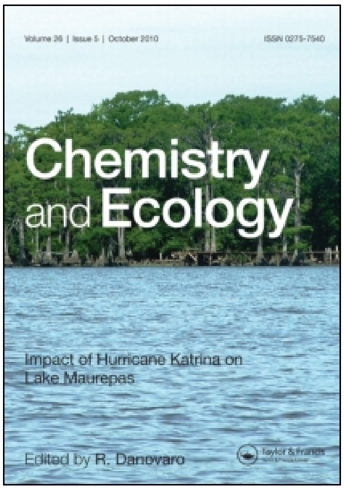

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Prospects for Development of an Index of Biotic Integrity for Evaluating Habitat Degradation in Coastal Systems

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PROSPECTS FOR DEVELOPMENT OF AN INDEX OF BIOTIC INTEGRITY FOR EVALUATING HABITAT DEGRADATION IN COASTAL SYSTEMS

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A multivariate index for assessment of habitat quality and its degradation, the Index of Biotic Integrity (IBI), has been developed for stream fish communities. The extension of the IBI concept to coastal waters is proposed by the development of an estuarine IBI based on macrobenthos and submerged aquatic vegetation. Eleven variables are proposed for inclusion in this index, although further consideration of the appropriateness of several variables is required. It is concluded that development of an IBI for coastal systems is feasible.

INTRODUCTION

The development of the coastal zone is resulting in increased pressure on the biological resources of this region throughout the world. Thus, the need for techniques to evaluate the changes in estuarine and coastal marine habitats is becoming ever more pressing. The development of methods to allow surveillance of trends in habitat status, and especially to allow the early detection of habitat degradation, is particularly needed.

The continued pressure for the development of indices of habitat degradation stems from the need to provide summations of underlying data in a form which can be easily used and understood by those faced with making environmentally important decisions. O'Connor and Dewling (1986) point out that appropriately designed indices can be more readily interpretable not only to the layman, but also to the scientist. They list five criteria which are deemed important to a suitable index of ecosystem degradation (O'Connor and Dewling, 1986). These are that the index be relevant, simple and easily understood by laymen, scientifically defensible, quantitative and acceptable in terms of cost.

Over the last 30 years, many approaches have been used to derive standardized and repeatable measures of habitat status for the marine environment. A partial list of these approaches and representative citations which provide examples of their use are given in Table 1. There has been some tendency to move away from the more reductionist, univariate type of approach represented by diversity indices or single indicator species to multivariate measures of the habitat (Bernstein & Smith, 1986; USFWS, 1980).

Table 1 A partial list of proposed approaches to the assessment of habitat degradation in marine and estuarine ecosystems

Assessment method	*Typical reference*
1. Benthic Pollution Index	Leppakoski, 1975
2. Size Class Distributions	Leppakoski, 1975 Hargrave & Thiel, 1983
3. Indicator Species	Pearson & Rosenberg, 1978
4. Log-normal Plotting	Gray & Mirza, 1979
5. Infaunal Trophic Index	Word, 1979
6. Nematode/Copepod Ratio	Raffaelli & Mason, 1981
7. *H'* Diversity Index	Gray & Pearson, 1982
8. Rarefaction	Gray & Pearson, 1982
9. Benthic Resources Assessment Technique	Lunz & Kendall, 1982
10. Dominance	Shaw *et al.*, 1983
11. Ordination	Bernstein & Smith, 1986
12. Indices of Coastal Degradation	O'Connor & Dewling, 1986
12.1 Dietary risks from toxicants in marine foods	
12.2 Pollutant stress in sediments	
12.3 Pollutant stress in water columns	
12.4 Human pathogen risks	
12.5 Benthic species composition and abundance	
12.6 Fish and shellfish diseases	
12.7 Fecundity in fish and shellfish	
12.8 Mortality in eggs and larvae of fish and shellfish—field measurements	
12.9 Mortality in eggs and larvae of fish and shellfish—laboratory measurements	
12.10 Reproductive success in marine birds	
12.11 Oxygen depletion effects	

THE INDEX OF BIOTIC INTEGRITY

The development of a multivariate index for stream fish communities, the Index of Biotic Integrity (IBI), appears to be a potentially powerful method for assessment of habitat quality and its degradation. The methodology was initially developed by Karr (1981) for stream communities of the midwestern United States. The technique was further refined and tested in a series of studies (Karr *et al.*, 1984; Fausch *et al.*, 1984; Karr *et al.*, 1986; Karr *et al.*, 1987). The IBI concept has also been successfully modified for US streams in California, Oregon, West Virginia, New England (references cited in Miller *et al.*, 1988), and in Ontario, Canada (Steedman, 1988).

The IBI approach is based on the premise that physical/chemical parameters used in monitoring of water quality are only surrogate measures of the biological integrity of aquatic systems (Karr and Dudley, 1978; Karr *et al.*, 1986; Miller *et al.*, 1988). It is possible that maintenance of high water quality standards in terms of physico-chemical parameters may not stop biotic degradation if habitat destruction is of greater importance (Miller *et al.*, 1988). Therefore, biotic integrity is measured using a variety of parameters (termed "metrics" by Karr *et al.*, 1986) which vary in sensitivity to habitat degradation (Table 2). The IBI in its

Table 2 List of metrics utilized in the Index of Biotic Integrity (Karr *et al.*, 1986)

Species Richness and Composition
1. Total number of species
2. Number and identity of intolerant species
3. Number and identity of darter species
4. Number and identity of sucker species
5. Number and identity of sunfish species
6. % of individuals as green sunfish
Trophic Composition
7. % of individuals as omnivores
8. % of individuals as insectivores
9. % of individuals as top carnivores
Fish Abundance and Condition
10. Number of individuals per sample
11. % of individuals as hybrids
12. % of individuals with disease

original form used 12 metrics divided into 3 categories:

Species Richness and Composition
Trophic Composition
Fish Abundance and Condition

The IBI establishes relative values for each of the 12 metrics based on comparison of values for the best available habitat (with minimal human disturbance) to those areas which are strongly disturbed. The values used are: 5—metric close to natural value, 3—metric deviates somewhat from natural value, 1—metric strongly deviates from natural value. An example from Karr *et al.* (1986) illustrates the assignment of values to the metric of total species number based on 72 collection sites along an Illinois river (Figure 1). Values for all metrics are then summed (with equal weighting) to produce the Index of Biotic Integrity, and ranges of scores are characterized as representing regions of environmental quality from very poor to excellent. A further example from Karr

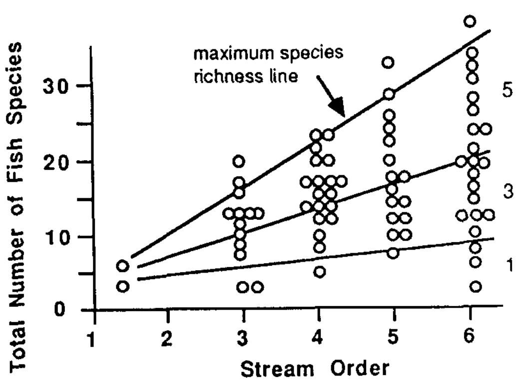

Figure 1 Illustration of the assignment of metric values (5, 3, 1) for the parameter "total number of fish species" based on observations from 72 sites along a river in Illinois. (From Karr *et al.*, 1986)

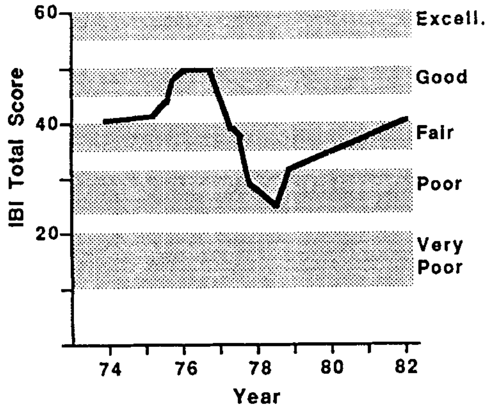

Figure 2 An example of the response of the IBI to changes in habitat quality caused by disturbance of the stream banks in a small stream in the midwestern U.S. The disturbance occurred in 1976 and resulted in increased sedimentation in the stream. (From Karr *et al.*, 1986)

et al. (1986) illustrates how the IBI responded to an environmental perturbation of a midwestern stream (Figure 2).

The IBI developed by Karr (1981), and used by all subsequent investigators, utilized fishes because it was felt that fishes were easier to identify and that they had more relevance to the public than other organisms. The technique assumes that the entire fauna is sampled such that all species are recorded in their true relative abundances without bias.

The IBI is based on ecological concepts of the community, and appears to be relatively flexible in its application. The use of the index in other geographic areas has required substitution of varied numbers of the original metrics (Miller *et al.*, 1988), yet the index appears to remain effective. In a recent modification, Steedman (1988) was able to find good correlation between the IBI and such measures of general habitat degradation as "proportion of the drainage basin in urban land use".

EXTENSION OF THE IBI CONCEPT TO COASTAL SYSTEMS

The IBI concept has been extended to estuaries, again using fishes only (Thompson & Fitzhugh, 1986). The prototype estuarine IBI used 13 metrics in order to include a parameter to deal with high levels of seasonal variability (Table 3). The use of fishes in habitat assessments in estuaries has the advantages that fishes are relatively easy to identify, life history information is often available, and social relevance is readily identified (Nelson, 1987). However, estuarine systems have particular difficulties for the application of general habitat status models with fishes. Daily and seasonal movements can confound correlations with habitat parameters (Nelson, 1987). Estuarine fishes are adapted to a dynamic environment and may be tolerant of a wide variety of physico-chemical changes such that they may be sensitive only to large-scale perturbations of the environment (Nelson, 1987). Of particular difficulty is the fact that sampling difficulties for fishes may be extremely high relative to the more confined habitats of higher order streams for which the IBI was originally developed. Thus, meeting the assumption that all species are sampled in their true relative proportions without bias may be difficult to meet for fishes in these systems.

As a potential alternative to the use of fishes in an esturine IBI, the combined use of macrobenthos and submerged aquatic vegetation (SAV) offers certain

Table 3 A list of metrics used in the prototype IBI developed for estuarine fish communities in Louisiana by Thompson and Fitzhugh (1986)

Category
Species Composition
1. Total number of fish species
2. Number and identity of estuarine species
3. Number and identity of marine species
4. Number and identity of scianids
5. Number and identity of freshwater species
6. Proportion of individuals as bay anchovies
7. Seasonal overlap of fish community
8. Number of species it takes to make up 90% of collection
Trophic Composition
9. Proportion of individuals as benthic feeders
10. Proportion of individuals as planktonic grazers
11. Proportion of individuals as top carnivores
Fish Condition
12. Proportion of young of year in sample or number of individuals in sample
13. Proportion of individuals with disease, tumors, fin damage, and other anomalies

advantages. Both groups are much more sedentary than fishes, and thus may tend to integrate environmental conditions over the longer term (Reish, 1986; Nelson, 1987). In contrast, planktonic and nektonic organisms will tend to reflect conditions only at the time of sampling (Reish, 1986). Based on long-term studies of the macrobenthos both Reish (1986) and Holland and Shaughnessey (1986) suggested that the macrobenthos is a sensitive indicator of pollutant effects. Because benthic assemblages are closely linked to both lower and higher trophic levels, as well as to processes influencing water and sediment quality, such as sedimentation rates and nutrient inputs, they appear to integrate responses of the entire system (Leppäkoski, 1979; Holland and Shaughnessey, 1986). A major advantage of using macrobenthos or SAV in an estuarine IBI is that either can be easily sampled both quantitatively and efficiently.

Several clear disadvantages of using macrobenthos are that these species are generally more difficult to identify than fishes, they may have little life history information available, and they may have a less clearly defined relevance to the public (Nelson, 1987). While seagrasses might not be subject to these problems, algae, which might make up a major component of SAV, would probably be subject to constraints similar to those affecting the macrobenthos.

While the use of fishes in the development of an estuarine IBI should clearly be explored further, the remainder of this paper will deal with the prospects for the development of an estuarine IBI based on the use of macrobenthos and SAV.

AN ESTUARINE MACROBENTHOS IBI

Adaptation of the IBI concept to macrobenthos requires considerable modification, and at this initial stage of development several of the metrics proposed must be considered highly tentative. The basic approach follows that of Karr (1981, Karr *et al.*, 1986) of dividing metrics into the three categories of species composition, trophic composition and abundance. Condition metrics such as the number of individuals with diseases would be much more difficult to apply to macrobenthos at the present state of knowledge and are not included. Disease condition and its relationship to toxicants (e.g., Bryan *et al.*, 1987) may be promising direction for future study, however.

Because of clear differences in the faunal composition between habitat types within estuaries (e.g. between seagrass beds and unvegetated substrates (Virnstein *et al.*, 1984)), development of any IBI will be largely habitat specific. While metrics may be applicable across several different habitats, the guidelines for assignment of values to these metrics will vary widely with habitat, and therefore must be established for each habitat type to be examined. As an example, density of macrobenthos in seagrass beds in the Indian River lagoon of Florida may typically exceed that in adjacent sand habitat by 300% (Virnstein *et al.*, 1984). Maximum abundance levels for undisturbed vegetated and unvegetated habitats clearly must be established separately.

As a sample habitat for development of a macrobenthos-based IBI, the seagrass beds of the Indian River lagoon, Florida, will be considered. The list of proposed metrics for a macrobenthos-based IBI is given in Table 4. The rationale for each metric will be examined briefly in turn.

1. *Species richness and composition of macrofauna*. The total number of benthic species within a community tends to decrease with increasing pollutant input (Pearson & Rosenberg, 1978; Botton, 1979; Mirza & Gray, 1981). Factors influencing this parameter which must be considered include seasonality and the biomass of the seagrass bed present. Examination of seasonality of amphipods over a 4-yr period from 5 seagrass beds in the Indian River lagoon indicates that regular seasonal cycles in species richness are present (Nelson *et al.*, 1982; Figure

Table 4 A list of proposed metrics for assessment of macrobenthic communitites of seagrass beds in the Indian River lagoon region of Florida

Metric	*Expected response to disturbance*
Species richness and composition	
1. Total number of macrobenthic species	decrease
2. % of individuals as amphipoda	decrease
3. % of individuals as opportunistic species	increase
4. % of individuals as polychaetes/oligochaetes	increase
5. % of individuals as molluscs	increase (?)
Trophic composition	
6. % of individuals as deposit feeders	increase
7. % of individuals as carnivores	decrease (?)
Abundance and size	
8. Mean number of individuals per sample	various
9. Dominance of most abundant species	increase
10. Mean size of organisms in habitat	decrease
Vegetation cover	
11. % cover of submerged aquatic vegetation	decrease

3). Seasonal patterns would allow the establishment of maximum species richness versus season for undisturbed seagrass beds, and this relationship would in turn allow a reasonable assignment of metric values for disturbed areas with seasonality factored out. While different taxa may respond to biomass of SAV in different ways, Virnstein *et al.* (1984) have found a clear relationship for amphipod crustaceans versus biomass which suggests that this variable may also be dealt with (Figure 4) in assigning metric values.

2. *Percent of individuals as amphipods.* Botton (1979) has found that ampeliscid amphipods were largely absent from a sludge dumping site relative to controls and suggested that absence of this group may be an indicator of

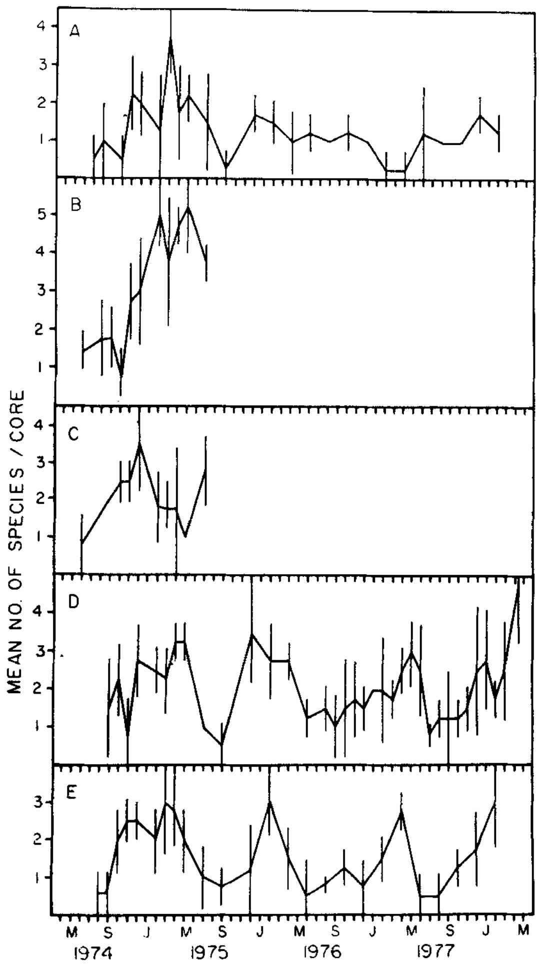

Figure 3 Seasonality of number of amphipod species per core at five sampling stations in seagrass beds in the Indian River lagoon, Florida. Variation in species number shows generally repeatable patterns from year to year. Variability would be decreased by using total number of species rather than mean number per core. (From Nelson *et al.*, 1982)

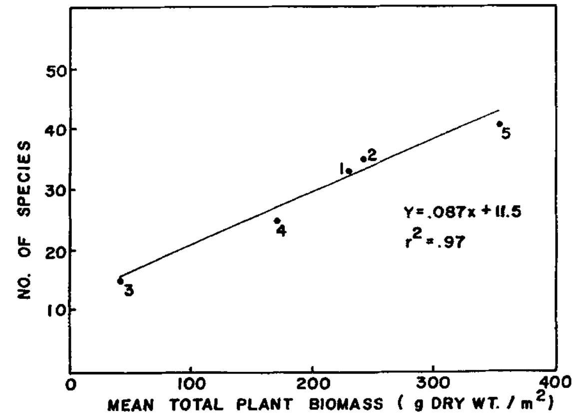

Figure 4 Regression of mean plant biomass per m^2 versus total number of amphipod species at five sampling sites. Such relationships could form the basis for establishment of metric values adjusted for habitat variability for the parameter "total number of species" and "total number of amphipod species" (From Virnstein *et al.*, 1984).

contamination. Similar observations of the disappearance of amphipod crustaceans in the proximity of organic enrichment have been made for both coasts of the U.S. (Boesch, 1982; Mearns & Word, 1982). McCall (1977) found ampeliscids recruited rapidly to colonization trays, a typical opportunistic species response, yet they tended to be absent from organically enriched areas. Ampeliscid amphipods may thus be a sensitive indicator of organic enrichment rather than general physical disturbance. Swartz *et al.* (1982) have also found a negative correlation between the presence of amphipods in the field and the toxicity levels of sediments as measured in laboratory bioassays.

3. *Percent of individuals as opportunistic species.* A comparison of the life history characteristics of opportunistic and equilibrium macrobenthic species was given by McCall (1977). Opportunistic species tend to increase in the vicinity of disturbances, whether natural or derived from disturbances such as organic enrichment of the bottom (Pearson & Rosenberg, 1978). Mirza & Gray (1981) suggested that opportunistic species will be relatively more dominant in disturbed areas, not necessarily because of higher tolerances to the environmental stress, but because these species possess better recruitment abilities when conditions moderate.

4. *Percent of individuals as polychaetes/oligochaetes.* Disturbed benthic areas are often characterized by a dominance by polychaetes and/or oligochaetes (McCall, 1977; Pearson & Rosenberg, 1978; Botton, 1979; Gray, 1979; Rosenberg *et al.*, 1987). Oligochaetes may be particularly tolerant of disturbed conditions which result in low oxygen levels (Hunter & Arthur, 1978). Pearson & Rosenberg (1978) have suggested that the use of groups of characteristic species rather than single indicator species may be more useful in characterizing pollution effects on the community. Gray & Pearson (1982) have proposed a method for objective selection of such pollution sensitive groups. It is possible that their technique might lead to a more precise definition of this particular metric, such as "percent capitellid polychaetes and oligochaetes".

5. *Percent of individuals as molluscs.* Certain molluscs have been suggested as pollution indicator species (reviewed in Pearson & Rosenberg, 1978). Botton (1979) found increased densities of a deposit feeding bivalve in a sewage sludge disposal area in the New York Bight. Gray (1979) lists several bivalve species as indicators of slight pollution levels, and Caspers (1987) has observed increased densities of bivalve molluscs near a sludge disposal site. However, Gray (1976) reported strong reduction in a number of species of molluscs in response to pollution of the Tees estuary. Josefson and Rosenberg (1988) have also found decreased abundances of molluscs in shallow fjords of Sweden over a 10 year period which they attribute to increased frequency of bottom water anoxia caused by regional eutrophication. Therefore, the predicted response pattern of molluscs is considered highly tentative at present, and further work is needed to determine if a substitute metric would be preferred.

6. *Percent of individuals as deposit feeders.* Pearson & Rosenberg (1978) have suggested that there will be a decrease in the relative composition of suspension feeders and an increase in the composition of deposit feeders as sediment organic concentrations increase. Reish (1986), based on 35 years of study of benthic polychaetes and pollution, concluded that there is a trend towards exclusion of all trophic groups except deposit feeders with increased pollution. Botton (1979) observed an increase of deposit feeders in a sludge disposal area based on abundance, but little difference based on biomass. In a series of papers, Word (1979; Mearns & Word, 1982; Word, 1980a, 1980b) developed the infaunal trophic index which is based on the shift in dominance of suspension feeders to deposit feeders as organic enrichment takes place. This index has been useful in describing areas affected by organic pollution off the coast of southern California.

7. *Percent of individuals as carnivores.* The utility of this particular metric is highly tentative. It is based on the assumption that the changes which occur in terms of the tendency of disturbed systems to shift to opportunistic species may affect top benthic infaunal or epifaunal carnivores adapted to a different set of prey items. Both Botton (1979) and Boesch (1982) found that the epibenthic predator *Cancer irroratus* decreased in benthic areas affected by waste discharges. Josefson and Rosenberg (1988) observed decreases in carnivores in bottom communities of shallow Swedish fjords which they related to eutrophication effects. One alternative may be to restrict this metric to percent predatory infauna, because these species may be less variable in space and time as compared with mobile, demersal crabs and fishes.

8. *Mean number of individuals per sample.* The response pattern of abundance to benthic pollution may vary depending on the intensity, longevity, or the nature of the disturbance. Pearson & Rosenberg (1978) have described the classic response pattern of abundance along an organic enrichment gradient. With increased organic enrichment to moderate levels, abundance tends to increase to a maximum at some distance from the enrichment source. Recent observations in the Skagerrak and Kattegatt off the coast of Sweden and Denmark suggest that such faunal increases due to organic enrichment have occurred in many parts of this region (Pearson *et al.*, 1985; Rosenberg *et al.*, 1987). The maximum point tends to be associated with maximum density of opportunist species (Pearson & Rosenberg, 1978). As enrichment levels continue to rise, abundance of benthos decreases, and if pollution levels are high enough, abundance may approach zero. The response of abundance may also vary depending on the nature of the

pollutant or disturbance. In the original IBI, assignment of values to this metric was rather qualitative (Karr *et al.*, 1986). Miller *et al.* (1988) have attempted to develop more quantitative methods to score this metric which involved plots of fish abundance versus watershed area. It is not clear that macrobenthos abundance would respond to a similar area variable in any consistent fashion. For unvegetated systems, a potential means for scoring this metric may be to define a maximum density line for macrobenthos versus a measure of organic loading in the sediment. Rather than directly measuring sediment organic content, it may be possible to measure depth of the oxidized sediment layer and develop plots of density versus this parameter (Rhoads and Germano, 1982). Alternately, direct measurement of sediment redox potential might be used (Pearson and Stanley, 1979). For seagrass beds, development of maximum density lines versus seagrass biomass would probably be more meaningful. Gore *et al.* (1981) have demonstrated a relationship between macrocrustacean abundance and seagrass biomass (Figure 5), which suggests that it may be possible to describe similar relationships for the entire macrobenthos. In both cases, the high degree of seasonal variability would also have to be considered in the development of scaled scoring values for the abundance parameter.

9. *Dominance of most abundant species.* Increased dominance by one or a few benthic species appears to be a typical response to benthic pollution or disturbance (Pearson & Rosenberg, 1978; Mirza and Gray, 1981). Shaw *et al.* (1983) have suggested that one of the best indices of pollution for benthic nematodes is a simple dominance index such as percent dominance. Lambshead *et al.* (1983) have extended the simple dominance index to the "*k*-dominance" index which appears to be a reasonable approach to quantifying this variable. Gray (1985), in a review of several concepts of marine pollution monitoring for the benthos, concurred that the dominance approach of Lambshead *et al.* (1983) appeared promising.

10. *Mean size of organisms in the habitat.* Pearson and Rosenberg (1978) found evidence that the mean physical size of macrofauna tended to decrease under

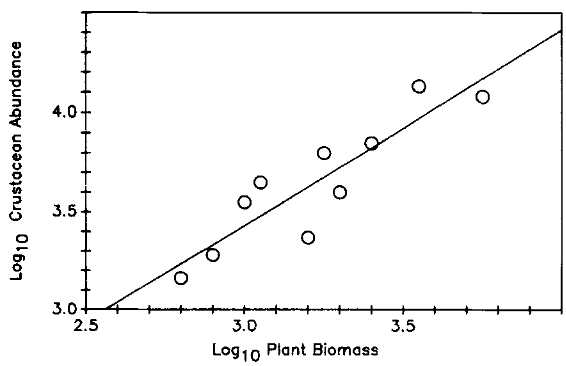

Figure 5 An example of a regression between log transformed biomass of seagrass and the abundance of macrocrustaceans collected from a seagrass bed in the Indian River lagoon, Florida. This figure suggests that it may be possible to establish criteria for scoring the metric "mean number

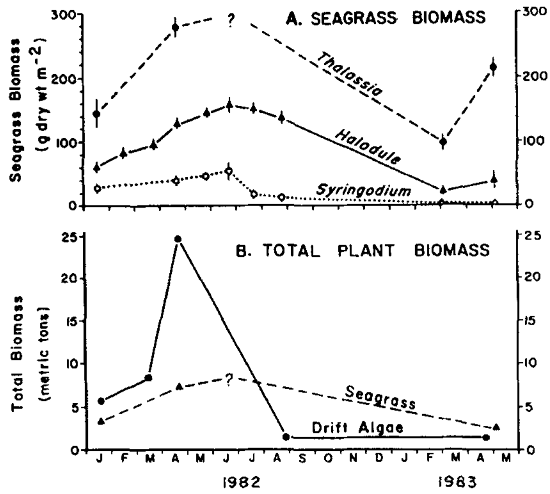

Figure 6 Example of the seasonal variation in biomass of seagrasses and drift algae from the Indian River lagoon, Florida. Such natural variation must be accounted for in determining metric values for the parameter "% cover of submerged aquatic vegetation". (From Virnstein and Carbonara, 1985)

organically polluted conditions. Hargrave and Thiel (1983) reviewed this evidence and suggested that size distribution may be a useful approach to pollution monitoring. Pearson *et al.* (1985) observed evidence of a shift in benthic populations to smaller organisms in the Kattegatt off Sweden which they attribute to the general eutrophication of the area.

11. *Percent cover of submerged aquatic vegetation.* The importance of SAV in aquatic systems is reasonably well documented (Diaz, 1982; Diaz *et al.*, 1982a, b). Declines of commercial catches of shrimp, crabs and squid in Japan have been associated with declines in SAV. Also, declines of waterfowl in the U.S. have been attributed to declines of SAV both in the massive natural declines of the 1930's and in the recent declines observed in the Chesapeake Estuary (Diaz *et al.*, 1982a). Virnstein *et al.* (1984) found 13 times more epifauna and 7 times more crustaceans in seagrass beds as compared with adjacent bare sand areas. Thus, declines in seagrass standing stock should be associated with declines in associated macrofauna, and percent cover of seagrass may be a reasonable index of decline in an element of great trophic importance. Seagrass biomass undergoes clear seasonal variation (Figure 6), and assignment of values to this metric must clearly be adjusted on the basis of season.

SUMMARY

The formulation of an Index of Biotic Integrity modified for coastal systems appears to be feasible in principle. The attraction of this particular approach to evaluation of coastal habitat status is that it is a multivariate approach based on changes in several levels of ecological parameters. As such, it is clear that the initial establishment of scores for each metric for a given habitat would not be a simple task. Establishment of these scores requires a reasonable level of understanding of the ecological system under consideration. In that use of this approach forces a careful ecological examination of the habitat on a regional basis, the benefits of the technique should far outweigh the temptingly simple use of single index values such as H' diversity. Once scoring criteria for metrics are established for a particular regional habitat, application of the IBI as a monitoring tool should be a relatively straightforward and routine task.

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