Species-abundance-biomass responses by estuarine macrobenthos to sediment chemical contamination

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Abstract

Macrobenthic community responses can be measured through concerted changes in univariate metrics, including species richness, total abundance, and total biomass. The classic model of pollution effects on marine macrobenthic communities recognizes that species/abundance/biomass (SAB) curves vary distinctively in a nonlinear manner with the magnitude of organic enrichment. For example, at moderate levels of organic enrichment, small-bodied opportunistic species boost the abundance curve, while species richness falls. Ratios among the metrics form useful indicators of how the community changes with organic enrichment. However, the classic SAB model is based on organic enrichment effects over small spatial and temporal scales, and the applicability of the SAB model to sediment chemical contamination and across broad natural estuarine gradients is largely unknown. Here, SAB responses were examined with respect to primary gradients in metals and organic chemicals based on an extensive dataset comprising 319 estuarine sites from throughout the northern Gulf of Mexico. Each SAB metric was first adjusted with respect to the three primary natural estuarine gradients, salinity, depth, and sediment silt/clay content. Adjusted SAB relationships varied in their details with respect to different classes of sediment contamination, but all types of SAB stress responses appear to exhibit similar basic characteristics. As in the SAB model, all three SAB metrics were notably low at the highest concentrations of both metal and organic-chemical contaminants. Moreover, rapid decreases in the B/A ratio with increasing contamination supported the concept that relatively long-lived, large-bodied, equilibrium taxa decline markedly at high concentrations of toxicants.

1. Introduction

Macrobenthic assemblages are good biotic indicators of environmental quality (Reice & Wohlenberg, 1993; Gray et al., 1992), especially in estuaries, where the biota may often be stressed by sediment contamination (Diaz & Rosenberg, 1996). However, dynamic estuarine gradients involving both hydrological and sedimentary factors also elicit variable macrobenthic responses (Mannino & Montagna, 1997). Thus, macrobenthic taxa respond individually to sediment contamination, which may also covary with natural estuarine gradients (Rakocinski et al., 1997). Because they are broadly tolerant, many opportunistic estuarine taxa show facultative resistance to pollution stress (Boesch & Rosenberg, 1981). Consequently, detecting the effects of contamination on macrobenthic community structure or on trophic structure can be difficult (Gaston et al., 1998).

The SAB approach developed by Pearson and Rosenberg (1978), is an intelligible method for detecting macrobenthic community responses to envir-
onmental stress based on established principles of benthic ecology. The SAB approach integrates three broad community metrics (species richness, total abundance, and total biomass), into a visual model of how the metrics covary along pollution gradients. For example, the abundance ratio (A/S) reflects changes in dominance and the size ratio (B/A) reflects changes in body size (Warwick, 1986, 1988). An attractive feature of the SAB approach is its use of explicit faunal metrics to define general relationships that can be compared across different ecosystems. Despite the general utility of the SAB approach, the effects of natural environmental gradients still must be reconciled before the effects of anthropogenic stress can be detected in estuarine systems.

Pearson and Rosenberg (1978) presented a graphical model of macrobenthic succession along an organic enrichment gradient, in which an undisturbed, stable macrobenthic community maintains relatively high species richness (S) and biomass (B) as well as moderate total abundance (A) (Figure 1). Several subdivisions along the pollution gradient can be recognized, corresponding to threshold changes in the SAB metrics. All three metrics are minimal at the point of highest stress. A peak of opportunists (PO) represents the point along the enrichment continuum where a few opportunistic species of relatively small body-sizes become superabundant, causing a secondary biomass maximum. As signaled by a decrease in abundance and a rapid rise in species richness as pollution decreases, the ecotone point (E) is marked by a change in dominance to small-bodied opportunists. Increasing species richness and decreasing total abundance with decreasing organic enrichment mark the transition zone (TR), which is represented by assemblages composed of both tolerant and intolerant species. A primary peak in biomass within this region of the continuum represents the point where some larger-bodied taxa respond to the biostimulation provided by moderate organic enrichment.

The SAB model originally exemplified the effects of organic enrichment across local spatial or temporal scales. However, the summary metrics that make up the SAB model are also useful for elucidating community responses across much larger spatial scales (Weisberg et al., 1997). Moreover, the SAB approach may also be useful for assessing macrobenthic responses to other kinds of environmental stress, including different kinds of sediment contamination (Diaz, 1992). Biostimulatory trophic effects at moderate concentrations of contamination character-

ize SAB responses to organic enrichment. Due to a dearth of empirical studies, the behavior of SAB metrics with respect to other kinds of anthropogenic stress, such as sediment chemical contamination, is largely unknown (Gray, 1989).

The purpose of this study was to examine broad-scale relationships between SAB metrics and estuarine gradients, with an emphasis on discerning effects of sediment chemical contamination. Three specific objectives were: (1) to identify relationships between SAB metrics and broad natural estuarine gradients; (2) to examine how SAB metrics vary with respect to sediment chemical contamination, after removing the influence of primary natural gradients; (3) to compare SAB relationships representing sediment metal and organic chemical contamination with the classic SAB model.

2. Methods

2.1. Field Sampling

The extensive dataset for this study originated from the U.S. Environmental Protection Agency’s (U.S. EPA) Environmental Monitoring and Assessment Program—Estuaries (EMAP-E) for the Louisianaian Province, and represents estuaries from throughout the northern Gulf of Mexico, USA (Summers et al., 1991, 1993; Macauley et al., 1994). During July–August of 1991 and 1992, 333 sites were sampled from Anclote Key, Florida, to the Rio Grande River, Texas. This
study region covered 25,725 km² of estuarine area. A surface area-based, probabilistic sampling design was used to equitably represent regional estuarine resources (Summers et al., 1991). Two hundred and one (201) probability-based sites (i.e., base sites) were sampled following a stratified random design. One-hundred and thirty-two (132) additional sites mostly represented areas of known or suspected environmental degradation. Eighteen percent of all the sites occurred in large estuaries (> 250 km²), sixty-nine percent in small estuaries (> 2 km² and < 250 km²), and thirteen percent in large tidal rivers (> 250 km², aspect > 18) with measurable tides (> 2.5 cm) (e.g., Mississippi River).

Loran-C (Long-range radio navigation) was used to locate sites, where water quality was measured and sediments were sampled for contaminants, sediment composition, and macrobenthos (Heitmuller & Valente, 1991; Summers et al., 1991). At each site, three benthic grabs were taken using a Young-modified VanVeen grab (Theodore Young Engineering, Sandwich, Massachusetts, USA) that sampled a sediment surface area of 440 cm². Small single superficial sediment cores (60 cc) were removed from the top 2 to 3 cm of each grab and frozen for later analysis of contaminants and sediment characteristics (Heitmuller & Valente, 1991). Macrobenthic samples were washed on a 0.5 mm-mesh sieve, preserved in 10% formalin-neph bean solution, and stored for at least 30 d prior to processing. Instantaneous water quality parameters, measured near the bottom with a Datasonde Surveyor II interfaced with an on-board computer (Hydrolab Corp), included water temperature, salinity, dissolved oxygen, and pH. Water depth was measured and sediment redox-potential discontinuity-depth (RPD) was determined from the sediment horizon. Diurnal tidal changes in depth are low in this region, averaging only 0.3 m in amplitude.

2.2. Laboratory Methods

Macrobenthic samples were rewashed in the laboratory on a 0.5 mm-mesh sieve, and transferred from formalin to 70% ethanol solution. All macroinvertebrates were sorted, identified to the lowest practical taxonomic level (usually species), and enumerated using stereo and compound microscopy. As a quality control measure, ten percent of all samples were resorted. When resorting showed more than 10% error in removal of organisms, the batch of 10 samples associated with the deficient sample was reprocessed.

Voucher collections were maintained and identifications were verified by expert taxonomists.

Benthic biomass was determined following a standard protocol (U.S. EPA, 1995). In order to standardize biomass measurements, benthic samples were preserved in 10% formalin for at least one month so that weight loss from preservation had stabilized. Biomass was obtained for each dominant macrobenthic group, such as amphipods, polychaetes, and bivalves. Soft-bodied organisms were dried and weighed directly after being identified, whereas hard bodied organisms, such as small molluscs and echinoderms, were acidified with HCl to remove calcium carbonate prior to weighing. Large molluscs (> 2 cm) were shucked instead of being acidified. Organisms were placed in a drying oven at 60°C for 24 hours and stored in a desiccator until weighed. Using a microbalance, weights were taken to the nearest 0.1 mg at successive 24 hour intervals until they stabilized (i.e., by 5% margin).

Sediments were characterized following the methods of Folk (1974) and Plumb (1981). Sediment silt/clay content was measured using standard procedures. Total organic carbon (TOC) was determined by combustion of an acidified, dried, and ground subsample using a Leco CR-12 carbon analyzer. Sediment characterizations included the analysis of sample duplicates, laboratory blanks, reagent blanks, and standard reference materials associated with each batch of ten samples (Heitmuller & Valente, 1991).

Concentrations of approximately 150 sediment contaminants were determined by established analytical chemistry methods (Summers et al., 1993), including 15 metals, 27 alkanes, 43 petroleum aromatic hydrocarbons, 20 polychlorinated biphenyls (PCBs) and 24 pesticides (MacLeod et al., 1985; Krah et al., 1988). Quality assurance procedures for sediment chemical analyses followed Taylor (1987).

2.3. Data analysis

2.3.1. Environmental Variables

Environmental variables of concern for this study included eight natural variables and 13 contaminant variables. Leading variables representing primary natural gradients included salinity, site depth (m), and percent silt/clay (Rakociński et al., 1997). Many of the contaminant variables represented concentrations (mg/g dry weight) of individual trace metals, including nickel, chromium, tin, zinc, lead, mercury, and arsenic (Ni, Cr, Sn, Zn, Pb, Hg, As). Copper,
silver, and cadmium (Cu, Ag, Cd) were excluded from analyses a-priori, because their concentrations always occurred well below recognized 10% biological effects threshold levels (Long et al., 1995). The remaining contaminant variables were treated as composite measures, including total alkanes, total PCBs, total low molecular weight polynuclear aromatic hydrocarbons (PAHs), high PAHs (PAH+-), total DDTs, and total chlorinated compounds (CHL) (ng/g).

2.3.2. Principal component analysis
We used rotated Principal Component Analysis (PCA) to identify interrelated suites of environmental variables representing complex environmental factors (Rakocinski et al., 1997). Varimax rotation maximally resolved variation in the 21 original variables and distinguished natural variables from contaminant variables. The first five PCA factors had eigenvalues greater than one and together explained 70.8% of the covariance in the original 21 transformed variables. Variation in the contaminant variables dominated this analysis, and they accounted for a large portion (13/21) of the variables and variance included in the PCA. Accordingly, the first PCA factor showed high loadings by metals, and moderately high loadings by silt/clay, total organic carbon (TOC) and alkanes. Covariation in sediment metal concentrations mainly drove PC1, as most metals (5/6; except mercury) loaded between 0.86 and 0.93 on this axis. PC2 showed high loadings by organic-chemical contaminants, including both PAH categories, PCB's, and total chlorinated compounds, and moderately high loadings by Hg and DDT.

2.3.3. SAB metrics
After adjusting SAB metrics for the influences of primary natural gradients, residuals were examined with respect to the first two PCA axes in order to assess SAB responses to sediment contamination. Estimates of species richness, total abundance, and total biomass were obtained by summing up the number of species, the number of benthic organisms, and the biomass of major benthic groups for sets of three grabs representing each site. Of the 201 probability based sites, 195 were complete for all environmental variables needed for adjusting SAB relationships, as well as for the three SAB metrics. These 195 sites provided a baseline used for adjusting the SAB metrics for the effects of three primary natural gradients identified by prior analysis, salinity, depth, and sediment composition (Rakocinski et al., 1997).

All three SAB metrics were transformed as ln (X + 1) to reduce skewness. To adjust for the heteroscedastic increase in log biomass with depth, transformed biomass data were weighted by the reciprocal of their associated depth values (i.e., 1/depth × ln (biomass + 1)). After excluding one extreme biomass value, the final regression of this transformed metric was homoscedastic and unrelated to depth. Multiple regression models using the remaining 194 base sites also were used to adjust for trends in species richness and total abundance with respect to the primary natural gradients. Independent variables were included in multiple regression models when they contributed significantly (i.e., α < 0.05) to the fit. Quadratic terms also were included in these models where non-linear, parabolic relationships were indicated. For all 318 complete sites (i.e., excluding 15 disqualified observations from the original 333), SAB metrics were adjusted for the effects of natural gradients by calculating residuals from multiple regression models.

SAB relationships were examined with respect to sediment contamination by averaging adjusted SAB metrics within intervals of 0.5 on each of the two contamination scales (i.e., PC1 & PC2). PC1 represented sediment metal contamination, while PC2 represented sediment organic-chemical contamination. Individual sites which were isolated either within the high end of PC1 (low end of metal levels) or within the low end of PC2 (high end of organic chemical levels) were excluded from the analysis (i.e., n = 317). Hence, the number of data points per interval of PC1 ranged between 3 and 75, and the number per interval of PC2 ranged between 3 and 63. After reversing their signs to reflect the same directionality in contamination as depicted in the original SAB graphic model, average PC1 interval values ranged between −1.7 and 2.7 and average PC2 interval values ranged between −2.6 and 1.7. Average values for all three SAB metrics were standardized on a scale ranging between zero and one. SAB curves were obtained by fitting averaged SAB values within contamination gradient intervals using Distance Weighted Least Squares (DWLS) with the tension parameter set to 0.05 (SYSTAT; Wilkinson et al., 1992). This procedure produces nonlinear curves that are flexible with respect to individual data points.
Table 1. Multiple regression model of the number of benthic species as a response to the three primary natural estuarine gradients for 194 base sites. DEPTH2 = squared depth term.

<table>
<thead>
<tr>
<th>Dependent</th>
<th>N: 194</th>
<th>Multiple</th>
<th>Squared</th>
</tr>
</thead>
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<tr>
<td>Var: Ln</td>
<td>R: 0.67</td>
<td>Multiple</td>
<td>R: 0.45</td>
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<td>(No. BSP + 1)</td>
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Source

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<tr>
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<th>DF</th>
<th>Mean square</th>
<th>F</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
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<td>4</td>
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<td>38.65</td>
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<tr>
<td>Residual</td>
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Variable

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<td>Salinity</td>
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<tr>
<td>Depth</td>
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<td>0.611</td>
<td>2.89</td>
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<tr>
<td>Depth2</td>
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<td>0.191</td>
<td>-3.29</td>
</tr>
<tr>
<td>Silt/Clay</td>
<td>-0.485</td>
<td>0.152</td>
<td>-3.18</td>
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Table 2. Multiple regression model of the total benthic abundance as a response to two significant primary natural estuarine gradients for 194 base sites. DEPTH2 = squared depth term.

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<td>multiple</td>
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Source

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Variable

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<th>STD error</th>
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<tbody>
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<td>3.34</td>
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<td>Depth</td>
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<td>3.10</td>
</tr>
<tr>
<td>Depth2</td>
<td>-1.297</td>
<td>0.383</td>
<td>-3.38</td>
</tr>
<tr>
<td>Silt/Clay</td>
<td>-0.657</td>
<td>0.296</td>
<td>-2.22</td>
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3. Results

3.1. SAB metrics vs. natural gradients

Trends were apparent between species richness and all three primary natural gradients: species richness increased with salinity, varied quadratically with depth, and decreased with sediment silt/clay content (Figure 2). Forty-five percent of the variation in species richness was explained by a multiple regression model in which both salinity and silt/clay effects were linear, and in which depth effects were quadratic (Table 1). Scatterplots of adjusted species richness for the inclusive dataset of 318 sites showed flat and even dispersion patterns, indicating that the effects of the three natural gradients had been effectively removed (Figure 2).

Trends were also apparent between total abundance and the three primary natural gradients: total abundance increased with salinity, varied quadratically with depth, and decreased with sediment silt/clay content (Figure 3). However, only 8.5% of the variation in transformed total abundance was explained by the final multiple regression model in which depth and silt/clay effects were included (Table 2). The lack of an apparent salinity effect in the final regression model indicated that the effect of this gradient was eliminated through correlations involving the other two gradients. Significant depth effects were quadratic, whereas total abundance decreased linearly with silt/clay content. Scatterplots of adjusted total abundance values for the inclusive dataset comprising the 318 sites showed flat and even dispersion patterns against all three natural gradients (Figure 3).

Relationships were not apparent between total biomass and either of two primary natural gradients, salinity and sediment silt/clay content (Figure 4). However, log biomass increased with depth, although the dispersion pattern was heteroscedastic. After log biomass values were weighted by the reciprocal of their associated depth values, scatterplots of the resulting adjusted biomass values against all three natural gradients showed flat and even dispersion patterns (Figure 4). Since no other trends remained between transformed SAB metrics and the three natural primary estuarine gradients, the adjusted SAB data could subsequently be examined with respect to sediment contamination.
Figure 2. Plots of the number of benthic species vs. three primary natural estuarine gradients before and after adjusting according to the multiple regression model of Table 1.
Figure 3. Plots of total benthic abundance vs. three primary natural estuarine gradients before and after adjusting according to the multiple regression model of Table 2.
Figure 4. Plots of total benthic biomass vs. three primary natural estuarine gradients before and after adjusting for the heteroscedastic increase in log biomass with depth.
3.2. Adjusted SAB metrics vs. contaminant gradients

Interpretable relationships were obtained by fitting adjusted SAB data across implicit contaminant gradients (i.e., PC1 & PC2). All three SAB metrics reached their minima at the highest levels of metal contamination (PC1), implying that few organisms tolerated the worst sediment contamination (Figure 5A). The B/A ratio also reflected relatively small body sizes at high metal concentrations. All three SAB metrics climbed rapidly as metal contamination decreased, leading to a peak in the number of presumably tolerant species under moderately high metal contamination. An ecotone point was indicated where both species richness and total abundance fell off together with decreasing contamination. Beyond the ecotone point, a transition zone of mixed tolerant and intolerant taxa was marked by a primary biomass maximum and a continued increase in both species richness and total abundance. An increasing B/A ratio leading up to the primary biomass peak reflected a corresponding increase in body size. As metal contamination decreased to relatively low levels, a primary maximum in total abundance coincided with declining biomass and increasing species richness. A decreasing B/A ratio as metal contamination declined also indicated a reduction in body size. Finally, species richness stabilized, total biomass moderated, and total abundance declined slightly at the lowest levels of metal contamination.

When examined with respect to organic-chemical contamination (i.e., PC2), all three SAB metrics were relatively low at the highest sediment levels (Figure 5B). However, a few small-bodied opportunistic taxa apparently occurred in large numbers where organic chemical contamination was highest, as shown by both relatively high A/S and low B/A ratios. As levels of organic chemicals decreased from very high levels, a secondary peak appeared in the number of presumably tolerant species. An ecotone point was indicated by depressions in all three SAB metrics as organic-chemical levels decreased. Beyond the ecotone point, an implied transition zone of both tolerant and sensitive taxa presumably contributed to the rapidly increasing biomass, and to the continued increase in species richness and total abundance. The B/A ratio over this range of contamination indicated an increase in body size across the transition zone. A primary biomass maximum marked an apparent biostimulation point at moderate levels of organic chemicals, beyond which species richness and total abundance increased rapidly. The resultant decrease in the B/A ratio implied that body size decreased somewhat at the lowest concentrations of organic chemicals. Finally, all three SAB metrics reached their maxima at the lowest levels of organic chemicals.

SAB relationships could be used diagnostically to identify critical levels of sediment contamination through correlations between individual contaminant variables and complex contamination gradients (i.e., PC factors). Concentrations of selected sediment contaminants marking critical changes in SAB relationships were extrapolated from contamination gradients (i.e., PC1 & PC2) (Figure 6). For example, a value of \( \approx 0.25 \) on the PC1 scale reflected the SAB ecotone point on the metal contamination gradient, which in turn corresponded with specific concentrations of Zn, Sn, and Ni. In contrast, a value of \( \approx 0 \) on the PC2 scale corresponded with the SAB biostimulation point on the organic-chemical gradient, and with concentra-
resistant and resilient benthic assemblages consisting largely of eurytolerant opportunists (Boesch & Rosenberg, 1981). For example, in Nueces Bay, Texas, Manino and Montagna (1997) found strong associations between physico-chemical variables such as salinity, sediment composition, and nutrient concentrations, and broad macrobenthic metrics, including biomass, abundance, and diversity. Thus, regardless of the approach used to measure macrobenthic effects in estuarine systems, the effects of natural gradients must often be removed before anthropogenic stresses can be detected. Indeed we saw that SAB relationships varied with respect to the three primary natural estuarine gradients in ways that could obscure SAB responses to sediment contamination; however, properly adjusted broad-scale SAB relationships varied meaningfully with sediment contamination.

Despite obvious environmental health concerns, the macrobenthic effects of sediment chemical contamination are not well understood (Gray, 1981; Diaz, 1992; Dauer, 1993; Dauer & Alden, 1995), although effects of organic enrichment are well documented (Weston, 1990). Dauer (1993) noted that contaminated sediments pose a multitude of stresses on benthic communities that collectively lead to reduced biomass and species richness, and to dominance by shallow-dwelling opportunistic taxa. Because of complex interactions involving varied chemicals, their concentrations, sensitivities of individual species, and varied life-history responses, Gray (1981) maintained that a general model of macrobenthic effects from chemical pollution would be difficult to apply. In addition, Underwood (1989) pointed out that macrobenthic responses to stress may vary unpredictably, depending on the timing, magnitude, and complexity of stressors. Others have taken the more unified view of macrobenthic effects by placing all forms of stress on a sliding scale of disturbance (Johnson, 1970; Pearson, 1981; Rhoads & Germano, 1986; Diaz & Rosenberg, 1996).

Recently, Diaz and Rosenberg (1996) recognized that single stressors seldom act alone in the real world, and that macrobenthos seem to react analogously to various kinds of stressors, including sediment contamination. The basic idea is that any environmental stressor, whether anoxic, toxic, eutrophic, or physical, resets the macrobenthic successional sequence to an earlier stage. Indeed, Wilson and Jeffrey (1994) pointed out that few of the 90 opportunistic species distinguished by Pearson and Rosenberg (1978) exclusively characterize organic enrichment condi-

**Figure 6.** Sediment concentrations of selected contaminants vs. contamination gradients for metals and organic chemicals. (SN = diamonds; NI = circles; ZN = squares) (PAHLMW = low molecular weight PAHs (squares); PAHHMW = high molecular weight PAHs (circles)).

Tions of both low and high molecular weight PAH’s of about 145 ng/g.

4. Discussion

Using the SAB approach to detect effects of environmental stress facilitates comparative assessments of different macrobenthic communities (Warwick, 1988). Indeed, the utility of the SAB approach has stimulated recent efforts to develop estuarine biotic indices based on precepts of the SAB model (Weisberg et al., 1997). Yet, even when using these broad metrics as indicators of stress, the extraneous effects of natural gradients on macrobenthos must still be taken into account (Dauer, 1993). This is especially true for estuarine systems, which contain particularly steep physical and contamination gradients, as well as
tions, and that most of these taxa respond similarly to any form of disturbance. However, some evidence indicates that macrobenthic responses do differ between metal contamination and organic enrichment. For example, metal contamination should detrimentally affect all taxa, whereas organic enrichment can actually favor certain opportunistic taxa (Rygg, 1986; Wilson & Jeffrey, 1994). Still, many of the same opportunistic species may prevail under different forms of sediment contamination, since the same small-bodied organisms will initially occupy the very upper surface layer of sediment regardless of the type of disturbance (Rhoads & Germano, 1986). Furthermore, surface layers often contain relatively lower concentrations of contaminants and are more oxidized than pore waters of deeper sediments (Huggett et al., 1992). Moreover, the rapid life cycles of small-bodied taxa can be completed before they are detrimentally affected by contaminants (Diaz & Rosenberg, 1996). Diaz and Rosenberg (1996) note further that as a consequence of the contamination-induced shift to small-bodied opportunistic taxa, benthic production rates, as well as fluctuations, should be amplified.

Key parallels between the two sediment contaminant SAB relationships in this study and the SAB organic enrichment model included notably low SAB metrics at the highest concentrations of both metal and organic-chemical contaminants. Notwithstanding some overlap between the metal and organic-chemical data, the relatively low B/A ratio at the highest concentrations of organic chemicals reflected higher abundances of some small-bodied opportunistic taxa, analogous to the ‘peak of opportunists’, whereas such increases in abundance were not evident for metal contamination. Organic chemicals either possibly conferred additional trophic benefits (Gray, 1989), or were less severe than metal contamination. Macrobenthic organisms may be better able to cope with organic-chemical contamination, as some organic compounds occur naturally (e.g., alkanes). For both metal and organic chemical contamination, there were peaks in the number of presumably tolerant species analogous to the ‘peak of opportunists’. Such changes in species richness along a pollution gradient will be produced from the interaction between both losses and gains in the incidence of taxa (Cao et al., 1997).

Although ‘ecotone points’ were indicated for both classes of sediment contamination, they were signaled by different responses in the SAB metrics, suggesting somewhat different models. According to the SAB model, the ‘ecotone point’ occurs at a juncture where biomass drops while increasing species richness and decreasing total abundance intersect. By contrast, the implied ‘ecotone point’ for metal contamination occurred where both species richness and total abundance dipped, whereas for organic-chemical contamination, it occurred where all three metrics dipped. For both classes of sediment contamination, dips in species richness were immediately followed by analogous ‘peaks of opportunists’ that likely reflected small increases in numbers of tolerant and/or opportunistic taxa. Biomass maxima analogous to the ‘biosimulation point’ in the SAB model occurred at moderate concentrations of contaminants, especially for organic-chemical contamination, implying that certain large-bodied organisms tolerated moderate concentrations of contaminants. However, rapid decreases in the B/A ratio with increasing contamination support the idea that relatively long-lived, large-bodied, equilibrium taxa decrease at high levels of toxicants (Dauer, 1993; Dauer & Alden, 1995).

In previous studies utilizing the same comprehensive dataset, we detailed changes in macrobenthic community structure as well as trophic structure with respect to sediment contamination (Rakocinski et al., 1997; Gaston et al., 1998). Many opportunistic and/or tolerant estuarine taxa that were associated with moderate or high contaminant levels included various polychaetes, such as Capitella capitata, Cossura delta, Cossura soyeri, Mediomastus californiensis, Pseudoeurythoe paucibranchiata, Sigambra tentaculata, and Streblospio benedicti; tubificid oligochaetes, such as Limnodrilus cervix, and Tubificoides heterochaetus; hydrobiid gastropods, such as Probythinella louisiana and Texadina sphinctostoma; bivalves, such as Corbicula fluminea and Rangia cuneata; chironomid larvae, such as Coelotanypus sp. and Cryptochironomus sp.; and the opportunistic amphipod, Corophium cf. lacustrae. Many of the same taxa occurred among sites contaminated with both metals and organic chemicals, however two opportunistic low-salinity taxa, Limnodrilus cervix and Corbicula fluminea characterized the worst contamination by organic chemicals, and two opportunistic polychaetes, Mediomastus californiensis and Cossura delta, characterized high metal contamination in marine sediments (Rakocinski et al., 1997). Thus, the abrupt increase observed in the total abundance of organisms at the highest concentrations of organic chemicals was likely due to the tubificid oligochaete, Limnodrilus cervix. This supports the hypothesis that dominant taxa occurring under the most severe stress
are opportunists (Gray, 1989), rather than tolerant taxa.

Although most taxa associated with sediment contamination were small-bodied opportunists, two relatively large-bodied taxa that tolerated moderate sediment contamination included the burrowing ghost shrimp, *Lepidophthalmus louisianensis*, and the estuarine bivalve, *Rangia cuneata*. Apparently, shifts in abundance/biomass distributions in disturbed marine benthos typically result from underlying taxonomic changes involving the general loss of crustaceans, molluscs, and echinoderms which tend to be larger bodied, in concert with a shift to smaller bodied species within the Polychaeta (Warwick & Clarke, 1994). However, the underlying taxonomic effect may sometimes confound attempts to find a consistent “dose-response” SAB effect. Indeed, expected abundance/biomass patterns in response to heavy metals and PAH’s were not apparent in a study by Dauer et al. (1993), largely due to confounding by the presence of the relatively large-bodied polychaete, *Diopatra cuprea*.

Macrobenthic trophic diversity declined with increasing sediment contamination in our dataset (Rakocinski et al., 1997; Gaston et al., 1998); implying that trophic function was affected along with reductions in faunal complexity with increasing sediment contamination (Gaston & Young, 1992). For example, the proportion of subsurface deposit feeders increased with respect to both metal and chemical contamination gradients, while the proportions of other trophic categories, like carnivores, filter feeders, and surface deposit feeders, decreased. Thus, sediment contamination by either metals or organic chemicals ultimately leads to greatly reduced trophic complexity, as represented by an altered macrobenthic community dominated by opportunistic, small-bodied, surface-dwelling organisms with rapid life-cycles. With increasing sediment contamination, macrobenthic shifts to small-bodied, abundant, opportunistic species, such as tubificid oligochaetes in oligohaline sediments or capitellid polychaetes in mesohaline sediments, translate into shortened food chains and a greater pulsing of energy flow (Diaz & Rosenberg, 1996). Linking related structural and functional responses is critical for a complete assessment of ecological stress (Cairns et al., 1992).

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