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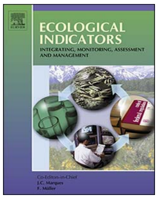

Original Articles

Relationships between biotic indices, multiple stressors and natural variability in New Zealand estuaries

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ABSTRACT

In response to the need to assess the ecological quality or health of marine benthic habitats, there has been a proliferation of biotic indices based on soft sediment macrofaunal communities. While shown to be useful in areas where they have been developed, some indices may not be readily transferrable to other regions due to differences in species ecology or composition, stressor type or magnitude, and natural variability. Using a national New Zealand dataset compiled from estuary monitoring data for 2001–2016, we used linear mixed models to determine the effect of multiple stressors (sediment mud content, metals and total phosphorus) and natural variability (associated with space, estuary type and time) on nine indices developed in New Zealand and overseas. The Richness-Integrated AZTI Marine Biotic Index (RI-AMBI), a modification of a popular overseas biotic index, had the most variation explained by stressors overall (marginal pseudo-$R^2 = 0.22$ compared to ≤ 0.15 for all other indices). This variation was primarily explained by a single stressor, sediment mud content, which is the dominant stressor in New Zealand estuaries. However, although the overall variation explained by stressors was lower for all other indices, multiple, rather than single, stressors had significant effects on some indices. For example all three stressors had a significant effect on the Traits Based Index, and the variation explained by metals was highest for this index. Relatively high amounts of natural and unexplained variation for all indices suggested that further understanding is required before operational implementation of indices at a national scale. Thus, the use of more than one index, i.e. a weight of evidence approach, is suggested to minimise uncertainty related to inaccuracy and misclassification of ecological health in New Zealand estuaries.

1. Introduction

Macrofaunal communities inhabiting marine soft sediments are often used as indicators of ecological quality or health (e.g. Borja et al., 2015). These globally common habitats are the receiving environment for many human impacts, and the sensitivity of benthic communities to human impacts has long been recognised (Pearson and Rosenberg, 1978). Many methods have been developed to assess ecological health based on analysis of multivariate community data. These include the development of biotic indices, which distil multivariate data to a univariate measure that aims to describe ecological health. Simple metrics (e.g. number of species and individuals) have been widely used. However, these have been out-performed as indicators of ecological health by biotic indices, such as those that reflect the sensitivities of different taxa to environmental gradients (Ellis et al., 2015; Simboura and Zenetos, 2002).

There has been a proliferation of biotic indices over the past 20 years and Borja and Dauer (2008) and Diaz et al. (2004) recommended that existing indices be considered before developing new ones. An important feature of an index is its capability to convey information that is meaningful for decision making, including being directly tied to management questions relating to human stressors, over a range of spatial and temporal scales (Cairns et al., 1993; Rees et al., 2008). Index responses can be tested against individual stressors (e.g. Simboura and Zenetos, 2002; Van Hoey et al., 2010). However, with increasing environmental pressures associated with both urban and rural intensification (e.g. nutrient run-off, sedimentation and metals contamination), it is preferable for indices to reflect the impacts of multiple stressors concurrently (Cairns et al., 1993; Van Hoey et al., 2010). Sensitivity to multiple stressors also increases the likelihood that an index will enable managers to identify changes in ecological health due to new and unanticipated perturbations (Cairns et al., 1993).

Most indices have been developed for northern hemisphere conditions, mostly in Europe and USA, and may not be readily transferable to other regions due to differences in species ecology or composition, stressor type or magnitude, and sources of natural variability (Gillett

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et al., 2015; Rodil et al., 2013; Van Hoey et al., 2010). Testing relationships between indices, stressors and natural variability is therefore crucial for assessing the suitability of indices in new regions (Borja et al., 2007; Diaz et al., 2004).

Some biotic indices developed overseas have been tested in New Zealand. For example, the AZTI Marine Biotic Index (AMBI, Borja et al., 2000) and the Benthic Index of Biotic Integrity (Weisberg et al., 1997) were assessed in estuaries of the Auckland region, with both performing poorly due to insensitivity to sediment and metal gradients (Rodil et al., 2013). For AMBI at least, this may be because the European ecogroups (EGs) to which taxa of differing sensitivities were assigned were based on response to enrichment (e.g. Borja et al., 2000), rather than on local stressor gradients (Rodil et al., 2013). Overseas studies have demonstrated that regionally-specific EGs can provide more accurate results due to taxon-specific responses to stressors (e.g. Gillett et al., 2015).

In New Zealand, Keeley et al. (2012a) developed regionally-specific EGs for subtidal macrofaunal taxa based on organic enrichment, and demonstrated that this increased the ability of EG-based indices to respond to organic enrichment gradients related to aquaculture farms (Keeley et al., 2012b). Local EGs for estuaries based on the response to mud have been developed by Robertson et al. (2015), as sediment mud content is a key estuarine stressor in New Zealand (Norkko et al., 2002; Robertson et al., 2016; Thrush et al., 2004). Robertson et al. (2016) subsequently developed a modified version of the AMBI (hereafter RI-AMBI) that incorporates proportional taxon richness in addition to proportional abundance of EGs used in the original AMBI. They found that using a combination of New Zealand specific and internationally defined EGs for index calculations improved the relationship with stressor gradients and furthermore that the RI-AMBI outperformed AMBI and another variation of this index, the Multivariate-AMBI (M-AMBI), which incorporates richness and diversity metrics.

As well as these EG-based approaches, an estuarine index was developed in New Zealand using a functional based approach. The Traits Based Index (TBI, Rodil et al., 2013) was developed to respond to sediment mud content and metal contamination gradients in the Auckland region, and is calculated using the richness of macrofaunal taxa in seven functional groups.

Despite the local refinement and development of biotic indices in New Zealand, there has been no comprehensive and consistent nationwide testing of the response of a range of indices (i.e. based on different approaches) to multiple stressors and natural variability. We have therefore undertaken such an assessment as a step toward selection of indices that provide accurate measures of ecological health in New Zealand estuaries and to test their comparability across wider scales. Our aim was to compile and then use a national dataset, collated from data collected using a standardised estuary monitoring protocol (Robertson et al., 2002), to assess the relationship between various biotic indices, developed both in New Zealand and overseas, multiple stressors, and natural variability (i.e. bioregion, estuary, estuary type and year).

2. Materials and methods

2.1. Macrofaunal and physico-chemical dataset

Data were obtained from intertidal estuarine surveys undertaken by New Zealand's regional government authorities during two seasonal periods (October − December and January − April) between 2001 and 2016. Surveys were conducted following a standardised estuarine monitoring protocol (Robertson et al., 2002) at unvegetated sites located at mid–low tidal height. Sites were positioned away from immediate point source discharges in order to capture overall cumulative stressor effects.

Macrofaunal samples were collected using a cylindrical core, 150 mm deep, and either 130 mm (82% of samples) or 150 mm (18% of samples) in diameter, and sieved through a 0.5 mm mesh. All individuals were identified to the lowest taxonomic level practicable by experts throughout the country. Taxonomic nomenclature followed the World Register of Marine Species (WoRMS Editorial Board, 2017). Where there were taxonomic uncertainties, we aggregated to higher groups. Taxa belonging to Plantae, Vertebrata, Bryozoa, Cirripedia, Insecta, Acari, and those identified to relatively coarse taxonomic groups (e.g. Gastropoda, Polychaeta, Annelida, Bivalvia, Decapoda and Brachyura), were removed from the dataset as recommended by Borja and Muxika (2005). Macrofaunal abundance was standardised to a 130 mm diameter core (i.e. results from samples taken with 150 mm cores were scaled down). However, initial exploratory analyses indicated no significant differences in species richness between the two core diameters ($p = 0.6$, results not shown). Thus, this potential source of bias was considered negligible and sampling events using cores of either diameter (130 mm core data and scaled-down 150 mm data combined) were included in order to maximise data availability for the development of subsequent models.

Physico-chemical sediment samples were collected at each site ($n = 1 - 12$) concurrently with the macrofaunal sampling. The large range in replicate numbers was due to compositing of samples prior to laboratory analyses in some surveys, resulting in a lower number of replicate samples than originally collected, as well as differences in sampling effort in some cases. Measured variables represented common stressors in New Zealand that are natural to some extent but are exacerbated by human-induced pressures affecting estuaries, e.g. sedimentation, eutrophication and contamination (Robertson et al., 2002, Robertson et al., 2015; Thrush et al., 2004; Edgar and Barrett, 2000). The stressors included in analyses were: mud (grain size < 63 μm), nutrients (total phosphorus), and the metals copper (Cu), zinc (Zn) and lead (Pb). These stressor variables were chosen based on data availability and quality, as well as ecological relevance.

Although estuaries both overseas and in New Zealand are often limited by total nitrogen (TN) rather than total phosphorus (TP) (Howarth and Marino, 2006; Robertson et al., 2002), poor data quality for TN led us to choose TP to represent nutrients in our statistical models. TP was moderately correlated with TN values above analytical detection limit (Pearson $r = 0.68$), and more strongly correlated with measures of organic content (Ash Free Dry Weight $r = 0.71$ and Total Organic Carbon $r = 0.95$). We therefore considered TP to be a relatively good proxy for catchment-level nutrient and organic enrichment. Although there was some variation in laboratory analysis methods, particularly for grain size, no significant differences in the relationships between biotic indices and stressors were detected, providing confidence that sample processing methods were not biasing our results.

Sites were assigned to seven wider biogeographical regions based generally on the coastal physical habitats and biological communities characterised by Shears et al. (2008). Because of limited data availability, three bioregions were combined with others, resulting in four bioregions overall: 'Northern' and 'Eastern' (North Island), 'Cook Strait' (North and South Island), and 'East South' (South Island) (Fig. 1). Additionally, each site was assigned to one of two Geomorphic Classes (hereafter 'estuary type'); GC 7, tidal lagoon, or GC 8, shallow drowned valley. These estuary types were based on landscape and waterscape characteristics (e.g. geology, basin morphometry), as well as hydrodynamic features due to river and ocean forcing of the estuary (Hume et al., 2016).

2.2. Index selection and calculation

Nine biotic indices and metrics were selected for testing. Some were identified in a recent international review (Borja et al., 2015), while others were developed, tested or modified in New Zealand (Keeley