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Benthic Response Index for Assessing Infaunal Communities on the Mainland Shelf of Southern California

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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. Here 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.

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. Benthic scientists typically employ great rigor in quantifying abundance

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changes of species over time or space, but then use subjective approaches to assess whether the sum extent of these changes across species are indicative of an improving or a declining environment (O'Connor and Dewling 1986). This subjective aspect in interpretation often causes dissension among scientists and frustrates environmental managers.

Several efforts have been undertaken to increase objectivity in interpretation of benthic data. The efforts generally fall into three categories. Single community attribute measures, including species diversity and 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 different levels of response. A second approach is the multi-metric index, which combines 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, VanDolah *et al.* in press).

The third approach uses species composition information 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 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 1980a, 1980b, 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:

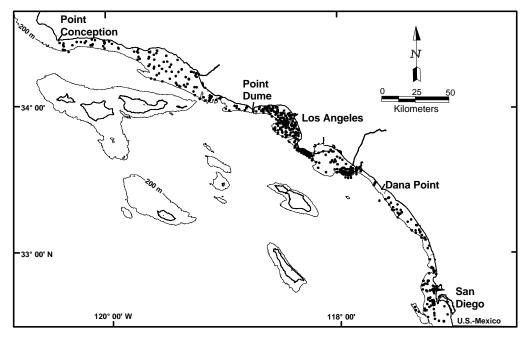$$I_s = \frac{\sum_{i=1}^{n} p_i \sqrt[3]{a_{si}}}{\sum_{i=1}^{n} \sqrt[3]{a_{si}}} \qquad (1)$$

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). Equation 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 discussed in more detail below.

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 to 324 m of water depth in the area between Point Conception and the United States-Mexico international border (Figure 1). Sediment grain size ranged from 0 to 99.96% fines. Sampling dates ranged from 1973 to 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. The 1973 samples were taken with a 0.04 m^2 Shipek grab. 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

FIGURE 1. Location of sites used in the calibration data set.

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 into 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 the 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 Smith and Bernstein (1985) and Bernstein and Smith (1986). The average position of the endmembers in the ordination space provided the general direction of the pollution gradient in the ordination space. Endmembers were defined using information from the monitoring reports for the larger outfalls in the area, and also from Word and Mearns (1979), Stull *et al.* (1986b), and Stull (1995). The endmembers were chosen to include a wide range of sediment sizes. Alternately, 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 1970's , and the seemingly "least polluted" stations were found in areas with coarser sediments associated with water currents that would prevent buildup of pollutant materials

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 from July 1 to September 30. 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 the Ordination Analysis subsection below), and contrasting each replicate with the other replicates. The replicate with the lowest average dissimilarity was selected as the typical replicate.

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

TABLE 1. Origin of data used in the calibraion data set.

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	1985	Infauna, sediment metals	
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)

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 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 (see below) 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 m, 25-130 m, 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 analyses. For the 10-35 m 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. Inclusion of the rarest species does not materially affect the ordination results (Field 1971, Orloci and Mukkattu 1973, and Smith 1976), but they were removed because we wanted to avoid misclassification of species with a small sample size.

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 higher than 0.80 were re-estimated 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 represented the pollution gradient as a projection onto a straight line (connecting endmembers), 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:

$$p_i = \frac{\sum\limits_{j=1}^{t}}{t}, \qquad (2)$$

where t is the number of samples to be used in the sum, with only the highest t species abundance values included in the sum. The g_j is the position on the pollution gradient in the ordination space for sample j (i.e. g_i is the projection onto the direction vector representing the pollution gradient.) Equation (2) is the arithmetic average of the pollution gradient positions of the stations at which species i occurs, with only the stations corresponding to the t highest abundance values of species i used in the average. The value for t was determined as part of the optimization procedure described in Appendix A. The numeric value of t determined in the optimization varied by the depth zone of the sample. For the 10-35 m, 25-130 m, and 110-324 m depth zones, the t values are 7, 41, and 48, respectively. The p_i values computed in Equation (2) are used as pollution tolerance scores in Equation (1) to compute the index values. The final form of Equation (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 0 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 less than 0 or greater than 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, 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) 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 BRI at sites throughout the 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 District (OCSD), which included a gradient of stations on the 60 m isobath, from 0 to 7,840 m from the outfall (CSDOC 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 over 3,800 m from the outfall.

The temporal analysis was conducted using data from two Los Angeles County Sanitation Districts (LACSD) collection sites, which have been sampled annually since 1972. Stull *et al.* (1986b) and Stull (1995) have shown that the first site, Station 6C (located 2,220 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

Figure 2 shows the ordination spaces and pollution gradient projections for the three depth zones, with the arrows showing the 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 (Figure 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 3 ordination plots shown in Figure 2 were from separate analyses, and it is not meaningful to compare the directions of the pollution gradients in the different spaces.

Figure 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 (Figure 4). The regression equations shown in Figure 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 re-scaled so that the index values for the 25-130 m depth calibration data ranged from 0 to 100. The final standardized and rescaled p values for all species are provided in Table 2.
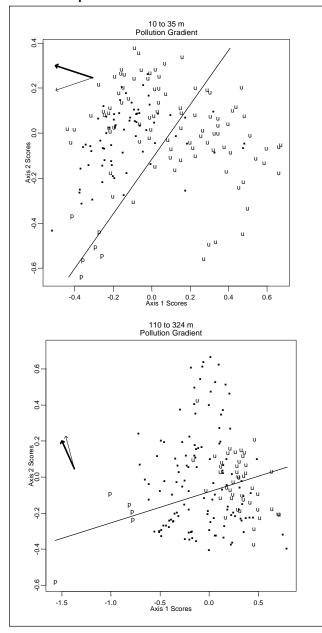
Threshold Development

The index values for samples from uncontaminated sites varied between 0.5 and 33.2 (Figure 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 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 allowed 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, the 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.

The one-tailed 95% tolerance interval size for replicates at a particular location and time was computed to be 3.4, which provides an estimate of the uncertainty associated with a specific index value. This means that 90% of the time, index values for replicate samples for a particular location-survey will be within 3.4 units of the mean value for that location-survey. For example, if the
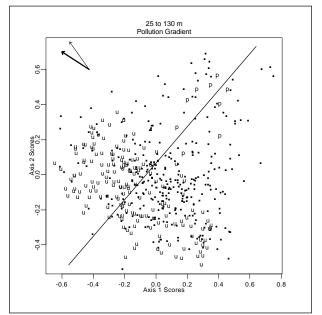
FIGURE 2. Plot of ordination results for the three depth zones. The "p" symbols signify polluted endmembers and the "u" symbols signify unpolluted endmembers. The dots show the positions of the remaining samples. The line in each ordination space connects the average positions of the polluted and unpolluted endmembers. Projections of the points onto the line provide the pollution gradient positions for the samples. The projections are scaled from 0 to 100, with a scaled value of 0 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 regular arrow shows the direction of increasing percent fines for the samples in the ordination space.

values from 26.1 to 33.4, while Station ZB2, also within the influence of the outfall, had values from 28.6 to 33.9. Index values at the three stations outside of the outfall influence, Stations 13, C, and Con, ranged from 14.9 to 19.3, below the reference threshold. Stations between these spatial extremes had intermediate index values. This is consistent with previous characterizations of the area based on conventional comparison of infaunal species composition. There are relatively strong currents in the area that prevent a large local build-up of solids. While alterations to the benthic community have been observed, they have been primarily limited to minor changes in relative composition, rather than large losses in diversity or abundance (CSDOC 1991).

The index also correctly characterized the temporal gradients near the LACSD outfall (Figure 7). At Station 6C, where Stull *et al.* (1986b) found dramatic improvements in benthic condition, index values decreased from 120 in 1972 to an average of 40-45 in each of the last three years. The decrease in index values in 1975-76 reflects the reported improvement in benthic communities associated with the invasion of the echiuroid *Listriolobus pelodes* (Stull *et al.* 1986a,b). Similar to Stull *et al.* (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.

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.

Index Validation

The index correctly characterized benthic condition across the spatial gradient near the OCSD outfall (Figure 6). Station 0 (located nearest to the outfall) had index

Figure 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 are the ascending rank order of species *p* values along the gradient. There were 477 species used in this depth range.
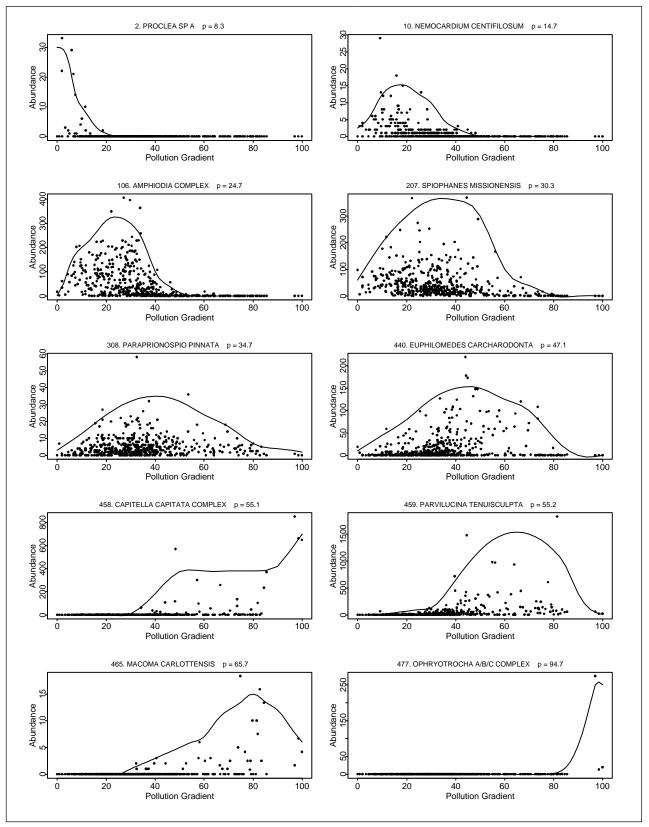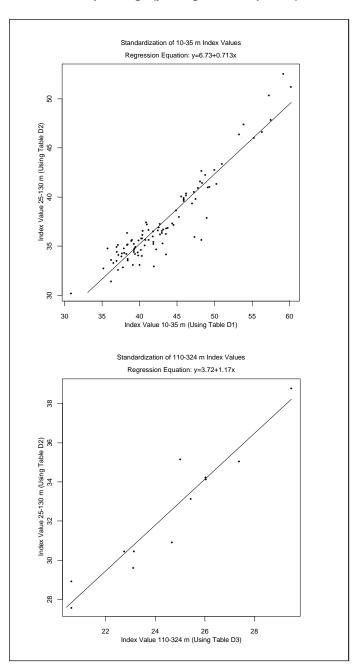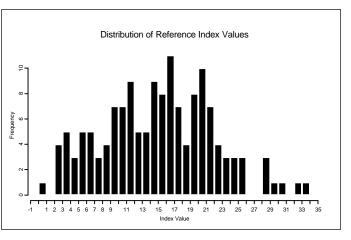
FIGURE 4. The index value pairs computed in the overlapping parts of the depth ranges. The regression equations were used to rescale index values from the shallow and deep depth ranges (x in the regression equation) to the scale of the middle depth range (y in regression equation).

Standardization of 10-35 m Index Values

Regression Equation: y=6.73+0.713x

Standardization of 110-324 m Index Values

Regression Equation: y=3.72+1.17x

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

Distribution of Reference Index Values

The first two validation efforts tested the predictive capability of the index when physical habitat, particularly 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 ERM threshold had index values ranging from 19.5-69.6, while every sample from sites with more than one chemical exceeding ERM had an index value exceeding 36 (Figure 8). Within each impact category, index values were consistent across depth.

Index values at chemically unimpaired (reference) sites were consistent across sediment size and latitudinal gradients (Figure 9b,c). While there was no linear relationship between reference site index values and depth, index values were generally highest in shallow water and lowest in the 70-130 m range. This may reflect the naturally low levels of organic matter in this mid-depth range, rather than a direct effect of depth. The flux of organic matter is relatively high in shallower areas closest to the onshore sources, while sites deeper than 130 m are typically depositional and accumulate organic matter.

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

TABLE 2. *p* values used to calculate benthic response index.

Species	*p* Shallow	*p* Middle	*p* Deep
Amphiura acrystata	22.1	-24.9	-28.9
Amygdalum pallidulum		-23.4	-20.1
Anchicolurus occidentalis	-39.1		
Ancistrosyllis sp.	40.1	56.3	25.8
Anobothrus gracilis		-11.2	8.3
Anonyx lilljeborgi		-24.4	
Anotomastus gordiodes	-11.0	11.2	
Aoroides sp.	0.8	14.2	-26.0
Aphelochaeta/monticellina complex	68.8	85.6	39.3
Aphrodita sp	11.4	3.3	
Apistobranchus ornatus	6.1	-21.0	
Aplacophora	37.9	19.2	9.9
Apoprionospio pygmaea	17.2	26.2	
Arabella sp.	48.9	36.4	
Araphura sp. A	13.8	-0.4	6.7
Araphura sp. B		-3.3	3.9
Argissa hamatipes	22.8	26.2	
Arhynchite californicus			59.5
Aricidea wassi	13.0	24.5	
Armandia brevis	129.0	142.0	138.5
Armina californica	44.0	32.5	
Artacamella hancocki	3.1	-9.9	-26.7
Aruga holmesi		-18.8	-22.4
Aruga oculata	26.5	10.5	
Asabellides lineata	19.8	-18.6	-6.9
Asteropella slatteryi	-6.1	0.3	16.4
Astropecten verrilli	-6.0	6.5	-14.1
Autolytus sp.		3.8	
Axinopsida serricata	69.7	27.0	60.4
Bathyleberis sp.	46.0	30.6	-15.4
Bathymedon pumilus		-15.1	-14.9
Bathymedon vulpeculus			2.8
Bemlos audbettius	-3.4	-4.5	-43.2
Bittium complex	-3.6	15.5	18.1
Blepharipoda occidentalis	-21.3		
Boccardia basilaria	50.7	39.3	
Boccardiella hamata			89.0
Brada pluribranchiata		-42.2	
Brada villosa	29.3	-5.3	
Branchiostoma californiense	-19.5		
Brisaster latifrons			3.0
Brissopsis pacifica			-8.8
Byblis veleronis	33.4	-4.7	-34.5
Caecum crebricinctum	2.9	-15.1	-34.6
Calinaticina oldroydii	36.2	23.9	
Calyptraea fastigiata	26.8	19.6	
Campylaspis canaliculata	-2.0	-6.3	
Campylaspis hartae		-16.9	
Campylaspis rubromaculata	4.2	-2.5	-13.5
Campylaspis sp. D	24.6		
Cancer gracilis	3.3	38.1	

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 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 (Figure 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); variations in 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 BRI scale across the three depth zones, we found high correlations between indepen-

Table 2 continued.

Species	p Shallow	p Middle	p Deep
Cancer jordani	8.3	10.8	
Capitella capitata complex	67.1	83.8	89.5
Carazziella sp.	54.9	29.1	
Cardiomya sp.		-0.7	9.6
Caudina arenicola		18.2	
Caulleriella alata	103.3	137.8	
Caulleriella gracilis		-8.6	0.1
Cephalophoxoides homilis		-29.7	-34.2
Cerapus tubularis complex	-13.8	12.2	
Cerantharia	36.2	12.5	-18.4
Cerithiopsis sp.		13.6	
Chaetopterus variopedatus	28.6		
Chaetozone armata	31.1	28.7	
Chaetozone corona	36.0	51.1	
Chaetozone setosa complex	25.1	28.7	2.8
Chione sp.	77.8	133.3	
Chloeia pinnata	37.6	19.0	26.9
Chone complex	4.6	14.8	26.7
Cirratulus sp.	41.3	26.9	
Cirriformia sp.	4.3	36.4	
Cirrophorus branchiatus			-2.8
Cirrophorus furcatus	15.5	20.4	
Clymenella complanata	16.9	-9.6	
Clymenura gracilis		-36.0	-31.4
Compsomyax subdiaphana	51.8	41.2	
Conus californicus	100.2	132.7	
Cooperella subdiaphana	47.2	42.8	
Corbula sp.		34.2	
Corophium sp.	25.7	34.0	
Corymorpha sp.	6.6	-25.3	
Cossura spp.	60.9	42.2	20.9
Crangon alaskensis	-0.3		
Crenella decussata	35.9	8.9	
Crepidula sp.	31.3	30.7	
Cryptomya californica	83.4		
Cumella sp. A		-5.0	
Cuspidaria parapodema		18.1	20.7
Cyclaspis nubila	-5.6		
Cyclocardia spp.		-31.5	30.4
Cylichna diegensis	26.5	37.1	52.2
Decamastus gracilis	74.8	54.5	58.2
Deilocerus planus	29.6	0.8	
Delectopecten vancouverensis		-42.0	-4.6
Dendraster excentricus	-10.2	10.5	
Dentalium sp.	-3.0	8.4	5.7
Diastylis californica	13.3	16.9	
Diastylis paraspinulosa			-0.7
Diastylis pellucida			39.5
Diastylis sp. A		0.5	8.1
Diastylopsis tenuis	-33.2	18.2	
Diopatra ornata	20.2	36.0	25.3
Diopatra splendidissima	13.0	9.2	
Diopatra tridentata	8.5	15.5	

dently calculated index values in the overlapping depth zones (Figure 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 (Figure 9). We attribute this robustness to our reliance on the types of species present, not on the abundance of individual species.

Alternate 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 distribution 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. Figure 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), 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, some with

Table 2 continued.

Species	p Shallow	p Middle	p Deep
Dougaloplus sp.		-47.7	-12.5
Drilonereis sp.	13.3	19.9	0.3
Eclysippe trilobata		-30.7	-41.3
Edotia sp.	-9.8	8.1	
Edwardsiidae	15.3	34.7	5.8
Ennucula tenuis	15.6	-0.4	-4.5
Ensis myrae	-1.5	26.5	
Enteropneusta	55.1	11.4	2.5
Entodesma pictum		2.2	
Ephesiella brevicapitis	26.0	-7.5	
Epitoniidae	1.1	29.5	
Eranno lagunae	32.4	-7.7	41.8
Ericthonius brasiliensis	36.2	8.9	
Ericthonius rubricornis			-4.4
Erileptus spinosus		-11.7	
Eteone sp.	87.0	26.9	
Euchone sp.	-11.4	-11.5	2.2
Eudorella pacifica		-9.7	-8.3
Eudorellopsis longirostris		-29.0	-23.1
Eulalia sp.	13.9	7.1	9.2
Eulima californicus	1.7	13.0	
Sige sp. A	19.7	23.4	18.9
Eumida longicornuta	35.1	61.0	49.5
Eunice americana	20.6	8.1	25.9
Malmgreniella baschi	24.3	-2.6	9.5
Euphilomedes carcharodonta	71.1	59.5	42.6
Euphilomedes producta		-9.8	26.8
Euphysa sp. A	14.3	9.0	
Eupolymnia heterobranchia	45.7	49.1	
Eurydice caudata	26.6	28.4	
Eusarsiella thominx	34.0	9.7	
Eusyllis transecta	-8.4	-7.6	
Exogone lourei	13.5	10.5	-2.4
Exogone molesta		-10.3	
Exogone dwisula	-8.5	11.2	
Exogone breviseta	13.1	0.1	
Eyakia robusta		-40.4	-28.6
Fauveliopsis sp.			-11.5
Foxiphalus cognatus	16.0	36.0	
Foxiphalus golfensis	12.9	19.7	
Foxiphalus obtusidens	15.8	24.5	23.2
Foxiphalus similis		-22.4	-36.4
Galeommatidae sp. A			-5.7
Gammaropsis ociosa		-30.9	-48.3
Gammaropsis thompsoni	16.3	18.1	
Gari californica	22.6		
Gastropteron pacificum	15.5	12.0	11.1
Gibberosus myersi	-18.0		
Gitana calitemplado	-4.2	-8.9	
Glottidia albida	7.4	10.9	
Glycera americana	56.9	79.3	55.2
Glycera convoluta	-10.0		

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 gradient, 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 between 25 and 33 represent only minor deviation from reference condition, and confirmatory sampling is recommended before concluding that the site is altered.

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, 1980a, 1980b, 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

Table 2 continued.

Species	p Shallow	p Middle	p Deep
Glycera nana	50.6	39.8	53.6
Glycera oxycephala	-4.1	10.4	-67.8
Glycinde armigera	19.1	19.1	38.5
Glyphocuma sp. A		-64.4	
Gnathia crenulatifrons	18.4	22.6	9.4
Goniada brunnea	-0.8	16.6	22.5
Goniada littorea	22.2	12.2	
Goniada maculata	49.3	19.3	30.5
Gymnonereis crosslandi	64.1	14.1	14.8
Halicoides synopiae		-32.5	-15.4
Haliophasma geminatum	26.4	1.9	-9.2
Halistylus pupoides	-42.0	-15.5	
Halosydna brevisetosa	28.4	29.2	
Hamatoscalpellum californicum	42.7	2.7	-21.4
Malmgreniella scriptoria		-0.7	-0.1
Harpiniopsis fulgens			2.2
Hemilamprops californica	-19.9	21.2	
Hemiproto sp. A		-30.0	-60.3
Heptacarpus stimpsoni	28.5		
Hesperonoe laevis	54.8	39.0	42.6
Heterocrypta occidentalis	0.9	10.4	
Heteromastus filobranchus		118.9	80.2
Heterophoxus sp.	38.9	16.4	-0.7
Hiatella arctica	58.8	12.7	
Hippomedon sp.	-20.0	-2.9	-8.7
Hornellia occidentalis	-17.4		
Huxleyia munita			-66.3
Hyale sp.	109.0	135.9	
Hyalinoecia juvenalis	-0.9	-9.1	
Idarcturus allelomorphus		-6.7	
Ilyarachna acarina			-25.5
Joeropsis dubia	37.1	29.7	
Kurtzia arteaga	26.3	14.3	3.4
Kurtziella beta	23.8	38.5	25.6
Kurtziella plumbea	-1.5	54.4	
Lamprops carinata		-17.8	
Lamprops quadriplicata	-35.8		
Lanassa spp.	5.8	-37.4	-19.1
Lanice conchilega	37.0	33.3	0.2
Laonice appelloefi		5.5	8.4
Laonice cirrata	25.0	38.6	23.2
Leitoscoloplos pugettensis	42.5	47.5	8.0
Lepidasthenia berkeleyae		35.5	42.9
Lepidepecreum sp. A	17.9	7.9	-23.3
Leptochelia dubia	11.9	6.6	-22.3
Leptochiton sp.		-37.0	
Leptopecten latiauratus	12.5	21.6	
Leptostylis sp. A	50.5	8.1	
Leptostylis villosa		-21.2	
Leucon subnasica		11.7	
Leuroleberis sharpei	-2.0	11.2	
Levinsenia sp.	59.1	25.0	12.6

cases led Word (1980b) to ascribe all members of a family 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 (non-calibration) data from outfall monitoring programs, we found that the ITI uses an average of approximately 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 (Figure 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 approximately 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.

Our approach to index development differs significantly from approaches used on the east and Gulf coasts of the United States, where multi-metric 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, abun-

Table 2 continued.

Species	*p* Shallow	*p* Middle	*p* Deep
Listriella diffusa	-17.8	31.8	
Listriella eriopisa	35.0	30.0	
Listriella goleta	60.7	63.1	61.7
Listriella melanica	26.1	34.3	
Listriolobus pelodes	83.9	38.6	63.7
Loimia medusa	43.1	19.9	
Lovenia cordiformis	-27.1	8.5	
Lucinisca nuttalli	107.8	85.7	
Lucinoma annulatum	39.4	53.3	80.1
Luidia sp.		-18.9	
Lumbrinerides platypygos	-3.1	-17.6	
Lumbrineris spp.	49.4	30.2	19.1
Lyonsia californica	21.1	0.1	
Lysippe sp.	54.0	20.1	4.1
Lytechinus pictus	8.0	-9.2	-16.4
Macoma carlottensis	106.0	115.8	76.7
Macoma nasuta	125.3	154.6	
Macoma yoldiformis	19.7	70.0	73.7
Mactridae	18.0		
Maera simile	20.7	-22.7	-40.3
Magelona pitelkai		52.9	
Magelona sacculata	-8.3	32.3	
Magelona spp.	26.8	29.5	-29.1
Malacoceros punctata		-34.4	
Maldane sarsi	18.3	9.3	17.3
Marphysa sp.	34.6	23.2	41.1
Mayerella banksia	4.7	-15.9	-37.0
Mediomastus spp.	96.3	59.3	20.5
Megalomma pigmentum	8.7	21.8	
Megasurcula carpenteriana		7.2	
Melanella sp.	4.1	16.5	-6.5
Melanochlamys diomedea	11.7	65.2	
Melinna heterodonta	37.4	20.8	24.7
Melinna oculata	12.9	31.6	-2.0
Melphisana bola complex	5.3	6.0	
Mesochaetopterus sp.		2.5	
Mesocrangon munitella	4.7		
Mesolamprops bispinosa		-0.7	
Metaphoxus frequens	20.1	12.3	5.0
Metasychis disparidentatus	9.4	11.8	14.0
Metopa dawsoni		-21.6	
Micropodarke dubia	50.5	55.0	
Microspio pigmentata		0.0	2.4
Modiolus sp.	48.5	43.2	
Molpadia intermedia		-13.4	8.2
Monoculodes sp.	1.3	-0.9	12.3
Mooreonuphis nebulosa	-2.8	4.5	-34.4
Mooreonuphis spp.		-35.4	-32.4
Mooresamytha bioculata	13.0	19.8	20.8
Munnogonium tillerae		-5.7	
Myriochele spp.	-2.1	-3.2	-2.6
Myriowenia californiensis		-15.8	

dance, and biomass occurring at increasingly higher levels of stress. In Chesapeake Bay and the Gulf of Mexico, where multi-metric 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 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.

Table 2 continued.

Species	p Shallow	p Middle	p Deep
Mysella sp.	45.6	59.2	51.6
Myxicola infundibulum		0.1	
Nassarius fossatus	25.1		
Nassarius insculptus			105.0
Nassarius mendicus	118.3	138.8	
Nassarius perpinguis	26.1	42.9	77.2
Neastacilla californica	20.6	10.5	
Nebalia sp.	-3.0	26.0	
Nemocardium centifilosum		-39.0	-5.5
Neocrangon zacae		-14.6	
Neomysis kadiakensis	-2.2	11.1	
Neotrypaea sp.	45.0	38.9	4.9
Nephtys caecoides	8.2	32.8	24.6
Nephtys cornuta	65.2	54.4	51.3
Nephtys ferruginea	76.0	23.5	15.8
Nereiphylla castanea	22.2	12.7	23.7
Nereis latescens	-3.1		
Nereis procera	46.2	71.7	66.5
Neverita reclusiana	29.7	57.1	54.2
Nicippe tumida		-26.9	4.3
Ninoe tridentata	42.0	23.2	41.6
Notocirrus californiensis	-1.3	4.5	
Notomastus sp.	73.9	96.9	54.1
Notoproctus pacificus		-31.1	
Nuculana sp.	13.5	19.5	16.8
Odontosyllis phosphorea	31.0	38.6	
Odostomia sp.	27.0	30.1	26.9
Ogyrides Sp. A	-14.2		
Olivella baetica	37.5	76.6	
Onuphis iridescens complex	19.2	29.7	35.7
Ophelia pulchella	-20.4		
Ophelina acuminata	1.9	-1.6	-12.0
Ophiodermella sp.	7.1	-2.9	
Ophiura luetkeni		-7.5	-21.9
Ophiuroconis bispinosa	-2.4	-17.0	-26.7
Ophryotrocha A/B/C complex		204.1	198.8
Opisa tridentata		-5.9	2.7
Orchomene anaquelus	23.1	7.8	
Orchomene decipiens	33.5	8.6	11.6
Orchomene pacificus			-31.5
Orchomene pinguis		29.9	
Orthopagurus minimus	13.6	22.8	
Owenia collaris	-9.3	24.7	
Oxyurostylis pacifica	-5.7	25.7	
Pachynus barnardi	16.3	23.3	-19.4
Paguristes bakeri		55.6	
Paguristes turgidus		36.5	
Pagurus sp.	119.1	78.6	
Paleanotus bellis	4.5		
Pandora bilirata		-10.8	
Pandora filosa		-0.3	7.1
Paradiopatra parva	10.7	-21.6	11.8

Table 2 continued.

Species	*p* Shallow	*p* Middle	*p* Deep
Paradoneis eliasoni		-12.1	
Paramage scutata	21.5	-13.8	-2.1
Parametaphoxus fultoni	40.5	35.8	
Paramicrodeutopus schmitti	13.9		
Paranaitis polynoides		40.9	
Parandalia sp.	23.0	38.0	
Parapagurodes laurentae		9.5	-8.8
Paraprionospio pinnata	10.6	21.7	38.6
Parasterope sp.		17.9	
Pardaliscella sp.			2.2
Parougia caeca	35.7	72.9	
Parvilucina tenuisculpta	61.3	84.1	76.7
Pectinaria californiensis	40.8	28.1	31.2
Periploma/thracia complex	18.8	33.8	
Petaloproctus sp.		-7.4	
Petricola sp.	73.6	101.0	
Pherusa neopapillata	29.3	19.0	7.8
Philine bakeri	6.2		
Philine Sp. A	34.3	12.7	
Pholoe glabra	39.9	-4.8	15.4
Pholoides asperus		-21.1	-56.6
Phoronida	17.9	8.1	-2.3
Photis spp.	14.7	7.9	-9.4
Phyllochaetopterus limicolus	63.5	9.4	12.8
Phyllochaetopterus prolifica	-6.2	-17.8	
Phyllodoce sp.	6.2	35.7	34.9
Pilargis berkeleyae	49.5	43.8	
Pinnixa franciscana	67.8	50.0	
Pinnixa hiatus	61.6	79.1	
Pinnixa longipes	-1.7		
Pinnixa occidentalis	39.2	24.2	41.2
Pinnixa tomentosa	21.7		
Pinnixa tubicola	20.1	22.8	
Pionosyllis sp.		-10.1	
Piromis Sp. A		-10.8	
Pista alata	36.4	25.1	-2.9
Pista fasciata	19.9	36.2	30.2
Pista moorei	31.8	12.1	
Pista Sp. B	27.5	9.5	-10.3
Platynereis bicanaliculata	5.1	32.2	
Pleurobranchaea californica		-14.7	
Pleurogonium californiense		-24.6	-22.0
Pleusymtes subglaber	15.4	27.1	
Podarke pugettensis	121.4	156.2	
Podarkeopsis glabrus	51.9	47.5	95.6
Podarkeopsis Sp. A	34.5	24.6	
Podocerus sp.		-34.4	
Podochela sp.	26.2	3.4	
Poecilochaetus johnsoni	30.4	42.0	8.2
Poecilochaetus Sp. A	33.4	16.3	
Polinices draconis	25.2	61.2	
Polinices lewisii		32.8	

Table 2 continued.

Species	*p* Shallow	*p* Middle	*p* Deep
Polycirrus sp.	-1.1	-5.0	1.9
Polydora sp.	18.7	28.9	-0.4
Potamethus Sp. A	32.9	-31.1	-15.3
Prachynella lodo		19.1	23.0
Praxillella sp.	13.0	15.6	10.4
Praxillura maculata	3.7	6.4	
Prionospio A/B complex	55.5	31.5	32.6
Prionospio ehlersi			36.6
Prionospio lighti	63.6	27.8	34.6
Procampylaspis Sp. A		-28.0	-30.9
Proceraea sp.		-5.4	
Proclea Sp. A		-58.2	
Protodorvillea gracilis	-9.3	-1.1	
Protomedeia sp.		-9.1	-13.0
Prototthaca sp.	66.5		
Gadila aberrans	19.6	26.0	-14.5
Pyromaia tuberculata	31.6	40.3	
Randallia ornata	18.5	23.8	
Rhabdus rectius	34.8	41.9	30.0
Rhachotropis sp.		-9.4	37.5
Rhamphidonta retifera		63.5	
Rhamphobrachium longisetosum		-17.8	
Rhepoxynius abronius	-33.5		
Rhepoxynius bicuspidatus	12.5	-15.0	-16.3
Rhepoxynius heterocuspidatus	-8.1	5.5	
Rhepoxynius lucubrans		-20.1	
Rhepoxynius menziesi	-11.8	11.3	
Rhepoxynius stenodes	-2.4	4.4	
Rhepoxynius variatus	-30.8	-10.1	
Rhodine bitorquata		-20.8	2.4
Rictaxis punctocaelatus	74.7	76.8	63.6
Rocinela angustata			30.0
Rudilemboides stenopropodus	4.0	9.2	
Rutiderma sp.	10.9	6.5	-15.2
Neosabellaria cementarium	16.8	11.0	
Sabellides manriquei	-15.2	-34.6	
Samytha californiensis		-5.7	2.7
Saxicavella nybakkeni	35.8	12.3	23.9
Saxicavella pacifica	54.8	23.7	59.5
Saxidomus nuttalli	51.1		
Scalibregma inflatum	11.6	4.3	-11.0
Dorvillea (schistomeringos) longi	114.5	123.5	104.1
Scleroconcha trituberculata			-7.3
Scleroplax granulata	61.2	74.5	
Scolelepis occidentalis		8.5	
Scolelepis spp.	7.7	18.7	
Scoloplos armigera complex	-1.5	-9.2	-20.7
Serolis carinata	13.5	-6.6	
Sigalion spinosa	-19.6	-12.0	
Sigambra tentaculata	79.2	77.7	34.8
Siliqua lucida	36.8		
Sinum scopulosum	37.4	29.4	
Siphonodentalium quadrifissatum		-8.2	-5.4

Table 2 continued.

Species	p Shallow	p Middle	p Deep
Solamen columbianum	22.4	4.9	
Solariella peramabilis			-58.4
Solemya reidi	91.3	98.8	133.4
Solen sp.	24.7	27.5	
Sosane occidentalis		-53.0	
Sphaerosyllis sp.	22.7		
Spio sp.	4.9	-20.0	-18.3
Spiochaetopterus costarum	36.4	54.5	16.6
Spiophanes berkeleyorum	24.2	33.8	38.8
Spiophanes bombyx	-2.3	12.1	-23.7
Spiophanes fimbriata	22.4	-17.4	-2.6
Spiophanes missionensis	6.1	8.5	-1.6
Spiophanes wigleyi			8.5
Stenothoides bicoma	9.5	10.0	
Sternaspis fossor	34.2	-17.3	-1.5
Sthenelais spp.	12.8	3.4	3.8
Sthenelais verruculosa	-7.5	5.0	
Sthenelanella uniformis	9.8	-6.6	-3.4
Streblosoma sp.	40.4	25.4	17.4
Stylatula elongata	28.7	34.0	
Subadyte mexicana		-13.4	-1.4
Sulcoretusa xystrum	25.4	27.8	
Syllis (ehlersia) heterochaeta	5.1	5.8	15.1
Syllis (ehlersia) hyperioni	19.9	9.2	
Syllis (typosyllis) farallonensis	0.0		
Syllis (typosyllis) spp.	7.4	14.3	102.7
Synaptidae	4.3	-11.2	-20.6
Synchelidium sp.	-6.9	29.8	11.7
Synidotea sp.	6.0	13.6	
Syrrhoe Sp. A			6.1
Tellina carpenteri	36.8	51.3	49.1
Tellina idae	16.6	35.5	
Tellina modesta	-3.0	39.5	4.2
Tenonia priops	14.6	44.8	
Terebellides sp.	24.9	-6.8	0.3
Thelepus setosus		12.8	
Thesea Sp. B		-3.6	
Thyasira flexuosa	40.0	45.5	42.7
Tiron biocellata	7.9	14.6	
Trachycardium quadragenarium	18.7	11.0	
Transenella tantilla	92.7	47.9	
Travisia brevis		-37.0	-11.5
Tritella pilimana		-14.5	
Turbonilla sp.	46.1	45.8	13.6
Upogebia sp.		1.0	
Urothoe varvarini		-41.7	-41.9
Vargula tsujii		0.0	
Vitrinella sp.		61.3	
Volvulella californica		-11.6	-0.6
Volvulella cylindrica	-2.7	12.9	
Volvulella panamica	51.1	31.8	22.6
Westwoodilla caecula	40.4	17.6	2.5

Figure 6. Benthic Response Index values for a gradient of stations near the Sanitation Districts of Orange County's outfall in 1990.

Figure 7. Benthic Response Index values for stations on the Palos Verdes Shelf from 1972-1995.
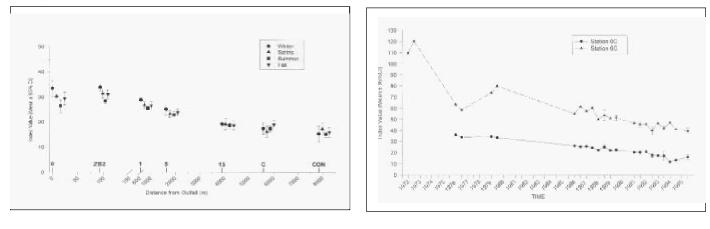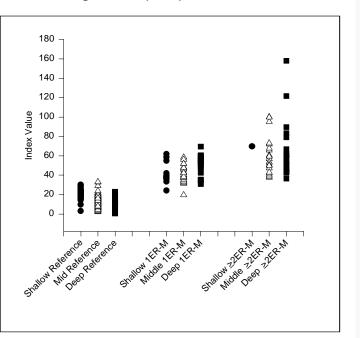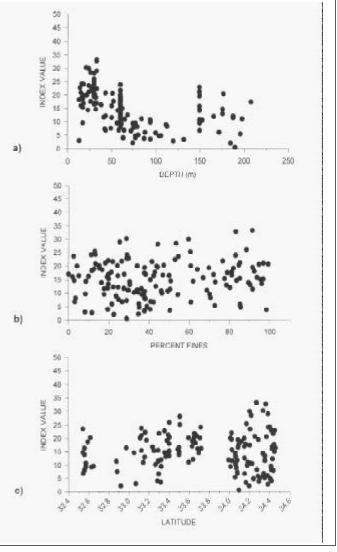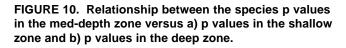

FIGURE 8. Benthic Response Index values within shallow, mid-depth and deep reference sites and at stations with one or more than two chemicals above the Effects Range Median (ER-M).

FIGURE 9. Benthic Response Index Values for reference stations versus:a) depth, b) percent fines and c) latitude

FIGURE 10. Relationship between the species p values in the med-depth zone versus a) p values in the shallow zone and b) p values in the deep zone.

FIGURE 11. Effect on the Benthic Response Index and Infaunal Trophic Index 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 computed from the calibration data.

LITERATURE CITED

Austin, M.P., and I. Noy-Meir. 1971. The problem of non-linearity in ordination experiments with two-gradient models. *Journal of Ecology* 59:762-773.

Beals, E.W. 1973. Ordination: Mathematical elegance and ecological naiveté. *Journal of Ecology* 61:23-35.

Bergen, M., S.B. Weisberg, D. Cadien, A. Dalkey, D. Montagne, R.W. Smith, J.K. Stull, and R.G. Velarde. 1998. Southern California Bight 1994 Pilot Project: IV. Benthic infauna. Southern California Coastal Water Research Project. Westminster, CA.

Bergen, M., E. Zeng, and C. Vista. 1995. The Southern California Bight Pilot Project: An experiment in cooperative regional monitoring. IEEE Oceans '95 Conference Proceedings: 526-536.

Bernstein, B.B., and Smith, R.W. 1986. Community approaches to monitoring. IEEE Oceans '86 Conference Proceedings: 934-939.

Boesch, D.F. 1973. Classification and community structure of macrobenthos in the Hampton Roads area, Virginia. *Marine Biology* 21:226-244.

Boesch, D.F. 1977. A new look at the zonation of benthos along the estuarine gradient. pp. 245-266 *in:* B.C. Coull (ed.), Ecology of Marine Benthos. University of South Carolina Press. Columbia, SC.

Boesch, D.F., and R. Rosenberg. 1981. Response to stress in marine benthic communities. pp. 179-200 *in:* G.W. Barret and R. Rosenberg (eds.), Stress Effects on Natural Ecosystems. John Wiley and Sons. New York, NY.

Bradfield, G.E., and N.C. Kenkel. 1987. Nonlinear ordination using shortest path adjustment of ecological distances. *Ecology* 68:750-753.

Bray, J.R., and J.T. Curtis. 1957. An ordination of the upland forest communities of southern Wisconsin. *Ecological Monographs* 27:325-349.

Chang, S., F.W. Steimle, R.N. Reid, S.A. Fromm, V.S. Zdanowicz, and R. Pikanowski. 1992. Association of benthic macrofauna with habitat types and quality in the New York Bight. *Marine Ecology Progress Series* 99:237-251.

City of Los Angeles, Environmental Monitoring Division. 1992. Marine Monitoring in Santa Monica Bay: Annual Assessment Report for the Period July 1990 through June 1991.

City of San Diego. 1987. Point Loma Ocean Monitoring Program Benthic Report April 1985 - January 1986. City of San Diego, Water Utilities Department, Metro Wastewater Division. San Diego, CA.

City of San Diego. 1991. City of San Diego Ocean Monitoring Program 1990 Annual Benthic Monitoring Report. City of San Diego, Water Utilities Department, Metro Wastewater Division. San Diego, CA.

Clifford, H.T., and W. Stephenson. 1975. An Introduction to Numerical Classification. Academic Press. New York, NY.

County Sanitation Districts of Orange County (CSDOC). 1986. 1985 Annual Report. County Sanitation Districts of Orange County. Fountain Valley, CA.

County Sanitation Districts of Orange County. 1991. 1991 Annual Report. County Sanitation Districts of Orange County. Fountain Valley, CA.

Dauer, D.M. 1993. Biological criteria, environmental health and estuarine macrobenthic community structure. *Marine Pollution Bulletin* 26:249-257.

Dauer, D.M, R.M. Ewing, and A.J. Rodi, Jr. 1987. Macrobenthic distribution within the sediment along an estuarine salinity gradient. Internationale Revue der Gesamten *Hydrobiologie* 72:529-538.

Dauer, D.M., T.L. Stokes, Jr., H.R. Barker, Jr., R.M. Ewing, and J.W. Sourbeer. 1984. Macrobenthic communities of the lower Chesapeake Bay. IV. Baywide transects and the inner continental shelf. Internationale Revue der Gesamten *Hydrobiologie* 69:1-22.

Elliott, M. 1994. The analysis of macrobenthic community data. *Marine Pollution Bulletin* 28:62-64.

Engle, V.D., J.K. Summers and G.R. Gaston. 1994. A benthic index of environmental condition of Gulf of Mexico estuaries. *Estuaries* 17:372-384.

Field, J.G. 1971. A numerical analysis of changes in the soft-bottom fauna along a transect across False Bay, South Africa. *J. Exp. Mar. Biol. & Ecol. 7: 215-253*.

Field, J.G., K.R. Clarke and R.M. Warwick. 1982. A practical strategy for analyzing multispecies distribution patterns. *Marine Ecology Progress Series* 8:37-52.

Fore, L.S., J.P. Karr, and R.W. Wisseman. 1996. Assessing invertebrate responses to human activities: Evaluating alternative approaches. *Journal of the North American Benthological Society* 15:212-231.

Gauch, H.G., Jr. 1982. Multivariate analysis in community ecology. Cambridge Studies in Ecology. Cambridge University Press. New York, NY.

Gerritsen, J. 1995. Additive biological indices for resource management. *Journal of the North American Benthological Society* 14:451-457.

Goff, F.G., and G. Cottam. 1967. Gradient analysis: The use of species and synthetic indices. *Ecology* 48:793-806.

Gower, J.C. 1966. Some distance properties of latent root and vector methods used in multivariate analysis. *Biometrica* 53:325-338.

Gower, J.C. 1967. Multivariate analysis and multidimensional geometry. *The Statistician* 17: 13-28.

Gray, J.S. 1979. Pollution-induced changes in populations. *Transactions of the Royal Philosophical Society of London* (B) 286:545-561.

Hahn, G.J., and W.Q. Meeker. 1991. Statistical Intervals. A Guide for Practitioners. John Wiley & Sons, Inc. New York, NY.

Heip, C., and J.A. Craeymeersch. 1995. Benthic community structures in the North Sea. *Helgolander Meeresuntersuchungen* 49:313-328.

Hilsenhoff, W.L. 1977. Use of arthropods to evaluate water quality of streams. Technical Bulletin. Wisconsin Department of Natural Resources 100. 15 pp.

Holland, A.F., A. Shaughnessey, and M.H. Heigel. 1987. Long-term variation in mesohaline Chesapeake Bay benthos: Spatial and temporal patterns. *Estuaries* 10:227-245.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environmental Management* 19:81-97.

Los Angeles County Sanitation Districts. 1990. Palos Verdes Ocean Monitoring, Annual Report 1990. Submitted to the Los Angeles Regional Water Quality Control Board. Los Angeles County Sanitation Districts. Whittier, CA.

Nelson, W.G. 1990. Prospects for development of an index of biotic integrity for evaluating habitat degradation in coastal ecosystems. *Chemistry and Ecology* 4:197- 10.

Norris, R.H. 1995. Biological monitoring: The dilemma of data analysis. *Journal of the North American Biological Society* 14:440-450.

O'Connor, J.S., and R.T. Dewling. 1986. Indices of marine degradation: Their utility. *Environmental Management* 10:335-343.

Orloci, L. and M.M. Mukkattu. 1973. Effects of species number and type of data on the resemblance structure of a phytosociological collection. *Journal of Ecology* 61: 37-46.

Pearson, T.H., and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology Annual Review* 16:229-311.

Pielou, E.C. 1984. The Interpretation of Ecological Data. John Wiley and Sons. New York, NY.

Rhoads, D.C., P.L. McCall, and J.Y. Yingst. 1978. Disturbance and production on the estuarine sea floor. *American Scientist* 66:577-586.

Schaffner, L.C., R.J. Diaz, C.R. Olson, and I.L. Larsen. 1987. Faunal characteristics and sediment accumulation processes in the James River Estuary, Virginia. *Estuarine, Coastal and Shelf Science* 25:211-226.

Smith, R.W. 1976. Numerical Analysis of Ecological Survey Data. Ph.D. dissertation. University of Southern California. Los Angeles, CA.

Smith, R.W., and B.B. Bernstein. 1985. Index 5: A multivariate index of benthic degradation. Report prepared for NOAA, under contract to Brookhaven Nat. Lab.

Smith, R.W., B.B. Bernstein, and R.L. Cimberg. 1988. Community-environmental relationships in the benthos: Applications of multivariate analytical techniques. Chapter 11, pp. 247-326 *in*: Marine Organisms as Indicators. Springer-Verlag. New York, NY.

Sneath, P.A., and R.R. Sokal. 1973. Numerical Taxonomy. W.H. Freeman and Co. San Francisco, CA.

Snelgrove, P.V., and C.A. Butman. 1994. Animal-sediment relationships revisited: Cause vs. effect. *Oceanography and Marine Biology Annual Review* 32:111-177.

Stull, J. 1995. Two decades of biological monitoring, Palos Verdes, California, 1972 to 1992. *Bulletin of the Southern California Academy of Sciences* 94:21-45.

Stull, J.K., C.I. Haydock, and D.E. Montagne. 1986a. Effects of Listriolobus pelodes (Echiura) on coastal shelf benthic communities and sediments modified by a major California waste water discharge. *Estuarine, Coastal and Shelf Science* 22:1-17.

Stull, J., C.I. Haydock, R.W. Smith, and D.E. Montagne. 1986b. Long-term changes in the benthic community on the coastal shelf of Palos Verdes, Southern California. *Marine Biology* 91:539-551.

Swan, J.M.A. 1970. An examination of some ordination problems by use of simulated vegetational data. *Ecology* 51:89-102.

Tapp, J.F., N. Shillabeer, and C.M. Ashman. 1993. Continued observation of the benthic fauna of the industrialized Tees estuary, 1979-1990. *Journal of Experimental Marine Biology and Ecology* 172:67-80.

Thompson, B.E., J.D. Laughlin, and D.T. Tsukada. 1987. 1985 reference site survey. Technical Report 221. Southern California Coastal Water Research Project. Westminster, CA.

Thompson, B.E., D.T. Tsukada, and D. O'Donohue. 1993. 1990 reference site survey. Technical Report 269. Southern California Coastal Water Research Project. Westminster, CA.

Van Dolah, R.F., J.L. Hyland, A.F. Holland, J.S. Rosen, and T.R. Snoots. 1999. A benthic index of biological integrity for assessing habitat quality in estuaries of the southeastern United States. *Marine Environmental Research* (in press).

Vardeman, S.B. 1992. What about the other intervals? *The American Statistician* 46:193-197.

Warwick, R.M., and K.R. Clarke. 1993. Increased variability as a symptom of stress in marine environments. *Journal of Experimental Marine Biology and Ecology* 172:215-226.

Warwick, R.M., and K.R. Clarke. 1994. Relearning the ABC: Taxonomic changes and abundance/biomass relationships in disturbed benthic communities. Marine Biology 118:739-744.

Weisberg, S.B., J.A. Ranasinghe, D.M. Dauer, L.C. Schaffner, R.J. Diaz, and J.B. Frithsen. 1997. An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. *Estuaries* 20:149-158.

Whittaker, R.H. 1973. Direct gradient analysis: Techniques. pp. 54-73 *in*: Handbook of Vegetation Science. Part V. Ordination and Classification of Communities. Dr. W. Junk b.v., The Hague.

Williamson, M.H. 1978. The ordination of incidence data. *Journal of Ecology* 66:911-920.

Wilson, J.G., and D.W. Jeffrey. 1994. Benthic biological pollution indices in estuaries. pp. 311-327 *in:* J.M. Kramer (ed.), Biomonitoring of Coastal Waters and Estuaries. CRC Press. Boca Raton, FL.

Word, J.Q. 1978. The infaunal trophic index. pp. 19-39 *in*: Southern California Coastal Water Research Project Annual Report. El Segundo, CA.

Word, J.Q. 1980a. Extension of the infaunal trophic index to a depth of 800 meters. pp. 95-101 *in*: Southern California Coastal Water Research Project Biennial Report 1979-1980. Long Beach, CA.

Word, J.Q. 1980b. Classification of benthic invertebrates into Infaunal Trophic Index feeding groups. pp. 103-121 *in*: Southern California Coastal Water Research Project Biennial Report 1979-1980. Long Beach, CA.

Word, J.Q. 1990. The infaunal trophic index, a functional approach to benthic community analyses. Ph.D. dissertation. University of Washington. Seattle, WA.

Word, J.Q., and A.J. Mearns. 1979. 60-m control survey off southern California. Technical Memorandum 229. Southern California Coastal Water Research Project. Long Beach, CA.

Appendix A. Optimization Procedure for Index Development

To produce the final formulation of our index, three factors needed to be determined, one factor for calculating the index (Equation 1) and two factors for calculating the positions (p_i values) of the species on the pollution gradient (Equation 2). For the index, it was necessary to determine the optimal weighting factor for the abundance. To calculate the p_i values for the species, it was necessary to determine the optimal weighting factor for the abundance and the number of values to be used in the calculation.

Since the pollution gradient from the ordination space is the "standard" upon which the index is based, the degree to which index values correlate with this standard is a measure of the success of the index. In other words, the success of the weighting and other factors used to compute the index can be evaluated by how well the index values for the *calibration* samples correlate with their gradient positions along the pollution gradient (i.e., projections onto the lines in Figure 2) defined in the ordination space. Thus, a Pearson correlation coefficient can be used to determine the optimal configuration of factors in the equation. Our strategy was to vary the factors used to compute the index and the species positions on the gradient and to evaluate each formulation with the resulting correlation coefficient. The optimal approach will have the highest correlation coefficient. Since all three factors affect the final outcome, the factors were optimized concurrently.

To describe the optimization procedure, Equations (1) and (2) must be expressed in a more general form. The general form of the index weighted average (Equation 1) is

$$
p_i = \frac{\displaystyle\sum_{j=1}^{t} a_{ij}^{e} g_j}{\displaystyle\sum_{j=1}^{n} a_{ij}^{e}} \qquad (a1)
$$

where I_s is the index value for sample s, n is the number of species in the sample s, p_i is the position for species i on the gradient, and a_{si} is the abundance of species i in sample s. The exponent f is for transforming the abundance weights. For example, if $f=1$, the raw abundance values are used. If $f=0$, I_s is the arithmetic average of the p_i values greater than zero (all), and if $f=0.5$, the square root of the abundances are used. Species in the sample without species positions are ignored, and species with abundances of zero are not used in the sum when $f=0$. In the optimization procedure, different values of the abundance weight transformation parameter f in Equation (a1) are tested.

The general form of the equation to compute the species positions on the gradient is

$$
I_s = \frac{\displaystyle\sum_{j=1}^{n} a_{si}^{f} p_i}{\displaystyle\sum_{i=1}^{n} a_{si}^{f}}, \qquad (a2)
$$

where e is a variable exponent for transforming the abundance, and t is the number of samples to be used in the sum, with only the sampling units with highest t species abundance values included. Thus, if $t=4$, j will range from 1 to 4. When $j=1$, j signifies the sample with the highest abundance count for species i; when $j=2$, j signifies the sample with the second highest abundance count for species i; and so on up to the fourth highest abundance value. The g_j is the position on the ordination gradient for sample j, and a_{ij} is the abundance of species i in sample j.

In the optimization procedure, the two parameters that can be varied in Equation (a2) are t and e. The t value is used to adjust the number of samples used in the weighted average, with precedence given to the samples with higher abundances. The e parameter is used to transform the abundance weights and avoid overemphasis on one or a few relatively high abundance values.

Since the combined values set for e and t in Equation (a2) and f in Equation (a1) will all affect the index values computed with Equation (a1), the optimization procedure must involve testing different combinations of e, t, and f values for the maximum correlation.

In the optimization, we computed correlation coefficients for all combinations of $e = 0, 1, 0.5, 0.33, 0.25$, and $t=1$ to 100 in Equation (a2), and $f = 0, 1, 0.5, 0.33, 0.25$ in Equation (a1). The optimum e, t, and f values for the different depth ranges are summarized in Table A1. The high correlations show that by using these parameter values, the weighted averages from Equation (a1) accurately reproduce the pollution gradients extracted from the ordination spaces. Note that the parameters in Table A1 result in Equations (1) and (2).