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BENTHIC RESPONSE INDEX FOR ASSESSING INFAUNAL COMMUNITIES ON THE SOUTHERN CALIFORNIA MAINLAND SHELF

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Abstract. Although benthic infaunal communities are commonly measured to assess the effectiveness of environmental management in protecting biological resources, the tools used to interpret the resulting data are often subjective or site specific. We present an objective, quantitative index for application throughout the southern California coastal shelf environment that measures the condition of a benthic assemblage, with defined thresholds for levels of environmental disturbance. The index was calculated using a two-step process in which ordination analysis was employed to quantify a pollution gradient within a 717-sample calibration data set. The pollution tolerance of each species was determined based upon its distribution of abundance along the gradient. The index is calculated as the abundance-weighted average pollution tolerance of species in a sample. Thresholds were established for reference condition as well as for four levels of biological response. Reference condition was established as the index value in samples taken distant from areas of anthropogenic activity and for which no contaminants exceeded the effects range low (ERL) screening levels. The four response levels were established as the index values at which key community attributes were lost. Independent data sets were used to validate the index in three ways. First, index sensitivity to a spatial gradient of exposure to a discharge from a point source was tested. Second, index response to a temporal gradient of exposure to a discharge from a point source was examined, testing index robustness to natural temporal variation. Third, the effect of changes in natural habitat (e.g., substrate, depth, and latitude) on index sensitivity was tested by evaluating the ability of the index to segregate samples taken in areas with high and low chemical exposure, across a gradient of physical habitats.

Key words: average pollution tolerance; benthic infaunal communities; benthic response index (BRI); index of biological response; infauna; marine pollution index; measure of environmental disturbance; reference communities; southern California; threshold of biological response.

Introduction

Effective environmental management requires biological indicators to assess the status of and/or trends in resources of interest. Benthic infauna have been used extensively as indicators of environmental status in the marine environment. Repeated studies have demonstrated that benthos respond predictably to various types of natural and anthropogenic stress (Pearson and Rosenberg 1978, Dauer 1993, Tapp et al. 1993, Wilson and Jeffrey 1994, Weisberg et al. 1997). Benthos have many characteristics that make them useful indicators, including their potential for high exposure to stress. Because benthic organisms have limited mobility and cannot avoid adverse conditions, they are exposed to contaminants accumulated in sediments and low concentrations of oxygen in near-bottom waters. As a result, benthic assemblages, unlike most pelagic fauna, reflect local environmental conditions (Gray 1979).

Another advantage of using benthic infauna as biological indicators is their taxonomic diversity. Benthic organisms have a wide range of physiological tolerances, feeding modes, and trophic interactions, making them sensitive to a wide array of environmental stressors (Pearson and Rosenberg 1978, Rhoads et al. 1978, Boesch and Rosenberg 1981). However, this diversity of responses can be difficult to interpret. Environmental managers typically employ a great deal of rigor in quantifying species that are increasing or decreasing over time (or space). A high degree of subjectivity is required, often creating dissension among scientists in their attempts to integrate and assess whether the sum extent of the changes are indicative of an improving or a declining environment (O'Connor and Dewling 1986).

Several efforts have been undertaken to address these issues. The efforts generally fall into three categories. First, single community-attribute measures, including

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species diversity or abundance:biomass ratios, have been used to summarize data beyond the level of individual species (Warwick and Clarke 1993, 1994). While these measures can be useful in some circumstances, Pearson and Rosenberg (1978) have suggested that benthos respond to pollution stress in stages, with different measures necessary to capture the varying responses. Second, the multimetric index approach has been used to combine multiple measures of community response into a single index to more effectively capture the different types of response that occur at different levels of stress (Nelson 1990, Engle et al. 1994, Weisberg et al. 1997).

Third, species composition information has been used directly, usually by describing the assemblage patterns in a comparative multivariate space (Field et al. 1982, Smith et al. 1988). Norris (1995) has suggested that multivariate approaches provide higher sensitivity in assessing perturbation than do methods based upon assemblage metrics. However, the implementation of multivariate approaches and the assessment of their output are often too complex to transmit easily to managers (Gerritsen 1995). Individual-species information has also been used in several indices by assigning pollution tolerance scores to various members of the community and then calculating an average pollution tolerance score of the species found at a site (Hilsenhoff 1977, Word 1978, 1980*a, b,* 1990). This approach is easily communicated to managers, but assignment of pollution tolerance scores has typically been subjective. Here, we develop a new technique for assigning pollution tolerance scores based upon multivariate analysis, with the objective of combining the ease of communication of the tolerance score approach with the analytical rigor of multivariate statistics.

Methods

The benthic response index (BRI) is the abundance-weighted average pollution tolerance of species occurring in a sample and is similar to the weighted average approach used in gradient analysis (Goff and Cottam 1967, Whittaker 1973, Gauch 1982). The index formula is given by the following expression:

$$I_s = \frac{\sum_{i=1}^{n} p_i \sqrt[3]{a_{si}}}{\sum_{i=1}^{n} \sqrt[3]{a_{si}}} \qquad (1)$$

where I_s is the index value for sample s, n is the number of species for sample s, p_i is the position for species i on the pollution gradient (pollution tolerance score), and a_{si} is the abundance of species i in sample s. Species in the sample without p_i values are ignored. In this and subsequent descriptions, "sample" is used equivalently with "sampling unit" and is defined as one grab taken at a station in an individual time period (survey). Eq. 1 is simply the weighted average p_i value for the species in sample s, with the cube root abundances of the individual species as the weights in the weighted average. The cube root of abundance was determined to be the optimal weighting factor, based upon an optimization procedure described in Appendix A.

Determining the pollution tolerance score (p_i) for the species involved four steps: (1) assembling a calibration infaunal data set, (2) conducting an ordination analysis to place each sample in the calibration data set on a pollution gradient, (3) computing the average position of each species along the gradient, and (4) standardizing and scaling the positions to achieve comparability across depth zones. These steps are now discussed in greater detail.

Assembling the calibration data set

The calibration data set included 717 samples selected to provide a range of benthic responses to pollution, across several decades and over a range of depth and sediment habitats. Samples were taken in 10–324 m of water depth in the area between Point Conception and the United States–Mexico international border (Fig. 1). Sediment grain size ranged within 0–99.96% fines. Sampling dates ranged 1973–1994.

Macrobenthic infaunal and sediment chemistry data from six Southern California Bight (SCB) sampling programs were used in the analysis (Table 1). All samples, except those collected in 1973 by the County Sanitation Districts of Los Angeles County, were taken with a 0.1-m^2 modified Van Veen grab (Kahl Scientific Instrument Corporation, El Cajon, California). The 1973 samples were taken with a 0.04-m^2 Shipek grab (Elcee Instrumentation, Selangor, Malaysia). All samples were screened through 1.0-mm sieves, and identified to the lowest possible taxonomic level. To make the data from Shipek grabs comparable, two replicate Shipek grabs were combined, and the abundances were multiplied by 1.25. The macrobenthic infaunal data were used to develop the index, while the sediment chemistry data were used mainly for index validation.

Taxonomic inconsistencies among programs were eliminated by cross-correlating the species lists, identifying differences in nomenclature or taxonomic level, and consulting taxonomists from each program to resolve discrepancies. In some cases, species were assigned to higher categories to maintain comparability with historical data. Data were limited to the summer period during 1 July–30 September. One sample was used for each station/sampling event. If replicate samples were taken at a station, the most "typical" of the replicates was selected. Typical replicates were determined by computing the average dissimilarity value (see *Methods: Ordination analysis*), and contrasting each replicate with the other replicates. The replicate with the lowest average dissimilarity was selected as the typical replicate.

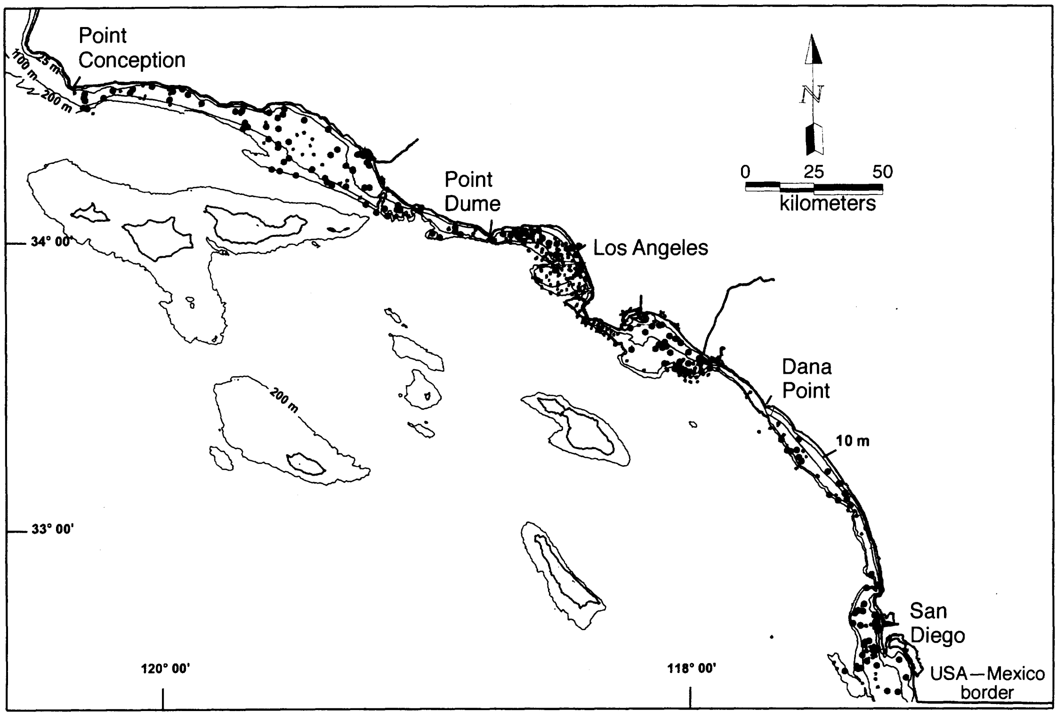

FIG. 1. Location of sites (dots) used in the calibration data set.

Ordination analysis of the macrobenthic infaunal data

Ordination was used to quantify gradients of species change presumably caused by environmental gradients (Pielou 1984). With ordination analysis, samples are displayed as points in a multidimensional space, with the distance between the points proportional to the differences in species composition found in the respective samples. Different environmental gradients causing gradients of species change will often correlate with vectors extending through the space in different directions. To quantify the species gradient corresponding to increasing levels of pollution in the calibration data, we performed an ordination analysis of the calibration data and then defined a vector in the ordination space that separated the known polluted stations from known unpolluted stations. Projections of the sample points onto this vector were used as the position of the sample on the pollution gradient.

Specifically, the pollution gradient within the ordination space was defined as a direction vector connecting the average position of a group of samples representing known polluted stations (polluted endmembers) with a group of samples from known unpolluted stations (unpolluted endmembers), similar to the approach used by R. W. Smith and B. B. Bernstein (*unpublished manuscript*) and Bernstein and Smith (1986). The average positions of the endmembers were computed only from the two-dimensional ordination subspace containing the pollution gradient. A large amount of information useful for defining endmembers was found in the monitoring reports for the larger outfalls in the area, and also in Word and Mearns (1979), Stull et al. (1986*b*), and Stull (1995). The endmembers were chosen to include a wide range of sediment sizes. The average positions of such endmembers in the ordination space should provide the *general direction* of the pollution gradient in the ordination space. Alternatively, we could have used as endmembers the positions of a small number of stations that we thought were the least and most polluted, without regard to sediment size. We rejected this approach, because it would have defined a pollution gradient highly correlated with sediment size, since the most highly polluted stations were in fine sediments on the Palos Verdes Shelf in the early 1970s, and the seemingly "least polluted" stations were found in areas with coarser sediments associated with water currents that would prevent buildup of pollutant materials.

To quantify the position of a sample on the pollution gradient defined in the ordination space, the sample point in the subspace containing the pollution gradient was projected onto the direction vector representing the pollution gradient (using simple geometry with the known positions of the sample points and the angle of the projection line in the space). The projections were rescaled so that the sample closest to the unpolluted end of the gradient was given a gradient score of 0, and the sample closest to the polluted end of the gradient was given a gradient score of 100. This approach assumes that the pollution gradient can be represented by a single direction in the ordination space and that

TABLE 1. Origin of the macrobenthic infaunal and sediment chemistry data used in the calibration data set from six Southern California Bight sampling programs.

Agency	Year	Type of data	Reference
City of Los Angeles	1985	Infauna	
City of Los Angeles	1990	Infauna, sediment metals, grain size, organic carbon	City of Los Angeles (1992)
City of San Diego	1985	Infauna, sediment grain size	City of San Diego (1987)
City of San Diego	1990	Infauna, sediment metals, grain size, organic carbon	City of San Diego (1991)
CSDLAC	1973	Infauna, sediment metals, grain size	CSDLAC (1990)
CSDLAC	1985	Infauna, sediment metals	CSDLAC (1990)
CSDLAC	1990	Infauna, sediment metals, grain size, organic carbon	CSDLAC (1990)
CSDOC	1985	Infauna, sediment metals, grain size, organic carbon	CSDOC (1986)
CSDOC	1990	Infauna, sediment metals, grain size, organic carbon	CSDOC (1991)
Southern California Bight Pilot Project	1994	Infauna, sediment metals, grain size, organic carbon	Bergen et al. (1998)
SCCWRP	1977	Infauna, sediment metals, grain size, organic carbon	Word and Mearns (1979)
SCCWRP	1985	Infauna, sediment metals, grain size, organic carbon	Thompson et al. (1987)
SCCWRP	1990	Infauna, sediment metals, grain size, organic carbon	Thompson et al. (1993)

Note: CSDLAC, County Sanitation Districts of Los Angeles County; CSDOC, County Sanitation Districts of Orange County; SCCWRP, Southern California Coastal Water Research Project.

changes in the ordination space in this direction are linearly related to the amount of pollution present at the respective sample location/times. We do not expect the individual species will always be linearly related to the pollution gradient (Swan 1970), but the ordination methodology we used is designed to represent gradients linearly in the ordination space.

Ordination analysis was conducted separately for three different depth zones, based upon Bergen et al.'s (1998) demonstration that benthic communities within the SCB segregate by depth; separate ordinations were developed for 10–35, 25–130, and 110–324 m. The depth ranges were selected to overlap so that index values could be standardized across depth ranges.

Rare species were eliminated prior to all analyses. For the 10–35 and 110–324 m depth ranges, all species occurring in fewer than three samples were eliminated; for the 25–130 m depth range, all species occurring in fewer than four samples were eliminated. The numbers of species remaining for the shallow, mid-, and deep depth ranges were 379, 477, and 267, respectively. Elimination of the rarest species would not affect the ordination results (Field 1971, Orloci and Mukkattu 1973, and Smith 1976). Also, in the index computations, we only wanted to include species with at least some minimal number of occurrences, in order to avoid misclassification of species due to sampling error.

For this project, the ordination was based upon principal-coordinates analysis (Gower 1966, 1967, Sneath and Sokal 1973, Pielou 1984), in which the ordination space is computed directly from a dissimilarity matrix contrasting all pairs of samples. Dissimilarity was quantified using the Bray-Curtis dissimilarity index (Bray and Curtis 1957, Clifford and Stephenson 1975). Prior to the dissimilarity index computations, data were square-root transformed and standardized by the species mean of values higher than zero (Smith 1976, Smith et al. 1988). Dissimilarity values >0.80 were reestimated using the step-across procedure (Williamson 1978, Bradfield and Kenkel 1987). The step-across procedure corrects for loss in sensitivity of the dissimilarity index as the amount of community change increases. This correction is important when quantifying extended gradients of biological change with ordination (Swan 1970, Austin and Noy-Meir 1971, Beals 1973), since it allows for accurately representing gradients as linear structures in the ordination space. Without this correction, the pollution gradient would be represented as a curvilinear multidimensional structure. Since we are representing the pollution gradient as projections onto a straight line (connecting end-members), distortion would result from projecting to a linear structure from a curvilinear structure.

Position of species on the gradient

The average position of species i (p_i) on the pollution gradient defined in the ordination space was computed as follows:

$$p_i = \frac{\sum_{j=1}^{t_i} g_{ij}}{t_i} \qquad (2)$$

where t_i is the number of samples to be used in the sum, with only the highest t_i species abundance values included in the sum. The g_{ij} is the position of species

i on the pollution gradient in the ordination space for sample j (i.e., g_{ij} is the projection onto the direction vector representing the pollution gradient). Eq. 2 is simply the arithmetic mean of the pollution gradient positions of the stations at which species i occurs, with only the stations corresponding to the t_i highest abundance values of species i used in the average. The value for t_i was determined as part of the optimization procedure described in Appendix A. The numeric value of t_i determined in the optimization varied by the depth zone of the sample. For the 10–35, 25–130, and 110–324 m depth zones, the t_i values are 7, 41, and 48, respectively. The p_i values computed in Eq. 2 are used as pollution tolerance scores in Eq. 1 to compute the index values. The final form of Eq. 2 was determined by the optimization procedure described in Appendix A.

Standardization and scaling of species positions

To enhance the interpretability of our index, we standardized the scales of the index values from the three different depth ranges so that a particular index value indicates the same level of effect, regardless of the depth range. The index standardization was accomplished by regressing shallow and deep depth index values against mid-depth index values for samples falling in the overlapping areas of the depth zones, and then predicting the index values for the shallow or deep depth range using the pertinent regression equation. We further expanded our index scale so that a value of zero corresponds to the lowest original calibration index value found within the mid-depth range, and a value of 100 corresponds to the highest original index value found within the mid-depth range. For future index calculations and for calibration index values from the shallow and deep depth ranges, this scale is open ended. Samples "less polluted" or "more polluted" than all the calibration samples in the mid-depth range can result in index values <0 or >100, respectively.

Threshold development

To place index values in perspective, four thresholds of biological response to pollution were identified. First, we identified the reference threshold, the index value below which natural benthic assemblages normally occur. The reference threshold was defined as a value toward the upper end of the range of index values of samples taken at sites that had minimal known anthropogenic influence. Sites were included if (1) no chemical concentration was higher than the Long et al. (1995) effects range median (ERM) level; (2) no more than one chemical was higher than the Long et al. (1995) effects range low (ERL) level; (3) total organic carbon (TOC) concentration was equal to that expected based upon the regression between sediment grain size and TOC (Bergen et al. 1995); and (4) the sample was collected distant from known contaminant sources (sewage discharges, rivers or storm drains, Santa Monica Bay, and Los Angeles/Long Beach Harbor, or the head of submarine canyons).

The other three thresholds involved defining levels of deviation from the reference condition. These thresholds were based upon a determination of the index values above which species, or groups of species, no longer occurred along the pollution gradient. The first of these response thresholds, which we called loss of biodiversity, was defined as the index value above which 25% of the species pool found in reference samples no longer occurred. The second threshold, which we termed loss in community function, occurred at the point where major taxonomic groups were lost from the assemblage (in our data, the first major taxonomic groups lost were echinoderms and arthropods). The last response threshold, which we referred to as defaunation, was the point at which 90% of the species pool in the reference samples no longer occurred. Index values between reference condition and the loss in biodiversity threshold were identified as marginal deviation, as benthic assemblages in this category primarily reflect a change in relative abundance among species, rather than species replacement.

The 90% upper tolerance interval bound (Hahn and Meeker 1991, Vardeman 1992, Smith, *in press*) for the reference samples was used for the threshold between reference condition and marginal deviation. Specifically, the computed tolerance interval was an upper 95% confidence limit for the 90th percentile of the reference distribution of index values.

Index validation

Three types of validation were performed. The first involved testing whether the index reproduced known spatial gradients of benthic conditions near a southern California ocean outfall. The second involved reproducing known temporal gradients at a set of historically monitored sites. The third involved testing the relationship between chemical exposure and the benthic response index (BRI) at sites throughout the Southern California Bight (SCB). In the first two tests, the validation data sets were independent of the calibration data.

The spatial-gradient test was conducted using data from the Orange County Sanitation Districts (OCSD), which included a gradient of stations on the 60-m isobath, within 0–7840 m from the outfall (County Sanitation Districts of Orange County 1991). Previous studies have shown that two sites located near the outfall (Stations 0 and ZB2) have altered species composition in comparison to three reference stations (13, C, and Con), which are >3800 m from the outfall.

The temporal analysis was conducted using data from two County Sanitation Districts of Los Angeles County (CSDLAC) collection sites, which have been sampled annually since 1972. Stull et al. (1986*b*) and Stull (1995) have shown that the first site, Station 6C (located 2220 m from the outfall) was severely impacted in the early 1970s and has improved since that time. The second site, Station 0C (located 14 720 m from the outfall) was less affected than Station 6C, but has also improved. Our premise in the validation is that index values should decrease over time at Stations 6C and 0C and that index values will be higher and decrease more at Station 6C than at Station 0C.

The relationship between the BRI and chemical exposure was assessed by separating samples into three categories based upon the number of chemicals exceeding Long et al.'s (1995) ERM threshold and examining the degree to which BRI values overlapped among these categories. The analysis was conducted separately for our three depth strata. Our hypotheses were that (1) index values in impact categories will be higher than in reference categories, and (2) index values will be consistent across depths for each impact category.

Results

Fig. 2 shows the ordination spaces and pollution gradient projections for the three depth zones. The arrows in the figure also show the general direction of depth and sediment size (percent fines) gradients in the space. Within each depth zone, the depth gradient was orthogonal to the pollution gradient (Fig. 2); for the mid- and deep depth zones, the sediment grain size gradient was also orthogonal to the pollution gradient. In the shallow habitat, the sediment grain size gradient demonstrated a moderate correlation with the pollution gradient, indicating that "pollution" or organic input is associated with finer sediment input in shallow depths. It should be noted that the three ordination plots shown in Fig. 2 were from separate analyses, and it is not meaningful to compare the directions of the pollution gradients in the different spaces.

Fig. 3 shows the distribution of 10 selected species along the mid-depth pollution gradient. The corresponding unscaled p_i values that summarize the species' positions on the pollution gradient are included.

A high correlation was found between index values in the overlapping sections of the depth zones (Fig. 4). The regression equations shown in Fig. 4 were used to standardize the shallow and deep species p values (and therefore the index values) to a common scale corresponding to the mid-depth scale (p_{di}). These species p_{di} values were then rescaled so that the index values for the 25–130-m depth calibration data ranged 0–100. The

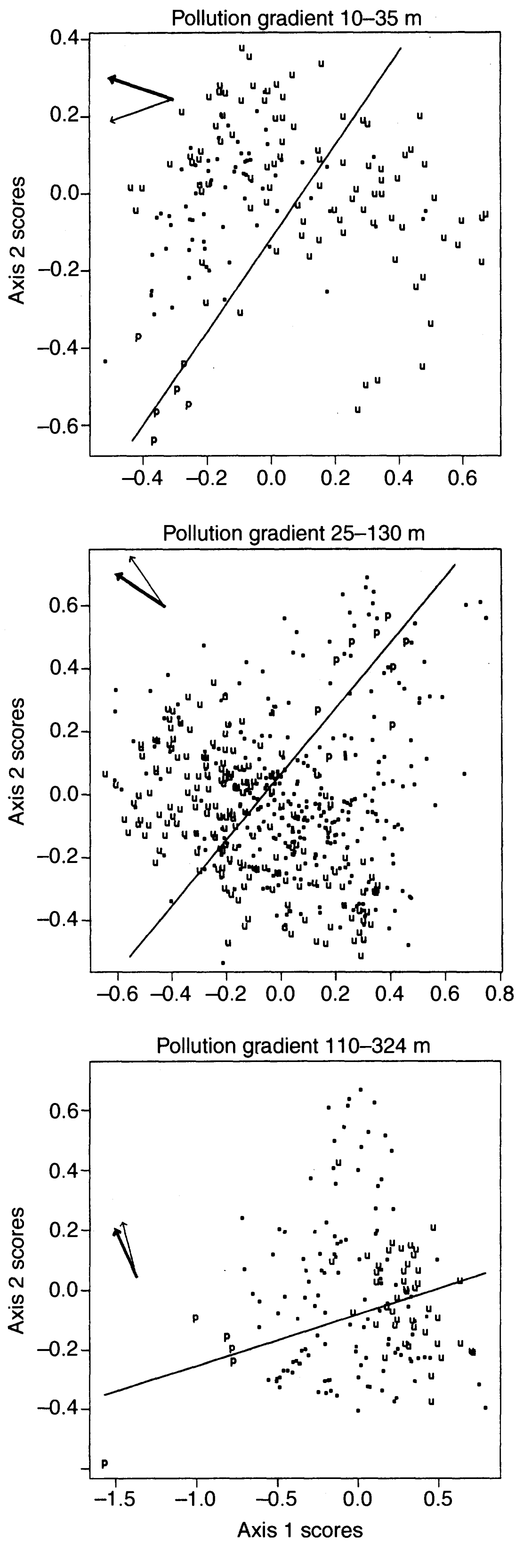

Fig. 2. Plot of ordination results for the three depth zones. The "p" symbols signify polluted endmembers, and the "u" symbols signify unpolluted endmembers. The dots show the positions of the remaining samples. The line in each ordination space connects the average positions of the polluted and unpolluted endmembers. Projections of the points onto the line provide the pollution gradient positions for the samples. The projections are scaled 0–100, with a scaled value of zero for the least polluted sample and a value of 100 for the most polluted sample. The bold arrow shows the direction of increasing depth, and the other arrow shows the direction of increasing percent fines for the samples in the ordination space.

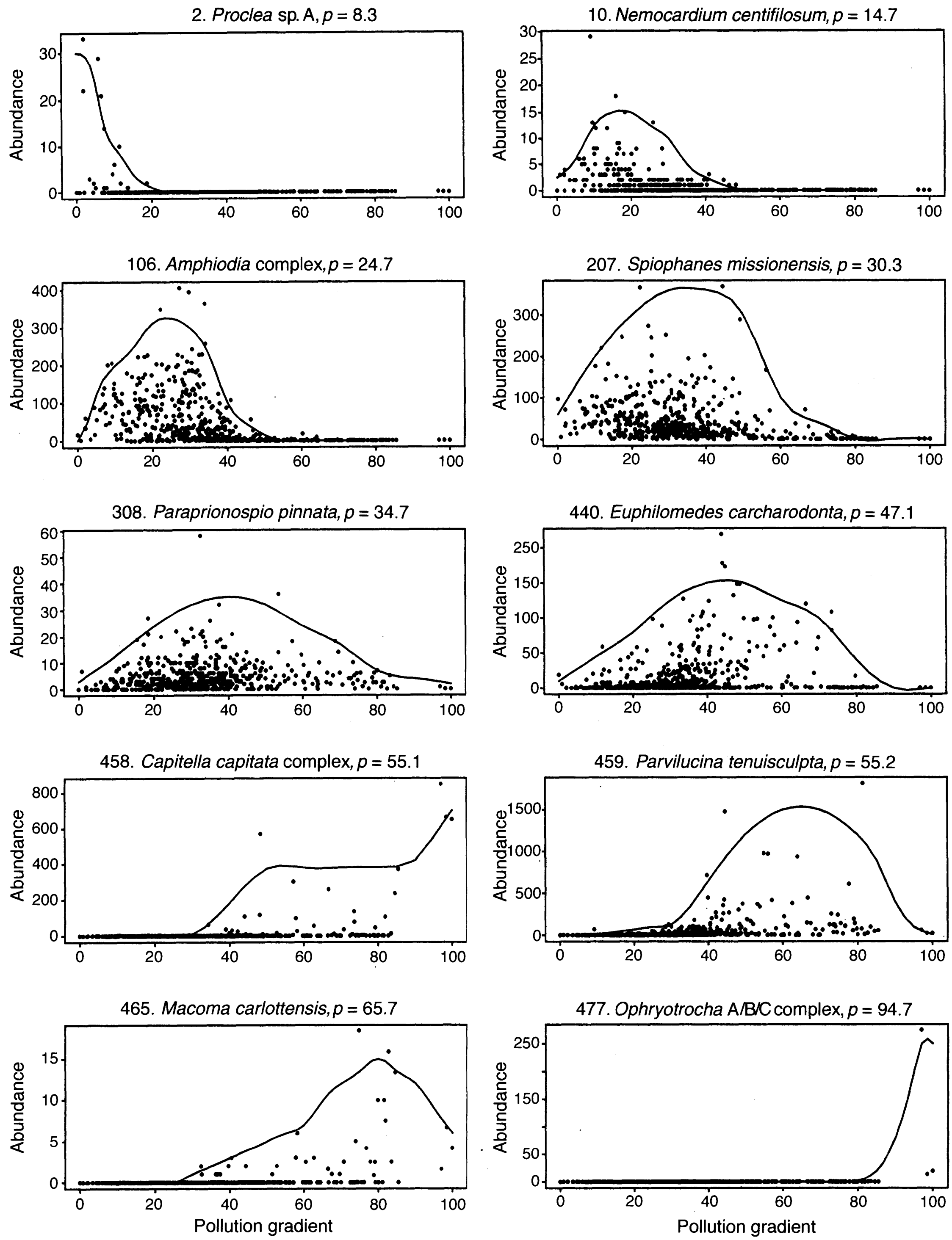

FIG. 3. Distributions of selected species along the mid-depth (25–130 m) pollution gradient. The unscaled p value for each species is indicated. The integers preceding the species names indicate the ascending rank order of species p values along the gradient. There were 477 species used in this depth range. The curves were drawn freehand to summarize the potential abundance of the species along the gradient.

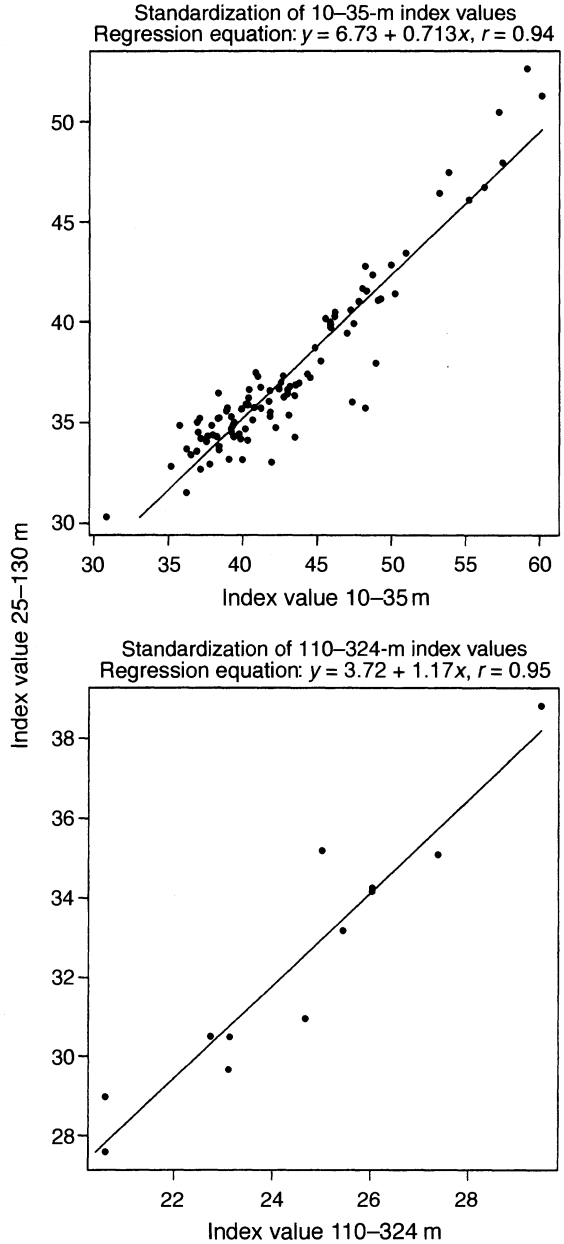

FIG. 4. The index value pairs computed in the overlapping parts of the depth ranges. The regression equations were used to rescale index values from the shallow and deep depth ranges (x in the regression equation) to the scale of the middle depth range (y in regression equation).

final standardized and rescaled p values for all species are provided in Appendix B. Table 2 contains the p values for a selected subset of 32 species.

Threshold development

The index values for samples from uncontaminated sites varied within 0.5–33.2 (Fig. 5). The threshold for reference condition was set at 25, which was the 90% tolerance interval bound for the reference index values. This tolerance interval bound is the upper 95% confidence interval bound of the 90th percentile of the underlying distribution of reference index values (Hahn and Meeker 1991, Vardeman 1992). We chose to use a percentile of the distribution instead of the highest observed value to allow for the possibility that some of the sites in our reference data set were anthropogenically altered by unmeasured pollutants and/or other human activities.

The threshold for loss in biodiversity was set at index value 34, a point where 25% of the species occurring at the reference sites were no longer encountered. The threshold for loss in community function was set at index value 44, the point where 90% and 75% of the species pool of echinoderms and arthropods, respectively, were excluded. The threshold value for defaunation was set at index value 72, the point where 90% of the pool of species occurring at reference sites was excluded.

As an estimate of the uncertainty associated with a specific index value, the one-tailed 95% tolerance interval size for replicates at a particular location and time was computed to be 3.4. This means that 90% of

TABLE 2. Scaled species positions (p_i) along the pollution gradient for a selected subset of species.

Species	Depth zone: Shallow	Middle	Deep
Ampelisca pacifica	50.4	−8.0	−2.4
Amphiodia complex	48.7	−8.6	−12.2
Armandia brevis	129.0	142.0	138.5
Asteropella slatteryi	−6.1	0.3	16.4
Axinopsida serricata	69.7	27.0	60.4
Caecum crebricinctum	2.9	−15.1	−34.6
Capitella capitata complex	67.1	83.8	89.5
Euphilomedes carcharodonta	71.1	59.5	42.6
Euphilomedes producta	···	−9.8	26.8
Leptochelia dubia	11.9	6.6	−22.3
Listriolobus pelodes	83.9	38.6	63.7
Macoma carlottensis	106.0	115.8	76.7
Macoma yoldiformis	19.7	70.0	73.7
Mediomastus spp.	96.3	59.3	20.5
Nemocardium centifilosum	···	−39.0	−5.5
Nephtys caecoides	8.2	32.8	24.6
Nephtys cornuta	65.2	54.4	51.3
Ophryotrocha A/B/C complex	···	204.1	198.8
Paraprionospio pinnata	10.6	21.7	38.6
Parvilucina tenuisculpta	61.3	84.1	76.7
Pectinaria californiensis	40.8	28.1	31.2
Proclea sp. A	···	−58.2	···
Rhepoxynius bicuspidatus	12.5	−15.0	−16.3
Rictaxis punctocaelatus	74.7	76.8	63.6
Solemya reidi	91.3	98.8	133.4
Spiophanes berkeleyorum	24.2	33.8	38.8
Spiophanes bombyx	−2.3	12.1	−23.7
Spiophanes fimbriata	22.4	−17.5	−2.6
Spiophanes missionensis	6.1	8.5	−1.6
Spiophanes wigleyi	···	···	8.5
Thyasira flexuosa	40.0	45.5	42.7
Westwoodilla caecula	40.4	17.6	2.5

Notes: The p_i values are used in Eq. 1 to compute index values. A table of p_i values for all species is included in Appendix B.

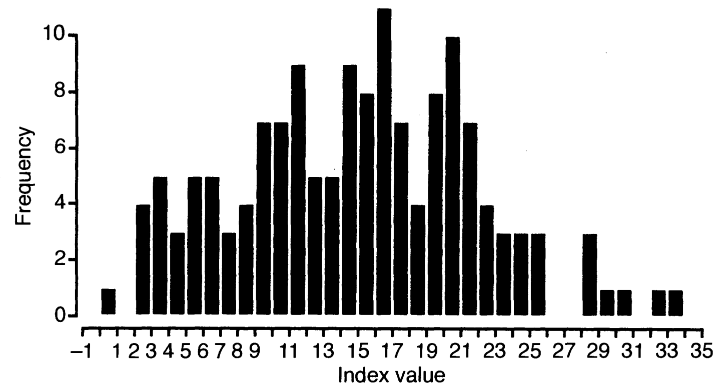

FIG. 5. Histogram showing the distribution of reference index values.

the time, index values for replicate samples for a particular location survey are expected to be within 3.4 units of the mean value for that location survey. For example, if the index value for a specific sample was 39 (second response level), then it is very unlikely that replicates from the same location survey would be found in either of the adjacent response levels (since the adjacent response levels are more distant than 39 ± 3.4).

Index validation

Our index correctly characterized benthic conditions across the spatial gradient near the Orange County Sanitation Districts (OCSD) outfall (Fig. 6). Station 0 (located nearest to the outfall) had index values 26.1–33.4, while Station ZB2, also within the influence of the outfall, had values 28.6–33.9. Index values at the three stations outside of the outfall influence, Stations 13, C, and Con, ranged 14.9–19.3, below the reference threshold. Stations between these spatial extremes had intermediate index values. The interpretation that locations near the outfall caused minor deviation from reference in 1990 would be consistent with the intent of the index, and with our personal experience. It has to be kept in mind that we are comparing with a broad-based reference, and not with a local reference. The community found near the outfall is not much unlike what is found in many unpolluted parts of the Bight. This does not mean that the index values would not be even lower if the outfall was not present, but if one were looking for places with serious pollution problems, this outfall would not stand out (given this interpretation of the benthic response index [BRI]). This is consistent with our experience and knowledge of the situation. There are relatively strong currents in the area that prevent a large local buildup of solids, and the effluent quality had been considerably improved by 1990.

Our index also correctly characterized the temporal gradients near the County Sanitation Districts of Los Angeles County (CSDLAC) outfall (Fig. 7). At Station 6C, where Stull et al. (1986*b*) found dramatic improvements in benthic condition, index values decreased from 120 in 1972 to a mean value of 40–45 in each of the last three years. The decrease in index values in 1975–1976 reflects the reported improvement in benthic communities associated with the invasion of the echiuroid *Listriolobus pelodes* (Stull et al. 1986*a, b*). Similar to Stull (1995), we also found that index values at Station 0C (located at the margins of outfall influence) also improved; however, the change was smaller than at Station 6C.

The first two validation efforts tested the predictive capability of the index when physical habitat, partic-

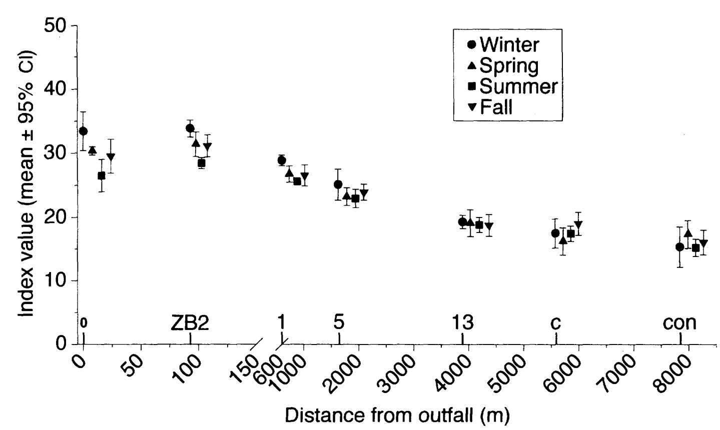

FIG. 6. Benthic response index (BRI) values for a gradient of stations near the Sanitation Districts of Orange County's (California) outfall in 1990.

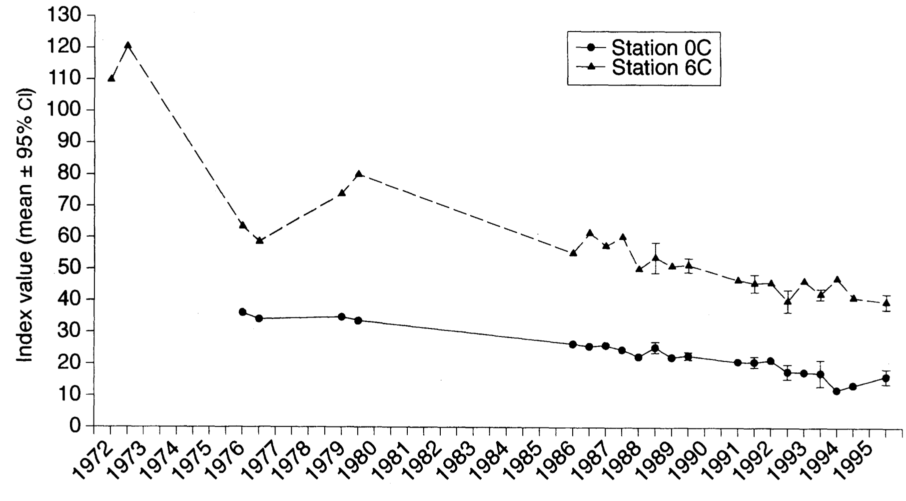

FIG. 7. Benthic Response Index values for stations on the Palos Verdes Shelf during 1972–1995.

ularly depth, was held relatively constant. The third test examined response relative to chemical exposure across a wide array of depth, substrate, and latitudinal gradients. A relatively high differentiation was found between index values for reference sites and samples from sites with known chemical exposure. Samples having at least one chemical exceeding the effects range median (ERM) threshold had index values ranging 19.5–69.6, while every sample from sites with more than one chemical exceeding ERM had an index value >36 (Fig. 8). Within each impact category, index values were consistent across depth.

Index values at chemically unimpaired (reference) sites were found to be consistent across sediment size and latitudinal gradients (Fig. 9b, c). Index values were generally lower in the 70–130-m range than in shallower and deeper water (Fig. 9a). This pattern with depth does not necessarily indicate that the index will confuse depth effects with pollution effects; rather, it is possible that the index is sensitive to the flux or presence of organic matter. The flux of organic matter should be relatively high in shallower areas, as they are closest to onshore sources of organic matter. Deeper sites will usually have finer sediments with associated higher levels of organic matter. Sites distant from outfalls in the mid-depth range would be farther from organic sources than the shallower locations and experience less deposition of finer sediments (and the associated organic matter) than the deeper locations. Thus, lower index values in the middle range might be expected.

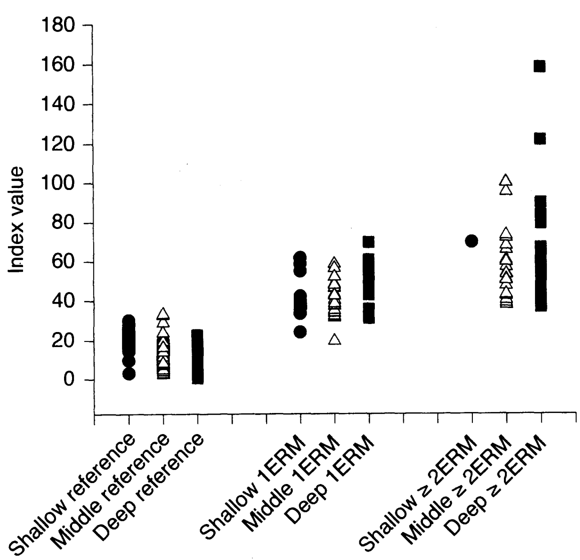

FIG. 8. Benthic response index (BRI) values within shallow, mid-depth, and deep reference sites and at stations with one or more than one chemical above the effects range median (ERM). Key: Shallow, solid circles; middle, open triangles; deep, solid squares.

DISCUSSION

Multivariate ordination analyses have been found to be powerful tools for assessing perturbations to benthic infaunal assemblages (Smith et al. 1988, Norris 1995). The concern with multivariate approaches has been their complexity in application (Gerritsen 1995) and distance from simple biological explanation (Elliott 1994, Fore et al. 1996). Our index resolves many of these challenges by converting the complex multivariate information into an easily interpreted and testable set of individual-species pollution tolerance scores. The pollution tolerance values captured most of the information in the ordination analysis of the calibration data, as a high correlation was found between our index values and the ordination scores depicting the pollution stress gradient (Table A1 in Appendix A). This high correlation means that little information is lost by computing the index value instead of performing an additional ordination analysis. When computing index

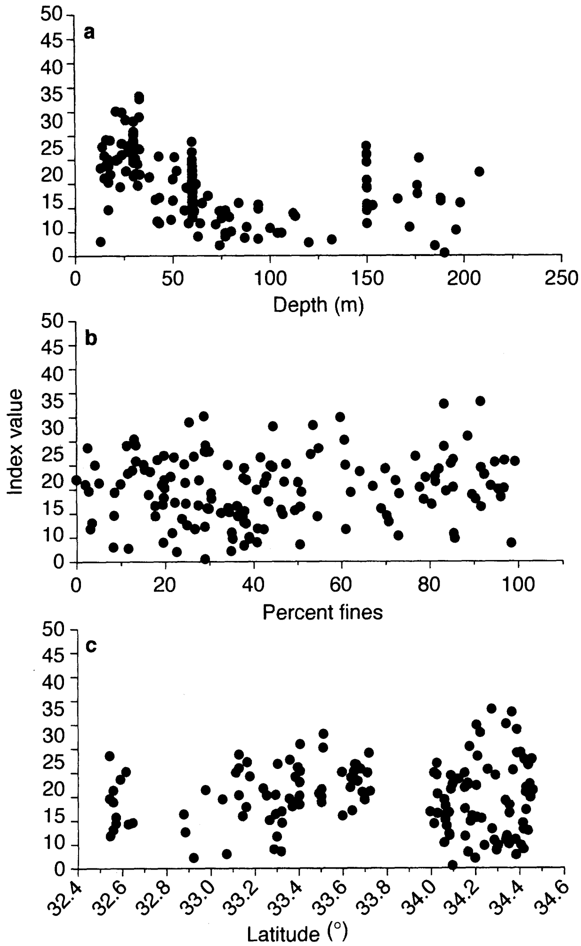

FIG. 9. Benthic response index (BRI) values for reference stations vs. (a) depth, (b) percent fines, and (c) latitude.

values for new data, conducting ordination analyses for each set of data is impractical. Calculating index scores, however, can be done by most biologists.

Benthic assessments have traditionally been conducted by examining changes in community or individual species abundance, an approach that is confounded by natural temporal variability associated with annual and intra-annual recruitment processes. Since our index is based upon the type (pollution tolerance) of species in a sample, it is less sensitive to peaks in abundance of individual species. We observed low seasonal variability in index values, especially at the less stressed stations where the condition of the benthic community should be relatively constant (Fig. 6).

Previous assessments have also focused primarily on characterizing environmental conditions and gradients at local spatial scales, in which depth, latitude, and grain size have been held constant to the degree possible. Benthic assemblages have rarely been used to assess ecological condition across habitats, because the structure of benthic assemblages also reflects natural variation related to salinity, sediment type, latitude, and depth (Boesch 1973, 1977, Dauer et al. 1984, 1987, Holland et al. 1987, Schaffner et al. 1987, Snelgrove and Butman 1994, Heip and Craeymeersch 1995). Furthermore, variation in the condition of the assemblage caused by habitat differences is difficult to separate from variation caused by anthropogenic stresses. This habitat confounding has been minimized in site-specific assessments by limiting comparisons to nearby reference sites from the same type of habitat. Confounding has been avoided in trend studies by continually returning to the same site, which keeps habitat constant.

Our index appears to be robust to this natural habitat variability. In standardizing the benthic response index (BRI) scale across the three depth zones, we found high correlations between independently calculated index values in the overlapping depth zones (Fig. 4). These high correlations indicate a consistency in relative pollution stress levels. We also found that index values at reference stations were not systematically related to grain size or latitude (Fig. 9). A pattern with lower index values at midde depths probably indicates a depth-related pattern of organic matter input rather than a sensitivity to depth per se. We again attribute this robustness to our reliance on the types of species present, not on the abundance of individual species.

Alternative index development methods

Three separate sets of species tolerance scores were developed, corresponding to the three depth zones identified by cluster analysis (Bergen et al. 1998). To assess the need for independent index calibration by depth zone, we attempted to develop a single index from an ordination analysis of all depths combined. We found that a single vector could not characterize the pollution gradient adequately at all depths, and the pollution direction vectors computed separately for the depth zones were not parallel in the ordination space. Presumably, the influence of depth on individual species distributions interacts with the response to stress over such a large depth gradient, reinforcing our decision to conduct separate ordination analyses for the three depth zones.

Most species were found in more than one depth zone. Our inability to identify a unidirectional pollution vector when all depth zones were combined in a single ordination space suggests an inconsistency of pollution response across depth zones for at least some species. Fig. 10 shows the relationship between the species-scaled p_i values for the different depth zones. If the same species indicated the same relative level of stress at all depths, the points for the p_i values would tightly cluster around a straight line and the correlation for the different depths would be high. Although the correlation is moderately high (r = 0.73, 0.78 for shallow and deep zones, respectively), some species differed significantly among the depth zones. Some of this variability can be attributed to measurement error associated with calculating p values for hundreds of species,

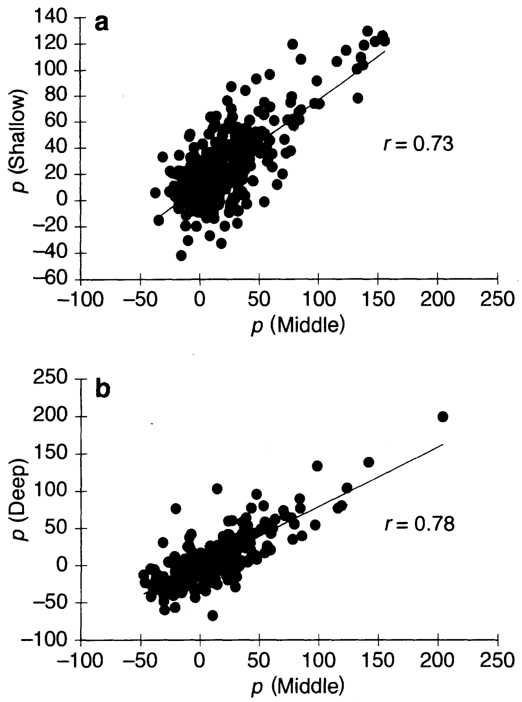

FIG. 10. Relationship between the species p values in the mid-depth zone vs. (a) p values in the shallow zone and (b) p values in the deep zone.

some with low densities in selected habitats. We suggest, however, that some of the differences in pollution tolerance of a species among depth zones may be valid; as a species gets closer to the edge of its distribution range, its tolerance to pollution may decline.

We established the threshold for reference condition at 25, rather than 33, which is the maximum score for reference sites in the calibration and validation data sets. By using a threshold below the maximum score, we allowed for the possibility that some sites in our reference data set may have been impacted by unmeasured pollutants or activities. Similar allowances have been made in the development of other benthic indices (Weisberg et al. 1997). Establishing the threshold at 25 could result in the overestimation of the magnitude of biological response when our index is applied. Philosophically, we believe it is a more conservative approach to classify sites that may exceed reference as falling in a marginal deviation category and to use the index as a screening tool. Users of the index are cautioned that sites with index values within 25–33 represent only minor deviation from reference condition, and confirmatory sampling is recommended before concluding that the site is altered.

General application of the method

The index development methodology could be applied to other geographic areas, habitat types (e.g., terrestrial environments), and gradients (e.g., a sediment size gradient instead of a pollution gradient). Whether an index can be successfully developed will depend on several factors. First of all, sufficient and relevant calibration data must be available. Depending on how the index is to be used, varying amounts of geographic coverage and habitat variation will need to be represented in the calibration data. For example, we developed the BRI index to apply to a large geographic area of offshore benthos, and we required extensive coverage in space and habitat in our data. On the other hand, one could develop an index for a more confined area such as a single harbor. In this case, the same methodology could be used with more limited data.

Second, the data must include appropriate endmembers for defining the gradient of interest in the ordination space. The choice of endmembers should be based on thorough preliminary analyses and understanding of the community patterns displayed in the calibration data. There will be cases where the available data do not contain the gradient of interest, precluding the development of any such index. In other cases, the gradient of interest will be confounded with other gradients, making interpretation of index values difficult. However, as with the choice of BRI endmembers, the use of endmembers spanning different levels of other potentially confounding gradients (e.g., sediment size) can help produce a pollution gradient less confounded with the other gradient.

Third, some attention must be paid to the temporal component of the calibration data. It is best that data toward both extremes of the gradient of interest be available for each sampling period. Otherwise, temporal changes could become confused with gradient changes. For example, if a species invaded the entire geographic area of interest during a specific time period, but samples were obtained for only one extreme of the gradient during that time period, then the presence of the that species would be associated with the gradient extreme that was sampled. On the other hand, if data toward both extremes were available, the species would be found toward both extremes and thus given its proper position on the gradient.

Comparison with other index approaches

The use of abundance-weighted pollution tolerance scores in the BRI is similar to the use of feeding modes as a measure of pollution tolerance in the infaunal trophic index (ITI), an index widely used in southern California (Word 1978, 1980*a*, *b*, 1990). Our application expands upon the ITI in several ways. First, we used an empirical approach to develop pollution tolerance scores for individual species, rather than extrapolating pollution tolerance from feeding mode. Despite differences in methodology, a high correlation was found between the ITI species scores and values we applied to individual species. When differences do occur, they can usually be attributed to a lack of information about the feeding mode of a species, which in some cases led Word (1980*b*) to ascribe all members of a family

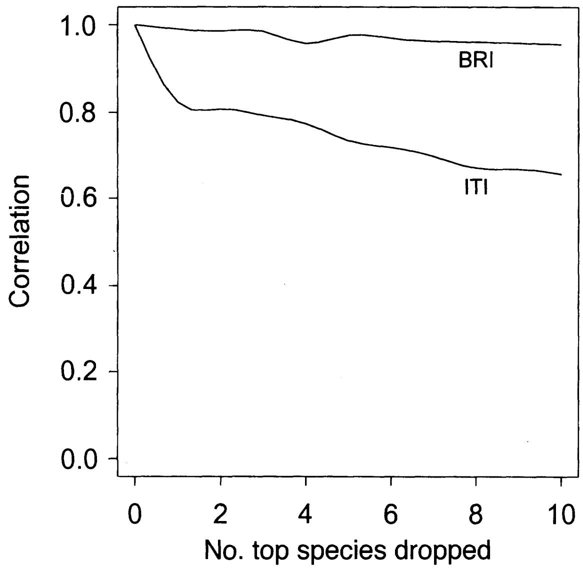

FIG. 11. Effect on the benthic response index (BRI) and infaunal trophic index (ITI) of dropping the most abundant (top) species in each sample. The horizontal axis indicates the number of species dropped, and the vertical axis gives the correlation between the index value with all species and the index value with the species dropped. Indices were computed from the calibration data.

to the same trophic group. We found that *p* values can differ substantially among members of the same family, similar to the findings of Chang et al. (1992).

The second major difference between our method and the ITI is that we developed pollution tolerance values for a larger number of species. In part, the expanded range reflects the larger, more encompassing data sets that are available now compared to the period during which the ITI was developed. Also, incomplete knowledge of trophic categories and inconsistency of trophic modes across different habitats for several species limited the number of species used in the ITI development. Using external (noncalibration) data from outfall monitoring programs, we found that the ITI uses a mean value of ~50% of the species in a sample, compared to 84% for our index. The use of fewer species (along with the use of untransformed abundance weights) makes the ITI subject to greater fluctuation in individual species' abundances. We tested the sensitivity of the BRI and ITI to individual species by systematically removing the most abundant species and correlating the revised index values with the original values (Fig. 11). Even when the 10 most abundant species for each sample were dropped from the computations, the correlation with the original BRI values was still as high as 0.96, confirming the robustness of our index. On the other hand, the correlation for the ITI was ~0.66 when the top 10 species were removed. The correlation for the ITI showed the largest reduction when the single most abundant species was eliminated, indicating that a single abundant species can have a major effect on ITI values.

Weighted averages have also been used in paleolimnological applications to define indices of environmental condition. "Enviornmental condition" here is tied to a specific chemical measurement such as pH (Charles and Smol 1988) or total phosphorus (Hall and Smol 1992). This differs from our approach where the environmental condition is derived from an ordination analysis of the biological data. The paleolimnological indices are also associated with ordination of the biological data, but here the ordination is only used as a preliminary analysis showing a relationship between the biota and the environmental condition of interest. Demonstration of such a relationship is then used as a justification for using the chemical measurement directly in a weighted average.

Our approach to index development differs significantly from approaches used on the east and Gulf coasts of the United States, where multimetric indices are widely used (Engle et al. 1994, Weisberg et al. 1997). The difference in our approach reflects the different levels of stress in the two areas. Pearson and Rosenberg (1978) have suggested that benthos respond sequentially to different levels of stress, with species replacement occurring at the lowest level, and loss in diversity, abundance, and biomass occurring at increasingly higher levels of stress. In Chesapeake Bay and the Gulf of Mexico, where multimetric indices have been developed, hypoxia was prevalent; sites with low diversity and abundance were an integral part of the index calibration and validation data sets. Hypoxia was virtually absent in our study area and the impacts on the benthos were more subtle. Weisberg et al. (1997) noted that the most sensitive metrics in Chesapeake Bay, particularly in lower stress environments, were based upon species replacement.

While the BRI appears to have immediate applicability along the continental shelf of the Southern California Bight (SCB), opportunities exist for further development. We have not yet tested its applicability in harbors or bays, where a higher level of exposure may exist. We have also not attempted to differentiate the effects of natural stress from anthropogenic stress. For example, benthos at sites near rivers experience natural salinity stress during the rainy season and may experience higher sediment organic content from natural runoff sources. Similarly, natural oil seeps in southern California can mimic the effect of anthropogenic pollution. Weisberg et al. (1997) recognized similar difficulties in differentiating the effects of natural and anthropogenically generated hypoxia in Chesapeake Bay. While these natural forms of stress do not invalidate the use of the index, they do lead to caution in interpretation of alterations from background communities and provide a focus for future research efforts to determine the cause of these effects.

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APPENDIX A

A description of the optimization procedure used to derive Eqs. 1 and 2 is available online in ESA's Electronic Data Archive: *Ecological Archives* A011-014-A1.

APPENDIX B

A table of p_i values for all species is available at ESA's Electronic Data Archive: *Ecological Archives* A011-014-A2.