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Assessing ecological community health in coastal estuarine systems impacted by multiple stressors

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ABSTRACT

Increasing population pressure, urbanisation of the coastal zone and nutrient and sediment run-off from agriculture and forestry has increased the number of large-scale and chronic impacts affecting coastal and estuarine systems. The need to assess cumulative impacts is a major motivation for the current desire of managers and ecologists to define ecosystem "health" and "stress". A number of univariate metrics have been proposed to monitor health, including indicator species, indicator ratios and diversity or contaminant metrics. Alternatively, multivariate methods can be used to test for changes in community structure due to stress. In this study we developed multivariate models using statistical ordination techniques to identify key stressors affecting the 'health' of estuarine macrofaunal communities. Macrofaunal and associated environmental samples were collected across 75 sites from within Tauranga Harbour, a large estuary located on New Zealand's North Island. The harbour receives discharges from urbanised, industrial, agricultural and horticultural catchments. Distance-based linear modelling identified sediments, nutrients and heavy metals as key 'stressors' affecting the ecology of the harbour. Therefore, three multivariate models were developed based on the variability in community composition using canonical analysis of principal coordinates (CAP). The multivariate models were found to be more sensitive to changing environmental health than simple univariate measures (abundance, species richness, evenness and Shannon–Wiener diversity) along an anthropogenic stress gradient. This multivariate approach can be used as a management or monitoring tool where sites are repeatedly sampled over time and tracked to determine whether the communities are moving towards a more healthy or unhealthy state. Ultimately, such statistical models provide a tool to forecast the distribution and abundance of species associated with habitat change and should enable long-term degradative change from multiple disturbances to be assessed.

1. Introduction

The need to better understand the interactive and cumulative effects of multiple stressors is cited as one of the most pressing questions in ecology and conservation (e.g. Crain et al., 2008; Sala et al., 2000). With growing human population comes an increase in the diversity and intensity of anthropogenic stressors (Halpern et al., 2007) including habitat loss, over-exploitation of key species (Jackson et al., 2001) pollution (particularly excess nitrogen), invasive species and more recently climate change (Kappel, 2005; Sala et al., 2000; Venter et al., 2006; Wilcove et al., 1998).

In New Zealand increasing population pressure, urbanisation of the coastal zone and nutrient and sediment run-off from agriculture and forestry has increased the number of large-scale and chronic impacts affecting coastal and estuarine systems. Potentially, this can lead to broader-scale changes to coastal and estuarine systems due to modification of habitats (Saiz-Salinas and Urkiaga-Alberdi, 1999; Smith and Kukert, 1996) and by influencing the health, abundance and distribution of benthic organisms. For example, elevated sediment loading to estuaries and coastal environments can lead to broad scale changes in ecology through modifying habitats (e.g., Saiz-Salinas and Urkiaga-Alberdi, 1999; Smith and Kukert, 1996) and, in particular, by influencing the health, abundance and distribution of benthic suspension feeders (Ellis et al., 2002). Benthic macroinvertebrates are frequently used as indicators of ecosystem health (Borja et al., 2000) because they provide and influence many critical ecosystem services including provisioning services such as food for birds, fish and humans and regulating and maintenance services such as primary and secondary production and the storage and cycling of nutrients (Thrush et al., 2013). Increased concentrations of silts and clay in suspension may significantly increase pseudofaeces production, decrease the amount

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of algal food actually ingested, and may also damage bivalve gills (Bricelj and Malouf, 1984; Iglesias et al., 1996; Morse et al., 1982; Navarro and Widdows, 1997; Robinson et al., 1984; Stevens, 1987; Willows, 1992). Exposure to increased concentrations of suspended sediments for an extended time can, therefore, result in decreased amounts of energy available for growth and reproduction, and have deleterious effects on local populations. Ultimately this can result in functional changes including a loss of key suspension feeding species and a switch to deposit feeding communities. Low levels of nutrient enrichment in estuarine and coastal environments can have a positive effect on the benthos due to improved primary productivity, and therefore food availability. Beyond a critical point, however, excessive nutrient discharges can lead to accelerated eutrophication of coastal environments and adverse symptoms of over enrichment (Cloern, 2001; McGlathery et al., 2007). Metals can be essential for organisms as trace elements, however, at higher concentrations they can become toxic (ANZECC, 2000a). High exposure to heavy metals can cause physiological stress, reduced reproductive success, and outright mortality in associated invertebrates and fishes (Fleeger et al., 2003; Gagnaire et al., 2004; Nicholson, 1999; Peters et al., 1997; Radford et al., 2000). Estuaries and coastal ecosystems are particularly vulnerable to these stressors as they act as natural retention systems for sediments and heavy metal contaminants and are affected by nutrient run-off. The need to assess cumulative impacts is a major reason for the current desire of managers and ecologists to define ecosystem “health” and “stress”.

Over the past decades, there has been a rapid development of indices of “ecosystem health” to assess the status of marine environments which are increasingly required under regulations like the US Clean Water Act and the EU Water and Marine strategy framework directives (de Juan et al., 2015). A number of univariate metrics have been proposed to monitor health including indicator species, indicator ratios and diversity or contaminant metrics (Borja et al., 2000; Gray and Mirza, 1979; Labrune et al., 2012; Margalef, 1958; Pielou, 1966; Rosenberg et al., 2004; Sanders, 1968; Shannon, 1948). In reviewing existing methods of defining and measuring ecological ‘health’ it was noted that many of the existing biological diversity indices do not differentiate amongst different types of taxa and are strongly affected by sample size (Dunn, 1994; Gappa et al., 1990). This limits their ability to detect changes in composition across different communities and habitats. Furthermore, it is not immediately apparent what differences or similarities in these indices actually mean to ecological functioning, as a similar diversity value can be obtained from communities with very different species (Clarke, 1993; Dufrene and Legendre, 1997).

Alternatively, multivariate models that focus on community composition have been developed (see Anderson, 2008; Anderson and Willis, 2003; Hewitt et al., 2005). Community composition comprises both the number and type of taxa (or animals) that make up a biological community at a site, together with their relative abundances or biomasses. Defining community composition requires the same information needed to generate many univariate biological diversity indices; however, by preserving all the information on the abundance of specific taxa, a more sensitive, and more ecologically meaningful, response could be expected (Attayde and Bozelli, 1998; Gray, 2000; Hewitt et al., 2005, 2009; Pohle et al., 2001). The community composition found in areas largely unaffected by anthropogenic disturbances versus that found in more ‘impacted’ areas can be used as a benchmark against which to assess the relative health of community composition found at specific sites. Thus, relative ‘health’ can be defined in terms of the range of communities present in comparable locations that are not considered to be affected by anthropogenically-derived inputs and should serve to identify both acute effects and broader-scale chronic degradation.

Community composition is generally determined using multivariate techniques, including ordination. Multivariate techniques have been applied successfully to indicate the effects of pollution (Ellis et al., 2000; Olsgard and Gray, 1995; Warwick et al., 1990) and subsequent studies have now shown that multivariate methods are better at determining differences between communities with different degrees of contaminant disturbance than univariate measures (Hewitt et al., 2005). In the present study we examined benthic communities in a large harbour, which receives discharges from urbanised, industrial, agricultural and horticultural catchments. Therefore, we were interested in investigating whether multivariate methods would be useful to rank the health of intertidal sites based on predicted responses to multiple stressors including sedimentation, nutrients and contamination. The results of the ordination models were compared with some frequently used diversity indices.

2. Methods

2.1. Study site

Tauranga Harbour is a large estuary (approximately 200 km^2) located on the western edge of the Bay of Plenty on New Zealand's North Island. The harbour is predominantly shallow (<10 m deep), with intertidal flats comprising approximately 66% of the total area (Inglis et al., 2008). Catchment land use is predominantly pasture and indigenous forest with urbanisation concentrated in the south-east, near the city of Tauranga.

Sampling was carried out from the December 2011 to February 2012. A total of 75 sites across the harbour were sampled for benthic macrofauna and associated sediment characteristics. Sites were chosen to reflect a range of habitats, including intertidal sand flats, shellfish beds, seagrass meadows and areas likely to be impacted by pesticides. Due to the large area sampled within a harbour, gradients in salinity and exposure are expected to be present with distance from river mouths. All sites were intertidal, therefore, depth and temperature were relatively constant across all study sites. Salinity was measured and did not vary much over the sites. Wave exposure was not measured directly, however, the coarse fraction of the sediment grain size was used as a surrogate in initial investigations. The results confirmed those of Hewitt et al. (2005) that wave exposure did not confound identification of effects of pollution on community composition. At each site, a 2 × 5 grid of ten plots (each 10 m × 10 m) was marked out and one replicate collected from each plot, yielding 750 samples overall.

2.2. Physico-chemical variables

At each site, one 2 cm diameter core extending 2 cm deep into the sediment was collected from each of the 10 plots in the grid yielding 10 replicates for each site. The replicates were composited into a single sample and the sediment was analysed for grain size, organic matter (loss on ignition, LOI), nutrients (total nitrogen, TN; total phosphorous, TP), heavy metals (lead, Pb; zinc, Zn; copper, Cu) and chlorophyll-α (chl-α).

Sediment grain size was determined by wet sieving and calculation of dry weight percentage fractions. Grain size fractions were gravel (>2 mm), very coarse sand (<2 mm and >1 mm), coarse sand (<1 mm and >500 μm), medium sand (<500 μm and >250 μm), fine sand (<250 μm and >125 μm), very fine sand (<125 μm and >63 μm) and mud (<63 μm). Loss on ignition was measured by dry sediment loss after combustion at 550 °C (American Public Health Association 21st Edn 2540 D + E (Mod); APHA, 2005).

Total nitrogen was measured through the analysis and addition of total Kjeldahl nitrogen (TKN) and total oxidized nitrogen (TON). TKN was determined, after digestion with sulphuric acid using copper sulphate as a catalyst, using a Flow Injection Analyser (APHA 21st Edn 4500N C; APHA, 2005). TON was extracted from the sediment sample using potassium chloride and also analysed by a Flow Injection Analyser (APHA 21st Edn 4500N C; APHA, 2005).

Sediment samples for total phosphorous and metals were dried at 30 °C then digested using a combination of nitric and hydrochloric acids, with heating to 95 °C for 30 min. Analysis for individual metal