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Effects of coastal development on nearshore estuarine nekton communities

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ABSTRACT: Coastal development affects estuarine resources by severing terrestrial–aquatic linkages, reducing shallow water habitats, and degrading ecosystem services, which is predicted to result in measurable declines in nekton community integrity. We assessed the effects of landscape features on nearshore habitats and biological communities, relating subtidal habitat, shoreline condition, upland land use and nearshore fish communities in a Chesapeake Bay tributary, the James River, Virginia. Both upland development and the placement of erosion control structures on the shoreline were associated with reduced fish community integrity, and shoreline alterations were linked with the amount of subtidal structural habitat in the nearshore. Ecological thresholds in nekton community integrity were evident at ≥23% developed land use within 200 and 1000 m buffer increments. Nekton assemblages at sites with low development (<23%) and natural or riprap shorelines were different from all other combinations of altered conditions (low development with bulkhead, and high development with riprap or bulkhead). Species composition along natural or riprap revetment shorelines with low upland development tended to be diverse and inclusive of tidal marsh species, while highly developed sites or bulkhead shorelines were dominated by a few generalist species. The complex interaction between watershed (both nearshore and inland) and shoreline development presents a unique challenge for coastal planning. Alternate moderating approaches for coastal development may include preservation of riparian buffers, the placement of living shorelines for erosion control where appropriate, and development of targeting tools to identify landscapes near an ecological threshold.

KEY WORDS: Coastal development · Fish · Nekton · Chesapeake Bay · Shallow water habitats · Ecological threshold · Shoreline alteration · Bulkhead · Biotic integrity

INTRODUCTION

Coastal plain estuarine systems are critical resource areas providing extensive economic and ecological services, such as essential spawning and nursery habitats for a variety of aquatic species, and fishing opportunities. However, these systems often exist under intense and increasing pressure from a varied set of uses and users without a directed comprehensive management plan. A prime example in the USA is the Chesapeake Bay watershed that incorporates parts of 6 states (Delaware, Maryland, New York, Pennsylvania, Virginia and West Virginia) and the heavily developed region of the District of Columbia, thus inheriting multiple regulatory procedures for managing coastal impacts.

Significant anthropogenic coastal stressors are shoreline and watershed developments that affect aquatic resources on a variety of levels and reduce ecosystem integrity, which is defined as 'the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region' (Karr & Dudley 1981). Shoreline development can directly affect local water quality and aquatic communities through the loss of intertidal habitat, changes in hydrology, increases in nutrient inputs, loss of allochthanous material, increased recreational use and a loss of natural erosion control. Shoreline hardening (generally related to upland development) affects benthic or

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interstitial invertebrate communities (Bilkovic et al. 2006a, Seitz et al. 2006), fish egg mortality (Rice 2006), predator abundances (Seitz et al. 2006) and fish community integrity (Beauchamp et al. 1994, Jennings et al. 1999, Bilkovic et al. 2005). Watershed development can have far-reaching impacts on hydrology and habitat quality that may affect aquatic communities downstream from the actual site of disturbance. Changes in water quality due to development (such as increased nutrient and sediment loads) affect benthic invertebrate communities (Lerberg et al. 2000, Bilkovic et al. 2006a), and other development effects (such as habitat fragmentation and increased impervious surfaces) can affect fish populations (Scheuerell & Schindler 2004) and degrade marsh and riparian bird community integrity (Hennings & Edge 2003, DeLuca et al. 2004).

Watershed development within the James River, Virginia's largest tributary to the Chesapeake Bay, shows a trend towards the conversion of forested and agricultural lands to residential and commercial land use. The James River has a drainage area that encompasses approximately 26 512 km^2, and land use varies considerably along its length. From the fall line at Richmond to the mouth, the river is characterized by high residential and commercial development on the lower shore. The land adjacent to the rural upper James River is primarily forested (84 % of land area), with 15 % agriculture (primarily pasture) and <1 % urban. Conversely, the more populated area surrounding the lower James River is 31 % forested, 12 % agriculture, and 48 % urban (Commonwealth of Virginia 2005). The major land use changes between 1992 and 2001 were loss of forested land (from 46 to 39 % of the land area) and an increase in urban area (USGS 1999, 2001). In the future, the lower portion of the drainage basin is expected to become increasingly urbanized with associated loss of agricultural and forested land. The cumulative impact of this intense coastal development on nearshore ecosystems is unknown.

Since highly productive nearshore habitats (such as seagrass beds, shallow water, marshes and oyster reefs) may serve as nursery and feeding grounds for many nekton and invertebrate species, impacts to these areas can resonate throughout trophic webs. The modification of subtidal habitats driven by coastal development may alter the biodiversity, trophic interactions and community assemblages of the nearshore ecosystem (Toft et al. 2007). Biotic responses may be due to degradation and loss of shallow water habitat, resulting in part from the construction of shoreline structures. Alterations of the riparian and nearshore environments associated with shoreline hardening projects can result in increased nearshore depth (Jennings et al. 1999, Peterson et al. 2000, Bilkovic et al. 2006a), the loss of complexity and refugia (Jennings et al. 1999, Scheuerell & Schindler 2004), a reduction in desirable allocthanous inputs, such as woody debris (Jennings et al. 1999, Christensen et al. 1996), and the creation of microclimates unsuitable for local species (Rice 2006). Nearshore ecosystems are probably the most affected by human activities, and changes in ecosystem integrity reflected in the responses of biological communities are a means to measure condition and manage systems (Beck et al. 2003). For example, the Chesapeake Bay Program has used multimetric indices, such as the Benthic Index of Biotic Integrity (B-IBI), to elucidate regional and local water- and sediment-quality impairments in Chesapeake Bay (Dauer et al. 2000).

Coastal development (residential/urban land use and shoreline hardening) within the James River is predicted to result in measurable declines in nearshore nekton community integrity. As efforts to manage fisheries advance toward an ecosystem approach in the Chesapeake Bay and elsewhere, information on the habitat quality of the nearshore and riparian zones and species associations becomes vital (NOAA Chesapeake Bay Fisheries Ecosystem Advisory Panel 2006). Therefore, we assessed relationships among (1) subtidal habitat, (2) shoreline condition, (3) upland development and (4) nearshore nekton communities (defined here as an assemblage of fish and select decapod crustaceans) within the James River, Virginia.

MATERIALS AND METHODS

Nekton survey on the James River. Prior to site selection, the nearshore subtidal benthic habitat of the James River was continuously surveyed with a bow-mounted Marine Sonics Sea Scan PC 600 kHz unit appropriate for shallow water conditions (<5 m depth). An external JRC D/GPS system (accuracy 3 to 5 m) was used to acquire ship position and control line planning. The Sea Scan side-scan sonar has the ability to map swath transects of subtidal habitat parallel to the shore, and was towed to collect real-time, georeferenced, riverbed mosaic data with overlapping edges matched to form a continuous profile of the bottom. The area was surveyed in 40 m swaths following shorelines. Approximately 127 km (survey area = 6.7 km^2) were surveyed on the north and south shores of the James River from the James River Bridge upriver to the Chickahominy River (Fig. 1). Georeferenced profiles were then converted to Geographic Information Systems (GIS) coverages for the depiction of areas of classified habitats. The classification of bottom type is based on an unsupervised classification of the imagery signatures. Combined with ground-truth sampling,

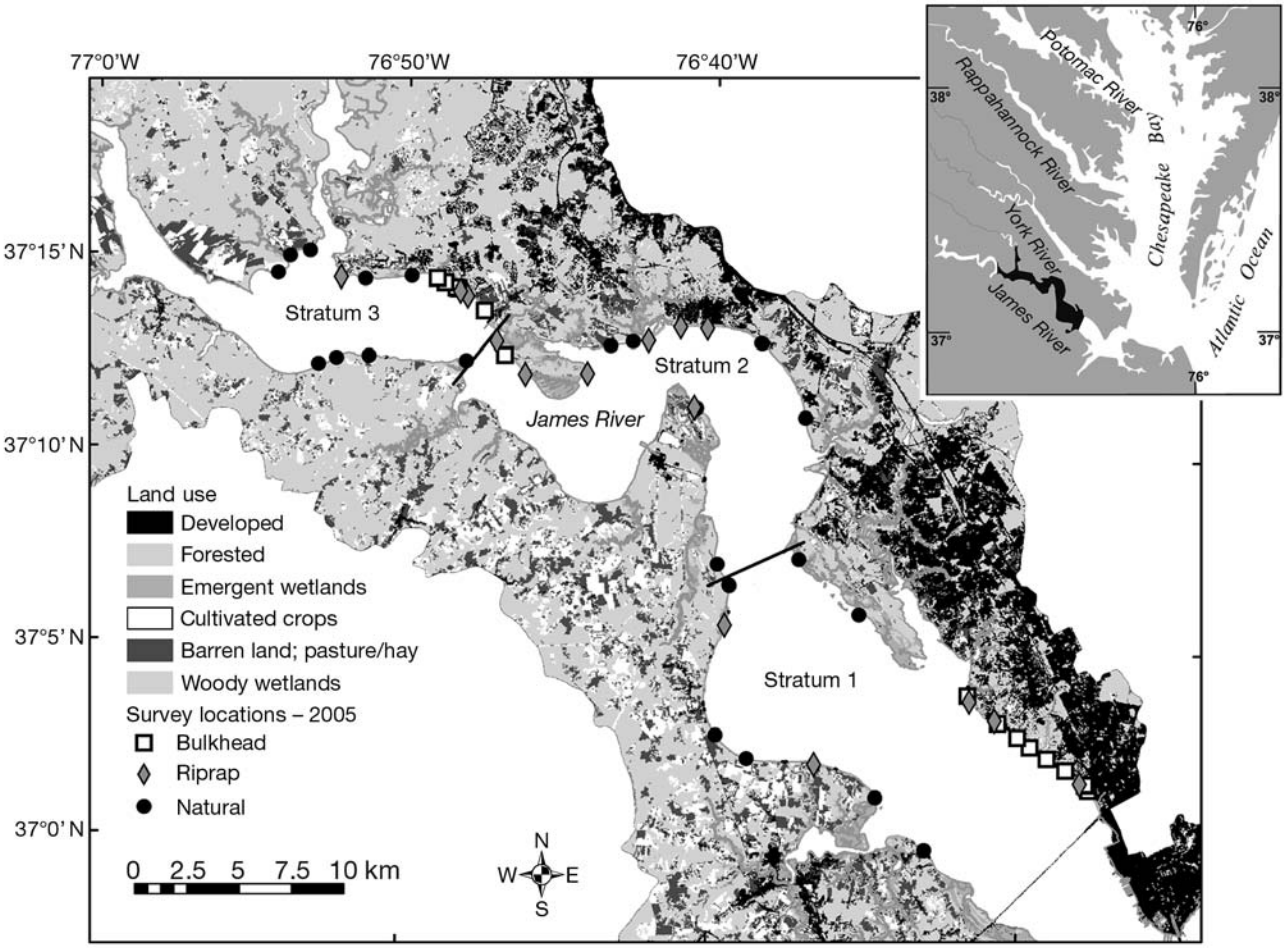

Fig. 1. Fish community survey locations and land use on the James River, 2005 (National Land Cover Database)

this enables separation of major classes of bottom type (e.g. sand, submerged aquatic vegetation [SAV], coarse debris). Since structural nearshore habitats (such as reefs) may support higher productivity than structureless habitat in estuarine systems (Beck et al. 2003), and our subtidal habitat mapping only allowed for coarse discrimination of bottom features, 2 broad benthic habitat classifications were considered for associations with fish communities in the James River: featureless bottom (soft) and structural habitat (hard). Hard bottom was defined to include classes associated with structure present in the James River that consisted of oyster or mussel beds and coarse woody debris, and soft bottom encompassed classes associated with structureless benthic habitat, typically sand and/or silt sediments. For the majority of the area surveyed, benthic habitat was classified as soft (featureless) with only approximately 29% of the area classed as hard (Bilkovic et al. 2006b).

The James River was initially divided into three 20 km sections for systematic sampling to ensure the longitudinal extent of the river was uniformly sampled. Each stratum was further segmented into nearshore reaches no larger than 100 m based on adjacent shoreline condition (riprap revetment, bulkhead, natural [unmodified] and surveyed bottom type [hard or soft]). Site categories were a combination of estimated nearshore seabed type and associated shoreline: hard bottom natural, hard bottom riprap revetment, hard bottom bulkhead, soft bottom natural, soft bottom riprap revetment and soft bottom bulkhead (Fig. 1). Attempts were made to randomly select at least 4 sites from each category in each stratum; however, some categories were not present in each stratum or in the same abundance as other categories due to the rarity of hard bottom throughout the entire survey area. Because of the limited availability of nearshore locations with structural habitat (hard bottom), detailed analyses were restricted to assessing shoreline conditions and land use in relation to fish communities.

Two replicate seine hauls (30.5 × 1.22 m bagless seine of 6.4 mm bar mesh) were conducted at each site during July through August 2005. One end of the seine was held on shore or as close to shore as possible. The other was fully stretched perpendicular to the shore and swept with the current over a quarter-circle quadrant. Ideally, the area swept was equivalent to a 729 m^2 quadrant. When depths of 1.22 m or greater were encountered, the offshore end was deployed along this depth contour. After encircling an area, the mouth of the seine was closed by crossing over the lead lines of each wing of the net. The seine was slowly hauled closed and the lead line continually checked to ensure contact with the bottom. For each replicate, counts and total lengths were recorded for each finfish species (or a subsample of at least 25 ind.); select crustacean species were also enumerated. Replicates were averaged for each date and location to avoid pseudoreplication at the habitat level (sensu Hurlbert 1984). Community measures were calculated for each site, including relative abundance, diversity and fish community indices (Bilkovic et al. 2005). At each site, auxiliary data were collected, including dissolved oxygen, salinity, conductivity, pH, turbidity, current speed, tides, air and water temperature, and wind speed and direction. Onsite visual evaluation and field photographs were used to categorize shoreline condition as bulkhead, riprap revetment or natural based on the predominant structure, and the percentage of subtidal habitat (i.e. shell, woody debris) was estimated by visual inspection and walking the survey area.

Analyses. Efforts to assess ecosystem health based on biological communities have often involved the reduction of large datasets to integrative indices, such as Index of Biotic Integrity (IBI). The use of single measures of condition affords many advantages including its ease of translation to end-users, and representation of multi-trophic responses to aquatic condition. A potential limitation of integrative measures is the difficulty in extracting species-specific information from a single indicator. Recent developments in multivariate approaches in community analyses allow for inclusion of all available species data to elucidate biological patterns and the easy extraction of species specific results that managers often seek. Nonparametric multidimensional scaling maps dissimilarities across communities and analysis of similarities (ANOSIM) provide statistical comparisons of complete species assemblages in relation to hypothesized environmental drivers. While the relative merits of multivariate and multimetric approaches have been debated (e.g. Suter 1993, Karr & Chu 1999), empirical evidence for the exclusive use of either approach is limited and inconclusive because the real biological impairment at a site is often unknown. Both approaches may supply useful complementary information that provides assessments of ecosystem integrity, guides the establishment of causative agents in ecosystem degradation and allows for the characterization of the nature of relationships (i.e. estimating thresholds of change). Therefore, we evaluated fish community integrity with a multimetric index and multivariate approaches.

Guild development and metric selection. The fish community index (FCI) is a multimetric indicator that measures biotic integrity and was developed and applied previously in the nearshore estuarine environs of the Chesapeake Bay; for detailed methods see Bilkovic et al. (2005). The FCI was applied in the James River system to assess relative measures of fish community structure and function. Fish species were initially placed into several guilds based on their documented life histories. Guilds were constructed based on (1) reproductive strategy (anadromous, marine, freshwater or estuarine spawner), (2) trophic level (carnivore, planktivore or benthivore), (3) primary life history (marine, estuarine, freshwater, diadromous or estuarine-dependent nursery), (4) habitat preference (pelagic or benthic) and (5) origin (estuarine resident or non-resident). Primary sources of life history information included Lippson & Moran (1974), USFWS (1978) and Murdy et al. (1997).

Eight metrics were assessed for consistency as indicators of aquatic ecosystem health based on fish community structure and function. Metrics were extracted from current literature that addressed similar estuarine environments, including the Chesapeake Bay, and were responsive to disturbance (Deegan et al. 1997, Jordan & Vaas 2000). These metrics represent key

Table 1. Fish community metrics assessed for use in a multimetric index

Metric	Description
Species richness/diversity	
Species richness	No. of species – 1/log(abundance)
Proportion of benthic-associated species	No. of benthic-associated species/total no. of species
No. of dominant species	No. of species that make up 90% of total abundance
No. of resident species	No. of estuarine resident species
Abundance	
Ln abundance	Natural log of abundance
Trophic composition	
Trophic index	Relative proportions of 3 broadly defined trophic guilds based on primary prey items: carnivores, planktivores and benthivores (scaled to 5)
Nursery function	
No. of estuarine spawning species	No. of species that predominately spawn in estuarine systems
No. of estuarine nursery species	No. of species that use estuarine systems as nursery habitat

aspects of fish community integrity, as well as the elements of life history that are dependent on estuarine condition. Metrics reside in 4 broad categories: taxonomic richness and diversity, abundance, trophic composition, and nursery function (for details see Bilkovic et al. 2005, Table 1). For each site, individual metric values were calculated based on observed species composition and abundance in 2005.

Metric analyses. All metrics were examined for normality and transformed when necessary using an appropriate transformation. The metrics (1) abundance and (2) proportion of benthic species were normalized with natural logarithms and square-root transformations, respectively. All other metrics had normal distributions and were not transformed. Individual metrics were standardized based on each metric distribution and aggregated, without weighting, into a FCI. For example, each species richness metric value was divided by the largest observed richness measure to standardize values (scale = 0 to 1) based on existing conditions for the year (no reference condition was considered); standardized metrics were then added to obtain the aggregate FCI.

The applicability and variability of metrics were assessed by calculating correlation coefficients for metric values, and examining principal component analysis (PCA) coefficients of the metrics. PCA was applied to individual fish community metrics to evaluate the usefulness of the multimetric index (FCI) as a descriptor of fish community structure and function. Those metrics that are supported in a multimetric index should exhibit similar associations. Metrics that exhibited similar trends in correlation (high and positive) with the aggregate FCI of all 8 tested metrics were combined into a final FCI by summing standardized individual metric values.

Nekton and habitat comparisons. Relationships among nekton community measures (FCI, abundance) and habitat measures (shoreline condition, developed lands and bottom habitat) were examined with univariate (1-way ANOVA and nonparametric changepoint analysis), and multivariate (multi-dimensional scaling, analysis of similarities, *k*-dominance curves) methods. Developed land use data were obtained from the National Land Cover Dataset (NLCD) (30 m raster coverage, USGS 2001), and impervious surface estimates were extracted from the data set RESAC 2000 CBW Impervious Surface Product[1]. Low, medium and high intensity development classifications from NLCD were combined into a 'developed lands' category. Scatterplots of fish community indices and developed land metrics suggested a potential threshold response, so changepoint analysis (nCPA) (King & Richardson 2003, Qian et al. 2003) was used to test for the presence of an ecological threshold in the FCI due to (1) developed land use and (2) impervious surface at 3 spatial scales: 100, 200 and 1000 m buffer widths. Buffers were generated in ARCGIS using the survey location as the central point. The nCPA detects changes in the mean and variance of a response variable (in this case FCI) due to variation in a forcing factor (in this case land use and impervious surface at 3 spatial scales). It examines every point along a continuum of predictor values (developed lands) and determines the probability that a value can split the data into 2 groups that have the greatest difference in means and/or variance. With bootstrap simulations repeated 1000 times, a distribution of changepoints is estimated and illustrated with a cumulative probability curve that describes the probability (frequency) of a changepoint occurring at various levels of disturbance. When probabilities were <0.05, the cumulative probability curves were assumed to accurately assess the likelihood of an ecological threshold occurring. Changepoint analyses were conducted in S-Plus using the custom function nopar.chngp (Qian et al. 2003).

Nearshore nekton community similarities were examined with nonparametric multidimensional scaling (nMDS) and analysis of similarities (ANOSIM) in PRIMER 6.0 for a subset of data within the mesohaline salinity range (5 to 18 ppt) to minimize the complicating effects of salinity on biological communities. Since MDS ordinates sites based on similarities in species makeup, using rank order of distances to map out relationships, it is critical to restrict samples to those within the same salinity regime that dictates species assemblages. Sites with high similarity are placed close together on the MDS map. A stress coefficient represents the goodness of fit of the data to a nonparametric regression; higher stress indicates more scatter about the line and perfectly represented data tend towards zero. Typically, stress is minimized with the addition of dimensions, and 2-dimensional and 3-dimensional stress values are estimated. Acceptable ordinations of data occur when stress values are <0.2 (Clarke & Warwick 2001). Prior to the MDS ordination, species abundances were square-root transformed to moderately downweight the effect of dominant species, and a Bray-Curtis coefficient was used to calculate the similarity matrix. Factors were overlaid on a MDS plot to visualize community groupings in relation to habitat features, such as shoreline condition. Subsequently, ANOSIM was used to test relationships among (1) shoreline condition (bulkhead, riprap revetment or natural), (2) local development within 1000 m buffer (above or below the ecological thresh-

[1]Mid-Atlantic Regional Earth Science Applications Center. Chesapeake Bay watershed impervious surface product (data set), 2000

old) and (3) shoreline condition × local development. Shoreline condition was categorized as bulkhead, riprap revetment or natural shoreline. Development was categorized as 'high' or 'low' based on observed ecological thresholds from changepoint analyses (23 %) at both 200 and 1000 m buffered areas around sites. To assess the effect from interactions between shoreline condition and local development on nekton community integrity, 5 observed scenarios were compared with pairwise ANOSIM: (1) bulkhead shoreline with high development, (2) riprap shoreline with high development, (3) bulkhead shoreline with low development, (4) riprap shoreline with low development and (5) natural shoreline with low development. No high development areas sampled had natural shorelines; therefore, this category was not included in the analysis. Exploration of species contributions to describing similarities within and dissimilarities among groups was completed with similarity percentages (SIMPER) procedure (PRIMER 6.0). This method uses relative abundances, represented by Bray-Curtis similarities, to determine those species contributing the most to overall dissimilarity between pairs of groups (Clarke & Warwick 2001).

Cumulative dominance (*k*-dominance) curves were estimated as a way to corroborate multivariate community analyses. The *k*-dominance curves depict cumulative ranked abundances plotted against species rank to examine differences in communities at grades of shoreline condition, local land use and combinations of condition. Curves with relatively shallow slopes are indicative of communities dominated by a single or a few species and are thought to be representative of affected sites (Attrill 2002). Since the curves express the level of dominance by species number and are not dependent on species specific comparisons, the entire dataset was included (meso- and oligohaline regimes). Cumulative dominance curves were plotted by (1) shoreline condition (bulkhead, riprap revetment, and natural), (2) amount of developed lands within 1000 m categorized in relation to the observed ecological threshold from changepoint analyses (high development ≥23 %, low development <23 %) and (3) 5 categories of shoreline condition × local land use described in the previous section.

RESULTS

Nekton collections

A total of 8626 nekton consisting of 33 species were collected from July to August 2005 at 54 sites. By percentage of catch, the most abundant species were Atlantic menhaden *Brevoortia tyrannus* (61.4 %), Atlantic silverside *Menidia menidia* (14.8 %), white perch *Morone americana* (9.6 %), bay anchovy *Anchoa mitchilli* (2.6 %), and spot *Leiostomus xanthurus* (2.3 %). Number of species collected at each site ranged from 2 to 14, and FCI values ranged from 1.2 to 6.7 (since metrics were standardized on a scale of 0 to 1, the maximum possible FCI value is 7.0 for the index that aggregates 7 metrics). Nekton average abundance (±SE) generally decreased across treatment groups (Group 1 = bulkhead shoreline with high development: 175.9 ± 101.4; Group 2 = riprap shoreline with high development: 118.9 ± 85.9; Group 3 = bulkhead shoreline with low development: 71.6 ± 33.9; Group 4 = riprap shoreline with low development: 30.3 ± 6.0; and Group 5 = natural shoreline with low development: 69.6 ± 15.6. A disproportionately high abundance of Atlantic menhaden in Groups 1 and 2 (151.4 and 97.2, respectively) accounted for the high average abundance of these groups. Number of species comprising 99 % of the catch and the average fish community index values were the lowest at sites with high development and bulkhead shoreline (Group 1: 6 species, FCI = 2.27 ± 0.23), and the highest at sites with low development and riprap or natural shoreline (Group 4: 18 species, FCI = 4.21 ± 0. 34; Group 5: 14 species, FCI = 3.82 ± 0.17). Groups 2 and 3 had intermediate values of species comprising 99 % of the catch and average FCI values (9 species, FCI = 3.39 ± 0.64; 10 species, FCI = 3.52 ± 0.58, respectively).

Fish community metrics

All but one of the examined fish community metrics were positively and highly correlated (r ≥ 0.5) with the summed metrics (FCI). The majority of correlations among metrics were positive. Total number of individuals (transformed into natural logarithms) had low, non-significant correlations with the FCI and negative

Table 2. Eigenvectors and accountable variances of the first 2 principal components (PC) based on individual fish community metrics. PC1 and PC2 accounted for 83 % of the variance in the data

Metric	PC1	PC2
Species richness	0.43	–0.06
Proportion of benthic-associated species	0.28	–0.37
No. of dominant species	0.41	–0.16
No. of resident species	0.38	0.28
Ln total abundance	–0.12	0.68
Trophic index	0.38	–0.22
No. of estuarine spawning species	0.34	0.36
No. of estuarine nursery species	0.39	0.34
% variance accounted for	60	23

correlations with other individual metrics. Principal components analysis of individual fish community metrics supported the use of all but one of the metrics (i.e. abundance, natural logarithm transformed) in a composite FCI. The first and second principal components accounted for 83% of the variance in the dataset (Table 2). All metrics were positively associated with PC1, except for low negative loading for total abundance. When considering correlation patterns and PCA analyses, the use of all the metrics, with the exception of total abundance, was supported for the application of a nearshore FCI in the James River.

Nekton communities and habitat

The lack of hard bottom locations on the James River in the nearshore became evident only after sites were surveyed for fish collection and restricted our ability to quantify differences between fish communities and bottom type. Only 11 sites could be designated as hard bottom, and many of these sites consisted of a seabed layer of shell hash, not large structural reef features. No significant difference in fish community structure measures (individual fish metrics and FCI) was evident between hard and soft bottom locations. Nonetheless, the amount of hard bottom cover was highest at sites with natural shoreline (30%) conditions as opposed to hardened shoreline (riprap revetment or bulkhead, 6%) indicating a potential land–water nexus (1-way ANOVA, $p = 0.009$; Fig. 2).

The lowest FCI values were associated with bulkhead shorelines, while sites with natural or riprap revetment shorelines reflected similar values (1-way

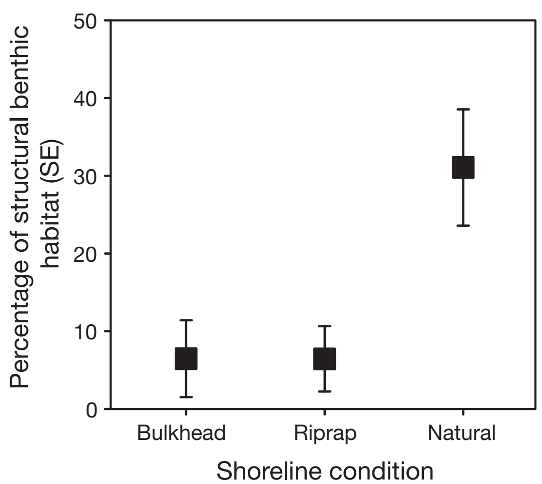

Fig. 2. Mean ± SE structural subtidal habitat by shoreline condition: bulkhead, riprap revetment or natural, for fish survey sites on the James River. Structural habitat, such as oyster reefs, clam beds or woody debris was reduced adjacent to hardened shorelines (1-way ANOVA, $p = 0.009$)

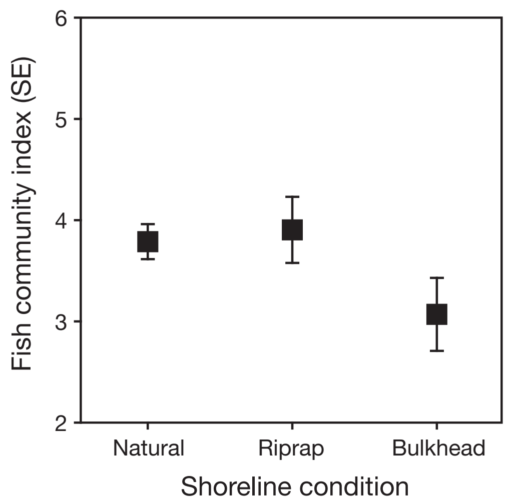

Fig. 3. Mean ± SE fish community index by shoreline condition: bulkhead, riprap revetment or natural. Values associated with bulkhead shorelines were significantly lower than for riprap revetment or natural conditions (1-way ANOVA, $p = 0.04$)

ANOVA, $p = 0.04$; Fig. 3). Of the measured chemical and physical variables, only salinity and dissolved oxygen were significantly related to the biotic endpoints ($p < 0.0001$, $r = -0.598$; $p = 0.031$, $r = -0.306$, respectively). Dissolved oxygen was also positively correlated with water temperature and time of day, suggesting the possibility that as shallows warm up, fish migrate into deeper waters, which is reflected as slight depressions of FCI values in relation to dissolved oxygen. However, this trend may be spurious in that conditions were never hypoxic and dissolved oxygen ranged from 5.7 to 10.7 mg l^{-1}. Since salinity is correlated with FCI values and diversity measures, distinguishing robust relationships with shoreline conditions is problematic. However, species diversity minimums in the James River have previously been observed at salinities between 8 and 10 ppt (Wagner 1999), while our data indicated that species depressions occurred between 10 and 18 ppt and this trend was primarily driven by sites with bulkhead shoreline in large stretches of intensely developed reaches on the lower north shore. Notably, the higher salinity region where species diversity is depressed is also the area of the river with the most intense development (Fig. 1). It is possible that in river reaches where species numbers are expected to be higher then observed, intense development has suppressed this effect. In support, single metrics that are independent of salinity regime limitations (e.g. trophic index) also have their lowest values associated with the highest development density in the farthest downstream reaches of the river.

Changepoint analyses indicated that ecological thresholds existed in response to developed land use (urban and suburban) at all 3 spatial scales, 100,

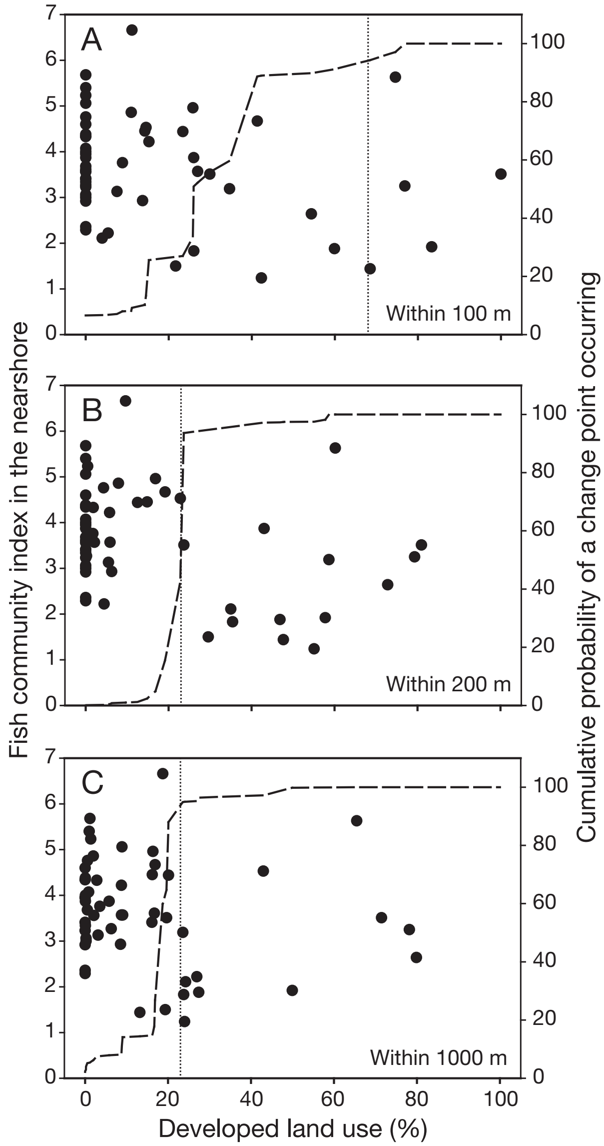

Fig. 4. Significant fish community responses ($p \leq 0.05$) were measured with the fish community index (FCI) in relation to the amount of developed lands within a (A) 100, (B) 200 and (C) 1000 m buffer. Scatterplots indicate the fish community response at different levels of development. The dashed line indicates cumulative probability of a change in the fish community response at a given level of development. There was a 94 % cumulative probability of an ecological threshold occurring at 23 % developed lands for the FCI at the 200 and 1000 m spatial scales. At the 100 m scale, the ecological threshold (94 % cumulative probability) occurred at 68 % developed lands

200 and 1000 m. Particularly strong patterns were evident at the 200 and 1000 m spatial scales, where the cumulative probability curve indicated a 94 % probability of a changepoint occurring at ≥23 % developed land use for the FCI values (Fig. 4). At the smaller 100 m scale, the ecological threshold (94 % cumulative probability) occurred at 68 % developed lands. Impervious surface at all 3 scales did not produce an ecological threshold in relation to fish integrity.

Table 3. Nearshore fish communities at 29 mesohaline sites within the James River, Virginia. ANOSIM R statistic for testing differences among sites with varying shoreline condition (bulkhead, riprap revetment or natural) and local development above or below established ecological threshold conditions (within 1000 m: high ≥ 23 %; low < 23 %). Global R compares differences among all sites; pairwise comparisons are in subsequent rows, with significance level from 9999 permutations given in brackets. The largest R indicates the best site separation. *p = 0.06 (marginally insignificant), **p ≤ 0.05, ***p ≤ 0.001

Group	ANOSIM comparisons	R	p
Global R		0.250	0.007**
1 vs. 2	Bulkhead, high vs. riprap, high	−0.145	0.909
1 vs. 3	Bulkhead, high vs. bulkhead, low	−0.115	0.607
1 vs. 5	Bulkhead, high vs. natural, low	0.403	0.001***
2 vs. 3	Riprap, high vs. bulkhead, low	0.164	0.286
2 vs. 4	Riprap, high vs. riprap, low	−0.097	0.679
2 vs. 5	Riprap, high vs. natural, low	0.233	0.061*
3 vs. 4	Bulkhead, low vs. riprap, low	1.000	0.100
3 vs. 5	Bulkhead, low vs. natural, low	0.684	0.010**

nMDS ordination plots exhibited inter-sample resemblances with overall 2-dimensional stress of 0.16 and 3-dimensional stress of 0.11 (Fig. 5). Analysis of similarities (ANOSIM) testing indicated differences in sites based on single habitat variables: shoreline condition (Global R = 0.232, pairwise difference for bulkhead versus natural: R-statistic = 0.393, p = 0.0002), and developed land use levels (high versus low: Global R = 0.169, p = 0.014).

Shoreline × local development exhibited different influences on fish communities in 3 circumstances:

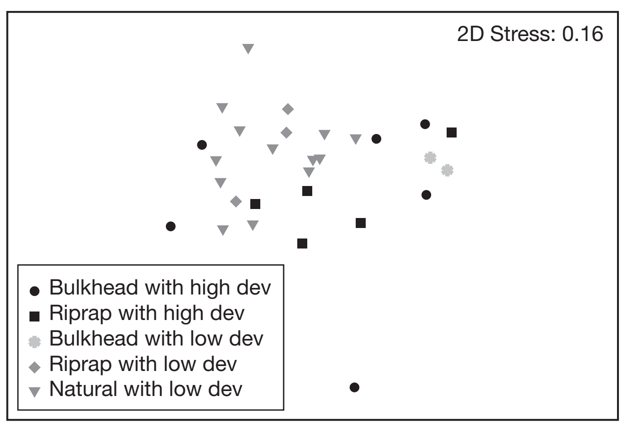

Fig. 5. Multidimensional scaling ordination of James River nearshore nekton assemblages adjacent to sites categorized into 5 arrangements of upland land use and shoreline condition: bulkhead shoreline with high development (●), riprap shoreline with high development (■), bulkhead shoreline with low development (✱), riprap shoreline with low development (◆) and natural shoreline with low development (▼)

(1) natural shorelines with low development (Group 5) versus bulkhead shorelines with high development (Group 1); (2) natural shorelines with low development (Group 5) versus bulkhead shorelines with low development (Group 3); and (3) natural shorelines with low development (Group 5) versus riprap shorelines with high development (Group 5) (Table 3, Fig. 5). Essentially, low developed, natural or riprap shoreline sites were different from all the combinations with high development and/or bulkhead conditions. Differences between the natural or riprap, low development categories (4 and 5) and others (1, 2 and 3) were typically due to the high contribution of species that defined the similarity of sites within Groups 4 and 5, such as Atlantic silverside, white perch, spot, striped bass *Morone saxatilis*, blue crab *Callinectes sapidus* and mummichog *Fundulus heteroclitus* (shallow-water habitat users), as opposed to species characteristic of the other groups, such as Atlantic menhaden and gizzard shad *Dorosoma cepedianum* (generalist species with wide habitat ranges) (Table 4).

Cumulative dominance curves indicated that highly developed lands in excess of 23%, and severed land–water interfaces (bulkhead) reflected fish

Table 4. Species contributions to dissimilarities between groups. Average dissimilarity (Avg diss) represents the contribution of each species to the overall dissimilarity between groups. The ratio of Avg diss to standard deviation (diss/SD) signifies good discriminating species for the groups with relatively large values. The percentage each species contributes to dissimilarities (Contrib%) is rescaled to to the cumulative percent (Cum%) of total dissimilarity. Species are ordered in decreasing contribution. Average abundance by group (Avg abund) is based on values in the Bray-Curtis similarity matrix and does not represent true abundance estimates. Nat: natural; Bulk: bulkhead; Rip: riprap revetment; Dev: development

Species	Group 5 Nat-low dev Avg abund	Avg abund	Avg diss	Diss/SD	Contrib%	Cum%
Group 1: Bulk-high dev[a]						
Atlantic menhaden	2.91	8.06	21.6	1.26	31.9	31.9
Atlantic silverside	4.43	2.19	10.51	1.14	15.52	47.42
White perch	2.35	2.06	8.25	1.46	12.18	59.59
Spot	1.51	0.58	4.9	1.25	7.24	66.83
Bay anchovy	1.04	0.63	3.88	1.02	5.73	72.56
Striped bass	0.8	0	2.88	1.07	4.26	76.82
Atlantic croaker	0.15	0.66	2.23	0.74	3.29	80.11
Gizzard shad	0.42	0.63	2.1	1.05	3.1	83.22
Mummichog	0.61	0	2.05	0.56	3.03	86.24
Blue crab (young of the year)	0.52	0.17	1.89	0.83	2.79	89.03
Hickory shad	0.4	0.24	1.73	0.82	2.56	91.59
Group 2: Rip-high dev[b]						
Atlantic menhaden	2.91	6.29	17.84	0.95	30.33	30.33
Atlantic silverside	4.43	2.19	7.32	1.36	12.44	42.77
White perch	2.35	1.34	6.33	1.37	10.76	53.53
Bay anchovy	1.04	1.56	4.53	1.13	7.71	61.24
Spot	1.51	1.01	3.58	1.26	6.09	67.32
Striped bass	0.8	0.32	2.68	1.22	4.55	71.87
Hickory shad	0.4	0.6	2.26	0.94	3.84	75.72
Gizzard shad	0.42	0.63	2.12	1.17	3.61	79.32
Atlantic croaker	0.15	0.54	1.88	1.08	3.2	82.53
Mummichog	0.61	0	1.86	0.57	3.17	85.7
Blue crab (young of the year)	0.52	0	1.64	0.82	2.78	88.48
Blue catfish	0.2	0.24	1.3	0.59	2.21	90.69
Group 3: Bulk-low dev[c]						
Atlantic menhaden	2.91	13.93	35.67	2.28	49.21	49.21
Atlantic silverside	4.43	0.85	10.41	2.29	14.36	63.57
White perch	2.35	0	6.92	1.94	9.55	73.12
Spot	1.51	0.87	3.64	1.44	5.02	78.14
Bay anchovy	1.04	0	2.94	1	4.05	82.19
Striped bass	0.8	0.61	2.11	1.17	2.91	85.1
Mummichog	0.61	0	1.68	0.58	2.32	87.42
Blue crab (young of the year)	0.52	0	1.47	0.84	2.03	89.45
Gizzard shad	0.42	0.71	1.4	1.31	1.93	91.38

[a]Average dissimilarity between natural shoreline with low development (Group 5) and bulkhead shoreline with high development (Group 1) sites is 67.71

[b]Average dissimilarity between natural shoreline with low development (Group 5) and riprap revetment shoreline with high development (Group 2) sites is 58.82

[c]Average dissimilarity between natural shoreline with low development (Group 5) and bulkhead shoreline with low development (Group 3) sites is 72.49

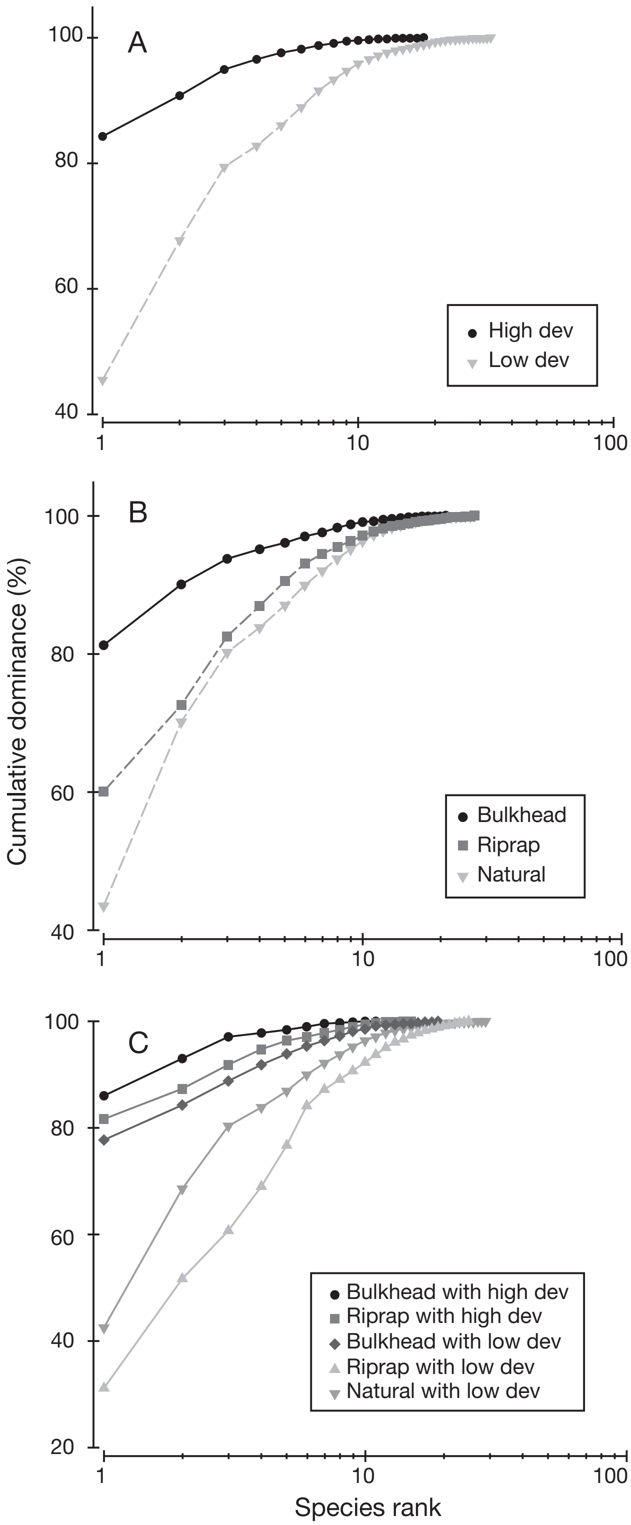

Fig. 6. Cumulative ranked abundances of nekton plotted against species rank as depicted by *k*-dominance curves to examine differences in communities at grades of (A) land use, (B) shoreline condition and (C) land use × shoreline condition. Curves with relatively shallow slopes are indicative of communities dominated by one or a few species and are considered to be representative of degraded sites

communities with the lowest integrity (dominated by few species). Natural and riprap revetment conditions resulted in similar *k*-dominance curves, while bulkhead conditions severely reduced the number of dominant species present. As expected, the lowest integrity sites had high development and/or bulkhead shoreline, the highest integrity sites had low development with natural or riprap shoreline. Sites with low development and bulkhead shorelines fared worse than those with natural or riprap shoreline, (Fig. 6).

DISCUSSION

Ecological threshold

Both upland development and the placement of erosion control structures on the shoreline were associated with reduced fish community integrity. Upland development impacts were most discernable in fish communities at large spatial scales (200 and 1000 m), as opposed to local scales (100 m), with evident ecological thresholds in biotic responses at relatively low development (>23%). Ecological thresholds that mark breakpoints at which a system or community notably responds (perhaps irreversibly) to a disturbance have been supported in a variety of systems and scales. The current literature suggests that tributary development (e.g. land use, impervious surface) exceeding 10 to 25% compromises the integrity of the ecosystem and its ability to perform functions (Limburg & Schmidt 1990, Wang et al. 1997, Paul & Meyer 2001, DeLuca et al. 2004, Bilkovic et al. 2006a, Brooks et al. 2006). DeLuca et al. (2004) observed responses in marsh bird community integrity at land-use disturbance thresholds of approximately 14%. As little as 10% watershed development within a large estuary and between 10 to 20% urbanization within streams have been linked with degradation of fish communities (Limburg & Schmidt 1990, Wang et al. 1997). A review of reported thresholds of impervious surface area within stream catchments indicated that between 10 and 20% was associated with stream and fish community degradation (Paul & Meyer 2001).

It is yet uncertain what are the most appropriate spatial or temporal scales at which threshold values apply, and a confounding factor is that scales will vary depending on the biological community affected (Groffman et al. 2006). Deciding what the appropriate scales should be will determine how this information may shape management and planning decisions. However, the relatively low development threshold observed for nearshore fish communities indicates that watershed planning may need to focus at spatial scales typically larger than a single parcel, and that comprehensive shoreline and watershed land use plans become imperative. Development in both waterfront and upland regions of a watershed may affect biotic communities, so should be considered. The variation in biological community responses over time still needs to be explored. Temporal variability was not addressed in this study, which was meant to be a comparative 'snapshot' over a large spatial area; therefore, patterns due to diel, seasonal and interannual variability are unknown. Fish populations are expected to vary over long time periods and may be affected by changes in water temperature, turbidity and productivity from year to year as well as anthropogenic influences. To capture communities that were representative of nearshore assemblages, surveys were conducted during periods of the year when abundance and diversity are generally highest in temperate estuaries (e.g. Hoff & Ibara 1977, Ayvazian et al. 1992, Rountree & Able 1992). Therefore, observed patterns may pertain to the majority of species that compose shallow water fish assemblages in the Chesapeake Bay.

Shoreline development and subtidal habitat

Even in areas with low development, the presence of shoreline erosion control structures had a negative impact on local fish community integrity. Shoreline bulkhead structures sever the connection between riparian, intertidal and subaqueous areas, alter the natural curve of the shoreline, remove undercut crevice habitat, change nearshore wave dynamics, and reduce shallow water habitat. James River historically possessed substantial reaches of subtidal structural habitat, such as oyster reefs (McCormick-Ray 2005) and seagrass beds (Moore et al. 1999) that supported fish production. The reduction of these structural habitats driven by coastal development (particularly shoreline hardening) may have altered community assemblages of the nearshore ecosystem. These changes are reflected in the differences in fish community along a continuum of shoreline conditions. Fish community integrity was lowest along bulkheaded shorelines, which arguably represent the most altered habitat. Fish community integrity was similar in natural and riprap revetment structures, which may mimic natural shorelines by accentuating habitat opportunities in the form of crevices and hard structure, as well as having a lesser impact on nearshore wave dynamics than bulkheads. Similar patterns have been observed in fish and macrobenthic communities (Jennings et al. 1999, Trial et al. 2001, Bilkovic et al. 2006a, Seitz et al. 2006).

Combined impacts

Coastal development impact on nearshore communities is dependent on the combination of upland land use and shoreline condition. Fish communities in low developed, natural or riprap shoreline reaches were distinct from other development scenarios (high development with shoreline hardening, low development with bulkhead shoreline). Bulkheads affected the nearshore environment and nekton community, regardless of the level of upland development, while the influence of riprap structures varied depending on the amount of upland development. The similarity between natural and riprap shorelines in areas of low development may be partially due to the condition of the surrounding watershed. With increased development, fish community composition shifted from diverse assemblages with tidal marsh species such as mummichog and juvenile blue crabs to predominately few generalist, pelagic species such as Atlantic menhaden. The shift may be due to the removal or reduction during development of fringe marsh environs, which are essential nursery habitats for several observed nekton species and their prey. Juvenile blue crabs and a variety of commercially or ecologically important finfish exhibit a strong preference (higher abundance and/or biomass) for fringe marsh habitat over altered shoreline (Peterson et al. 2000, Carroll 2003, King et al. 2005). Links among fringe marshes, infaunal prey in subtidal habitats, and predator abundance and diversity (blue crab and finfish) may reflect the effects of shoreline degradation on secondary production in shallow waters (Seitz et al. 2006). In low development areas, there are likely to be more surrounding natural shorelines than in areas with high development. The surrounding natural shoreline may be subsidizing riprap shorelines in these areas (Seitz et al, 2006), leading to more diverse communities than seen in highly developed areas. Therefore, the preservation of reaches with low development and natural shorelines should be a priority.

Management implications

Relatively low levels of development appear to trigger a shift in nearshore assemblages and may be due to a complex set of impacts. Effects from both shoreline and upland development within a watershed need to be considered during the planning process. However, since landscapes tend to be developed piecemeal, translating and applying coastal development controls to practical management scales is a challenge. Watersheds and shorelines already beyond the development threshold most probably will not be reversed to natural conditions; therefore, alternative approaches to mitigate impacts are required. Preservation of riparian buffers along shorelines, and the use of 'living shoreline' alternatives for erosion protection where appropriate, may help mitigate the stress from upland development. Living shorelines act to address erosion, incorporate vegetation, stone, sand fill and/or other structural and organic materials, and avoid severing connections among riparian, intertidal and subaqueous areas. Shoreline management and permitting programs should strive to consider cumulative along with local effects during risk assessments and prior to decision-making.

Application of ecological thresholds to coastal management has been limited due to the complexity of multiple factor controls operating on diverse spatial and temporal scales that can confound its utility (Groffman et al. 2006). While examples of dramatic regime shifts in aquatic and terrestrial ecosystems in response to anthropogenic activity (e.g. ecosystem state shifts from corals to fleshy brown macroalgae dominating) have been well documented (e.g. Scheffer et al. 2001, Walker & Meyers 2004), the extraction of key extrinsic factors that are manageable is problematic. For example, while thresholds of 10 to 15% impervious watershed surface resulting in declines of stream ecological health have been reported, others have noted strictly linear declines in species richness in relation to imperviousness (Morley & Karr 2002). Likewise, in this study, impervious surface did not exhibit thresholds in relation to fish communities at the 3 examined spatial scales. This metric may not completely synthesize development stressors important to nekton integrity, thereby confounding results (Karr & Chu 2000, Allan 2004). However, developed land use and shoreline condition were consistently reflected in biotic responses. Identification of landscapes that are at the point of crossing a threshold and further research on underlying mechanisms driving shifts in condition are important next steps for supporting coastal management.

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