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Validation and limitations of a cumulative impact model for an estuary

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ABSTRACT

Comprehensive marine management approaches, such as ecosystem based management and marine spatial planning, would benefit from quantitative, spatially explicit estimates of the cumulative impact of human activities on marine ecosystems. In this study, a method to map and quantify cumulative impact was applied to estimate the combined impact of multiple stressors on Tauranga Harbour, a large estuary in New Zealand. The impact of eight stressors on seven ecosystems was assessed at a 100 m resolution, using New Zealand-specific expert judgement on the vulnerability of different ecosystems to each stressor. Estimated cumulative impact tended to be highest in the southern basin and inner estuaries, corresponding with sensitive ecosystems and multiple stressors and reflecting what is known about the distribution of pressures in Tauranga Harbour. Using benthic community data as an independent estimate of ecological condition, we had the novel opportunity to validate cumulative impact predicted from the model. Only a weak relationship was found between the estimated cumulative impact and measured ecological condition and several reasons for this are considered. Substitution of a more realistic sediment layer improved model outputs, highlighting the importance of accurate input data, particularly for stressors or ecosystems with high impact weights. Different standardisation methods did not greatly affect the spatial distribution of cumulative impact patterns in the harbour. The study highlights some fundamental issues for consideration when using this cumulative impact mapping approach, such as the importance of involving the expert panel throughout the course of the study and the availability and quality of the data used to construct the model.

1. Introduction

Coastal ecosystems are among the most intensively used and most threatened natural systems on Earth (Lotze et al., 2006; Worm et al., 2006). Coastal areas have an average population density three times the global average (Small and Nicholls, 2003). Stressors and anthropogenic activities impacting the marine environment are numerous and include sediment (Lohrer et al., 2006; Norkko et al., 2006), nutrient (Pearson and Rosenberg, 1978) and pollutant inputs (Derraik, 2002; Bolton et al., 2004), overfishing (Jackson et al., 2001; Pauly et al., 2005), climate change (Brierley and Kingsford, 2009; Hoegh-Guldberg and Bruno, 2010; Doney et al., 2012), species invasion (Carlton and Geller, 1993; Grosholz, 2002), oil drilling (Ellis et al., 2012) and destructive fishing (Watling and Norse, 1998; Collie et al., 2000; Thrush and Dayton, 2002).

The number and variety of anthropogenic stressors acting on marine ecosystems can make it difficult for decision makers to account for all stressors and their interactions. Additionally, both human activities and marine habitats vary in their spatial distribution and an understanding of where these activities occur is necessary to evaluate trade-offs (or compatibility) between human uses of the oceans and protection of ecosystems and the services they provide. Managing stressors in isolation is insufficient to protect marine ecosystems as co-occurring human activities create multiple impacts on communities (Halpern et al., 2008a). The shift towards more comprehensive management of activities, as with the recent focus on ecosystem based management and marine spatial planning, requires a consideration of the spatial patterns of the cumulative impacts of human activities on ecosystems (Crowder et al., 2006; Crowder and Norse, 2008; Halpern et al., 2008a; Crain et al., 2009). Visualising the patterns of overlap in human activities can facilitate effective management of resources

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to reduce impact to areas with multiple stressors and conserve areas that are relatively unused (Selkoe et al., 2008).

Halpern et al. (2008b) developed a framework for evaluating and mapping the cumulative impacts of human activities that is adaptable to a variety of scenarios and scales. Expert judgement was used to rank threats to marine ecosystems in a manner that accounted for differences in ecosystem response (Halpern et al., 2007). The resulting impact weights were used at a global scale to combine the impacts of 17 anthropogenic drivers of ecological change on 20 marine ecosystems into a spatially explicit estimate of cumulative human impact.

Since Halpern et al.'s (2008b) global estimate of human impact on marine ecosystems, adaptations of this approach have been used to assess threats to a Hawaiian coral reef ecosystem (Selkoe et al., 2009), the California Current (Halpern et al., 2009), Canada's Pacific seaboard (Ban et al., 2010), the Baltic Sea (Korpinen et al., 2012), the eastern North Sea (Andersen et al., 2013), and the Mediterranean and Black Seas (Micheli et al., 2013). However, despite the widespread uptake of the model, and Halpern et al.'s (2008b) recommendation to compile regional and global databases of empirical measurements of ecosystem condition to further validate the efficacy of the approach, very little has been done to ground truth the framework.

Halpern and Fujita (2013) identified a number of assumptions that underlie most cumulative mapping efforts: 1) stressor categories are of equal importance, 2) stressors are uniformly distributed within a grid cell, 3) ecosystems either exist or are absent in a grid cell, with no differentiation for habitat quality, 4) decisions are required on how to transform and normalise (standardise) the stressors, 5) ecosystems exhibit linear responses to stressors and cumulative impacts, 6) ecosystem response to a given stressor is consistent over time, 7) vulnerability of ecosystems to stressors can be estimated with sufficient accuracy and 8) stressor impacts are additive. Challenges also exist around data availability, incorporating movement and spatial connections, including historical impacts, accounting for temporal dynamics of stressors and ecosystem responses, the three dimensionality of the oceans, conflating stressors and drivers, and uncertainty in data and their combinations (Halpern and Fujita, 2013). All of these points raise questions about the ability of the Halpern approach to generate realistic estimates of cumulative impacts and their relationship to ecological condition. Given these assumptions and challenges, it is important to test whether this relatively simple model does in fact reflect the complexity of coastal ecosystems and their response to cumulative impacts.

In the present study, Halpern et al.'s (2008b) cumulative impact mapping framework was applied on a finer scale than previous studies, using New Zealand-specific impact weights to estimate the spatial distribution of cumulative impacts in Tauranga Harbour, a large estuary in New Zealand. If cumulative impact scores are ecologically relevant, one would expect to find a strong relationship between those scores and ecological condition, reflected in variance in composition of benthic communities. Using soft-sediment benthic community data, we had the novel opportunity to compare actual ecological condition with the estimated cumulative impact from multiple stressors. Although we had data for a limited number of stressors and a limited number of habitats, these included the most significant stressors and sensitive habitats. 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 environmental health (Pearson and Rosenberg, 1978; Dauer, 1993; Weisberg et al., 1997; Borja et al., 2000). By comparing ecological condition to estimated cumulative impact scores, we hoped to validate for the first time the efficacy of the Halpern approach at a local scale.

2. Methods

2.1. Case study area

Tauranga Harbour is a natural estuarine embayment located on the western edge of the Bay of Plenty on New Zealand's North Island. With an area of 200 km^2, it is sheltered from the Pacific Ocean by low lying Matakana Island, with navigable entrances at the far northern and southern ends. Two harbour basins are separated by large intertidal flats in the central area of the harbour with little or no exchange of water between the two (Barnett, 1985; de Lange, 1988). The two main entrances to the harbour have strong tidal flows (4–7 knots, Ellis et al., 2008) and residence times ranging from a few hours up to a month (Heath, 1976). Mean freshwater inflow is 4.1 m^3/s from the northern harbour catchments and 30.5 m^3/s from the southern catchments (Park, 2009). The freshwater inflow represents only 0.1% of the harbour volume per tidal cycle in the northern basin and 0.48% in the southern basin (Park, 2003).

The sixth largest city in New Zealand (population of approximately 115,000), Tauranga, is situated at the southern end of the harbour, while land in the catchment inland to the west has been developed for agriculture and horticulture. The Port of Tauranga, also situated at the southern end of the harbour, is New Zealand's largest port in terms of cargo volume.

2.2. Ecosystems and stressors

Ecosystems, stressors and the resulting impacts were mapped on a grid with cells of 100 m $\times$ 100 m (one hectare). Dominant ecosystem types within the harbour (mangroves, seagrass, shellfish beds, rocky reef, mud, intertidal sand, subtidal sand) were mapped according to their presence or absence in each grid cell (Fig. 1).

Key pressures on Tauranga Harbour were identified from MacDiarmid et al.'s (2012) assessment of anthropogenic stressors to New Zealand's marine habitats. The scores published in MacDiarmid et al. (2012) describe the vulnerability of ecosystems to stressors. The scores for the dominant ecosystems and stressors present in the harbour were taken, and regrouped into categories for which data was available or estimable: sedimentation, recreational fishing, metals, dredging, physical structures, reclamation, causeways and nutrients (Fig. 2).

No data on sedimentation were available for the entire harbour, so the sedimentation stressor layer was derived using a simple dispersion model (Goodwin and Sinner, 2013). In order to assess the effect of data quality on the resulting cumulative impact scores, the cumulative impact model was also re-run using a more realistic, physically-based model that covers only the southern portion of Tauranga Harbour (Green, 2010).

Stressors associated with climate change (*i.e.* ocean acidification, sea level rise, increased intertidal/sea temperature, increased storminess and change in currents) were excluded from the study because global change is outside the scope of local management action and sufficiently accurate spatial data were either not available or required major processing efforts. While these global scale stressors would likely have a uniform distribution across the study area, different ecosystems may be more sensitive than others to these threats, resulting in differences in cumulative impact across the study domain.

Each of the eight stressors was estimated across the study domain and then log transformed and standardised to allow stressors with different distributions and different measurement units to be compared with each other (Table 1). Standardisation requires choosing a maximum value to set equal to 1.0 and can have important consequences for resulting impact assessments, as every