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Classification efficiency of the B-IBI comparing water body size classes in Chesapeake Bay

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ABSTRACT

The Benthic Index of Biotic Integrity (B-IBI) was developed and is currently employed for environmental assessment in Chesapeake Bay. The index consists of a variety of benthic community metrics (e.g. abundance, biomass, diversity, stress tolerance groups, etc.) scored by thresholds applied to seven benthic community habitats (tidal freshwater, oligohaline, low mesohaline, high mesohaline mud, high mesohaline sand, polyhaline mud, and polyhaline sand) This index was verified as being a sensitive and robust tool for summarizing the status of benthic communities. In our study we tested the classification efficiency of the index using new benthic data by characterizing each sample a priori as degraded or undegraded using criteria of sediment contaminant levels, bioassays and bottom dissolved oxygen levels. A primary objective of our study was to test the classification efficiency of the B-IBI in small water bodies connected to larger water bodies of the mainstems of the large rivers of Chesapeake Bay, as well as the efficiency of the index over time (1990 through 2009). The B-IBI was affected by the size of the water body, e.g., index accuracy was higher for water bodies in small watersheds in lower salinity habitats, whereas large water bodies of the mainstem of rivers were better classified by the B-IBI in habitats with higher salinities. Across the seven benthic habitat types overall correct classification was moderate to low and lower for correctly classifying undegraded sites. In general the index metrics showed some deficiencies that suggest improvements could be made by recalibrating existing metric thresholds or selecting new suitable metrics.

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1. Introduction

The need for protecting and assessing aquatic environments has led to classifying the ecological condition of coastal areas and the development of several tools using common ecological indicators to supply synoptic information (Salas et al., 2006). The condition of the macrobenthos is widely used as an indicator in marine environmental assessment. Benthic organisms make good ecological indicators because they are relatively sedentary, unable to avoid deteriorating water/sediment quality, have relatively long life-spans and show marked responses to stress, depending on their species-specific sensitivity/tolerance levels. They also serve a critical role in cycling nutrients and materials between the underlying sediment and the overlying water column (Borja et al., 2000; Dauer, 1993; Dauvin et al., 2007; Ferraro and Cole, 1995).

Numerous biotic indices have been developed to summarize information provided by the status of benthic communities, and many are widely accepted as effective ecological tools (see summary in Diaz et al., 2004). These indices are useful tools of communication of ecosystem condition to environmental managers because they reduce complex scientific data, integrate different types of information, and produce results that can be easily interpreted in the perspective of water quality management (Chainho et al., 2007; Wilson and Jeffrey, 1994). Among these indices, multimetric indices are ecosystem management tools that can provide a robust indication of ecosystem status (Williams et al., 2009). One of these, the Chesapeake Bay benthic index of biotic integrity (B-IBI), was developed for environmental assessment in the Chesapeake Bay, the largest estuary in the United States (Weisberg et al., 1997).

Estuaries are especially difficult to assess due to the high level of heterogeneity of the natural parameters (e.g., salinity, substrata, depth, fine particles, sediment types) produced by the complexity and variability of physical and chemical processes, tidal mixing and salinity gradients (Dauer et al., 2000; Dauvin and Ruellet, 2009;

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Hopkinson and Vallino, 1995). In order to cope with estuarine natural spatial variability, B-IBI metrics and scoring systems were developed independently for each of seven different habitat types. These habitat types were defined based on salinity and sediment type, which were identified as principal factors structuring estuarine benthic biological assemblages in Chesapeake Bay (Llansó et al., 2002; Weisberg et al., 1997). In this way, application of the B-IBI with habitat specific metrics and metric scoring thresholds should minimize the natural variance of these dynamic systems and allow for a more robust assessment of anthropogenic impacts (Dauer et al., 2008). The Chesapeake Bay B-IBI has been verified as being sensitive, stable, robust and statistically sound (Alden et al., 2002).

However, the assessment of estuarine ecological condition using macrobenthic communities may require alternative spatial or temporal stratification schemes of the ecosystem in order to determine ecological status (Dauer et al., 2008; Gibson et al., 2000). A useful monitoring strategy must be regionally and temporally consistent and clearly linked to specific ecosystem management needs. In order to reach these aims, both adaptive management and adaptive monitoring strategies (Boesch, 2006; Borja and Dauer, 2008; Ringold et al., 1996) should be used to periodically improve the understanding of these variability sources and change management remediation and restoration actions as well as monitoring program metrics and sampling designs.

Although assemblages of benthic organisms in estuaries are mainly associated with salinity gradients and sediment type (Llansó et al., 2002; Weisberg et al., 1997), these are not the only environmental factors responsible for species distributions within a certain region of an estuary. Important physical factors such as water depth, shoreline modification, watershed land-use, tidal ranges and/or current velocity have an important role in the variability of benthic community condition (Tenore et al., 2006; Teske and Wooldridge, 2003; Ysebaert and Herman, 2002). These characteristics could interact to affect nutrient levels, primary production and trophic level interactions in the system, producing discernable differences in benthic communities. In this way, Tenore et al. (2006) suggest that estuaries can be divided into geomorphological units or modules that have inherent characteristics such as size, shape, depth, tidal ranges and spatial position within the system.

The Chesapeake Bay has a drainage basin of approximately 166,000 km^2 (Boesch, 2006) and its watershed is comprised of different components, including the mainstem of Chesapeake, large semi-enclosed basins, several large tidal rivers, and multiple small tributaries and embayments (Dauer et al., 2008; Tenore et al., 2006). Each component can be characterized by attributes such as their size, shape, hypsography or tidal regimes (Roy et al., 2001). Welsh et al. (1982) found that the area: volume ratio of small estuaries in Long Island Sound was highly positively correlated with benthic secondary production. Features of the surrounding landscape such as water body size and type have a direct effect on benthic trophic composition (Tenore et al., 2006). In general smaller embayments are affected by smaller watersheds that have a greater surface area: volume relationship with both the shoreline and watershed size. These factors could result in stronger relationships between benthic community structure and function and watershed nutrient and sediment loads as well as contaminant loads.

The primary objective of this study was to assess the spatial variability in classification efficiency of the B-IBI in relation to water body size, given the potential for the large differences in both natural and anthropogenic stressors that might be associated with differences in primary forcing factors associated with water body size. Efficiency of the B-IBI was evaluated at two spatial scales (small discontinuous water bodies versus larger and more continuous water bodies). In addition, an assessment was made to determine if the classification efficiency of the B-IBI has changed over a period of 19 years (1990–2009) both overall and in relation to habitat type and embayment size.

2. Materials and methods

Data from a total of 1003 sites belonging to three different monitoring programs were used in the analysis (Table 1). All samples were collected during the Benthic IBI index period (July 15th–September 30th, see Weisberg et al., 1997) using a Young grab with a sampling surface area of 440 cm^2. Each sample was sieved on a 0.5 mm screen and preserved in the field. Samples were sorted, enumerated and identified to the lowest possible taxon. Ash-free dry weight biomass was determined for each taxon. These samples were selected because sediment contaminant, dissolved oxygen and sediment composition (particle size and organic content) were collected and sediment bioassays were conducted. These additional data allowed each site to be classified as degraded or undegraded following the criteria established in Llansó et al. (2009b).

Sediment contamination was scaled employing criteria established by Long et al. (1995). Two guideline values, the effects range-low concentrations (ERL) and effects range-median concentrations (ERM) were used to delineate concentration ranges for each chemical. Concentrations equal to or above ERL threshold but below ERM values, represent a possible effect range within which adverse biological effects occur occasionally. Concentrations above the ERM value represent a "probable-effects" range at which adverse biological effects would frequently occur. In addition, the mean ERM quotient (M-ERM-Q) was calculated according to Long et al. (1998). The M-ERM-Q is obtained by first dividing the concentration of each chemical contaminant by its respective ERM value and then taking the average of all the resultant quotients. Mean ERM quotients lower than 0.044 are associated with a medium or low risk level, whereas values above 0.044 with a high or very high risk levels (Hyland et al., 2003).

Sites were defined as degraded if any of the following criteria were met: dissolved oxygen concentrations were less or equal to 2.0 ppm, any chemical contaminant concentration exceeded the ERM, more than 10 chemical contaminants exceeded the ERL, M-ERM-Q was higher than 0.044 or sediments were toxic based on the results of *Ampelisca* bioassays. Sites were defined as undegraded if all of the following criteria were met: dissolved oxygen concentrations were greater than 3.0 ppm, no chemical contaminant concentration exceeded the ERM, no more than two chemical contaminants exceeded the ERL, M-ERM-Q was lower than 0.044 and sediments were not toxic based on *Ampelisca* bioassays Llansó et al. (2009b).

The B-IBI was calculated for each site following Weisberg et al. (1997) and Alden et al. (2002) and using corrections as described in Llansó and Dauer (2002). Various measures of benthic community structure, referred to as metrics, are employed to calculate the index. These metrics were previously established and scaled in Weisberg et al. (1997) and Alden et al. (2002) for each of seven benthic habitat types (tidal freshwater, oligohaline, low mesohaline, high mesohaline mud, high mesohaline sand, polyhaline mud, and polyhaline sand) (Table 2). For each habitat type, thresholds were established based on percentiles of each metric from reference conditions, i.e. sites considered to be undegraded with respect to dissolved oxygen, contaminants and sediment organic content (Weisberg et al., 1997). To calculate the index, each metric is scored as 1, 3 or 5 based on its value with respect to the established thresholds. The B-IBI is the mean of these scores. Index values <3, are considered to represent degraded or stressed conditions, whereas values ≥3, are considered to represent undegraded or unstressed benthic community conditions.

Table 1
Environmental monitoring programs employed during the study.

Program	Survey	Number of sites	Sampled by	Sampling method
National Status and Trends	August–September 1998, 1999, 2001, 2002	198	NOAA	Young grab
National Coastal Assessment Program	July–September 2005–2009	222	EPA	Petite Ponar or Young grab
Environmental Monitoring and Assessment Program	July–September 1990–1992, 1996–1999	583	EPA	Young grab

Table 2
Number of sites and means with standard errors of depth, bottom salinity, silt-clay content and dissolved oxygen for each benthic community habitat.

Habitat	Number of sites	Depth (m)	Bottom salinity	Silt-clay content by weight (%)	Dissolved oxygen (ppm)
Tidal Freshwater	117	3.70 ± 0.25	0.12 ± 0.01	56.96 ± 3.29	7.02 ± 0.16
Oligohaline	109	2.93 ± 0.21	2.37 ± 0.12	66.90 ± 3.31	6.76 ± 0.14
Low Mesohaline	143	3.66 ± 0.20	9.79 ± 0.16	60.23 ± 3.17	5.83 ± 0.21
High Mesohaline Mud	206	6.26 ± 0.30	14.94 ± 0.12	82.18 ± 1.17	4.83 ± 0.16
High Mesohaline Sand	119	4.16 ± 0.26	15.24 ± 0.17	10.14 ± 1.19	6.28 ± 0.16
Polyhaline mud	128	6.39 ± 0.59	22.50 ± 0.37	78.90 ± 1.53	5.12 ± 0.18
Polyhaline sand	181	5.56 ± 0.36	24.02 ± 0.34	10.48 ± 0.86	6.07 ± 0.11

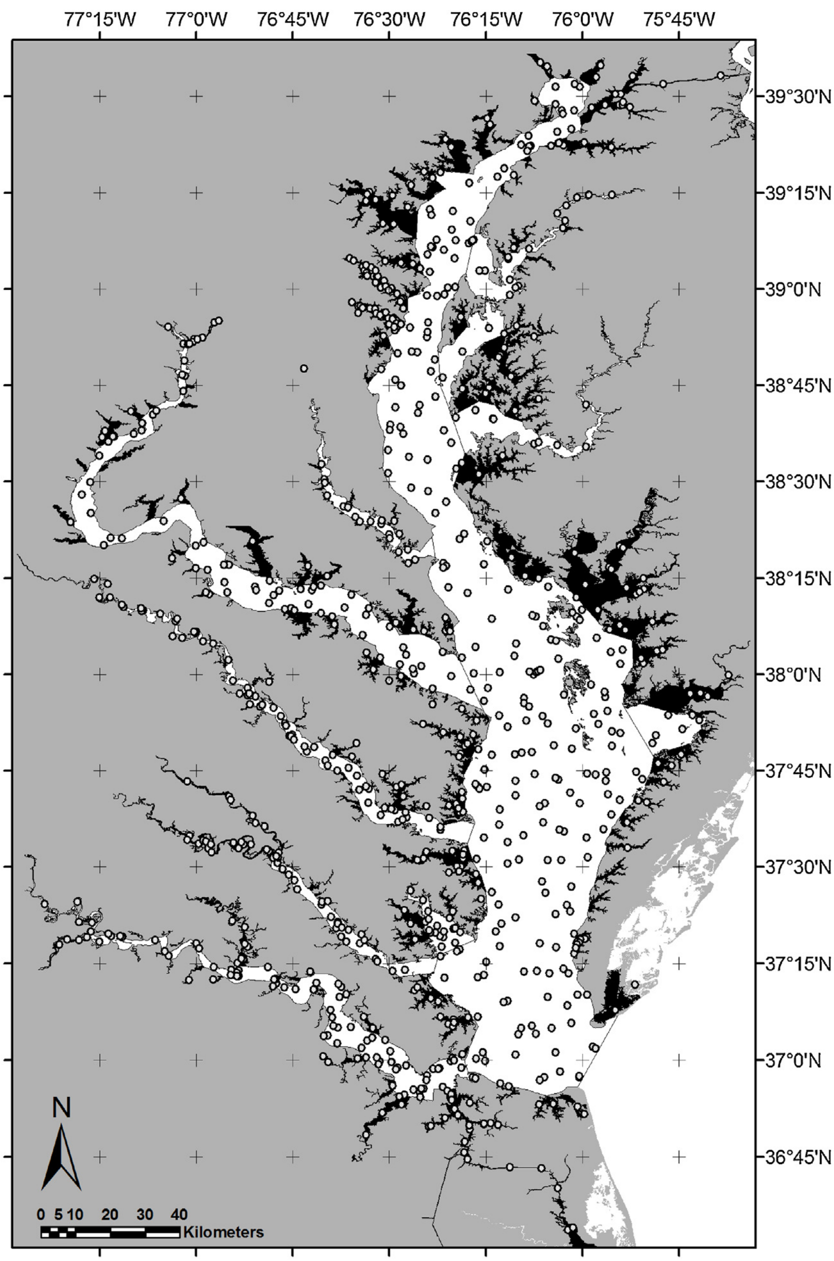

Fig. 1. Map of Chesapeake Bay showing size water body classifications. Lines divide each water body. White areas: large water body. Black areas: small water bodies. Grey points: sampling sites.

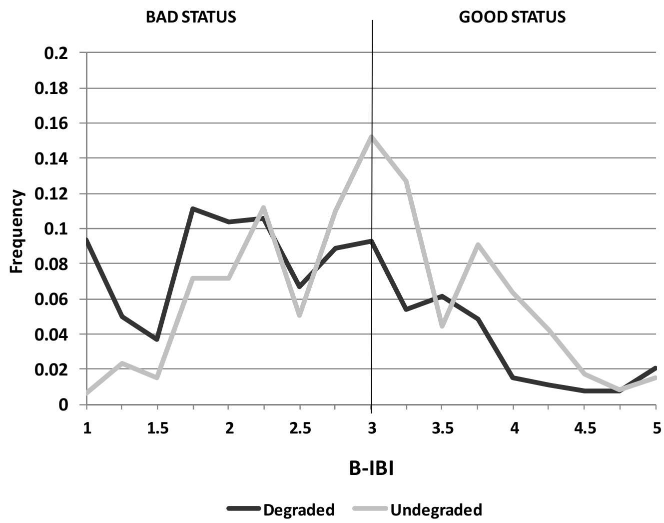

Fig. 2. Frequency of B-IBI values at degraded and undegraded sites.

To determine if B-IBI values are related to the chemical criteria used to differentiate degraded and undegraded sites, the value and accuracy of the index was calculated for each criterion. Since the response of benthic communities to stress could vary among water body size and habitat types, percentages of correct and incorrect B-IBI classifications for degraded and undegraded sites were plotted and compared between water body size, habitat types, and combinations thereof. With respect to water body size, two size classes were established based on surface area of the water body: small for those water bodies less than 100 km^2 and large for those with areas greater than 100 km^2 (Fig. 1).

To determine if specific metrics worked better than others, classification efficiencies of each metric were also obtained. The Mann–Whitney and Kolmogorov–Smirnov tests were used to validate each metric (Weisberg et al., 1997) by testing for differences in the mean of each metric between degraded and undegraded sites, and between sites corresponding to small and large water bodies. A probability level of 0.1 was used to reduce the risk of Type II errors or declaring degraded sites as unstressed (Llansó et al., 2002). Finally, the correct classification of the B-IBI and of each metric was correlated with year of the sampling to assess possible temporal changes in efficiency.

3. Results

3.1. Overall classification efficiency

Among the 521 sites classified as degraded, the B-IBI correctly assigned degraded or stressed status to 350 sites (67%), and among the 482 undegraded sites the B-IBI correctly assigned good status to 265 sites (55%). The distribution of the index values (Fig. 2) showed a higher frequency of undegraded sites classified as stressed than of degraded sites classified as unstressed.

3.2. Relationships with abiotic variables

When degraded sites were evaluated by the B-IBI value, the index accuracy was higher when hypoxic events occurred, classifying 93.2% of the degraded sites with low DO into stressed status (Table 3). For sites classified as degraded due to contaminants, the efficiency rate was lower (67–68%). Mean B-IBI values were always lower in degraded sites with respect to undegraded sites (Table 3); however, mean index values in undegraded sites were generally below 3.0 producing the high percentage of undegraded sites classified as bad or stressed status (Fig. 2). For sites a priori designated as degraded, this designation was due solely to chemical contaminants for 86% of the sites, to only low dissolved oxygen for 4% of the sites and to both chemical contaminants and low dissolved oxygen for 10% of the sites. Comparing the two primary stressors (hypoxia and chemical contaminants), (1) sites with neither stressor had a mean B-IBI value of 2.9 and were correctly classified at a rate of 55%, (2) sites with both stressors had a mean B-IBI value of 1.5 and were correctly classified at a rate of 96.4%, and (3) sites with only one of the two stressors had intermediate values of the B-IBI (2.3–2.6) and intermediate correct classification rate (61.1–81.3%, Table 4).

Table 3
Mean B-IBI in degraded and undegraded sites and percentage of correctly classified degraded sites by B-IBI for all chemical criteria and each criterion.

	Mean B-IBI		Percentage of degraded sites correctly classified by B-IBI
	Degraded	Undegraded	
All chemical criteria	2.5	2.9	67.1
Dissolved oxygen	1.7	2.8	93.2
ERM	2.4	2.7	67.8
ERL	2.5	2.8	67.1
M-ERM-Q	2.5	2.9	66.6

Table 4
Mean B-IBI and percentage of sites correctly classified by B-IBI in different chemical status based on dissolved oxygen and chemical contaminant concentrations.

		Concentration of contaminants	
	Mean B-IBI	Degraded	Undegraded
Dissolved oxygen	Degraded	1.5	2.3
	Undegraded	2.6	2.9
		Concentration of contaminants	
	Percentage of sites correctly classified by B-IBI	Degraded	Nondegraded
Dissolved oxygen	Degraded	96.4%	81.3%
	Undegraded	61.1%	55.0%

3.3. Differences in efficiency among habitat types and water body size classes

The classification efficiency of the B-IBI varied among different habitats and estuarine water body size type (Fig. 3). The lowest percentage of correct classification for large water body sites was observed in tidal freshwater habitats (41%), and for small water body sites the B-IBI showed the worst classification efficiency in high mesohaline sand (33%) and polyhaline sand habitats (30%). Misclassifications in tidal freshwater habitats of large water body sites tended to correspond to degraded rather than undegraded sites resulting in an underestimate of degradation in these areas. Classification of sites in tidal freshwater habitats of small water body sites, however, was generally better (67% correct classification). Meanwhile misclassifications in polyhaline habitats of small water body sites occurred because undegraded sites were classified as degraded (Fig. 3).

3.4. Metric accuracy

Metrics of the B-IBI were not always significantly different between degraded and undegraded sites. In tidal freshwater, most of metrics were significantly different between degraded and undegraded sites only in small water bodies, while in other habitats, significant differences were mainly detected in large water bodies,

Table 5
Mean values for each candidate metric at degraded versus undegraded sites for each water body size class. Shading in the Table: numbers in boldface indicate significant differences in mean values (Mann–Whitney test; $p \leq 0.1$) for each pair while underlined pairs indicate a significant difference in the frequency distribution (Kolmogorov–Smirnov test; $p \leq 0.1$). Abundance–individuals m^{-2}; Biomass g AFDW m^{-2}; Shannon-Weiner using base 2.

	Undegraded × Degraded		Large modules × Small modules	
	Large modules	Small modules	Undegraded	Degraded
Tidal freshwater				
Abundance	4136 < 4312	3927 < 4075	4137 ≥ 3927	4313 > 4075
Abundance of pollution-indicative taxa (%)	45.7 > 44.8	**34.2** ≤ **54.4**	45.7 > 34.2	44.8 < 54.4
Tolerance score	8.38 < 8.39	**8.18** ≤ **9.07**	8.38 > 8.18	**8.39** ≤ **9.07**
Deep-deposit feeders (%)	64.0 > 60.6	**42.0** ≤ **71.3**	**64.0** ≥ **42.0**	**60.6** ≤ **71.3**
Oligohaline				
Abundance	2307 > 2277	3055 > 2120	2307 < 3055	2277 > 2120
Abundance of pollution-sensitive taxa (%)	6.6 < 9.0	6.5 < 9.7	6.6 > 6.5	9.0 < 9.7
Abundance of pollution-indicative taxa (%)	55.2 < 66.4	75.1 > 71.7	55.1 ≤ 75.1	66.4 < 71.7
Tolerance score	7.66 < 8.0	8.37 > 8.2	7.66 < 8.37	8.0 < 8.2
Tanypodinae to chironomidae	**12.7** ≤ **81.7**	**40.8** ≤ **68.4**	12.7 < 40.8	81.7 > 68.4
Abnd. of carnivores and omnivores (%)	23.5 > 15.4	33.4 > 26.8	23.5 < 33.4	**15.4** < **26.8**
Low mesohaline				
Abundance	**3285** ≥ **2123**	**3996** ≥ **2444**	3285 < 3996	**2123** ≤ **2444**
Shannon-Weiner	2.2 > 2.1	**2.5** ≥ **1.4**	2.2 ≤ 2.5	**2.1** ≥ **1.4**
Biomass	7.5 < 8.5	1.2 ≤ 2.4	**7.5** ≥ **1.2**	**8.5** ≥ **2.4**
Abundance of pollution-indicative taxa (%)	12.5 < 15.5	21.2 < 40.6	12.5 ≤ 21.2	**15.5** ≤ **40.6**
Biomass of pollution-sensitive taxa (%)	27.3 < 42.7	13.5 ≤ 16.8	**27.3** ≥ **13.5**	**42.7** ≥ **16.8**
High Mesohaline Mud				
Abundance	**1850** ≥ **1390**	1946 ≤ 2065	1850 < 1946	1390 ≤ 2065
Shannon-Weiner	**2.7** ≥ **1.5**	**2.4** ≥ **1.6**	**2.7** ≥ **2.4**	1.5 < 1.6
Biomass	**0.8** ≤ **1.1**	1.3 ≥ 0.9	0.8 < 1.3	1.1 > 0.9
Biomass of pollution-indicative taxa (%)	28.1 < 28.2	24.0 < 28.1	28.1 > 24.0	28.2 > 28.1
Biomass of pollution-sensitive taxa (%)	**25.7** ≥ **18.4**	**26.6** ≥ **16.5**	25.7 < 26.6	18.4 > 16.5
Abnd. of carnivores and omnivores (%)	**24.6** ≥ **16.2**	**21.3** ≥ **17.2**	24.6 > 21.3	16.2 < 17.2
High Mesohaline Sand				
Abundance	**4482** ≥ **2897**	1487 < 2288	**4482** ≥ **1487**	2897 > 2288
Shannon-Weiner	**2.8** ≥ **2.2**	2.4 > 2.4	**2.8** ≥ **2.4**	2.2 < 2.4
Biomass	**4.9** > **2.5**	0.6 > 0.3	**4.9** ≥ **0.6**	2.5 > 0.3
Abdn. of pollution-indicative taxa (%)	16.6 > 14.8	20.0> 4.6	16.6 < 20.0	14.8 > 4.6
Abdn. of pollution-sensitive taxa (%)	**31.6** > **20.4**	**30.4**≤**67.0**	31.6 > 30.4	**20.4** ≤ **67.0**
Abnd. of carnivores and omnivores (%)	29.2 > 24.3	27.4 < 34.7	29.2 > 27.4	**24.3** ≤ **34.7**
Polyhaline mud				
Abundance	**2580** ≥ **970**	1210 < 2357	2580 > 1210	**970** ≤ **2357**
Shannon-Weiner	**2.9** ≥ **2.0**	2.4 > 2.0	2.9 > 2.4	2.0 < 2.0
Biomass	**4.2** ≥ **2.1**	0.5 < 0.9	4.2 > 0.5	**2.1** ≥ **0.9**
Biomass of pollution-indicative taxa (%)	**17.5** ≤ **38.6**	16.1 < 26.2	17.5 > 16.1	**38.6** > **26.2**
Biomass of pollution-sensitive taxa (%)	25.9 < 28.6	21.8 < 27.5	25.9 > 21.8	28.6 > 27.5
Abnd. of carnivores and omnivores (%)	**39.2** ≥ **25.4**	**35.6** ≥ **19.0**	39.2 > 35.6	**25.4** ≥ **19.0**
Polyhaline sand				
Abundance	3714 < 3914	**3928** ≥ **263**	**3714** ≤ **3928**	**3914** ≥ **263**
Shannon-Weiner	3.2 > 3.1	2.8 > 1.4	**3.2** > **2.8**	3.1 > 1.4
Biomass	5.3 < 6.7	1.3 > 0.6	**5.3** ≥ **1.3**	6.7 > 0.6
Biomass of pollution-indicative taxa (%)	7.9 < 15.5	21.3 < 51.9	**7.9** ≤ **21.3**	15.5 > 51.9
Abundance of pollution-sensitive taxa (%)	47.6 > 38.5	44.6 > 32.3	47.6 > 44.7	38.5 > 32.3
Abundance deep-deposit feeders (%)	22.5 > 22.4	25.5 > 10.3	22.5 < 25.5	22.5 > 10.3

e.g. in high mesohaline mud, high mesohaline sand and polyhaline mud (Table 5). In other habitats, significant differences between degraded and undegraded sites were only detected in some metrics but in both small and large water bodies, e.g., the Tanypodinae to Chironomidae ratio in oligohaline, or abundance in the low mesohaline habitat. Finally, metrics used for polyhaline sand habitats did not show significant differences between degraded and undegraded sites, with the exception of abundance which was significantly different between degraded and undegraded sites in small water bodies (Table 5).

3.5. Differences over time

The classification efficiency of the B-IBI declined from about 70% in 1990 to less than 50% in 2009 (Fig. 4a). This trend was more pronounced in mesohaline and polyhaline sandy habitats. In these sandy habitats, a high decrease of the efficiency of the index was observed in the last years with significant negative correlations between B-IBI efficiency and sampling year. Classification efficiencies for the tidal freshwater and polyhaline mud habitat types remained relatively stable (Fig. 4b and g) with rates of around 50% and 80%, respectively.

Among the metrics of the high mesohaline and polyhaline sand habitats, abundance and biomass showed the largest significant efficiency decline. Abundance showed also a significant temporal decrease in the tidal freshwater and polyhaline mud habitats. While, biomass in high mesohaline mud habitat and abundance of deep-deposit feeders in polyhaline sand habitat showed a significant increase (Fig. 5).

4. Discussion

4.1. Overall classification efficiency

The classification efficiency of the Chesapeake Bay B-IBI index was generally lower in our study compared to the original

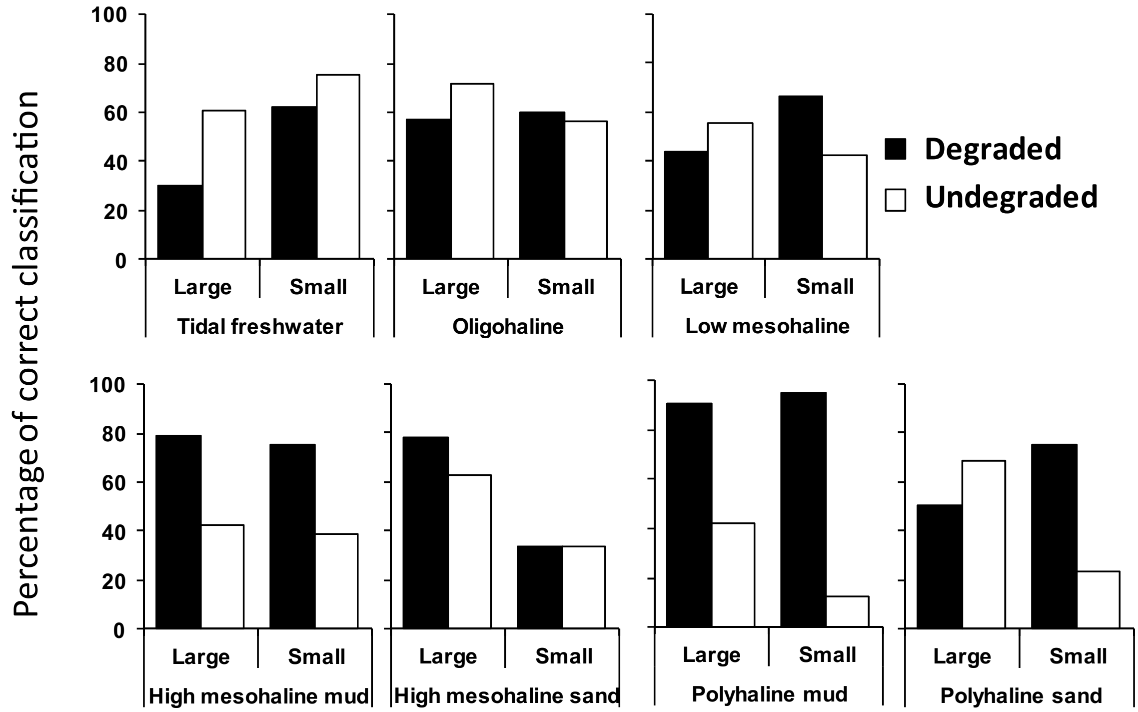

Fig. 3. Percentage of degraded and undegraded sites correctly classified by B-IBI index, in each habitat and water body size class.

validation values when the index was created (Weisberg et al., 1997) and later modified to improve classification efficiency of low salinity habitat types (Alden et al., 2002). Classification efficiencies were low, particularly with respect to correctly identifying undegraded sites. This type of error has been attributed to high levels of natural variability in both freshwater streams (Southerland et al., 2007) and estuaries (Dauvin and Ruellet, 2009). The index had a broad range of values for undegraded sites with three frequency peaks at 3.75, 3.0 and 2.3 (Fig. 2). The peak in frequency at 2.0–2.5 corresponds to a classification of bad status. These low scores may be the result of unmeasured anthropogenic stressors or other natural factors (Van Dolah et al., 1999). The misclassification rate can also be affected by the high natural variability as expressed by the Estuarine Quality Paradox (Dauvin and Ruellet, 2009; Elliot and Quintino, 2007). Some sites could contain benthic communities whose characteristics do not allow them to be classified clearly using a biotic index (Paul et al., 2001). As a result biotic index scores for such communities represent values within ranges of uncertainty for the index (Ranasinghe et al., 2002) where the distributions in B-IBI score for degraded and undegraded sites overlap (Alden et al., 2002). Llansó et al. (2009a) introduced an approach that used criteria that take into consideration habitat specificity and uncertainty. Incorporating this uncertainly into the ecological status classification process improves the utility of the B-IBI in ecological assessments, because this approach focuses on minimizing error that might occur from the inherent variability of benthic communities, the effects of natural stressors, and sampling and methodological error Llansó et al. (2009a). Since, B-IBI values for degraded and undegraded sites overlapped, the B-IBI threshold between good and bad status would be adjusted as the 5th percentile B-IBI score of the undegraded sites, and in order to declare a module/segment as impaired the proportion of sites below this B-IBI threshold had to be higher than 5% Llansó et al. (2009a).

4.2. Relationships with abiotic variables

Sites with hypoxic effects showed the highest correct classification and lowest B-IBI scores compared to chemical contaminants (Table 4). Dissolved oxygen was reported as an important predictor of benthic density, biomass and diversity (Seitz et al., 2009). Dauer et al. (1992) observed that benthic community condition in Chesapeake Bay was strongly related to the frequency of low dissolved oxygen events. Low oxygen effects on benthic communities are influenced by several factors: water depth, duration of water column stratification, the critical oxygen level, the temporal duration of this oxygen level, and other environmental factors such as temperature (Seitz et al., 2009). For the dissolved oxygen criterion we used, the B-IBI is an efficient indicator of community condition (Table 5).

With respect to criteria adopted to establish contamination in sediments, the M-ERM-Q was the most relevant criterion when it comes to defining degraded sites. However, sediment quality guidelines such as the M-ERM-Q, have important limitations and underlying assumptions that should be considered (Long et al., 2006). One of these assumptions is that samples with the same or similar mean guidelines but with a different set of chemical characteristics have the same probability of being toxic. Sediment quality guidelines are correlated with toxicity (Long et al., 2006) but factors other than toxicity probably contribute significantly to differences in the abundance and diversity among sites (Long et al., 2001). Although benthic community condition responded to the M-ERM-Q violations, the B-IBI variability observed in undegraded sites may be attributable to the effects of covarying natural factors such as sediment texture, total organic carbon, depth, or salinity. Research is needed to augment our understanding of the effects of these natural factors on benthic community composition (Long et al., 2006).

There can be situations in which assessments of the relationship between benthic community structure and possible sediment

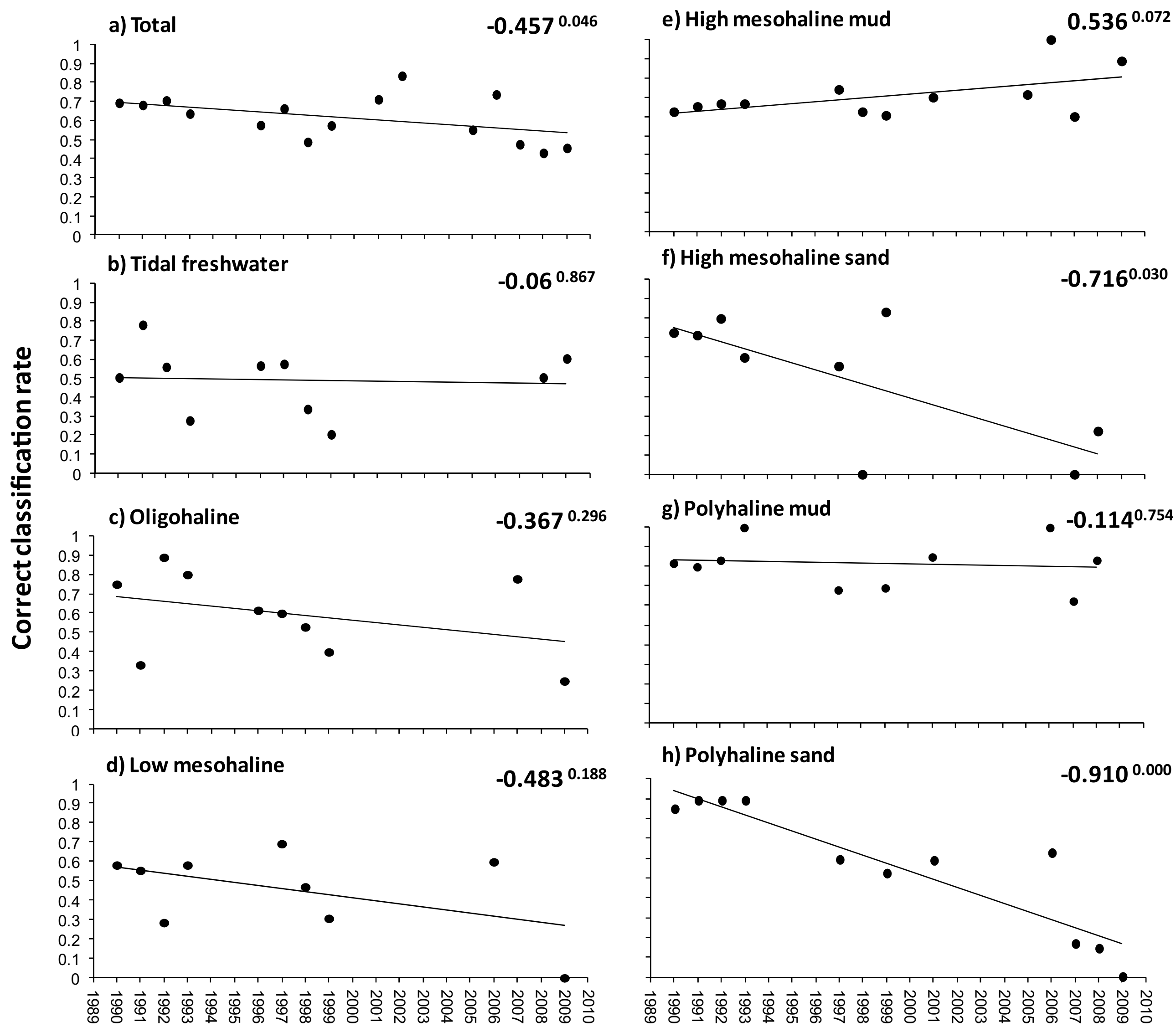

Fig. 4. Pearson correlation (R^p) between B-IBI and years by habitats for years with more than 3 samples per habitat type.

effects are not realistically possible (Chapman and Anderson, 2005). Dauer et al. (2000) did not find a strong relationship between the presence of contaminants and the condition of the benthic communities. In fact, a variety of factors that contribute to toxicity do not affect benthic community status because contaminants may be not present in biologically available forms, or if present in bioavailable forms, they are not harmful to the benthos due to avoidance or resistance by organisms. Alternatively, benthic degradation could be caused by unmeasured contaminants or other natural sources of disturbances or concentrations exhibit small-scale spatial variations that could not be detected during analysis despite adverse effects on the benthos (Hyland et al., 2000, 2003).

Therefore, the use of chemical and toxicological exposure measures to identify sites can produce incorrect differentiations between degraded and undegraded sites, and it is critical to demonstrate a linkage between chemical exposure and biological effects to classify a site affected by contamination (Bay et al., 2007). Other approaches, such as expert judgement present new opportunities for index validation (Weisberg et al., 2008). Expert judgement reduces false negatives of sites affected by unmeasured chemicals, or avoids false positive designations due to contaminants that are measured in the chemical analysis but are tightly bound to sediments and unavailable in situ to benthic organisms (Batley et al., 2005; Ranasinghe et al., 2009; Weisberg et al., 2008).

4.3. Spatial differences in B-IBI efficiency

In previous papers (Alden et al., 2002; Weisberg et al., 1997), correct classifications rates in oligohaline and tidal freshwater areas from Chesapeake Bay were generally lower than correct classification rates in mesohaline and polyhaline areas. Classification efficiencies between undegraded (reference) and degraded sites appeared to increase with increasing salinity. The development and application of biocriteria is more difficult in the tidal freshwater and oligohaline regions because of the high variability in benthic community composition in those habitats (Dauer, 1993). Our results also indicate lower correct classification efficiencies for the B-IBI in low salinity systems. Only polyhaline mud habitats in large estuaries had efficiencies close to those reported in previous papers. This pattern is also consistent with the tenets of the Estuarine Quality Paradox (Dauvin and Ruellet, 2009; Elliot and Quintino, 2007).

Our results showed different patterns of misclassification errors between low and high salinity habitats. In low salinity habitats Type

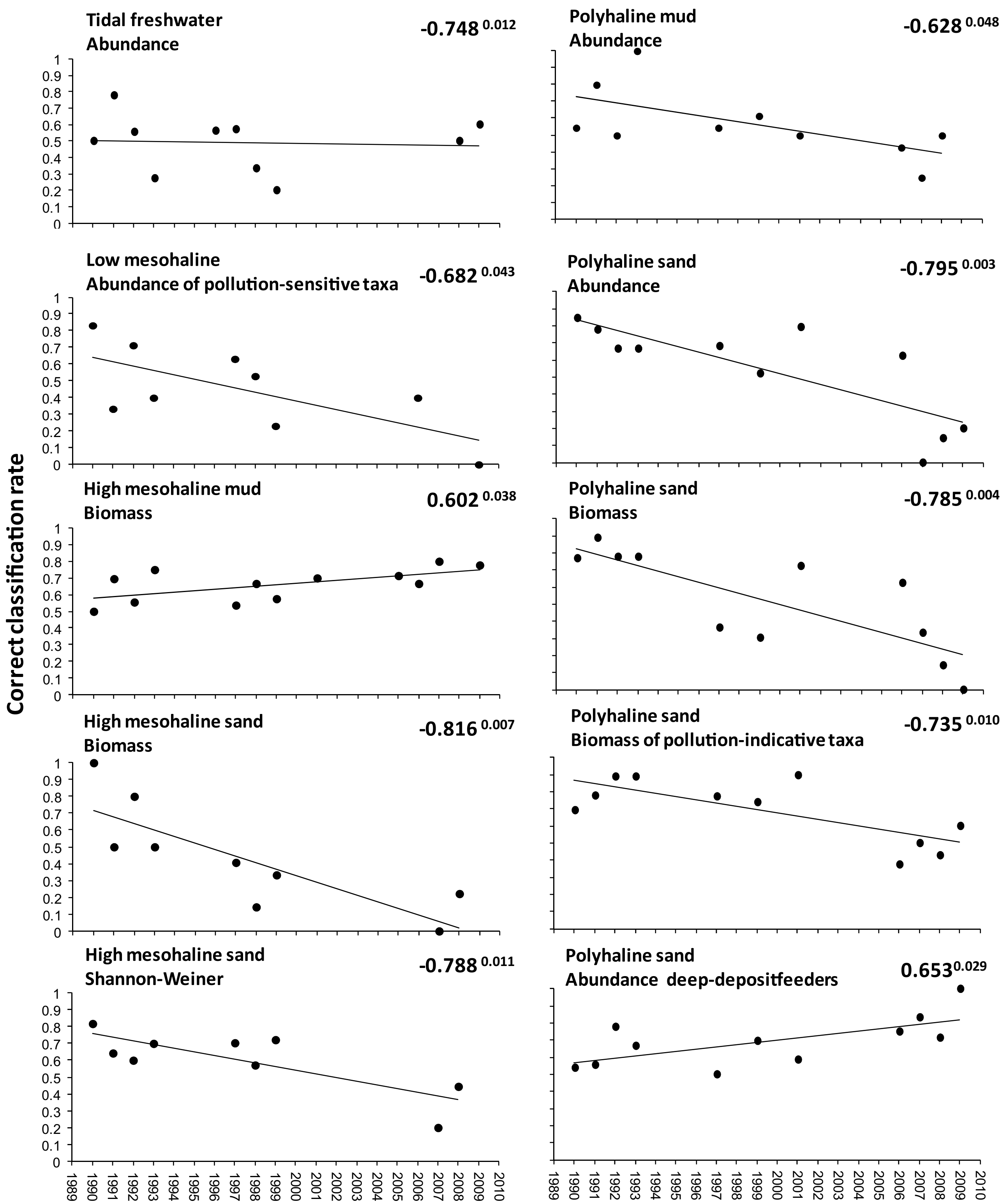

Fig. 5. Pearson correlation (R^p) between metrics and years by habitats for years with more than 3 samples per habitat type. Metrics with significant correlation.

II errors (classifying degraded sites as undegraded) were greater while Type I errors (classifying undegraded sites as degraded) were greater in higher salinity habitats. Consistent with the precautionary principle of improving the power of monitoring programs by minimizing Type II errors (Gray and Bewers, 1996; Ranasinghe et al., 2002) the application of the B-IBI in low salinity habitats is of greater concern than the higher salinity habitat Type I error rates.

The geomorphology of an estuarine region, in this case defined by size classes, affects a suite of primary forcing factors, such as water depth, temperature, energy and disturbances; and/or the characteristics of the surrounding landscape (Tenore et al., 2006). In this analysis, the index tended to have a higher correct classification rate for small compared to large water bodies in tidal freshwater habitats, whereas in polyhaline and mesohaline sand habitats the index had a higher correct classification in larger water bodies. Although parameters such as depth did not previously show any relation to B-IBI values (Dauer et al., 2008), other parameters, such as factors associated with the surrounding landscape, such as land-use type, exhibited strong correlations with benthic community condition (Dauer et al., 2000; Bilkovic et al., 2006).

4.4. Metric accuracy

With respect to individual metrics of the B-IBI, we observed that some metrics did not work well in low salinity habitats, where their classification performance was relatively poor, although there were improvements when metrics were combined (Dauer, 1993; Llansó et al., 2002). In tidal freshwater habitats, some metrics responded to degradation differently in small and large water bodies and can be used for assessment in small estuaries, although recalibration of these metrics is recommended. If some of the data used for the reference distributions were from sites that were naturally stressed, the resulting scoring system could be a less reliable indicator of community condition. Better results may be achieved in the future if the confounding effects of natural stress could be reduced or metrics that were chosen were less influenced by natural stressors. However, this search for the holy grail of metrics to solve the Estuarine Quality Paradox issue has a long and unsuccessful history in estuarine benthic community science.

Selection of the right combination of metrics that reliably differentiates reference from degraded sites, and ensures that the results are ecological plausible (Paul et al., 2001) avoids possible misinterpretations produced by temporal or geomorphologic variability. In this study, only the Tanyponidade to Chironomidae ratio for the oligohaline habitat type showed an efficiency that would be acceptable for this application. The Tanypodinae-Chironomidae percent abundance ratio is a measure of the relative contribution of the Tanypodinae, considered tolerant to pollution (Lenat, 1993), to all the other midges found in a sample. Similar ratios have been used in other studies (Barbour et al., 1996). Including more of this kind of ratio among the list of the proposed metrics could increase the efficiency of the index, and reduce possible temporal or spatial natural variability. Metrics such as total abundance or biomass are useful to differentiating between degraded and undegraded sites, as exhibited in low mesohaline habitats. However, the high natural variability of these metrics, perhaps due to changes in species composition, requires recalibration.

In high salinity muddy habitats, most metrics were able to distinguish degraded from undegraded conditions. Only biomass of pollution-indicative taxa in high mesohaline mud habitats and biomass of pollution-sensitive taxa in polyhaline mud habitats did not show differences between degraded and undegraded sites. Both metrics were previously reported among the “key metrics” (Alden et al., 2002) that appear to be the most important. However, the response of a given species is dependant on the kind of perturbation (Bustos-Baez and Frid, 2003) and classification of species along a sensitivity-tolerance continuum is thus a very difficult task and still a matter of debate (Labrune et al., 2006). Direct experimental testing of responses of different sensitive or indicative pollution groups to natural and anthropogenic stressors could help to choose the most suitable indicator species for each habitat and define more appropriate species for each group avoiding unexpected responses (Llansó et al., 2002; Paul et al., 2001; Van Dolah et al., 1999).

Finally, the low classification rates observed for many metrics for both high mesohaline and polyhaline sand habitats may be due to temporal variability in the metrics for these habitats. As a result, finding suitable metrics for these habitats may prove difficult and/or require periodic recalibration of their thresholds. Variability between size classes for these habitat types may warrant the selection of different metrics for small and large estuaries.

4.5. Temporal differences in B-IBI efficiency

While in low salinities areas efficiency of the B-IBI varied highly among years, decreases of the efficiency was observed in habitats with higher salinity, especially in sandy bottoms, where a high efficiency was expected. The B-IBI was developed employing data from 1971 to 1994. Benthic community structure in the Chesapeake Bay could have changed with time. Ecological assessment in estuaries is particularly challenging due to the seasonal and environmental characteristics that influence benthic communities, such as hydrological conditions that change based on the volume of freshwater discharges (Chainho et al., 2007). In Chesapeake Bay, significant interannual changes in the physical environment and benthos were previously reported in relation to precipitation and freshwater flow (Bilkovic et al., 2006; Kemp et al., 2005; Llansó et al., 2009b). These changes could produce variability in the range of certain metrics. If the B-IBI metrics are less responsive to immediate changes in sediment quality and are sensitive to variable flow regimes as previously reported (Williams et al., 2009), then it will require several successive years in either degraded or undegraded areas before a response to improving conditions can be observed.

4.6. Final considerations

Adaptive management is a structured approach to allow managers to take action in the face of uncertainties. This principle requires testing B-IBI efficiency and modifying it based on new estuarine knowledge and dynamic changes. The issues of the B-IBI detected in this study could be solved or reduced using different approaches that assume natural variability inherent in estuarine systems. Firstly, a new stratification system, in addition to habitat classification, considering water body size could minimize the natural spatial variance of the estuary. On the basis of this new classification system, new metrics might be found that are less influenced by natural stressors. In order to define metrics that differ between degraded and reference areas, establishing reference zones using different criteria, such as Best Expert Judgment (Bay et al., 2007; Ranasinghe et al., 2009; Weisberg et al., 2008), could reduce incorrect differentiations produced by the use of chemical and toxicological measures. Finally incorporating the uncertainly approach Llansó et al. (2009a) into the B-IBI employment for ecological assessments could increase index efficiency and minimize misclassifications

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