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Response of multi-metric indices to anthropogenic pressures in distinct marine habitats: The need for recalibration to allow wider applicability

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ABSTRACT

Sustainable exploitation of coastal ecosystems is facilitated by tools which allow reliable assessment of their response to anthropogenic pressures. The Infaunal Quality Index (IQI) and Multivariate-AMBI (M-AMBI) were developed to classify the ecological status (ES) of benthos for the Water Framework Directive (WFD). The indices respond reliably to the impacts of organic enrichment in muddy sand habitats, but their applicability across a range of pressures and habitats is less well understood. The ability of the indices to predict changes in response to pressures in three distinct habitats, intertidal muddy sand, maerl and inshore gravel, was tested using pre-existing datasets. Both responded following the same patterns of variation as previously reported. The IQI was more conservative when responding to environmental conditions so may have greater predictive value in dynamic habitats to provide an early-warning system to managers'. Re-calibration of reference conditions is necessary to reliably reflect ES in different habitats.

1. Introduction

Impacts caused by anthropogenic activities are a global concern in marine ecosystems (Borja et al., 2012; Halpern et al., 2009, 2012, 2008). Estuarine and coastal waters are particularly vulnerable to a variety of pressures due to their proximity to land and the subsequently high levels of human activity they experience (see O'Gorman et al., 2012). Estuarine and coastal habitats provide some of the most productive and highly biologically diverse ecosystems and as such have a high social, economic and ecological value (see Barbier et al., 2011; Beaumont et al., 2007). The high intensity of pressure experienced by coastal ecosystems highlights the need to develop methods to ensure their sustainable exploitation now and into the future.

Legislation has been adopted in an attempt to encourage sustainable uses of ecosystems (e.g. Convention on Biological Diversity, Water Framework Directive (WFD) and the Marine Strategy Framework Directive (MSFD)) (Borja et al., 2012, 2009a; Fitch and Crowe, 2010, 2012). The characterisation of ecological responses to anthropogenic pressures, and their subsequent impacts, aims to allow managers and regulators to make informed decisions based on the ecology of an area (Karakassis et al., 2013). The development of indices to distil large sets of ecological information into a single number which reflects the ecological integrity of an area is an ongoing requirement of managers and policy makers (e.g. Neto et al., 2013).

The WFD is the primary legislative framework for the protection of estuarine and coastal waters in EU member states (Van Hoey et al., 2010). The WFD requires the characterisation of biological quality elements (BQE's) to determine the ecological status of a water body. BQE's used in the characterisation of ecological status are angiosperms, macroalgae, phytoplankton, fish and benthic invertebrates (Best et al., 2007a, 2007b; Coates et al., 2007; Desrosiers et al., 2013). Benthic invertebrates are one of the BQE's considered under the WFD for which there is a large body of background information and research (e.g. Borja et al., 2000; Dauvin and Ruellet, 2007; Reiss and Kroncke, 2005; Simbora and Zentos, 2002; Teixeira et al., 2008). Benthic invertebrates are considered to be good indicators of disturbance as they are relatively sedentary, long-lived and incorporate changes in the physico-chemical environment rapidly (Fitch and Crowe, 2011; Garcia-Marin et al., 2013; Smith and Rule, 2001). There have been a number of benthic

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invertebrate indices developed in response to the WFD (Borja et al., 2011a; Kroncke and Reiss, 2010; Reiss and Kroncke, 2005), many of which are based on the characteristic response of benthos to gradients of organic matter described by Pearson and Rosenberg (1978). Biotic indices are increasingly being used in the assessment of environmental quality, but the performance of such indices and their comparability across different environments requires further investigation (Keeley et al., 2013).

The Multivariate-Azti Marine Biotic Index (M-AMBI) and the Infaunal Quality Index (IQI) are based on AMBI, which is the most widely used biotic index in the implementation of the WFD. AMBI analyses the proportion of taxa assigned to each of 5 ecological groups (Sensitive to Opportunistic, see www.azti.es) (Borja et al., 2000, 2009b; Muxika et al., 2007). M-AMBI also incorporates measures of richness and Shannon diversity, it has been shown to be effective in coastal waters and has been tested in estuarine and intertidal waters (Borja et al., 2009b; Muxika et al., 2007). Reference conditions for different environmental conditions are determined for discreet habitats for M-AMBI. Whilst this approach has been shown to be effective for the number of habitats tested it does not take account of within-habitat variability. AMBI and M-AMBI have been shown to respond to a variety of anthropogenic pressures including hypoxia, eutrophication, oil platform discharges, dredging and fish aquaculture (Borja et al., 2003, 2006, 2009b, Muxika et al., 2007; Callier et al., 2009), and have been globally validated in different habitats and inter-calibrated with other methodologies e.g. (Borja et al., 2011b; Borja et al., 2007, 2008; Ruellet and Dauvin, 2007; Blanchet et al., 2008; Bouchet & Sauriau 2008).

The IQI is the index developed for the assessment of ecological status in coastal and transitional waters in the UK and Ireland (Phillips et al., 2014). The development of the IQI and associated reference conditions was heavily dependent upon the quality of the data available and the range of environmental conditions (habitats) over which the data were collected (Phillips et al., 2014). The IQI was developed for use in transitional and coastal waters (up to 3 km offshore). It can be used to calculate the ecological status of benthic invertebrates for WFD with a high degree of confidence in muddy and sandy habitats but has not yet been shown to be reliable in course sands and gravel or intertidal and inshore waters (inshore waters are between 1 and 12 km offshore) (Phillips et al., 2014). Reference conditions for the IQI are determined on a sliding scale for the environmental factors salinity and sediment type (Phillips et al., 2014). Whilst this does, to some degree, account for within-habitat variability, the number of samples the reference conditions are based on is low for some sedimentary habitats (see Supplementary material Table S1), and therefore interpretation of EQR's should be undertaken with caution. Despite the narrow range of pressures and habitats the IQI was developed in and tested against, there is evidence that it may be able to predict ecological status in a number of habitats and against a range of pressures (Borja et al., 2011a; Fitch and Crowe, 2010; Kennedy et al., 2011; Van Hoey et al., 2010). The calculation of the IQI against known impacts in previously untested habitats provides an opportunity to validate the tool against a wider range of pressures in a variety of habitats, such as intertidal and coarse sediments.

The aim of this study was to validate the ability of the M-AMBI and the IQI to detect changes in benthic invertebrate assemblages in response to anthropogenic pressures in different habitats. To achieve this they were tested using data comparing control and putatively impacted conditions in three distinct habitats: (i) intertidal muddy sand subject to organic matter and inorganic nutrient inputs; (ii) maerl in highly tidal coastal waters subject to pressures associated with fin fish aquaculture and (iii) inshore sand and gravel subject to aggregate extraction.

2. Methods

This study utilised data from impact studies in three distinct habitats which were subject to different anthropogenic pressures. The raw data used to calculate M-AMBI and the IQI have previously been used to characterise the response of benthic invertebrates to: (i) the individual and combined effects of organic matter in intertidal muddy sand sediments (Fitch and Crowe, 2012; O'Gorman et al., 2012); (ii) fin fish aquaculture in maerl in highly tidal coastal waters (Hall-Spencer and Bamber, 2007; Hall-Spencer et al., 2006) and (iii) aggregate dredging in inshore gravel sediments (Cooper et al., 2007, 2008a).

2.1. Data collection

2.1.1. Intertidal muddy sand

The individual and combined effects of inorganic nutrients and organic matter were tested during a field experiment. The experimental site was a sheltered intertidal soft sediment bay (Finavarra) in Galway Bay (Ireland) (Fitch and Crowe, 2010). The experiment was setup along the midshore where sediments were characterised by muddy sands. Three levels of nutrient addition (0, 100 and 200 g m^{-2} per dose, or ambient (=N), medium (+N) and high (++N) respectively) were crossed with 2 levels of organic matter (0 and 200 g m^{-2} per dose, or ambient (=OM) and addition (+OM) respectively). The experiment was setup in June 2007. Two sampling times were planned (after 3 and 11 months), and for each sampling time 4 replicate plots were randomly assigned to each treatment prior to the start of the experiment. Plots (50 $\times$ 50 cm) were marked at least 5 m apart to ensure independence, and treatments and sampling times were randomly assigned. Samples were taken using a 10 cm diameter by 20 cm deep cylindrical core. Full details of the experimental setup are available in Fitch and Crowe (2012) and O'Gorman et al. (2012).

2.1.2. Maerl in strongly tidal coastal waters

The impact of aquaculture cages on benthic invertebrate communities inhabiting maerl beds was tested. To obtain a geographic spread, two farms that were approximately 350 km apart and located over shallow sublittoral maerl beds in Shetland (North Sandwick) and Orkney (Puldrite Bay) were sampled. Diving surveys were carried out between 24 May and 29 June 2003 when these farms at Shetland and Orkney were permitted to stock 995 t and 980 t of salmon, respectively.

At each farm, four weighted transect lines were laid out on the sea bed at right angles from cage edges to locate four sites with stations at 0, 25 and 50 m, and two sites with stations also at 75 and 100 m from the cages. Near each farm, pairs of shallow sublittoral reference maerl beds were surveyed at sites 500–1000 m distant from any known anthropogenic sources of organic enrichment.

In Shetland and Orkney, divers took five samples from reference sites and five from each of the transect line sampling stations around cages, using cylindrical capped cores (0.01 m^2) inserted to a sediment depth of 20 cm. For full details of the sampling programme and aquaculture regime see Hall-Spencer et al. (2006).

2.1.3. Gravel in inshore waters

The impact of aggregate dredging on benthic invertebrate communities inhabiting gravelly sediment was tested. Two areas of seabed previously subjected to relatively high (H) and low (L) levels of dredging intensity were identified on the Hastings Shingle Bank. Two reference (i.e., undredged – R) areas were also selected for comparative purposes. Benthic invertebrates from all four sites were monitored annually over the period 2001–2004, using a 0.1 m^2 Hamon grab.