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Tracking environmental stress gradients using three biotic integrity indices: Advantages of a locally-developed traits-based approach

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ABSTRACT

Estuarine and coastal ecosystems are productive and functionally diverse areas that provide a wide range of societal benefits. Along with human exploitative uses comes an array of anthropogenic disturbances that can affect ecological integrity, including changes to the composition and resilience of benthic macroinvertebrate communities. To understand the responses of ecological communities to anthropogenic disturbance and to manage and mitigate effects, indices for assessing the ecological integrity of estuarine and coastal waters have proliferated worldwide. Using data from 84 intertidal sites in Auckland, New Zealand, we evaluated the suitability of two widely used measures of ecological integrity that were developed in USA and Europe, respectively: the Benthic Index of Biotic Integrity (B-IBI) and the AZTI's Marine Biotic Index (AMBI). We then developed a local index based on macrofaunal traits and verified its utility using independent data from >100 additional sites. The local traits based index (TBI), constructed from the richness of macrofaunal taxa in seven functional groups, responded to changes in sediment mud percentage and heavy metal contaminant concentration gradients below international guidelines. The TBI performed better than the indices developed overseas, probably because they were designed to track organic enrichment and hypoxia, which are not the predominant stressors in New Zealand at present. The TBI successfully tracked the stressors that were the most relevant locally and indicated the relative levels of within-group taxonomic richness at various sites. As within-group richness is a component of functional redundancy and ecological resilience, the TBI offers a trifecta of simplicity, robustness and meaningfulness that will facilitate management.

1. Introduction

Human activities on land and at sea are modifying marine ecosystems globally and, with increasing numbers of people living near the coast, the health and resilience of estuaries and coastal marine areas are under increasing threat (Loreau et al., 2001; Beaumont et al., 2007). With growing concern about the ability of these systems to provide us with valued goods and services in perpetuity, easily understandable yet scientifically defensible indicators of marine ecosystem integrity have been sought (see Díaz et al., 2004 for a review). Some of these tools have been adopted by resource managers and used in relation to environmental policy at the national and international level (e.g., Borja et al., 2008a; Teixeira et al., 2010; van Hoey et al., 2010). Yet questions remain about how well these indicators perform for locations far from where they were developed.

Soft-sediment macroinvertebrates have been used repeatedly to assess the effects of natural and anthropogenic disturbances because they are considered accurate and sensitive indicators of the environmental status (e.g., Pearson and Rosenberg, 1978; Dauer, 1993; Weisberg et al., 1997; Borja et al., 2000). The number of available indices based on macroinvertebrates has risen dramatically in recent years, particularly in Europe and the USA, and several indices are now widely used and publically available as freeware on the worldwide web (e.g., Díaz et al., 2004; Borja et al., 2008a; Pinto et al., 2009).

The proliferation of new indices during the last decade reflects not only societal concerns and urgent legislation demands, but also doubts that one or a few metrics can be meaningful and applicable in all management situations. Nevertheless, it is important to evaluate the suitability of existing indices that have proven useful prior to the development of new ones (Díaz et al., 2004). There have been reviews and comparisons of previously developed indices, generally within or between Europe and USA (e.g., Borja et al.,

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2003, 2008b; Quintino et al., 2006; Teixeira et al., 2010), though their applicability in southern hemisphere locations such as New Zealand has not yet been tested.

Among all the available environmental indicators of biological integrity used worldwide, the AZTI's Marine Biotic Index (AMBI; Borja et al., 2000) and the Benthic Index of Biotic Integrity (B-IBI; Weisberg et al., 1997) are the best known and most applied indices. AMBI is based on the degree of sensitivity of selected species to an environmental stress gradient. Strengths of the index include its free availability on the internet (www.azti.es) and the production of easily interpretable plots by the software in a standard format. However, this index is generally designed to compare marine communities of the same basic type, and thus habitat variation reduces its utility. Conversely, B-IBI stratifies habitats based on benthic assemblage differences, identifies diagnostic metrics and thresholds based on the distribution of values at reference sites, and combines metrics into an index by a process that uses a simple scoring system that weights all measures equally. An advantage of B-IBI over AMBI is that it accounts for habitat variation by using reference sites, although this in turn raises the problem of defining such sites. Both indices are said to be able to be used to assess different types of stressors (Muxika et al., 2005; Borja et al., 2008b).

In the present study we tested the applicability of two consolidated environmental indicators from overseas (AMBI and B-IBI) on data obtained at latitudes and within environments different from those in which they were originally developed. We compare them and discuss their validity for detecting environmental stressors such as elevated sediment mud and heavy metal concentrations, which are recognised as major threats to the health and functioning of New Zealand coasts and estuaries (Hewitt et al., 2005, 2009; Rodil et al., 2011; Lohrer et al., 2012). Because of the relative isolation of New Zealand in the southwest Pacific Ocean and its low human population density, the problems of coastal eutrophication and organic enrichment that affect large swathes of Europe, North America and Asia are limited in New Zealand at present. Thus, indices developed overseas to detect this type of stress were predicted to perform poorly here in New Zealand. We hypothesised that a locally developed index would perform better, and we provide details of a new traits based index (TBI) designed to measure the ecological integrity of invertebrate communities in New Zealand marine ecosystems. More specifically, the TBI provides an assessment that is related to the concept of functional redundancy and ecological resilience (e.g., Naeem, 1998; Walker et al., 1999; Díaz and Cabido, 2001; Petchey and Gaston, 2002a,b), as it is derived from the richness of taxa within functional groups. Greater taxonomic richness within functional groups provides the means by which communities can maintain functioning in the face of stochastic or stress induced losses of one or a few taxa (Walker, 1992; Fonseca and Ganade, 2001; Rosenfeld, 2002; Micheli and Halpern, 2005). To develop the TBI, we quantified variation in the richness of macrofauna taxa in seven functional trait groups that responded to gradients of sediment mud and heavy metals. Following development, the performance of the TBI was validated using independent data collected from >100 estuarine and coastal sites in northern New Zealand.

2. Materials and methods

2.1. Study sites and sampling procedure

Data from 84 intertidal soft-sediment sites from Manukau and Waitemata Harbours near Auckland, New Zealand (Fig. 1a), were used for AMBI and B-IBI scoring and testing and for the development of the TBI. All sampling took place between 2002 and 2006. Eleven of the 84 sites were sampled twice during this period to produce a data set with 95 data points. The dataset contained identifications and average abundances at each site for all macrofaunal taxa present in each location. The dataset also contained sediment particle size and sediment heavy metal contaminant information. The positions of the sites were specifically selected to encompass a gradient of storm water contaminants and a range of sandy and muddy sites; subsequent analyses of sediment metal concentrations confirmed the gradient, and mud and metals were correlated (Supplementary Material, Figure S1a).

Benthic macrofauna were sampled using a 13 cm diameter, 15 cm deep corer, with 10 randomly located replicates collected and analysed per site. Sampling was conducted in October to avoid the peak recruitment period for most taxa and limit the effects of seasonal and inter-annual variability. Samples were sieved across a 0.5 mm mesh screen, preserved in 70% IPA and stained with 0.2% Rose Bengal prior to identification to the lowest taxonomic level practicable. Surface sediment particle size was sampled using a 2 cm diameter, 2 cm deep corer, with 6 replicate cores collected per site and aggregated prior to analysis. Particle size analysis was performed via standard wet sieving methods that have been described elsewhere (Hewitt et al., 2009); "mud content" is defined as the percentage by weight of particles < 63 μm. Sediment heavy metal concentrations were assessed by collecting 30 subsamples per site and aggregating them into 3 composite samples prior to analysis. These samples were analysed for total recoverable Cu, Pb and Zn concentrations in the <500 μm sediment fraction using previously described standard methods (Hewitt et al., 2009).

2.2. Application of consolidated indices: AMBI and B-IBI

The AMBI index is based upon the proportion of species assigned to one of five levels of sensitivity (ecological groups) to increasing levels of disturbance, from very sensitive to opportunistic soft-bottom macrofauna species. Guidelines for interpreting AMBI outputs are given by Borja et al. (2000), where Biotic Index (BI) scores of 0–1 reflect unpolluted sites, 2–4 slightly or moderately polluted sites, and 5–7 heavily or extremely polluted sites. To enable us to apply AMBI in a New Zealand context it was necessary to assign the New Zealand macrobenthic species to one of the five ecological groups (following Borja et al., 2008b).

The B-IBI is calculated by comparing the value of a metric (related to benthic community structure and function; e.g., species diversity, productivity, species and trophic composition) from a sample of unknown quality to thresholds established from reference data distributions (Weisberg et al., 1997). These thresholds called "restoration goals" were established from the 5th or 95th, and 50th percentile values of metrics measured at reference sites. Both sandy (<40% mud) and muddy (>40% mud) sites were used as reference sites, although all of the reference sites were of one salinity class (>18 ppt). Importantly, all of the reference sites had low levels of sediment heavy metal contamination (Cu < 12, Zn < 106 and Pb < 24 mg kg^{-1}). Each metric in the B-IBI calculation is scored on a 5, 3, or 1 scale; individual metric scores are averaged to produce the combined B-IBI score. We used abundance based metrics for B-IBI calculations where biomass data was unavailable (Llansó and Dauer, 2002), thus Shannon–Weiner diversity index, total species abundance, % abundance of pollution sensitive taxa, % abundance of carnivores and omnivores, and % abundance of deep-deposit feeders were used as metrics. According to Weisberg et al. (1997), samples with combined B-IBI scores ≥ 3 are indicative of good habitat quality.