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Linkages between tidal creek ecosystems and the landscape and demographic attributes of their watersheds

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Abstract

Twenty-three headwater tidal creeks draining watersheds representative of forested, suburban, urban, and industrial land cover were sampled along the South Carolina coast from 1994 to 2002 to: (1) evaluate the degree to which impervious land cover is an integrative watershed-scale indicator of stress; (2) synthesize and integrate the available data on linkages between land cover and tidal creek environmental quality into a conceptual model of the responses of tidal creeks to human development; and (3) use the model to develop recommendations for conserving and restoring tidal creek ecosystems. The following parameters were evaluated: human population density, land use, impervious cover, creek physical characteristics, water quality, sediment chemical contamination and grain size characteristics, benthic chlorophyll *a* levels, porewater ammonia concentration, fecal coliform concentration, and macrobenthic and nekton population and community characteristics.

The conceptual model was developed and used to identify the linkages among watershed-scale stressors, physical and chemical exposures, and biological responses of tidal creeks to human development at the watershed scale. This model provides a visual representation of the manner in which human population growth is linked to changes in the physiochemical environment and ultimately the nursery habitat function of tidal creeks and the safety of seafood harvested from headwater tidal creeks. The ultimate stressor on the tidal creek ecosystem is the human population

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density in the watershed and associated increases in the amount of impervious land cover. Measurable adverse changes in the physical and chemical environment were observed when the impervious cover exceeded 10–20% including altered hydrography, changes in salinity variance, altered sediment characteristics, increased chemical contaminants, and increased fecal coliform loadings. Living resources responded when impervious cover exceeded 20–30%. The impacts on the living resources included reduced abundance of stress-sensitive macrobenthic taxa, reduced abundance of commercially and recreationally important shrimp, and altered food webs. Headwater tidal creeks appear to provide early warning of ensuing harm to larger tidal creeks, tidal rivers and estuaries, and the amount of impervious cover in a watershed appears to be an integrative measure of the adverse human alterations of the landscape. Through education and community involvement, a conservation ethic may be fostered that encourages the permanent protection of lands for the services they provide.

Keywords: Tidal creeks; Impervious cover; Watershed development; Landscape indicators; Ecosystem responses; Nursery habitat

1. Introduction

The estuaries of the southeastern United States are dynamic environments characterized by low relief, shallow depth, broad fluctuations in water quality and expansive intertidal areas dominated by shallow tidal creeks and salt marshes (Teal, 1962; Hackney et al., 1976; Day et al., 1989; Wiegert and Freeman, 1990; Kneib, 1997; Dame et al., 2000; Lerberg et al., 2000; Wenner et al., 2001). These creeks and marshes provide nursery habitat and feeding grounds for fish, shellfish and wading birds (e.g., Hackney et al., 1976; Cain and Dean, 1976; Shenker and Dean, 1979; Weinstein, 1979; Dodd and Murphy, 1996; Kneib, 1997). Over the next several decades, the human population in the watersheds that drain into southeastern estuaries is expected to increase by over 60% from the 1960 levels (Culliton et al., 1990). This growth will result in conversion of substantial portions of the existing forested and agricultural ecosystems into low and high density suburban housing, transportation infrastructure, shopping centers, resorts, and industrial sites. These land cover changes are projected to adversely affect the productivity, biodiversity, and ecological functioning of coastal ecosystems, particularly the tidal creeks which are the first-order connections between uplands and estuaries (Olsen et al., 1982; Arnold and Gibbons, 1996; Vitousek et al., 1997; Sanger et al., 1999a,b; Lerberg et al., 2000).

The response of freshwater streams to land cover changes associated with urbanization has been well studied (e.g., Schueler, 1994; Arnold and Gibbons, 1996; Beach, 2002). These studies link increases in human population density with increases in the amount of impervious cover (e.g., roofs, roads, parking lots) and degradation of stream environmental quality and ultimately biological integrity. When the degree of imperviousness exceeds about 10%, adverse changes in water quality have generally been reported. Severe biological degradation occurs when more than about 30% of the watershed is converted to impervious cover (e.g., Schueler, 1994; Arnold and Gibbons, 1996). In summary, these studies suggest that the amount of impervious cover is a sensitive indicator of the consequences land development has on the ecological integrity of freshwater ecosystems. These findings have contributed to the identification and development of a plethora of stormwater controls for protecting and restoring stream quality (Schueler and Holland, 2000).

This study was initiated to provide regulatory and resource management agencies with information for conserving and restoring tidal creek ecosystems. Some of this research has been previously published (Sanger et al., 1999a,b; Lerberg et al., 2000); however, in this publication, we will expand upon these previous findings to: (1) evaluate the degree to which impervious cover is a watershed-scale indicator of tidal creek systems; (2) synthesize and integrate the available data on linkages between land cover and tidal creek environmental quality into a conceptual model of tidal creek responses to landscape development; and (3) use the model and findings to develop recommendations for conserving and restoring tidal creek ecosystems.

2. Methods

2.1. Watershed classification and land use

Twenty-three tidal creeks draining watersheds representative of forested, suburban, urban and industrial development were sampled from 1994 to 2002 (Fig. 1). Criteria used to classify watersheds were: (1) forested or reference, < 30% urban/suburban land cover and < 10% of watershed as impervious cover; (2) suburban, >30% but < 70% urban/suburban land cover with a human population density >5 but < 20 individuals/ha or >10% but < 50% of watershed as impervious cover; (3) urban, >70% urban/suburban land cover with a human population density of >20 individuals/ha or >50% of watershed as impervious cover; and (4) industrial, >45% urban/suburban land cover with industrial facilities and >50% of watershed as impervious cover (Table 1). These criteria were developed based upon information provided by Anderson et al. (1976), Schueler (1994), and Arnold and Gibbons (1996) as well as knowledge of land use patterns in South Carolina. Threshold values represented a gradient of landscape development from relatively pristine forested habitat to highly urbanized and industrialized landscapes.

The tidal creeks sampled formed a first-order connection between the landscape and the adjacent estuary. The upper creek boundary was the point where water depth was ~ 1 m on the average high tide or where an impassable obstacle (e.g., a dike or security fence) was reached. The lower boundary was the point where the creek converged with another water body or where channel water depth exceeded 3 m on the average high tide. Water depth in the creek channel ranged from < 10 cm at low tide to ~ 3 m at high tide. The creeks represented the range of salinity distributions, sediment characteristics, creek lengths (231–1491 m), watershed sizes, land cover types and levels of human disturbance that occur in South Carolina. The developed creek population drained watersheds along the Ashley River (BL, KP, DL), the Cooper River (SH, NM, VR), the Wando River (HB, SM), the lower Charleston Harbor (CC, MC, PC, YC), as well as regions of the state north (MI) and south (AS) of Charleston. The reference creek population drained

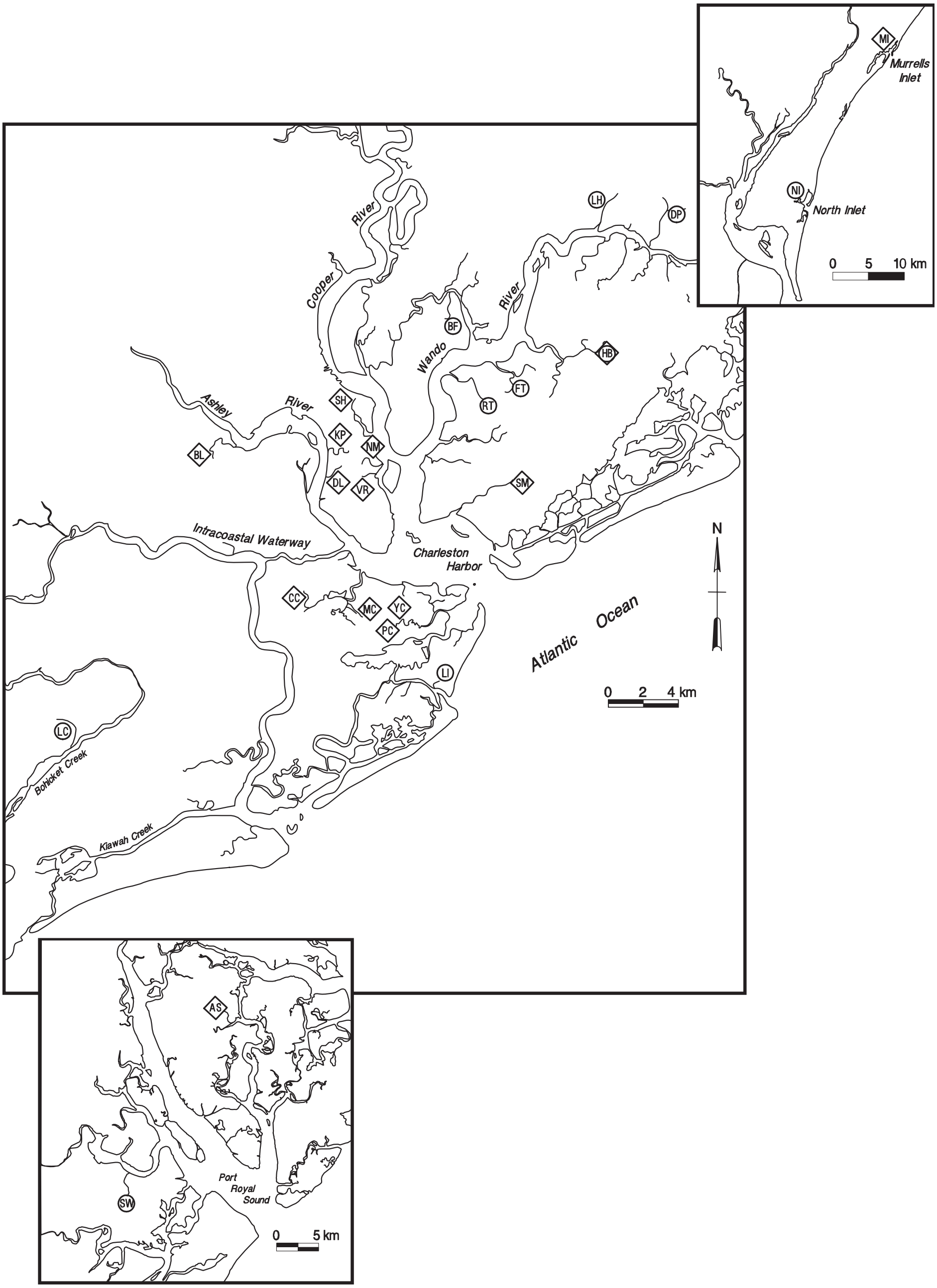

Fig. 1. Map showing location of tidal creeks sampled in South Carolina. The reference (○) and developed/impacted (◇) creeks are presented by their abbreviations (see Table 1). The inserts are the north (top) and south (bottom) sampling sites.

Table 1
Summary of the creeks sampled for each study (SC = summer calibration, SV = summer validation, C = chemistry, WC = winter calibration, SF = summer fecal coliforms) including watershed size

Creek name	Creek abbreviation	1992 Watershed type	1999 Watershed type	1994 SC	1994 SV	1995 SV	1995 C	2000 WC	2002 SF	Watershed size (ha)
Beresford	BF	forested	forested	×			×	×	×	25
Crab Hall	NI	forested	forested			×	×			197
Deep	DP	forested	forested	×	×		×	×	×	65
Foster	FT	forested	forested	×			×	×	×	16
Lachicotte	LH	forested	forested	×	×		×	×	×	13
Lighthouse Inlet	LI	forested	forested	×			×	×		37
Long	LC	forested	forested	×	×	×	×	×	×	412
Rathall	RT	forested	forested	×		×	×	×	×	72
Sawmill	SW	forested	forested			×	×			524
Horlbeck	HB	forested	suburban	×			×	×	×	238
Bull	BL	suburban	suburban	×			×	×	×	434
Cross	CC	suburban	suburban	×			×	×	×	313
MCAS	AS	suburban	suburban			×	×			434
Metcalf's	MC	suburban	suburban	×	×		×	×	×	130
Murrells Inlet	MI	suburban	suburban			×	×			83
Parrot	PC	suburban	suburban	×			×	×	×	147
Yacht Club	YC	suburban	suburban	×			×	×	×	69
Shem	SM	suburban	urban	×	×	×	×	×	×	428
New Market	NM	urban	urban	×	×		×	×	×	198
Vardell	VR	urban	urban	×			×	×	×	69
Diesel	DL	industrial	industrial	×		×	×	×	×	105
Koppers	KP	industrial	industrial	×	×		×	×	×	117
Shipyard	SH	industrial	industrial	×			×	×	×	280

watersheds on Wadmalaw Island (LC), the Wando River (DP, LH, RT, FT, BF), the lower Charleston Harbor (LI), and relatively pristine habitats north (NI) and south (SW) of Charleston (Fig. 1).

Watershed boundaries of the study creeks were developed based on elevation contours defined by 1:24,000 United States Geological Survey topographic maps. The outline of each creek's drainage basin was then digitized to produce an estimate of watershed boundaries at a 1:4800 scale. Watersheds ranged in size from 13 to 524 ha.

Land cover data were developed for 2 years (1992 and 1999) by: (1) superimposing a transparent overlay of the watershed upon 1:4800 blackline aerial photographs taken in 1992 and 1:4800 high-resolution color infrared (CIR) National Aerial Photographic Program (NAPP) photographs taken in February 1999; (2) defining polygons that represented areas of homogeneous land cover within each watershed; (3) estimating the area of each land cover type with a planimeter; and (4) summing the estimates of each land cover category (Lerberg et al., 2000). The six land cover categories established were derived from the Anderson Level II Land Cover Classification System and included: (1) estuarine salt marsh; (2) freshwater ponds/lakes; (3) forested land; (4) suburban/urban land; (5) agricultural land; and (6) barren land or land in transition from one category to another (Anderson et al., 1976). Land cover estimates for watersheds that experienced little or no development between 1992 and 1999 were similar for the blackline aerial photographs and the CIR NAPP photographs suggesting the two types of photographs produced similar results. Human population statistics were obtained from block level 1990 and 2000 U.S. Census data using TIGER/Line Files.

The proportion of each watershed that represented impervious cover was estimated by applying a triangular sampling grid to the 1:4800 blackline or the NAPP photography. The number of points sampled in the watersheds ranged from 34 to 1373. The percent impervious cover was determined to be the number of points on the overlay that fell on roads, sidewalks, parking lots, building roofs, road medians, and railroad tracks divided by the total number of non-salt marsh grid points within the watershed boundary multiplied by 100.

2.2. Sample design

This study consisted of four sampling time periods: July–September 1994, July–September 1995, January–February 2000, and June–July 2002 (Table 1). The parameters that were measured varied slightly among sampling dates. These differences are summarized in Table 2. The creeks were stratified into 300-m reaches for sampling in 1994/1995. For the winter 2000 and summer 2002 studies, only the upper reaches of creeks were sampled.

Nineteen creeks located in Charleston Harbor were sampled for the July–September 1994 sampling effort to evaluate linkages between the degree and type of watershed development and indicators of tidal creek ecological condition (Table 1, 1994 summer calibration study). Sampling was conducted on randomly selected days. Six of the creeks representing forested (3), suburban (1), urban (1), and industrial (1) watershed categories were sampled a second time in summer 1994 to estimate within-season variability and to determine the reliability of the linkages identified during the calibration study (Table 1, 1994 summer validation study).

Between July and September 1995, four of the original Charleston Harbor creeks representing the forested (2), suburban (1), and urban (1) watershed categories were

Table 2
Summary of the parameters sampled for each study (see Table 1 for study abbreviations)

Measured parameter	1994 SC/SV	1995 SV	1995 C	2000 WC	2002 SF
1994 Blackline land use	×	×	×		
1999 NAPP land use				×	×
1990 Population	×	×	×		
2000 Population				×	×
Water quality	×	×		×	
Sediment composition	×	×	×	×	
Sediment chemistry		×	×	×	
Porewater ammonia				×	
Benthic chlorophyll *a*				×	
Macrobenthic community	×	×		×	
Nektonic community	×	×			
Fecal coliforms					×

sampled to evaluate if the response patterns observed in summer 1994 persisted in a different year (Table 1, 1995 summer validation study). Four additional creeks (i.e., one forested and one suburban creek in Beaufort County, SC, about 100 km south of Charleston Harbor and one forested and one suburban in Georgetown/Horry Counties, SC, about 100 km north of Charleston Harbor) were also sampled in summer 1995 to evaluate if the response patterns observed in Charleston Harbor in 1994 were applicable over a broader region (Table 1, 1995 summer validation study). Because the sediment contaminant data collected in summer 1994 were determined to be unreliable, the type and degree of sediment chemical contamination were evaluated for the upper and lower reaches of the 19 Charleston Harbor creeks as well as the 4 additional creeks north and south of Charleston (Table 1, 1995 sediment chemistry study). The 1995 chemistry values will be summarized with the 1994 summer calibration study.

During January and February 2000, the upper reaches of 19 creeks located in Charleston Harbor were sampled for a reduced suite of parameters to determine if the response patterns observed in summer 1994 and 1995 applied to a different season and to identify changes that may have occurred over the 5-year period (Table 1, 2000 winter calibration study). The limited suite of parameters measured in winter 2000 mainly included selected water quality, sediment quality, and macrobenthic community properties (Table 2). In addition, benthic chlorophyll *a* and porewater ammonia were measured in the winter 2000 study. Sediment chemical contamination samples were only processed for 8 of the 19 creeks. No nekton samples were collected due to the extremely low winter abundances. The sampling schedule for the winter 2000 study was not random in time but was based upon weather conditions, tidal stage and accessibility.

In June and July 2002, water samples for fecal coliform determinations were collected from 18 of the Charleston Harbor creeks (Table 1, 2002 summer fecal coliform study).

2.3. Field collection and laboratory processing

Macrobenthic community samples were collected randomly along the length of each creek at approximately 1 m below the mean high tide line (mid-tide level). Six within-creek replicate samples were collected in 1994/1995, and three within-creek replicates were collected in 2000. Samples were collected with a 45.6-cm^2 coring device to a depth of 15 cm and sieved through a 500-μm screen. All organisms retained on the screen were preserved and identified to the lowest possible taxonomic level and counted. Two benthic community metrics that were developed by Lerberg et al. (2000), one of which was modified by Gawle (2002), were used to evaluate the benthic data: (1) stress-sensitive taxa (i.e., relative abundance of Nemertinea, *Streblospio benedicti*, *Tharyx* cf. *acutus*, *Tubificoides heterochaetus*) and (2) stress-tolerant taxa (i.e., *Laeonereis culveri*, *Monopylephorus rubroniveus*, *Tubificoides brownae*, *Paranais litoralis*). The taxa composing each of the community metrics had similar responses to watershed development during all seasons and years. Detailed discussions of the response of individual taxa and community-level responses to watershed development are provided by Lerberg et al. (2000) and Gawle (2002) and are not described here. Paired surface (upper 2 cm) sediment samples were collected at each sample site and processed to determine percent moisture, percent silts and clays, and percent sand using methods modified from Plumb (1981).

Seine samples of fish and crustaceans were collected at a random site in each reach during 1994. The seines were square mesh with 0.6-cm bar and 16-kg weight webbing. They were pulled on the ebbing tide for 25 m in water depth less than 1-m deep but greater than 0.25-m deep. Seine samples were processed by two methods depending on the weight of the preserved sample. Samples with a wet weight < 2 kg were completely processed. Samples weighing >2 kg were subsampled. Large (>20 cm) organisms and samples of rare taxa were removed and placed in 50% isopropanol before subsamples were defined. The sample was then divided into 10 approximately equal-weight subsamples. Two of the ten subsamples were randomly selected and processed. Organisms in the subsamples were identified to the lowest possible taxon and counted. Abundance data from the subsamples were summed and multipled by 5. Then, the abundance data for rare taxa were added to the total.

A Hydrolab Datasonde 3 (DS3) water quality monitoring system was deployed in the lower reach of each creek for 2–7 days prior to other sampling. The DS3 was positioned approximately 5–10 cm above the creek bottom and measured salinity, water temperature, dissolved oxygen (DO) concentration, pH, and water depth at 30-min intervals. The 48 h of water quality data most proximal to biological sampling events were used for analysis.

Surface sediments (upper 2 cm) were collected for chemical analyses at one randomly selected site from the upper- and lowermost reach of each creek in 1995 and the upper reach of each of the eight creeks in 2000. Sample collection and analyses were performed for 14 trace metals, 24 polycyclic aromatic hydrocarbons (PAHs), 20 polychlorinated biphenyls (PCBs), and 15 pesticides using methods described by Sanger et al. (1999a,b).

Benthic chlorophyll *a* and porewater ammonia were measured in winter 2000 adjacent to each macrobenthic sample. Benthic chlorophyll *a* samples were obtained by inserting a 7-mm-diameter core into the top 1 cm of sediment. Porewater ammonia samples were collected by inserting a 4-cm-diameter core 3 cm into the sediment. The three benthic chlorophyll *a* and porewater ammonia samples were combined and homogenized to obtain a single sample representing each creek. Sediment chlorophyll *a* and porewater ammonia samples were kept on ice in the dark until processed.

Sediment chlorophyll *a* samples were processed within 24 h of field collection using a modification of the spectrophotometric method (Strickland and Parsons, 1972). Sample extraction occurred in the dark with 10 ml of 100% acetone at 4 °C for 24 h. The samples were vortexed four times during extraction: immediately following the addition of acetone, 12 and 24 h after the addition and 1 h prior to analysis. Samples were allowed to equilibrate to room temperature and were centrifuged at 3000 rpm for 2 min. Three milliliters of the supernatant was transferred to a 1-cm pathlength quartz cuvette and absorption was read with an UV–Vis spectrophotometer. Chlorophyll *a* was calculated from an equation converting UV-absorption to a concentration (Strickland and Parsons, 1972). Sediment porewater ammonia samples were homogenized and placed into a 50-ml centrifuge tube and spun at 4500 rpm for 10 min. Ammonia concentrations of the supernatant were measured using the salicylate–cyanurate method following the protocols of the manufacturer (Hach®, 1994).

In summer 2002, the concentration of fecal coliform bacteria was measured in the water of the upper most reaches of creeks. Samples were collected at low tide when Mallin et al. (1999) reported concentrations were generally highest. Water samples were collected with

Table 3
Summary of regression results for each parameter versus the impervious cover for the four studies

Parameter	1994 Summer calibration				1994 Summer validation				1995 Summer validation				2000 Winter calibration			
	n	r^2	*p*	Slope	*n*	r^2	*p*	Slope	*n*	r^2	*p*	Slope	*n*	r^2	*p*	Slope
Population (no. of people/hectare)	19	0.63	< 0.001	0.27									19	0.52	0.001	0.17
Mean temperature (°C)	19	0.03	0.501	2.51	6	0.38	0.193	−10.48	8	0.11	0.422	−6.45	19	0.01	0.704	0.95
Mean pH	19	0.09	0.202	29.26	6	0.21	0.360	63.32	8	0.00	0.944	2.45	19	0.07	0.276	39.04
Mean salinity (psu)	19	0.05	0.345	−0.80	6	0.02	0.786	−0.60	8	0.11	0.425	−0.98	19	0.05	0.352	−0.92
Salinity range	19	0.26	0.025	2.00	6	0.51	0.114	4.84	8	0.00	0.933	0.09	19	0.04	0.402	0.72
Mean DO (% saturation)	19	0.02	0.537	−0.29	6	0.09	0.562	1.33	8	0.00	0.955	−0.04	19	0.05	0.335	−0.69
Percent of time DO < 28% sat. [arcsine]	19	0.00	0.797	−6.00	6	0.34	0.224	−221.1	8	0.00	0.910	−3.61	19	0.00	0.933	−11.41
Silt/clay (upper)	19	0.31	0.014	−0.46	6	0.44	0.152	0.76	8	0.06	0.568	−0.35	19	0.06	0.318	−0.26
Mean ERMQ [log]	19	0.78	< 0.001	53.10					4	0.70	0.164	36.30	8	0.27	0.228	32.47
Ammonia concentration (μM/l)													19	0.18	0.070	0.21
Benthic chlorophyll *a* (mg/m^2)													19	0.14	0.118	0.21
Stress-sensitive taxa (upper/lower)	19	0.46	0.001	−0.95	6	0.71	0.034	−0.96	8	0.04	0.649	−0.21				
Stress-tolerant taxa (upper/lower)	19	0.32	0.012	0.62	6	0.93	0.002	0.88	8	0.00	0.872	0.08				
Stress-sensitive taxa (upper)	19	0.44	0.002	−0.82	6	0.65	0.053	−0.80	8	0.04	0.615	−0.23	19	0.28	0.019	−0.69
Stress-tolerant taxa (upper)	19	0.12	0.140	0.35	6	0.87	0.006	0.70	8	0.02	0.729	−0.13	19	0.33	0.010	0.67
F. heteroclitus [log]	19	0.03	0.502	−13.36	6	0.11	0.525	51.66	8	0.02	0.722	−15.87				
L. xanthurus [log]	19	0.10	0.185	−71.22	6	0.40	0.181	−223.7	8	0.14	0.365	−96.36				
Palaemonetes spp. [log]	19	0.16	0.095	−10.44	6	0.05	0.673	−8.62	8	0.21	0.255	−12.86				
Penaeoid shrimp [log]	19	0.31	0.013	−26.37	6	0.39	0.184	−34.25	8	0.19	0.284	−33.03				
Fecal coliform (2002 summer [log])	18	0.48	0.001	23.06												

Data were transformed as noted in brackets.

sterile, autoclaved plastic bottles with the mouth of the bottle positioned so that water flowed into the bottle. Samples were processed within 6 h of collection to minimize bacterial cell replication and death. The fecal coliform densities were measured by the membrane filtration method (mFC) described by Eaton et al. (1995).

2.4. Statistical analyses

Changes in land cover and impervious cover between 1992 and 1999 were examined using a paired *t*-test and/or a Wilcoxon signed rank test depending on the assumptions of normality. Regression analysis (SAS, 1989) was performed to determine the relationship between measures of water quality, sediment quality, and biological condition and impervious cover (i.e., a hypothesized integrative measure of the degree of watershed development and condition). The amount of impervious cover was selected as the integrative measure because it was not associated with watershed size, creek length, or other measured creek attributes and is reported to be a good indicator of watershed condition in freshwater environments (Schueler, 1994; Arnold and Gibbons, 1996). Parameters that were transformed for analysis are identified in Table 3. A Wilcoxon signed rank test was used to determine if the within-season variability was significant. An alpha of 0.05 was used for all statistical tests. We did not attempt to develop multivariate linear and nonlinear models of the tidal creek responses to changes in the degree and type of watershed development. Such complex analyses were beyond the scope of this publication which had the objective of defining first-order linkages between watershed condition and tidal creek environmental condition.

3. Results

3.1. Land use and creek classification

Drainage basins of the study creeks ranged in size from 13 to 524 ha (Table 1). The watersheds of developed creeks were slightly larger (range = 69–434 ha, mean size = 216 ha) than those for the forested/reference creeks (range = 13–524 ha, mean size = 160 ha). The watersheds of reference creeks were comprised mostly of forest and salt marsh land cover (Fig. 2A). The mean suburban/urban land cover in the forested watersheds was less than 1.0% and 2.5% in 1992 and 1999, respectively, for creeks evaluated in both years. The mean suburban/urban land cover in the developed creek watersheds was 62.9% and 67.0% in 1992 and 1999, respectively, for creeks evaluated in both years. Changes in land cover for individual creeks and watershed classes between 1992 and 1999 were generally small (Fig. 2A).

Forested watersheds consisted of little or no impervious cover (range = 0–6%, mean = 2%) in 1999. Impervious cover ranged from 16% to 44% (mean = 30%) for suburban and 54% to 85% (mean = 67%) for urban/industrial watersheds in 1999 (Fig. 2B). For the test watersheds evaluated in both years, impervious cover increased by about 8% between 1992 and 1999. Furthermore, suburban and urban/industrial watersheds had significantly more impervious cover in 1999 than in 1992 ($p = 0.0035$ and $p = 0.0001$,

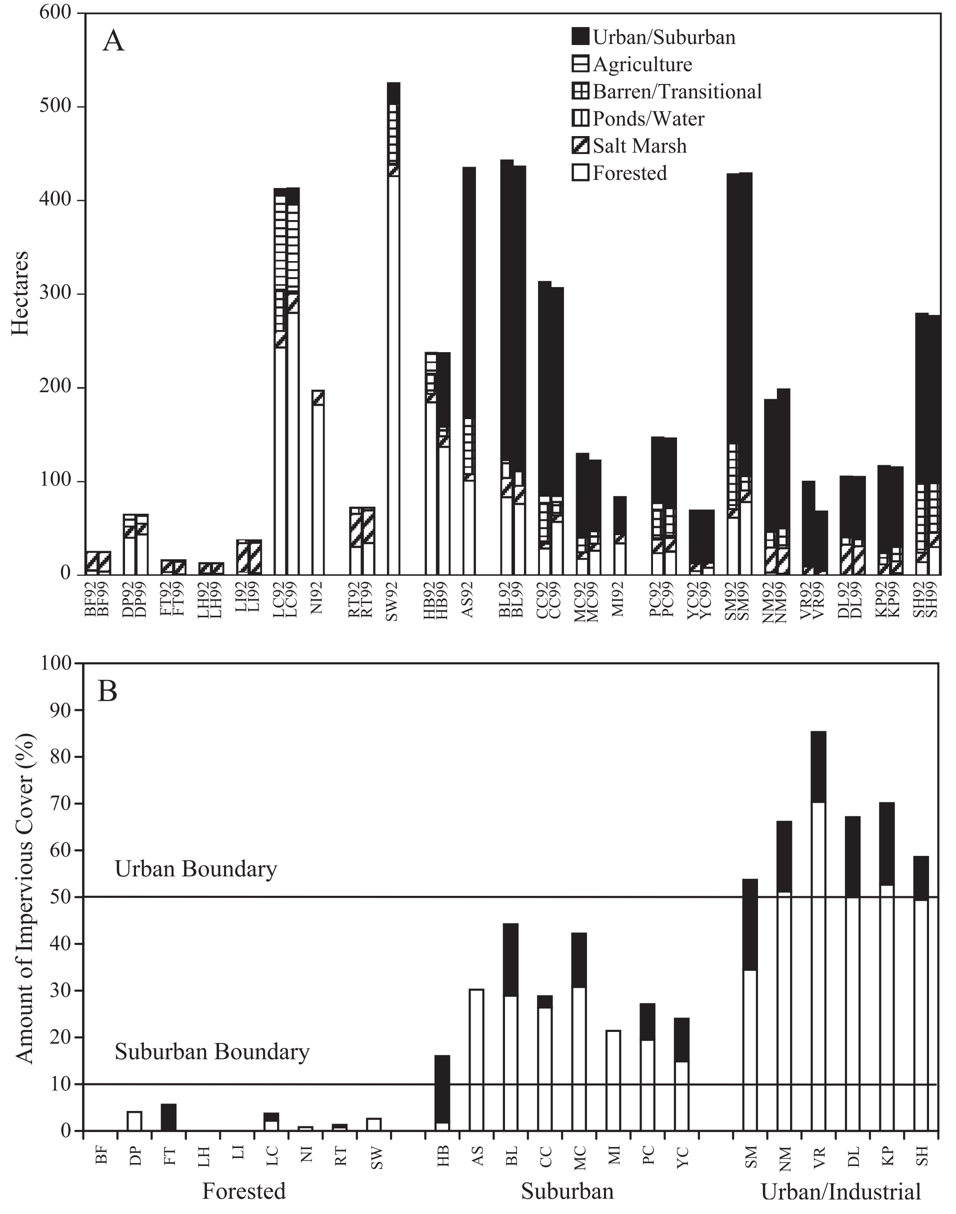

Fig. 2. Summary of land cover and watershed size for each creek based upon 1992 and 1999 aerial photographs (A). The amount of impervious cover in the watersheds for the 1992 aerial photographs (white) and the increase in impervious cover between 1992 and 1999 (shaded) (B). The creek abbreviation (see Table 1) and photograph year are presented along the *x*-axis.

respectively). Impervious cover increased slightly for forested watersheds between 1992 and 1999 (Fig. 2B). The increase in impervious cover in Horlbeck (from 2% to 16%) and Shem (from 35% to 54%) creeks between 1992 and 1999 warranted a change in watershed classification for these creeks (Fig. 2B). Several other reference creeks (i.e., Foster and Rathall creeks) will likely be re-classified as suburban creeks within the next few years.

Human population density was significantly associated with the amount of impervious cover using both 1990 and 2000, suggesting impervious cover is a reasonable measure of the impact of population growth on watershed characteristics (Table 3; Fig. 3A). Because industrial watersheds had high impervious cover values and low population densities, data from these watersheds tended to reduce the strength of the relationship between impervious cover and human population density. The relationship between impervious cover and population density was much stronger ($r^2 = 0.89$, $p < 0.0001$) when only data for reference, suburban, and urban watersheds were used for the analysis.

3.2. Physical/chemical indicators

A number of parameters were evaluated to determine if creek water quality was related to the degree of watershed development as represented by impervious cover. These parameters included the mean temperature, mean pH, mean salinity, and mean DO as well as salinity range (maximum minus minimum) and the percent of time the DO was less than 28% saturation.

Water temperature ranged from 23.0 to 37.2 °C (mean = 28.5 °C) during the summer and from 4.1 to 19.4 °C (mean = 8.9 °C) during the winter. The mean water temperature was not associated with the amount of impervious cover (Table 3) which is in contrast to studies of freshwater streams where strong relationships between stream temperature and the amount of impervious cover in the watershed have been found (e.g., Galli, 1991). The mean creek pH was 7.2 and 7.6 for summer and winter collections, respectively, and was not associated with the amount of impervious cover. The range in creek pH over the 48-h sampling periods was from 6.6 to 8.2. These values are typical of southeastern headwater tidal creeks.

Mean creek salinities ranged from 2.2 to 34.4 psu and each development class included two or more creeks representing mesohaline, polyhaline, and euhaline environments. Creek salinities were on average 2 psu higher in winter 2000 (drought conditions) compared to summer 1994 (high rainfall period). The mean creek salinity was not associated with the amount of impervious cover (Table 3). The salinity range in summer 1994 data was positively associated with the amount of impervious cover (Table 3; Fig. 3B), suggesting that as the degree of development increased so did the frequency and magnitude of salinity fluctuations. Salinity range also tended to increase with increasing impervious cover for the 1994 and 1995 summer validation data sets, suggesting the pattern was consistent across multiple sample events and summers. During winter 2000, salinity range was not associated with the amount of impervious cover (Table 3; Fig. 3B). The winter 2000 salinity range in forested creeks was nearly double the value observed in the summer studies (~ 12 vs. 6 psu). The summer to winter differences in salinity range were probably related to seasonal differences in the magnitude of evapotranspiration between warm, dry and cold, wet seasons.

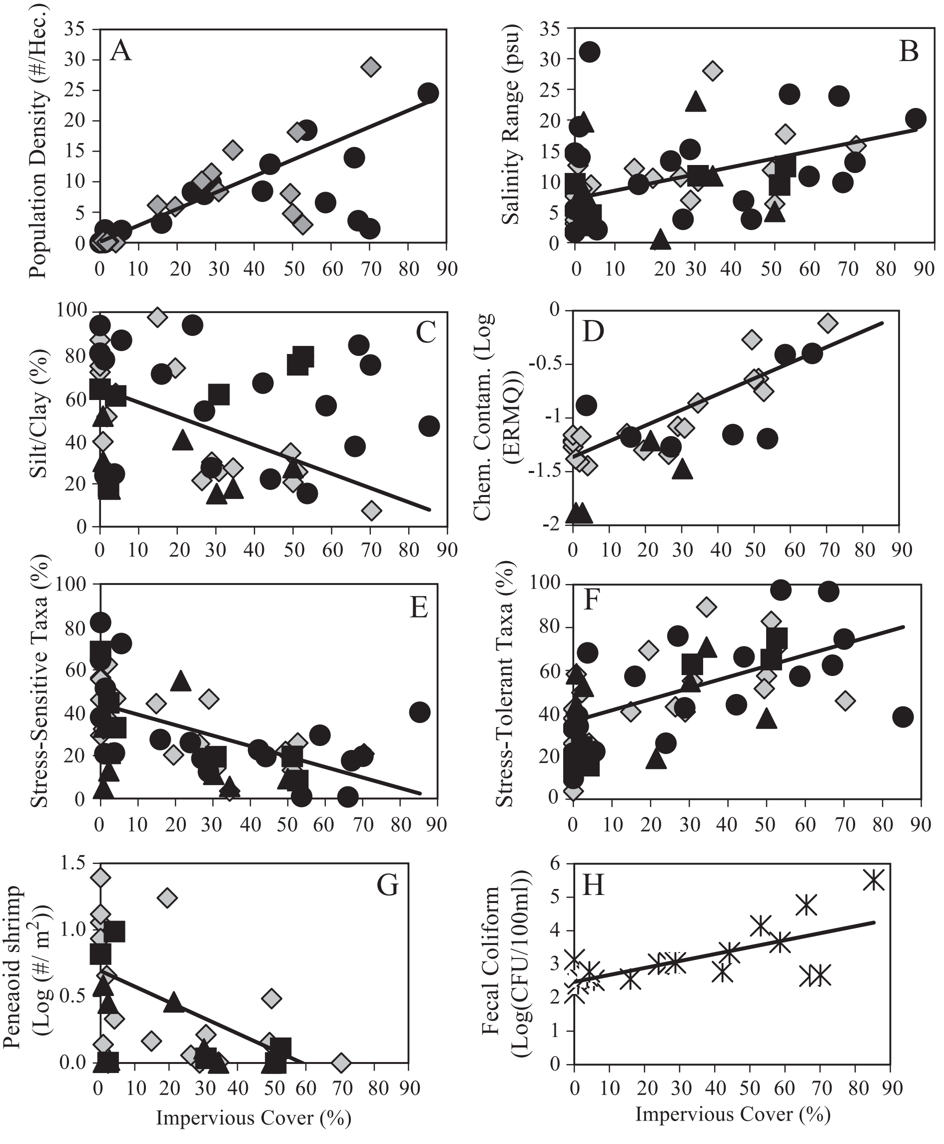

Fig. 3. Comparison of the relationship between eight parameters evaluated in this study and impervious cover (%) for the 1994 summer calibration (diamonds), 1994 summer validation (squares), 1995 summer validation (triangles), 2000 winter calibration (circles), and 2002 summer fecal coliform calibration (asterisks). The line indicates the linear regression for the summer calibration data sets.

Mean DO values were 51.3% saturation during the summer and 88.9% saturation during the winter, with DO levels ranging from 0% saturation to 200% saturation over the 48-h sampling periods. Hypoxic conditions (DO values $<28\%$ saturation, a level at which Diaz and Rosenberg, 1995 report biological effects are frequently observed) were found during the summer in forested and developed watersheds. Hypoxia was only observed in one creek in the winter. The mean DO and the percent of time DO values were below 28% (a critical low value) were not associated with the amount of impervious cover (Table 3).

The tidal creeks sampled represented all major sediment types (sand, mixed, and silt/clay sediments). The silt/clay content of sediments in summer 1994 was negatively associated with the amount of impervious cover (Table 3; Fig. 3C). This general pattern also occurred in the 1995 summer validation and winter 2000 calibration studies but to a lesser degree. The sediment silt/clay content in the summer 1994 validation study was not related to the amount of impervious cover (Table 3; Fig. 3C). When all of the data collected for sediment characteristics were combined, there was not a significant relationship between the silt/clay content of the sediments and the amount of impervious cover.

Sanger et al. (1999a,b) provided a detailed assessment of the association between the degree and type of landscape development and the concentrations of chemical contaminants in creek sediments. They also provided a detailed analysis of among creek differences and similarities in sediment contaminants for the creeks. In general, this study found sediment trace metal sediment concentrations commonly associated with urban and industrial sources, including Cu, Cr, Pb, Zn, Cd, and Hg, were 2–10 times higher in creeks located in urban and industrial watersheds. Sediment trace metal levels were similar for creeks located in suburban and forested watersheds. Creeks located in industrial/urban watersheds also had significantly higher concentrations of PAHs, PCBs, and DDT compared to creeks located in suburban and forested watersheds. Concentrations of about half of the PAH analytes and the total PCBs were significantly higher in suburban creeks compared to the forested reference creeks.

The mean effects range medium quotient (ERMQ) is an approach developed by Long et al. (1998) to integrate chemical contaminant data by assessing cumulative contaminant loading based on normalizing the chemical concentrations to the level of probable biological effects. The ERMQ for creek sediments during summer 1995 was positively associated ($r^2 = 0.78$, $p < 0.0001$) with the amount of impervious cover (Table 3; Fig. 3D). In the limited regional sampling conducted in summer 1995 ($n = 4$) and winter 2000 ($n = 8$), a positive relationship between the mean ERMQ and impervious cover was also found. A positive relationship ($r^2 = 0.59$, $p < 0.0001$) was found between the mean ERMQ and impervious cover when all of the information available for sediment contaminants were combined (Fig. 3D).

Sediment biological activity as assessed by porewater ammonia and benthic algal biomass as assessed by sediment chlorophyll *a* were not associated with the amount of impervious cover in the associated watersheds (Table 3). Values for both of these measures, however, increased with increasing levels of development (Fig. 4).

3.3. Biological

The relative abundance of stress-sensitive macrobenthic taxa in the upper and lower creek reaches in summer 1994 were negatively associated ($r^2 = 0.46$, $p = 0.0013$) with the amount of impervious cover (Table 3; Fig. 3E). A similar pattern was found for the 1994 and 1995 summer validation studies. Only the upper reaches of creeks were sampled in the

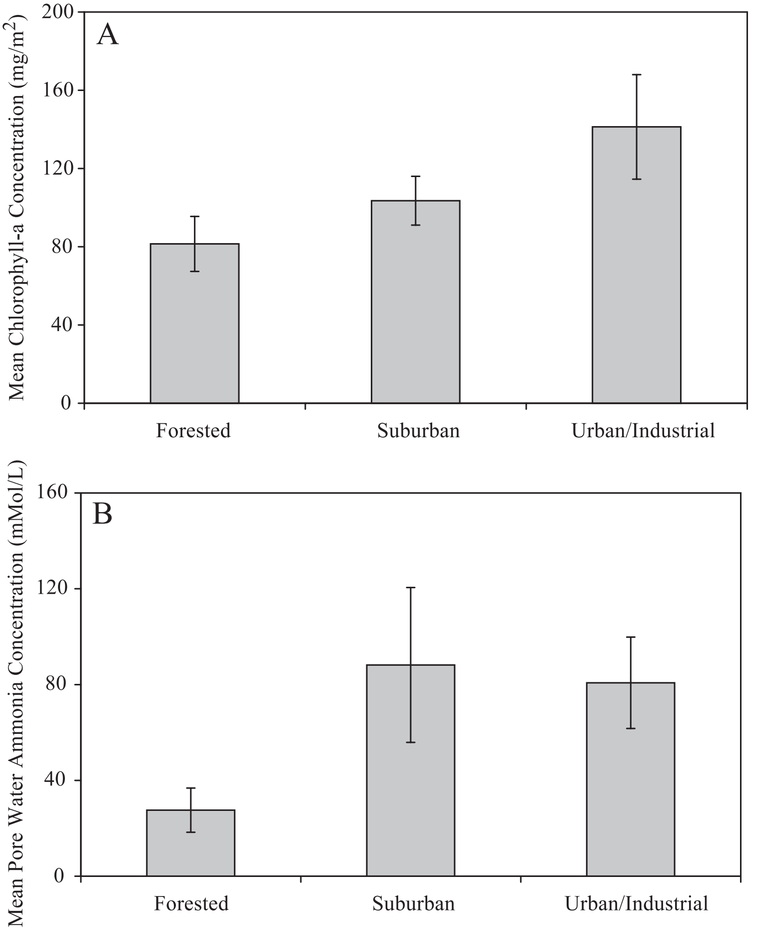

Fig. 4. The mean surface sediment chlorophyll *a* (A) and pore water ammonia (B) concentrations in 2000 winter calibration study for each watershed type. Error bars represent ± 1 standard error.

winter 2000 calibration study, and for this sampling effort, the relative abundance of stress-sensitive taxa was also negatively associated ($r^2 = 0.28$, $p = 0.0186$) with the amount of impervious cover. To determine if the observed declines in the relative abundance of stress-sensitive taxa with increasing amounts of impervious cover were concordant over the length of the creek, only the data collected in upper creek reaches in summer 1994 were regressed against the amount of impervious cover (Table 3). The r^2 (0.44), p-value (0.0020), and slope (-0.82) for this regression were similar to values reported above for the upper and lower reaches combined. When all of the data for stress-sensitive taxa were included in a single analysis, a negative relationship was found between the relative abundance of stress-sensitive taxa and the amount of impervious cover.

The relative abundance of stress-tolerant macrobenthic taxa in the upper and lower reaches of the 1994 calibration and validation studies were positively associated ($r^2 = 0.32$,

$p=0.0117$; $r^2=0.93$, $p=0.0020$, respectively) with the amount of impervious cover (Table 3; Fig. 3F). Similar patterns were found when only samples from the upper reaches were included in the analysis and for the summer 1995 validation study. The relative abundance of stress-tolerant taxa in the winter 2000 samples were also positively associated ($r^2=0.33$, $p=0.01$) with the amount of impervious cover (Table 3; Fig. 3F). When all of the samples were combined, a positive relationship ($r^2=0.34$, $p<0.0001$) was found between the relative abundance of stress-tolerant taxa and the amount of impervious cover.

Fundulus heteroclitus (mummichogs) and *Palaemonetes* spp. (grass shrimp including *Palaemonetes pugio*, *Palaemonetes intermediatus*, and *Palaemonetes vulgaris*, but predominately *P. pugio*) are the preferred prey of many recreationally important fish including juvenile *Sciaenops ocellata* (red drum), *Cynoscion nebulosus* (spotted sea trout), and *Paralichthys lethostigma* (southern flounder) and were found to be the numerically dominant organisms collected from sample creeks during summer. Penaeoid species (common commercial shrimp including *Litopenaeus aztecus* (brown), *Farfantepenaeus setiferus* (white), and *Farfantepenaeus duorarum* (pink), but predominately *F. setiferus*) and *Leiostomus xanthurus* (spot) were the numerically dominant recreationally important living resources collected. No winter seine samples were collected because relatively few fish and crustaceans occur in tidal creek habitats in colder months (e.g., Hoffman, 1991).

The abundances of *F. heteroclitus*, *L. xanthurus*, and *Palaemonetes* spp. were not associated with the amount of impervious cover for the 1994 calibration or validation studies (Table 3). However, the abundance of all three taxa appeared to decline as the amount of impervious cover increased (Table 3). When the data for all summer sampling events were combined, the abundance of *L. xanthurus* ($r^2=0.12$, $p=0.0464$) and *Palaemonetes* spp. ($r^2=0.13$, $p=0.0366$) were negatively associated with the amount of impervious cover. No relationship was found between *F. heteroclitus* abundance and the impervious cover for the combined samples.

Penaeoid species abundance was negatively associated ($r^2=0.31$, $p=0.0134$) with the amount of impervious cover for the summer 1994 calibration study (Table 3; Fig. 3G). Penaeoid abundance also declined in the summer 1994 validation and summer 1995 validation studies with increasing levels of impervious cover. When the data from all summer sampling events were combined, the abundance of Penaeoid species was negatively associated ($r^2=0.26$, $p=0.0023$) with the amount of impervious cover (Fig. 3G).

3.4. Fecal coliform bacteria

The average concentration of fecal coliform bacteria at low tide was 2448, 1060, 134,333, and 1806 CFU/100 ml for the creeks located in forested, suburban, urban, and industrial watersheds, respectively, and was positively associated ($r^2=0.35$, $p=0.0081$) with the amount of impervious cover (Fig. 3H). The concentration of fecal coliform bacteria from Long Creek, a reference creek, was 12,200 CFU/100 ml, approximately three times higher than other reference creeks. Long Creek has a large (approximately 70 ha) freshwater swamp in its headwaters and the sampling event for this creek occurred following several large rainfall events. Long Creek was sampled a second time a few weeks later to validate the unusually high bacterial loading initially observed and to

Table 4
Results of the paired Wilcoxon rank sum test to determine if a difference occurred between the 1994 summer calibration and validation samples

Parameter	*t*-statistic	*p*
Temperature	2.5	0.6875
pH	9	0.0938
Salinity	− 0.5	1
Salinity range	3.5	0.4375
Dissolved oxygen (% saturation)	− 6.5	0.2188
Percent of time DO < 28% sat.	5.5	0.3125
Silt/clay	− 1.5	0.8125
Stress-sensitive taxa (upper reach)	1.5	0.8438
Stress-tolerant taxa (upper reach)	− 0.5	1
L. xanthurus	1	0.8750
F. heteroclitus	3.5	0.5625
Palaemonetes spp.	0.5	1
Penaeoid shrimp	− 0.5	1

evaluate if the fecal coliform bacteria present were of animal or human origin. A Multiple Antibiotic Resistance Test (MAR) was used to determine the source of the bacteria (Parveen et al., 1997). The fecal coliform concentration for the second sampling in Long Creek was 440 CFU/100 ml, a value that was similar to the concentrations observed in other forested creeks. The *Escherichia coli* bacteria which were present in Long Creek during the second sampling exhibited little resistance to commonly used antibiotics suggesting that the source of these bacteria was not human but probably wildlife (Gooch, personal communication). When the count data for the second sampling event for Long Creek were used for analyses a stronger relationship between the concentration of fecal coliform bacteria and impervious cover was found ($r^2 = 0.48$, $p = 0.0010$) (Table 3; Fig. 3H).

3.5. Within-season variation

The parameters which were evaluated to determine the magnitude of within-season variance that occurred over the ∼75 day summer 1994 sampling period included the silt/clay content of the sediment, mean water temperature, mean pH, mean salinity, salinity range, mean DO concentration (% saturation), the amount of time DO was less than 28% saturation, the relative abundance of stress-sensitive and -tolerant macrobentic taxa, and the abundance of the numerically dominant fish and crustaceans. A Wilcoxon rank sum test found no within-season difference for any of these parameters (Table 4).

4. Discussion

4.1. Conceptual model of tidal creek responses to watershed development

Fig. 5 identifies the linkages defined by this study among watershed-scale stressors, physical and chemical exposures, and biological responses of tidal creeks to watershed

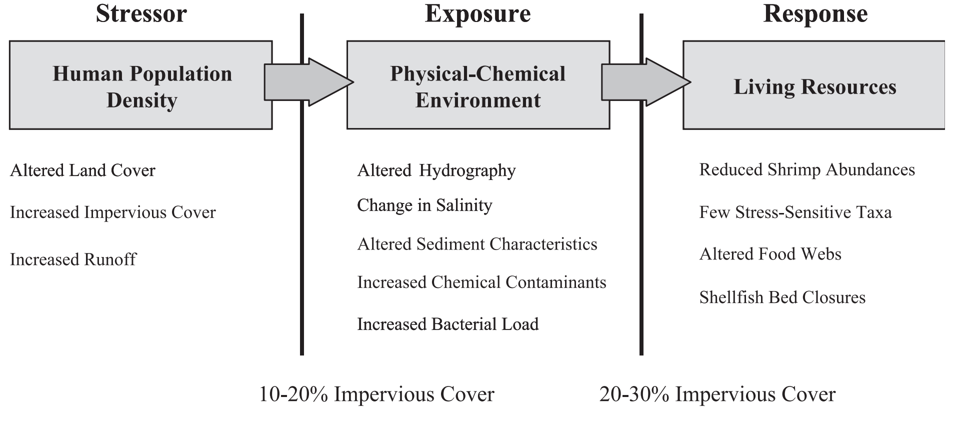

Fig. 5. Summary of the conceptual model of linkages developed for headwater tidal creeks.

development. This conceptual model provides a visual representation of the manner in which human population growth is linked to changes in the physiochemical environment and the kinds and abundances of biological resources. Most importantly, however, the model provides a framework for assessing the risks of watershed development on tidal creek ecosystems. The information provided by the conceptual model is critical for defining management actions that reduce impacts on ecological processes and for designing monitoring programs to evaluate the effectiveness of management actions. The conceptual model represents a first-order synthesis of the data we collected. In the future, multivariate analyses and empirical modeling using artificial neural nets will undoubtedly result in additional insights into causative factors and interactions among watershed attributes and environmental conditions in tidal creeks.

4.1.1. Stressors

Coastal population growth and changes in population density are the major stressors identified for tidal creeks (Fig. 5) and are generally considered to be the greatest threats to the ecological functioning of estuarine environments (Culliton, 1998; Beach, 2002). The coastal zone is a narrow corridor that comprises less than 13% of the nation's land but contains more than half of the human population potentially magnifying the effects of nonpoint source runoff on receiving water bodies (as reviewed by Beach, 2002). Major pollution problems that have been identified for the coastal zone include eutrophication and associated hypoxia, chemical and microbial contamination, nonindigenous species introductions, and habitat modification (Diaz and Rosenberg, 1995; USEPA, 1999, 2001; NRC, 2000).

Over the next decade, the population density of U.S. coastal counties is projected to increase by 18%, from about 1.0 to 1.2 individuals/ha (Culliton, 1998). As Beach (2002) notes, "Coastal population growth is not the whole story, however. Runaway land consumption, dysfunctional suburban development patterns, and exponential growth of automobile use are the real engines of pollution and habitat degradation on the coast." The rate at which forested and agricultural systems are being converted into low density, pedestrian unfriendly suburban and urban uses is occurring at twice the rate the human population is growing (Beach, 2002). For example, the urban area of Charleston, SC, increased by 250% between 1973 and 1994, but the human population only increased by 40% (Allen and Lu, 2000).

As forest ecosystems are urbanized, the amount of the landscape that is impervious increases from a few percent in low density suburban development to values approaching 90% in urban centers (Soil Conservation Service, 1975; Arnold and Gibbons, 1996). Typical examples of impervious cover include roads, parking lots, roofs, and sidewalks. Impervious cover increases the volume and rate of runoff adversely altering the water cycle and creek hydrology, including rapid rises and declines in stream flow and decreased infiltration into groundwater (Arnold et al., 1982; Arnold and Gibbons, 1996; Schueler and Holland, 2000). In a typical forested ecosystem, approximately 40% of the runoff is returned to the atmosphere by evapotranspiration and approximately 50% infiltrates into the soil with the remaining 10% returned to receiving waters via surface runoff (e.g., Dunne and Leopold, 1978; Harbor, 1994; Arnold and Gibbons, 1996). Impervious cover for the forested watersheds sampled for this study was about 1.7% and we estimate approximately 5% of the rainfall on the watersheds was converted into runoff using a standard curve developed by Schueler (1994) for over 40 watersheds. Impervious cover for the developed watersheds sampled ranged from 16% to 85% and we estimate approximately 15–75% of the rainfall on these watersheds was converted into runoff. These data suggest that for rainfall events of similar magnitude the volume of runoff was 3–25 times greater in the developed watersheds studied than in the forested watersheds.

In the period covered by this study, impervious cover within many of the watersheds increased even though the population density in the suburban and urban/industrial watersheds decreased. The increase in impervious cover in forested, suburban, and urban/industrial watersheds was 1.7%, 8.1%, and 12.0%, respectively. Most of these increases were primarily the result of infrastructure (e.g., roads) and housing expansion into forested and agricultural habitat. The average population density decreased in the urban/industrial watershed from 14.3 to 11.6 individuals/ha. Decreases in the human population density in urban and industrial watersheds were mainly associated with urban renewal projects. Relatively small human population increases occurred in the suburban and forested watersheds.

4.1.2. Impacts of increased impervious cover on pollution exposure

The major physical and chemical attributes of tidal creeks used to measure pollution exposure were water quality parameters, sediment contaminant levels and physical characteristics, benthic algae abundance, porewater ammonia concentration, and fecal coliform bacteria levels (Fig. 5). We did not directly measure the changes in freshwater flow for the tidal creeks studied, and therefore could not directly evaluate the relationship between runoff and the amount of impervious cover in the watershed. The increased salinity range that occurred in developed creeks during summer 1994, however, appeared to provide an indirect measure of the alterations to freshwater inflow associated with increased levels of impervious cover (Lerberg et al., 2000). The lack of differences in salinity range between reference and developed creeks in winter was probably related to the low transpiration rates that characterize forested creeks during cold periods. The South

Carolina coastal zone was historically underwater in geological time. This resulted in the coastal zone having naturally high sandy soils which appear to be transported from the sandy upland into the muddy tidal creeks. In addition, the creek channel in developed creeks were also composed of coarser sediments (i.e., greater amounts of sand particles) than similar habitats in forested creeks. In southeastern coastal watersheds, we hypothesize that increases in imperviousness result in increases in the volume and flashiness (i.e., erratic and rapid inputs) of runoff which leads to the naturally sandy soils being transported into the headwaters of tidal creeks. Salinity range and upland erosion appear to increase when impervious cover levels exceeded 10–20%.

Benthic algal biomass tended to be higher in creeks draining suburban and urban/industrial watersheds. Porewater ammonia levels were also higher in suburban and urban/industrial creeks, suggesting increased microbial activity in the sediments of creeks draining developed watersheds. These patterns suggest increased loading of conventional pollutants may have been associated with the transition from forested to suburban categories. Others researchers have documented increases in the nonpoint source loadings of conventional pollutants, particularly dissolved nitrogen, as coastal watersheds are developed (e.g., Vernberg et al., 1996; Mallin et al., 2000).

The degree of watershed development and the amount of impervious cover was strongly related to the degree of sediment contamination as defined by the mean ERMQ. Increased sediment contamination appears to begin during suburban development (i.e., when impervious cover exceeds 10–20%). There appears to be a substantial increase in the degree of contamination when impervious cover exceeds 40%, which is probably related to urban population densities and associated industrial land uses. The greatest sediment contamination occurred in the upper reaches of urban/industrial creeks which have a long history of pollutant inputs and are repositories for chemical pollutants. Specifically, urban/industrial watersheds contained greater amounts of organic (PAHs, PCBs, and DDT) and trace metal (Cu, Cr, Pb, Zn, Cd, and Hg) contaminants than other creek classes (Sanger et al., 1999a,b). Chemical contaminant levels in suburban creeks were only slightly higher than the levels in forested creeks (Sanger, 1998; Sanger et al., 1999a,b). We hypothesize that the increased flashiness that characterizes the hydroperiod of developed creeks transported contaminants into adjacent estuarine environments through the tidal creek system. Sediment chemical contaminant levels did not change appreciably between 1995 and 2000, suggesting that they are relatively stable over time scales of 5 years.

Improperly treated human or animal wastes are the primary concerns associated with fecal coliform contamination (Mallin et al., 2000; Van Dolah et al., 2000). Pathogens associated with fecal coliform contamination are transmitted to humans through consumption of filter feeding shellfish and water contact recreation. Seventeen of the eighteen creeks sampled did not meet the state fecal coliform standard for water contact recreation, and none of the 18 creeks met the state standard for shellfish harvesting (SCDHEC, 1998). Bacterial source tracking or typing has been conducted for only one creek. Van Dolah et al. (2000), however, found that pet and animal wastes as well as septic tank failures appear to be the major sources of fecal coliform bacteria in forested and suburban watersheds. Two of the urban/industrial creeks, New Market and Vardell creeks, were characterized by fecal coliform counts (i.e., 59,000 and 330,000 CFU/100 ml, respectively) that suggest substantial anthropogenic sources (i.e., raw sewage). If the reference and suburban creeks are used to develop a regression equation [$FC = 22.3(\%ImpSur) + 414$; $r^2 = 0.38$, $p = 0.0323$] to compare to the equation Mallin et al. (2001) developed [$FC = 5.4(\%ImpSur) - 29$], then the slope of this studies regression is approximately four times steeper than the Mallin et al. (2001) equation. This difference may be because the Mallin et al. (2001) equation was developed using fecal coliform values collected at high tide while our data were collected at low tide. Our samples also represent a broader range of impervious cover levels (1–85%) compared to Mallin et al. (2001) levels (12–33%). At higher impervious cover levels, the relationship between impervious cover and fecal coliform concentration may be curvilinear. Major changes in fecal coliform contamination occurred when impervious cover levels exceeded 30–50%.

4.1.3. Impacts of increased impervious cover on biological resources

As others describe, oligochaetes and polychaetes were the numerically dominant macrobenthic organisms in tidal creeks (e.g., Lerberg et al., 2000; Gawle, 2002). Increases in the amount of impervious cover from watershed development were associated with changes in the relative abundance of the specific taxa in summer and winter. The relative abundance of stress-tolerant species, particularly the oligochaetes *M. rubroniveus* and *P. litoralis*, increased as the amount of impervious cover increased. The relative abundance of stress-sensitive species, particularly the polychaete *S. benedicti*, decreased as impervious cover increased.

The high densities (~ 50,000 individuals/m^2) of *M. rubroniveus* and *P. litoralis* in urban/industrial creeks in winter suggest that these organisms potentially thrive in disturbed environments including chemically contaminated sediments. Oligochaetes are frequently resistant to organic and chemical contamination and often maintain high standing stocks in disturbed habitats (Pearson and Rosenberg, 1978; Chapman et al., 1982; Giere and Pfannkuche, 1982; Chapman and Brinkhurst, 1984; Sarda et al., 1996). Weinstein et al. (2003) report that *M. rubroniveus* is tolerant of PAHs in concentrations an order of magnitude greater than those that were observed in the urban/industrial creeks. Research on a closely related Pacific-coast species, *Monopylephorus cuticulatus*, found that this species was tolerant to anoxia and changes in pH, temperature, trace metal contamination, sewage sludge, and pulp mill effluents (Chapman et al., 1982). *P. litoralis* is infrequently mentioned as a species typical of chemically contaminated sediments. A few studies, however, report *P. litoralis* to be numerically dominant in organic enriched environments (Gray, 1976; Wiltse et al., 1984; Sarda et al., 1996).

Previous research has generally classified *S. benedicti* as tolerant to stress (Levin, 1984; Weisberg et al., 1990; Engle et al., 1994; Van Dolah et al., 1999, 2000), and as a result, our classification of this species into the stress-sensitive category may be controversial. The response of *S. benedicti* was, however, consistent over seasons and years and indicates that in the southeastern U.S. intertidal creek habitat *S. benedicti* responds as a stress-sensitive taxon. Some of the recent work on this species indicates that *S. benedicti* is not as tolerant of chronic pollution exposure or hypoxia as many oligochaete species (Llanso, 1991, 1992; Sarda et al., 1996; Lerberg et al., 2000, Weinstein, unpublished data). Also, recent bioassays suggest it is much less tolerant of exposure to PAHs than the oligochaete

M. rubroniveus (Weinstein and Sanger, 2003). The high densities (~ 7000 individuals/m^2) of *S. benedicti* within urban/industrial creeks during our winter sampling suggest this species is capable of tolerating the contaminant stress characteristic of these habitats when it is not simultaneously exposed to compounding stressors (e.g., low DO levels, high temperatures, H_2S, etc.).

The shallow tidal creeks we sampled functioned as nursery habitat for many species of crustaceans and fish throughout the summer period, supporting previous findings (Cain and Dean, 1976; Hackney et al., 1976; Shenker and Dean, 1979; Weinstein, 1979; Kneib, 1997). Highest densities were observed in the upper most reaches of creeks suggesting these shallow, physiologically stressful habitats provide a refuge to juvenile organisms especially at stressful low tide periods when large numbers are concentrated in the creek channel and tidal pools. At low tide during summer, these creeks frequently experienced water temperatures exceeding 30 °C, highly variable salinities and hypoxia. These conditions exceed the tolerances of most large finfish and crustacean predators. We hypothesize that the stressful water quality and shallow depth that occur in headwater tidal creeks provides a barrier to larger predators.

Nursery habitat and function was severely degraded in urban and industrial creeks. Tidal creeks draining suburban development supported nursery habitat function although at reduced capacity, particularly in creeks with high levels of suburban development. The abundance of economically important Penaeoid species and bottom feeding fish, *L. xanthurus*, decreased as the amount of impervious cover increased with lowest densities occurring in urban and industrial creeks. Given the mobility of these species over tidal and diel cycles, these differences were surprising. No relationship was found between the amount of impervious cover and the abundance of the numerically dominant resident crustaceans (*Palaeomonetes* spp.) and fish, *F. heteroclitus*. This finding suggests some resident nekton may have adapted to the stressful conditions which exist in urban and industrial creeks.

Accepted biological criteria or threshold values have not been developed for determining when tidal creek macrobenthic and nekton assemblages are degraded. In addition, traditional macrobenthic and nekton community-level attributes including species richness, dominance, and diversity were not useful in discriminating between forested and developed watersheds (Lerberg et al., 2000; Gawle, 2002). We, therefore, classified sample sites using the stress-sensitive and -tolerant macrobenthic metrics and the relative abundance of Penaeoid species into the following categories: (1) good condition represented by values that were within one standard error of the mean condition in undeveloped forested watersheds; (2) impacted condition represented by values that were between one and two standard errors of the mean condition in undeveloped forested watersheds; and (3) degraded condition represented by values that were greater than two standard errors of the mean condition in undeveloped forested watersheds (Fig. 6). About 80% of the samples from forested watersheds represented good biological condition using these metrics. Sample sites from the North Inlet National Estuarine Research Reserve were consistently near the mean value for sites in the good category. Biological degradation generally occurred at sites where impervious cover exceeded 20–30%.

Establishing a two standard error threshold for biological degradation to tidal creek biological resources is a stringent criterion. This, in effect, amounts to requiring a

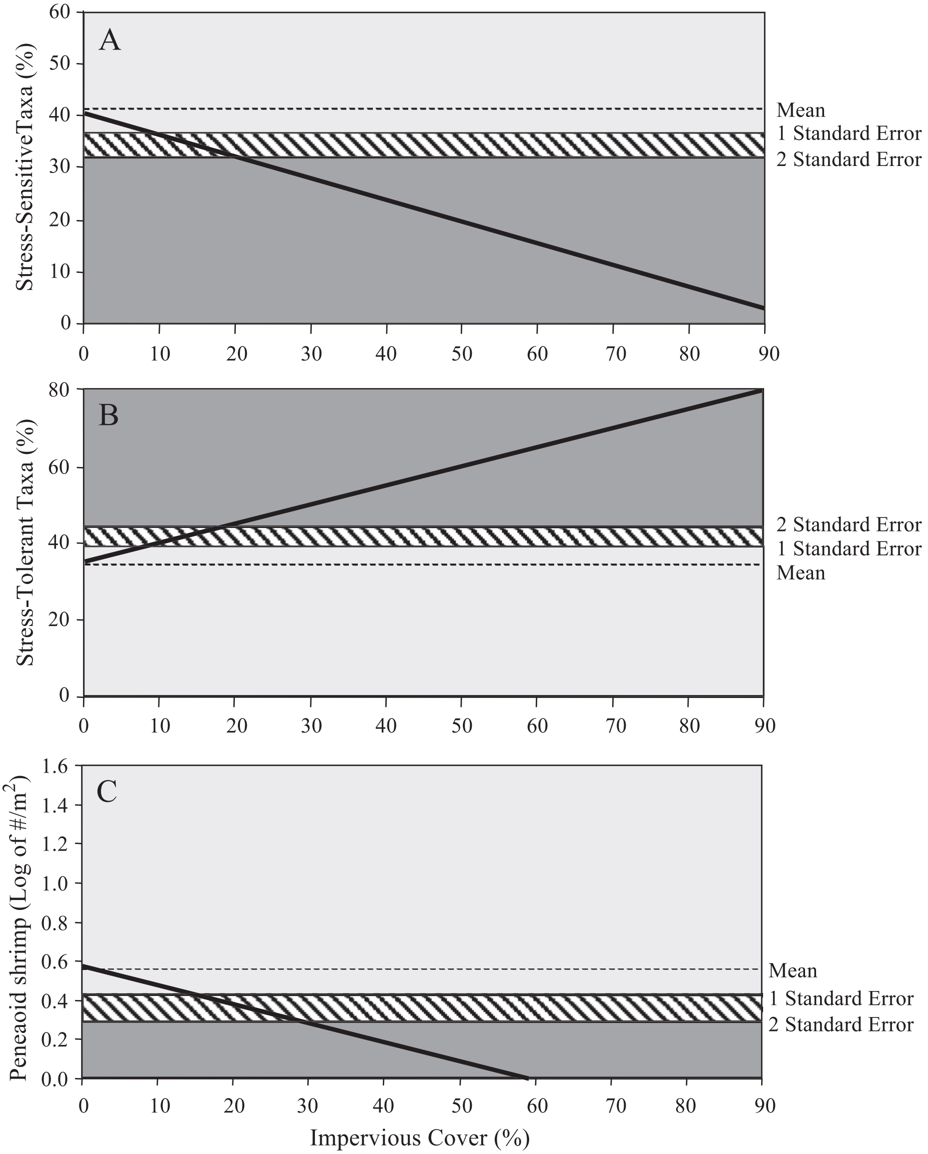

Fig. 6. Diagram of the stress-sensitive (A) and stress-tolerant (B) macrobenthic metrics and the relative abundance of Penaeoid species (C) into three categories based upon the mean of the forested watersheds: (1) good condition (light gray) represented by values that were within one standard error; (2) impacted condition (striped) represented by values that were between one and two standard errors; and (3) degraded condition (dark gray) represented by values that were greater than two standard errors. The regression line for all of the data is also provided.

statistically significant difference before the level of impact is considered to be degraded. We, however, felt this stringent threshold was justified given the stress-tolerant nature of most of the biota inhabiting tidal creeks. In the future, it may be possible to combine the two macrobenthic metrics with the information for the relative abundance of Penaeoid species as well as other information into an overall multi-metric index of biological integrity for tidal creeks that has greater sensitivity and less uncertainty (e.g., Karr, 1991; Weisberg et al., 1997).

4.2. Impervious cover as a watershed-scale indicator

Impervious cover is a physical attribute of the landscape that provides a reliable indicator of the cumulative impacts of watershed development on receiving water bodies and aquatic habitats, including tidal creeks (Schueler, 1994; Arnold and Gibbons, 1996; Mallin et al., 2000; Lerberg et al., 2000; this study). It is also related to the human population density and housing unit density (Stankowski, 1972; Arnold and Gibbons, 1996; Scott et al., 1998; this study). Increases in impervious cover adversely affect the processes linking upland and aquatic systems by increasing the flashiness of freshwater runoff which adversely affects the natural processing of pollutants, thereby providing a rapid transport system for delivering pollutants to receiving water bodies and decreasing freshwater infiltration into groundwater (Arnold and Gibbons, 1996). A range of methods are available for estimating the imperviousness of a watershed including: (1) point-sampling from aerial photographs (this study), (2) direct survey techniques, (3) census of impervious cover using GIS, and (4) remote sensing of land cover and applying literature values of imperviousness to the various land cover categories represented (Arnold and Gibbons, 1996). Future studies linking watershed process and aquatic ecosystem function would benefit from collecting impervious cover information. The uncertainty and comparability of estimates of imperviousness obtained by the above methods is unknown and likely varies.

All types of impervious cover do not have equal impact on hydroperiod and volume of runoff (Schueler and Holland, 2000). For example, runoff from roofs is generally released over pervious areas (e.g., lawns) and does not enter the stormwater transport system. This provides additional infiltration and transpiration. A study of 11 watersheds in Olympia, WA, estimated that only about 40% of runoff from roofs was transported to streams as surface runoff (Arnold and Gibbons, 1996). From 90% to 100% of the rain that falls onto roads, sidewalks, and parking lots is rapidly directed into the stormwater transport systems and ultimately makes it into receiving water bodies.

The threshold values we defined for physical, chemical, and biological indicators of tidal creek health are consistent with the general response pattern for freshwater environments (e.g., Schueler, 1992, 1994; Arnold and Gibbons, 1996). When the level of imperviousness is less than 10–20%, physical, chemical, and biological attributes of tidal creeks are similar to those reported for relatively pristine conditions and natural ecological processes are able to sustain a high level of environmental quality over seasons and years. When the amount of impervious cover exceeds 10–20% physical, chemical, and biological processes are altered or impacted (Fig. 5). The relationships that were defined for impervious cover and environmental parameters by this study appear to apply over much of South Carolina. We do not know if these relationships are valid in other areas that have greater and lesser tidal height and flushing.

4.3. Solutions

Humans are moving, and will likely continue to move, to coastal environments for at least the next several decades (Culliton, 1998). Smart growth and new urbanism represent efforts to modify current development patterns (i.e., urban sprawl) in manners which creates diverse, livable communities that conserve the ecological services provided by estuarine habitats (Beach, 2002). Beach (2002) suggested, and we agree, that actions are required at three scales to conserve important coastal processes and habitats: regional, neighborhood, and local. Regional scale actions should focus on determining where new development should occur including identification of habitats and watersheds that need to be protected from increases in impervious cover. The tools of managing regional development are zoning, infrastructure planning, and land-protection programs. Zoning ordinances and land use plans are an important step for mitigating the environmental impacts of watershed development on tidal creek habitats (Holland, 2000). However, these ordinances only protect the land until the next land use plan is developed or until a new governing body is elected. The role of permanent protection offered by Purchase of Development Rights Programs and Land Trusts cannot be overemphasized. The goal at the regional scale must be to prevent sprawl from encroaching upon rural areas. At the neighborhood scale, actions must be implemented that return our neighborhoods to more traditional designs including mixed uses and street connectivity that reduce reliance upon personal automobiles. Site scale actions should focus on controlling the volume, rate and quality of runoff and effectively reducing the deleterious effects of impervious cover. Specific actions include: (1) limiting the degree of alteration to freshwater inflow (e.g., stormwater retention ponds, buffers, setbacks); (2) minimizing the creation of new impervious surfaces (e.g., gravel driveways, pervious parking areas, cisterns, cluster housing); (3) directing surface runoff into swales and vegetated areas to trap pollutants and increase infiltration; and (4) maintaining vegetated open spaces to decrease runoff (Holland, 2000). Public education efforts should probably target the local scale. Through education and community involvement, a conservation ethic may be fostered that encourages the permanent protection of lands for the services they provide.

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