Phaeo) using fluorometric techniques (Welschmeyer 1994).

2.4.3 Pathogen Indicators

Water collected for pathogen indicators was analyzed for both bacterial and viral indicators within 24 hours of collection. Fecal coliforms (FC) and enterococci (ENT) were enumerated by membrane filtration according to standard methods (APHA 1998). Coliphages were enumerated and characterized as described in Stewart et al. (2006). Both male-specific (F+) and somatic (F-) coliphages were enumerated by the single agar layer method, adapted from USEPA Method 1602 (USEPA 2001b).

Actual values were not obtained for FC and ENT at three of the GoM sites because the concentrations at these sites were not sufficiently diluted such that they exceeded the level of quantification (600 colony forming units [CFU]) for that dilution set. One site was located in the intertidal section of Bayou Chico and the other two sites were located in the intertidal and subtidal sections of Bayou Pattassa. This lack of realistic values resulted in our removing the ENT data at sites with greater than quantification levels recorded and developing estimates of the FC water concentration at two of the sites based on FC oyster concentrations. A regression of oyster tissue concentrations to water concentrations from the 2006 SE sites was performed ($y = 1.0257x - 0.3858$ on log transformed data, $R^2 = 0.58$) and used to estimate the water concentrations for two of the three sites in the GoM.

Oysters to be tested for pathogen body burdens were first homogenized and composited for each collection site to obtain approximately 100 g (wet weight) tissue. The tissue homogenate was tested for the microbial indicators FC and ENT were enumerated most probable number according to standard methods (APHA 1998).

In 2008, water and oyster samples were also analyzed for norovirus and enterovirus using viral RNA detection protocols. Each water sample was passed through a 0.45 μm filter membrane until the filter clogged. A subsample of shellfish homogenate was archived at –80 ºC until RNA isolation was performed. RNA was extracted from each matrix using an RNeasy kit as described in Noble et al. (2006). The norovirus reverse transcription-quantitative polymerase chain reaction (RT-qPCR) assay was performed as in Gregory et al. (2011). Samples were similarly analyzed for enterovirus according to Gregory et al. (2006).

2.4.4 Chemical Contaminants

Sediments and tissues (oyster or mussel) were analyzed for a suite of 22 trace metals, 22 pesticides, 25 polycyclic aromatic hydrocarbons (PAHs), 79 polychlorinated biphenyls (PCBs), and 13 polybrominated diphenyl ethers (PBDEs) (Appendix A). PBDEs are used as flame retardants and are considered to be an emerging contaminant of concern. Data quality was assured using a series of spikes, blanks, and standard reference materials (NIST 1944 and NRC MESS-3 for sediments; and NIST 1566b, 1974b, and NRC DOLT-3 for tissues). Sediment samples were kept frozen at approximately - 40 ºC until analyzed. To thaw, samples were left in closed containers in a 4 ºC cooler for approximately 24 hours. Sediment samples were thoroughly homogenized by hand with a stainless steel spatula prior to extraction. Tissues from multiple oysters (maximum of 12 individuals) were composited to obtain 15 g of wet weight. The tissue was well homogenized using a ProScientific homogenizer in 500 mL Teflon containers. The homogenized tissue sample was split into an organic (pre-cleaned glass container) and inorganic (pre-cleaned polypropylene container) sample and stored at - 40 ºC until extraction or digestion. Tissues were removed from the freezer and stored overnight at 4 ºC and allowed to partially thaw. A percent dry-weight determination was made gravimetrically on an aliquot of each wet sediment and tissue sample.

Inorganic sample digestion and analysis consisted of the following steps. Dried sediment was ground with a mortar and pestle and transferred to a 20 mL plastic screw-top container. A 0.25 g sub-sample of the ground material was transferred to a Teflon-lined digestion vessel and digested

in 5 mL of concentrated nitric acid using microwave digestion. The sample was brought to a fixed volume of 50 mL in a volumetric flask with deionized water and stored in a 50 mL polypropylene centrifuge tube until instrumental analysis of Li, Be, Al, Fe, Mg, Ni, Cu, Zn, Cd, and Ag. A second 0.25 g sub-sample was transferred to a Teflon-lined digestion vessel and digested in 5 mL of concentrated nitric acid and 1 mL of concentrated hydrofluoric acid in a microwave digestion unit. The sample was then evaporated on a hotplate at 225 ºC to near dryness, and 1 mL of nitric acid was added. The sample was brought to a fixed volume of 50 mL in a volumetric flask with deionized water and stored in a 50 mL polypropylene centrifuge tube until instrumental analysis for V, Cr, Co, As, Sn, Sb, Ba, Tl, Pb, and U. Selenium was analyzed by hotplate digestion using a 0.25 g sub-sample and 5 mL of concentrated nitric acid. Each sample was brought to a fixed volume of 50 mL in a volumetric flask with deionized water and stored in a 50 mL polypropylene centrifuge tube until instrumental analysis. Additionally, 2-3 g wet tissue were microwave digested in Teflon-lined digestion vessels using 10 mL of concentrated nitric acid along with 2 mL of hydrogen peroxide. Digested samples were brought to a fixed volume with deionized water in graduated polypropylene centrifuge tubes and stored until analysis.

A separate inorganic aliquot was used for mercury analysis. Approximately 0.5 g of wet sediment or tissue was analyzed on a Milestone DMA-80 Direct Mercury Analyzer. All remaining elemental analysis was performed using Inductively Coupled Plasma Mass Spectrometry (ICP-MS) except for silver, which was determined using Graphite Furnace Atomic Absorption (GFAA) spectroscopy. Data quality was controlled by using a series of blanks, spiked solutions, and standard reference materials including NRC MESS-3 (Marine Sediments) and NIST 1566b (freeze dried mussel tissue).

Organic extraction and analysis consisted of the following steps. An aliquot (10 g sediment or 5 g tissue wet weight) was extracted with anhydrous sodium sulfate using Accelerated Solvent Extraction (ASE) in either 1:1 methylene chloride:acetone for sediments or 100% dichloromethane for tissues (Schantz et al. 1997). Following extraction, samples were dried and cleaned using Gel Permeation Chromatography and Solid Phase Extraction to remove lipids and then solvent-exchanged into hexane for analysis. Samples were analyzed for PAHs, PBDEs, PCBs, and a suite of chlorinated pesticides using appropriate Gas Chromatograph and Mass Spectrometer (GC-MS) technology. Data quality was ensured by assessing a spiked blank, a reagent blank, and appropriate standard reference materials with each set of samples to ensure the integrity of the analytical method.

2.4.5 Macrobenthic Community

Benthic samples were sieved through a 0.5 mm standard sieve, and the material retained on the screen was transferred to a polyethylene bottle and preserved in 10% buffered formalin containing Rose Bengal. Samples were capped and stored until macrobenthic invertebrates could be removed from the detritus, identified to the lowest practical taxonomic unit, and counted. The following quality control/quality assurance (QA/QC) procedures were used. One out of every 10 samples was re-sorted to ensure 90% sorting efficiency. If 10% of the organisms remained in the sample after sorting, then all 10 samples were re-sorted. The samples were identified to the lowest taxonomic level using dissecting and compound microscopes. One out of every ten samples was re-identified by a taxonomist for QA/QC purposes. If 10% of the

dominant organisms or 25% of the rare organisms were misidentified, then all 10 samples were re-identified. Densities of all organisms and number of taxa were calculated for each sample.

2.4.6 Nekton Community

Fish and crustaceans collected by seine net (0.635 cm bar mesh) were rinsed carefully and preserved in 10% formalin in seawater. Preserved organisms were sorted in the laboratory and identified to the lowest practical taxonomic level (usually species). A 90% identification efficiency was ensured using similar methods to the macrobenthic samples.

2.5 Data Summary and Statistical Analyses

Only a limited number of tidal creek systems could be sampled for this study in the GoM which resulted in a focus on tidal creeks in or of interest to the Grand Bay National Estuarine Research Reserve (NERR), MS and Weeks Bay NERR, AL. The NERRs provided regional reference sites and local knowledge of the tidal creek systems and creeks near their sites. We combined the data for GoM creeks with previous data collected for SE creeks for three reasons: (1) to determine the similarities and differences in tidal creek ecosystems between the two regions; (2) to evaluate if the responses of tidal creek ecosystems to coastal development were similar for the two regions; and (3) to increase the power of statistical tests. In general, relationships between the degree of coastal development and the amount of impairment increased as a result of combining the two data sets.

Resulting data (e.g., land cover, nutrient concentrations, infaunal abundances, pathogen abundances) were stored in a relational database. This database, initially managed in a PostgreSQL database and then migrated to a Microsoft SQL Server, was accessed by individual users via a Microsoft Access interface (White et al. 2008). This interface allows individual users to query the database to develop summaries and datasets for statistical analysis. Data are available upon request by contacting the senior author.

Tidal creek data from 2005, 2006, and 2008 summer sampling periods have been compiled into one data set; no attempt was made to examine year-to-year variability. The main unit of statistical inference was creek order, and the resulting data set was comprised of 43 observations (29 from intertidal systems, 24 from subtidal systems). In cases involving multiple measures per order, data were averaged within each order to obtain a single value for each indicator. Data were also averaged across the second and third orders to obtain a single value representing the subtidally-dominated sections. Lastly, for creeks that were sampled in both 2005 and 2006 (i.e., Guerin, James Island School, New Market, and Village; Table 2-1), data were averaged across years (2005 and 2006) resulting in a single value for each intertidal system and each subtidal system for each parameter.

Statistical analyses were designed to address three questions: 1) Do measured parameters vary across the two geographic regions?, (2) Do measured parameters vary across the sampled land use classes?, and 3) Do measured parameters vary along the creek longitudinal gradient? To address these questions, we employed Analysis of Variance (ANOVA) and regression analyses. The basic ANOVA model was a three-way, fixed factor model, with Region (GoM, SE), Land Use Class Type (forested, suburban, urban), and Creek Order (intertidal, subtidal) as the main effects. The interaction terms were included in all models and excluded if determined to be

nonsignificant ($p \geq 0.05$). Type III sums of squares were used for evaluating the significance of the factors. Pairwise differences were examined by comparing least square means (using PDIFF in SAS). Lastly, individual response variables were regressed against impervious cover by creek order to identify predictive relationships. Regression analysis included both the SE and GoM data sets. Regressions were considered significant at $p < 0.05$. If data were found to be non-normal or heteroscedastic, appropriate basic transformations (log, square root, arcsine) were used. Analyses were performed using SAS 9.1 (SAS Institute Inc., Cary, NC).

In addition, the macrobenthic invertebrate and nekton community data were analyzed by multivariate analyses. Multidimensional scaling (MDS) ordination was employed using Primer v6 statistical package (Clarke and Gorley 2006). The samples collected were averaged for each creek and habitat type. All data were fourth-root transformed while salinity and sediment composition were log transformed. Similarities among land use class were then examined. The analysis of similarity (ANOSIM) function was used to test for differences in biological communities. These analyses compared creek order between and within region.

To summarize sediment contaminant data, values less than method detection limits (MDL) were set to 0 before analysis. Total PAH, total PCB, total pesticides, and total PBDE concentrations were determined by summing the values for the 25, 79, 22, and 13 individual analytes measured for each group of contaminants, respectively (Appendix A). Concentrations of trace metals, PAHs, PCBs, and pesticides from the study creeks were compared to sediment quality guidelines. Long et al. (1995) developed sediment quality guidelines by summarizing the published literature on the effects of a suite of sediment contaminants on a wide range of marine biota and derived two threshold values, an effects range-low (ERL) and an effects range-median (ERM) for individual analytes. An ERL was defined as the sediment concentration of a given contaminant where 10% of all published studies reported an adverse effect, and an ERM was defined as the sediment concentration where 50% of all published studies reported an adverse effect. Values below the ERL would rarely be expected to be associated with measurable biological effects. Values between the ERL and ERM represent a range in which there are possible biological effects for a wide range of organisms. Values above the ERM represent a range above which there are probable biological effects.

The mean ERM quotient (mERMQ) was calculated for all contaminants (Total mERMQ) and for each major class of contaminant (i.e., trace metals, PAHs, PCBs, and pesticides) using the 24 analytes identified in Appendix B. Calculations were made by (1) dividing the concentration of each analyte or analyte group by the published ERM value, (2) summing the ratios of analytes within each contaminant class (e.g., trace metals), and (3) dividing by the number of contaminants in that class. In addition, total mERMQ values, which encompassed all four contaminant classes, were calculated for each sample. These values were calculated in the same fashion except that analytes were not combined within contaminant classes, instead the ratios of all 24 analytes were summed, and the total was divided by 24 (Long and MacDonald 1998). mERMQs provide a way to compare potential cumulative effects of contaminants after weighting them on a toxicological basis and were used in subsequent analyses.
Bivalve tissue chemical contamination was not analyzed using statistical methods due to the low sample size. Bivalve tissue concentrations on wet weight basis were compared to Food and Drug Administration (FDA) environmental chemical contaminant action levels (FDA 2001) and the

USEPA (2000) human-health consumption limits for cancer and non-cancer endpoints. The FDA action levels are simply threshold values for comparison against tissue concentrations (non-consumption based).

3. Results

3.1 Stressors

The five Gulf of Mexico (GoM) tidal creek systems surveyed for this study each consisted of 2 or 3 sub-watersheds depending upon the number of intertidal and subtidal creek segments sampled (Table 3-1). All systems included at least some upland areas and were classified as either forested, suburban, or urban based upon their land use designation.

Table 3-1. Creek system, land use class, watershed area, impervious cover, population density (individual hectare^{-1}), and median income for each creek segment. Subtidal watershed area includes the related intertidal area.

Creek System	Land Use Class	Watersheds	Order	Station Type	Watershed Area (ha)	Impervious Cover (%)	Population Density	Median Income
Alabama								
Weeks Bay	Forested	Weeks Bay	1	Intertidal	33	2.9	0.12	$50,208
		Weeks Bay	2	Subtidal	141	2.9	0.12	$50,208
Bear Creek	Suburban	Bear Creek	1	Intertidal	54	16.0	0.60	$41,250
		Bear Creek	2	Subtidal	222	11.6	0.60	$41,250
Mississippi								
Bayou Heron	Forested	Bayou Heron S	1	Intertidal	12	6.3	0.05	$29,773
		Bayou Heron	2 & 3	Subtidal	2402	3.5	0.11	$33,349
Bayou Chico	Urban	Bayou Chico	1	Intertidal	136	48.8	10.36	$34,996
		Bayou Chico	2 & 3	Subtidal	654	58.4	11.61	$32,235
Bayou Pattassa	Urban	Bayou Pattassa	1	Intertidal	53	55.0	6.99	$22,107
		Bayou Pattassa	2	Subtidal	68	52.0	6.39	$31,685

In the GoM, intertidal watersheds ranged in size from 12 ha (Bayou Heron South, forested) to 136 ha (Bayou Chico, urban; Table 3-1) and impervious cover ranged from 2.9% (Weeks Bay, forested) to 48.8% (Bayou Chico, urban). Subtidal watersheds included the intertidal area and ranged from 68 ha (Bayou Pattassa, urban) to 2402 ha (Bayou Heron, forested) in size. Impervious cover in these watersheds ranged from 2.9% (Weeks Bay) to 58.4% (Bayou Chico, urban). The GoM intertidal watersheds examined here were smaller than the average SE intertidal watershed (Figure 3-1) but still fell within the range of SE watershed sizes (18 - 2,425 ha). This same pattern generally held for subtidal watersheds with the exception of the rather large Bayou Heron watershed (Figure 3-1). However, all GoM watersheds fell within the range of SE subtidal watersheds (59 - 5,501 ha). The GoM watersheds also had a similar range of impervious cover levels to SE intertidal (2.9 - 70.4%) and subtidal (0.0 – 47.7%) watersheds.

Land cover in each watershed was determined using NLCD categories of developed-high, developed-low, agricultural, forest, scrub, palustrine wetland (swamp, floodplain, etc.), marsh (primarily salt marsh), or water. The percent composition of land cover shows a progressive change from scrub, palustrine wetland, and salt marsh to low and high development along the forested-suburban-urban gradient (Figure 3-2). GoM creek watersheds classified as forested were primarily scrub, marsh, and palustrine land cover, as compared to the primarily forest land cover of SE creeks. For suburban and urban watersheds, there was a general increase in urban land cover (both developed-low and developed-high) with the two urban watersheds having

greater than 90% total developed land cover (a percentage greater than SE urban watersheds previously examined). Subtidal impervious cover was generally higher than the intertidal impervious cover in urban watersheds in the GoM, opposite the overall pattern seen in the SE.

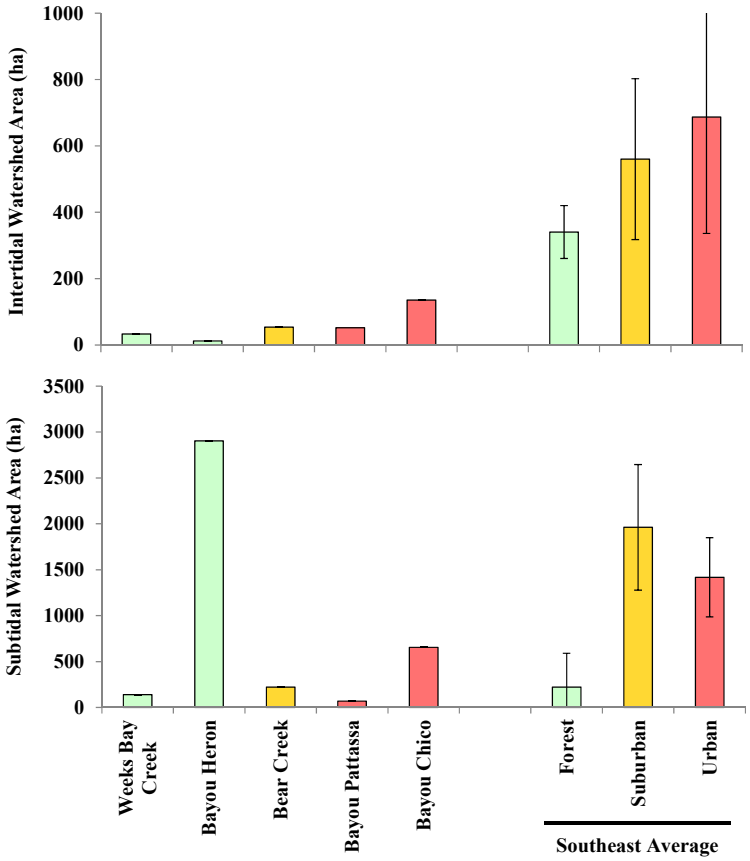

Figure 3-1. Area of intertidal (upper) and subtidal (lower) watersheds examined within this study. The subtidal watersheds include the associated intertidal areas. Land use class is marked by color.

Classifying watersheds based on impervious cover provides a useful framework for analyzing and interpreting study results. Similarly, impervious cover is a valuable indicator of the general level of development of watersheds that describes more complex conditions and attributes than other metrics such as the level of development or population density. For GoM intertidal and subtidal creek watersheds, human population density (individuals ha^{-1}) is linearly related to the impervious cover (%) which explains 90% and 82%, respectively, of the total variability (Figure 3-3). The slopes of the regression lines describing the relationship between total population density and impervious cover in the GoM are almost identical to those for the SE, indicating this is a very consistent predictor of human population density in coastal watersheds.

3.2 Exposures

3.2.1 Basic Water Quality

The basic water quality metrics that were sampled included temperature, pH, salinity, and dissolved oxygen (DO). Temperature affects the rate of chemical reactions, and organisms have differing physiological tolerances to temperature. Extreme values of pH can occur when acids or caustic materials enter creek waters thus this measure may indicate the presence of pollutants. pH also changes in response to photosynthesis and respiration by algae and to salinity through tidal fluctuations and rainfall. Salinity levels influence the distribution and diversity of many invertebrates and fish species and can be stressful to many estuarine organisms when large variations occur over short time periods. Low DO levels can limit distribution or survival of most biota, especially if conditions persist for extended periods (Van Dolah et al. 2004).

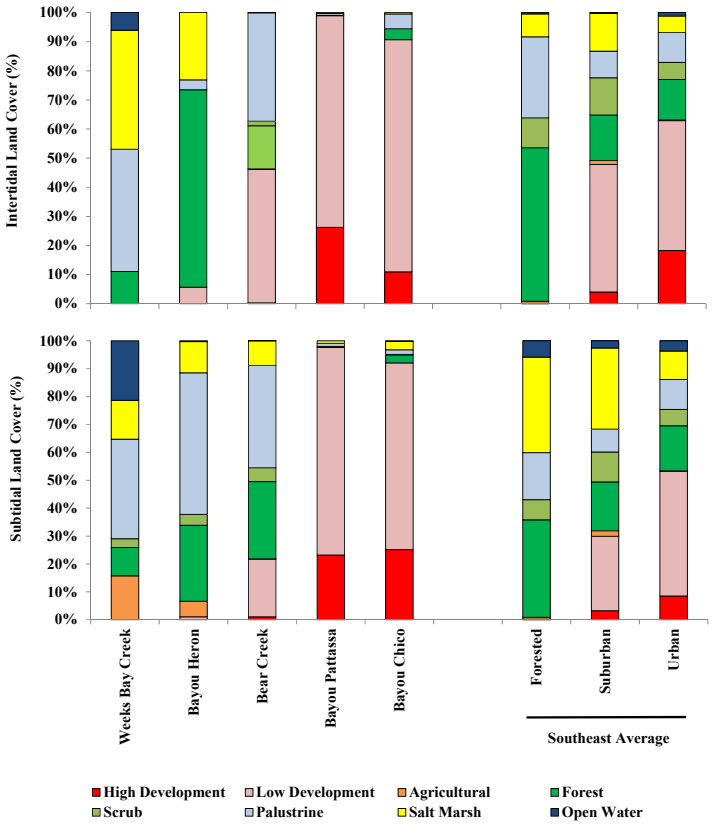

Figure 3-2. Proportional land cover categories (from NLCD 2001) within intertidal (upper) and subtidal (lower) watersheds examined within this study.

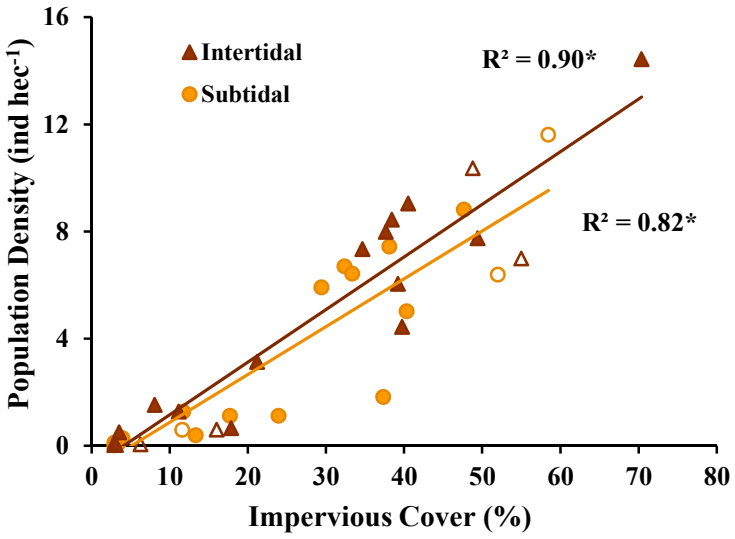

Figure 3-3. Relationship between population density and impervious cover within watersheds. Model R^2 is shown for each regression with asterisk (*) indicating significance ($p < 0.05$). Open markers represent GoM sites.

Among GoM creeks during the study period, average water temperature values varied from 28.7 ^{o}C (Weeks Bay, intertidal, forested) to 32.4 ^{o}C (Bayou Heron, intertidal, forested), and average pH values varied from 7.08 (Bayou Heron, intertidal, forested) to 8.11 (Bayou Chico, intertidal, urban) (No pH data were available for AL creeks). Average salinity values varied from 6.13 ppt (Weeks Bay, intertidal, forested) to 24.0 ppt (Bayou Chico, subtidal, urban), and average DO values ranged from 3.1 mg L^{-1} (Bayou Pattassa, subtidal, urban) to 10.5 mg L^{-1} (Weeks Bay, subtidal, forested). Temperature ranges (maximum minus minimum) varied from 2.3 ^{o}C (Bear Creek, intertidal, suburban) to 8.4° C (Bayou Pattassa, intertidal, urban), and pH ranges varied from 0.49 (Bayou Heron, subtidal, forested) to 1.03 (Bayou Chico, intertidal, urban). Salinity ranges varied from 3.74 ppt (Bayou Heron, subtidal, forested) to 22.0 ppt (Bayou Chico, intertidal, urban), and DO ranges varied from 3.02 mg L^{-1} (Bayou Pattassa, subtidal, urban) to 9.36 mg L^{-1} (Weeks Bay, intertidal, forested).

Average water quality values were not different in the GoM compared to the SE except for the average salinity values which were lower in the GoM (Appendix C). Average DO, pH and salinity values were significantly lower in the intertidal areas compared to the subtidal areas. Average DO

levels were the only water quality parameter which showed a land use class effect with the forested creeks having significantly higher DO levels than the urban creeks (Figure 3-4).

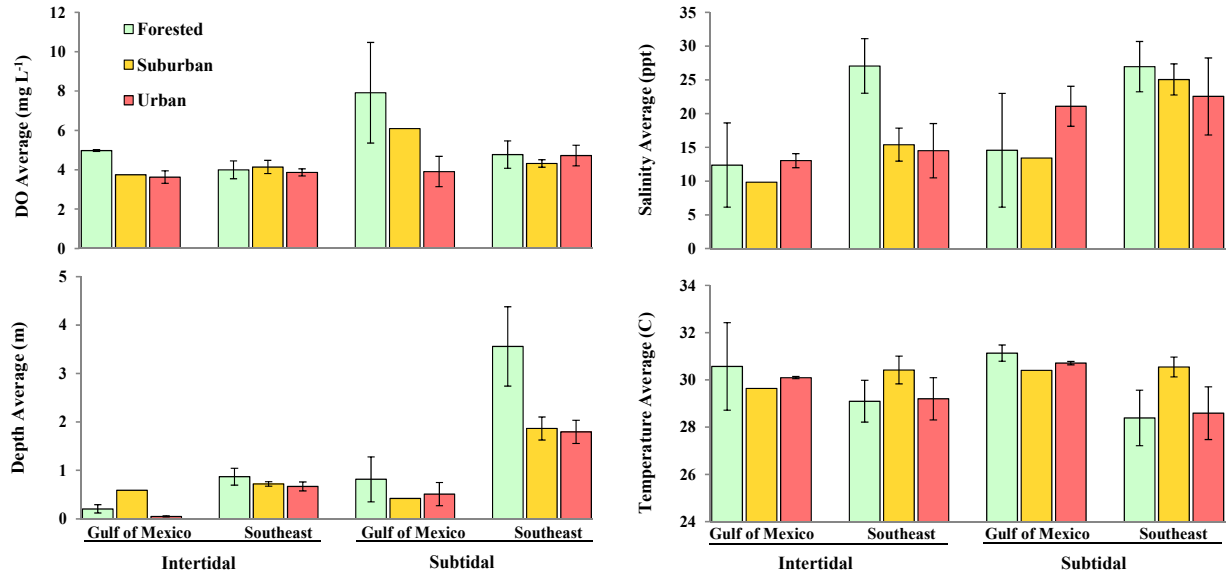

Figure 3-4. Average basic continuous water quality levels by land use, region, and longitudinal gradient. Bars represent average concentrations. Error bars are +/- 1 standard error.

Only the DO and pH ranges (max-min) were significantly different between regions with ranges being higher in GoM creeks than in SE creeks (Appendix C). Salinity range and pH range varied significantly with land use. The salinity ranges in GoM urban and suburban creeks were significantly larger than in forested creeks (Figure 3-5). Unlike salinity range, pH range was significantly greater in the suburban creeks than in urban and forested creeks. In addition, ranges of water quality metrics varied along the longitudinal spatial gradient, with intertidal creeks having significantly larger ranges for salinity, temperature, DO, and pH compared to the subtidal creeks.

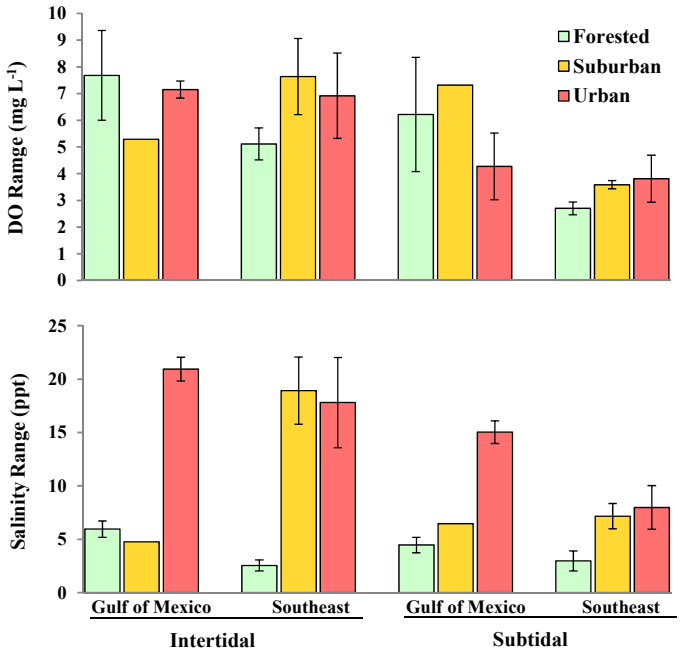

Figure 3-5. Range of basic continuous water quality levels by land use, region, and longitudinal gradient. Bars represent average concentrations. Error bars are +/- 1 standard error.

Intertidal and subtidal salinity ranges showed a significant positive relationship with the amount of impervious cover in the watersheds (Figure 3-6, Appendix D). This pattern in the intertidal areas is similar to that found by previous research in SC that attributed the pattern to flashier runoff (i.e., larger volumes and faster rates) from developed watersheds due to increased impervious cover (Lerberg et al. 2000, Holland et al. 2004). The

strength of the pattern in the subtidal areas (a pattern not observed in SE tidal creeks) may be due to the lower tidal range and presumably reduced oceanic exchange in creeks of the northern GoM. In addition, intertidal average DO % saturation (but not mg L^{-1}) showed a significant relationship decreasing with increasing amounts of impervious cover in the watersheds. This pattern has not been observed previously in the SE. The other basic water quality metrics had no statistically significant relationships with impervious cover.

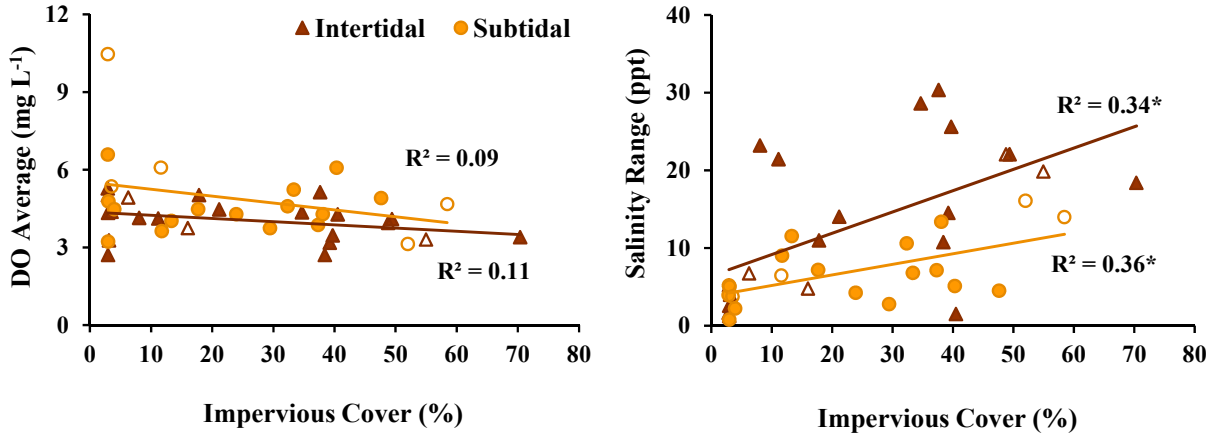

Figure 3-6. Relationship between basic continuous water quality and impervious cover for the study watersheds. Model R^2 is shown for each regression with asterisk (*) indicating significance ($p<0.05$). Open markers represent GoM sites.

3.2.2 Nutrients and Phytoplankton

Sampled nutrient and phytoplankton metrics included ammonium (NH_4^+), nitrate plus nitrite ($NO_{2/3}$), total dissolved nitrogen (TDN), total nitrogen (TN), orthophosphate (PO_4^{3-}), total dissolved phosphate (TDP), total phosphate (TP), silicate (DSi), chlorophyll-*a* (Chl-*a*), and phaeophytin (Phaeo). Enriched nutrient levels can be related to anthropogenic influences and may cause adverse impacts on creek biota. In particular, stormwater runoff from developed land carries $NO_{2/3}$ and PO_4^{3-} from fertilizer applications into creek waters. High $NO_{2/3}$ concentrations indicate possible creek eutrophication which can lead to algal blooms and enhanced organic matter deposition. This may result in increased respiration and low and fluctuating DO levels that can adversely impact creek biota. Chl-*a* is a measure of phytoplankton biomass, and high concentrations can indicate the presence of excessive algal standing stocks from eutrophication.

Nutrient and phytoplankton measures generally were not significantly different between the GoM and SE regions with the only exception being significantly higher TSS in the SE (Appendix C). Similarly, there were no significant interactions between region and land use class (forested, suburban, urban) or station type (intertidal, subtidal). The lack of interactions indicates that the model developed in the SE is directly applicable to the GoM in terms of nutrients and phytoplankton, and thus the two regions can be pooled and analyzed simultaneously for other effects (e.g., land use class, order).

All nutrient and phytoplankton measures were significantly higher in intertidal compared to subtidal creeks (with the exception of Chl-*a*, which was marginally significant – $p<0.10$). These

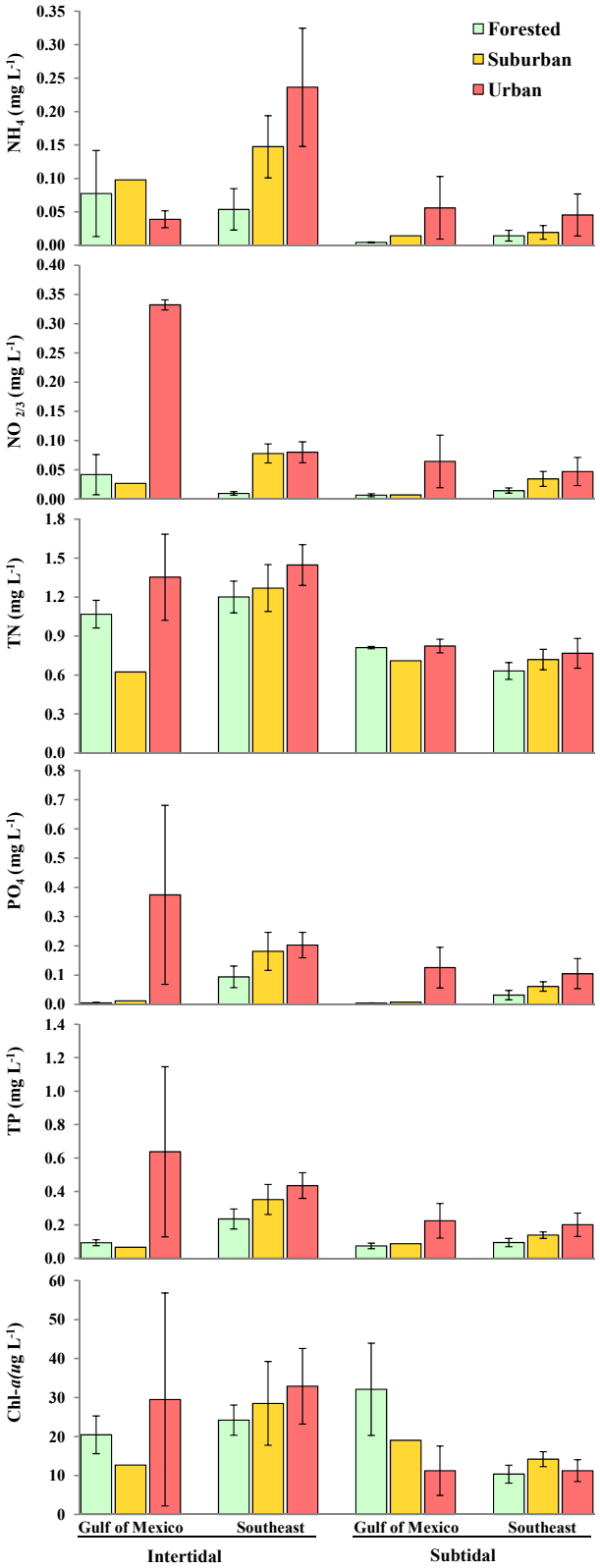

Figure 3-7. Nutrients and phytoplankton levels by land use, region, and longitudinal gradient. Bars represent average concentrations. Error bars are +/- 1 standard error.

measures were also generally higher in watersheds classified as urban with various measures of phosphorous (TP, TDP, and PO_4^{3-}) and inorganic nitrogen ($NO_{2/3}$ and NH_4) being significantly different across land classes (Figure 3-7, Appendix C). Although not significant, the phytoplankton pigments (Chl-*a* and Phaeo) were the only measures that were lower in urban creeks than in forested or suburban creeks. The results for phosphorus may reflect a combination of fertilizer use and stormwater runoff in developed watersheds. They may also reflect human land disturbance (including historical mining activities) interacting with natural phosphate deposits. In the GoM, land use class differences were primarily driven by elevated phosphorus and $NO_{2/3}$ levels in the intertidal and subtidal sections of both urban systems sampled, Bayou Chico and, to a lesser extent, Bayou Pattassa.

Similar but generally weaker relationships emerged when analyzing nutrients and phytoplankton measures as a function of impervious cover (Appendix D). $NO_{2/3}$ increased significantly in both intertidal and subtidal areas; measures of NH_4 in intertidal areas and phosphorous in both intertidal and subtidal areas increased marginally ($p<0.10$) with increasing impervious cover (Figure 3-8).

Overall, the incorporation of the GoM data into the larger SE dataset resulted in the nutrient and phytoplankton measures having stronger (higher R^2 and lower p-values) relationships among land classes and creek station types. This suggests that data across a larger geographic scale and a broader range of conditions supported our conceptual model and increased the predictive power accordingly.

Based on categorical guidelines developed for coastal waters by NOAA (Bricker et al. 1999), concentrations found in this study for TDN and TDP ranged from medium to high, and Chl-*a* from low to hypereutrophic. In general, intertidal creek concentrations were classified into the higher categories compared to the subtidal creeks. The only TDN concentrations (n=5) classified as high were for intertidal creeks draining suburban and urban watersheds in the SE. All other sites had TDN concentrations classified as medium. Twenty sites had TDP concentrations classified as high with the majority of the sites being located in the SE in intertidal areas of suburban and urban watersheds. All other sites had TDP concentrations classified as medium. Chl-*a* concentrations in two intertidal sites from suburban and urban watersheds in the SE were classified as hypereutrophic. In an additional thirteen sites, Chl-*a* concentrations were classified as medium, primarily from intertidal areas. Only five sites were classified as low based on Chl-*a* concentrations.

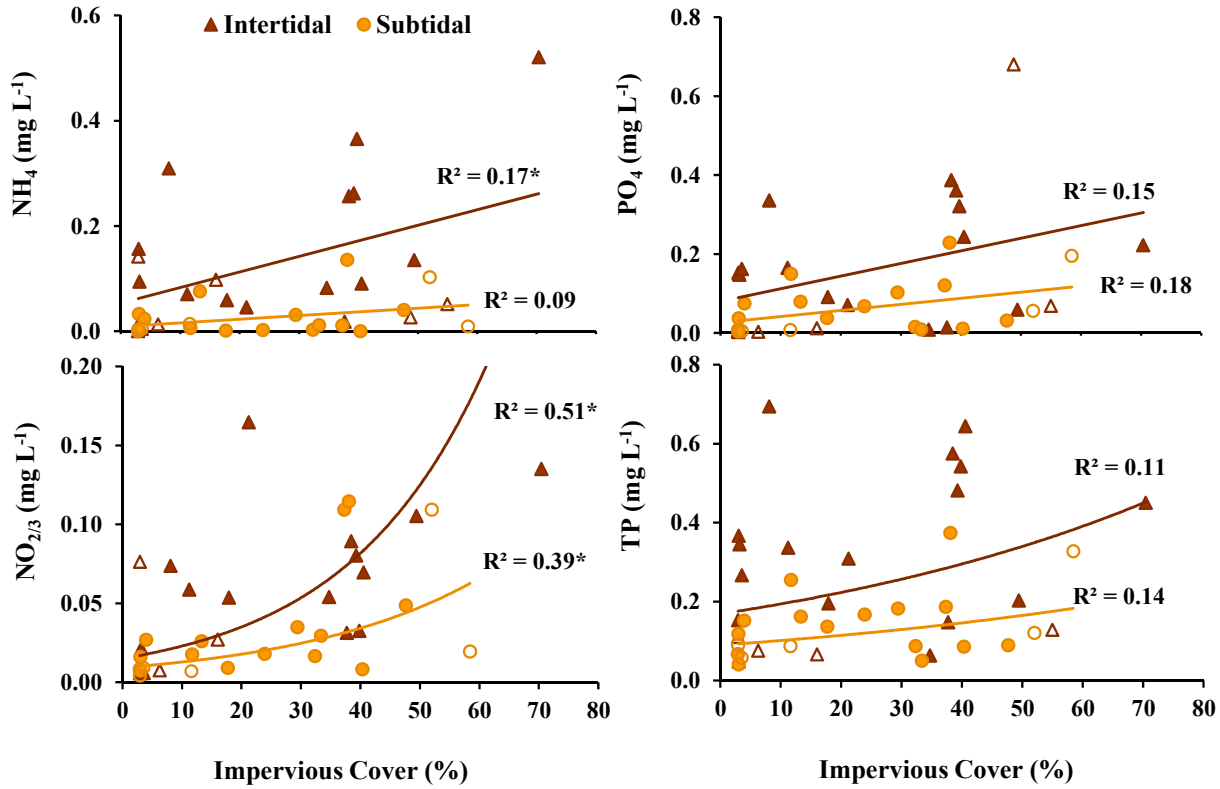

Figure 3-8. Relationship between nutrient concentrations and impervious cover for the study watersheds. Model R^2 is shown for each regression with asterisk (*) indicating significance ($p<0.05$). Open markers represent GoM sites. The regressions for $NO_{2/3}$, and TP were performed on log transformed data but are shown here as quadratic on non-transformed data scale.

3.2.3 Pathogens

Fecal coliform (FC) and Enterococcus (ENT) bacteria have been used extensively as indicators of fecal pollution and enteric pathogens. However, they may be inadequate indicators for all pathogens that are associated with fecal pollution, particularly enteric viruses. F+ and F-coliphages are viruses that infect *Escherichia coli*; these are being investigated to determine if

they are a more appropriate indicator for water-borne pathogens (IAWPRC 1991, Havelaar et al. 1993).

Fecal coliform concentrations ranged from <1 to 91,000 colony forming units (CFU) 100 ml^{-1} in the SE and 2 to 89,550 CFU 100 ml^{-1} (estimated based on oyster tissue concentrations) in the GoM. Enterococci concentrations ranged from 3 to 21,000 CFU 100 ml^{-1} in the SE and 0 to >600 CFU 100 ml^{-1} in the GoM. Levels of indicator viruses tended to be lower than those of the bacteria. F+ coliphages ranged from 0 to 453 plaque forming units (PFU) 100 ml^{-1} in the SE and 0 to 14 PFU 100 ml^{-1} in the GoM. F- coliphages ranged from <1 to 2,615 PFU 100 ml^{-1} in the SE and 0 to 253 PFU 100 ml^{-1} in the GoM.

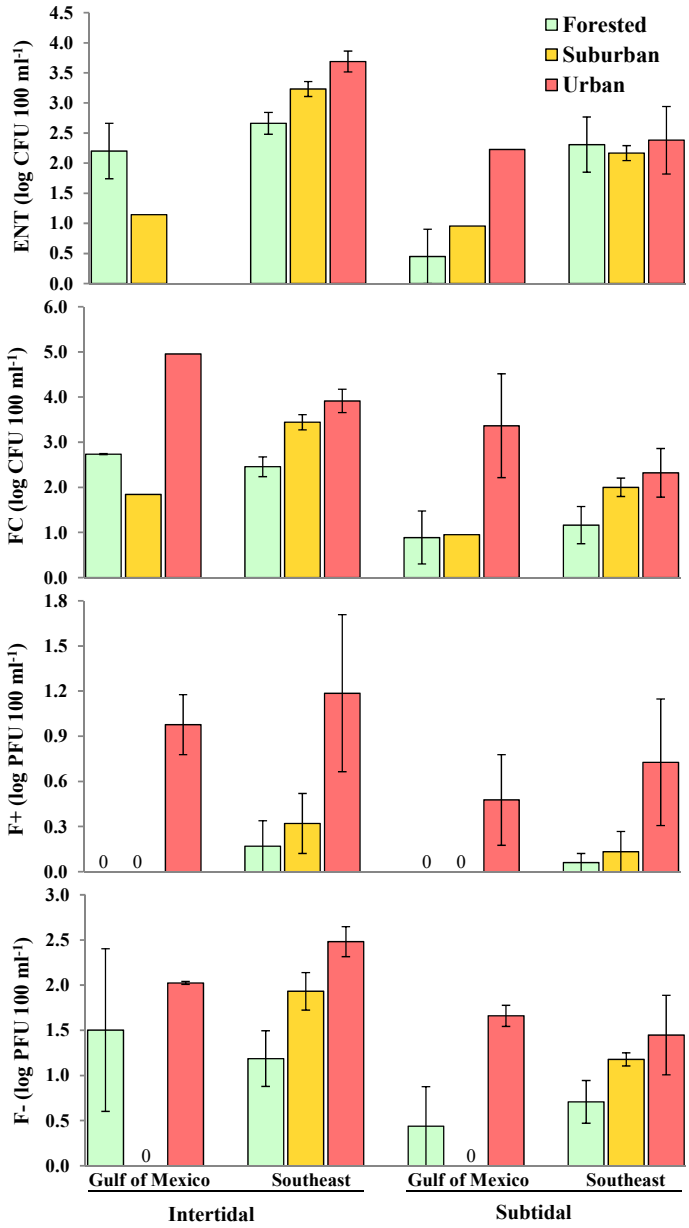

Figure 3-9. Pathogen indicator levels by land use, region, and longitudinal gradient. Bars represent average concentrations. Error bars are +/- 1 standard error.

The land use class of the watershed surrounding the tidal creeks, the spatial longitudinal gradient sampled, and the geographic region were found to affect both bacterial (FC and ENT) and viral pathogen (F+ and F-) indicator densities (Appendix C, Figure 3-9). Levels of ENT and F- were found to be significantly higher in the SE compared to the GoM; however, FC and F+ were similar between the two geographic regions. Levels of FC, ENT, and F+ showed a similar pattern of increasing values from forested to suburban to urban watershed classes in both regions. For FC and F+, the urban watershed classes were significantly higher than the forested and suburban classes which were similar. For ENT, the urban watershed class was significantly higher than the forested class but both were similar to the suburban class. FC, ENT, and F- concentrations were significantly higher in the intertidal compared to the subtidal areas. The F+ coliphage concentration showed a similar trend but this trend was not statistically significant (Appendix C).

Concentrations of pathogen indicators significantly increased with increasing levels of impervious cover in the watersheds (except ENT in subtidal areas) (Appendix D). The relationships were stronger in the intertidal systems

compared to the subtidal systems (Figure 3-10). Enterovirus and norovirus were measured in the GoM only, but no detectable concentrations were found in any of the creeks sampled.

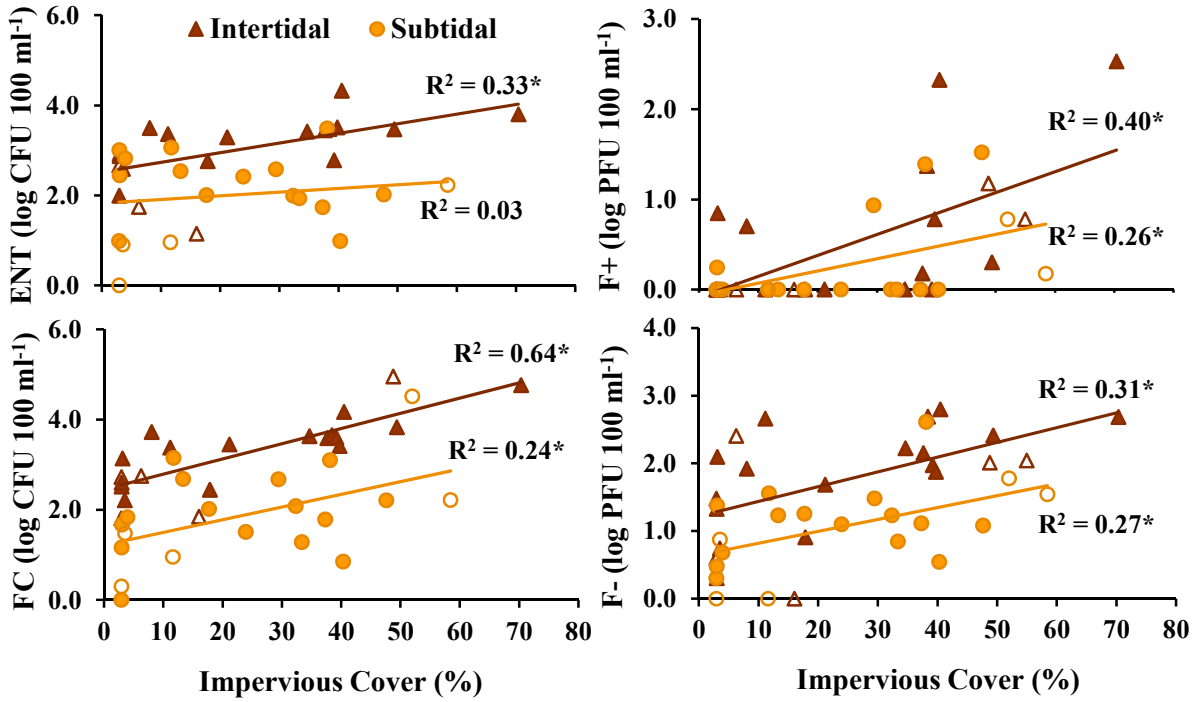

Figure 3-10. Relationship between pathogen indicators and impervious cover for the study watersheds. Model R^2 is shown for each regression with asterisk (*) indicating significance ($p < 0.05$). Log transformation is x + 1 for ENT, F+ and F-. Open markers represent GoM sites.

3.2.4 Sediment Quality

Tidal creek sediments are often characterized as pluff mud which is a soft sediment rich in organic matter and high in clay content. These types of sediments are often repositories for chemical contaminants entering tidal creeks from stormwater runoff and recreational use of the waters. For example, trace metal contaminants bind to clay particles while organic contaminants bind to organic carbon (Beeftink et al. 1982, Boehm and Farrington 1984, Barrick and Prahl 1987). Therefore, higher clay and total organic carbon (TOC) levels will often be associated with higher contaminant levels. Ammonious nitrogen (TAN) is also an important indicator of sediment quality, and some components of TAN may be toxic to benthic organisms.

3.2.4.1 Sediment Composition

The coarse grain composition (% sand) and the clay content (% clay) were not significantly related to region, surrounding land use or station type (Appendix C, Figure 3-11). The silt content of the sediment was significantly affected by the region, surrounding land use, and station type with significant interaction terms for region by land use class and region by station type. This indicates that the patterns were more complex and varied depending on the two regions for this sediment component. Sediment characteristics were not associated with the levels of impervious cover in the surrounding watershed for the intertidal or subtidal habitats (Appendix D).

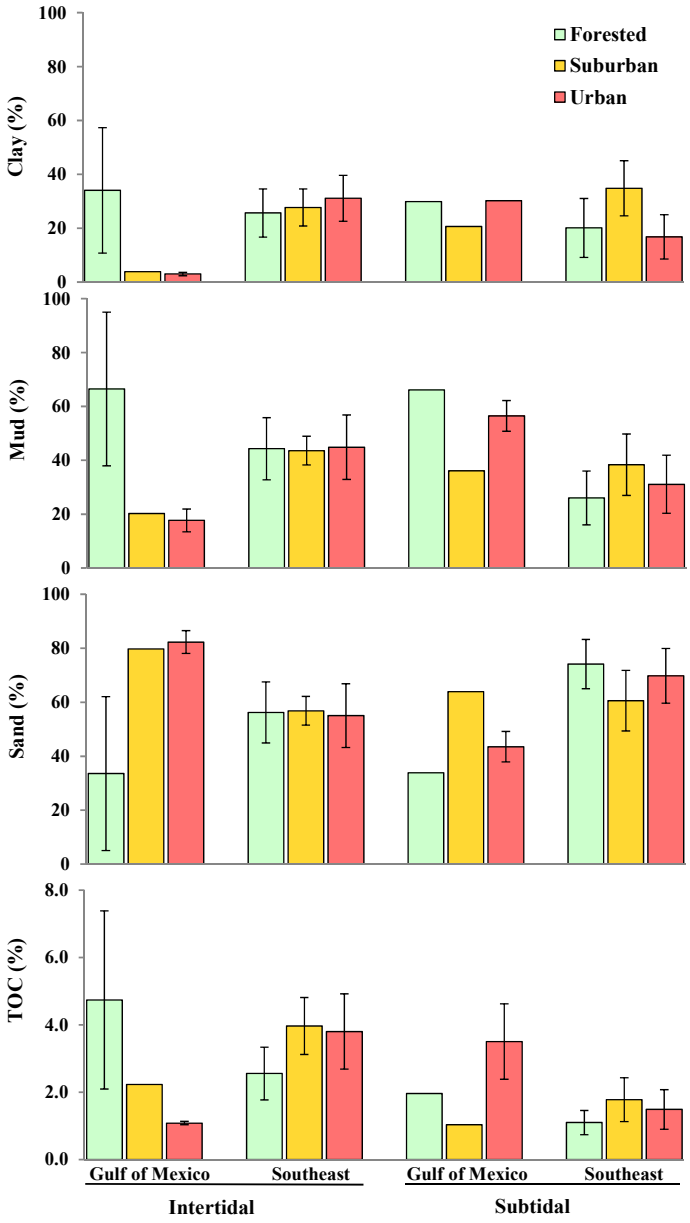

Figure 3-11. Sediment characteristic levels by land use, region, and longitudinal gradient. Mud is the sum of clay and silt which were not distinguished for every sample. Bars represent average concentrations. Error bars are +/- 1 standard error.

Porewater TAN was not significantly related to region, land use class or station type (Appendix C). Porewater TAN was not significantly associated with the impervious cover level of the surrounding watersheds for either the intertidal or subtidal habitats (Appendix D).

3.2.4.2 Sediment Contamination

Polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), pesticides, and polybrominated diphenyl ethers (PBDEs) were measured during this study. PAHs are a major component of lubricating oils and fossil fuels which are released into the environment when these products are spilled or combusted. Potential sources include runoff from highways and parking lots, street dust, fuel spills, marinas and recreational boating activities, and atmospheric fallout (reviewed by Weinstein 1996). In addition, forest fires provide a natural source of many PAH compounds. The production of PCBs was banned in the 1970s, but these compounds have been reported to accumulate and persist in estuarine environments (reviewed by Weinstein 1996). Most of the pesticides measured for this study are historical pesticides such as DDT, mirex, and chlordane (this group was also banned in the 1970s). Current-use pesticides were not measured because they are generally more water soluble and do not accumulate as readily in sediments. The PBDEs are contaminants of emerging concern that are used as flame retardants in furniture, plastics, and clothing. Although trace metals are naturally occurring elements with concentrations influenced by natural weathering of basement rock (Williams et al. 1994), several are also anthropogenically enhanced from industrial and urban uses (e.g., lead, chromium,

copper, cadmium, zinc, and mercury). Due to the large number of analytes measured, only the totals and a few of the anthropogenically introduced metals will be discussed.

The intertidal portions of the most urbanized creeks in the SE including New Market, Shem, and Burnett creeks had a number of exceedences of the ERL for PAHs, PCBs, and pesticides. Similarly, Bayou Pattassa, one of the two urban creeks in the GoM, had ERL exceedences for 4 individual PAHs and total PAH, total DDT, and cis-Chlordane (Table 3-2). Weeks Bay Creek also exceeded the ERL for total DDT and 4,4-DDE. Elevated concentrations of DDT have also been observed in other forested and developed creeks (Sanger et al. 1999a, Sanger et al. 2008). No metal concentrations exceeded the effects range low (ERL) or effects range median (ERM) in the intertidal habitats of any of the GoM creeks sampled (Table 3-2). In contrast, metal contamination in the SE intertidal creeks was not uncommon. In particular, arsenic concentrations, a naturally elevated metal throughout the SE, were found to commonly exceed the ERL. None of the sediment contaminant levels exceeded the ERM level for any parameter.

Table 3-2. Characteristics and contaminant concentrations in tidal creek sediments for selected parameters. Total PAH is the sum of 23 analytes and total PCB is the sum of 79 analytes. Organics are ng g^{-1} dry weight and metals are µg g^{-1} dry weight. Italicized numbers indicate concentrations that exceed the ERL (as defined by Long et al. 1995) for that parameter.

		Sediment			Organics			Trace Metals					
Land Use	Creek	Clay (%)	Mud (%)	TOC (%)	Total PAHs	Total PCBs	Total DDTs	As	Cr	Cu	Hg	Pb	Zn
Intertidal													
Forested	Weeks Bay Creek	57.34	92.61	6.87*	216	0	***2.69***	***13.5***	46.0	11.8	0.093	28.2	106
	Bayou Heron	10.75	34.48	2.1*	30	0	0	1.7	16.8	1.8	0.016	10.3	11
Suburban	Bear Creek	3.87	7.13	0.51*	7	0	0	2.1	14.3	1.3	0.007	4.3	6
Urban	Bayou Chico	2.39	3.98	0.36*	41	0	0	0.5	6.7	1.6	0.005	5.2	28
	Bayou Pattassa	3.63	8.19	3.27	***5072***	11.8	***6.34***	1.6	11.2	7.0	0.026	27.2	88
Subtidal													
Forested	Weeks Bay Creek				156	0	***1.93***	***13.9***	62.4	12.2	0.090	27.2	109
	Bayou Heron	29.90	85.04	1.96**	161	0.15	0	5.6	33.0	9.5	0.049	22.2	64
Suburban	Bear Creek	20.68	40.99	1.03**	68	0	0	6.8	29.4	6.1	0.031	10.1	41
Urban	Bayou Chico	30.20	63.86	2.38**	849	10.4	***1.73***	7.6	41.9	25.9	0.082	46.0	***225***
	Bayou Pattassa		53.49	5.41	***7245***	***66.2***	***16.21***	5.8	28.9	26.9	0.121	***69.2***	***302***
	ERL				***4022***	***22.7***	***1.58***	***8.2***	***81***	***34***	***0.15***	***46.7***	***150***
	ERM				***44792***	***180***	***46.1***	***70***	***370***	***270***	***0.71***	***218***	***410***

* paired site data not available - average of the reach is provided
** paired site data not available – average of the station type is provided

The subtidal areas in the GoM creeks had more chemical contaminant exceedences than the intertidal areas which is in contrast to the spatial pattern observed in the SE where exceedences were more common in the intertidal areas. This occurred in the GoM for select metals and total DDT (Table 3-2). The subtidal habitat in Bayou Pattassa, one of the two urban creeks in the GoM, had ERL exceedences for 7 individual PAHs, total PAH, 4,4-DDE, and total DDT concentrations. The ERM was exceeded for Dibenz(a,h)anthracene and cis-Chlordane. Total PCBs, lead, and zinc concentrations were also found to exceed the ERL in the subtidal area of Bayou Pattassa. Bayou Chico, the other urban GoM creek, exceeded the ERL for total DDT and zinc in the subtidal area. The subtidal portion of Weeks Bay Creek, an AL NERR forested site, exceeded the ERL for total DDT and 4,4-DDE (Table 3-2).

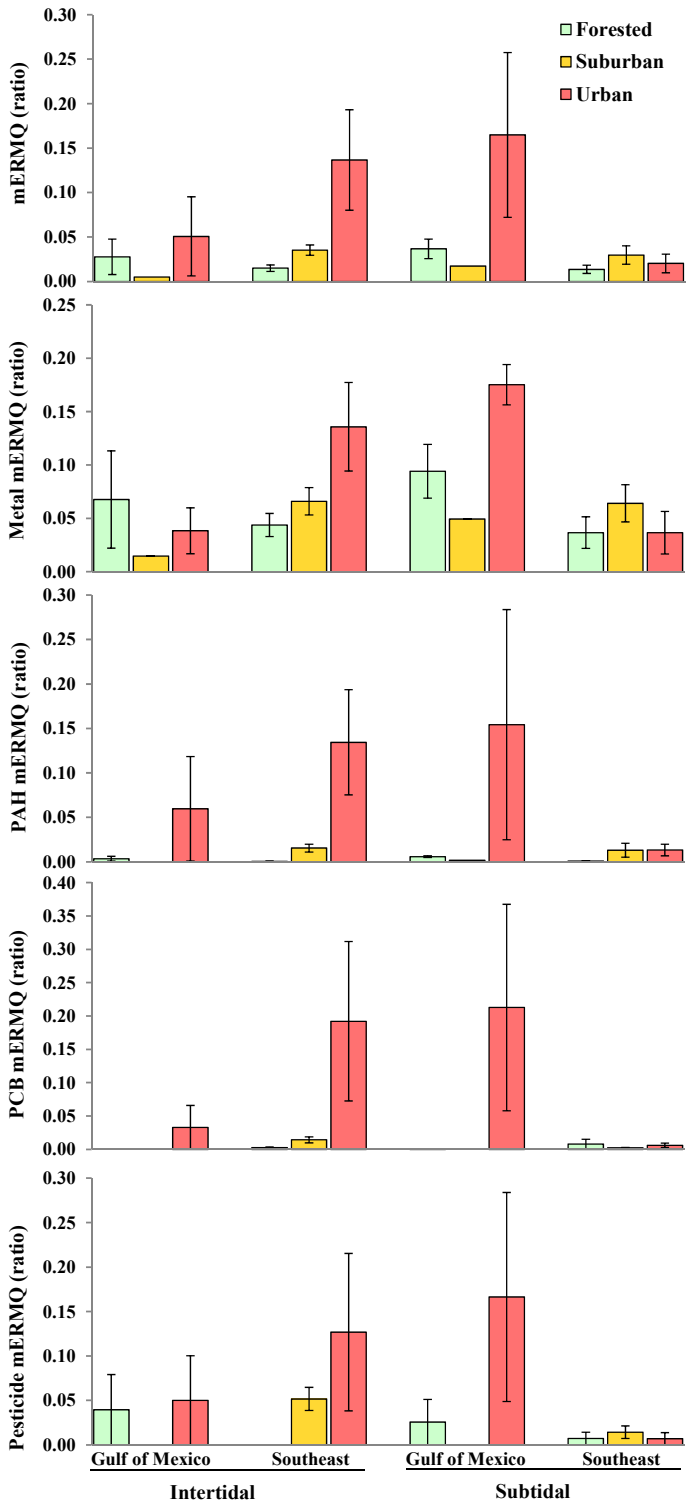

Figure 3-12. mERMQ levels by land use, region, and longitudinal gradient. Bars represent average concentrations. Error bars are +/- 1 standard error.

The total mean effects range median quotient (mERMQ) values ranged from 0.005 to 0.257 in GoM creeks and 0.0036 to 0.375 and SE creeks. When mERMQ values were analyzed in a 3-way ANOVA, region and station type were not significant. The total mERMQ levels were similar in the forested and suburban land use classes and both were significantly lower than total mERMQ levels in the urban land class (Appendix C, Figure 3-12). The region by station type interaction was significant due to a significantly higher total mERMQ levels in the intertidal compared to the subtidal habitats in the SE versus higher total mERMQ levels (although not significantly different) in the subtidal compared to the intertidal habitats in the GoM. The metals mERMQ values were similar across region, land use class, and station type. The PAH mERMQ and PCB mERMQ values were significantly higher in the urban land use class compared to the other classes (Appendix C, Figure 3-12). The pesticide mERMQ levels were significantly higher in the urban class compared to the forested class, and both were similar to the suburban class.

Regression analysis demonstrated that mERMQs increased with increasing levels of impervious cover (Appendix D, Figure 3-13). Regressions of all quotients versus impervious cover were statistically significant in the intertidal creeks. Regressions of all quotients except metals ERMQ versus impervious cover were statistically significant in the subtidal creeks. The relationships were stronger in the intertidal creeks compared to the subtidal creeks for all of the quotients except PCB mERMQ.

PDBEs were only detected in the more developed creeks: Bayou Chico and Bayou Pattassa (urban) in the GoM, and James Island and Orangegrove (suburban) and Bulls, Shem, and New Market (urban) in the SE (Figure 3-14). In the SE, PBDE levels above the detection limit were only found in the intertidal creeks; however, in the GoM detectable concentrations were found in both the intertidal and subtidal areas. Intertidal creeks as well as subtidal creeks in GoM appeared to be potentially valuable sentinels for detecting emerging contaminants of concern.

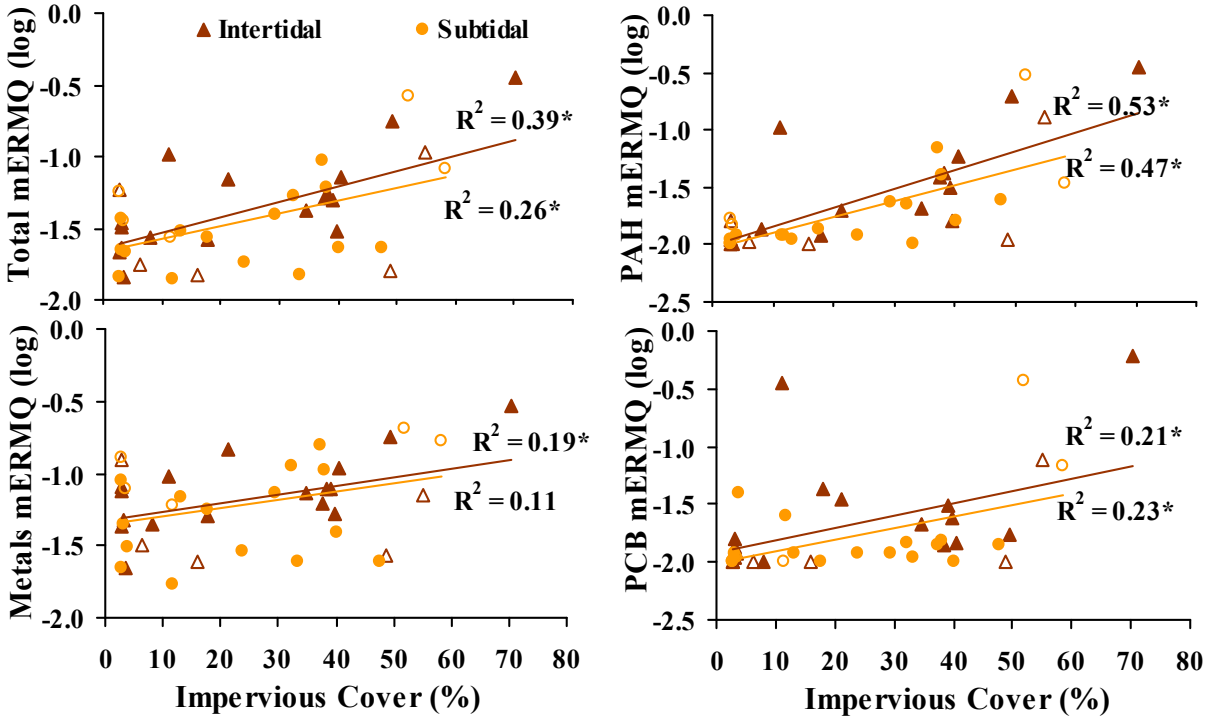

Figure 3-13. Relationship between mERMQ (log+0.01) and impervious cover for the study watersheds. Model R^2 is show for each regression with asterisk (*) indicating significance ($p<0.05$). Open markers represent GoM sites.

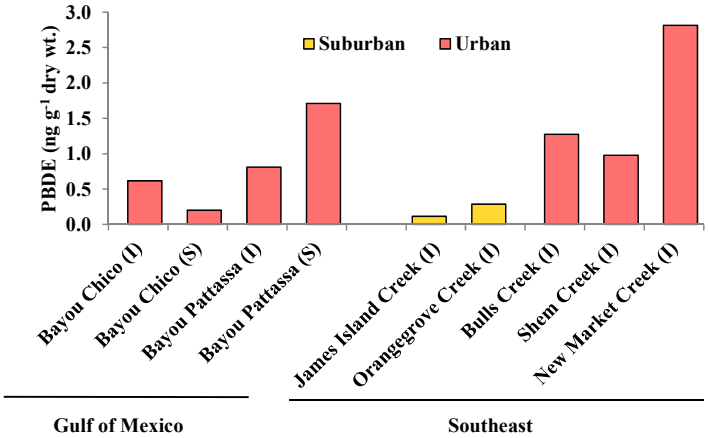

Figure 3-14. Concentrations of polybrominated diphenyl ethers (PBDEs) in creek sediments. Only the sites with detectable levels are shown. I indicates an intertidal area and S indicates a subtidal area. Land use class is marked by color. Bar represents measured concentrations.

3.3 Ecological Response

3.3.1 Macrobenthic Community

The macrobenthic community was sampled to examine how different levels of development affect abundances and distributions of these organisms. In the GoM, a total of 2,530 organisms representing 40 taxa were collected in 66 samples. In the SE, a total of 18,296 organisms representing 279 taxa were collected in 303 samples. Sampling methods varied between intertidal and subtidal sections of the creeks system, as did volume of sediment collected. Because of these differences, abundances were converted to density (expressed as ind. m^{-2}) prior to analysis.

In the GoM, intertidal creek macrobenthic communities had higher densities (668,222 ind. m^{-2} and 27 taxa) than subtidal creeks (37,957 ind. m^{-2} and 29 taxa) but similar number of taxa. Within individual GoM creeks, the largest numbers of macroinvertebrate taxa were collected from Bear Creek and Bayou Heron (16-20 taxa depending on creek and station type). Eight or fewer taxa were collected from all other GoM creek station type. The amphipod *Corophium simile* and tanaid *Paratanais* sp. numerically dominated the intertidal macrobenthic invertebrate community with most of those being found in Bear Creek. Other numerically abundant species found in the intertidal GoM creeks included the polychaete *Ceratocephale oculata*, various tubificid worms and *Grandidierella* amphipods. Among subtidal creeks, *C. oculata* was the numerically most abundant taxon in subtidal collections, primarily from Bayou Pattassa. Nemerteans were the next most abundant taxon, and these were most abundant in the forested creeks, Weeks Bay and Bayou Heron.

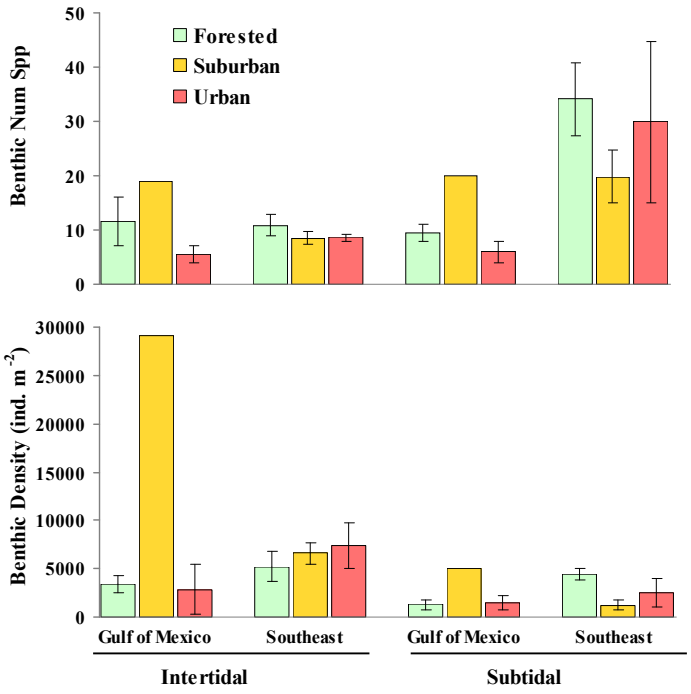

Figure 3-15. Macrobenthic community number of species and total abundance by land use, region, and longitudinal gradient. Bars represent average concentrations. Error bars are +/- 1 standard error.

Average macrobenthic invertebrate abundances and number of species per sample were not significantly different between the GoM and SE in either intertidal or subtidal creeks (Appendix C, Figure 3-15). Both measures varied significantly with land use class in intertidal creeks but not in subtidal creeks (Appendix C). Overall, densities and numbers of species were significantly higher in GoM suburban intertidal creeks than in forested or urban creeks. This was due to particularly high infaunal densities in Bear Creek, the only suburban system sampled in the GoM (Figure 3-15). Macrobenthic invertebrate densities were not significantly different between forested and urban creeks in this region. The significant difference in number of species among intertidal

creeks representing different land use classes was due to a significant difference between suburban and urban intertidal creeks; all other comparisons were not significant (Appendix C).

Macrobenthic invertebrate abundance was not significantly related to impervious cover in either intertidal or subtidal creeks (Figure 3-16). The relationship between number of species and impervious cover was negative and marginally significant ($p = 0.067$) in intertidal creeks and negative and not significant in subtidal creeks (Figure 3-16), except for a single large macrobenthic invertebrate density in Bear Creek, AL.

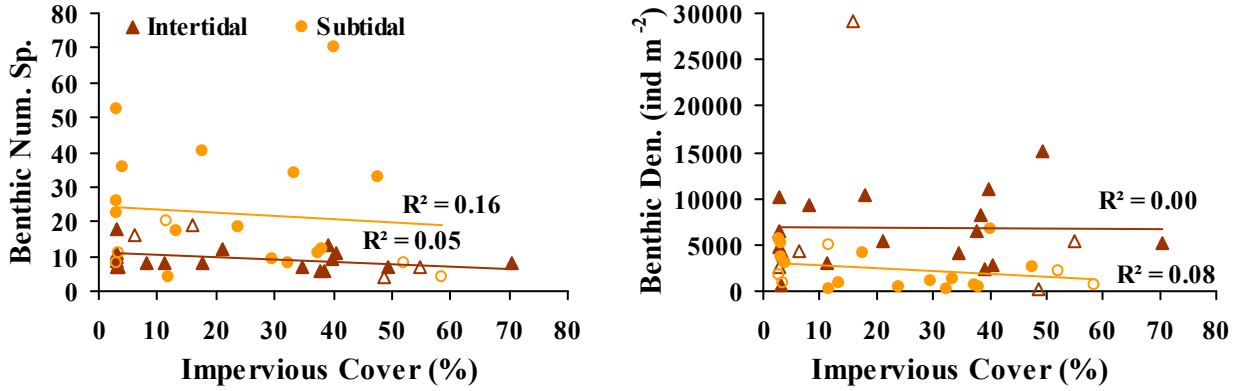

Figure 3-16. Relationship between macrobenthic number of species and density versus impervious cover for the study watersheds. Model R^2 is show for each regression with asterisk (*) indicating significance ($p<0.05$). Open markers represent GoM sites.

When examined using multivariate analyses (i.e., MDS), the composition of the macrobenthic communities, the GoM communities were significantly different from those of the SE in both intertidal (ANOSIM: $R = 1.000$, $p < 0.001$) and subtidal creeks (ANOSIM: $R = 0.462$, $p = 0.011$). Most of these differences were due to the identities of the dominant fauna differing between the two biogeographic regions. Intertidal communities within each region clearly clustered together, and the communities of the two regions did not overlap (Figure 3-17A). Eight of the ten taxa most responsible for the dissimilarity between intertidal creeks of the GoM and SE were present in only one region (Table 3-3). The subtidal macrobenthic community of the SE had far more taxa than the intertidal communities. In addition, the subtidal communities in the two regions shared a larger proportion of taxa compared to the intertidal communities. As a result, the clusters representing subtidal communities from the GoM and SE regions did not diverge as strongly as they did for intertidal habitats (Figure 3-17B). Five of the ten taxa most responsible for the dissimilarity between the subtidal creeks of the two regions were, however, present in only one region (Table 3-3). Of the taxa most responsible for the differences between the two regions in both intertidal and subtidal creeks, most were either polychaetes or oligochaetes. Exceptions included the amphipod *Corophium simile*, the gastropod *Ilyannasa obsoleta* and Nemerteans.

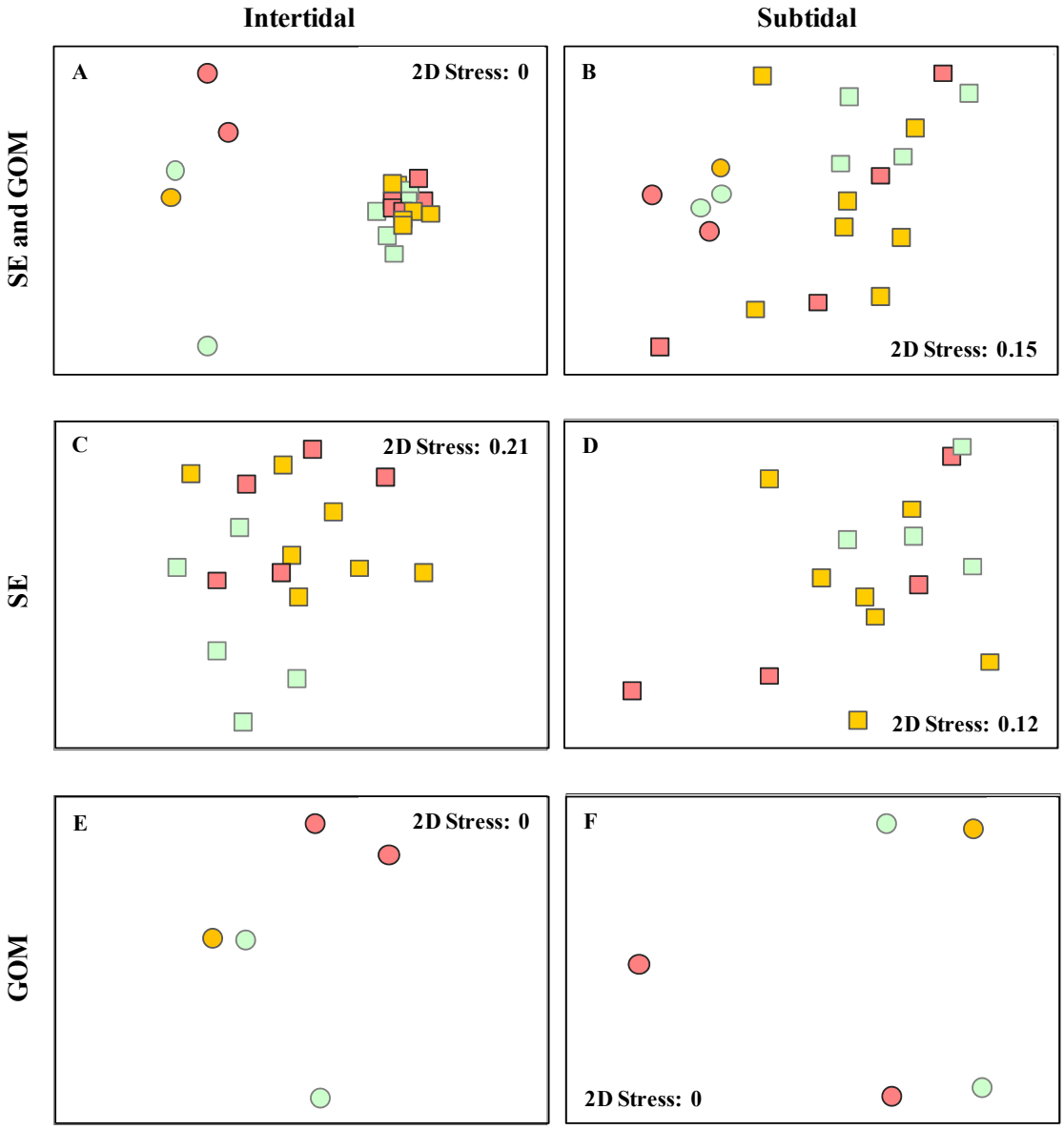

Figure 3-17. Multidimensional scaling (MDS) ordination of benthic communities with SE and GoM regions combined (A and B), just the SE region (C and D) and just the GoM region (E and F). Boxes = SE region creeks, circles = GoM region creeks. Green = forested creeks, yellow = suburban creeks and red = urban creeks.

Due to the dissimilarity of the benthic communities between the SE and GoM, the relationships with land use class were analyzed for each region separately. In SE creeks, benthic community composition varied significantly with land use class for intertidal creeks (ANOSIM: R = 0.229, p = 0.012), but no relationship with land use class was found for subtidal creeks (ANOSIM: R = 0.098, p = 0.208). Intertidal macrobenthic community composition in suburban creeks was significantly different than in forested creeks (R = 0.349, p = 0.008) and forested creeks were marginally significantly different than urban creeks (R = 0.212, p = 0.071). The macrobenthic community inhabiting suburban and urban creeks were not different (R = -0.060, p = 0.634). In

intertidal creeks, the species responsible for the differences among land use classes were primarily annelids. An unidentified gastropod and the nemerteans were notable exceptions (Table 3-4). In subtidal creeks, annelids were also most important in detecting differences among land use classes, but non-annelid taxa played a more important role here and included the amphipods *Ampelisca abdita* and *C. simile*, the gastropod *I. obsolete,* and the bivalve *Tellina agilis* (Table 3-4).

Due to a small sample size in the GoM, differences among land use classes were not analyzed using ANOSIM. Although not statistically analyzed, macrobenthic communities of urban creeks clearly separated from those of suburban and forested creeks. In GoM intertidal creeks as in the SE intertidal creeks, annelids were most responsible for differentiating among land use classes, but two amphipods C. *simile* and *Grandidierella* sp., an unidentified pelecypod bivalve, and the tanaid *Hargeria rapax* were also important (Table 3-5). In subtidal creeks, annelids also dominated the taxa most responsible for the differences among land use types with three amphipods *A. abdita*, *C. simile,* and *Ericthonius brasiliensis*, an unidentified gastropod, an unidentified pelecypod bivalve and nemerteans also being important (Table 3-5).

3.3.2 Nekton Community

Table 3-4. Results of SIMPER analysis of land use differences in macrobenthic community structure in southeast creeks.

	Abundance in			% Contribution to Difference		
Taxon	For	Sub	Urb	For vs. Sub	For vs. Urb	Sub vs. Urb
Intertidal						
Tubificoides brownae	4.69	1.35	3.61	8.15	6.2	9.62
Tubificoides heterochaetus	1.57	3.66	4.37	7.82	9.12	9.25
Laeonereis culveri	3.67	5.34		6.7	7.06	--
Heteromastus filiformis	3.29	1.38	2.18	6.31	5.94	5.67
Monopylephorus	5.89	8.16	7.16	6.01	6.05	6.14
Fabricia sp.	0	2.8	0.55	5.67	--	7.58
Streblospio benedicti	4.77	2.96	3.48	5.29	6.6	8.53
Nereis succinea	3.34	2.42	3.3	5.25	--	6.61
Spionidae	2.83	1.58		5.14	--	--
Gastropoda	2.08	0	0.78	4.48	--	4.77
Nemertea	3.84	2.67	0.62	--	6.96	7.01
Sabellidae		0.93	2.03	--	--	6.28
Capitomastus aciculatus	2.11		1.12	--	4.8	--
Subtidal						
Polydora cornuta	2.46	1.68	2.21	2.27	2.04	--
Mediomastus sp.	5.29	2.98	2.71	2.27	2.95	4.01
Nereis succinea	4.05	2.49	3.19	2.22	1.82	2.93
Ampelisca abdita	2.69	1.28		2.11	--	--
Tubificoides brownae	4.38	3.55	2.21	2.04	2.7	4.27
Tharyx acutus	1.92	0	0	2.02	1.8	--
Scoletoma tenuis	2.75	2.2	2.68	1.98	2.05	2.88
Streblospio benedicti	6.16	4.94		1.97	--	--
Scoloplos rubra	2.56	1.43	1.91	1.94	1.9	--
Corophium simile	2.51	0		1.94	--	--
Tubificoides wasselli	2.47	2.28	1.4	--	2.01	3.56
Ilyanassa obsoleta	2.23	1.68	1.5	--	2.11	3.54
Tubificoides heterochaetus		0.31	1.76	--	--	3.33
Tellina agilis		0.23	2.75	--	--	3.23
Spionidae		1.87	2.57	--	--	2.99
Heteromastus filiformis	3.08	2.31	1.24	--	2.15	2.88

Nekton, primarily larger crustaceans and fish, were only collected from four intertidal creeks in the GoM with two representing forested and two representing urban land use (Bear Creek was not sampled). Of the intertidal creeks examined, the forested Bayou Heron hosted the highest average densities (9,170 ind. ha^{-1}) and average numbers (8 taxa) of nekton taxa, followed by the urban Bayou Chico (5,288 individuals ha^{-1}, 5.7 taxa) and the forested Weeks Bay Creek (3,593 ind. ha^{-1}, 4.0 taxa). The urban creek Bayou Pattassa hosted the lowest densities (1,970 ind. ha^{-1}) and number (3.3 taxa) of nekton taxa. The most abundant nekton taxon collected in the GoM was the unidentified menhaden *Brevoortia* sp. (average density 1,212 ind. ha^{-1}), but it was only caught in Bayou Heron at a density of 4,847 ind. ha^{-1}. This was followed by unidentified juvenile shrimp in the genus *Penaeus* (average density 1,032 ind. ha^{-1}) which was found in all four creeks, the molly *Poecilia latipinna* (725 . ha^{-1}) found primarily in urban creeks, and the unidentified anchovy *Anchoa* sp. (477 ind. ha^{-1}) found only in the forested creeks.

Table 3-5: Results of SIMPER analysis of land use differences in macrobenthic community structure in GoM creeks.

	Abundance in			% Contribution to Difference		
Taxon	For	Sub	Urb	For vs. Sub	For vs. Urb	Sub vs. Urb
Intertidal						
Paratanais sp.	1.96	8.7	0	11.18	--	12.98
Corophium simile	3.42	9.67	0	9.48	10.48	14.43
Hargeria rapax	0	5.41	0	8.7	--	8.07
Grandidierella sp.	2.7	4.58	0	8.3	5.39	11.3
Heteromastus filiformis	0	4.6	0	7.41	--	6.87
Pelecypoda	1.62	5.77	0	6.96	--	8.61
Capitella capitata	0	4.32	3.25	6.95	7.14	--
Amphicteis gunneri	2.92	6.82	1.62	6.76	6.48	7.93
Ceratocephale oculata	3.71	7.4	6.18	6.59	10.55	--
Eteone heteropoda	1.62	5.51	0	6.54	--	8.22
Mediomastus sp.	6.46	5	0	--	16.44	7.47
Mediomastus californiensis	2.8		0	--	8.59	--
Streblospio benedicti	6.52		3.25	--	8.13	--
Ceratonereis longicirrata	2.41		3.58	--	6.77	--
Tubificoides sp.	4.85		5.69	--	5.77	--
Subtidal						
Eteone heteropoda	0	4.08	0	12.49	--	11.17
Ampelisca macrocephala	1.05	4.76	0	11.66	--	13.05
Amphicteis gunneri	1.9	4.76	1.37	9.31	7.41	9.42
Corophium simile	0	2.74	0	8.4	--	7.51
Streblospio benedicti	4.78	7.44	5.95	8.21	--	4.12
Capitella capitata	2.95	5.32	0	7.01	12.54	14.57
Cossura soyeri	2.41	0	1.05	6.71	9.17	--
Gastropoda	1.97	2.74	0	5.49	6.99	--
Leitoscoloplos fragilis	1.05	5.24	0	5.46	--	7.51
Heteromastus filiformis	4	0	2.24	5.45	12.21	8.04
Ceratocephale oculata	1.56	2.74	3.66	--	14.53	9.95
Nemertea	4.87		3.07	--	7.76	--
Pelecypoda	1.9		0	--	6.72	--
Ericthonius brasiliensis	0		1.37	--	5.36	--
Mediomastus sp.	6.26	7.83	5.2	--	5.04	7.16

Using a main effects two-way ANOVA, average nekton densities and numbers of taxa were significantly different in the GoM compared to the SE with the GoM region having lower values for both measures (Figure 3-18, Appendix C). Neither measure varied significantly with land use class (Figure 3-18). Because nekton data were unavailable for suburban creeks in the GoM, a full interaction ANOVA was not possible; however, a second analysis was performed in which SE suburban creeks were excluded and a full two-way ANOVA was performed with only forested and urban creeks. This analysis confirmed the results of the main effects model tested above. SE creeks supported higher densities and number of taxa than did GoM creeks, and neither measure varied significantly with land use (Appendix C).

Nekton density was negatively although not significantly ($p = 0.819$) related to impervious cover in intertidal creeks across both the SE and GoM regions (Figure 3-19, Appendix D). Similarly, the relationship between number of species and impervious cover was negative and not significant in intertidal creeks (Figure 3-19, Appendix D).

When examined using multivariate analyses (nMDS), the nekton communities of intertidal GoM creeks were significantly different from those of the southeastern US (ANOSIM: $R = 1.000$, $p = 0.002$). Nekton communities within each region clearly clustered together, and the clusters representing the communities of the two regions did not overlap (Figure 3-20A). Four of the ten taxa most responsible for the dissimilarity between intertidal creeks of the GoM and SE were present in only one region (Table 3-6). The species most responsible for the difference between regions were primarily small fish such as mummichogs (*Fundulus heteroclitus* and *Fundulus*

grandis), silversides (an unidentified species of *Menidia*), mollies (*Poecilia latipinna*), and mosquitofish (*Gambusia holbrooki*) as well as shrimp in the genera *Palaemonetes* and *Penaeus (Litopanaeus, Farfantepanaeus)*.

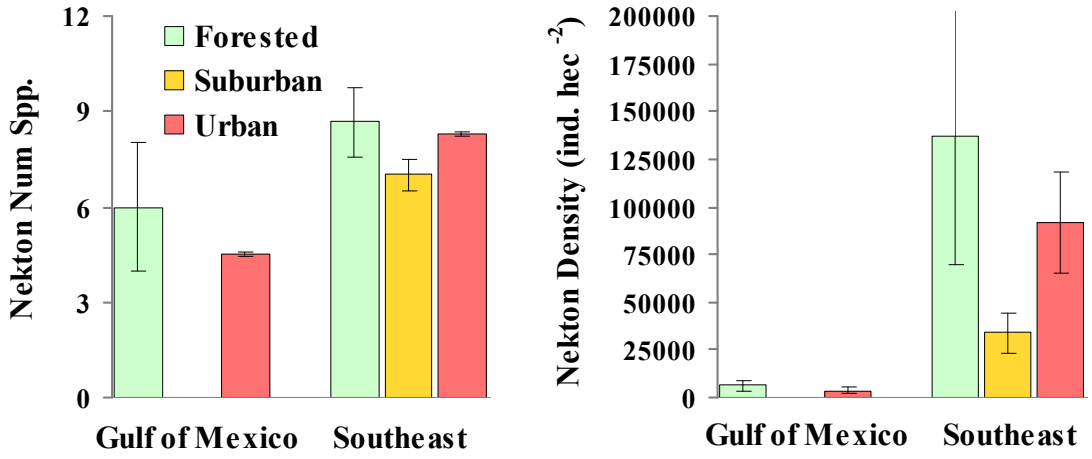

Figure 3-18. Nekton community number of species and total abundance by land use and region for only the intertidal systems. Bars represent average concentrations. Error bars are +/- 1 standard error.

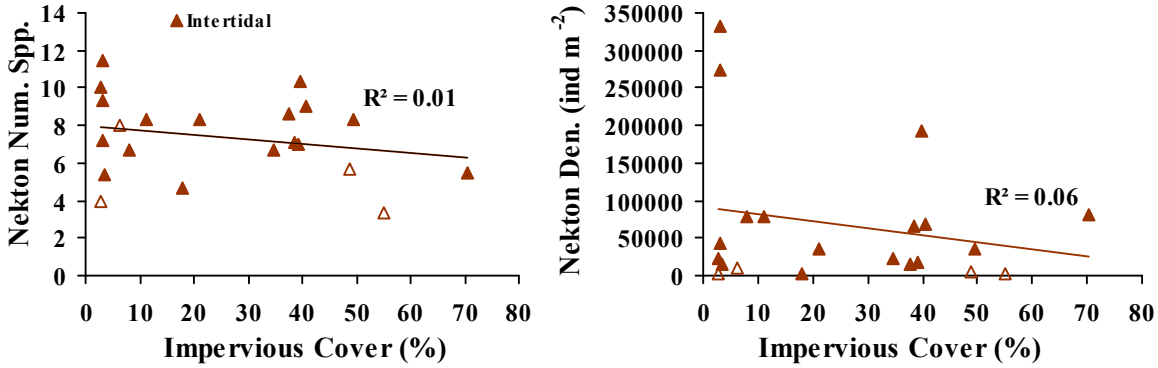

Figure 3-19. Relationship between nekton number of species and density versus impervious cover for the study watersheds. Model R^2 is show for each regression with asterisk (*) indicating significance ($p<0.05$). Open markers represent GoM sites.

Due to the dissimilarity of the nekton communities between the SE and GoM, the impact of land use class was analyzed for each region separately. Nekton community composition did not vary significantly with land use class for intertidal creeks in the SE (ANOSIM: R = 0.027, p = 0.339; Figure 3-20B). In the SE, nekton communities did not differ significantly between forested and urban (R = -0.06, p =0.706) or urban and suburban (R = -0.082, p = 0.755) creeks and only differed marginally significantly between forested and suburban creeks (R = 0.18, p = 0.08). Due to the small sample size, the GoM creek nekton communities could not be analyzed alone, but the forested and urban creeks communities clearly separated on the MDS ordination (Figure 3-20C). In the SE, the differences between creeks representing forested, suburban, and urban land use types were primarily due to differences in the abundance of various fish species (Table

3-7). In the GoM region, the differences between forested and urban creeks were also dominated by differences in fish species, but decapod crustaceans (*Panaeus setiferus* and *Uca minax*) also contributed (Table 3-7).

Table 3-6. Results of SIMPER analysis for nekton community for the SE and GoM tidal creeks in the intertidal and subtidal systems.

Taxon	Average Abundance in Southeast	Average Abundance in Gulf of Mexico	% Contribution to Difference
Palaemonetes sp.	9.99	1.25	9.09
Fundulus heteroclitus	8.2	0	8.75
Panaeus sp.	8.5	2.93	6.15
Panaeus setiferus	0	4.46	4.71
Menidia sp.	0.23	3.99	4.1
Poecilia latipinna	4.93	4.3	3.86
Gambusia holbrooki	3.97	2.98	3.62
Bairdiella chrysoura	3.4	0	3.41
Lagondon rhomboids	1.73	3.58	3.22
Fundulus grandis	0	2.95	3.2

Table 3-7. Results of SIMPER analysis of land use differences in nekton community structure in southeast and GoM intertidal creeks.

Taxon	Abundance in For	Abundance in Sub	Abundance in Urb	% Contribution to Difference For vs. Sub	% Contribution to Difference For vs. Urb	% Contribution to Difference Sub vs. Urb
Southeast						
Poecilia latipinna	2.98	5.65	5.87	6.05	5.94	4.81
Bairdiella chrysoura	4.7	2.01	4.07	5.36	4.79	5.87
Gambusia holbrooki	2.45	4.36	4.95	5.23	4.89	5.07
Gerreidae	4.04	0.54	2.44	5.17	4.72	3.85
Leiostomus xanthurus	3.28	4.88	5.82	5.16	4.08	3.96
Mugil cephalus	1.31	3.42	4.66	4.55	5.18	4.64
Cynoscion nebulosus	9.58	7.46	2.15	4.17	3.59	--
Brevoortia tyrannus	2.5	0.88	0.92	3.46	3.44	--
Fundulus majalis	2.68	0	1.38	3.43	--	--
Symphurus plagiusa	1.66	2.31	0.76	3.31	--	--
Cyprinodon variegatus	0	1.4	2.75	--	--	4.35
Fundulus luciae	1.04	1.58	2.84	--	3.41	4.27
Penaeus sp.	9.58	7.46	8.87	--	--	4.24
Lagodon rhomboides	1.5	1.43	2.36	--	3.4	4.11
Gulf of Mexico						
Gambusia holbrooki	0	--	5.97	--	7.97	--
Penaeus setiferus	7.27	--	1.66	--	7.92	--
Poecilia latipinna	1.49	--	7.12	--	7.87	--
Anchoa sp.	6.13	--	0	--	7.74	--
Fundulus grandis	0	--	5.9	--	7.69	--
Menidia sp.	1.74	--	6.24	--	6.45	--
Cyprinodon variegatus	0	--	3.76	--	4.94	--
Uca minax	2.98	--	0	--	4.9	--
Lagodon rhomboides	5.17	--	2	--	4.43	--
Brevoortia sp.	4.24	--	0	--	4.33	--

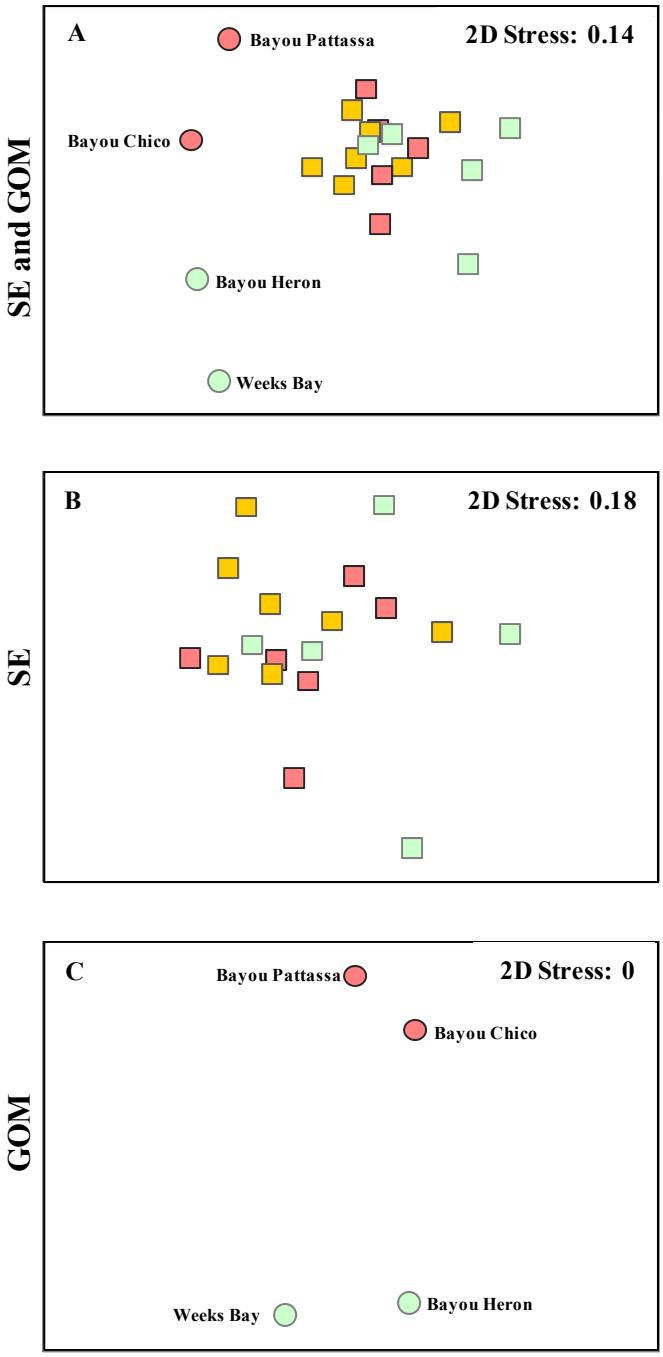

Figure 3-20. Multidimensional scaling (MDS) ordination of nekton with SE and GoM regions combined (A), just the SE region (B) and just the GOM region (C). Boxes = SE region creeks, circles = GoM region creeks. Green = forested creeks, yellow = suburban creeks and red = urban creeks.

3.4 Human Responses

3.4.1 Oyster Tissue Pathogens

Oysters were only collected in the three Mississippi creeks in the GoM in 2008: Bayou Heron and Bayou Chico (intertidal) and Bayou Pattassa (subtidal); and along the entire spatial gradient in ten tidal creeks in the SE in 2006: Sapelo Island NERR-Duplin, Sapelo Island NERR -Oakdale, Guerin, Burnett, Postell, New Market, James Island, ACE Basin NERR-Village, Hewlitts, and Whiskey. Tissues were examined for FC and ENT body burdens only.

Pathogen concentrations for FC and ENT body burdens varied over 3-4 orders of magnitude. FC ranged from 23 to 220,000 CFU 100 g^{-1} tissue weight in the SE and from 480 to 160,000 CFU 100 g^{-1} tissue weight in the GoM. ENT concentrations varied from 3,240 to 320,000 CFU 100 g^{-1} tissue weight in the SE and from 4,800 to 10,000 CFU 100 g^{-1} tissue weight in the GoM.

Because the overall sample size was small, as oysters were only collected from a subset of the sampled systems, ANOVAs were not performed on these data. Regression analysis showed that there was a significant positive relationship between watershed impervious cover and FC concentrations in oysters collected in intertidal and subtidal creeks (Figure 3-21). No significant relationship was found for ENT body burdens and impervious cover.

3.4.2 Bivalve Tissue Contaminants

Bivalve tissues were collected from the 11 SE tidal creek systems sampled in 2006 and 3 GoM tidal creek systems sampled in 2008. In addition, mussel tissue was collected from 3 GoM tidal creek systems sampled in 2008. Tissues were analyzed for lipids and contaminants including PAHs, PCBs, PBDEs, pesticides, and metals. Statistical analyses were not performed on these data due to the high number of non-detects and small sample size. Oyster and mussel tissues are summarized as a group.

Oyster and mussel tissue lipid concentrations ranged from 5.7% to 38% (average = 21.4%) in the SE and 4.7% to 11.2% in the GoM (average = 7.4%).

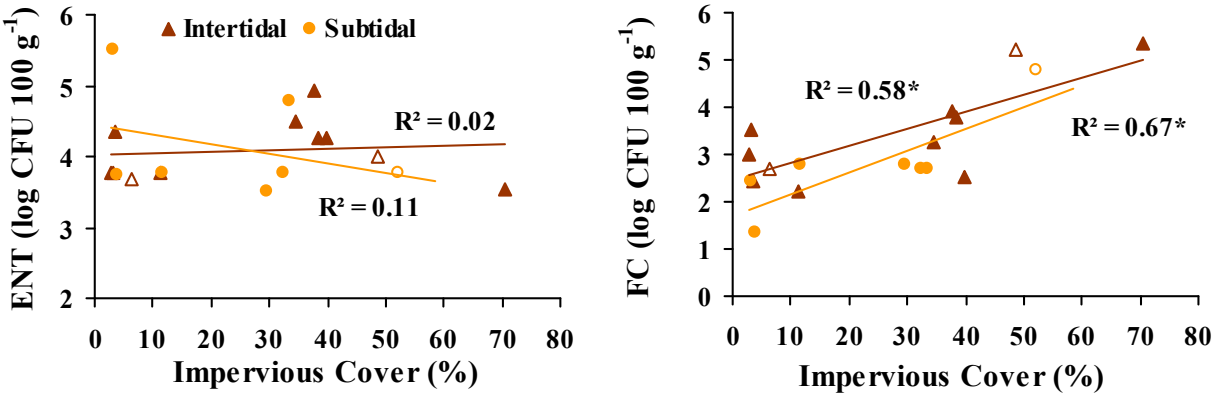

Figure 3-21. Relationship between pathogen indicator levels in oyster tissues and impervious cover for the study watersheds. Impervious cover was regressed against Enterococcus (ENT; upper) and fecal coliform (FC; lower). Model R^2 is shown for each regression with asterisk (*) indicating significance ($p < 0.05$). Log transformation is x + 1 for ENT. Open markers represent GoM sites.

Total PAH tissue concentrations ranged from 0 to 2,161 ng g^{-1} dry weight (average = 297) in the SE and 28.4 to 1,947 ng g^{-1} dry weight (average = 648) in GoM (Figure 3-22). In the SE, PAHs detected included acenaphthylene, anthracene, benzo(a)anthracene, benzo(b)fluoranthene, benzo(g,h,i)perylene, fluoranthene, and pyrene. All except acenaphthylene and naphthalene are high molecular weight PAHs typical of pyrogenic sources, and were predominantly found in developed systems. The GoM oyster and mussel tissues had detectable PAH concentrations for all but one of the analytes measured (dibenz(a,h)anthracene). Thirteen of the 24 detectable analytes were found in all six samples. Urban creeks in both the GoM and SE consistently had higher PAH concentrations particularly higher molecular weight PAHs from pyrogenic sources (Figure 3-22).

Total PCB concentrations ranged from 0 to 244 ng g^{-1} dry weight (average = 21.2) in the SE and 10.6 to 105.5 ng g^{-1} dry weight (average = 46.3) in the GoM (Figure 3-22). In the SE, PCB concentrations above the detection limit were found in the two urban intertidal creeks and in one urban and one suburban subtidal creek. In the GoM, all six samples collected from the forested, suburban and urban creeks had detectable concentrations of PCBs. The two urban GoM creek samples contained lower concentrations of PCBs than the samples from Burnett Creek (a superfund site in GA for PCB contamination) but were almost double the values observed in oysters from the next highest site in the SE, New Market Creek (urban, SC).

PBDEs, flame retardants, were detected in oyster tissues at New Market (SE, intertidal, urban), Guerin (SE, intertidal, forested), Bayou Chico (GoM, intertidal, urban), and Bayou Pattassa (GoM, subtidal, urban). Total PBDE concentrations ranged from 4.04 to 37.9 ng g^{-1} dry weight. PBDE 47 was detected in the three urban systems, PBDE 99 was detected in Bayou Chico and Guerin Creek, and PBDE 100 and 17 were only detected in Bayou Chico. The three urban creeks had detectable PBDE levels in the sediments; however, detectable levels were not observed in the forested creek sediments.

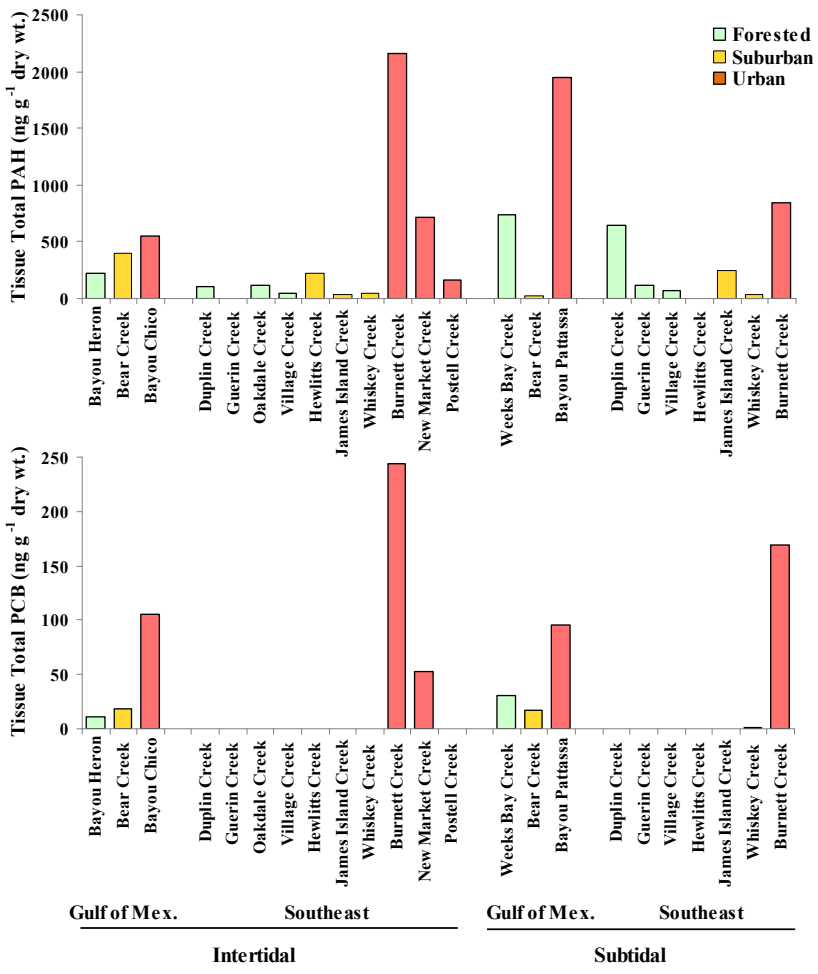

Figure 3-22. Concentrations of total PAHs and total PCBs in creek sediments for the intertidal and subtidal systems. Land class is marked by color.

Pesticide concentrations in oyster tissues were dominated by DDT and its derivatives. Total DDT concentrations ranged from 2.53 to 20.84 ng g^{-1} dry weight (average = 7.31) in the SE and 3.86 to 119.9 ng g^{-1} dry weight (average = 34.7) in the GoM. In general, total DDT tissue concentrations were higher in the developed systems compared to the forested systems in the SE. This trend was similar in MS but not in AL. The highest concentration observed was for the Weeks Bay subtidal mussels. In addition, Bear Creek, a low density suburban system, had the next highest concentrations in the intertidal and subtidal areas. Detectable pesticide contaminants included mirex, endosulfan I and s, dieldrin, chlorpyrifos, and trans-nonachlor. Dieldrin was detected in all six tissue samples in the GoM and in only one tissue sample in the SE.

Concentration data for metals (Cd, Cr, Cu, Pb, Hg) that are often elevated as a result of anthropogenic uses plus arsenic (As) are discussed below. Lead concentrations in bivalve tissues were similar across the two regions and three land use classes and were generally low (< 0.8 µg g^{-1} dry weight) except for three heavily urbanized creeks: New Market (SE, 1.58 µg g^{-1} dry weight), Bayou Chico (GoM, 1.84 µg g^{-1} dry weight) and Bayou Pattassa (GoM, 1.64 µg g^{-1} dry

weight). Mercury concentrations were similar across regions and land use classes and were generally low (< 0.19 μg g^{-1} dry weight) except for Burnett (SE, urban) in both the intertidal (0.35 μg g^{-1} dry weight) and subtidal (0.42 μg g^{-1} dry weight) creek segments. The highest concentrations of arsenic (ranged from 14.4 to 33.4 μg g^{-1} dry weight) were found in the North Carolina creeks, similar to the fish tissue contamination findings of Cooksey et al. (2008). In general, the SC and GA creeks were similar to the As concentrations in the GoM creeks with concentrations <14.3 μg g^{-1} dry weight. The cadmium, copper, and chromium concentrations were approximately double in the SE creeks compared to the GoM creeks. These values were also generally higher in the forested and suburban creeks compared to the urban creeks in the SE but this pattern was not apparent in the GoM creeks. In particular, Guerin, a forested creek in the Francis Marion National Forest, had some of the highest concentrations of metals.

In addition, the bivalve tissue chemical concentrations were compared to Food and Drug Administration (FDA; FDA 2011) environmental chemical contaminant action levels and the USEPA (2000) human-health consumption limits for cancer and non-cancer endpoints (Table 3-8). The FDA action levels are simply threshold values for comparison against tissue concentrations (non-consumption based). None of the concentrations observed in oyster tissue on a wet weight basis exceeded the molluscan bivalve or fish action levels for As, Cd, Cr, Pb, Ni,

Table 3-8. Contaminant wet weight concentrations in tidal creek bivalves (oyster for SE and MS and mussel for AL). Concentrations given in ng g^{-1} wet weight for PAH, PCB and DDT, and in μg g^{-1} wet weight for metals (As and Hg). Inorganic Arsenic calculated as 2% of total arsenic. Methyl mercury calculated as 100% of total mercury. Italicized numbers indicate concentrations that exceeded the EPA cancer-endpoint consumption limits for fish, based on four 8-ounce meals per month. Bold and italicized numbers indicate concentrations that exceed non-cancer endpoints. No measurements were found to exceed FDA levels of concern for shellfish.

Creek Name	Station Type	Hg	Total As	Estimated Inorganic As	Total DDT	Total PAH	Total PCB
Weeks Bay Creek	Subtidal	0.012	1.24	*0.02474*	15.3	*94.0*	3.9
Bayou Heron	Intertidal	0.012	0.79	*0.01588*	0.5	*27.2*	1.3
Bear Creek	Intertidal	0.022	1.24	*0.02473*	4.1	*46.2*	2.1
Bear Creek	Subtidal	0.008	1.03	*0.02055*	1.7	*2.5*	1.5
Bayou Chico	Intertidal	0.004	0.51	*0.01012*	1.1	*35.8*	*6.9*
Bayou Pattassa	Subtidal	0.012	0.58	*0.01162*	0.9	*138.7*	*6.5*
FDA		1		86	5000		
EPA noncancer		>0.12-0.23		>0.35-0.7	>59-120		>23-47
EPA cancer				>0.0078-0.016	>35-69	>1.6-3.2	>5.9-12

methyl mercury, PCBs, DDT, heptachlor epoxide, or mirex. The USEPA values are based on a consumption rate of four 8-ounce meals of fish per month for an adult population, and that comparing oyster tissue to fish tissue values represents a level of potential risk. It should also be noted that for many of the systems studied, shellfish grounds are closed for harvest, thus reducing risks associated with human consumption. Inorganic arsenic (estimated as 2% of total arsenic [USEPA 2004]) values exceeded the lower cancer endpoint for all sites sampled for

oyster tissue in the SE and GoM. Total PCB values exceeded the lower EPA cancer endpoint at two sites (Bayou Chico, intertidal, urban; Bayou Pattassa, subtidal, urban). This is in comparison to only one creek, Burnett Creek (Sanger et al. 2008), exceeded the upper cancer endpoint at both the intertidal and subtidal sites with the interidal site also exceeded the lower non-cancer endpoint. Burnett is an urban creek that was listed and cleaned up as a Superfund site for PCB and Hg contamination. Total PAH values exceeded the lower and sometimes upper cancer endpoint at all SE and GoM sites except for Guerin (interidal, forested) and Hewlitts (subtidal, suburban) both of which are in the SE (Sanger et al. 2008). No other published EPA values were exceeded.

4. Discussion

Tidal creeks have been used as a sentinel habitat to provide early warning and assess the impacts of pollution from coastal development on estuarine areas in the southeastern US (e.g., Mallin et al. 2000a, Holland et al. 2004, Van Dolah et al. 2007, Sanger et al. 2008) and Tampa Bay, FL (Greenwood et al. 2008a, Greenwood et al. 2008b, Sherwood et al. 2008). Most of the previous studies have shown that the amount of impervious cover (e.g., roads, parking lots and other paved areas, roofs) is a reliable indicator of the degree of coastal watershed development and the amount of pollution released into tidal creeks. Changes in the volume and rate (or flashiness) of stormwater runoff associated with increases in watershed imperviousness (e.g., percent impervious cover) is the major mechanism through which coastal development impairs water quality, public health, and the quality of coastal living (Holland et al. 2004, Holland and Sanger 2008). Impervious surfaces impede rain from penetrating the soils and returning to groundwater. Coastal development also removes natural forests decreasing the amount of rain that is returned to the atmosphere through evapotranspiration. In forested settings, only about 10 to 20 percent of the rain that falls on a coastal watershed is transferred to tidal creeks as runoff. In suburban and urban areas, 15 to 75 percent of rainfall is discharged to tidal creeks as runoff. When the amount of impervious cover in a tidal creek watershed exceeds 10 to 20 percent of the area, measurable increases in the volume and rate of freshwater runoff generally occur. Increased runoff contains non-point source pollution which impairs water quality, introduces excessive amounts of anthropogenic chemicals into sediments, and increases the amount of pathogens in water and shellfish. Similar relationships have been observed between the degree of watershed imperviousness and the quality of freshwater stream habitats, particularly intermittent and lower order streams (e.g., Schueler 1994, Yoder and Rankin 1999, Davies et al. 2010, Coleman et al. 2011).

4.1 Regional Comparisons

The Gulf of Mexico (GoM) and the southeastern (SE) US coasts share features of their coastlines which include abundant and productive tidal creek and salt marsh habitats. These habitats are important nursery and feeding areas for numerous recreational, commercial, and ecological resources including birds, finfish, crustaceans, and their food sources. Both the GoM and SE are characterized by low topographic relief. The most significant difference between these two regions is the magnitude of tidal forcing. Southeast tidal creeks are characterized by semi-diurnal tides that have a tidal range of 0.5 m to 2 m from North Carolina to Georgia. GoM tidal creeks experience diurnal tides that have an average tidal range of about 0.5 m in the Alabama and Mississippi focus areas of this study. Tidal range in GoM marshes and tidal creeks is more strongly influenced by wind than in the SE. Differences in tidal range influence the amount of intertidal habitat that occurs within a creek. Tidal creeks in areas with larger tidal range have a greater area of intertidal habitat and broader, deeper headwaters. Changes in the amount of intertidal habitat influence the capacity of headwaters to dilute land-based stormwater runoff without impairment of ecosystem services. In addition, the flushing rate and residence time of pollutants may also be different in the GoM versus the SE.

The GoM and SE also differ in the overall environmental cond ition of their estuarine and coastal resources as defined by the US EPA Coastal Condition Report III (USEPA 2008). In this report, the GoM had an overall rating of fair to poor with a score of 2.2 in comparison to the SE overall

rating of fair with a score of 3.6. The water quality ratings of these two regions were similar but sediment quality, benthic condition, and cumulative coastal habitat condition are higher in the SE. This finding suggests that the estuarine systems in the SE generally experience less impairment from chemical contaminants associated with sediments than similar habitats in the GoM. Tidal creek habitats were not sampled for the Coastal Condition Report because they were too shallow during low tides for the methods used. These shallow tidal creek habitats represent from about 30-60% of area of SE estuaries (Nummedal et al. 1977). The findings of this report suggest the ecological condition of tidal creeks in the SE generally experienced less impairment than similar tidal creek habitat in the GoM which is supported by the findings of the Coastal Condition Report.

Potential regional differences were examined and accounted for in the statistical analyses (ANOVAs) by using least squares means for comparisons in order to determine whether GoM tidal creeks can be used as sentinel habitats with similar patterns for the classification scheme and conceptual model. A variety of stressors were evaluated for this study including the amount of impervious cover, population density, and land cover characteristics. In general, tidal creek watershed sizes were within the same range for the two regions. Land cover varied between the GoM and SE primarily in the composition of salt marsh (less), palustrine (more), and forested (less) land covers in the GoM compared to the SE. In addition, the GoM urban creeks sampled had higher proportions of developed (low and high) land compared to the SE creeks. The range in impervious cover was similar, but only one urban subtidal watershed was sampled in the SE.

In general, the physical-chemical responses of GoM creeks to coastal development were similar to that found for the SE (e.g., increased variability in salinity, increased abundances of pathogen indicators, and increased amounts of chemically contaminated sediments). As was the case in the SE, we hypothesize that these differences are due to increases in the rate and volume of stormwater runoff that occur as coastal watersheds are developed. These changes in hydrology are mainly due to increases in the amount of impervious cover and decreases in the abundance of vegetation that are associated with development. A few interesting differences in physical-chemical responses were, however, observed between the GoM and SE. The average salinity was lower in the GoM compared to the SE (probably an artifact of creeks sampled), and the range in DO and pH was higher in the GoM compared to the SE. This probably resulted because the GoM creeks were generally more eutrophic than the SE creeks which is in agreement with the Coastal Condition Report (USEPA 2008). The significantly higher amounts of suspended sediment in the SE compared to the GoM may be related to the larger areas of salt marsh that were associated with SE creeks. Nutrient concentration patterns were generally similar, however, nitrate/nitrite levels in the GoM urban intertidal systems were extremely high compared to similar systems in the SE or to GoM suburban and forested systems. Orthophosphate and total phosphorus concentrations were also exceptionally high in the intertidal area of Bayou Pattassa. High nitrate/nitrite and orthophosphate concentrations in tidal creeks are indicative of commercial fertilizer input from adjacent upland sources. The two GoM urban creeks sampled were in downtown Pascagoula, MS with limited fringing marsh areas and abundant lawns and bulkheaded suburban properties adjacent to creek banks.

Enterococcus and F- coliphage, two of the four pathogen indicators, were higher in the SE compared to GoM. We are unsure of the mechanism causing these differences. GoM sediments

were more contaminated with pesticides than SE sediments. In addition, higher levels of chemical contamination of sediments were generally found further downstream in subtidal portions of GoM creeks. This was likely because the tidal range in GoM creeks was small compared to the SE creek resulting in substantially less intertidal area. As a result, it appears that sediment deposition areas occur in the subtidal portion of GoM creeks and in intertidal portions of SE creeks.

Comparison of biological response between the GoM and SE was focused on a limited number of metrics for infaunal benthic and shallow water nekton communities. Macrobenthic community total abundance and number of species were similar between the GoM and SE in both the intertidal and subtidal areas. Only the intertidal areas were sampled for the nekton community. Nekton total abundance and number of species were significantly higher in the SE compared to the GoM. Because the amount of intertidal habitat is limited in the GoM compared to the SE, the subtidal regions of tidal creeks may represent the most expansive and functional nursery habitats in the GoM.

The societal responses measured in the SE and GoM were focused on oyster tissue contamination with pathogens and chemicals. Oyster tissue fecal indicator concentrations were generally similar in the GoM and SE. Organic chemical (PCBs, DDT, PAHs) shellfish tissue levels tended to be higher in the GoM than in the SE; however, lipid concentrations in the SE were higher than in the GoM. All five GoM sites where shellfish tissues were collected exceeded the EPA cancer endpoint for total PAHs and inorganic arsenic, and two of the sites for total PCBs. Similar patterns were found by the NOAA Mussel Watch Program when the data for NC, SC, GA, MS, and AL were compared (Kimbrough et al. 2008).

4.2 Classification Framework

Classification frameworks have been developed and extensively used to integrate the ecological attributes of freshwater ecosystems in the context of their biogeography, hydrology, and short- and long-term ecological history (e.g., Horton 1945, Frissell et al. 1986). These classification systems assist in understanding the complexity and natural variability associated with freshwater ecosystems. In estuarine ecosystems, classification approaches have made only a limited contribution to understanding the complexity and explaining spatial and temporal variability (e.g., Anderson et al. 1976, Odum 1984). The reasons estuarine ecologists have not embraced classification as a means of partitioning and understanding complexity may include: (1) standardized approaches for resolving scale, spatial, temporal, and location differences within and among habitats and estuaries have not been broadly applied; (2) natural factors vary on multiple temporal and spatial scales (e.g., tidal, diel, extreme events, seasonal, year-to-year, climatic, geological); and (3) much of the ecosystem and habitat scale science for estuaries is based on indirect evidence from relatively few places.

The tidal creek classification system we developed in the SE and applied to the GoM facilitated the integration of the findings and enhanced the understanding of the complexity and variability in tidal creek habitats. Further study should, however, be undertaken to fully explain the differences observed in the characteristics (e.g., length, depth, area) of creek orders between the GoM and SE. In the SE, the intertidal portion was a repository for many of the pollutants discharged with stormwater runoff to estuarine systems as evidenced by the higher pollutant

concentrations in the intertidal regions. This resulted in intertidal or first order tidal creeks having substantially stronger relationships between coastal development patterns and environmental quality than subtidal creek orders. This pattern is not as clearly defined in the GoM. The land use relationship in the intertidal and subtidal systems in the GoM was not analyzed separately due to the low number of creeks sampled (n=5); however, very few interaction terms were significant. This either indicates that even though there was a difference between the two regions regarding the creek classification (i.e., intertidal and subtidal) the overall relationship with land use patterns were similar or that our sample size was too small to detect significant interactions.

The reason for the potential difference in the location or place where land-derived pollutants accumulated in tidal creeks may be related to the substantially reduced tidal flushing that occurs in the GoM compared to the SE tidal creeks. GoM tidal creeks are characterized by small tides (~0.5 m) which occur once a day and are strongly influenced by rain and wind. Tides in the SE are diel, much larger (1-2 m), and less influenced by rain events and wind. In the SE, stormwater runoff and pollutants appear to enter the system and dissipate in the shallow intertidal or headwater areas before being transported downstream to the deeper systems. As a result, the headwaters of SE tidal creeks appear to function as a repository for sediments and the pollutants associated with them. This suggests the differences between intertidal and subtidal areas may not be as clearly defined in the GoM as in the SE.

4.3 Responses to Coastal Development

In general, the relationships between indicators of tidal creek environmental quality and the degree of coastal development in the associated watershed as indicated by the amount of impervious cover were similar in the GoM and SE. This was consistent with previous research (e.g., Mallin et al. 2000a, Lerberg et al. 2000, Holland et al. 2004) that has found relationships with a wide range of exposure (physical-chemical) indicators including salinity range, sediment contamination, nutrient enrichment, and water borne pathogens and the imperviousness of the drainage area or watershed.

The salinity range (one of the significant SE indicators of runoff rate and volume or flashiness) showed a similar relationship with significantly higher salinity ranges for the urban and suburban creeks compared to the forested creeks. A significant relationship was also observed between the salinity range and impervious cover for both the intertidal and subtidal sections. This indicates that the runoff entering both GoM and SE tidal creeks in developed watersheds is flashier (increased volume and rate of runoff) than in forested watersheds. Coleman et al. (2011) found similar relationships between urbanization and hydrological changes.

Our previous research in the SE did not find average DO concentration to be a significant indicator of coastal developmental effects; however, with the addition of the GoM data the average DO concentration was found to be significantly lower in the urban creeks compared to the forested systems. This may indicate that DO concentration may be an additional indicator of land use impacts in the GoM if confirmed with further testing. The only nutrient and phytoplankton indicator in the previous SE research that was found to increase with increasing levels of development was nitrate/nitrite, a nutrient indicative of fertilizer input. This parameter was also found to be a potential indicator for the combined GoM and SE datasets. Several other

nutrients were also found to be significantly different in ANOVA analyses but not with the regression analysis indicating there was not a clear pattern with increasing development. In combination, these findings agree with other research in microtidal and macrotidal areas that indicate microtidal systems are generally more eutrophic than macrotidal systems (Monbet 1992).

The waterborne pathogen indicators found to be important in the SE appear to be important in the GoM; however, laboratory detection limits were a concern. Water column fecal coliform concentrations in GoM creeks were estimated at several sites using oyster tissue concentrations (R^2=0.58). Despite the detection limit concerns, it is clear that fecal coliform, enterococcus, F+, coliphages, and F-coliphages are all potential indicators of the levels of development in both the SE and GoM tidal creeks. This is in agreement with other research in SE tidal creeks which has also found strong relationships between pathogen indicators and the level of urbanization (Mallin et al. 2000a, Line et al. 2008). Stewart et al. (2008) highlighted the use of sentinel habitats for evaluating pathogen levels to provide early warning of ecosystem health impacts.

In general, the sediment contaminant indicators (total mERMQ, PAHs mERMQ, PCBs mERMQ, and pesticides mERMQ) were found to be good indicators of the level of development in the watershed with the forested and suburban systems having similar concentrations and the urban systems being significantly higher. This was similar to what was observed with just SE research (Sanger et al. 1999a, 1999b, Sanger et al. 2008) with the highest sediment chemical contaminant concentrations in the heavily developed areas and the low to moderately developed areas not showing significant increases in chemical contaminant levels.

The biological response with urbanization was not as clear in the GoM as previously observed in the SE (Lerberg et al. 2000, Holland et al. 2004, Sanger et al. 2008). The total abundance and number of species patterns for the macrobenthic and nekton communities regarding land use were complex. In general, very few differences were found in the GoM (note the two regions were analyzed separately due to species composition). The limited sample size in the GoM probably resulted in a lack of any relationships between urbanization and the macrobenthic or nekton communities. Very few relationships have been found in other studies with regard to anthropogenic alterations and nekton communities in estuarine and coastal environments (Holland et al. 2004, Van Dolah et al. 2007, Bilcovic 2011). This is in contrast to freshwater studies which have found nekton community parameters to be indicators of urbanization (e.g., Coleman et al. 2011). One of the few relationships observed in a previous study was a negative relationship between impervious cover and abundance of *Panaeus* sp. shrimp (Holland et al. 2004); however, this relationship has only been observed in a small geographic area (SC) and with a large sample size (n>30 creeks). Further studies evaluating the macrobenthic and nekton communities in the GoM are necessary to determine if any patterns exist with regard to coastal development.

The societal responses measured in both the SE and GoM were focused on shellfish tissue contamination with pathogens and chemicals. The oyster tissue fecal coliform levels were found to increase with increasing levels of development for the intertidal and subtidal habitats. In contrast, enterococci levels in oyster tissues were not found to be related to the level of development in the surrounding watershed. The shellfish tissue chemical contaminant levels

observed in the GoM were complex. In general, the concentrations of organic contaminants were higher in the more developed tidal creeks systems; however, the differences were not as apparent as those observed in the SE.

4.3 NERRs as Regional References

Much of our understanding of the impacts of anthropogenic activities on natural environments relies heavily on the concept of reference sites or the reference condition (Stoddard et al. 2006). For stream ecosystems, for example, biological criteria used to characterize the quality of a stream segment are developed by comparison to a population of reference sites (e.g., Hughes 1995, Barbour et al. 1995). No such effort has been made to document appropriate coastal reference sites to evaluate the impacts of human activity in the coastal zone and adjacent marine and estuarine waters. This study, and the partnership with the NERRS, was undertaken in part to explore tidal creeks as sentinel habitats and to evaluate whether the NERRs network of protected coastal habitats could serve as reference sites for future research.

Over the last decade tidal creeks have been sampled in four SE NERRs (North Carolina; North Inlet-Winyah Bay, SC; ACE Basin, SC; Sapelo Island, GA) and two GoM NERRs (Grand Bay, MS; Weeks Bay, AL). For most of the indicators of tidal creek environmental quality the NERRs were found to be reasonable regional references with environmental quality of tidal creeks found to be higher within the boundaries of the NERRs than adjacent creeks draining developed watersheds. However, current and historical land use and management practices within and around the NERRs appears to impact the reserves and may diminish their capacity as regional references.

4.5 Summary

This study demonstrates the value of subdividing a creek network into orders in the SE but further research is needed before the SE tidal creek classification systems can be validated for creek networks in the GoM. The SE creek classification provided a hydrologic and biogeographic context for understanding scale and spatial patterns characteristic of SE tidal creek networks. The SE data consistently suggested that for most environmental quality and public health indicators, the signal of land use effects on tidal creek ecosystems and human risk was strongest in shallow first order creeks. The degree of impact and risk of environmental harm decreased in second and third orders. Thus, application of the classification system clearly demonstrated that failure to sample first order creeks may result in a Type II error or false negative relative to land use impact on tidal creek ecosystems. That is, a conclusion may be reached that there is no impact to the tidal creek network from stormwater runoff and the associated land-based pollution when in fact there is significant impact but it is confined to the first order or intertidal sections which serve as a repository for sequestering pollution inputs. However, in the GoM the strong difference in land use impacts between intertidally and subtidally dominated habitats is not as apparent. The flushing rate and residence times of pollutants in tidal creeks are likely important in understanding the differences in spatial distributions; however, little information exists on these for GoM or SE tidal creeks. These factors reinforce the importance of spatial scale at which research and monitoring activities occur and proximity to pollution sources are very important for identifying the impacts of land use changes on ecosystems.

Developing a conceptual model is a critical step to identifying and evaluating monitoring and management strategies including defining what parameters to measure and when and where measurements should be taken (NRC 1990). The conceptual model detailing the effects of coastal development on tidal creeks, a sentinel habitat, which was developed for the SE was found to be applicable to GoM tidal creeks (Figure 1-1). In general, the addition of the GoM data strengthened our understanding of the complexity and spatial patterns observed in the SE. The coastal development relationships were strongest for the physical-chemical exposure indicators and weaker for the biological response indicators. This hierarchy of response patterns as you move from the left side of the model to the right has also been observed in other studies (e.g., Limburg et al. 2005).

The societal response component of the conceptual model was not studied as thoroughly in the GoM compared to the SE. Further research in this area is warranted. There is an emerging consensus that patterns of coastal development are associated with evidence of increasing fecal pollution in tidal creeks, estuaries, and bathing beaches (Mallin et al. 2000a, Karn and Harada 2001, Holland et al. 2004, Mallin 2006) which was also evidenced by this study. From a human health perspective, the accumulation of pathogens in the water, sediments, and organisms may render seafood products unsafe to eat and water unsafe for body contact recreation. Flooding vulnerability, public health risk, and economic impacts are metrics just being added to the SE model which will require additional effort in the GoM to fully understand if these linkages are also applicable to the GoM. This is highlighted in two research studies from other areas that documented decreased property values from impaired water quality (Leggett and Bockstael 2000) and decreased ecosystem services values from increased urbanization (Zang et al. 2010) illustrating the continued importance for assessing these impacts.

Tidal creek networks are the primary hydrologic link between estuaries and land based activities. Therefore, tidal creeks serve as sentinel habitats that provide early warning of the ensuing harm that is likely to occur to seaward habitats and ecosystems in both the SE and GoM. As the first zone of impact for non-point source pollution runoff, the potential for alterations to freshwater inputs, increased levels of microbial and chemical contamination, and adverse effects on biological communities in tidal creek habitats is great. This is particularly evident given the high rate of coastal population growth which has the potential to lead to further impairment of natural resources as well as impairment of the free ecosystem services tidal creeks provide if current land use decisions continue. Educating decision-makers on the impact of existing land use practices is an important activity to ensure that the potential impacts of current and future land use decisions are considered.

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Appendix A

The expected range of method detection limits (MDLs) based on extracted sample mass for the sediment and oyster tissue contaminant analyses. Organics are ng g^{-1} dry weight and metals are µg g^{-1} dry weight.

Analyte	Sediment MDL Range	Tissue MDL Range
Aluminum	0.008 - 0.090	5.35 - 10
Antimony	0.786 - 0.804	0.196 - 0.368
Arsenic	0.091 - 0.933	0.023 - 0.124
Barium	0.108 - 1.1	0.026 - 0.050
Beryllium	0.159 - 0.167	0.019 - 0.037
Cadmium	0.0036 - 0.0038	0.002 - 0.004
Chromium	0.161 - 1.64	0.201 - 0.376
Cobalt	0.079 - 0.082	0.049 - 0.093
Copper	6.26 - 6.56	0.779 - 6.49
Iron	0.051 - 0.532	31.6 - 59.1
Lead	0.051 - 0.518	0.012 - 0.024
Lithium	0.652 - 0.683	0.081 - 0.152
Manganese	4.96 - 5.2	0.617 - 1.16
Mercury	0.0006 - 0.008	0.006 - 0.014
Nickel	1.1 - 1.16	0.027 - 0.051
Selenium	0.161 - 0.165	0.103 - 0.194
Silver	0.269 - 0.282	0.18 - 1.12
Thallium	0.030 - 0.032	0.019 - 0.036
Tin	0.141 - 0.144	0.035 - 0.066
Uranium	0.047 - 0.048	0.029 - 0.055
Vanadium	0.06 - 0.061	0.015 - 0.028
Zinc	16 - 16.8	99.5 - 186
2,4'-DDD	0.315 - 1.32	5.11 - 8.98
2,4'-DDE	0.0787 - 0.33	1.28 - 2.25
2,4'-DDT	0.355 - 1.49	5.76 - 10.1
4,4'-DDD	0.265 - 1.11	4.29 - 7.55
4,4'-DDE	0.263 - 1.1	4.26 - 7.49
4,4'-DDT	0.789 - 3.31	12.8 - 22.5
Aldrin	0.527 - 2.21	8.56 - 15
Hexachlorobenzene	0.214 - 0.897	3.47 - 6.1
Chlorpyrifos	0.0447 - 0.187	0.724 - 1.27
cis-Chlordane	1.39 - 5.85	22.6 - 39.8
Dieldrin	0.116 - 0.486	1.88 - 3.31
Endosulfan I	0.127 - 0.531	2.05 - 3.61
Endosulfan II	0.26 - 1.09	4.23 - 7.43
Endosulfan sulfate	0.38 - 1.59	6.16 - 10.8
Heptachlor	0.205 - 0.861	3.33 - 5.86
Heptachlor epoxide	0.585 - 2.45	9.49 - 16.7
Gamma-HCH (g-BHC, lindane)	0.087 - 0.366	1.41 - 2.49
Mirex	0.178 - 0.745	2.88 - 5.07
Trans-nonachlor	1.43 - 6	23.2 - 40.8
PBDE 100	0.124 - 0.522	2.02 - 3.55
PBDE 138	0.372 - 1.56	6.04 - 10.6

Appendix A (cont.)

Analyte	Sediment MDL Range	Tissue MDL Range
PBDE 153	0.119 - 0.5	1.93 - 3.4
PBDE 154	0.685 - 2.87	11.1 - 19.5
PBDE 17	0.106 - 0.446	1.72 - 3.03
PBDE 183	0.176 - 0.741	2.86 - 5.04
PBDE 190	0.735 - 3.08	11.9 - 21
PBDE 28	0.158 - 0.665	2.57 - 4.52
PBDE 47	0.142 - 0.598	2.31 - 4.07
PBDE 66	0.134 - 0.562	2.17 - 3.82
PBDE 71	0.14 - 0.589	2.28 - 4
PBDE 85	0.732 - 3.07	11.9 - 20.9
PBDE 99	0.134 - 0.562	2.17 - 3.82
PCB 101	0.157 - 0.66	2.55 - 4.49
PCB 103	0.148 - 0.62	2.4 - 4.22
PCB 104	0.128 - 0.535	2.07 - 3.64
PCB 105	0.14 - 0.589	2.28 - 4
PCB 107/108	0.272 - 1.14	4.42 - 7.77
PCB 110	0.197 - 0.825	3.19 - 5.61
PCB 114	0.231 - 0.968	3.74 - 6.58
PCB 118	0.485 - 2.03	7.87 - 13.8
PCB 119	0.234 - 0.981	3.79 - 6.67
PCB 12	0.189 - 0.794	3.07 - 5.4
PCB 123	0.926 - 3.89	15 - 26.4
PCB 126	0.164 - 0.687	2.66 - 4.67
PCB 128	0.095 - 0.401	1.55 - 2.73
PCB 130	0.174 - 0.732	2.83 - 4.98
PCB 132/168	0.161 - 0.674	2.6 - 4.58
PCB 138	0.089 - 0.375	1.45 - 2.55
PCB 141	0.125 - 0.526	2.04 - 3.58
PCB 146	0.473 - 1.99	7.68 - 13.5
PCB 149	0.319 - 1.34	5.17 - 9.1
PCB 15	0.165 - 0.691	2.67 - 4.7
PCB 151	0.151 - 0.633	2.45 - 4.31
PCB 153	0.394 - 1.66	6.4 - 11.3
PCB 154	0.14 - 0.589	2.28 - 4
PCB 156	0.12 - 0.504	1.95 - 3.43
PCB 157	0.11 - 0.459	1.78 - 3.13
PCB 158	0.183 - 0.767	2.97 - 5.22
PCB 159	0.090 - 0.379	1.47 - 2.58
PCB 169	0.119 - 0.5	1.93 - 3.4
PCB 170	0.14 - 0.589	2.28 - 4
PCB 172	0.13 - 0.544	2.1 - 3.7
PCB 174	0.154 - 0.647	2.5 - 4.4
PCB 177	0.169 - 0.709	2.74 - 4.82
PCB 18	0.422 - 1.77	6.85 - 12
PCB 180	0.115 - 0.482	1.86 - 3.28
PCB 183	0.149 - 0.625	2.41 - 4.25
PCB 184	0.102 - 0.428	1.66 - 2.91

Appendix A (cont.)

Analyte	Sediment MDL Range	Tissue MDL Range
PCB 187	0.086 - 0.361	1.4 - 2.46
PCB 188	0.092 - 0.388	1.5 - 2.64
PCB 189	0.129 - 0.54	2.09 - 3.67
PCB 193	0.191 - 0.803	3.1 - 5.46
PCB 194	0.087 - 0.366	1.41 - 2.49
PCB 195	0.254 - 1.07	4.12 - 7.25
PCB 198	0.173 - 0.727	2.81 - 4.95
PCB 2	0.146 - 0.611	2.36 - 4.16
PCB 20	0.134 - 0.562	2.17 - 3.82
PCB 201	0.117 - 0.491	1.9 - 3.34
PCB 202	0.115 - 0.482	1.86 - 3.28
PCB 206	0.103 - 0.433	1.67 - 2.94
PCB 207	0.296 - 1.24	4.79 - 8.43
PCB 209	0.116 - 0.486	1.88 - 3.31
PCB 26	0.125 - 0.526	2.04 - 3.58
PCB 28	0.383 - 1.61	6.21 - 10.9
PCB 29	0.224 - 0.941	3.64 - 6.4
PCB 3	0.098 - 0.415	1.6 - 2.82
PCB 31	0.325 - 1.37	5.28 - 9.28
PCB 37	0.184 - 0.772	2.98 - 5.25
PCB 44	0.116 - 0.486	1.88 - 3.31
PCB 45	0.231 - 0.968	3.74 - 6.58
PCB 48	0.107 - 0.451	1.74 - 3.06
PCB 50	0.21 - 0.883	3.42 - 6.01
PCB 52	0.146 - 0.611	2.36 - 4.16
PCB 56/60	0.216 - 0.906	3.5 - 6.16
PCB 61/74	0.285 - 1.2	4.62 - 8.13
PCB 63	0.218 - 0.915	3.54 - 6.22
PCB 66	0.215 - 0.901	3.48 - 6.13
PCB 69	0.372 - 1.56	6.04 - 10.6
PCB 70	0.44 - 1.85	7.14 - 12.6
PCB 76	0.292 - 1.23	4.74 - 8.34
PCB 77	0.156 - 0.656	2.54 - 4.46
PCB 8	0.367 - 1.54	5.95 - 10.5
PCB 81	0.188 - 0.79	3.05 - 5.37
PCB 82	0.215 - 0.901	3.48 - 6.13
PCB 84	0.342 - 1.44	5.55 - 9.77
PCB 87	0.203 - 0.852	3.29 - 5.8
PCB 88	0.219 - 0.919	3.55 - 6.25
PCB 9	0.403 - 1.69	6.54 - 11.5
PCB 92	0.134 - 0.562	2.17 - 3.82
PCB 95	0.102 - 0.428	1.66 - 2.91
PCB 99	0.182 - 0.763	2.95 - 5.19
Acenaphthene	0.048 - 21.97	117.76 - 207.16
Acenaphthylene	0.064 - 7.63	40.88 - 71.91
Anthracene	0.063 - 8.63	46.25 - 81.36
Benzo(b)fluoranthene	0.063 - 4.6	24.68 - 43.41
Benzo(a)anthracene	0.065 - 5.42	29.05 - 51.1

Appendix A (cont.)

Analyte	Sediment MDL Range	Tissue MDL Range
Benzo(a)pyrene	0.054 - 13.61	72.97 - 128.36
Benzo(e)pyrene	0.054 - 16.72	89.64 - 157.68
Benzo(g,h,i)perylene	0.051 - 9.16	49.07 - 86.32
Benzo(k+j)fluoranthene	0.133 - 11.87	63.62 - 111.91
Biphenyl	3.49 - 8.61	46.15 - 81.18
Chrysene+Triphenylene	0.047 - 14.07	75.39 - 132.63
Dibenz(a,h)anthracene	0.158 - 17.94	96.17 - 169.17
Dibenzothiophene	0.043 - 8.27	44.34 - 77.99
Fluoranthene	24.62 - 60.71	325.39 - 572.4
Fluorene	0.052 - 22.97	123.11 - 216.56
Naphthalene	0.037 - 4.64	24.88 - 43.77
1,6,7 Trimethylnaphthalene	0.123 - 32.6	174.73 - 307.36
1-Methylnaphthalene	0.036 - 17.35	93 - 163.6
2,6 Dimethylnaphthalene	0.03 - 18.7	100.23 - 176.31
2-Methylnaphthalene	0.059 - 27.06	145.02 - 255.1
Indeno(1,2,3-cd)pyrene	0.167 - 22.72	121.78 - 214.22
Perylene	2.33 - 5.75	30.84 - 54.25
Phenanthrene	4.06 - 10.02	53.7 - 94.46
1-Methylphenanthrene	0.037 - 11.12	59.59 - 104.83
Pyrene	4.62 - 11.38	61.01 - 107.32

Appendix B

The 24 analytes used to calculate the mean ERM quotient. For total PCB and DDT, refer to Appendix A for the individual analytes used.

Analyte
Arsenic
Cadmium
Chromium
Copper
Lead
Mercury
Silver
Zinc
4,4'-DDE
Total DDT
Acenaphthene
Acenaphthylene
Anthracene
Benzo(a)anthracene
Benzo(a)pyrene
Chrysene+Triphenylene
Dibenz(a,h)anthracene
Fluoranthene
Fluorene
Naphthalene
2-Methylnaphthalene
Phenanthrene
Pyrene
Total PCB

Appendix C

Results of 3-way or 2-way ANOVAs on indicator variables sampled in summer in the SE and GoM. Region factors are southeast (SE) and Gulf of Mexico (GoM). Land use class factors are Forested (F), Suburban (S), and Urban (U). Station type factors are Intertidal (I) and Subtidal (S). Post hoc multiple comparisons were performed using least squared means; model factors (arranged from low to high) with different superscripts are statistically different. $--interaction term not calculated due to lack of suburban systems in GoM for these variables.

Parameter	Model p-value	R^2	Region p-value	Land Use p-value	Station Type p-value	Inter-actions	Region LS Means	Land Use LS Means	Station Type LS Means
Water Quality Average									
DO (mg/l)	0.0062	0.39	0.0827	0.0188	0.0201	r*l	SE^a GoM^a	U^a S^{ab} F^b	I^a S^b
pH	0.1519	0.18	0.2687	0.7949	0.0397		SE^a GoM^a	S^a F^a U^a	I^a S^b
Salinity	0.0088	0.30	0.0107	0.1560	0.0172		GoM^a SE^b	S^a U^a F^a	I^a S^b
Temperature	0.1218	0.17	0.0515	0.0811	0.8867		SE^a GoM^a	U^a F^a S^a	S^a I^a
Turbidity	0.2449	0.13	0.2686	0.9477	0.0502		GoM^a SE^a	U^a S^a F^a	S^a I^a
Water Quality Range									
DO (mg/l)	0.0004	0.42	0.0214	0.3502	<0.0001		SE^a GoM^b	F^a U^a S^a	S^a I^b
pH	0.0126	0.31	0.0115	0.0391	0.0359		SE^a GoM^b	F^a U^a S^b	S^a I^b
Salinity	<0.0001	0.50	0.6097	<0.0001	0.0014		SE^a GoM^a	F^a S^b U^b	S^a I^b
Temperature	<0.0001	0.62	0.2769	0.7039	0.0007	r*l, r*s	SE^a GoM^a	S^a F^a U^a	S^a I^b
Turbidity	0.1003	0.19	0.0610	0.8871	0.0612		GoM^a SE^a	F^a U^a S^a	S^a I^a
Water Column Nutrients and Phytoplankton									
Chl-*a*	0.4105	0.10	0.9907	0.8363	0.0627		SE^a GoM^a	U^a S^a F^a	S^a I^a
Phaeophytin	0.0072	0.31	0.4946	0.9404	0.0003		GoM^a SE^a	U^a F^a S^a	S^a I^b
NH_4	0.0008	0.39	0.8866	0.0428	0.0002		SE^a GoM^a	F^a S^{ab} U^b	S^a I^b
$NO_{2/3}$	<0.0001	0.69	0.9881	<0.0001	0.0013	r*l*s	SE^a GoM^a	F^a S^b U^c	S^a I^b
TDN	0.0003	0.42	0.2080	0.1134	<0.0001		GoM^a SE^a	F^a S^{ab} U^b	S^a I^b
TN	<0.0001	0.51	0.6841	0.2319	<0.0001		GoM^a SE^a	S^a F^a U^a	S^a I^b
PO_4	0.0133	0.28	0.8036	0.0253	0.0164		GoM^a SE^a	F^a S^{ab} U^b	S^a I^b
TDP	0.0016	0.37	0.0937	0.0032	0.0307		GoM^a SE^a	F^a S^a U^b	S^a I^b
TP	0.0008	0.39	0.1619	0.0083	0.0024		GoM^a SE^a	F^a S^a U^b	S^a I^b
TSS	0.0019	0.36	0.0140	0.1833	0.0017		GoM^a SE^b	S^a F^a U^a	S^a I^b
Water Column Pathogens									
FC	<0.0001	0.74	0.7054	<0.0001	<0.0001	r*l	GoM^a SE^a	F^a S^a U^b	S^a I^b
ENT	<0.0001	0.63	<0.0001	0.0431	<0.0001		GoM^a SE^b	F^a S^{ab} U^b	S^a I^b
F+	0.0025	0.35	0.3639	0.0007	0.1833		GoM^a SE^a	F^a S^a U^b	S^a I^a
F-	<0.0001	0.61	0.0112	<0.0001	0.0002	r*l	GoM^a SE^b	S^a F^a U^b	S^a I^b

Parameter	Model p-value	R^2	Region p-value	Land Use p-value	Station Type p-value	Inter-actions	Region LS Means	Land Use LS Means	Station Type LS Means
Sediment Composition									
Sand	0.8814	0.03	0.5082	0.8440	0.5461		GoM^a SE^a	F^a S^a U^a	I^a S^a
Mud	0.8765	0.03	0.5252	0.8485	0.5194		SE^a GoM^a	U^a S^a F^a	S^a I^a
Clay	0.8315	0.04	0.4435	0.7207	0.9752		GoM^a SE^a	U^a F^a S^a	I^a S^a
Silt	<0.0001	0.72	<0.0001	0.0009	0.0005	r*l, r*s	SE^a GoM^b	S^a U^a F^b	I^a S^b
TOC	0.1409	0.17	0.9035	0.8130	0.0111		SE^a GoM^a	F^a U^a S^a	S^a I^b
Porewat. TAN	0.2680	0.13	0.3648	0.1133	0.5347		SE^a GoM^a	F^a U^{ab} S^b	S^a I^a
Sediment Contamination									
Total mERMQ	0.0071	0.35	0.4728	0.0104	0.6057	r*s	SE^a GoM^a	F^a S^a U^b	I^a S^a
mERMQ Metal	0.0545	0.25	0.4289	0.3576	0.2821	r*s	SE^a GoM^a	F^a S^a U^a	I^a S^a
mERMQ PAH	0.0005	0.41	0.8004	<.0001	0.2812		SE^a GoM^a	F^a S^a U^b	S^a I^a
mERMQ PCB	0.0111	0.29	0.9006	0.0028	0.3111		SE^a GoM^a	F^a S^a U^b	S^a I^a
mERMQ Pest.	0.0998	0.19	0.3670	0.0619	0.1806		SE^a GoM^a	F^a S^{ab} U^b	S^a I^a
Macrobenthos									
Intertidal									
Density (#/m^2)		0.30	0.5220	0.0120		r*l	GoM^a SE^a	S^a F^b U^b	
# of Species		0.27	0.2690	0.0250		r*l	SE^a GoM^a	S^a F^{ab} U^b	
Subtidal									
Density (#/m^2)		0.23	0.8820	0.5270			SE^a GoM^a	S^a F^a U^a	
# of Species		0.16	0.0860	0.4140			GoM^a SE^a	S^a F^a U^a	
Nekton									
Density (#/hec)		0.46	<0.001	0.1050		$	SE^a GoM^b	F^a U^a S^a	
# of Species		0.25	0.0090	0.2410		$	SE^a GoM^b	F^a U^a S^a	

Appendix D

Regression results for various parameters versus impervious cover for intertidal and subtidal systems separately. Bolded values are significant at $p < 0.05$.

Parameter	Trans-formation	Station Type	R^2	Model p-value	Slope	Intercept
Water Quality Average						
Depth (m)	log10	Intertidal	0.11	0.1285	-17.84	19.65
Depth (m)	log10	Subtidal	0.09	0.2049	-15.90	25.98
DO (% sat)		**Intertidal**	**0.20**	**0.0372**	**-0.85**	**75.28**
DO (% sat)		Subtidal	0.09	0.1955	-0.257	41.73
DO (mg/l)		Intertidal	0.11	0.1242	-9.20	62.45
DO (mg/l)		Subtidal	0.09	0.1944	-3.46	40.29
pH		Intertidal	0.06	0.3126	15.94	-88.82
pH		Subtidal	0.05	0.3967	13.821	-77.97
Salinity (ppt)		Intertidal	0.11	0.1304	-0.78	38.43
Salinity (ppt)		Subtidal	0.00	0.8249	0.12	20.63
Temperature (deg C)		Intertidal	0.00	0.8137	-0.65	44.44
Temperature (deg C)		Subtidal	0.00	0.8802	-0.36	34.19
Turbidity (NTU)		Intertidal	0.10	0.1541	-0.38	37.38
Turbidity (NTU)		Subtidal	0.01	0.7327	0.10	21.26
Water Quality Range						
Depth (m)	log10	Intertidal	0.05	0.2988	-20.59	26.61
Depth (m)	log10	Subtidal	0.04	0.4157	-10.90	25.18
DO (% sat)	log10	Intertidal	0.00	0.9522	1.48	22.23
DO (% sat)	log10	Subtidal	0.00	0.9520	1.61	20.51
DO (mg/l)	log10	Intertidal	0.01	0.6515	11.64	15.90
DO (mg/l)	log10	Subtidal	0.00	0.9794	0.69	22.95
pH		Intertidal	0.06	0.2806	18.84	14.25
pH		Subtidal	0.17	0.0922	47.82	0.10
Salinity (ppt)		**Intertidal**	**0.34**	**0.0041**	**1.26**	**8.43**
Salinity (ppt)		**Subtidal**	**0.36**	**0.0055**	**2.61**	**5.15**
Temperature (deg C)	log10	Intertidal	0.02	0.5390	-13.50	35.50
Temperature (deg C)	log10	Subtidal	0.09	0.1878	28.96	10.46
Turbidity (NTU)	log10	Intertidal	0.01	0.6472	-5.50	35.66
Turbidity (NTU)	log10	Subtidal	0.01	0.6562	4.84	15.23

Parameter	Trans-formation	Station Type	R^2	Model p-value	Slope	Intercept
Water Column Nutrients and Phytoplankton						
Chl *a*	log10	Intertidal	0.03	0.4513	-9.12	36.97
Chl *a*	log10	Subtidal	0.09	0.1950	-19.64	44.70
Phaeo		Intertidal	0.02	0.5475	-0.57	31.18
Phaeo		Subtidal	0.06	0.2927	-1.95	34.27
NH_4	log10	Intertidal	0.17	0.0506	13.49	41.27
NH_4	log10	Subtidal	0.09	0.2025	8.48	40.19
$NO_{2/3}$	**log10**	**Intertidal**	**0.51**	**0.0002**	**27.73**	**62.89**
$NO_{2/3}$	**log10**	**Subtidal**	**0.39**	**0.0034**	**27.15**	**69.57**
PO4		Intertidal	0.15	0.0766	46.40	17.71
PO4		Subtidal	0.18	0.0618	117.05	16.08
Si	log10	Intertidal	0.00	0.9606	-0.86	25.57
Si	log10	Subtidal	0.08	0.2141	-20.72	29.07
TDN	log10	Intertidal	0.10	0.1506	23.44	5.87
TDN	log10	Subtidal	0.00	0.7770	7.57	19.53
TDP	log10	Intertidal	0.14	0.0817	15.67	39.88
TDP	log10	Subtidal	0.17	0.0740	20.04	47.74
TN	log10	Intertidal	0.03	0.4203	28.17	22.91
TN	log10	Subtidal	0.00	0.9297	3.46	23.86
TP	log10	Intertidal	0.11	0.1323	18.01	36.31
TP	log10	Subtidal	0.14	0.1058	26.21	47.55
TSS	log10	Intertidal	0.00	0.7998	-3.50	30.75
TSS	log10	Subtidal	0.07	0.2550	-24.75	55.93
Water Column Pathogens						
FC - est./no >600	**log10**	**Intertidal**	**0.64**	**<0.0001**	**18.96**	**-37.88**
FC - est./no >600	**log10**	**Subtidal**	**0.24**	**0.0283**	**9.02**	**6.16**
ENT - no >600	**log10**	**Intertidal**	**0.33**	**0.0079**	**15.46**	**-23.85**
ENT - no >600	log10	Subtidal	0.03	0.5140	3.12	15.59
F+	**log10**	**Intertidal**	**0.40**	**0.0016**	**17.33**	**16.49**
F+	**log10**	**Subtidal**	**0.26**	**0.0219**	**19.12**	**18.52**
F-	**log10**	**Intertidal**	**0.31**	**0.0067**	**14.39**	**-0.33**
F-	**log10**	**Subtidal**	**0.27**	**0.0200**	**15.12**	**7.41**

Parameter	Trans-formation	Station Type	R^2	Model p-value	Slope	Intercept
Sediment Composition						
Sand (%)		Intertidal	0.01	0.7212	0.07	20.95
Sand (%)		Subtidal	0.07	0.2580	-0.21	37.62
Mud (%)		Intertidal	0.01	0.7268	-0.07	28.16
Mud (%)		Subtidal	0.07	0.2781	0.20	16.96
Clay (%)		Intertidal	0.00	0.8753	-0.04	26.10
Clay (%)		Subtidal	0.01	0.7199	0.07	20.94
Silt (%)		Intertidal	0.00	0.9438	0.04	24.70
Silt (%)		Subtidal	0.00	0.8484	-0.06	23.62
TOC (%)		Intertidal	0.05	0.3209	2.08	18.20
TOC (%)		Subtidal	0.17	0.0791	5.41	15.07
Porewater TAN	log10	Intertidal	0.18	0.0512	35.53	9.69
Porewater TAN	log10	Subtidal	0.09	0.1871	13.58	18.07
Sediment Contamination						
ERMQ	**log10**	**Intertidal**	**0.39**	**0.0020**	**35.98**	**74.41**
ERMQ	**log10**	**Subtidal**	**0.26**	**0.0229**	**28.96**	**65.64**
ERMQ Metals	**log10**	**Intertidal**	**0.19**	**0.0428**	**32.09**	**62.91**
ERMQ Metals	log10	Subtidal	0.11	0.1562	18.83	46.38
ERMQ PAHs	**log10**	**Intertidal**	**0.53**	**0.0001**	**32.46**	**76.98**
ERMQ PAHs	**log10**	**Subtidal**	**0.47**	**0.0008**	**34.97**	**83.32**
ERMQ PCB	**log10**	**Intertidal**	**0.21**	**0.0330**	**19.44**	**57.17**
ERMQ PCB	**log10**	**Subtidal**	**0.23**	**0.0309**	**22.97**	**64.18**
ERMQ Pesticides	**log10**	**Intertidal**	**0.47**	**0.0004**	**27.92**	**67.24**
ERMQ Pesticides	**log10**	**Subtidal**	**0.23**	**0.0343**	**20.98**	**58.82**
Oyster Tissue Pathogens						
ENT	log10	Intertidal	0.02	0.7063	7.00	-1.54
ENT	log10	Subtidal	0.11	0.4617	-8.28	57.85
FC	**log10**	**Intertidal**	**0.58**	**0.0061**	**16.41**	**-29.47**
FC	**log10**	**Subtidal**	**0.67**	**0.0234**	**14.67**	**-17.11**
Macrobenthos						
Density (ind/m^2)	1/3 root	Intertidal	0.00	0.9502	-0.0001	25.46
Density (ind/m^2)	1/3 root	Subtidal	0.08	0.2359	-0.008	29.17
Num of Species	log10	Intertidal	0.16	0.0623	-50.53	73.09
Num of Species	log10	Subtidal	0.05	0.3330	-11.87	37.82
Nekton						
Density (ind/hec)	log10	Intertidal	0.01	0.6179	-3.83	27.31
Num of Species		Intertidal	0.06	0.2869	-2.42	43.453

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