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Impacts of Coastal Development on the Ecology of Tidal Creek Ecosystems of the US Southeast Including Consequences to Humans

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Abstract Upland areas of southeastern United States tidal creek watersheds are popular locations for development, and they form part of the estuarine ecosystem characterized by high economic and ecological value. The primary objective of this work was to define the relationships between coastal development, with its concomitant land use changes and associated increases in nonpoint source pollution loading, and the ecological condition of tidal creek ecosystems including related consequences to human populations and coastal communities. Nineteen tidal creek systems, located along the southeastern US coast from southern North Carolina to southern Georgia, were sampled in the summer, 2005 and 2006. 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 then watersheds were delineated for each segment. Relationships between coastal development, concomitant land use changes, nonpoint source pollution loading, the ecological condition of tidal creek ecosystems, and the potential impacts to human populations and coastal communities were evaluated. In particular, relationships were identified between the amount of impervious cover (indicator of coastal development) and a range of exposure and response measures including increased chemical contamination of the sediments, increased pathogens in the water, increased nitrate/nitrite levels, increased salinity range, decreased biological productivity of the macrobenthos, alterations to the food web, increased flooding potential, and increased human risk of exposure to pathogens and harmful chemicals. The integrity of tidal creeks, particularly the headwaters or intertidally dominated sections, was impaired by increases in nonpoint source pollution associated with sprawling urbanization (i.e., increases in impervious cover). This finding suggests that these habitats are valuable early warning sentinels of ensuing ecological impacts and potential public health and flooding risk from sprawling coastal development. The results also validate the use of a conceptual model with impervious cover thresholds for tidal creek systems in the southeast region.

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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 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). As a result of the desire of humans to live along the coast, forested and agricultural land in coastal areas is being converted to urban development three to six times faster than the rate of population growth (Beach 2002; Allen and Lu 2003). These trends appear to be accelerating, with potentially serious impacts on coastal ecosystems and the quality of life of the people who live, work, and recreate in coastal areas (Cohen et al. 1997; Vitousek et al. 1997).

Recent reports have noted the diminished condition of coastal natural resources (e.g., USEPA 2001a; NMFS 2002; Pew Oceans Commission 2003; Millennium Ecosystem Assessment 2005a). Most of these reports conclude that nonpoint sources of pollution and the combined effects of multiple stressors are the major contributors to the diminished resources and declining condition. New approaches and collaborations are required to understand and resolve the complex, regional-scale environmental issues (Millennium Ecosystem Assessment 2005a). Existing observational systems do not provide sufficient early warning and have failed to link degraded ecosystem condition to human populations and quality of life issues. Public health, quality of life, and ecosystem science are not separate domains, but are interconnected and linked disciplines (Millennium Ecosystem Assessment 2005a, b). A paradigm of one health (human, wildlife, and ecosystem) is crucial to sustaining the critical ecological services and quality of life that currently exists in the coastal zone. For example, there is an emerging consensus that coastal development is associated with increasing fecal pollution in tidal creeks, estuaries, and bathing beaches (e.g., Mallin et al. 2000; Karn and Harada 2001; Holland et al. 2004; Mallin 2006). The accumulation of pathogens and chemicals in the water, sediments, and organisms may render seafood products unsafe to eat and water unsafe for primary contact recreation. Other consequences of sprawling coastal development to human populations include increased vulnerability of homes to flooding, potential decreases in the economic value of private property as environmental quality declines and flooding potential increases, increased public health risk from contaminated sediments and disease, and demographic and cultural changes that occur due to declining environmental quality (e.g., impacts on the Gullah-Geechee culture, creation of "brownfields," loss of subsistence fishing; Seabrook 2012).

One of the earliest symptoms of broad-scale coastal ecosystem impairment has historically been declines in the amount and condition of critical habitats that are sensitive to localized and relatively small changes in environmental conditions. Notable examples include declines in the extent and condition of sea grass beds, oyster reefs, kelp forests, coral reefs, and wetlands (e.g., Bayley et al. 1978; Dustan and Halas 1987). These "sentinel habitats" or "first responders" generally exhibit declines years to decades before system-wide impairment is documented by monitoring activities. Unfortunately, the scientific knowledge needed to understand the warning signals provided by sentinel habitats has only recently become available (e.g., Kemp et al. 1983; Hoegh-Guldberg 1999; Porter and Tougas 2001; Turgeon et al. 2002).

In estuarine environments of the southeastern USA (SE), tidal creeks may serve as a sentinel habitat for assessing the impacts from human landscape alterations in coastal areas (Holland et al. 2004; DiDonato et al. 2009). Tidal creeks and their associated salt marshes are the interface between the local landscape and estuaries where freshwater from the land mixes with saline water from the estuary. The resulting tidal creek–salt marsh networks are renowned for their dynamic nature, ecological complexity, pollutant retention and processing, nursery functions, biological productivity, and seafood production (Kneib 1997; Sanger et al. 1999a, b; Lerberg et al. 2000; Mallin et al. 2000; Holland et al. 2004). In the SE, the watersheds associated with headwater tidal creeks are among the most rapidly developing in the nation.

A conceptual model linking watershed development (stressors), the associated physical and chemical exposures, and ecological responses was developed for headwater tidal creeks in South Carolina (Holland et al. 2004). Changes in the rate and volume of storm water runoff resulting from increases in impervious cover were predominant factors driving ecological impairment of tidal creeks in this model. Adverse changes in the creek physical and chemical environment occurred when impervious cover levels exceeded 10–20 %. Ecological processes in creek ecosystems responded when impervious cover levels exceeded 20–30 %. This study and others (e.g., Odum 1984; Dame et al. 1992; Sanger et al. 1999a, b; Lerberg et al. 2000; Washburn and Sanger 2013) also suggest that the variability among and within tidal creek networks is large and that a classification framework would facilitate understanding this variability and identifying its causes. Classification frameworks developed for freshwater streams and wetland ecosystems have contributed to a greater understanding and integration of the ecological attributes of these ecosystems within their biogeography, hydrology, and short- and long-term ecological history (e.g., Horton 1945; Cowardin et al. 1979; Frissell et al. 1986). The overall goal of this study was to clearly define the relationships between coastal development, with its concomitant land use changes and associated increases in nonpoint source pollution loadings, and the ecological condition of tidal creek ecosystems including the consequences of creek impairment to human populations and coastal communities throughout the SE. The two primary hypotheses were: (1) there is no relationship between impervious cover, an indicator of sprawling coastal development, and the ecology and potential consequences of coastal development to coastal communities and human populations and (2) there is no difference in the ecological characteristics and impacts of coastal development down the length of tidal creeks (i.e., first orders compared to second/third orders).

Methods

Nineteen tidal creek networks between New Hanover County, NC, and Glynn County, GA, were sampled during the summers (June–August) in 2005 and 2006 (Fig. 1). Twelve SC networks were sampled in 2005, and four of these networks were resampled in 2006. In 2006, four creek networks in GA and three networks in NC were sampled (Table 1). Tides are semi-diurnal and range from approximately 1 m in NC to 3 m in GA. Vegetation surrounding the SE tidal creeks is primarily *Spartina alterniflora* and *Juncus roemarianus*.

A longitudinal gradient was defined for each creek system by applying a freshwater stream classification model (Horton 1945; Strahler 1957) to tidal creek systems (DiDonato et al. 2009; Fig. 2). The first order, or headwaters, of each creek was characterized by narrow widths (2–10 m) and consisted of primarily intertidally dominated habitat. The second and third orders of each creek were formed by the confluence of two or more first or second orders, respectively. Second- and third-order systems had widths of approximately 10–100 m and consisted primarily of subtidally dominated habitat. First-order sections will hereafter be referred to as intertidal. The second- and third-order systems, hereafter referred to as subtidal, were combined as the differences between them were generally small in SC and only one third-order creek was sampled in NC and GA. Each 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 sampled. Within any reach, stations were randomly located for sample collection.

Watersheds and sub-watersheds were identified and land use and impervious cover were determined for each creek and order using ArcGIS 9 (Fig. 2). Watersheds and their sub-watersheds were delineated based on elevation data including United States Geological Survey topographic data, digital elevation model data, and Elevation Derivatives for National Applications (http://edna.usgs.gov/). National Land Cover Data (NLCD; Homer et al. 2004; accessed at http://gisdata.usgs.net/website/MRLC/viewer.php) from 2001 were clipped by each watershed and sub-watershed to obtain the land cover and impervious cover data. Land

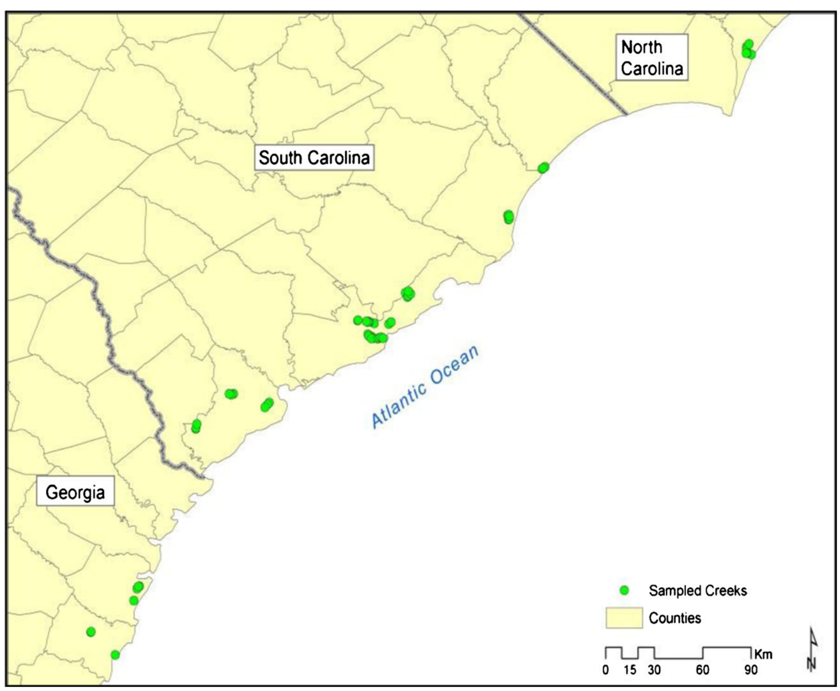

Fig. 1 North Carolina, South Carolina, and Georgia sampling sites

Table 1 Creek system, land use class, latitude and longitude, watershed area, and impervious cover for each creek segment

Creek system	Land use class	Latitude	Longitude	Order	Creek segment	Area (ha)	Impervious cover (%)
North Carolina							
Hewlitts	Suburban	34.189	−77.857	1	Intertidal	614	40.9
				1	Intertidal	459	34.5
				2	Subtidal	2,782	33.4
Masonboro	Marsh	34.152	−77.849	1	Intertidal	29	2.9
Whiskey	Suburban	34.161	−77.865	1	Intertidal	482	34.7
				2	Subtidal	712	32.4
South Carolina							
Albergottie	Suburban	32.448	−80.720	1	Intertidal	558	8.1
				2 and 3	Subtidal	2,096	23.9
Bulls	Urban	32.825	−80.027	1	Intertidal	369	40.5
				2 and 3	Subtidal	510	38.1
Guerin	Marsh	32.944	−79.766	1	Intertidal	25	0.0
				2	Subtidal	342	0.0
	Forested			1	Intertidal	219	3.0
				2 and 3	Subtidal	3,427	3.0
James Island	Suburban	32.744	−79.974	1	Intertidal	296	30.0
				2	Subtidal	773	29.1
	Suburban			1	Intertidal	144	41.3
				2 and 3	Subtidal	1,820	29.5
Murrells Inlet	Urban	33.564	−79.025	2 and 3	Subtidal	1,297	40.3
New Market	Urban	32.806	−79.940	1	Intertidal	199	70.4
North Inlet	Marsh	33.339	−79.189	1	Intertidal	55	0.0
				2	Subtidal	102	0.0
	Forested			1	Intertidal	184	2.9
				2 and 3	Subtidal	1,860	2.9
Okatee	Suburban	32.287	−80.929	1	Intertidal	2,415	17.9
				2 and 3	Subtidal	5,501	13.3
Orangegrove	Marsh	32.812	−79.978	1	Intertidal	18	0.0
				2	Subtidal	59	0.0
	Suburban			1	Intertidal	61	39.2
				2 and 3	Subtidal	322	37.3
Parrot	Marsh	32.733	−79.910	1	Intertidal	28	0.0
	Suburban			1	Intertidal	62	21.2
				2 and 3	Subtidal	501	17.7
Shem	Urban	32.801	−79.869	1	Intertidal	456	49.4
				2	Subtidal	1,269	47.7
Village	Forested	32.419	−80.522	1	Intertidal	630	3.6
				2 and 3	Subtidal	2,016	4.0
Georgia							
Burnett	Urban	31.234	−81.538	1	Intertidal	2,425	11.2
				2	Subtidal	2,589	11.8
Duplin	Forested	31.145	−81.285	1	Intertidal	385	3.0
				2 and 3	Subtidal	1,480	3.0
Oakdale	Forested	31.481	−81.272	1	Intertidal	286	3.1
Postell	Urban	31.417	−81.375	1	Intertidal	218	39.8

Subtidal watershed area includes the related intertidal area

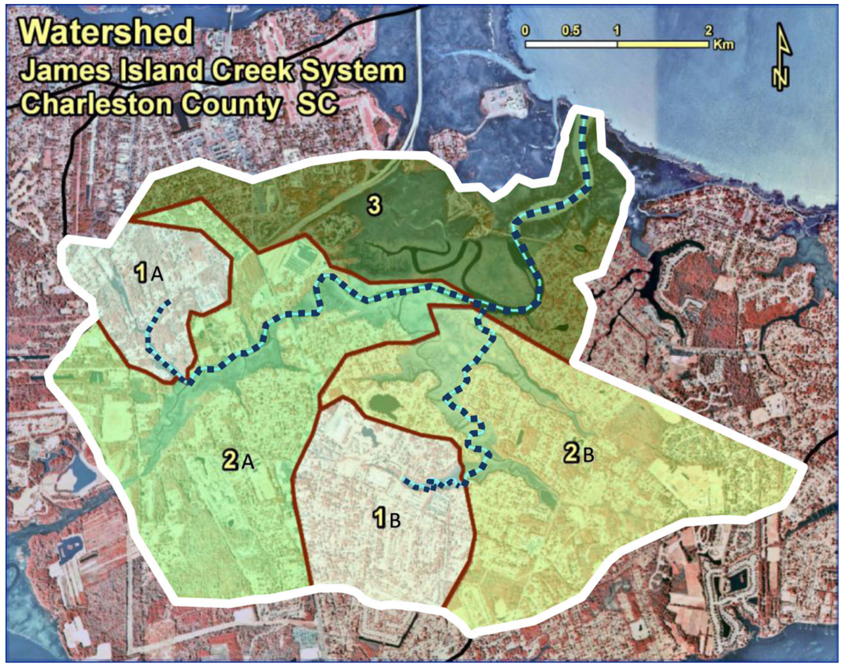

Fig. 2 Example of a study creek watershed with sub-watersheds identified for each order. The entire watershed is outlined in *white*, with sub-watersheds identified in *gray*. Creek order is identified by *1*, *2*, or *3*, with an *A* or *B* to distinguish the multiple orders sampled in a single system. The creek area sampled is shown with the *black dashed line*

cover data were determined from this layer and summed to obtain simplified categories of land cover. The impervious cover data were further modified by removing data that represented marsh and open water or undevelopable areas. Impervious cover levels were calculated from the NLCD for all sub-watersheds and watersheds and then adjusted using a quadratic relationship ($y=2.9301+2.16789x-0.01611x^2$), where y is the adjusted impervious cover percent and x is the NLCD-derived impervious cover, as described in DiDonato et al. (2009). Our findings and those of Jarnagin et al. (2006) have reported the NLCD impervious cover levels as underestimates. Storm water runoff volume was estimated for first-order upland creek watersheds using a modified version of the U.S. Department of Agriculture, Natural Resources Conservation Service curve number method. These results are presented in Blair et al. (2013) and the findings highlighted in the discussion.

Creek watersheds were classified at the largest order level into the following land use categories based on impervious cover: (1) forested (<10 % impervious cover); (2) suburban (≥10 % but <35 % impervious cover); (3) urban (≥35 % impervious cover); and (4) salt marsh (emergent marsh as the dominant land cover class). Several systems had upland creeks representing various levels of human development, but also had creek segments that were dominated by salt marsh. Specifically, within the North Inlet, Guerin, Parrot, and Orangegrove systems, both upland and salt marsh creeks were sampled. Upland and salt marsh segments were treated separately in statistical analyses. There were two exceptions to the above land use classification. The Orangegrove watershed was estimated to have 37.3 % impervious cover; however, since land cover was primarily light residential development, it was categorized as a suburban watershed. The Burnett watershed was estimated to have 11.8 % impervious cover; however, since this network was a Superfund site designated by the United States Environmental Protection Agency (USEPA), it was categorized as an urban watershed.

Each creek was sampled approximately 2–3 h prior to low tide, and sampling occurred over two consecutive days. Sampling was conducted in an upstream direction to minimize habitat disturbance. Sampling stations were selected using a stratified random method. The number of samples collected in each order varied by sample type.

Basic water quality data (temperature, dissolved oxygen, salinity, pH, turbidity, chlorophyll-*a*) were collected in bottom waters (0.3 m above bottom) using a YSI 6600 data logger. Data loggers were calibrated prior to deployment and a post-calibration check was conducted to ensure the instrument was functioning properly. A logger was deployed in the second reach of each creek order and collected data at 15-min intervals for up to two full tidal cycles (25 h).

Water samples were collected approximately 0.3 m below the surface and in the upstream direction. Water samples for bacterial and viral pathogen indicators were collected in the second reach in sterile 2-L polypropylene bottles and processed within 24 h. Fecal coliforms (FC) and enterococci (ENT) were enumerated using membrane filtration according to standard methods (APHA 1998). Male-specific (F+) and somatic (F−) coliphages were enumerated by the single-agar layer method, adapted from USEPA method 1602 (USEPA 2001b) and described in Stewart et al. (2006).

Water samples for nutrients were collected in an acid-washed 500-mL polyethylene bottle in each reach of each creek order. 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 the dissolved constituents [i.e., ammonium (NH_4^+), nitrite+nitrate ($NO_{2/3}$), total dissolved nitrogen (TDN), orthophosphate (PO_4^{3-}), total dissolved phosphorus (TDP), and silicate (DSi)]. Ammonium was analyzed via the Berthelot reaction using a Technicon AutoAnalyzer (Technicon Industrial Systems 1986). Both orthophosphate and nitrate+nitrite were analyzed using standard methods (USEPA 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).

Macrobenthic infauna was sampled using two different methods. In intertidal creeks, the benthos was sampled approximately 1 m below mean high water using a 0.0044-m^2 core sampler. A total of nine cores (three from each reach) were collected at randomly located stations along the entire intertidal habitat. A small scoop of mud was collected next to each core sample for sediment analysis [percent sand, percent silt, percent clay, total organic carbon (TOC)]. In subtidal creeks, the infauna was sampled using a 0.04-m^2 modified Van Veen grab sampler. One grab sample was collected in each reach. Sediment samples for grain size analysis determination were taken from the top 2 cm of a second intact grab from each site. Benthic samples were sieved through a 0.5-mm sieve and preserved in 10 % formalin containing Rose Bengal. Detailed information on macrobenthic methods and the response of individual taxa and community-level responses to watershed development are provided in Washburn and Sanger (2011) and only highlighted in the discussion in relation to the conceptual model.

Sediments were sampled for chemical contaminants in the second reach of each order. In intertidal creek segments, the top 2 cm of sediment was removed from the sediment surface 1 m below high tide and homogenized in a stainless steel bowl. In the subtidal creek segments, the top 2 cm of several successful Van Veen grabs was homogenized for chemical analysis. The homogenates were apportioned to appropriate pre-cleaned sample jars (i.e., metals in plastic and organics in glass) and placed on ice as soon as possible. Sediments were analyzed for 22 trace metals, 22 pesticides, 25 polycyclic aromatic hydrocarbons (PAHs), 79 polychlorinated biphenyls (PCBs), and 13 polybrominated diphenyl ethers (PBDEs). Data quality was assured using a series of spikes, blanks, and standard reference materials (NIST 1944 for sediments and NIST 1566b for tissues). All contaminants were analyzed by the National Oceanic and Atmospheric Administration (NOAA), National Ocean Service (NOS), Center for Coastal Environmental Health and Biomolecular Research (CCEHBR) using procedures similar to those described by Krahn et al. (1988), Fortner et al. (1996), Kucklick et al. (1997), Long et al. (1997), and Schantz et al. (1997).

Nekton, predominantly fish and epibenthic crustaceans, were sampled using two different methods. In intertidal creeks, the nekton was sampled using a 0.25-in. (0.635 cm) mesh seine net. One seine was pulled in each reach in an upstream direction for up to 25 m, stretching the net from bank to bank. Water width and depth were measured at the starting and end points of the seined area to calculate the area and volume of the creek sampled. In subtidal creeks, nekton were sampled using a four-seam trawl (5.5-m foot rope, 4.6-m head rope, and 1.9-cm bar mesh throughout) pulled at a constant speed in the downstream direction for 250 m. Nekton collected by seine were preserved in 10 % formalin in seawater. Preserved organisms were sorted and identified to the lowest practical taxonomic level (usually species). Animals collected in trawls were identified and counted in the field.

In 2006, oysters (*Crassostrea virginica*) were collected near where the data logger was deployed and sediment chemistry samples were collected when present. After collection, oysters were separated for pathogen determination (~20 oysters), determination of chemical contaminant body burdens (~12 oysters), and genomic transcriptome analyses (25 oysters). Oysters tested for pathogen body burdens were homogenized and composited to obtain at least 100 g (wet weight) tissue. Oyster homogenate was then tested for FC and ENT using the multiple fermentation tube most probable number (MPN) technique (APHA 1998). The homogenate was also tested for F+ and F− coliphages by plating the equivalent of 12.5 g using the single-agar layer method (USEPA 2001b). Oyster tissues for chemical contamination analysis were processed for the same chemicals as the sediment samples and similar methods were employed. Oyster genomic transcriptome methods and findings are provided in Chapman et al. (2009, 2011) and only highlighted in the discussion.

Tidal creek data from both 2005 and 2006 were combined for analyses; no attempt was made to examine year-to-year variability. The main unit of statistical inference was the creek order, and the resulting data set comprised 43 observations (24 from intertidal systems and 19 from subtidal systems). In cases involving multiple measures per order, data were averaged within each order to obtain one value for each indicator. Creek data were summarized by averaging across the second and third orders to get one value representing the subtidally dominated habitats. Data for creeks that were sampled across both 2005 and 2006 (i.e., Guerin, James Island School, New Market, and Village) were averaged across years.

Statistical analyses were designed to address the following null hypotheses: (1) no differences occurred for environmental quality parameters among land use classes; (2) measured environmental quality parameters in creeks did not vary among creek orders (i.e., along the creek longitudinal gradient); and (3) no predictive relationships occurred between the measured environmental quality parameters in creeks and impervious cover levels of associated watersheds. To address these hypotheses, we employed analysis of variance (ANOVA) and regression. The basic ANOVA model was a two-way, fixed-factor model, with land use class type (salt marsh, forested, suburban, urban) and creek order (intertidal, subtidal) as the main effects. The interaction term was included in all models and excluded if non-significant ($p \geq 0.05$). 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 document significant predictive relationships. Regressions were considered significant at $p < 0.05$. The regressions were performed with the forested, suburban, and urban creeks. The salt marsh creeks were excluded from this analysis because they had no developable uplands. The mean effects range median quotient (mERMQ) method was used to simplify the sediment contaminant data for 24 compounds (Long and Morgan 1990; Long et al. 1995; Long and MacDonald 1998). The use of these quotients provides a way to compare potential cumulative effects of contaminants after weighting them on a toxicological basis. If residuals were found to be non-normal or heteroscedastic, basic transformations (log, square root, arcsine) were attempted. If those transformations did not improve the distribution of the data, data were rank-transformed. Analyses were performed using SAS 9.1 (SAS Institute Inc., Cary, NC) or Systat 11 (Systat Software, Inc., San Jose, CA).

Results

Stressors

The 19 tidal creek systems surveyed consisted of one to five sub-watersheds or drainage basins depending upon the number of intertidal and subtidal creek segments sampled (Table 1). Intertidal creek watersheds ranged in size from 28 to >2,400 ha, and impervious cover levels ranged from 0 % up to about 70 %. Subtidal creek watersheds included the intertidal area and ranged in size from 59 to 5,501 ha; impervious cover levels ranged from 0 to 47.7 %. Land use classes were primarily determined using watershed impervious cover, and as expected, the amounts of impervious cover within each watershed class were significantly different from each other (ANOVA: $p < 0.0001$), with a progressive increase from salt marsh to forested to suburban to urban. Across land use classes, there was no significant difference between the intertidal and subtidal impervious cover amounts.

Classifying watersheds based on impervious cover provides a useful framework for analyzing and interpreting study results. Similarly, impervious cover is a useful indicator of the physical conditions and environmental quality attributes of watersheds (Schueler 1994; Arnold and Gibbons 1996). For intertidal and subtidal creek watersheds, human population density (individuals per hectare) was linearly related to the impervious cover (in percent) and explains 88 and 75 %, respectively, of the total variability (Fig. 3a).

Exposures

Basic Water Quality

Averages for creek water temperature values were from 25.0 to 32.6 °C. Average pH values ranged from 6.52 to 7.97. Average salinity values were from 0.51 to 35.1 ppt. Average dissolved oxygen (DO) values ranged from 2.68 to 6.89 mg L^{-1}. Temperature ranges (maximum minus minimum) were from 1.24 to 13.18 °C. pH ranges were from 0.18 to 1.53. Salinity ranges were from 0.8 to 31.3 ppt. DO ranges were from 2.68 to 15.02 mg L^{-1}.

Average water quality measurements were not significantly affected by land use class or longitudinal gradient. Land use, however, had a significant effect on salinity range, with the urban and suburban creeks having significantly larger ranges than the marsh and forested creeks (Table 2). In addition, salinity range as well as temperature range and DO range responded to the longitudinal spatial gradient, with the intertidal creeks having significantly larger ranges than subtidal creeks.

Intertidal salinity range showed a significant relationship with the amount of impervious cover in the watersheds (Fig. 3b). None of the other basic water quality metrics had statistically significant regressions with watershed impervious cover levels.

Nutrients and Phytoplankton

The concentration levels for nutrients ranged from one to three orders of magnitude among the creeks sampled. TN ranged from 0.37 to 4.65 mg L^{-1}. $NO_{2/3}$ ranged from 0.003 to 0.397 mg L^{-1}. TP ranged from 0.03 to 3.93 mg L^{-1}. Chl-*a* levels ranged from 0.57 to 174.15 μg L^{-1}. TDN and TDP ranged from 0.23 to 3.19 mg L^{-1} and from 0.02 to 2.85 mg L^{-1}, respectively. Based on categorical guidelines developed for coastal waters by NOAA (Bricker et al. 1999), the 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 TDN, TDP, and Chl-*a* concentrations were classified in the higher categories compared to the subtidal creeks. TDN concentrations for intertidal creeks draining suburban and urban watersheds were classified as medium for North Carolina study sites and either medium or high for South Carolina and Georgia study sites. TDN concentrations were classified as medium for all subtidal creeks and for intertidal creeks draining

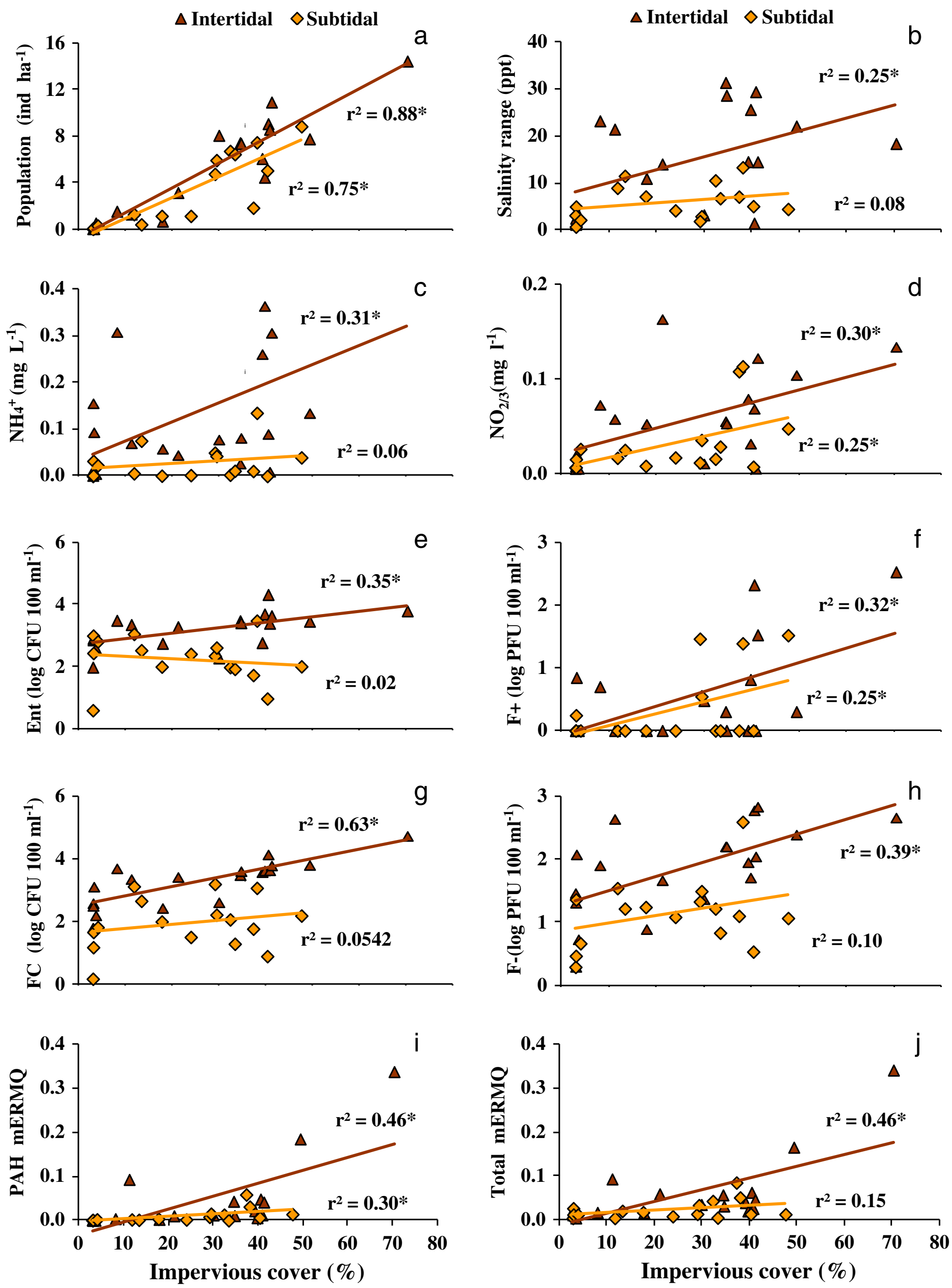

Fig. 3 Relationships between indicator variables and impervious cover within watersheds. Model r^2 is shown for each regression, with *asterisk* indicating significance ($p<0.05$). Marsh watersheds are excluded owing to lack of impervious cover

forested and marsh watersheds, with one exception—one marsh intertidal site was classified as high.

The type of land use surrounding the tidal creeks had little effect on most nutrient and Chl-*a* concentrations, whereas the longitudinal spatial gradient sampled showed a more consistent significant effect (Table 2). Land use class, however, did have a significant effect on $NO_{2/3}$ concentrations, and the $NO_{2/3}$ concentrations for creeks in marsh and forested watershed classes were significantly lower than those in developed watershed classes. All nutrient concentrations, with the exception of $NO_{2/3}$, exhibited similar spatial gradients, with intertidal creeks having significantly (or trending toward significance, $p<0.10$) higher levels than subtidal creeks (Table 2).

Intertidal concentrations of NH_4^+ and $NO_{2/3}$ and subtidal levels of $NO_{2/3}$ increased significantly with increasing levels of impervious cover in the watersheds (Fig. 3c, d). None of the other nutrients or phytoplankton measures were significantly related to watershed impervious cover levels.

Pathogens

FC concentrations in the water column ranged from <1 to 91,000 colony forming units (CFU) 100 mL^{-1}, and ENT

Table 2 Results of two-way ANOVA on the averages and selected ranges of water quality and sediment quality indicator variables sampled in summer, 2005 and 2006

Parameter	Model p value	r^2	Land use p value	Order p value	Interaction	Land use LS means	Order LS means
Basic water quality—range							
Temperature	<0.001	0.61	0.111	<0.001	ns		I^aS^b
Salinity	0.001	0.37	<0.05	<0.05	ns	$F^aM^aS^bU^b$	S^aI^b
Dissolved oxygen	<0.001	0.44	0.303	<0.001	ns		S^aI^b
pH	0.059	0.21	0.057	0.183	ns		
Basic water quality—average							
Temperature	0.144	0.16	0.085	0.599	ns		
Salinity	0.061	0.21	0.071	0.120	ns		
Dissolved oxygen	0.500	0.08	0.774	0.121	ns		
pH	0.450	0.09	0.855	0.092	ns		
Nutrients/phytoplankton							
Ammonium (NH_4^+)	0.003	0.33	0.169	0.001	ns		S^aI^b
Nitrate+nitrite ($NO_{2/3}$)	0.001	0.40	0.003	0.329	ns	$F^aM^aS^bU^b$	
Total dissolved nitrogen	0.057	0.21	0.663	0.006	ns		S^aI^b
Total nitrogen	<0.001	0.42	0.687	<0.001	ns		S^aI^b
Orthophosphate (PO_4^{3-})	0.062	0.21	0.084	0.092	ns		
Total dissolved phosphorous	0.059	0.21	0.111	0.056	ns		
Total phosphorous	0.003	0.34	0.120	0.001	ns		S^aI^b
Silicate	0.149	0.16	0.871	0.014	ns		S^aI^b
Chlorophyll-*a*	0.341	0.23	0.858	0.002	ns		S^aI^b
Phaeophytin	0.004	0.33	0.677	<0.001	ns		S^aI^b
Pathogen indicators							
Enterococcus (ENT)	0.001	0.37	0.015	0.002	ns	$M^aF^{ab}S^bU^b$	S^aI^b
Fecal coliform	<0.001	0.62	<0.001	<0.001	ns	$M^aF^aS^bU^b$	S^aI^b
F− coliphage	<0.001	0.49	<0.001	0.001	ns	$M^aF^aS^bU^b$	S^aI^b
F+ coliphage	0.007	0.30	0.004	0.300	ns	$M^aF^aS^aU^b$	
Sediment quality							
Sediment % clay	0.291	0.12	0.776	0.051	ns		
Sediment TOC	0.014	0.27	0.533	0.001	ns		S^aI^b
Total mERMQ	0.009	0.29	0.055	0.009	ns	$F^aM^{ab}S^bU^b$	S^aI^b
PCB mERMQ	0.043	0.22	0.046	0.149	v	$F^aM^aS^aU^b$	
Metal mERMQ	0.061	0.21	0.361	0.016	ns		S^aI^b
PAH mERMQ	0.001	0.49	0.007	0.040	0.016	$F^aM^aS^aU^b$	

Post hoc multiple comparisons were performed using least squared means. Model factors (arranged from low to high) with different superscripts are statistically different

Land use class factors: *M* marsh, *F* forested, *S* suburban, *U* urban. Order factors: *I* intertidal, *S* subtidal

concentrations ranged from 3 to 21,000 CFU 100 mL^{-1}. Levels of the measured indicator viruses tended to be lower than those of the bacteria, ranging from <1 to 450 plaque forming units (PFU) 100 mL^{-1} and from <1 to 1,200 PFU 100 mL^{-1} for F+ and F− coliphages, respectively.

The type of land use surrounding the tidal creeks and the spatial gradient sampled were found to affect both bacterial and viral pathogen indicator densities. The concentrations generally were lowest in salt marsh and forested watershed classes and highest in the urban watershed classes. This pattern was most apparent in the intertidal creeks. FC and F− coliphage levels differed significantly for the salt marsh and forested creeks compared to the developed (suburban and urban) watershed classes (Table 2). ENT concentrations were significantly higher in the suburban and urban classes compared to the salt marsh class, with the forested class not significantly different from the other land use classes. F+ coliphage levels were <1 PFU 100 mL^{-1} in all salt marsh creeks, and the F+ concentrations in the marsh, forested, and suburban classes were significantly lower than the concentrations in the urban class. ENT, FC, and

F− coliphage concentrations exhibited similar spatial patterns, with intertidal creeks having significantly higher densities of pathogen indicators than subtidal creeks. The F+ coliphage concentrations showed a similar trend, but were not statistically significant (Table 2).

The concentrations of pathogen indicators increased with increasing levels of impervious cover in the watersheds (except ENT in subtidal areas; Fig. 3e–h). In intertidal creeks, significant relationships were found between all of the pathogen indicators and the amount of impervious cover in the watershed. In the subtidal creeks, only the F+ coliphage showed a significant relationship with impervious cover in the watershed.

Sediment Quality

Neither the percent sand nor percent clay composition was significantly related to either the surrounding land use or longitudinal spatial gradient (Table 2). Percent clay concentrations ranged from 1.6 to 74.0 %. Sediment TOC ranged from 0.09 to 10.7 %, with significantly higher TOC concentrations in the intertidal creeks compared to the subtidal creeks (Table 2). There was no measurable effect of land use class.

Sediments from intertidal systems generally showed increasing concentrations of PAHs, PCBs, and pesticides from forested and salt marsh to suburban and urban creeks. Forested creeks had significantly lower Total mERMQ values than both suburban and urban creeks; marsh creeks had Total mERMQ values between the forested and the suburban/urban creeks (Table 2). The intertidal creeks showed a trend of increasing Total mERMQ values from forested to suburban to urban creeks. Salt marsh creeks were often similar to the suburban creeks, which may be due to the high levels of TOC in marsh creeks. The Pesticide mERMQ and Metal mERMQ values were similar across land use classes, while the PAH mERMQ and PCB mERMQ values were significantly higher in urban land use class compared to the other classes (Table 2). The intertidal creeks had significantly higher concentrations of overall contamination (Total mERMQ) as well as pesticide and metal contamination than the subtidal creeks. The PAH mERMQ and PCB mERMQ values were similar down the length of the creek.

Regression analysis demonstrated that mERMQ values generally increased with increasing levels of impervious cover. Regressions of Total mERMQ, Metal mERMQ, and PAH mERMQ values versus impervious cover were statistically significant in the intertidal creeks (Fig. 3i, j). In addition, PAH mERMQ was positively related with impervious cover levels in subtidal creeks.

PDBEs were only detected in the intertidal areas of the more developed creeks in SC: two suburban and three urban sites. This finding suggests that intertidal creeks may be potentially valuable sentinel habitats for providing early warning of emerging chemical contaminant pollution.

Ecological Response

Macrobenthic Community

A synthesis of the macrobenthic community results was published in Washburn and Sanger (2011) and will not be discussed here.

Nekton Community

Not unexpectedly, the nekton assemblages in intertidal and subtidal creeks differed substantially. These differences are most likely related to differences in nekton utilization patterns for intertidal and subtidal habitats as well as differences in gear sampling characteristics. Habitat structure of the different orders (e.g., water quality, volume) may also have had a role. Due to gear differences, communities between the two system types were analyzed separately. Fifty-nine intertidal and 59 subtidal species were identified. Only species occurring commonly across the creeks were analyzed statistically as individuals per square meter and individuals per hectare for the intertidal and subtidal, respectively. For the intertidal systems, the species occurring most commonly were *Palaemonetes* spp. (grass shrimp), Penaeidae (white and brown shrimp), *Fundulus heteroclitus* (mummichog), *Leiostomus xanthurus* (spot), and *Callinectes sapidus* (blue crabs). For the subtidal systems, eight species occurred most commonly: *Litopenaeus setiferus* (white shrimp), *L. xanthurus*, *Farfantepenaeus aztecus* (brown shrimp), *Lolliguncula brevis* (brief squid), *Lagodon rhomboides* (pinfish), *Bairdiella chrysoura* (silver perch), *C. sapidus*, and *Anchoa mitchilli* (bay anchovy). Differences were also observed based on geographical location, with the highest abundances of shrimp in GA and the lowest abundances in NC creeks.

In the intertidal creeks, none of the models were significant at $p<0.05$; however, *Palaemonetes* spp. showed a trend toward a significant land use class effect, with forested creeks having the highest abundance and suburban creeks significantly lower than the other land use classes (Table 3). In the subtidal creeks, *L. rhomboides* was the only species with a significant land use class effect (Table 3). The forested and suburban classes had significantly lower abundances of this species compared to the marsh class, and the urban class was similar to the other classes. The general trend regarding land use class was that the abundances of individual species in marsh creeks were different from the abundances in the other three land use classes.

Human Consequences

Oyster Tissue Pathogens

Oysters were only collected during the 2006 sampling period from 11 tidal creek systems. Pathogen indicator concentrations

Table 3 Results of one-way ANOVA examining differences in average abundance in intertidal and subtidal creeks separately by land use class

Creek parameter	Model *p* value	r^2	Land use LS means
Intertidal			
F. heteroclitus	0.884	0.03	
C. sapidus	0.297	0.17	
L. xanthurus	0.539	0.10	
Palaemonetes spp.	0.064	0.30	$S^aM^{ab}U^bF^b$
Penaeidae	0.233	0.19	
Subtidal			
B. chrysoura	0.422	0.17	
C. sapidus	0.304	0.21	
L. rhomboides	0.044	0.41	$F^aS^aU^{ab}M^b$
L. xanthurus	0.179	0.27	
L. brevis	0.462	0.15	
F. aztecus	0.851	0.05	
L. setiferus	0.416	0.17	
Penaeidae	0.806	0.06	

Post hoc multiple comparisons were performed using least squared means. Model factors (arranged from low to high) with different superscripts are statistically different

Land use class factors: *M* marsh, *F* forested, *S* suburban, *U* urban

for any particular parameter varied over two to four orders of magnitude. FC ranged from 23 MPN 100 g^{-1} tissue wet weight in a subtidal, forested creek to 2.2×10^5 MPN 100 g^{-1} wet weight in an intertidal, urban creek. ENT concentrations varied from 3.2×10^3 MPN 100 g^{-1} wet weight to 3.2×10^5 MPN 100 g^{-1} wet weight. F− coliphages in a few cases were not detected, but reached 6.1×10^3 PFU 100 g^{-1} wet weight in one urban creek. F+ coliphages were only detected in four samples; the highest concentrations were observed in a forested creek on Sapelo Island (4.8×10^3 PFU 100 g^{-1} wet weight) that partially drained a rural Gullah-Geechee island community on septic systems.

The overall sample size was small as oysters were only collected from a subset of the sampled systems. Nonetheless, ANOVA results indicated a significant land use class effect for F− coliphages, with concentrations in oysters collected from forested watersheds being lower than those collected from either suburban or urban watersheds (Table 4). Regression analysis showed that there was a significant ($p<0.05$) positive relationship between watershed impervious cover and FC concentrations in oysters collected in intertidal creeks (Fig. 4a). For F− coliphages, there were significant relationships with watershed impervious cover in both creek orders (Fig. 4b). There were no other significant regressions.

Oyster Tissue Contaminants

Oyster tissue lipid concentrations ranged from 5.7 to 38 %. In general, higher lipid values were observed in the forested creeks compared to the marsh and developed creeks. In addition, the lipid concentrations tended to be slightly lower in the subtidal creeks compared to the intertidal creeks.

Total PAH tissue concentrations ranged from 0 to 2,161 ng g^{-1} tissue dry weight. Naphthalene, a low-molecular-weight PAH, was most commonly detected. The other PAHs detected included acenaphthylene, anthracene, benzo(*a*)anthracene, benzo(*b*)fluoranthene, benzo(*g*,*h*,*i*)perylene, fluoranthene, and pyrene. All except acenaphthylene are high-molecular-weight PAHs typical of pyrogenic sources and were found in developed systems, except for one forested system which had high levels of benzo(*g*,*h*,*i*)perylene (488 ng g^{-1} dry weight).

Total PCB concentrations ranged from 0 to 244.3 ng g^{-1} dry weight. Concentrations above the detection limit were found in the three urban intertidal creeks and in the one urban subtidal creek; low concentrations (<2 ng g^{-1} dry weight) were found in the remaining subtidal creeks and one forested and one suburban intertidal creeks. PBDEs, flame retardants, were detected in oyster tissue at only one site, an intertidal, urban system. Pesticide concentrations in oyster tissues were dominated by DDT and its derivatives. Total DDT concentrations ranged from 2.53 to 20.54 ng g^{-1} dry weight. In general, total DDT tissue concentrations were higher in the developed systems. The only other detectable

Table 4 Results of two-way ANOVA examining differences in concentrations of selected pathogen indicators measured in tissue from oysters collected in study creeks

Parameter	Model *p* value	r^2	Land use *p* value	Order *p* value	Interaction	Land use LS means	Order LS means
Enterococcus	0.563	0.15	0.375	0.774	ns		
Fecal coliform	0.181	0.32	0.419	0.078	ns		
F− coliphage	0.005	0.65	0.004	0.076	ns	$F^aU^bS^b$	
F+ coliphage	0.352	0.23	0.677	0.153	ns		

The one sample from a Marsh creek (Masonboro) was excluded from these analyses. Post hoc multiple comparisons were performed using least squared means. Model factors (arranged from low to high) with different superscripts are statistically different

Land use factors: *F* forested, *S* suburban, *U* urban. Order factors: *I* intertidal, *S* subtidal

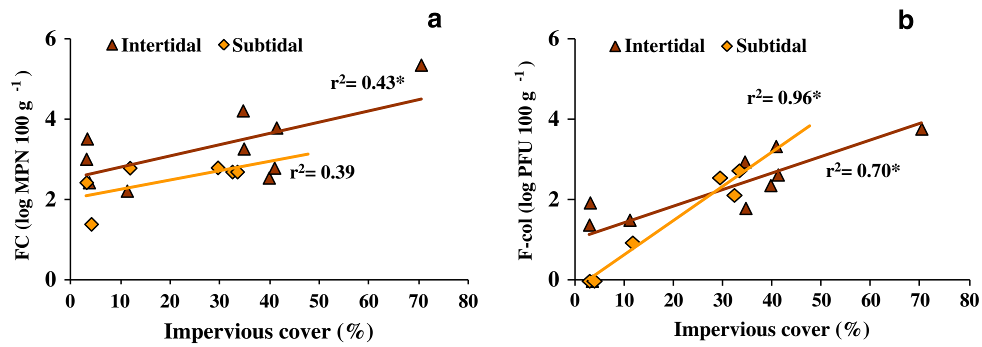

Fig. 4 Relationships between pathogen indicator levels in oyster tissues and impervious cover for the study watersheds. Model r^2 is shown for each regression, with *asterisk* indicating significance ($p<0.05$). Log transformation is $x+1$

pesticide contaminants were mirex (3.49 ng g^{-1} dry weight) in one intertidal, forested creek, endosulfan I (2.38 ng g^{-1} dry weight) in one intertidal, forested creek, and dieldrin (3.82 ng g^{-1} dry weight) in one intertidal, urban creek.

To evaluate the potential for bioaccumulation of PCBs, PCB congeners were grouped by the number of chlorines and then compared to sediment PCBs. PCBs with seven or more chlorines have an increased potential for transfer up the food web (Oliver and Niimi 1988). The highest total sediment PCB concentration was in an intertidal, urban creek (107 ng g^{-1} dry weight) and consisted primarily of lower chlorinated compounds (hexa- and tetrachlorobiphenyls). The oyster tissue concentrations in this creek were comparatively low (approximately 52 ng g^{-1} dry weight). This difference corresponds with conclusions that lower chlorinated compounds found in the sediments are not bioaccumulating in tissues. In comparison, Burnett, an urban, intertidal creek and Superfund site, had the second highest total PCB concentration in sediments (61 ng g^{-1} dry weight) and consisted primarily of higher chlorinated compounds. Oyster tissue concentrations were high in this system (244 ng g^{-1} dry weight), reflecting high bioaccumulation in this system.

Concentration data for only a few metals (As, Cd, Cr, Cu, Pb, Hg) are discussed here. Except for arsenic, these are the metals which are often elevated from anthropogenic sources. Lead concentrations were similar across the different land use classes and were generally low (<0.7 μg g^{-1} dry weight), except for one intertidal, urban creek (1.58 μg g^{-1} dry weight). Mercury concentrations were similar across land use classes and were generally low (<0.19 μg g^{-1} dry weight), except for one urban creek 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 were found in the NC creeks, similar to the fish tissue contamination findings of Cooksey et al. (2008). Cadmium, copper, and chromium concentrations were generally higher in the forested and suburban creeks compared to the marsh and urban creeks. In particular, Guerin, a forested creek in the Francis Marion National Forest, had some of the highest cadmium, copper, and chromium concentrations.

In addition, oyster tissue concentrations on a wet weight basis were compared to U.S. Food and Drug Administration (USFDA 2011) environmental chemical contaminant action levels and the USEPA (2000) human health consumption limits for cancer and non-cancer endpoints. The USFDA action levels are simply threshold values for comparison against tissue concentrations (non-consumption-based). None of the concentrations observed in oyster tissue exceeded any of the molluscan or fish actions levels for As, Cd, Cr, Pb, Ni, methyl mercury, PCBs, DDT, heptachlor epoxide, or mirex. The USEPA values are based on a consumption rate of four 8-oz meals of fish per month for an adult population. It should be noted that we are comparing oyster tissue to fish tissue values; however, the comparison represents a level of potential risk. It should also be noted that for a number of these systems, the shellfish are closed for harvest. Inorganic arsenic (estimated as 2 % of total arsenic) and total DDT values exceeded the cancer endpoint for all sites sampled for oyster tissue. Dieldrin values exceeded the cancer endpoint at one intertidal, urban site. Total PCB values only exceeded the USEPA cancer endpoint in both the intertidal and subtidal sites in Burnett Creek, an urban creek, and the intertidal site also exceeded the USEPA non-cancer endpoint.

Discussion

Previous research in tidal creek ecosystems has demonstrated that the environmental quality of these systems, particularly the intertidally dominated portions or headwaters, is sensitive to land use changes within their relatively small (hundreds to thousands of hectares) watersheds (Sanger et al. 1999a, b; Lerberg et al. 2000; Mallin et al. 2000; Holland et al. 2004; DiDonato et al. 2009). Because tidal creeks are sensitive to local land use changes, these systems provide an early warning of the degradation from surrounding upland land use well before changes would be detected in larger coastal waters (e.g., tidal rivers, estuaries). Tidal creeks are therefore useful and important sentinels for monitoring the impacts of human activities on coastal habitats at local scales.

The current study demonstrates that the sensitivity of tidal creeks to changes in these small coastal watersheds diminishes down their length (i.e., from small intertidal headwater creeks to larger subtidal creeks). This spatial variability must be recognized before assessing the environmental quality of these habitats. For many of the measured parameters, the intertidally dominated or headwater portions of tidal creeks were found to respond differently from the subtidally dominated or larger, deeper portions of tidal creeks. The smaller intertidal creeks generally had higher concentrations of nonpoint source pollutants, which are likely indications of higher proportional levels of upland runoff into headwater creeks as well as an estuarine dilution influence (i.e., tidal flushing) in the larger creeks. The biological parameters measured (e.g., nekton and benthos) also demonstrate significant variability along the longitudinal spatial (i.e., headwaters to tidal river) gradient. There is a marked shift in the macrobenthic infauna, from one dominated by oligochaetes in the headwaters to one dominated by polychaetes in the deeper subtidal creeks (Washburn and Sanger 2011). The nekton also appears to shift along this gradient, from more resident and nursery species in the headwaters to larger transient organisms in the deeper subtidal regions of creeks. Recognizing the spatial variability that occurs for headwaters to subtidal habitats of creeks not only allows more meaningful comparisons to be made across similar creek classes (with respect to surrounding land use for example) but should also provide better insight into how biological resources of tidal creeks ecosystems respond to coastal development and other stressors.

In addition to accounting for the spatial variability down the length of a creek, the type of land cover in the watershed is also an important factor to consider when comparing creeks. The creeks draining watersheds with only salt marsh land cover were found to respond differently to land use changes from creeks draining watersheds with forested upland land cover. The major pathway of contamination and pollution loadings is different between these two creek classes. Creeks draining only salt marsh primarily receive contaminants and other pollutants from downstream sources, particularly adjacent water bodies. In comparison, creeks draining upland terrestrial areas receive significant freshwater input and pollution loadings from the upstream upland areas. The input of freshwater is a critical factor to consider when assessing the impacts of land use change on tidal creek ecosystems. Freshwater input in the form of storm water runoff increases with increasing levels of impervious cover (Blair et al. 2013), carrying increased pollutant loadings from the surrounding watershed into tidal creeks.

To assist in understanding the complexity and variability associated with freshwater streams and rivers, classification frameworks have been developed that integrate the ecological attributes of these systems in the context to their biogeography, hydrology, and short- and long-term ecological history (e.g., Horton 1945; Frissell et al. 1986). Classification approaches have, however, made only limited contribution to the understanding of spatial and temporal variability and scale issues for tidal creek ecosystems (e.g., Anderson et al. 1976; Odum 1984). The reasons estuarine ecologists have not embraced classification as a means of partitioning and understanding tidal creek complexity include: (1) standardized approaches for resolving scale, space, time, and location differences within and among creeks have not been developed and applied, (2) environmental conditions vary on multiple temporal and spatial scales (e.g., tidal, diel, extreme events, seasonal, year-to-year, climatic, geological), and (3) much of tidal creek ecology is based on indirect evidence from relatively few places, with few studies evaluating ecological differences and similarities on regional scales. The findings of this study clearly demonstrate that the tidal creek classification framework (i.e., longitudinal and land use) applied in this study has general applicability for the southeast region and contributed to an improved understanding of spatial and temporal variability in tidal creek ecosystems. Future studies should be conducted to refine this preliminary classification framework.

Tidal creek networks are the primary hydrologic link between estuaries and adjacent land-based activities. As the first zone of coastal impact for nonpoint source pollution runoff entering the estuary from surrounding land use, the potential for microbial and chemical contamination in tidal creek habitats is great. Developing a conceptual model is a critical step for identifying and evaluating monitoring and management strategies (Saila 1979; NRC 1990; Barnthouse and Brown 1994). Holland et al. (2004) developed a conceptual model to identify and describe the source–receptor links between coastal development and anticipated impacts on tidal creek ecosystems. The model was based on the USEPA Ecological Risk Assessment paradigm with stressors leading to changes in the physical–chemical environment (i.e., exposures), which in turn leads to a biological response.

The conceptual model of Holland et al. (2004) developed for South Carolina intertidally dominated tidal creeks did not include a number of new indicators sampled by this study (e.g., nutrients, emerging contaminants of concern, indicators of viral pathogens) or show the potential consequences of an impaired tidal creek environment to human populations and coastal communities. Historically, scientists have only looked at how humans impact the natural environment, with little emphasis on how impairment to the natural environment affects human populations and coastal communities (e.g., Millennium Ecosystem Assessment 2005a, b). Based on the data collected in this study, the conceptual source–receptor model developed by Holland et al. (2004) has been expanded (Fig. 5). This updated model provides an overview of the linkages between coastal development and associated human activities, changes in the physical–chemical environment, anticipated responses of tidal creek ecosystems, and potential consequences to human populations and coastal communities. This model was developed using an integrated weight of evidence approach based on the information collected.

The sprawling coastal development activities in the surrounding watersheds (stressor) found in this study included changes in the land cover and increases in the population density and impervious cover. The changes in the physical–chemical environment (exposure) associated with increasing development included increases in the salinity range, increases in the levels of nitrate/nitrite and ammonium, increases in the amount of storm water runoff (Blair et al. 2013), increases in the concentrations of bacterial and viral pathogen indicators, and increases in chemical contamination of the sediment including some emerging chemicals of concern. The ecological response or impacts on the living resources identified in this study include impaired oyster health as evidenced by the changes in gene expression (Chapman et al. 2009, 2011), reduced secondary biological productivity (Lerberg et al. 2000; Holland et al. 2004; Washburn and Sanger 2011), and alterations to the food web (Jones 2008). The reduced biological productivity is associated with the impacts on the macrobenthic infauna such as changes in the species composition, abundance of organisms and diversity with increasing levels of development (Lerberg et al. 2000; Holland et al. 2004; Washburn and Sanger 2011), as well as changes in the nekton community, particularly reduced brown and white shrimps abundances, with increasing levels of development when small geographic areas were evaluated (Holland et al. 2004; Jones 2008). The absence of nekton community responses to changes in land use associated with coastal development, particularly over larger regional spatial scales, is likely the result of the variability in nekton recruitment patterns and population dynamics at larger geographic scales (e.g., regional), the mobile nature of nekton, the large salinity variance in creek headwaters, and the changes in food resources between intertidal and subtidal habitats. Jones (2008) evaluated the isotopic signatures of three shrimp species (*Palaemonetes pugio*, *L. setiferus*, *F. aztecus*), a fish (*F. heteroclitus*), a macrobenthic oligochaete worm (*Monopylephorus rubroniveus*), and various primary producers in a subset of our study creeks with varying levels of land use. In her study, the $\delta^{15}N$ ratios were found to increase with increasing levels of development for the four nekton species, and the relative contributions of the various food resources were different with varying levels of watershed development. This indicates that the food web was altered by the increasing levels of development.

Some of the consequences of sprawling coastal development to human populations and coastal communities include potential economic impacts from impaired environmental quality (Lovelace unpublished), increased public health risk (DiDonato et al. 2009; this paper), and potential for increased flooding potential (Blair et al. 2013). Identification of direct relationships between the level of coastal development at the small watershed scale and direct impacts on human health and well-being has been elusive and challenging. Lovelace (unpublished) evaluated the relationships between property values and environmental conditions within a subset of our study creeks for a range of development levels in Charleston County, SC. Her evaluation found some promising relationships, including higher property values associated with deeper creeks and lower turbidity and lower property values associated with higher nitrate/nitrite and total suspended solids levels. However, the broader application of Lovelace's evaluation across the full range of our study creeks was not feasible given the inconsistency of property value and demographic data across the counties studied. We also considered evaluating direct impacts on human health. It was, however, not possible to obtain human health data at the scale of our watersheds (hundreds to thousands of hectares). We concluded that formal studies to evaluate the impact of sprawling coastal development on human health and well-being were not feasible within our study constraints (e.g., time, money) due to the scale mismatch between environmental data and human health and well-being data. Our study did clearly indicate there was increased risk to human health from exposure to increased levels of pathogen indicators and chemicals in the water, sediments, and shellfish. Our results were also concordant with the finding of others that current patterns of coastal development are associated with increasing fecal pollution in tidal creeks, estuaries, and bathing beaches (Mallin et al. 2000; Karn and Harada 2001; Holland et al. 2004; Mallin 2006). In addition, Blair et al. (2013), which is a component of our study, showed that as the level of coastal development increased, the potential for flooding from flashy and episodic storm water runoff also increased, mainly due to increases in impervious cover. Maiolo and Tschetter (1981) evaluated shellfish bed closures in the coastal counties of North Carolina and found that as development increased, so did the increases in shellfish bed closures.

Our current findings, as evidenced by evaluating the graphics and statistics through a weight of evidence approach, agree with the broad impervious cover thresholds originally proposed by Holland et al. (2004), making the revised model applicable throughout the SE (Fig. 5). The tidal creek conceptual model identifies that adverse changes generally occur in the physical and chemical environment when impervious cover levels in the watershed reach 10–20 %. Ecological processes responded to and were generally impaired when impervious cover levels exceeded 20–30 % (Fig. 5). From a human consequence perspective, estimates of impervious cover levels defining where human uses are impaired continue to be a challenge, but it generally appears that health risks and flooding vulnerability of headwater regions become a concern when impervious cover values exceed 10–30 %. This research project has validated and

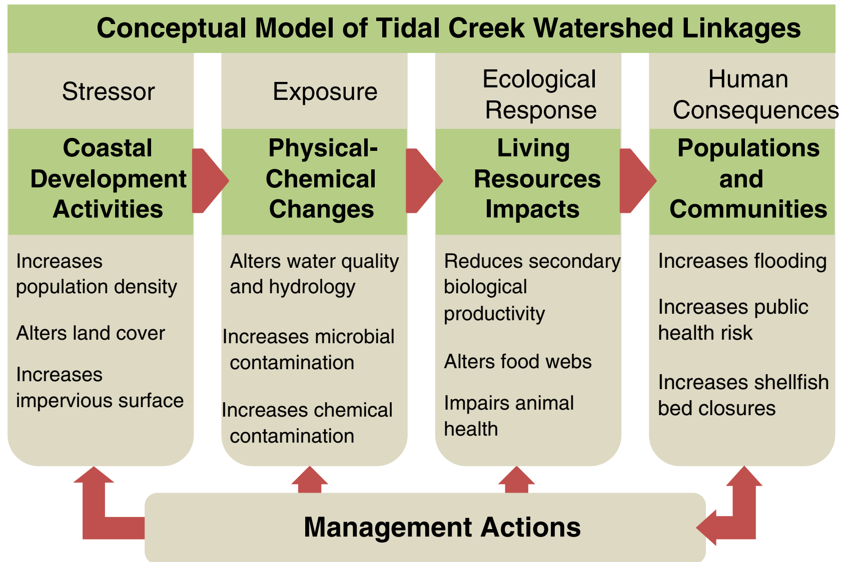

Fig. 5 Conceptual model of the relationships between the stressor (coastal development), the exposure (physical–chemical changes in the tidal creek), the ecological response (natural resources), and human consequences (human populations and communities) that make up the tidal creek ecosystem

expanded the model for the southeastern USA. It should also be noted that these thresholds are in good agreement with freshwater stream and other tidal creek studies (e.g., Schueler 1994; Arnold and Gibbons 1996; Mallin et al. 2000).

One of the primary purposes of developing the conceptual model was to clearly and succinctly outline study findings in a simple and concise format for a wide range of audiences (e.g., general public, municipal official, coastal managers). The tidal creek conceptual model also provides a framework for defining system feedback loops such that the level of government which is responsible for ensuring appropriate actions are taken to remediate and restore impaired systems can be identified (Fig. 5). County and municipal governments are responsible for regulating land use activities and make most zoning decisions, which ultimately controls impervious cover levels. State and federal governments mainly influence physical–chemical exposures (water and sediment quality), but also play a large role in enforcement and permitting activities related to near-marsh development.

Summary

In the southeastern USA, coastal uplands adjacent to tidal creeks and salt marshes are increasingly popular locations for human development. These tidal creek networks are also critical feeding grounds, spawning areas, and nursery habitats for many species of fish, shellfish, birds, and mammals. Tidal creeks form the primary hydrologic link between estuaries and adjacent land-based activities and, as such, reflect the impacts of coastal development earlier than larger coastal water bodies. Nonpoint source pollution (e.g., storm water runoff from adjacent upland development) carries sediment, chemicals, bacteria, viruses, and other pollutants into tidal creeks and salt marshes and degrades their environmental quality. The relationships between increases in coastal development levels and the environmental quality and public health risk indicators evaluated were strongest in the shallow, intertidally dominated headwater creeks.

The relationship between watershed development and the ecological condition of the headwater areas of tidal creeks in SC is fairly well understood, but spatial and temporal variability and patterns in ecological condition along tidal creek networks are often poorly characterized. Effective monitoring, assessment, and prediction of the effects of coastal urbanization on tidal creeks and estuaries require that this variability be characterized and understood. Stratification of tidal creek networks into units that represent relatively homogenous environments or creek classes is one tool for characterizing and understanding the variability within tidal creek networks. This stratification is crucial for understanding at what scale land use impacts are likely to be observed. Classifying watersheds that drain into specific creek networks based on the degree and type of development that exists is a tool and requirement for understanding variability among creek networks and forecasting the impacts of development.

The scale of our tidal creek study watersheds (hundreds to thousands of hectares) is also the spatial scale at which coastal land use decisions and remediation actions typically occur. Creeks draining the headwater portions of those watersheds are valuable indicators of impacts from land use activities and urbanization. Our conceptual model provides managers or land use planners with a valuable tool to understand the impacts of developments on the environmental quality and potential human consequences in nearby tidal creeks and thereby inform the decision-making process.

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