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Influence of Land Use on Macrobenthic Communities in Nearshore Estuarine Habitats

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ABSTRACT: Macrobenthic community indices were examined for their ability to characterize the influence of shoreline alteration and watershed land use in nearshore estuarine environments of the Chesapeake Bay, U.S.A. Twenty-three watersheds were surveyed in 2002 and 2003 for nearshore macrobenthic assemblages, environmental parameters (i.e., dissolved oxygen, pH, total suspended solids, salinity, and sediment composition), shoreline condition, and land use. Two indices of macrobenthic biological integrity, benthic index of biological integrity in the nearshore (B-IBI$_N$) and abundance biomass comparison (W-value), were evaluated for associations with environmental and shoreline condition, and riparian and watershed land use. Comparisons between nearshore measures of the B-IBI with offshore values (> 2 m; Chesapeake Bay benthic index of biological integrity [B-IBI$_{CB}$]) were conducted to assess the ability of the index to reflect land use patterns at near and far proximities to shore. Nearshore macrobenthic communities were represented by a total of 94 species (mean number of species = 9.2 ± 0.4 sample^{-1}), and were dominated by the phyla Arthropoda, Annelida, and Mollusca. Temporal variability in environmental conditions and macrobenthic abundance and biomass may be attributable to the notable increase in precipitation in 2003 that led to nutrient influxes and algal blooms. For the biotic indices applied in the nearshore, the highest scores were associated with forested watersheds (W-value, B-IBI$_N$). Ecological thresholds were identified with nonparametric change-point analysis, which indicated a significant reduction in B-IBI$_N$ and W-value scores when the amount of developed shoreline exceeded 10% and developed watershed exceeded 12%, respectively.

Introduction

Coastal plain estuaries have become progressively more degraded due to anthropogenic stressors, evident in increases of hypoxic events, algal blooms, and biodiversity losses. In response, development and use of indicators to define condition, health, and sustainability of estuaries have been emphasized. The characterization of ecosystem condition using integrative indices was initially developed for and applied in freshwater systems (Karr 1981; Kerans and Karr 1994; Karr and Chu 1999). The variability of estuarine habitats delayed similar advances in indicator development for those systems. Over the past decade, varied indicators of anthropogenic effects have been developed specific to estuarine environs including declines in habitat (Dennison et al. 1993), water and sediment quality degradation (Chapman 1996; Dauer et al. 2000), and decreases in diversity of aquatic biota (Whitfield 1996; Deegan et al. 1997; Dauer et al. 2000). In particular, multimetric biological indices, such as benthic indices of integrity, have shown promise as tools for assessing condition in estuaries due to their predictable and integrative response to stressors (Weisberg et al. 1997; Engle and Summers 1999; Van Dolah et al. 1999; Paul et al. 2001; Diaz et al. 2003).

Benthic macroinvertebrates have a long history as indicator organisms due to the ease of collection, their immediate and measurable response to impairments, and the fact that they are mostly sedentary, reflecting local conditions (Pearson and Rosenberg 1978; Dauer 1993). Previously developed indices for estuarine environs that incorporate structure and function of macrobenthic communities include the benthic index of biotic integrity (B-IBI) and abundance biomass comparison (ABC method). The B-IBI is comprised of several community measures specific to habitat type stratified by salinity and sediment composition. Each community measure is weighted equally and scored based on expected values for undegraded sites (Weisberg 1997; Alden et al. 2002; Llanso et al. 2002a,b). The ABC method compares abundance and biomass distributions among species. The stress level of a site can be interpreted based on the relationship between cumulative dominance (k-dominance) curves and summarized with a statistic termed W-value (Warwick 1986; Meire and Dereu 1990; Dauer et al. 1993; Je et al. 2003). These indices can be valuable indicators of estuarine health in critical nearshore environs reflecting gradients in local shoreline condition and watershed development.

Nearshore, shallow-water, tidal habitats provide essential nursery and spawning areas, protection from predators, and foraging opportunities for numerous fish, shellfish, and crustacean species. This critical resource area is under intense and

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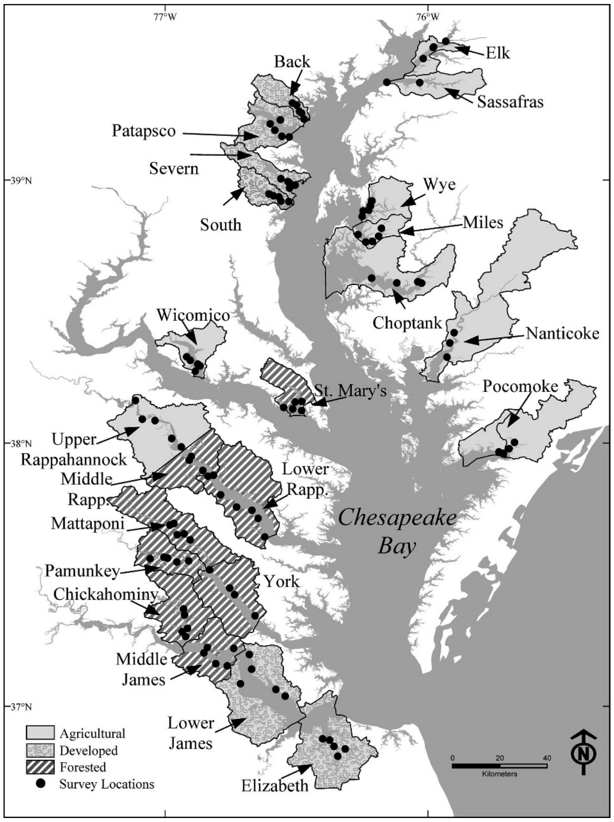

Fig. 1. Location of watersheds and sites surveyed in the Chesapeake Bay in 2002–2003 with predominant land use class described for each watershed (National Land Cover Database).

TABLE 1. Classification criteria for watersheds based on land use percentages derived from the National Land Cover Database (30-m raster coverage).

Land Use Category	Criteria
Forested	> 65% total forest covers (forest, mixed, forest wetland) and < 10% urban
Agricultural	> 50% total agricultural covers (pasture, crop)
Developed	> 50% total urban covers (low and high residential and industrial areas)
Mixed developed	20–50% total urban covers
Mixed agriculture	20–50% total agricultural covers

increasing pressure from a variety of uses and users and generally exists without an operative comprehensive management plan. Throughout the coastal plain of Virginia, the conversion of natural shoreline to stabilization structures is occurring at a rapid pace. Over the past 10 yr in Virginia, it is estimated that 342 km of tidal shoreline have been altered with riprap (stone revetments) and retaining walls (bulkheads; Center for Coastal Resources Management, Tidal Wetlands Impacts data [http://ccrm.vims.edu/wetlands.html]). The cumulative effect of shoreline armoring and watershed development has been demonstrated to drastically reduce available shallow-water habitat structure and associated fish (Beauchamp et al. 1994; Jennings et al. 1999; Jordan and Vaas 2000; Bilkovic et al. 2005) and benthic communities (Dauer et al. 2000; Lerberg et al. 2000; Hale et al. 2004). Land use patterns influence nutrient and sediment input into waterways, with increased nutrient loads corresponding to high levels of agricultural and urban land use in coastal watersheds (Dauer et al. 2000). High nutrient loads may lead to algal blooms, increased low dissolved oxygen events, and losses of trophic complexity and diversity in biotic communities. Watershed land use has also been strongly related to levels of estuarine sediment contaminants, which affect the condition and biodiversity of estuarine benthic communities (Comeleo et al. 1996; Paul et al. 2002; Morrisey et al. 2003; Kiddon et al. 2003; Hale et al. 2004).

The effect of shoreline and watershed land use on adjacent nearshore biotic communities is a critical ecosystem management question. Given the proximity of nearshore benthic communities to upland activities, these communities may be particularly sensitive to changes in land use and developmental pressures. Our objective was to examine the influence of shoreline alteration and watershed land use on nearshore macrobenthic communities using established indices for related estuarine environments.

Materials and Methods

This research is part of a broader study conducted under the Atlantic Slope Consortium, U.S. Environmental Protection Agency (USEPA) Estuarine and Great Lakes Research Program, examining the effect of riparian and nearshore habitat on biological communities, in particular, finfish and macrobenthos. Detailed methodology and complementary information can be found in Brooks et al. (2006) and Bilkovic et al. (2005), and is only briefly described here. Twenty-three watersheds were selected in the oligohaline to mesohaline (0.5–18‰) portions of Chesapeake Bay based on several criteria: land use classifications, salinity regime, and accessibility (Fig. 1). Each watershed was initially categorized as forested, agriculture, developed, mixed agriculture, or mixed developed based on principle land use percentages derived from the National Land Cover Database (30 m raster coverage; Table 1). U.S. Geological Survey (USGS) designated 14-digit hydrologic unit codes (HUC) were used as watershed sampling units. This constrained the number of available watersheds in each land use category. Designations were combined to represent major land use types throughout the Chesapeake Bay: forested, agricultural (includes mixed agriculture watersheds), and developed (includes mixed developed watersheds). Five or more representative watersheds in the oligohaline to mesohaline salinity regime were sampled from each of these three principle land use categories.

In each watershed, 5 sites were selected based on historic finfish seine survey locations between July and August in 2002 and 2003. If 5 historic survey locations were not available within a watershed, then a stratified random design (strata varied depending on watershed size) was applied to select additional sites (Bilkovic et al. 2005; Brooks et al. 2006). Sampling was limited temporally to reduce the potential effect of seasonal variability. In the first year, sites in 14 watersheds were sampled (65 sites total, 5 sites omitted due to sampling or processing inaccuracies), and in the second year, sites in 11 watersheds were sampled (51 sites total; 4 sites removed due to a toxic algal bloom that prevented proper sampling). Two of the watersheds were surveyed in both years to provide points for comparison. The two sample years differed greatly in climatic conditions in both Virginia and Maryland; 2002 was a drought year with low precipitation (108.92 and 106.66 cm, respectively), while 2003 had relatively high precipitation (159.36 and 159.49 cm, respectively; www.sercc.com).

At each site, three replicate benthic samples were collected in the nearshore subtidal ($<$ 2 m in depth) using an Ekman Grab that sampled a surface area of 0.025 m^2. Replicate samples were combined, rinsed into a 0.5-mm sieve to remove excess silt, and preserved in 10% buffered formalin. In addition to biological samples, daytime measures of environmental condition (dissolved oxygen [DO] concentration, pH, total suspended solids [TSS], salinity, and sediment composition) and shoreline condition were obtained. Water quality parameters were measured with a handheld YSI 6820 unit. Two replicate water samples (500 ml each) were analyzed for TSS at each site and the results averaged. On-site visual evaluation and field photographs were used to estimate percentages of shoreline alteration. Percentages were scored on a sliding scale (0–20) of shoreline condition similar to rapid bioassessment protocols developed by the USEPA (Barbour et al. 1999); low scores reflected high levels of alteration, high scores reflected natural systems. Riparian land use (30 m shoreline buffer; 150 m length) near the sampling sites was categorized as percentages of developed, agricultural, or forested land. A small core (60 cc) was taken for sediment characterization (i.e., grain size analysis). Using sieve fractionation, sediment samples were separated into four categories (clay, silt, sand, and gravel) by grain size (Folk 1980); silt and clay categories were combined for analyses. At each site, the proportion of each sediment category was estimated (e.g., percent sand). Supplemental environmental databases were used to assess climatic and hydrologic conditions and compare chlorophyll *a* (chl *a*) levels. Climate and hydrologic data were obtained from the National Climatic Data Center (http://lwf.ncdc.noaa.gov/oa/ncdc.html; precipitation) and the USGS (http://water.usgs.gov/osw/; stream flow). Chl *a* data were acquired from the Chesapeake Bay Program (CBP) Water Quality Database (2002–2003) for the Back and Severn Rivers (http://www.chesapeakebay.net/data/).

All macroinvertebrates were removed from each sample and identified down to the lowest practical taxonomic unit (generally species). Specimens were dried at 175°F for 48 h and combusted at 550°F for 6 h to calculate taxon ash-free dry mass (AFDM). When there were too few of a taxon in a sample to determine AFDM, the taxon was pooled across several samples. Total abundance (number of individuals m^{-2}) and biomass (g m^{-2}) for each site were estimated. Two indices of macrobenthic biological integrity were calculated for each site: the benthic index of biological integrity in the nearshore (B-IBI$_N$) and the W-value (a statistical measure of abundance biomass curve comparisons).

The B-IBI$_N$ was calculated for each site following established procedures developed for the Chesapeake Bay that distinguish metric benchmarks by salinity and sediment regimes (Weisburg et al. 1997; Alden et al. 2002). With this index, sites can be classified as either undegraded or stressed based on how their final score relates to the threshold value of 3 (on a scale of 1 to 5); i.e., sites with scores < 3 are considered stressed. The development and application of the B-IBI in the Chesapeake Bay has been discussed extensively in the literature; detailed methodology may be obtained from numerous resources and will not be repeated here at length (Weisburg et al. 1997; Alden et al. 2002; Llanso et al. 2002a,b, 2003). ABC curves were plotted, and the W-value employed to statistically define the relationship between curves as a comparable measure for standard univariate tests using PRIMER 5 (Clarke 1990; Clarke and Warwick 2001). As proposed by Warwick (1986), the ABC method relies on k-dominance curves to define the distribution of number of individuals (abundance) and biomass among species. The comparison of evenness of abundance to biomass distributions can indicate the level of stress in a community. If the abundance curve lies above the biomass curve, a highly stressed system is indicated, with the opposite relationship observed for unstressed systems. The W-value can be used to quantify the level of stress that a community experiences and is predicted with k-dominance curves. When the biomass curve is above the abundance curve the W-value will be strongly positive. The W-value will be strongly negative when the abundance curve is above the biomass curve, with intermediate cases tending toward zero. The W-value is sensitive to low sample or replicate sizes (Clarke and Warwick 2001), so sites and replicates were combined in each watershed prior to plotting ABC curves. The W-value represented the statistical relationship between abundance and biomass at a watershed level.

Within 21 of the sampled watersheds, data were available from long-term macrobenthic community monitoring efforts initiated in the mid 1980s and conducted by the States of Maryland and Virginia in cooperation with the USEPA CBP (http://www.chesapeakebay.net/). Monitoring data from this program are used to estimate a Chesapeake Bay benthic index of biotic integrity (B-IBI$_{CB}$) developed to assess benthic community health and environmental quality in the Bay (Weisberg 1997; Alden et al. 2002; Llanso et al. 2003). For each major category of watershed land use (developed, agriculture, and forested), previously assessed B-IBI$_{CB}$ scores for habitats outside the nearshore zone ($>$ 2 m depth; based on data from 1998 to 2004) were compared with nearshore habitat B-IBI$_N$ scores estimated in this study. Within the bounds of each watershed (14-digit HUC), B-IBI$_{CB}$ scores from fixed-point and probability-based random sample locations (1998–2004) at depths $>$ 2 m were used to represent offshore B-IBI responses to condition. B-IBI$_{CB}$ values from shallow-water sites ($<$ 2 m) were excluded from any analysis because comprehensive coverage of the sampled watersheds in this study was not available for comparison at this depth range. Watershed level influences, such as land use, were expected to resonate in both nearshore and offshore biotic indices.

Since land use categories were not randomly dispersed throughout the entire Chesapeake Bay (e.g., developed watersheds were clustered in extreme northern and southern locations), one-way analysis of variance (ANOVA) models were used to assess potential relationships among environmental parameters (DO, pH, TSS, and salinity), physical characters (shoreline condition, depth, and percent sediment composition), and watersheds stratified by land use. Watersheds were numerically scored based on predominant land use along a continuum towards forested conditions (developed = 1, agricultural = 2, forested = 3; Table 1). Temporal consistency was assessed between sample years with one-way ANOVA models in SPSS12.0 by comparing annual differences for environmental conditions across all the sampled watersheds, macrobenthic abundance and biomass among watersheds by land use categories, and biotic indices in a subset of two watersheds that were sampled both years (the Back and Severn Rivers). Abundance and biomass estimates were transformed using $\log_{10}$ prior to comparisons to approximate normality.

After obtaining the results of temporal consistency comparisons, data for both years were combined and each biotic index was compared with predominant watershed land use (one-way ANOVA, Tukey's pairwise multiple comparisons), environmental parameters (DO, TSS, pH, salinity, depth, and shoreline condition; Pearson bivariate correlation analysis), and the amount of developed lands (riparian [B-IBI$_N$] or watershed [W-value]; nonparametric change-point analysis). Scatterplots of biotic indices and developed land metrics suggested a potential threshold response, so change-point analysis (nCPA; King and Richardson 2003; Qian et al. 2003) was used to test for the presence of an ecological threshold in the biotic indices (B-IBI$_N$ and W-value) due to developed land use in the riparian area and watershed, respectively. Since the W-value was assessed solely at the watershed scale, change-point analysis could only test the influence of land use at a watershed scale, not on a site (riparian) level. The nCPA detects changes in the mean and variance of a response variable (in this case B-IBI$_N$ and W-value) due to variation in a forcing factor (in this case land use at riparian and watershed 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 two groups that have the greatest difference in means and variance. With bootstrap simulations repeated 1000 times, a distribution of change-points is estimated and illustrated with a cumulative probability curve that describes the probability (frequency) of a change-point 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. Change-point analyses were conducted in S-Plus using the custom function nopar.chngp (Qian et al. 2003).

To examine the relationship between B-IBI scores in the nearshore (this study) versus off-shore of a given watershed, the B-IBI$_{CB}$, consisting of macrobenthic community data from offshore ($>$ 2 m) sites only, was assessed in relation to watershed land use (one-way ANOVA) and compared with the associated B-IBI$_N$ scores (two-sample *t*-test; MINITAB 12.1).

Results

MACROBENTHIC COMMUNITY

Survey locations for macrobenthic communities were nearshore subtidal systems, with depths rang-

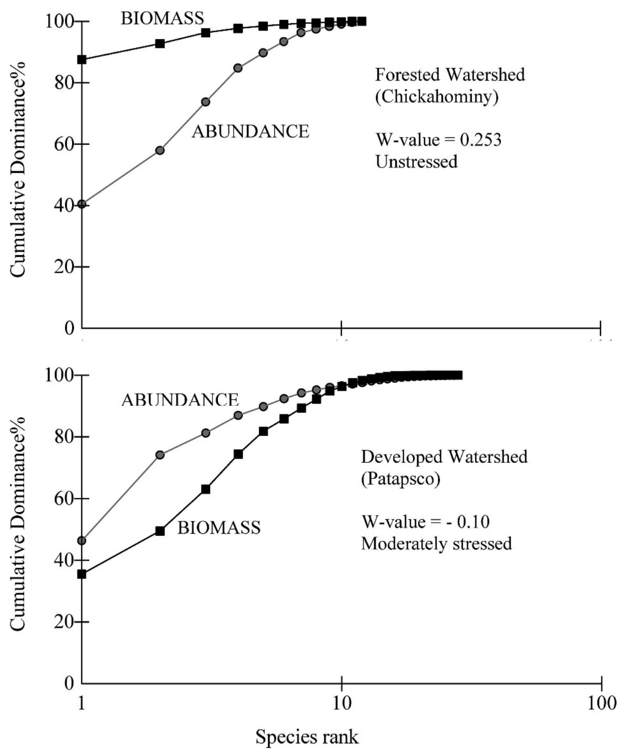

Fig. 2. Example of abundance and biomass k-dominance curves with respective W-values for a developed (Patapasco, Maryland) and forested (Chickahominy, Virginia) watershed.

ing from 0.3 to 1.5 m. A total of 37,211 individuals representing 9 phyla and 94 species were obtained from 116 samples. The assemblage was dominated by 3 phyla comprising 99.3% of the faunal abundance (97.3% of the biomass): Mollusca (13.4% of abundance, 85.7% of biomass), Arthropoda (61.3% of abundance, 5.6% of biomass), and Annelida (24.6% of abundance, 6.0% of biomass). Two groups were representative of samples with notably high average abundances: Arthropoda (2,652 individuals m^{-2}) and Annelida (1,067 individuals m^{-2}). The B-IBI_N site scores covered the entire possible range of scores (1 to 5) demonstrating both undegraded and stressed conditions in the survey area (based on the threshold value of 3). The W-value ranged from -0.10 to 0.43 indicating relatively undisturbed to moderately disturbed watersheds (Fig. 2).

Watershed Characteristics

The estuarine watersheds surveyed showed variability in environmental and physical characteristics among land use categories. The majority of the sites were oligohaline to mesohaline, with a few sites exhibiting fresh or polyhaline conditions at the time of sampling. Comparison of environmental conditions among predominant watershed land use coded categories (developed = 1, agriculture = 2, forested = 3) indicated that pH and DO mean values were higher in developed then forested watersheds (one-way ANOVA; $p < 0.0001$), but both parameters were well within standard optimal habitat suitability ranges for macrobenthos (Table 2). As conditions deviated from a forested watershed, physical characters varied; depth and shoreline alterations increased (one-way ANOVA; $p = 0.002$; $p < 0.0001$). The riverbed composition also varied significantly with salinity, reflecting longitudinal changes. Percent clay-silt remained constant while percent sand increased and gravel decreased with increasing salinity (Pearson bivariate correlation analysis: percent sand, $p = 0.04$, $r = 0.19$; percent gravel, $p = 0.01$, $r = -0.24$).

Temporal Variability

Environmental Conditions

Across watersheds, several environmental parameters, DO, salinity, pH, and TSS, varied significantly between years (one-way ANOVA; $p < 0.0001$, $p <$

TABLE 2. Comparison of data ranges and averages (± standard error) of environmental parameters and physical characters by land use class, with significant differences (one-way ANOVA) at 0.05 (*) and 0.01 (**) levels, for the survey period 2002–2003. Sample sizes (n) are indicated for each watershed land use class. ns = not significant.

		Watershed Land Use			
	Range	Developed (n = 35)	Agricultural (n = 52)	Forested (n = 29)	Significance
Environmental parameters					
Dissolved oxygen (mg l^{-1})	3.6–15.3	9.0 (0.4)	8.4 (0.4)	6.2 (0.2)	**
Salinity (‰)	0.04–23.0	9.6 (1.2)	8.2 (0.5)	8.3 (1.4)	ns
pH	6.2–9.1	8.0 (0.08)	7.6 (0.08)	7.2 (0.1)	*
Total suspended solids (g)	0.003–0.24	0.01 (0.002)	0.02 (0.005)	0.01 (0.002)	ns
Physical characteristics					
Shoreline condition	0–20	9.2 (1.2)	16.1 (0.8)	17.4 (0.7)	**
Depth (m)	0.3–1.5	1.1 (0.03)	1.0 (0.04)	0.9 (0.05)	**
Sediment: clay/silt (%)	0.2–95.0	6.6 (1.7)	6.8 (1.2)	12.2 (1.3)	ns
Sediment: sand (%)	4.3–99.0	74.7 (4.0)	76.8 (3.4)	73.6 (5.4)	ns
Sediment: gravel (%)	0.0–94.7	18.7 (4.1)	16.4 (3.5)	14.2 (4.2)	ns

TABLE 3. Comparison of the averages (± standard error) of environmental parameters and physical characters observed in 2002 and 2003 with significant differences (one-way ANOVA) at 0.05 (*) and 0.01 (**) levels noted. Sample sizes (n) are indicated for each year. ns = not significant.

	Sample Year		
	2002 (n = 65)	2003 (n = 51)	Significance
Environmental parameters			
Dissolved oxygen (mg l^{-1})	7.0 (0.2)	9.3 (0.3)	**
Salinity (‰)	10.7 (0.9)	5.7 (0.5)	**
pH	7.5 (0.06)	7.7 (0.09)	*
Total suspended solids (g)	0.01 (0.0007)	0.02 (0.004)	**
Annual precipitation (cm)	42.88	62.62	
Physical characteristics			
Shoreline condition	14.6 (0.7)	13.7 (1.0)	ns
Depth (m)	1.0 (0.03)	1.1 (0.03)	ns
Sediment: clay/silt (%)	10.7 (2.2)	4.4 (0.8)	*
Sediment: sand (%)	76.9 (2.9)	74.4 (3.5)	ns
Sediment: gravel (%)	12.5 (2.5)	21.2 (3.7)	*

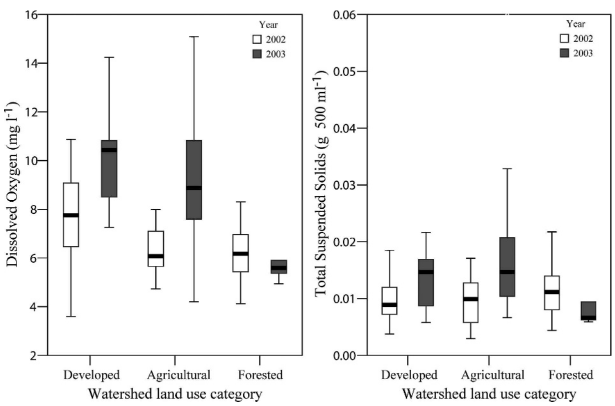

Fig. 3. Average dissolved oxygen and total suspended solids with 95th and 5th percentiles and outliers per land use class and year depicted on each associated bar graph.

0.0001, $p = 0.03$, and $p = 0.01$, respectively). Percent DO, pH, and TSS were higher in 2003, while salinity was higher in 2002. Although the predominant sediment component was sand for both years (76.9 ± 3.0% [2002], 74.4 ± 3.5% [2003]), silt and gravel composition were significantly different in each year with higher percent silt evident in 2002 and higher percent gravel in 2003 (one-way ANOVA; $p = 0.01$, $p = 0.05$, respectively). These changes may be related to differences in the watersheds sampled in a given year or the notable changes in precipitation between the two years (Table 3). Peaks in DO and TSS in 2003 occurred in agricultural or developed watersheds, possibly due to higher nutrient runoff (and resultant algal blooms) in relation to higher precipitation (Fig. 3). Summer (July–September) peaks in chl *a* were noted in August 2003 following high precipitation and stream flow events (Fig. 4) in the highly developed Severn and Back Rivers. While the Severn River exhibited higher average August chl *a* levels in 2003 than 2002, the Back River had high levels in both years due to the highly eutrophic condition of the system. On the Severn River in 2003, the chl *a* summer average was the greatest observed for the period of 1990–2003 coinciding

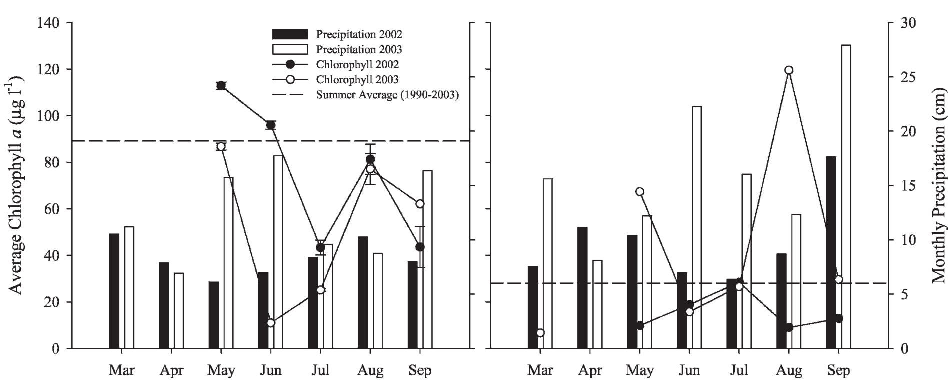

Fig. 4. Monthly average chlorophyll *a* (± standard deviation) and total precipitation for the Back and Severn Rivers during 2002 and 2003. Summer (July–September) chlorophyll *a* averages are depicted as dashed reference lines for the period 1990–2003.

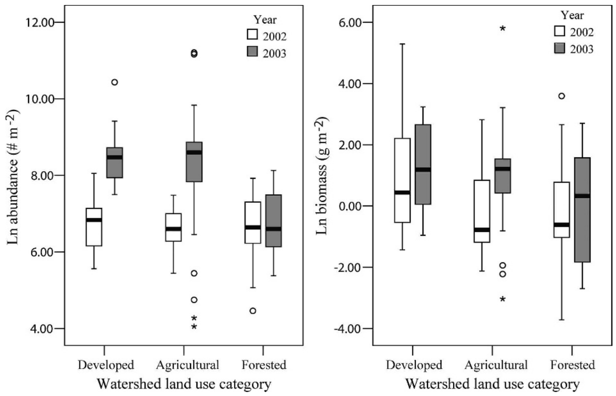

Fig. 5. Average abundance and biomass with 95th and 5th percentiles and outliers per land use class and year depicted on each associated bar graph.

closely with climatic influences. The Back River is a much shallower, smaller system with poor flushing; subsequently, chl *a* summer averages are consistently high in relation to other systems and exceed proposed chl *a* criteria for water quality standards, such as DO, in the Chesapeake Bay (USEPA 2003).

Macrobenthic Community Structure and Biotic Indices

Abundance and biomass were similar among land use categories for 2002. In 2003, there were peaks in abundance and biomass in developed and agricultural watersheds that may be related to high precipitation and subsequent nutrient inputs. Due to these peaks, significant differences were noted between years for abundance (one-way ANOVA; $p = 0.01$) and biomass (one-way ANOVA; $p = 0.01$; Fig. 5). Temporal variation in macrobenthic abundance and biomass was further explored in the subset of watersheds (Back and Severn Rivers) sampled both years. There were significant annual differences between total abundance and biomass for the Severn River watershed (one-way ANOVA; $p < 0.001$, $p = 0.045$) with both biomass and abundance being higher in 2003. In the Back River watershed, abundance differed between the two years (one-way ANOVA; $p = 0.006$), while biomass did not. This was due to higher abundances of *Rangia cuneata* (a relatively large invertebrate) at the site in 2002, while in the following year there was overall higher abundance, but fewer *R. cuneata*. The W-value (which is sensitive to differences in the abundance-biomass relationship) only showed differences between years in the Back River ($p < 0.0001$). The B-IBI$_N$ did not indicate significant differences between years for the two watersheds, so years were combined in subsequent analyses.

Macrobenthic Indices

Environmental Parameters in Relation to Indices

Only in a few instances were environmental parameters correlated with an index ($p \leq 0.05$), and in these cases the r values were low ($r < 0.5$), so the correlations may be spurious. The W-value was correlated with TSS, pH, and salinity ($r = -0.18$, $r = -0.26$, and $r = -0.31$, respectively), and the B-IBI$_N$ was correlated with salinity ($r = -0.49$).

Land Use in Relation to Indices

The highest biotic index scores in this study were associated with forested watersheds (one-way ANOVA; W-value $p = 0.001$, B-IBI$_N$ $p = 0.006$). For the B-IBI$_{CB}$, the lowest index scores were observed in developed watersheds, with agricultural and forested watersheds reflecting similar score ranges (one-way ANOVA; $p < 0.0001$). Change-point analyses indicated that ecological thresholds existed in response to developed land use at the site (B-IBI$_N$) and watershed (W-value) scale. The cumulative probability curve indicated a 98% probability of a change-point occurring at > 20% developed riparian land use for the B-IBI scores. As little as 10% developed shoreline had a 77% probability of resulting in an ecological threshold for macrobenthic communities. On the watershed scale, there was a 98% probability of a change-point at 14% developed land use for the W-value, and at relatively low levels of developed watershed (2%) there was a 60% probability of a change-point (Fig. 6). Both indices exhibited significant macrobenthic community responses when developed lands were between 10% and 12% ($p \leq 0.05$).

Comparison of B-IBI$_N$ and B-IBI$_{CB}$ scores by watershed types indicated that developed and forested watersheds had significantly higher scores for nearshore sites (B-IBI$_N$) relative to off-shore sites (B-IBI$_{CB}$). Agricultural watersheds were similarly scored in both nearshore and off-shore sites (Fig. 7).

Discussion

Macrobenthic indices exhibited patterns related to land use, which has also been observed in other biological components of the shallow-water ecosystem in the Chesapeake Bay (Bilkovic et al. 2005; King et al. 2005; Brooks et al. 2006). Index scores decreased with anthropogenic alterations to the landscape (e.g., developed watersheds), and thresholds were identified for riparian and watershed developed land use (10–12%) beyond which a response in macrobenthic communities is reflected in consistently low scores with reduced variability, which is indicative of stressed conditions. The addition of riparian land use information enhances the discriminatory ability of the indices in a given

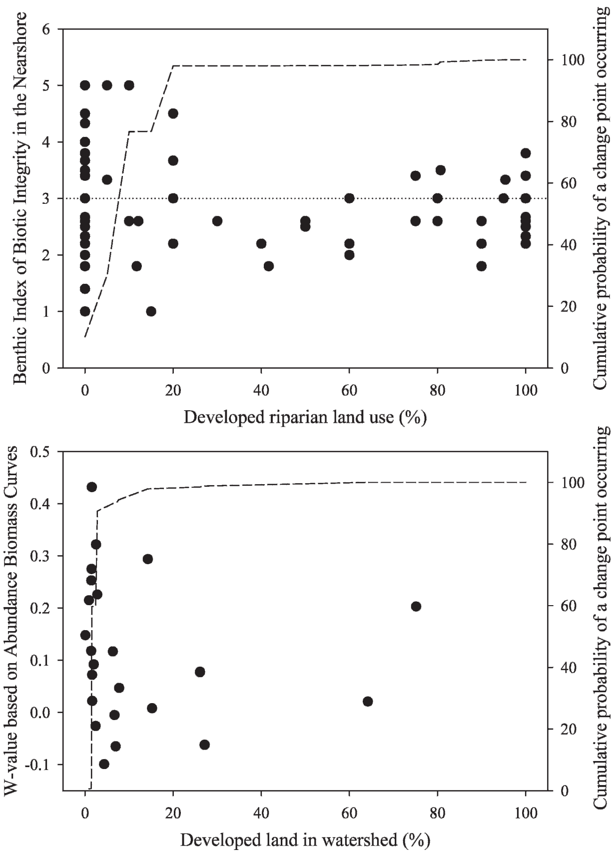

Fig. 6. Results of nonparametric change-point analyses for percent development within the riparian zone (150 m of shoreline) of study sites and the benthic index of biotic integrity in the nearshore (B-IBI$_N$), and percent development within the watershed and W-value. The cumulative probability curve represents the cumulative probability that a change-point occurred at various levels of development. Significant macrobenthic community responses ($p \leq 0.05$) were measured with the B-IBI$_N$ and W-value when developed lands were 10% and 12%, respectively. There was a 95% cumulative probability of an ecological threshold occurring at 20% and 14% developed lands for the B-IBI$_N$ and W-value, respectively.

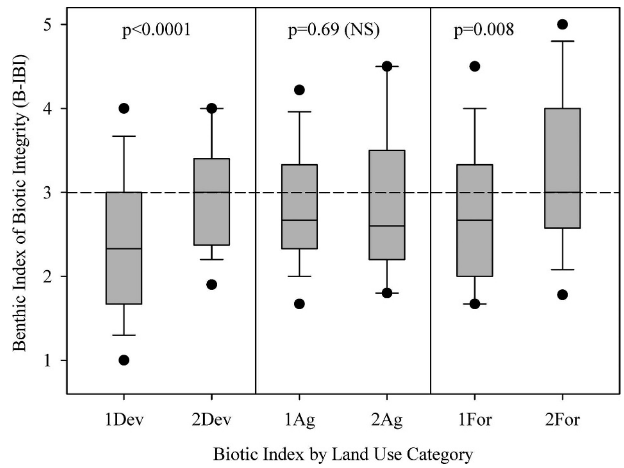

Fig. 7. Comparison of a subset of nearshore and offshore benthic index of biotic integrity scores within each of the watershed land use classes (Dev = developed, Ag = agricultural, For = forested). Offshore conditions (1) are represented with the benthic index of biotic integrity values obtained from the Chesapeake Bay Program (B-IBI$_{CB}$; n = 635) and nearshore conditions (2) are depicted with the benthic index of biotic integrity calculated from our survey (B-IBI$_N$; n = 106); e.g., 2Dev = B-IBI$_N$ data distribution for developed watersheds. Average benthic index of biotic integrity scores with 95th and 5th percentiles and outliers depicted on each associated bar graph.

landscape. The B-IBI$_N$ shows promise for elucidating gradients of condition within landscapes with varying degrees of shoreline alterations. Since riparian forests and wetlands may diminish the effects of urban land use in localized areas (Correll et al. 1992; Osborne and Kovacic 1993), the inclusion of detailed site-specific information may be indispensable for defining condition. If ecosystem health is defined as the optimal state of a system that has been modified by human activity (e.g., the best possible environmental condition in an agricultural or urban landscape; Karr and Chu 1999), then it is critical to designate multiple benchmarks of conditions based on societal choices for a watershed (Regier 1993); e.g., the optimum condition attainable in a highly developed harbor would differ from the condition attainable in a rural agricultural setting. The advantage of multiple references lies in the precision of guidance that can be developed for managers. Information on conditions in an area can be developed relative to an attainable outcome with the difference supposedly amenable to management. Local communities need the ability to gauge the health of their aquatic resources in relation to the social and economic conditions they have chosen for their watershed (Bilkovic et al. 2005). The use of habitat conditions, such as watershed and riparian land use, as indicators may help guide the development of multiple benchmarks of ecosystem condition.

Ecological thresholds that mark break points at which a system or community notably responds (perhaps irreversibly) to a disturbance have been supported in a variety of systems. As in this study, several studies of aquatic systems have noted thresholds ranging between 10% and 20%. 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% and 20% urbanization within streams have been linked with degradation of fish communities (Limburg and 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 and Meyer 2001).

The application of benthic indices show promise in their ability to discern watershed level effects defined by predominant land use patterns (Dauer et al. 2000; Jordan and Vaas 2000; Lerberg et al. 2000; Hale et al. 2004; Holland et al. 2004; this study). In fact, the B-IBI was able to distinguish watershed land use effects for both off-shore (B-IBI$_{CB}$) and nearshore stations (B-IBI$_{N}$). There were notable differences between nearshore and off-shore scores for land use classes. Sites are classified as either undegraded or stressed based on how their final B-IBI score relates to the threshold value of 3 (on a scale of 1 to 5); for instance, sites with scores < 3 are considered stressed (Weisberg et al. 1997). The B-IBI$_{N}$ has a tendency to score developed and forested watershed sites higher than associated B-IBI$_{CB}$ values resulting in differing site classifications (Fig. 7). This may be due in part to the limited number of hypoxic events in the nearshore subtidal (those that do occur are typically from the intrusion of hypoxic deep water due to sustained wind events) relative to deeper estuarine waters (Llanso et al. 2003; USEPA 2003). Our sampling was limited to daytime and the possibility of nighttime hypoxic events cannot be eliminated without further investigation.

Additional evaluation of site or region specific environmental parameters may be useful for explaining variability within watershed types. Variability of environmental parameters among watershed types may be either naturally or anthropogenically induced. Natural variability observed in systems included high clay-silt percentages in forested watersheds that may be due to contributions from vast emergent marsh areas located in several of these watersheds. Other environmental parameters associated with watersheds were more likely due to anthropogenic influences. Depth increased and shoreline condition decreased as conditions deviated from a forested watershed, as may be expected since increases in shoreline hardening have been associated with the reduction in nearshore, shallow-water habitat (Jennings 1999). The most effective index that could be developed for nearshore systems would have the ability to delineate natural from anthropogenically-induced variability, and subsequently target potential restoration areas. A limitation to multimetric indices, such as the B-IBI, is their inability to discern underlying causes of stress (natural or anthropogenic) that lead to delineations of degraded sites. These indices may be used as guidance to select areas that require further diagnostic evaluation.

A critical question to be addressed during any indicator development is whether interannual natural variability, such as climatic features, will elicit responses in biotic communities that result in inconsistent assessments of estuarine condition. The B-IBI$_{CB}$ was specifically designed to account for variable environmental conditions in estuarine systems (salinity, sediment), so differences in land use patterns with associated environmental parameters should not limit indicator use. The ABC method (and W-value) uses a relative relationship between biomass and abundance to define biotic condition and should be applicable in a variety of environmental states. Warwick et al. (1987) noted that this technique could be used in intertidal habitats and that physical disturbance, such as wave activity, did not preclude its usefulness as an indicator. Since we were interested in large scale process influences, such as watershed land use, on biotic communities, short-term temporal variability in site-specific communities may not be critical. In the 2 yr of study, climatic differences were extreme, yet patterns in watershed land use associations and macrobenthic communities were consistent, underscoring the ability of the indices to illustrate spatial variability. Exploring interannual variability in index scores can lead to a better comprehension of driving environmental influences on biotic communities and ultimately enhance indicator performance (Jackson et al. 2000).

The multimetric indices were able to integrate the effects of multiple anthropogenic stresses on macrobenthic communities more effectively than individual metrics. Watershed land use and shoreline condition may be effective representations of integrative measures of stress that relay the state of degradation in a system. Indicators of estuarine health, such as the B-IBI, in combination with landscape and habitat condition descriptions, can be used to elucidate ecological thresholds and guide research on processes and functions affecting ecosystem services. Future research that incorporates shoreline and watershed land use measures may lead to viable management tools with local and regional applications, in particular on small watershed scales.

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