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Optimising a widely-used coastal health index through quantitative ecological group classifications and associated thresholds

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ABSTRACT

Many globally applied biotic indices, including the AMBI benthic index, are based on species' sensitivity/tolerance to anthropogenic disturbances. The AMBI scoring primarily relies on the correct assignment of both taxon stressor-sensitivities and the disturbance thresholds or bands. Using an extensive, long-term monitoring dataset from New Zealand (NZ) estuaries, we describe how the AMBI has been strengthened through quantitative derivation of taxon-specific sensitivities and condition thresholds for two key estuarine stressors [mud and total organic carbon (TOC)], and the integration of taxon richness. The results support the use of the existing AMBI condition bands but improve the ability to identify cause; 2–30% mud reflected a 'normal' to 'impoverished' macrofaunal community; 30–95% mud and 1.2–3% TOC 'unbalanced' to 'transitional'; and >3–4% TOC 'transitional' to 'polluted'. The (refined) AMBI was also successfully validated (R^2 values >0.5 for mud, and >0.4 for TOC) for use in shallow, intertidal dominated estuaries NZ-wide. Most biotic indices lack the ability to differentiate between anthropogenic disturbances, which in turn undermine their effectiveness for applied purposes. By integrating key quantitative information to an existing benthic index, these results enable more robust identification of coastal stressors and facilitate defensible management decisions.

1. Introduction

Determination of the benthic condition of shallow coastal ecosystems focuses on monitoring both biotic and abiotic sediment quality indicators (e.g. the Australian Oceans Policy, the Canadian Oceans Act and Oceans Strategy, the USA Oceans Act, the European Water and Marine Strategy Framework Directives (WFD, 2000/60/EC and MSFD, 2008/56/EC), and the New Zealand Estuary Monitoring Protocol (EMP, 2002) and Estuarine Trophic Index (Robertson et al., 2016a,b)). In particular, indicators have been developed to reflect environmental degradation associated with increased sediment mud content (Robertson et al., 2015), organic enrichment (Hyland et al., 2005; Pusceddu et al., 2009; Sutula et al., 2014; Robertson et al., 2015), and toxicity (Brady et al., 2015). Macroinfaunal communities are generally selected as the primary biotic indicator due to their functional importance, their diversity of responses and their relatively sedentary existence. To facilitate the interpretation of macroinfaunal abundance data as it relates to environmental variables, 'sensitivity' groupings have been developed for many taxa, either quantitatively (Robertson et al., 2015) or through expert opinion (e.g. Gillett et al., 2015). Comparing the relative magnitudes of each of the taxon-specific sensitivity groupings at a particular site provides an indication of where the macroinfaunal community fits along the environmental gradient(s).

The most widely used coastal biotic index, the AZTI (AZTI-Tecnalia Marine Research Division, Spain) Marine Benthic Index (AMBI), has been verified in relation to a range of abiotic variables (Borja et al., 2000), environmental impact sources (Borja and Muxika, 2005) and regions, including Europe, the United States (Borja et al., 2008; Borja and Tunberg, 2011; Teixeira et al., 2012), South America (Muniz et al., 2005), and Canada (Callier et al., 2008). The AMBI biotic coefficient (BC) is generated by

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combining weighted abundances of each of five sensitivity groupings (called ecological groups – EGs) to anthropogenic disturbance, ranging from very sensitive to very insensitive, and is then used to categorise a particular site into one of seven 'disturbance bands' (Normal to Azoic, derived by comparing the macroinfaunal response to a range of environmental variables). Hence, the two key drivers of the AMBI BC scoring approach are the correct assignment of each taxon to an EG, and the disturbance thresholds or bands used.

While such an approach clearly provides an easy-to-use, cost effective tool for assessing the condition of benthic coastal habitat, in some cases the performance of the AMBI has been limited. For example, it can perform unsatisfactorily where samples have a low number of taxa with assigned EG values (Muxika et al., 2007; Gillett et al., 2015), or where sensitivity groupings are based predominantly on the international AMBI list (http://ambi.azti.es) due to an absence of local sensitivity data (e.g. Rodil et al., 2013; Gillett et al., 2015). Local sensitivity EG data, derived through expert opinion, significantly improves the performance of AMBI when augmented with international list values (Gillett et al., 2015). AMBI performance, particularly in its role in managing coastal benthic pollution in estuaries, is also limited by its inability to differentiate between various key anthropogenic disturbance stressors such as muddiness, organic matter, oxygen conditions and toxicants, and natural disturbance such as low salinity (Baritone et al., 2012)).

The present study addressed four objectives to improve the efficacy of the AMBI in New Zealand (NZ) estuaries: (1) to determine improvements in the AMBI from the inclusion of quantitative EGs derived from NZ macrobenthic data; (2) to validate the AMBI using an independent, nationwide dataset; (3) to derive thresholds of two primary stressors, sediment muddiness and organic enrichment, to better inform the current AMBI condition bands for use in NZ estuaries; and (4) to determine the usefulness of adding species richness to the AMBI, either as M-AMBI (AMBI's multivariate extension; Bald et al., 2005; Muxika et al., 2007), or through direct integration to the abundance-weighted AMBI equation.

2. Materials and methods

2.1. Study locations and sampling protocol

An extensive nation-wide estuary monitoring dataset collected over 15 years by NZ regional government authorities was used in Robertson et al. (2015) to model and derive taxon-specific EGs. Here we used data from 21 of the 25 estuaries (four tidal river estuaries with minimal intertidal regions were omitted during dataset standardisation as outlined below). The present study assesses whether the inclusion of local EGs improved the efficacy of the AMBI and facilitates establishing threshold values along primary stressor gradients. Benthic datasets were standardised to minimise variance in index values by: selecting moderate-high salinity zones (>25 psu) in representative mid-low water intertidal habitat with low sediment metal concentrations (apart from rare situations where metal concentrations are naturally high due to geological activity); targeting predominantly shallow, relatively large tidal lagoon systems, dominated by intertidal habitat and perpetually open to tidal exchange (see Fig. 1 for locations of estuaries, Table 1 for relevant attributes), a type that constitutes >150 of New Zealand's 400+ estuaries (NIWA's Coastal Explorer Tool available at: http://www.niwa.co.nz/coasts-and-oceans/nz-coast/coastal-explorer).

Variation among sites or estuaries due to different sampling or identification techniques is considered negligible since sampling was standardised in accordance with the NZ National EMP (Robertson et al., 2002) and one expert undertook the majority of the macroinvertebrate taxonomic work. Details of the sampling protocol, including the timing, replication and number of sampling events per estuary, and the biotic and abiotic parameters measured, are presented in Robertson et al. (2015). Briefly, benthic macrofauna were sampled using a 130 mm diameter (area = 0.0133 m^2) core manually driven 150 mm into the sediment, with 10–12 randomly located replicates collected and analysed per location. Samples were sieved on a 0.5 mm mesh and retained fauna were preserved in 95% isopropyl alcohol/seawater solution. Macrofauna were identified to the lowest possible taxonomic resolution and counted.

A sediment sample (approx. 250 g from the top 20 mm at the surface) was collected for analysis from each macrofaunal sampling location and analysed for: (1) grain size distribution (% mud, sand, gravel) using wet sieving and gravimetric calculations; (2) TOC via elemental analyser (628 Series CNS, Leco); and (3) metal contaminants (total recoverable Cd, Cr, Cu, Ni, Pb and Zn) using nitric/hydrochloric acid digestion, ICP-MS (low level) USEPA 200.2 (US EPA, 2009) – see Robertson et al. (2015) for estuary-specific metal concentrations.

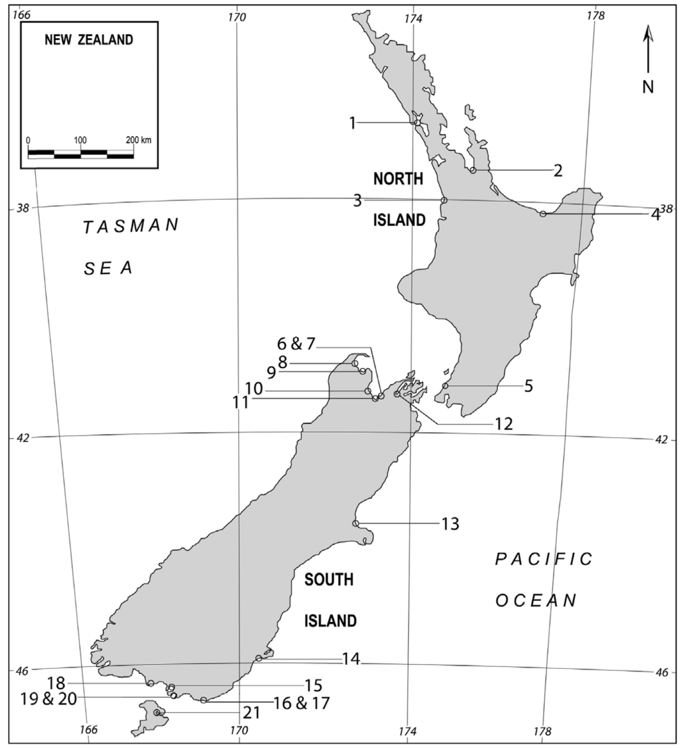

Fig. 1. Geographic locations of the 21 estuaries sampled in New Zealand used to calibrate and validate the AMBI benthic index. Refer to Table 1 for location and physical details relating to each estuary. Figure is modified from Robertson et al. (2015).

2.2. AMBI under three classification schemes

Of all the available benthic indices (reviewed by Borja et al., 2012), this study focused on the applicability of the AMBI biotic index for two reasons. First, the AMBI was designed to be responsive to a number of anthropogenic disturbance variables, including mud and organic enrichment, which are particularly important primary stressors of macrofaunal communities in shallow, intertidal, NZ estuaries (Robertson et al., 2015). Second, Robertson et al. (2015) used organic enrichment, grain size and macroinvertebrate abundance data to assign EGs to 99 (quantitatively for 39 and semi-quantitatively for 60) taxa according to their responses to mud content and organic enrichment. These NZEGs – labelled I, II, III, IV or V, with V assigned to the most tolerant taxa and I to the