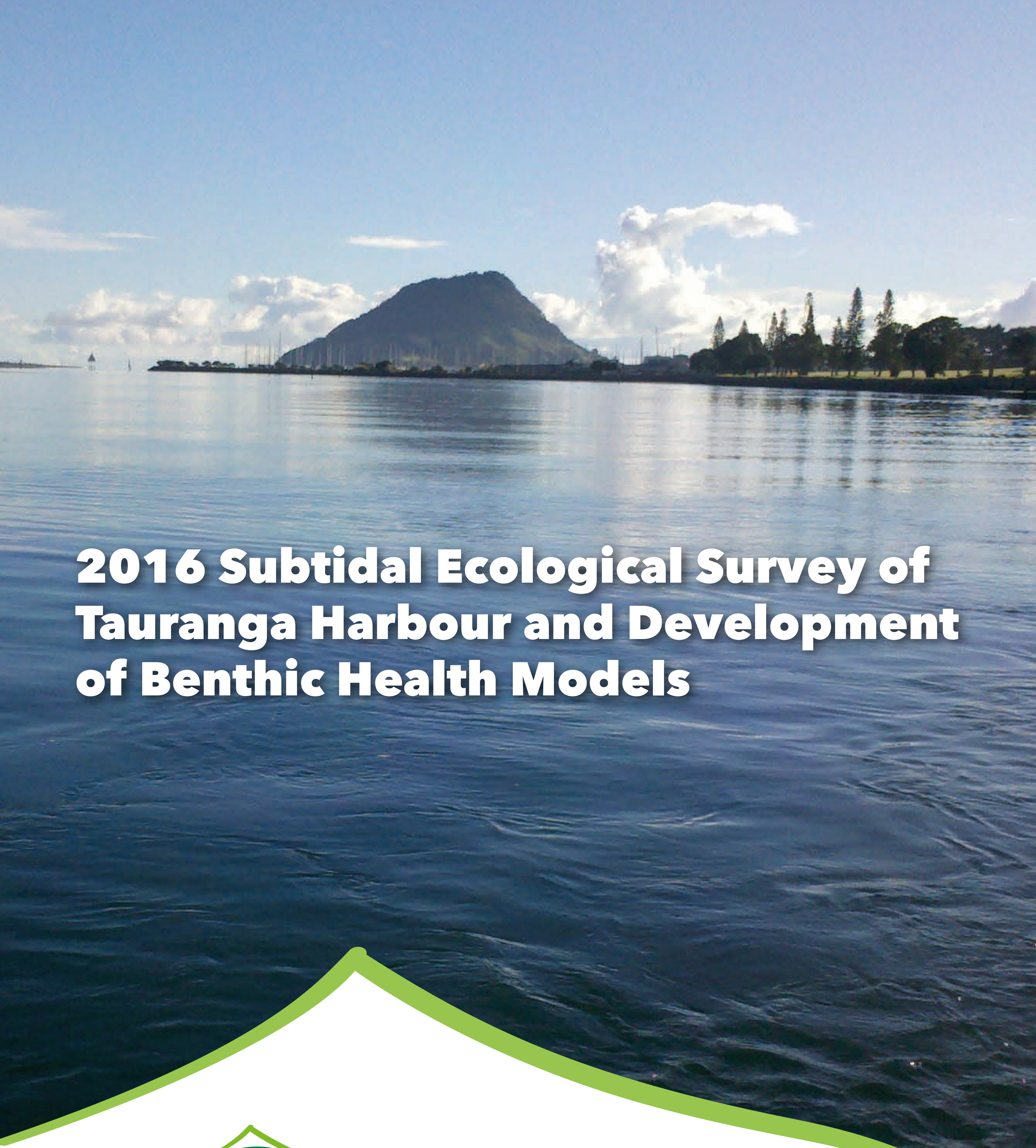

2016 Subtidal Ecological Survey of Tauranga Harbour and Development of Benthic Health Models

Oranga Taiao Oranga Tāngata

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2016 Subtidal Ecological Survey of Tauranga Harbour and Development of Benthic Health Models

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Mihi

I te tīmatanga, ko te kore

Ko te pō

Nā te pō ka puta ko te Kūkune

Ko te Pūpuke

Ko te Hīhiri

Ko te Mahara

Ko te Manako

Ka puta i te whei āo ki te āo mārama

Tihēi Mauri ora

Ki ngā maunga, ki ngā moana

Ki ngā whare maha e karopōti nei i Te Awanui

E rere ana ngā mihi

Ki a rātau kua moe ngā whatu

Takoto mai i te moenga roa

Kia tātau e pīkau ana i ngā āhuatanga o te āo tūroa

Tātau e kōwhaiwhai ana i ngā wawatā ō rātau mā, tēnā koutou

Ka rere tōnu ngā mihi ki te kaupapa

Ko te kaupapa he mea rangatira

He whakapiringa kōrero, he hononga tāngata

Ko te rangahau he kaupapa mutunga kore, ko te tūmanako ko tēnei kohinga kōrero he timatanga noa, kia whakahihiko i te hinengaro, i te wairua kia rere arorangi.

Executive Summary

This report summarises the results of biological and physical data collected from a broad-scale subtidal survey of Tauranga Harbour conducted between March and May 2016. The survey was designed to understand more fully the role of various anthropogenic stressors on ecosystem health and feed into management of the harbour. Community-based models of ecosystem health called Benthic Health Models (BHMs) were developed to assess ecosystem health in response to mud and metal loading. This is the first comprehensive quantitative survey of Tauranga Harbour's subtidal environment since 1990/91.

The research was conducted as part of the *Oranga Taiao Oranga Tangata* (OTOT) programme. The wider research project aims to provide knowledge and toolsets to support co-management of estuaries. The three phases of this programme include 1) gathering *Mātauranga Māori* (a body of knowledge of *Māori* experience in the area) from local *iwi/hapū*, 2) consolidating the ecological knowledge of the Tauranga Harbour and providing modelling and indicators of estuarine ecosystem health, resilience and functioning, and 3) creating an Integrative Spatial Planning Tool (ISPT) that can help inform decision making in the harbour. The subtidal survey fits into the latter two research objectives, as the information collected will be used to develop indicators of ecosystem health, which will be used as components of the ISPT.

Water, sediment and benthic macrofauna samples were collected from 45 subtidal sites across the harbour. Subtidal sediments were predominantly sandy with low levels of nutrients and metal loading. Upper reaches of channels tended to have higher mud, organic content and nutrient concentrations than sites closer to the main channels. Maximum subtidal sediment metal concentrations were well below guideline values and all metals, except lead, were less than median national values for intertidal estuarine sites. Highest metal concentrations were in the urbanised southern harbour or in areas of high mud deposition.

Many of the numerically dominant macrofaunal taxa were present in both the 1990/91 and 2016 subtidal surveys, with pipi the most abundant bivalve in both surveys. Although the 2016 survey was not designed to quantitatively measure large bivalves, compared with the 1990/91 survey, fewer scallops and horse mussels were observed. The apparent decline in these species is concerning as large bivalves stabilise the sediment and provide complex physical structure to soft sediment habitats, providing predation refuges and settlement substrate for epifauna. The invasive Asian date mussel, on the other hand, has also become common in the harbour.

Mud and metals were identified as key variables affecting the ecology of the harbour. To determine how environmental gradients in these variables affected ecosystem health, two Benthic Health Models (BHMs) were developed based on variability in community structure, one for each environmental gradient. The BHM approach is a useful management tool that can be used to determine the relative health of benthic communities at a single point in time, or track sites over time to assess whether communities are moving towards a more healthy or unhealthy state. It has been found to be more sensitive to changing ecosystem health than simple univariate community measures because it preserves more information about the community.

Most sites were ranked in the lower BHM groups, suggesting Tauranga Harbour had fairly healthy subtidal communities with regard to mud and metal impacts. This was supported by values from other biotic indices, which indicated that all subtidal sites in Tauranga would be classified as having ‘good’ ecological status. Sites identified as most impacted by elevated mud and metals were generally located in the upper reaches of estuaries or near the urbanised southern portion of the harbour. In terms of mud, this to some extent reflects the natural progression of an estuary from land to sea; however, the rates of accumulation of sediments have been accelerated because of anthropogenic land-based activities. Although currents and mud were highly coupled in Tauranga’s subtidal environment, the Mud BHM could draw out community responses that were only associated with mud.

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Glossary

a – the intercept of a regression model

ADL – Analytical Detection Limit

AFDW – Ash-Free Dry Weight, a measure of organic enrichment, also referred to as Loss on Ignition (LOI)

Akaike's Information Criterion (AIC) – a type of selection criterion used to identify the best statistical model

AMBI – AZTI Marine Biotic Index, an index used to establish the quality of soft-sediment benthic communities within estuarine and coastal environments

ANZECC - Australian and New Zealand Environment and Conservation Council

As – arsenic, a metalloid commonly found in marine sediments that is used to measure metal contamination

Ascidiacea – a taxonomic class of animals (phylum Chordata), which is composed of sac-like filter-feeding marine invertebrates

b – the slope of a regression model

BHM – Benthic Health Model, a name coined by the Auckland Council to describe a model derived using Canonical Analysis of Principal Coordinates (CAP), which differentiates benthic communities from estuarine sites on the basis of an environmental gradient of interest

Bivalve – animals in the taxonomic class Bivalvia (phylum Mollusca), which are commonly referred to as shellfish

Bray-Curtis dissimilarities – a measure of how dissimilar two samples are in terms of composition based on the Bray-Curtis coefficient, which is commonly used for ecological samples

Bryozoa – a phylum of aquatic invertebrate animals, sometimes incorrectly referred to as corals

CAP – Canonical Analysis of Principal Coordinates, a multivariate constrained ordination technique

Calinski-Harabasz stopping criterion – a stopping criterion that is used to evaluate the optimal number of clusters when carrying out cluster analysis

Cd – cadmium, a metal found in marine sediments used as a measure of contamination

Chl *a* – chlorophyll *a*, a photosynthetic pigment used as a proxy for phytoplankton biomass

CLUES – Catchment Land Use for Environmental Sustainability model, a GIS based modelling system which assess the effects of land use change on water quality and socio-economic indicators

Corr. – correlation between the canonical axis and the pollution gradient (success of model fit)

Corr. sq. (δ_1) – squared canonical correlation for the canonical axis (success of model fit)

Cr – chromium, a metal commonly found in marine sediments and used as a measure of metal contamination

Crustaceans – animals in the sub-phylum Crustacea (phylum Arthropoda), which includes taxa such as shrimps, crabs, isopods and ostracods

CTD – conductivity-temperature-depth profile

Cu – copper, a metal commonly found in marine sediments used as measure of metal contamination

DistLM – Distance-based Linear Modelling, a distance-based regression approach for the analysis of multivariate data in response to multiple predictor variables

ECHI – Estuarine Cultural Health Index

Ethanol – a chemical compound, also known as alcohol, which is used to preserve macrofaunal samples

ETI – Estuary Trophic Index

Gastropod – animals in the class Gastropoda (phylum Mollusca), which are commonly referred to as snails

GIS – Geographic Information System

H – Shannon-Wiener diversity, a diversity index that describes, in a single number, the different types and amounts of animals present in a collection. Varies with both the number of species and the relative distribution of individual organisms amongst the species. The index ranges from 0 for communities containing a single species to high values (> 5) for communities containing many species and each with a small number of individuals.

Hapū – clans or descent groups

Hg – mercury, a metal commonly found in marine sediments that is used as a measure of metal contamination

Hierarchical agglomerative group-average clustering – a type of clustering method where samples are grouped and the groups themselves form clusters at lower levels of similarity based on group averages. These usually take a similarity matrix as their starting point and successively fuse the samples into groups and the groups into large clusters, starting with the highest mutual similarities then lowering the similarity level when all samples are in a single cluster.

Hydrozoa – a taxonomic class of very small, predatory animals in the phylum Cnidaria

Intertidal – area that is above the water at low tide and under water at high tide

ISPT – Integrative Spatial Planning Tool

ISQG – Interim Sediment Quality Guidelines

Iwi – 'people' or 'nation' or 'tribe'

J – Pielou's evenness, a measure of equitability, or how evenly the individuals are distributed amongst the different species/taxa. Values can theoretically range from 0 to 1, where a high value indicates an uneven distribution or dominance by a few taxa.

***k*-means** – a non-hierarchical clustering method of vector quantisation that classifies data into a certain number of groups, fixed *a priori*, based on feature similarity. A stopping criterion, such as the Calinski-Harabasz stopping criterion, can be used to evaluate the optimal number of groups.

***k*-*R* CLUSTERING** – a non-hierarchical method of clustering which seeks to maximise the ANOSIM *R* statistic to obtain a *k*-group division of the samples, where *k* is the desired number of clusters

LOI – Loss on Ignition, a measure of organic enrichment, also referred to as Ash-Free Dry Weight (AFDW)

ln – natural logarithm

m – number of Principal Coordinate Analysis (PCO) axes

Macrofauna – small marine invertebrate animals that are greater in size than 0.5 mm, such as shellfish and worms. In this survey, we are focusing primarily on animals that live within the sediment.

Mātauranga Māori – a body of knowledge of *Māori* experience in an area

MBIE – Ministry of Business, Innovation and Employment

Mean sea level – the sea level halfway between the mean levels of high and low water

Meiofauna – small marine invertebrate animals, defined as being greater in size than 50 µm but less than 0.5 mm

MTM – *Manaaki Taha Moana* research programme

Multivariate statistics – a subdivision of statistics encompassing the simultaneous observation and analysis of more than one outcome or response variable

N – total abundance, which is a measure of the total number of individual organisms in a sample

NERMN – National Environment Regional Monitoring Network

NIWA – National Institute of Water and Atmospheric Research

Ni – nickel, a metal found in marine sediment, which is used as a measure of metal contamination

nMDS – non-metric Multidimensional Scaling, a multivariate ordination technique used to display the information contained in a dissimilarity matrix

Non-hierarchical clustering – a type of clustering method where samples are separated into a pre-determined number of groups using an iterative algorithm that optimises a chosen criterion, such as the Calinski-Harabasz stopping criterion

NZTM – NZGD 2000 New Zealand Transverse Mercator, a coordinate system

OTOT – *Oranga Taiao Oranga Tangata* research programme

Pb – lead, a metal found in marine sediments, which is used as a measure of metal contamination

PC1 – the first principal component axis, the projection of points onto the line of 'best fit' in two-dimensional space, in a Principal Component Analysis (PCA)

PCA – Principal Components Analysis, a multivariate unconstrained ordination technique, which is well suited to environmental data but not species abundance data

PCO – Principal Coordinates Analysis, a multivariate unconstrained ordination technique which can be applied completely generally to any resemblance measure

Pearson correlation coefficient – a measure of the linear correlation between two variables

Polychaete – worms in the class Polychaeta (phylum Annelida)

Porifera – a phylum of primarily marine invertebrate animals more commonly known as sponges

Prop. G – proportion of the total variation in the dissimilarity matrix explained by the first *m* principal coordinates analysis (PCO) axes

PSU – practical salinity unit

r – Pearson's correlation coefficient, a measure of the linear correlation between two variables

R^2 – coefficient of determination, is the proportion of the variance in the dependent variable that is predictable from the independent variable(s). It ranges from 0 to 1 and can indicate the strength of a relationship

RI-AMBI – Richness Integrated AMBI, a modification of the AZTI Marine Biotic Index, which is used to establish the quality of soft-sediment benthic communities within estuarine and coastal environments

S – species richness, which is a measure of the total number of taxa at a site

SIMPER – Similarity Percentages, a multivariate analysis which determines how similar or dissimilar groups are to each other in terms of community structure or composition

SIMPROF – Similarity Profiles, a statistical analysis that tests the hypothesis that within a given set of samples there is no genuine evidence of multivariate structure

SS_{RES} – the leave-one-out residual sum of squares, where a lower number indicates a better model fit

Subtidal – area that is below the low water mark

Te Awanui – Tauranga Harbour

TOC – total organic carbon, a measure of organic enrichment

TEL – threshold effects level, a level of metal contamination beyond which adverse effects on benthic species can occur

Tombolo – a deposition landform in which an island is attached to the mainland by a narrow piece of land such as a spit or bar

TN – total nitrogen, a measure of nutrient enrichment

TP – total phosphorus, a measure of nutrient enrichment

Zn – zinc, a metal commonly found in marine sediments that is used as a measure of metal contaminations

1. Introduction

1.1. Background and scope of the report

A previous Cawthron report summarised information about the ecosystem health of Tauranga Harbour — traditionally known to local iwi as *Te Awanui* — in order to inform the Tauranga community, iwi and stakeholders of the 'state of the harbour' and to identify information gaps and priorities for field research (Sinner et al., 2011). The report was based on a literature review of published scientific papers and technical reports and did not extend to new field work or new analysis and interpretation of data. The report indicated that while studies have been conducted on a wide range of ecological topics, studies that assess biodiversity of marine flora and fauna at the scale of the estuary had not been conducted since 1990. The spatial scale over which information has been collected also varied greatly between studies, reflecting the diverse purposes for which specific studies were undertaken. Thus, in order to understand more fully the role of various anthropogenic stressors on biodiversity, a broad scale survey of Tauranga Harbour was suggested (Sinner et al., 2011).

Following these recommendations, an estuary-wide intertidal survey of Tauranga Harbour was carried out during the summer of 2011/12 as part of the *Manaaki Taha Moana* (MTM) research programme (Ellis et al., 2013). The report from this survey provided general information on spatial trends of macrofaunal species distributions, sediment types, nutrient and metal concentrations across the harbour and developed community-based models of ecosystem health called Benthic Health Models (BHMs).

The current report summarises the results of a broad-scale subtidal survey that was conducted between March and May 2016 to provide information on the ecosystem health of subtidal portions of the harbour. This survey was carried out as part of the *Oranga Taiao Oranga Tangata* (OTOT) research programme. The subtidal survey followed similar methods to those used in the 2011/12 intertidal survey to ensure general comparability between surveys. Like the 2011/12 intertidal survey, the subtidal survey included information on sediment type, organic enrichment, nutrient and metal concentrations, chlorophyll *a* and macrofaunal community structure. In addition, water column variables were measured, and current velocities estimated. Two Benthic Health Models (BHMs) were developed for subtidal portions of the harbour, which rank the health of intertidal sites based on observed responses to mud (Mud BHM) and metals (Metals BHM).

This report focuses on the results of the subtidal survey, however, some 2011/12 intertidal survey results are presented for comparative purposes and to provide a comprehensive description of the harbour's benthic ecology. Given the surveys were carried out four years apart some caution should be exercised when drawing comparisons between the two datasets. Reference is also made to Park and Donald's 1990/91 survey (Park & Donald, 1994), which was previously Tauranga Harbour's most recent quantitative harbour-wide subtidal survey. However, direct comparison between the 1990/91 and 2016 subtidal datasets is difficult because macrofauna were only sieved on a 1 mm mesh during the former survey, biasing those samples towards larger organisms.

The aims of this report are to:

1. Provide information on spatial patterns in water and sediment physico-chemical variables across the harbour
2. Provide information on spatial patterns in subtidal macrofaunal community structure across the harbour
3. Identify key environmental gradients driving changes in subtidal community structure and develop Benthic Health Models (BHMs) to classify subtidal sites according to categories of relative ecosystem health, based on community structure and predicted responses to these environmental gradients.

1.2. Background to the Benthic Health Model approach

The Benthic Health Model (BHM) is the name coined by Auckland Council to describe a modelling approach that uses a constrained ordination technique called Canonical Analysis of Principal coordinates (CAP) to characterise changes in macrofaunal community structure in response to stressors. Although macrofaunal community structure is affected by many different environmental and biotic factors, the CAP model draws out responses that are only associated with the environmental gradient of interest. Community structure found in areas largely unaffected by anthropogenic disturbances (versus that observed in more 'impacted' areas) can be used as a benchmark against which to assess the relative health of community structure found at specific sites. Thus, relative health of a site or sites can be defined in terms of the range of communities present in a set of comparative locations that are not considered to be affected by anthropogenically-derived inputs. The difference should serve to identify both acute effects and broader-scale chronic degradation. Consistent with Hewitt et al. (2005), this study defines ecosystem health on the basis of the range of communities observed along gradients of environmental impacts. This definition identifies both acute effects and broader scale degradation in community structure.

The BHM approach was originally developed by Auckland University and the National Institute of Water and Atmospheric Research (NIWA) for the Auckland Council as a way of assessing ecosystem health for the council's estuarine monitoring programme. The models were developed as a tool to classify intertidal sites within the region according to categories of relative ecosystem health, based on community structure and predicted responses to storm-water contamination (Anderson, Hewitt, Ford, & Thrush, 2006) and mud content (Hewitt & Ellis, 2010). While the BHM can be used as a snapshot in time of the relative health of benthic communities, this approach is more useful 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. Once the BHM has been developed, new observations from existing or new sites can be easily placed along the canonical axis and community 'health' defined based on its position in the ordination space (Anderson & Robinson, 2003).

Most councils already collect macrofaunal information suitable for use in BHMs. Macrofaunal data are often used as a measure of ecosystem response because these communities respond relatively rapidly to anthropogenic and natural stress, integrate the effects of multiple stressors over time and consist of a diverse range species that exhibit different feeding behaviours, sensitivity to stress and roles in ecological succession (Dauer, 1993; Gray et al., 1979; Pearson & Rosenberg, 1978). Macrofaunal community information can be used in a variety of ways ranging from simple univariate measures (e.g. number of individuals, species richness, Pielou's evenness, Shannon-Wiener diversity) to more complex univariate measures that integrate information on taxa sensitivities to stress (e.g. AMBI, BENTIX; Borja, Franco, & Perez, 2000; Simboura & Zenetos, 2002) to multivariate approaches, which incorporate information on all species and their relative abundances (e.g. Ellis et al., 2015; Hewitt et al., 2005). The BHM approach has been found to be more sensitive to changing ecosystem health than simple univariate measures because it utilises all of the information on the abundance of each taxon, allowing a more ecologically meaningful response to be observed (Ellis et al., 2015; Hewitt et al., 2005).

1.3. Parent research programmes

1.3.1 Oranga Taiao Oranga Tangata (OTOT)

The subtidal survey of Tauranga Harbour was carried out under the research programme *Oranga Taiao, Oranga Tangata: Knowledge and Toolsets to Support Co-Management of Estuaries* (MAUX1502), which is funded by the Ministry of Business, Innovation and Employment (MBIE). This research programme ($4.4 million) focuses on Tauranga Harbour and its catchment as a case study. It is a four-year research programme (October 2015 to September 2019) that has three phases:

Phase 1 focuses on gathering *Mātauranga Māori* (a body of knowledge of *Māori* experience in the area) from local *iwi/hapū.* Based on this information, an Estuarine Cultural Health Index (ECHI), or other similar tool(s), will be constructed to help *iwi/hapū* to assess the state of local estuarine habitats, record changes over time and judge the effectiveness of factors such as local fishing rules and management strategies.

Phase 2 will consolidate the ecological knowledge of Tauranga Harbour and provide some preliminary modelling and indicators of estuarine ecosystem health, resilience and functioning.

Phase 3 will create an Integrative Spatial Planning Tool (ISPT). This tool is a hybrid GIS-modelling system that will use information from the estuarine ecology, land use, economic and cultural areas, where appropriate, to predict changes in ecosystem health under different socio-economic scenarios. It will enable users to evaluate future planning options for Tauranga Harbour. This integrative planning tool should be at the leading edge of socio-economic and ecological modelling developments worldwide, although such tools have been developed for the terrestrial environment, few if any spatial-modelling tools have been developed for the-whole-of catchment, including both land and coastal-marine ecosystems.

In all phases, the knowledge, frameworks and toolsets will be developed in such a way to foster transference and uptake to other *iwi* and regions throughout New Zealand, where possible, to enhance the health of estuaries nationwide and internationally.

1.3.2 Manaaki Taha Moana (MTM)

Enhancing Coastal Ecosystems for Iwi: Manaaki Taha Moana (MAUX0907), was a predecessor to OTOT and was also funded by MBIE. This six-year programme ($6.6 million) ran from October 2009 to September 2015, with case study research conducted primarily in two areas: Tauranga Harbour and the Horowhenua coast (north of Wellington).

The central research question of MTM was:

"How can we best enhance and restore the value and resilience of coastal ecosystems and their services, so that this makes a positive contribution to iwi identity, survival and welfare in the case study regions?"

Thus, the research aimed to restore and enhance coastal ecosystems and their services of importance to *iwi/hapū*, through a better knowledge of these ecosystems and the degradation processes that affect them. The key features of the research were that it was cross-cultural, interdisciplinary, applied/problem solving, and integrated the ecological, environmental, cultural and social factors associated with coastal restoration.

2. Materials and 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 (37° 40'S, 176° 10'E; Figure 1). The harbour is protected from the Pacific Ocean by a barrier island (Matakana Island) and two barrier tombolos: Bowentown at the northern entrance, and Mount Maunganui to the south. Two harbour basins are separated by large intertidal flats in the centre of the harbour. Although the two basins are connected there is little water exchange between the two (Barnett, 1985; de Lange, 1988). The harbour is predominantly shallow (< 10 m deep), with intertidal flats comprising approximately 66% of the total area (Inglis et al., 2008). Overall, Tauranga Harbour is considered to be well mixed and flushed, with an average residence time of 1.7 to 3.1 days in the main channel of the southern basin (Tay, Bryan, de Lange, & Pilditch, 2013). Sub-estuaries further from the harbour entrances, or with constricted entrances, are expected to have longer residence times (average of 3.1-8.2 days; Tay et al., 2013), making these areas more susceptible to accumulation of sediments and contaminants. The Wairoa River is the main freshwater input into the harbour with a mean inflow of 17.6 m^3/s (Park, 2004). The city of Tauranga (population of 138,000) is located on the edge of the southern basin. Near the Mt Maunganui entrance, the Sulphur Point region of the harbour has been progressively developed for port facilities, including channel dredging and land reclamation activities.

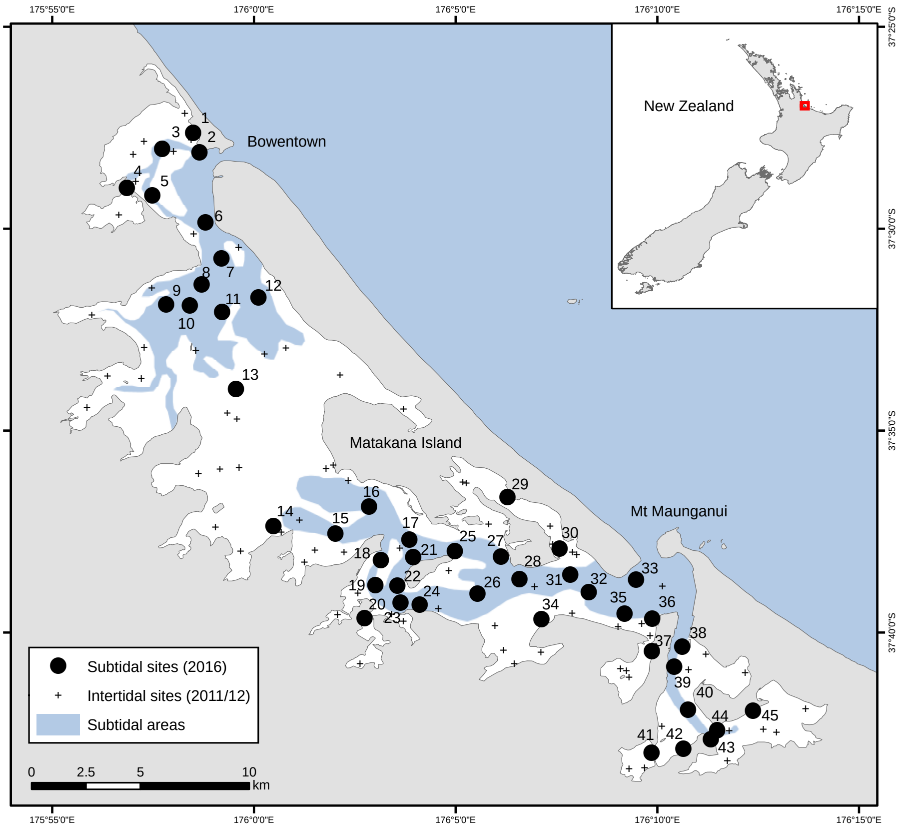

Figure 1. Map of Tauranga Harbour showing the location of the 45 subtidal sites (numbered dots) sampled in 2016 as well as the 75 intertidal sites previously sampled in 2011/12 (crosses). Blue shading indicates approximate extent of subtidal areas.

2.2. Sampling

2.2.1 Survey design

Sampling was carried out from 15 March to 4 May 2016 by researchers from the Cawthron Institute, Manaaki Te Awanui and the University of Waikato. The sampling design was chosen to provide data generally comparable to those generated from the 2011/12 intertidal ecological survey of Tauranga Harbour (Ellis et al., 2013). A total of 45 subtidal sites (refer Appendix 1 for site coordinates) were surveyed for water and sediment physico-chemical variables and benthic macrofauna. Sites were selected to provide good spatial coverage of subtidal portions of the harbour and to link in with existing sites that are monitored as part of the National Environment Regional Monitoring Network (NERMN) Coastal Estuarine Ecology Programme.

2.2.2 Water column physico-chemical variables

One-off conductivity-temperature-depth (CTD) profile measurements were taken at each of the 45 sites. The CTD casts were not necessarily collected on the same day as the benthic physico-chemical and macrofauna samples (refer Appendix 1 for sampling dates). CTD profiles were taken across the entire water column but only surface (0.1-0.2 m water depth) and bottom (0.1-0.2 m above the seafloor) measurements are reported. Mean and maximum current velocities were modelled by a hydrodynamic model called the Semi-implicit Cross Scale Hydroscience Integrated

System Model (SCHISM; Zhang, Ye, Stanev, & Grashorn, 2016), using the mean over 14 days (refer Appendix 2 for details).

2.2.3 Sediment physico-chemical variables

At each site, six sediment cores (6 cm diameter, 10 cm deep) were collected by divers within 5 m of a shot line located at each site. The top 2 cm of each core was sampled, and the six replicates composited into a single sample representing each site. This depth of sediment was targeted for sampling because previous work has shown that in most settling zones[1], the top 2 cm contains sediment deposited over a 0.2 to 7 year period (ARC, 2004). In terms of detecting trends, 2 cm is a compromise between shallower depths that may be biased by one or two large recent events and greater depths where recent changes in sediment parameters are diluted by levels laid down in the more distant past. Each sediment sample was analysed for sediment grain size, organic matter (total organic carbon, TOC; loss on ignition[2] or LOI), nutrients (total nitrogen, TN; total phosphorus, TP), metals (copper, Cu; lead, Pb; zinc, Zn; arsenic, As; cadmium, Cd; chromium, Cr; mercury, Hg; nickel, Ni), and chlorophyll *a* (Chl *a*). Two proxies for organic enrichment were measured; LOI was included to be consistent with the intertidal survey and TOC was measured to be consistent with a shifting trend towards using TOC as an indicator of organic enrichment. See Table 1 for details of respective methods and detection limits for each sediment physico-chemical parameter measured by RJ Hill Laboratories in Hamilton.

[1] Depositional areas where most contaminants (~75%) settle out of suspension and become incorporated into benthic sediments

[2] Loss on ignition (LOI) is the same as ash-free dry weight (AFDW), which was the terminology used in the Ellis et al. (2013) report for the intertidal survey.

Table 1. Sediment analysis methods and detection limits as reported by RJ Hill Laboratories Limited.

Parameter	Laboratory method	Detection limit (dry weight)
Grain-size	Wet sieving using dispersant. Seven size classes: > 2 mm (gravel) < 2 mm, ≥ 1 mm (sand) < 1 mm, ≥ 500 µm (sand) < 500 µm, ≥ 250 µm (sand) < 250 µm, ≥ 125 µm (sand) < 125 µm, ≥ 63 µm (sand) < 63 µm (mud)	0.1 g/100 g
TOC	Acid pre-treatment to remove carbonates followed by catalytic combustion (900°C, O2) separation, thermal conductivity detector [Elementar Analyser].	0.05 g/100 g
LOI	Ignition in muffle furnace at 550°C for 6 hrs, gravimetric. APHA 2540 G 22nd ed. 2012.	0.04 g/100 g
TN	Catalytic combustion (900°C, O2), separation, thermal conductivity detector [Elementar Analyser].	0.05 g/100 g
TP	Dried sample, nitric/hydrochloric acid digestion, ICP-MS, screen level. US EPA 200.2.	40 mg/kg
Metals (Cu, Pb, Zn, As, Cd, Cr, Hg, Ni)	Dried sample, < 2 mm fraction. Nitric/hydrochloric acid digestion, ICP-MS, trace level.	0.010-0.4 mg/kg
Chlorophyll *a*	Extraction with 95% ethanol, spectroscopy. Subcontracted to NIWA, Hamilton. In-house.	0.1 mg/kg as rcvd

2.2.4 Benthic macrofauna

To quantify benthic community structure at each site, samples of macrofauna living within the sediment (i.e. animals > 0.5 mm, such as worms and shellfish) were collected by divers. At each site five macrofauna cores (13 cm diameter, 15 cm deep) were collected within 5 m of a shot line. The macrofaunal samples were separated using stacked sieves with mesh sizes of 1 mm and 0.5 mm. Macrofauna retained on the sieves were preserved with ethanol (diluted to approximately 70% with seawater). For three replicates, macrofauna from both sieves were sorted, identified and counted to the lowest taxonomic resolution. For the remaining two replicates, only macrofauna from the 1 mm sieve were retained and only four shellfish species sorted, identified and counted. These shellfish were cockles (*Austrovenus stutchburyi*), pipi (*Paphies australis*), wedge shells (*Macomona liliana*) and nut shells (*Linucula hartvigiana*). All macrofaunal data presented in this report is based on the three replicates for which both 0.5 mm and 1 mm data were available, except for shellfish distributions (Section 3.3.2) that are based on the five replicates sieved to only 1 mm. Macrofaunal data for Site 16 was misplaced, thus this site was excluded from the analyses and results.

2.3. Data analyses

2.3.1 Water column and sediment physico-chemical variables

For all sediment physico-chemical parameters, values below the analytical detection limit (ADL) were divided by two (e.g. TN < 500 mg/kg was treated in any analyses as 250 mg/kg). Minimum, mean and maximum values were calculated for all water and sediment physico-chemical variables. Spatial patterns in key variables were examined by plotting values on a map of the harbour.

2.3.2 Benthic macrofauna

The raw macrofaunal count data were analysed to provide total abundance (average total number of individuals per site, N) and species richness (total number of taxa per site, S) and the top 10 most abundant taxa across all sites calculated. For each of the four key shellfish species (cockles, pipi, wedge shells and nut shells), average abundance was calculated by site. Spatial patterns in N, S and shellfish abundances were examined by plotting values on a map of the harbour.

The count data for each replicate sample were square-root transformed to de-emphasise the influence of dominant species (by abundance) before calculating Bray-Curtis dissimilarities. Differences in community structure amongst sites were visualised using non-metric multidimensional scaling (nMDS; Clarke, Gorley, Somerfield, & Warwick, 2014). Distances among centroids were calculated using site as a grouping factor and nMDS was then used to place sites in a 2-, or 3- or multidimensional space according to their similarities and differences. If a 2-dimensional (2-D) representation explains a sufficient proportion of the samples' differences observed, these can be assessed spatially on a 2-D plot, where the distance between sample points corresponds to the degree of difference observed between macrofaunal assemblages. A stress statistic provides a measure of how well the plot represents the differences between all the individual sites.

Comparisons between intertidal and subtidal sites were made by standardising the taxonomic resolution of intertidal macrofaunal data to be consistent with that used for the subtidal sites. Non-metric multidimensional scaling (nMDS) based on Bray-Curtis dissimilarities between site centroids of square-root transformed abundance data was used to visualise differences in community structure. Similarity Percentages (SIMPER) was used to determine how similar or dissimilar intertidal and subtidal sites were to each other in terms of community structure. Comparison of the proportions of different taxonomic groups between intertidal and subtidal macrofaunal communities was carried out to facilitate comparison with Park and Donald's 1990/91 survey (Park & Donald, 1994). All statistical analyses were carried out using the statistical software PRIMER 7 (v 7.0.13; Clarke & Gorley, 2015)

2.3.3 Benthic Health Model development

A general background to the multivariate analyses used in this report, with reference to canonical analysis of principal coordinates (CAP), can be found in Ellis et al. (2013). All statistical analyses were carried out using the statistical software PRIMER 7 (v 7.0.13) with the PERMANOVA+ add-on (Anderson, Gorley, & Clarke, 2008; Clarke & Gorley, 2015).

Data exploration

For all sediment physico-chemical parameters, values below the analytical detection limit (ADL) were divided by two. Histograms and draftsman plots were used to check for normality of water and sediment physico-chemical variables. Optimal transformations were performed to limit the distorting effects of outliers and improve normality (Table 2).

Table 2. Optimal transformations for physico-chemical and water column variables.

Transformation	Parameters
Log: ln(V)	TP, TN, Chl *a*, Cu, Pb, Zn, LOI, TOC, mud
Square-root: sqrt(V)	Gravel, depth
Reverse log: ln(100-V)	Sand
Reverse square-root: (36-V)^0.5	Bottom salinity
No transformation	Surface salinity, surface temperature, bottom temperature, mean currents, maximum currents

A lead/Pb value of 77 mg/kg at Site 36 was deemed to be an outlier as it was high compared to other estuary monitoring sites in New Zealand (only one site > 77 mg/kg in the intertidal National Estuary Dataset (140 mg/kg); Berthelsen, Clark, Goodwin, Atalah, & Patterson, 2018) and also in relation to measured Cu and Zn concentrations at the site, which are often highly correlated with Pb levels. Exploration of the macrofaunal data did not indicate that the benthic community at Site 36 was impacted by high levels of Pb. The high value could be due to a flake of lead paint or an analysis error, but most likely does not represent the true Pb value at that site. The missing routine in PRIMER 7 was used to estimate lead from transformed Cu and Zn concentrations (Clarke & Gorley, 2015) and Site 36 was excluded from the Benthic Health Model development (used only as a validation site).

Strong correlations (r > 0.8) were identified between some predictor variables (Table 3; Appendix 3). Thus, Principal Component Analysis (PCA; Jolliffe, 2002) using log-transformed metals data was used to derive a single metal variable (the first principal component axis, PC1) that characterised an overall gradient of all three key metals (Cu, Pb, Zn). The PC1 axis for metals (PC1 metals) explained 92% of the variance in these three metals. We followed Anderson et al. (2006) and carried out PCAs on raw, rather than normalised data, which resulted in no practical difference to the results. By doing the analysis on raw (log transformed) variables, we retained the relative ease of placing a new (validation) object into the PCA space based on the original metal concentration measured at a site. Although mud, LOI and TOC were also highly correlated, a PCA was not performed for these three variables because we wanted to try and tease apart the effects of mud on macrofaunal communities independently of the effects of LOI and TOC.

Table 3. Pearson correlation coefficients greater than 0.8. LOI = loss on ignition, TOC = total organic carbon, Cu = copper, Pb = lead, Zn = zinc, S = surface, B = bottom. Refer Appendix 3 for results in full.

	Mud	LOI	Cu	Pb	Temperature (S)	Salinity (S)
LOI	0.86					
TOC	0.84	0.86				
Pb			0.89			
Zn			0.84	0.90		
Temperature (B)					0.98	
Salinity (B)						0.90

Macrofauna from both 0.5 mm and 1 mm-size fractions were combined for each of the three replicates for which these data were collected. Some taxa were removed from the dataset before the analysis (Appendix 4): juveniles, larvae, meiofauna, insects and taxa that would not be well-represented by core sampling (e.g. Ascidiacea, Bryozoa, fish, Hydrozoa, Porifera). Macrofaunal data were square-root transformed to de-emphasise the influence of dominant taxa, but still allow differences in relative abundance to influence the results as this was considered meaningful in terms of ecosystem health.

Determining key environmental gradients

The first step in model development was to identify the key environmental gradients driving changes in Tauranga Harbour's subtidal benthic communities. Multivariate linear regression using Distance-based Linear Modelling (DistLM; McArdle & Anderson, 2001) was used to identify variables that explain the maximum variation in community structure. DistLM was conducted using Bray-Curtis dissimilarities (Bray & Curtis, 1957) of the square-root transformed macrofaunal data. Models were selected using a forward selection procedure based on the Akaike's Information Criterion (AIC). Fifteen transformed predictor variables were included in the analysis: % mud, % sand, % gravel, TN, TP, depth, bottom salinity, bottom temperature, Chl *a*, TOC, LOI, mean current velocity, maximum current velocity, PC1 metals and sampling day[3]. Surface temperature and salinity were not included because they were highly correlated with bottom temperature and salinity. Although strongly correlated, mud, LOI and TOC were retained as variables because we wanted to try and tease apart the effects of mud on macrofaunal communities independently of the effects of LOI and TOC.

DistLM identified % mud, mean current velocity and PC1 metals as key environmental variables driving changes in community structure (refer Section 3.4.1 for further details). As we were interested in developing BHMs to describe the effects of environmental gradients associated with anthropogenic stressors on subtidal communities, mud (represented by ln % mud) and metal loading (represented by PC1 metals) were chosen as the variables of interest for the BHMs. Mean current velocity was also identified as explaining a significant proportion of the variation in community structure so a number of sequential DistLMs were run (varying input variable order) to identify how much of the explained variation overlapped between mud, metals and currents.

Determining the relationship between environmental gradients and macrofaunal community structure

Canonical analysis of principal coordinates (Anderson & Robinson, 2003; Anderson & Willis, 2003) was used to test the relationship between macrofaunal community structure and each key environmental gradient (i.e. mud and metals). CAP allows a constrained ordination to be done on the basis of any dissimilarity or distance measure of choice (such as the Bray-Curtis measure; Bray & Curtis, 1957) and determines the axes that best discriminates an environmental gradient. Separate CAP models for mud and metals were constructed using thirty-eight sites to develop the model and six sites to validate model performance. All CAP analyses were performed on Bray-Curtis dissimilarities based on the square-root transformed macrofaunal data. Additional CAP analyses were carried out to determine if model performance could be improved by 1) the omission of taxa that were rare in abundance or occurrence, 2) higher taxonomic lumping of taxa that were rare in abundance or occurrence, 3) the omission of highly aggregative taxa (*Austrominius modestus, Balanus decorus*, *Balanus* sp.) or 4) the use of untransformed Bray-Curtis dissimilarities. Model performance was not improved, thus results are not shown.

[3] Day when sediment and macrofaunal samples were collected

Classifying sites along the environmental gradients

The sites were classified from 'less impacted' to 'more impacted' based on their position along each environmental gradient (i.e. mud and metals) and their community response. These classifications are only relative to the subtidal sites sampled in Tauranga Harbour. Four methods were used to identify possible groupings along each environmental gradient; 1) evenly splitting the environmental gradients into groups, 2) hierarchical agglomerative group-average clustering (Clarke et al., 2014), 3) non-hierarchical clustering using *k*-means (Hartigan & Wong, 1979) with the number of groups determined using the Calinski-Harabasz stopping criterion (Calinski & Harabasz, 1974) and 4) non-hierarchical clustering using *k*-*R* CLUSTERING with the optimal number of groups determined using SIMPROF (Clarke et al., 2014). Once the classification boundaries were defined along each environmental gradient these values were converted into CAP scores and used as cut-off points for the BHM groups. Conversion into CAP scores was carried out using the following equations:

$$\text{Mud BHM CAP score} = 0.293 + (-0.1658 * \ln \% \text{ mud})$$
$$\text{Metals BHM CAP score} = -0.0012 + (-0.1175 * \text{PC1 metals})$$

Comparison with other biotic indices

CAP scores were compared with values obtained using two other commonly-used biotic indices for assessing the health of estuarine benthic communities; the AZTI Marine Biotic Index (AMBI; Borja et al., 2000) and the Richness Integrated AMBI (RI-AMBI; Robertson, Savage, Gardner, Robertson, & Stevens, 2016). The AMBI is a widely used index that was developed to establish the ecological quality of European soft sediment benthic communities within estuarine and coastal environments. Index values are based upon the sensitivity or tolerance of benthic macrofaunal communities to stress gradients and range from 0 (unpolluted/normal benthic community health) to 7 (extremely polluted/azoic benthic community health). The index can be used to classify benthic communities into five ecological status groups ranging from high status to bad status (Borja, Franco, & Muxika, 2003). RI-AMBI is a modification of AMBI, which incorporates a measure of species richness into the equation. AMBI and RI-AMBI values were calculated following the methods and eco-groups used in Berthelsen et al. (2018).

2.3.4 Benthic Health Model validation

The accuracy of each CAP model at identifying and predicting real and repeatable patterns in the data, was measured by its ability to correctly place six validation sites onto the environmental gradient. This is an important step because high canonical correlation does not necessarily mean good predictive power (Anderson et al., 2006). For example, high canonical correlation can be achieved by simply increasing the number of principal coordinate analysis (PCO) axes (m) to be used in the CAP analysis. Validation sites were chosen by dividing the harbour into five sections and randomly choosing a validation site from each section. Each validation site was checked to make sure it did not contain the minimum or maximum value for any of the environmental predictor variables (mud, Cu, Pb, Zn), otherwise a different site was randomly selected. The validation sites were Site 2, 5, 15, 27 and 44. Site 36 was excluded from model development due to its unusually high Pb value, so this site was also used as an additional validation site.

Validation was carried out in accordance with Anderson et al. (2006). The aim of the validation was to determine how closely the final models were able to place each new validation site onto the existing canonical axes, and from this, to predict the true position of that site along the relevant environmental gradient. First, the physico-chemical data (either ln % mud or PC1 metals) at each validation site was used to place the site onto each of the environmental gradients. These positions were deemed to be the true or observed values for those sites along each environmental gradient. The CAP model (also referred to as the BHM) was then used to place each validation site onto the environmental gradient axes by calculating the Bray-Curtis dissimilarity between that site and the sites in the model. In a CAP model, the dissimilarity between any two sites does not depend on the other sites in the model so adding a new site to the model does not causes distances among other points to change. For each model, these were the predicted values along the environment gradient. The sum of squares deviations of the predicted values from the observed values (the residual sum of squares, SS_{RES}) were calculated for each model. The models with the smallest values for SS_{RES} were considered to have achieved the best predicted fit.

Linear regression of predicted versus observed values was used to identify sites whose predicted values deviated most from their observed values and in which direction. A 1:1 line (i.e. with slope (b) = 1 and intercept (a) = 0) was drawn to help interpret the positions of the points. If prediction is exact, the points would lie precisely on this line. The slope of the linear relationship, b, and the strength of the relationship (coefficient of determination, R^2), between the predicted and observed values was also used to determine validation success. Models were considered good if b and R^2 were close to 1.

The potential for interactions with other factors was examined by looking at Pearson's correlation coefficients between the CAP scores and other predictor variables. Potentially confounding predictor variables (i.e. highly correlated with CAP scores) were overlaid on the BHM plot to check no patterns were evident. Interactions between the mud and metals BHMs were checked by examining Pearson's correlation coefficients and graphing the relationship between CAP scores from each model.

3. Results

3.1. Water column physico-chemical variables

Table 4 provides a summary of water column parameters measured or modelled during the subtidal survey (see Appendix 1 for results in full). Study sites were relatively shallow with the deepest site only 9.4 m and a mean depth of 4.5 m across all sites. Water temperature ranged from 17.8 to 23 °C with a maximum of 1 °C difference between surface and bottom waters. Mean surface and bottom salinity was 30.3 to 31.2 PSU, respectively. Site 1, near Pios Beach by Bowentown (Figure 1), had particularly low salinity for both the surface and bottom (21.2 and 22.7 PSU, respectively), with the next lowest salinity at Site 23, which had surface salinity of 25.8 PSU. Modelled mean current velocities were relatively high at most sites with an average of 0.36 m/s and maximum of 1.52 m/s (Figure 2).

Table 4. Minimum, mean and maximum values for water column variables measured in Tauranga Harbour 17 March to 4 May 2016. Current velocities were modelled using the mean over a 14-day period.

Variable	Unit	Min.	Mean	Max.
Depth (m relative to mean sea level)	m	1.1	4.5	9.4
Surface temperature	°C	17.8	20.4	23.0
Bottom temperature	°C	18.2	20.4	22.6
Surface salinity	PSU	21.2	30.3	35.2
Bottom salinity	PSU	22.7	31.2	35.0
Mean current velocity	m/s	0.15	0.36	0.65
Maximum current velocity	m/s	0.46	0.89	1.52

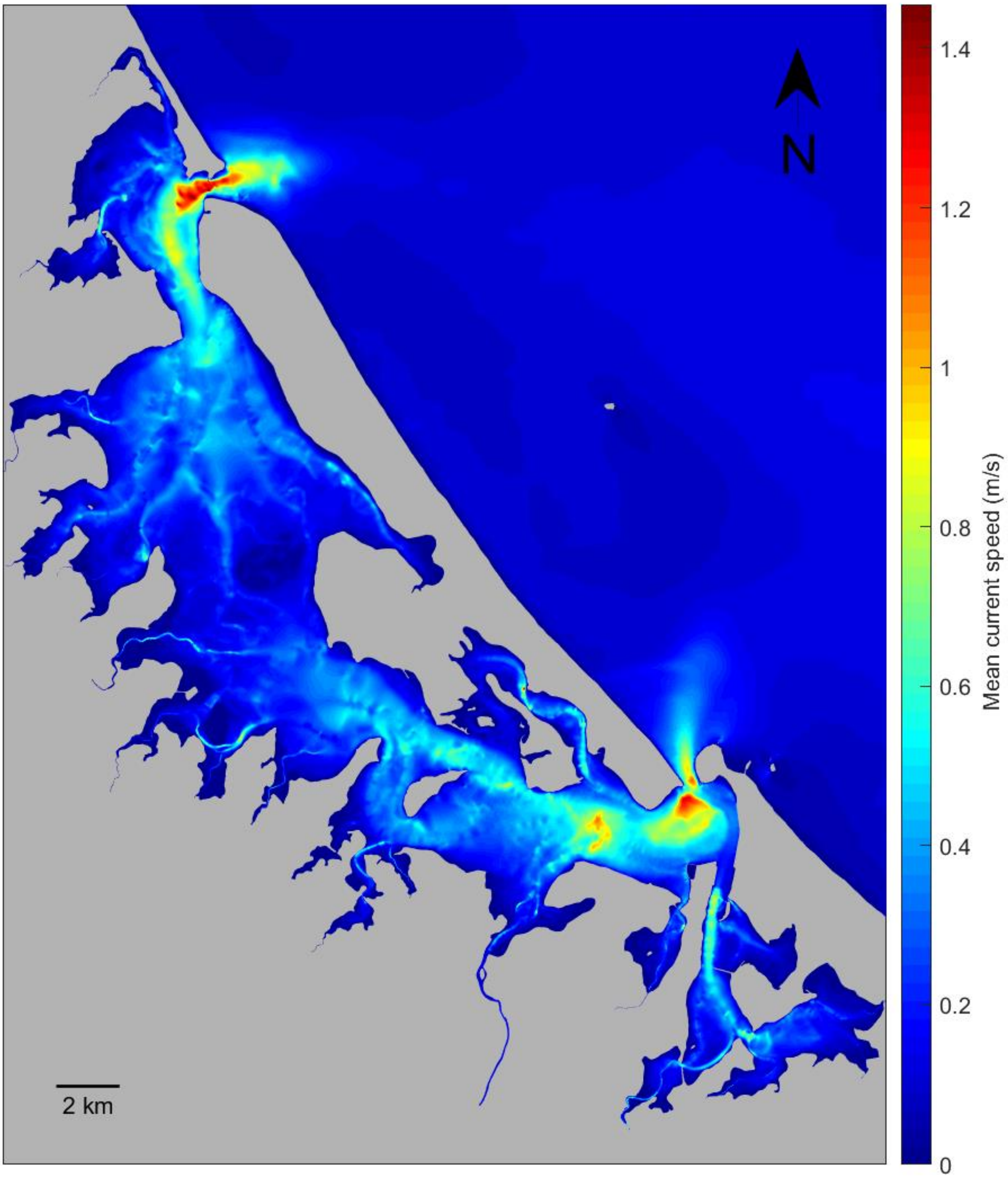

Figure 2. Modelled current velocities for Tauranga Harbour using the mean over a 14-day period.

3.2. Sediment physico-chemical variables

Table 5 provides a summary of all sediment physico-chemical parameters measured during the subtidal survey and Appendix 5 contains results in full. Similar to intertidal areas of Tauranga Harbour, subtidal sediments were predominantly sandy (67-97% sand), however, mud concentrations (< 23.4%) were generally lower than in intertidal sediments (Table 5; Figure 3). Organic content in subtidal sediments was comparable to intertidal sediments with greater enrichment in upper channel areas. Mean organic content was 2.5% LOI and 0.2% TOC, with maximum values of 6.2% and 0.9%, respectively (Table 5; Figure 4). There was a high correlation between LOI and TOC (r = 0.86) suggesting that both analyses are appropriate measures of organic content.

Table 5. Minimum, mean and maximum values for all sediment physico-chemical variables measured in Tauranga Harbour 15 March to 4 May 2016. Australian and New Zealand Environment and Conservation Council Interim Sediment Guidelines (ISQG-low) for metals are shown for comparison (ANZECC, 2000). TOC = total organic content, LOI = loss on ignition, TN = total nitrogen, TP = total phosphorus, Cu = copper, Pb = lead, Zn = zinc, As = arsenic, Cd = cadmium, Cr = chromium, Hg = mercury, Ni = nickel, Chl a = chlorophyll a, NA = not applicable.

Variable	Unit	Minimum	Mean	Maximum	ISQG-low
Mud	%	2.4	7.2	23.4	NA
Sand	%	67.4	87.6	97.0	NA
Gravel	%	< 0.1	5.2	17.8	NA
TOC	%	< 0.05	0.2	0.9	NA
LOI	%	1.0	2.5	6.2	NA
TN	mg/kg	< 500	342	1200	NA
TP	mg/kg	79	139	340	NA
Cu	mg/kg	0.3	1.0	3.5	65.0
Pb	mg/kg	1.4	4.5	6.4*	50.0
Zn	mg/kg	7.7	16.5	37.0	200.0
As	mg/kg	1.9	4.8	7.1	20.0
Cd	mg/kg	< 0.010	0.023	0.063	1.5
Cr	mg/kg	1.2	3.8	6.6	80.0
Hg	mg/kg	< 0.010	0.019	0.074	0.150
Ni	mg/kg	0.4	1.1	2.1	21.0
Chl *a*	mg/kg	20.0	18.8	56.3	NA

* A Pb value of 77 mg/kg was recorded, but this was deemed to be an outlier. See methods Section 2.3.3.for details.

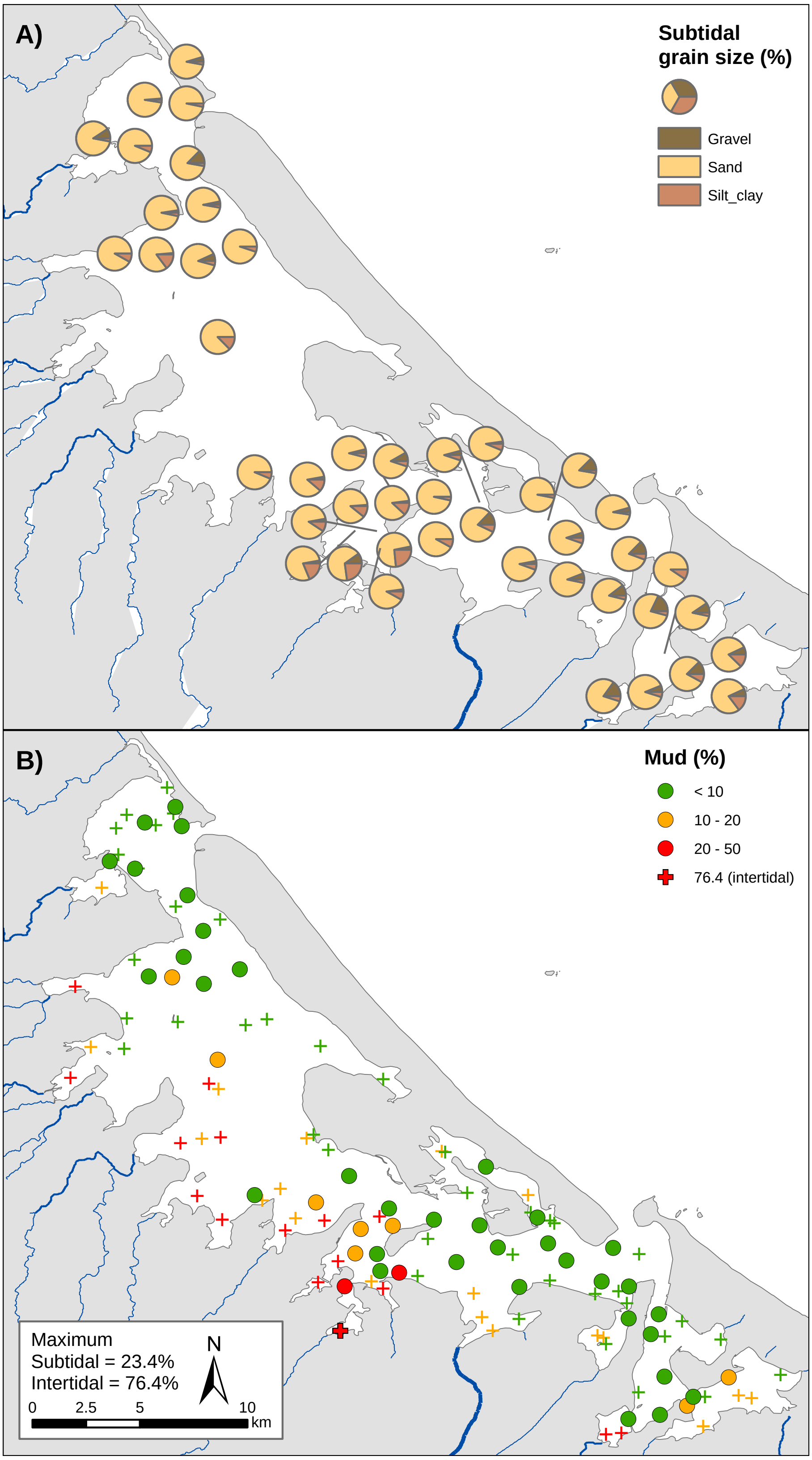

Figure 3. Map of Tauranga Harbour showing the distribution of grain size across the 45 subtidal sites sampled in 2016 (circles) with the 75 intertidal sites sampled in 2011/12 (crosses) for comparison. A) grain size, B) percentage mud. Major rivers and streams entering the harbour are shown in blue. Refer to Figure 1 for site numbers.

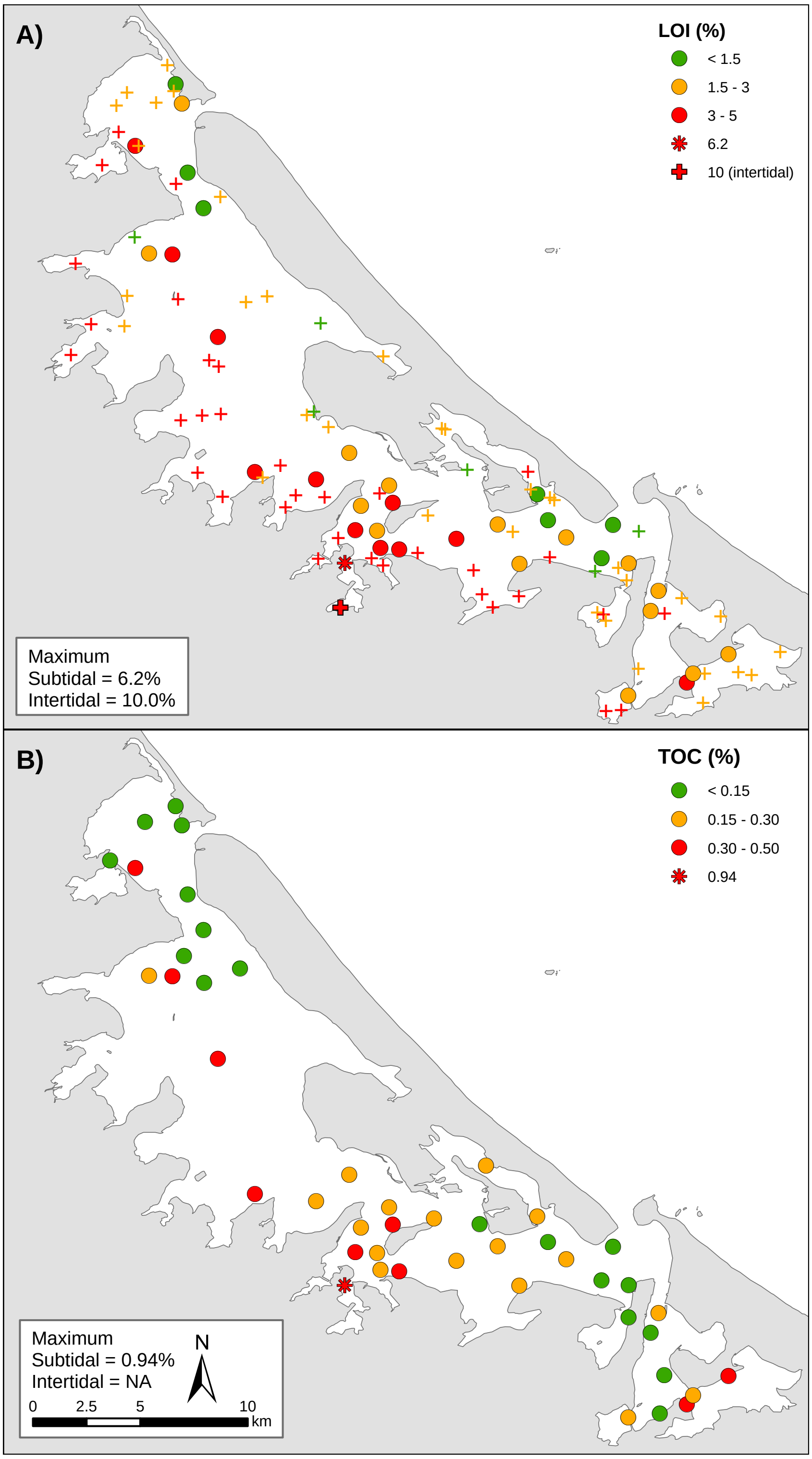

Figure 4. Map of Tauranga Harbour showing organic content across the 45 subtidal sites sampled in 2016 (circles) with the 75 intertidal sites sampled in 2011/12 (crosses) for comparison. A) loss on ignition (LOI), B) total organic content (TOC). TOC was not measured during the 2011/12 intertidal survey. Refer to Figure 1 for site numbers.

Nutrient concentrations (i.e. TN and TP) were slightly lower in subtidal sediments than observed in intertidal sediments (Table 5; Figure 5; Figure 6). Subtidal TN ranged from < 500 to 1200 mg/kg and TP from 79 to 340 mg/kg. The ADL for TN (500 mg/kg) used for the subtidal survey, was much higher than that for the intertidal survey meaning differentiation between lower TN levels was not possible (76% of subtidal sites). As with mud and organic content, upper reaches of the channels tended have higher nutrient concentrations than sites closer to the main channels. Site 20, in the Te Puna sub-estuary (Figure 1), had particularly high nutrients levels (TN = 1200 mg/kg and TP = 340 mg/kg) relative to other sites.

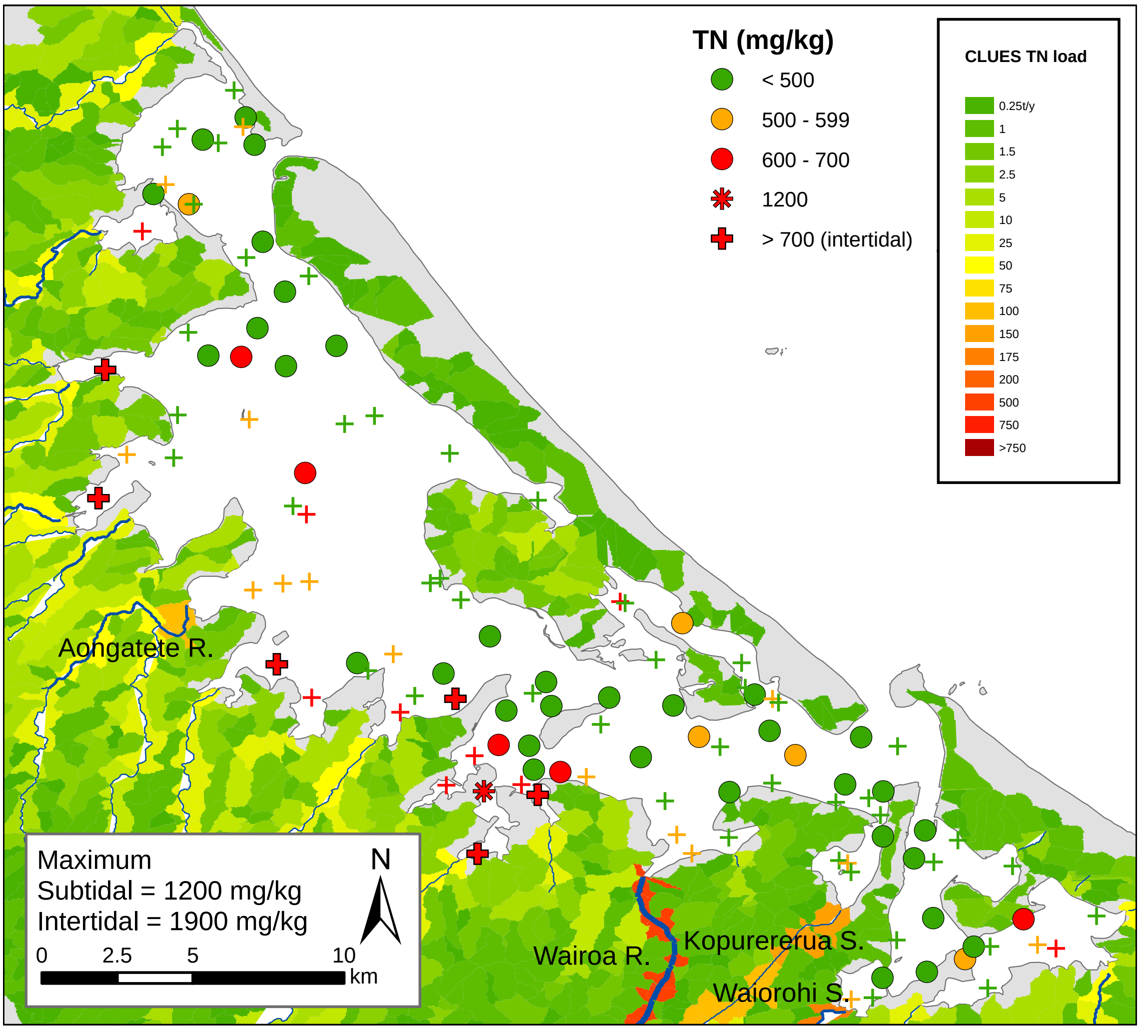

Figure 5. Map of Tauranga Harbour showing total nitrogen (TN) concentrations across the 45 subtidal sites sampled in 2016 (circles) with the 75 intertidal sites sampled in 2011/12 (crosses) for comparison. Major rivers and streams entering the harbour are shown in blue and catchment colours indicate modelled nitrogen loading (estimated from CLUES; Plew, Zeldis, Shankar, & Elliot, 2015). Refer to Figure 1 for site numbers.

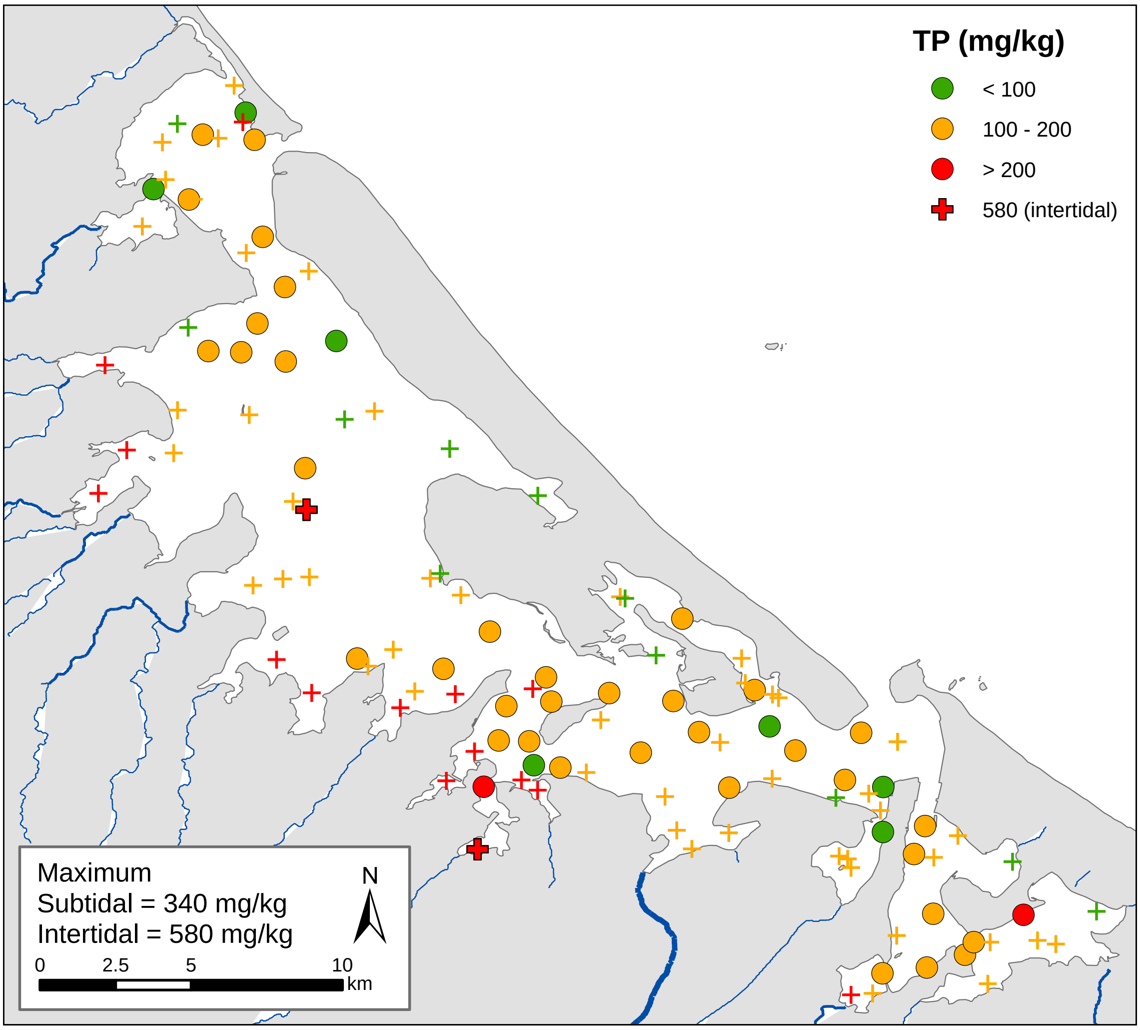

Figure 6. Map of Tauranga Harbour showing total phosphorus (TP) concentrations across the 45 subtidal sites sampled in 2016 (circles) with the 75 intertidal sites sampled in 2011/12 (crosses) for comparison. Major rivers and streams entering the harbour are shown in blue. Refer to Figure 1 for site numbers.

Metal concentrations (i.e. Cu, Zn and Pb) were lower in subtidal sediments than in intertidal sediments (Table 5; Figure 7). All metals were well below Australian and New Zealand Environment and Conservation Council (ANZECC) Interim Sediment Quality Guidelines (ISQG; ANZECC, 2000), with the exception of Pb at Site 36. Pb concentrations at this site (77 mg/kg) were above the low-ISQG of 50 mg/kg, which provides a threshold for possible biological effects. However, this high Pb value was deemed to be an outlier (refer Section 2.3.3 for details). Sites with the highest subtidal sediment Cu, Pb and Zn concentrations were situated in the southern harbour around the Tauranga Bridge marina, Rangataua Bay and the Te Puna sub-estuary/Omokoroa area.

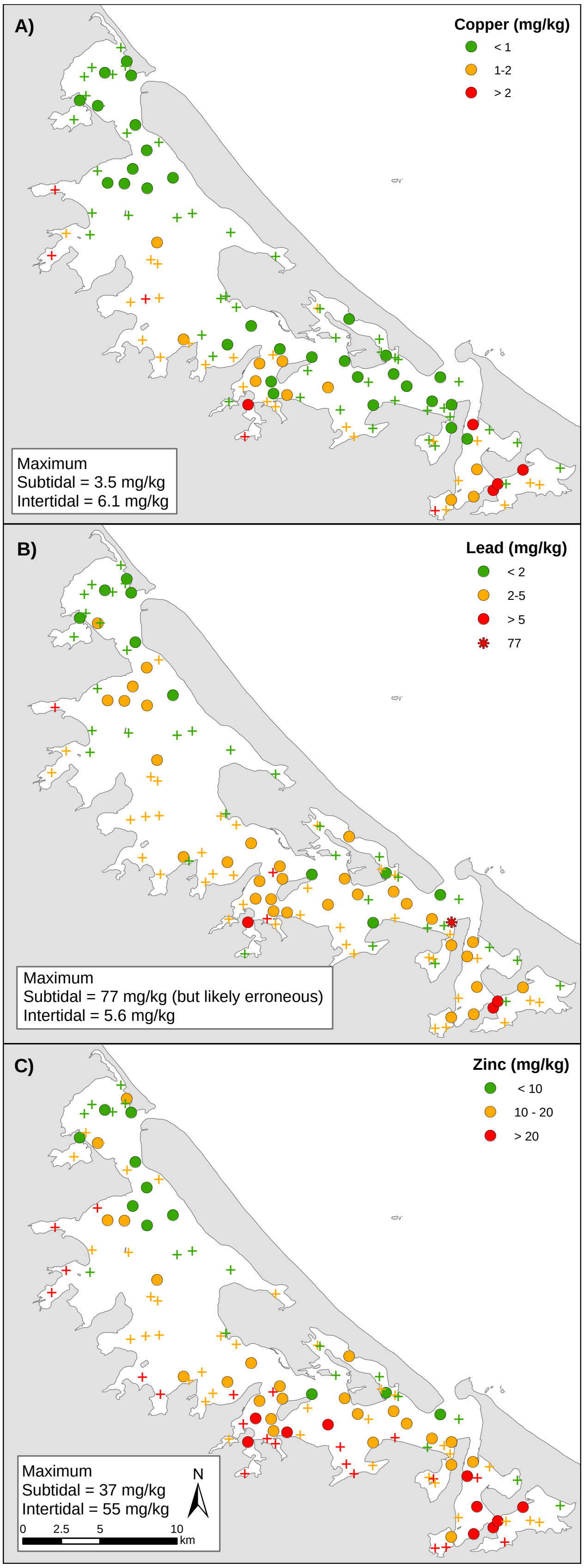

Figure 7. Map of Tauranga Harbour showing concentrations of three key metals measured at 45 subtidal sites in 2016 (circles) with the 75 intertidal sites sampled in 2011/12 (crosses) for comparison. A) copper (Cu), B) lead (Pb), C) zinc (Zn). Refer to Figure 1 for site numbers.

Chl *a* concentrations were higher in subtidal sediments than intertidal sediments, with an average of 19 mg/kg and a maximum of 56 mg/kg (Table 5; Figure 8). The main channel of the southern harbour had the highest concentrations, with maximum Chl *a* levels observed in its upper reaches at Sites 16 and 17 (Figure 1). In the northern harbour Site 5, near Tuapiro Point, had relatively high Chl *a* levels (32 mg/kg). No significant correlation was found between sediment Chl *a* and TN (r = 0.07) or TP (r = 0.19).

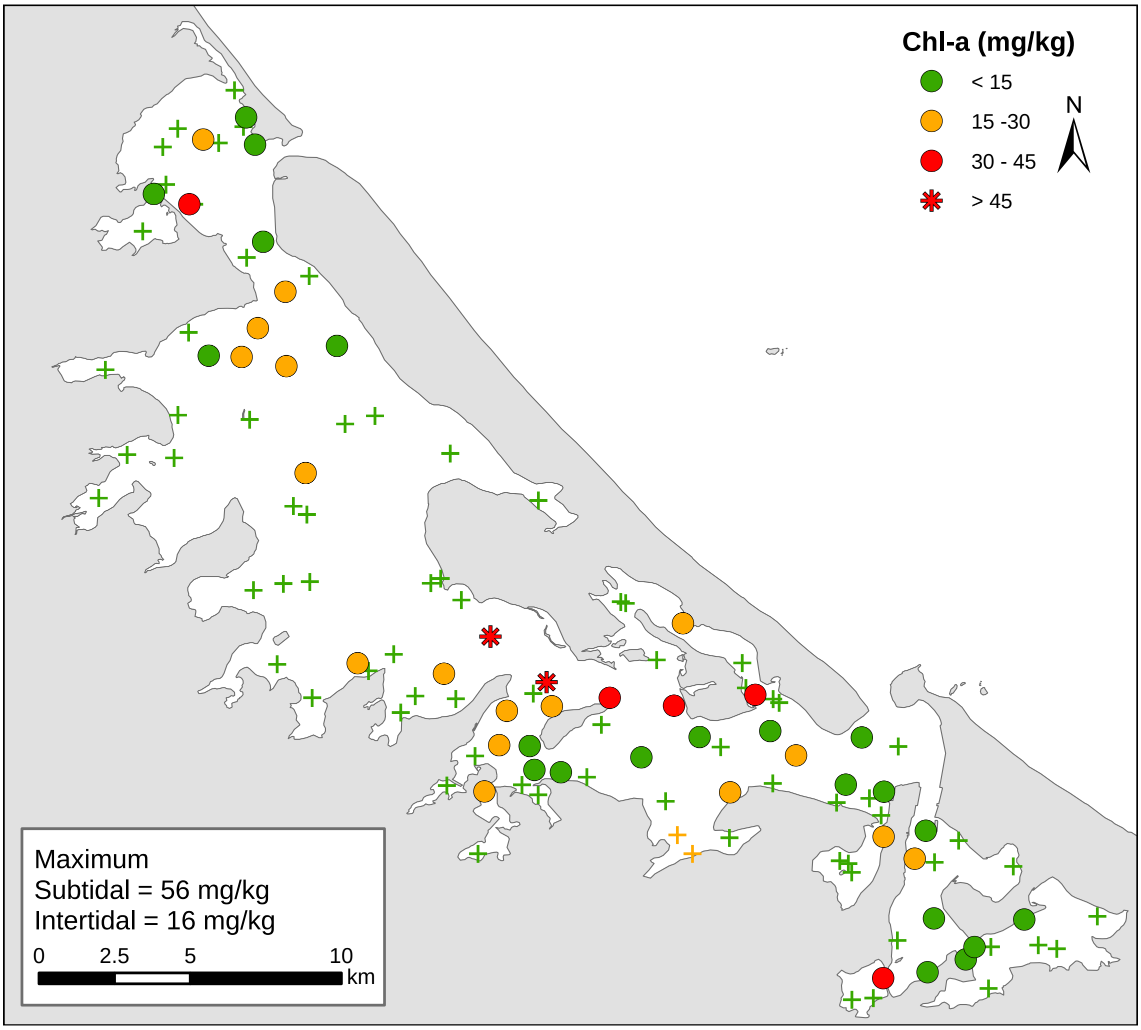

Figure 8. Map of Tauranga Harbour showing chlorophyll a (Chl a) concentrations across the 45 subtidal sites sampled in 2016 (circles) with the 75 intertidal sites sampled in 2011/12 (crosses) for comparison. Refer to Figure 1 for site numbers.

3.3. Benthic macrofauna

3.3.1 Patterns in taxa richness and total abundance

Overall, two hundred taxa were found in subtidal portions of the harbour. Taxa richness per site (n = 3) ranged from 11 to 55 taxa, with an average of 33 (Figure 9). Maximum taxa richness (> 43 taxa) was found in the central channel, near Oikimoko Point, Te Puna Beach, Motuhoa Island and

Rangiwaea Island, while the lowest richness (< 19 taxa) was found in the southern portion of the harbour between Tauranga City and Motuhoa Island.

Average total abundance at subtidal sites ranged from 26 to 785 individuals and averaged 214 individuals per site (Figure 9). Sites with particularly high or low abundances were not concentrated in a specific portion of the harbour. High abundances were primarily driven by large numbers of dominant taxa, such as amphipods (including Corophiidae amphipods), oligochaete worms and polydorid polychaete worms (Table 6). Both species richness and abundances were generally higher in the subtidal than the intertidal (average intertidal species richness of 24 taxa and abundance of 118 individuals[4]).

[4] Intertidal species richness and abundance values vary slightly from the Ellis et al. (2013) report because taxa lumping was adjusted to ensure consistency between the intertidal and subtidal datasets.

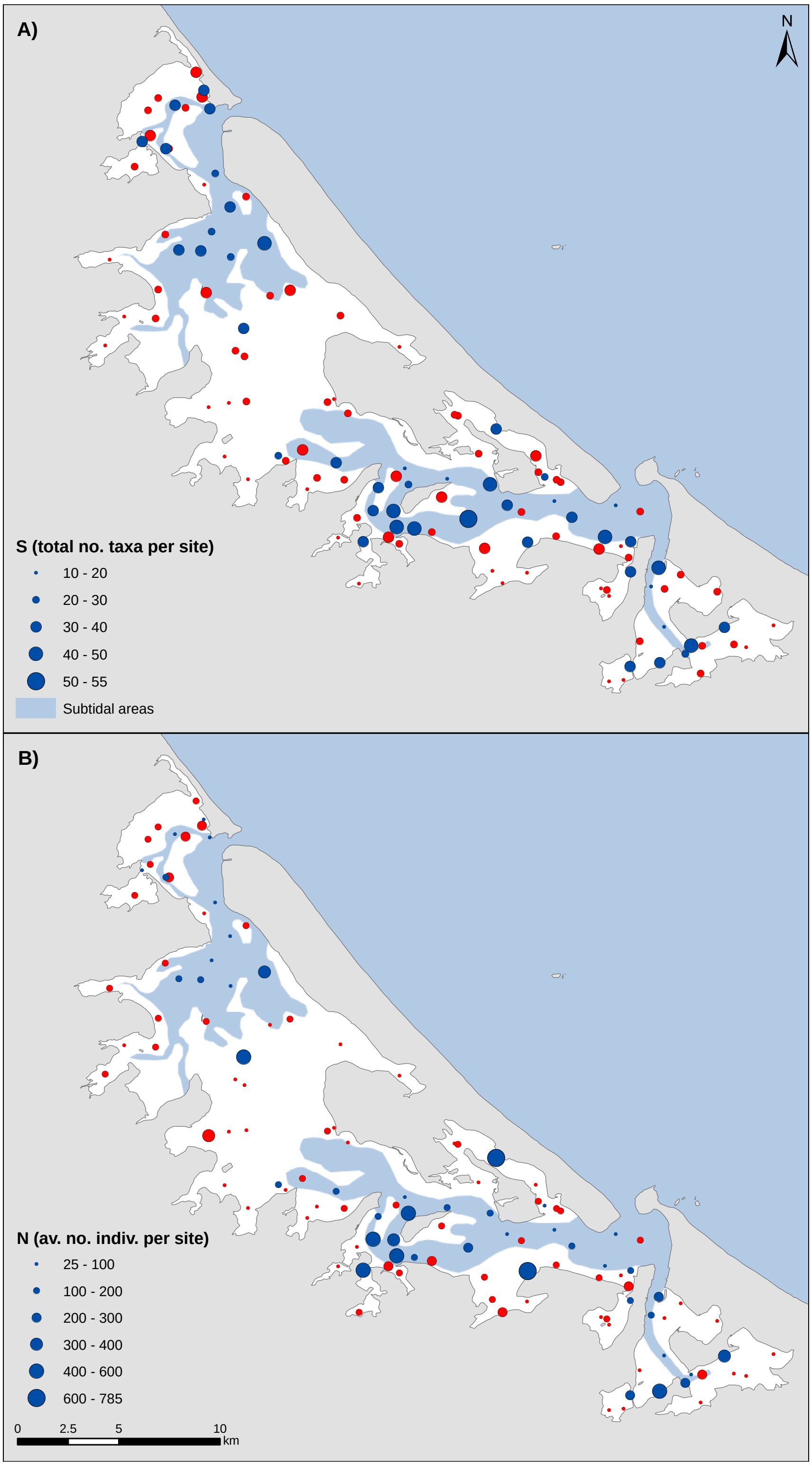

Figure 9. Map of Tauranga Harbour showing A) species richness (total number of taxa per site) and B) abundance (average number of individuals per site) across the 44 subtidal sites sampled in 2016 (blue) with the 75 intertidal sites sampled in 2011/12 (red) for comparison. Refer to Figure 1 for site numbers.

Table 6. Top 10 most abundant taxa in the 2016 subtidal survey of Tauranga Harbour (± standard error).

Taxa	Group	Average abundance (± SE)
Polydorid	Polychaete worm	30.6 (± 6.5)
Oligochaeta	Oligochaete worm	39.9 (± 10.3)
Corophiidae	Amphipod	18.1 (± 4.9)
Aricidea sp.	Polychaete worm	17.4 (± 2.5)
Amphipoda	Amphipod	14.9 (± 6.1)
Heteromastus filiformis	Polychaete worm	14.7 (± 1.8)
Paraonidae	Polychaete worm	8.5 (± 1.4)
Paphies australis	Bivalve (pipi)	7.3 (± 2.6)
Cumacea	Cumacean	6.7 (± 1.4)
Exogoninae	Polychaete worm	5.7 (± 0.8)

3.3.2 Distribution of key shellfish

Cockles, wedge shells and pipi were distributed across a similar number of sites (27-30% of subtidal sites), with nut shells more widely distributed (present at 41% of subtidal sites). Pipis had the highest average abundance (6.6 individuals per site ± 1.0 SE) with a maximum average abundance of 128 (± 46 SE) pipi at a site in the channel near Waipu Bay (Figure 10). Average abundances of cockles and nut shells were similar (0.8 ± 0.1 SE and 0.9 ± 0.1 SE individuals, respectively), with cockles particularly abundant at a shallow site near Tilby Point (mean = 26 individuals, ± 12 SE) and nut shells abundant in the channel off Otumoetai (mean = 21 individuals, ± 9 SE). Wedge shells were rare in subtidal areas of the harbour (mean = 0.2 ± 0.03 SE and maximum = 1.6 individuals per site).

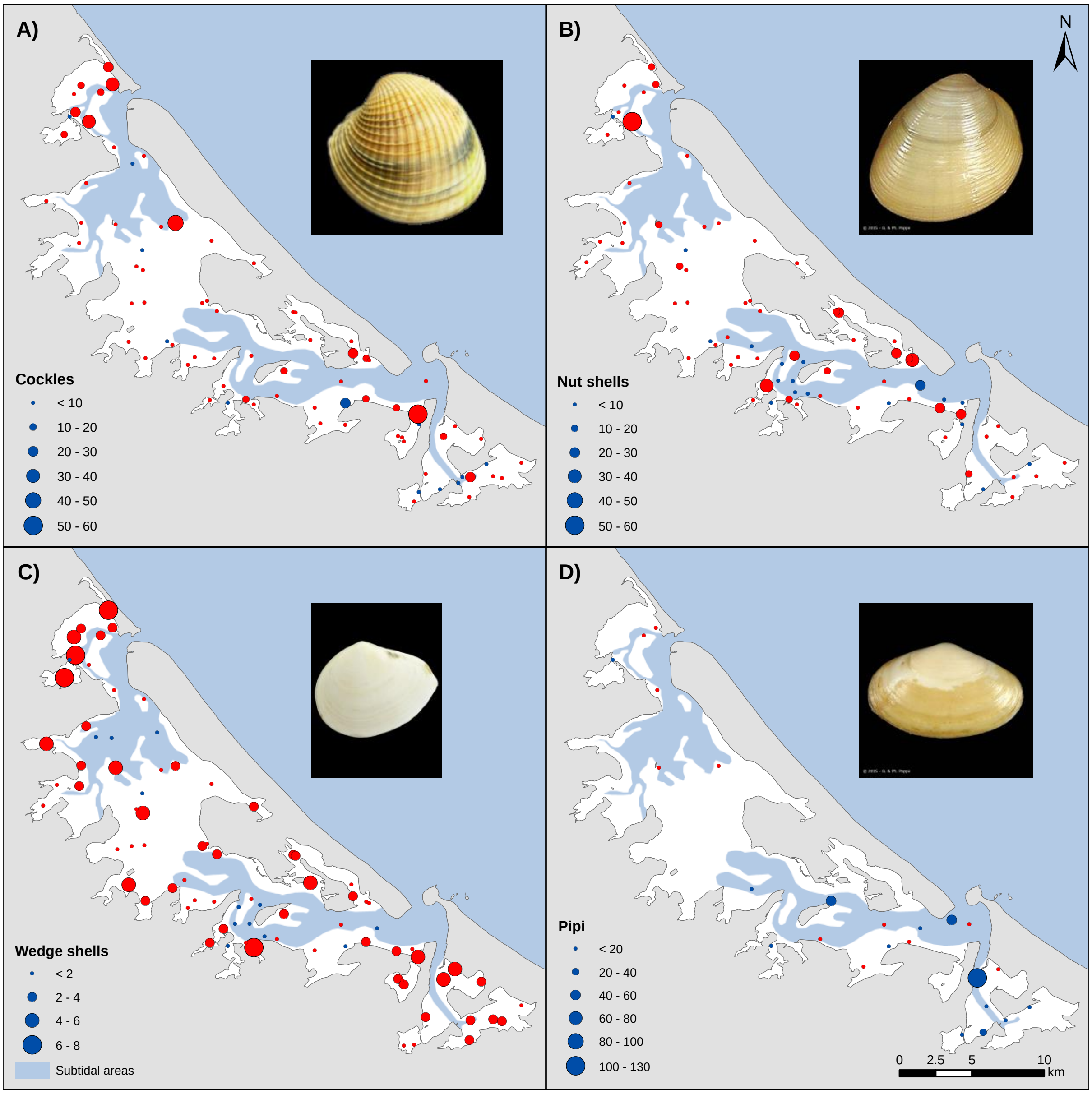

Figure 10. Map of Tauranga Harbour showing shellfish abundances (> 1 mm) across the 45 subtidal sites sampled in 2016 (blue) with the 75 intertidal sites sampled in 2011/12 (red) for comparison. A) cockles (Austrovenus stutchburyi), B) nut shells (Linucula hartvigiana), C) wedge shells (Macomona liliana), D) pipi (Paphies australis). Refer to Figure 1 for site numbers. Note the difference in scale between species.

3.3.3 Patterns in overall community structure

The nMDS plot showed no clear difference in community structure between sites, except Sites 25, 33 and 39 that appeared to be more separate from the others (Figure 11). These three sites had considerably higher numbers of pipi (average 43-163 pipi per site, ± 1.5-63.2 SE) than the other sites (average of 0-19 pipi per site) and were associated with relatively high mean current velocities (0.5-0.6 m/s).

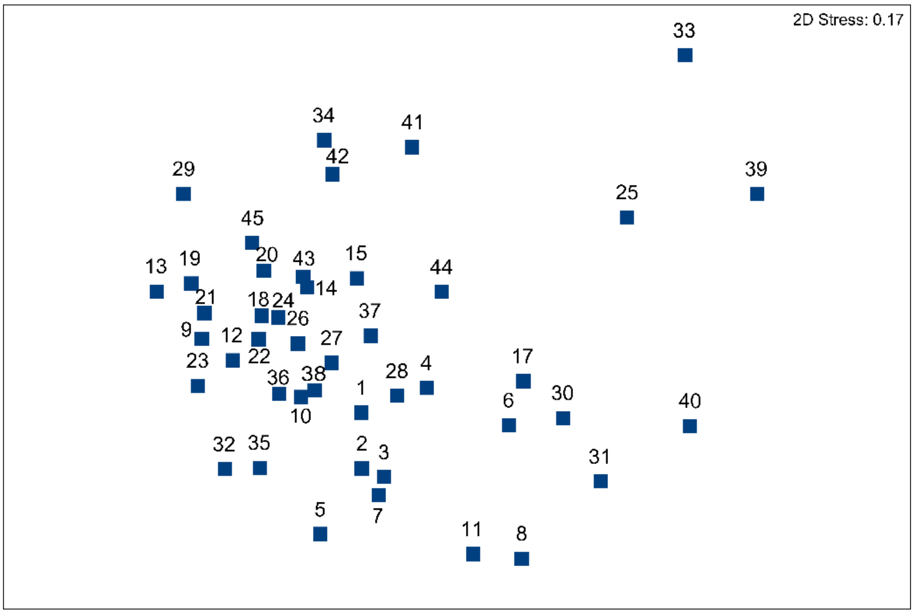

Figure 11. Non-metric multidimensional scaling (nMDS) showing differences in macrofaunal community structure, based on Bray-Curtis similarities between site centroids using the square-root transformed abundance data of subtidal sites.

There was a clear difference in the structure of macrofaunal communities inhabiting subtidal and intertidal portions of the harbour (Figure 12). SIMPER analysis showed that overall dissimilarity between the intertidal and subtidal was 79% and this was driven by differences in the abundances of a range of taxa including polychaetes (*Aricidea* sp., Exogoninae, Paraonidae, polydorids, Nereididae juveniles, *Prionospio aucklandica, Heteromastus filiformis*), nematodes, amphipods (including Corophiidae and Phoxocephalidae) and bivalves (*Macomona liliana, Linucula hartvigiana, Austrovenus stutchburyi;* Table 7). Intertidal communities had greater proportions of bivalves, sea anemones, crustaceans and gastropods than subtidal communities, which had greater proportions of polychaetes, sea stars and urchins (Figure 13).

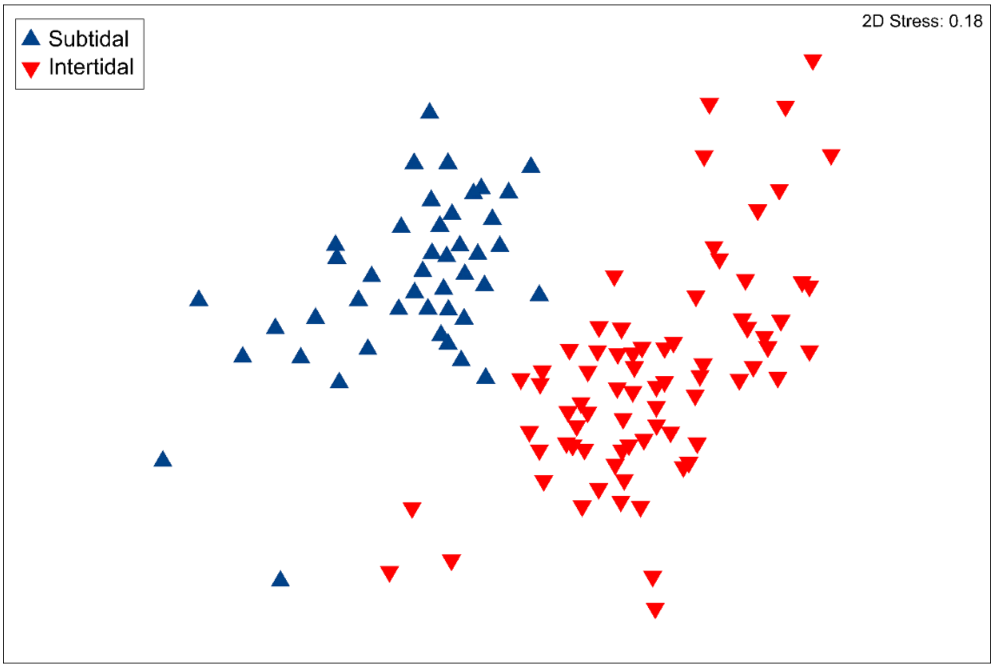

Figure 12. Non-metric multidimensional scaling (nMDS) illustrating differences in of subtidal and intertidal macrofaunal community structure, based on Bray-Curtis dissimilarities between site centroids of the square-root transformed abundance data.

Table 7. Similarity percentage (SIMPER) results comparing subtidal and intertidal macrofaunal communities based on square-root-transformed macrofaunal abundance data. Differences between groups are shown to a 40% level. Overall dissimilarity between the intertidal and subtidal was 79%. Av. Abund = average abundance, Av. Diss = average dissimilarity, Diss/SD = ratio of average contribution divided by standard deviation, Contrib. % = percent contribution, Cum. % = cumulative percent contribution.

Species	Subtidal Av. Abund	Intertidal Av. Abund	Av. Diss	Diss/SD	Contrib. %	Cum.%
Aricidea sp.	1.41	0.40	2.07	1.40	2.79	2.79
Exogoninae	1.19	0.00	2.03	1.96	2.73	5.52
Nematoda	1.08	0.03	1.94	1.72	2.61	8.13
Paraonidae	1.25	0.19	1.89	1.71	2.54	10.66
Polydorid	1.32	0.41	1.88	1.17	2.52	13.19
Nereididae (juvenile)	0.39	1.25	1.80	1.40	2.42	15.61
Corophiidae	0.88	0.53	1.69	0.88	2.28	17.89
Prionospio aucklandica	0.82	1.29	1.69	1.28	2.27	20.16
Amphipoda	1.16	0.32	1.69	1.44	2.27	22.43
Macomona liliana	0.21	1.05	1.60	1.61	2.15	24.58
Heteromastus filiformis	1.52	1.44	1.60	1.16	2.15	26.73
Phoxocephalidae	0.66	1.10	1.57	1.25	2.11	28.84
Linucula hartvigiana	0.40	1.02	1.55	1.22	2.08	30.93
Austrovenus stutchburyi	0.29	1.02	1.54	1.23	2.07	33.00
Oligochaeta	1.66	1.16	1.52	1.15	2.04	35.03
Scolecolepides benhami	0.02	0.77	1.43	1.25	1.92	36.95
Ostracoda	0.94	0.22	1.41	1.50	1.90	38.85
Armandia maculata	0.82	0.12	1.33	1.32	1.79	40.64

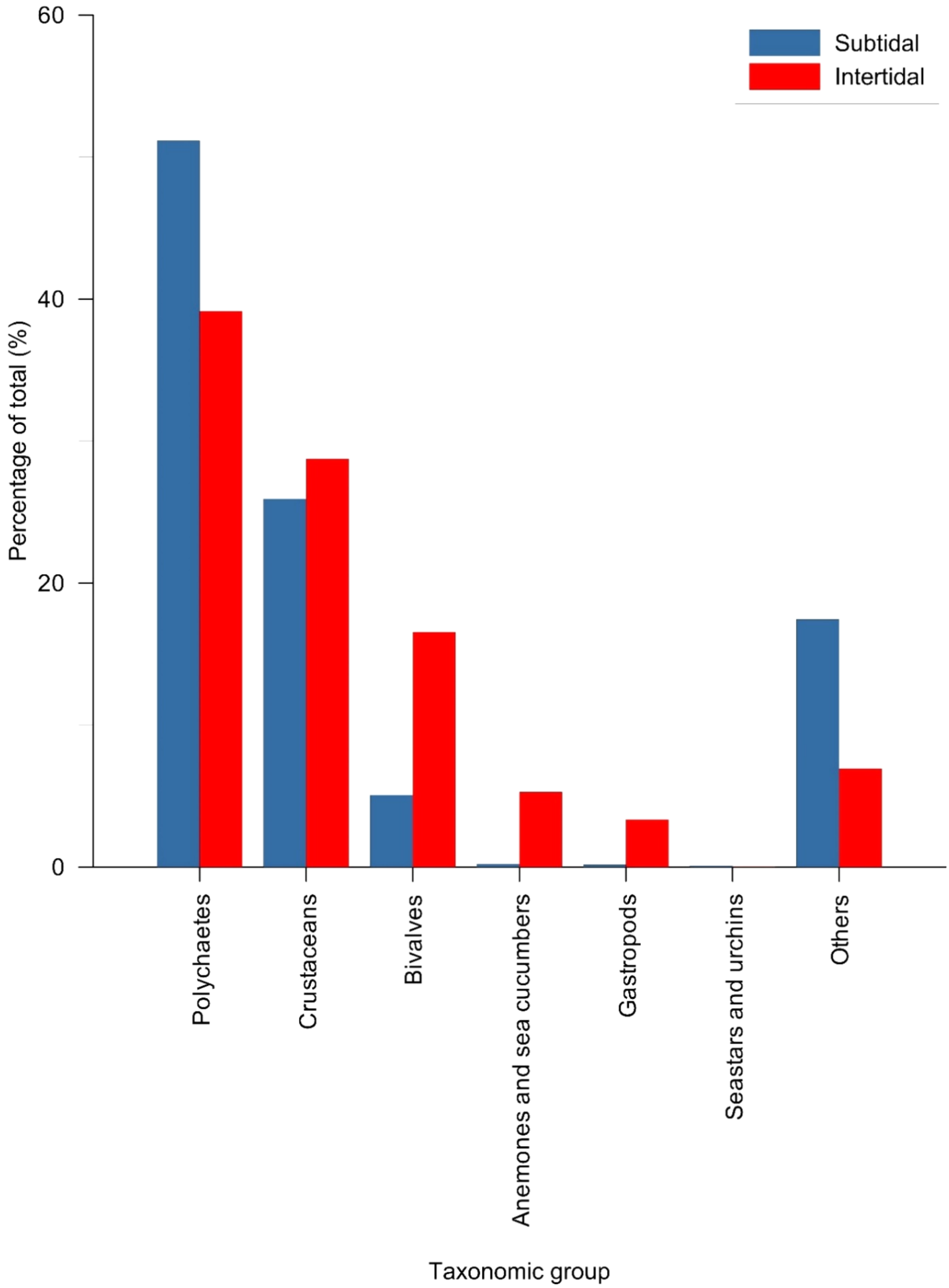

Figure 13. Comparison of proportions of different groups between intertidal and subtidal macrofaunal communities sampled across Tauranga Harbour in 2011/12 and 2016. Taxonomic groups were chosen to facilitate comparison with Park and Donald's 1990/91 survey.

3.4. Benthic Health Models

3.4.1 Model development

Marginal tests from DistLM selected mud (12.8%), TOC (10.7%), LOI (8.6%), mean current velocity (8.5%) and PC1 metals (7.5%) as explaining the most variation in benthic community structure (Table 8). In the sequential tests, mud was identified as explaining the most variation (12.8%) in benthic community structure, followed by mean current velocity and PC1 metals, with just over 21% of variation in the benthic communities explained by these three variables (Table 8).

Table 8. Results of the distance-based linear modelling (DistLM) based on Bray-Curtis dissimilarities of the square-root transformed data showing the percentage variation (% var) and cumulative variation (% cumulative var) explained for each variable individually (marginal) and variables together (sequential), identified using a forward selection based on the Akaike's Information Criterion (AIC). Only significant variables (p < 0.05) are displayed.

Marginal tests	*p* value	% var	
Mud	0.0001	12.8	
TOC	0.0001	10.7	
LOI	0.0001	8.6	
Mean current velocity	0.0001	8.5	
PC1 metals	0.0004	7.5	
Bottom salinity	0.0036	6.1	
Gravel	0.0114	5.4	
TN	0.0373	4.1	
Sequential tests	***p* value**	**% var**	**% cumulative var**
Mud	0.0001	12.8	12.8
Mean current velocity	0.0046	4.5	17.2
PC1 metals	0.0138	4.0	21.2

A series of sequential DistLMs, which varied the order of the input variables (Table 9), showed that despite some overlap in explained variation between mud and currents (4.1%)[5], a BHM based on mud should be able to partition out the effects of mud from currents because this variable always came out as explaining more variation in benthic community structure than currents. Although current velocity explained more variation in community structure than metals, the overlap in variation explained between these two variables was small (1.6%)[6].

[5] Value derived by subtracting the proportion of variation explained by mud when mean current velocity was the first variable (8.7%) from the proportion of variation explained by mud when mud was the first variable (12.8%).
[6] Value derived by subtracting the proportion of variation explained by mean current velocity when PC1 metals was the first variable (6.9%) from the proportion of variation explained by mean current velocity when mean current velocity was the first variable (8.5%).

Table 9. Results of a series of distance-based linear models (DistLM) based on Bray-Curtis dissimilarities of the square-root transformed data showing the percentage variation (% var) explained for all variables together (sequential) identified using specified selection based on the Akaike's Information Criterion (AIC).

Sequential tests	% var		
Variable order	Mean current velocity	Mud	PC1 metals
Mean current velocity, Mud, PC1 metals	8.5	8.7	4.0
Mean current velocity, PC1 metals, Mud	8.5	6.8	5.8
Mud, PC1 metals, Mean current velocity	4.4	12.8	4.0
Mud, Mean current velocity, PC1 metals	4.5	12.8	4.0
PC1 metals, Mud, Mean current velocity	4.4	9.3	7.5
PC1 metals, Mean current velocity, Mud	6.9	6.8	7.5

CAP analysis based on mud resulted in a canonical correlation of 0.81 where *m* (the number of PCO axes used for the analysis) was equal to 6 (Figure 14A; Table 10). The squared canonical correlation for the canonical axis (δ_1), or coefficient of determination (R^2), was 0.66. The proportion of the total variation in the dissimilarity matrix explained by the first PCO axis was 0.60. The permutation test (9,999 permutations) indicated that the correlation between CAP scores and the mud gradient was significantly different from zero ($p < 0.0001$). These results indicate that the BHM is suitable to describe the effects of mud on macrofaunal communities in Tauranga Harbour.

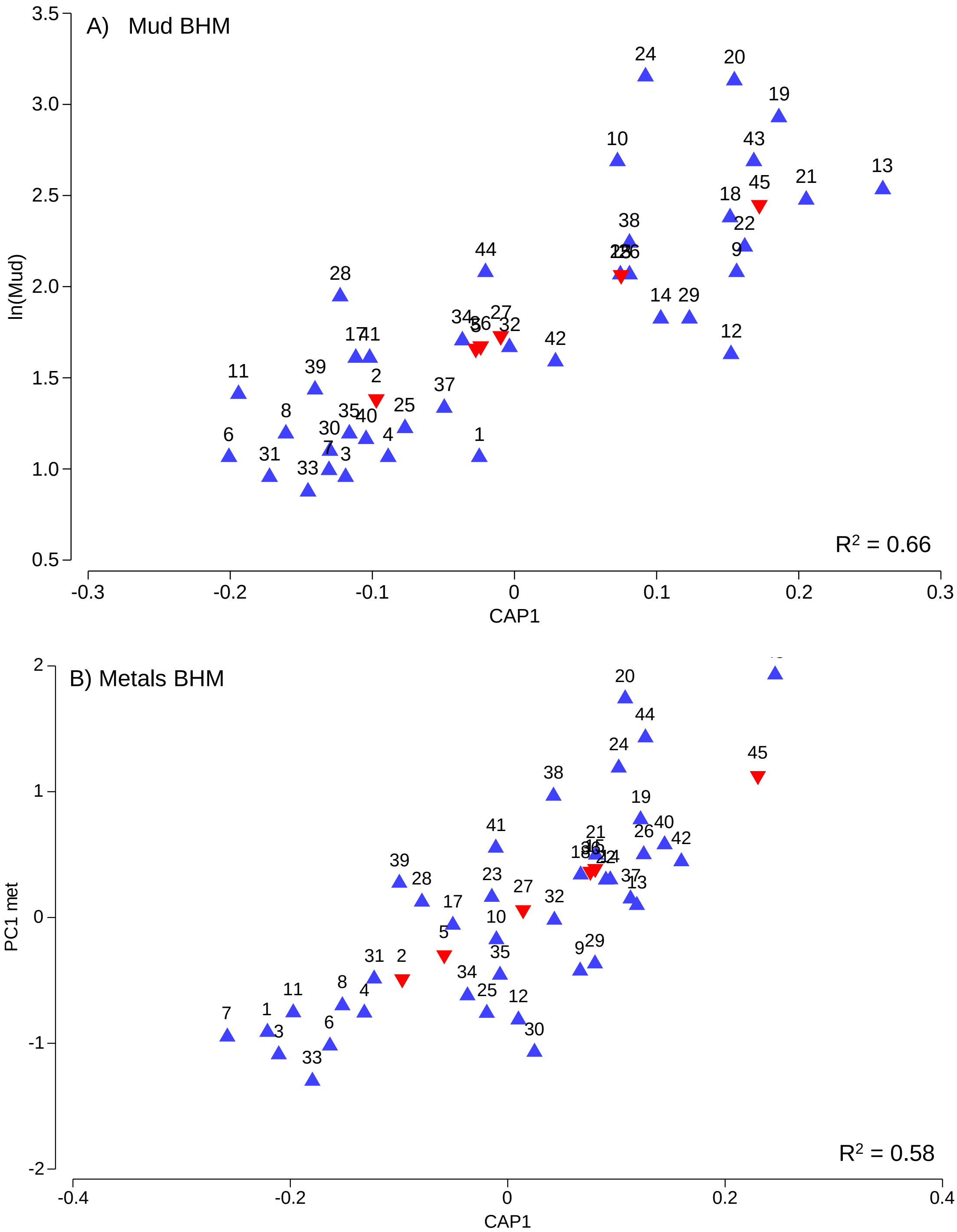

Figure 14. Benthic Health Models (BHMs) based on canonical analysis of principal coordinates (CAP) for mud (A) and metal loading (B), based on data from 38 subtidal sites in Tauranga Harbour. Sites used to develop the model are shown in blue and six new sites placed into the model to validate its success are shown in red. R^2 is the coefficient of determination for each model.

Table 10. Summary of canonical analysis of principal coordinates (CAP) models used to develop the mud and metals Benthic Health Models (BHMs). Corr. = correlation between the canonical axis and the pollution gradient (success of model fit). Corr. sq. (δ_1) = squared canonical correlation for the canonical axis (success of model fit). Prop. G = proportion of the total variation in the dissimilarity matrix explained by the first m principal coordinates analysis (PCO) axes. m = no. of PCO axes used for the analysis. SS_{RES} = the leave-one-out residual sum of squares (smaller is better).

	Corr.	**Corr. sq.**	***m***	**Prop. G**	**SS_{RES}**
Mud BHM	0.81	0.66	6	0.60	0.42
Metals BHM	0.76	0.58	10	0.76	0.72

CAP analysis based on metal concentrations (PC1 metals) resulted in a canonical correlation of 0.76 where *m* (the number of PCO axes used for the analysis) was equal to 10 and the proportion of the total variation in the dissimilarity matrix explained by the first PCO axes was 0.76 (Figure 14B; Table 10). The squared canonical correlation for the canonical axis (δ_1), or coefficient of determination (R^2), was 0.58. The permutation test (9,999 permutations) indicated that the correlation between CAP scores and the metals gradient was significantly different from zero (p = 0.0032). These results suggest that the BHM model is suitable to determine potential effects of metals (Cu, Pb, Zn) on macrofaunal subtidal benthic communities in Tauranga Harbour, despite the relatively low concentrations covered by the gradient in metals (Table 5).

3.4.2 Splitting up the gradient

As the mud gradient (ln % mud) was a continuous single variable gradient, and no indication of clustering structure was evident, it was decided that evenly splitting the gradients into five groups would be most appropriate. Five groups were chosen as a number that provides the ability to discriminate between sites but is not overwhelming for managers.

Clearer clustering was apparent along the metal loading gradient (PC1 metals) and both non-hierarchal clustering methods (*k*-means and *k*-*R* CLUSTERING) identified three as the optimal number of groups with the same sites allocated in each group by both methods. The classification boundaries for the mud groups were defined as five even splits of the mud gradient, while metal groups were defined as being half-way between the highest value obtained along the PC1 metals axis for one group and the lowest value obtained along the axis for the next group (Table 11). The boundaries of the groups were then converted into CAP scores and used as cut-off points for the BHM groups. Appendix 5 details the CAP scores and groups for each of the sites.

Just over 65% of sites were ranked in Mud BHM Group 3 or less, suggesting fairly healthy communities with regard to mud impacts (Figure 15). Subtidal areas impacted by mud were in the upper reaches of the harbour channels. Macrofaunal communities were 76% dissimilar between Group 1 and Group 5, with Group 5 communities characterised as having higher abundances of certain polychaetes (polydorids, *Aricidea* sp., *Heteromastus filiformis*, Paraonidae, *Pseudopolydora* sp.), amphipods (including Corophiidae) and Oligochaetes than sites in Group 1 and less pipi (refer Appendix 6 for full results).

Table 11. Boundaries for classification groups along each of the environmental gradients for the mud and metals Benthic Health Models (BHMs).

	Mud BHM					Metals BHM				
	ln(% mud)		CAPmud				PC1 metals		CAPmetals	
Group	Min	Max	Min	Max		Group	Min	Max	Min	Max
1	0.88	< 1.33	0.147	< 0.072		1	-1.300	-0.291	0.152	< 0.033
2	≥ 1.33	< 1.79	≥ 0.072	< -0.004		2	≥ -0.291	0.873	≥ 0.033	< -0.104
3	≥ 1.79	< 2.24	≥ -0.004	< -0.078		3	≥ 0.873	1.930	≥ -0.104	≤ -0.228
4	≥ 2.24	< 2.70	≥ -0.078	< -0.155						
5	≥ 2.70	≤ 3.15	≥ -0.155	≤ -0.229						

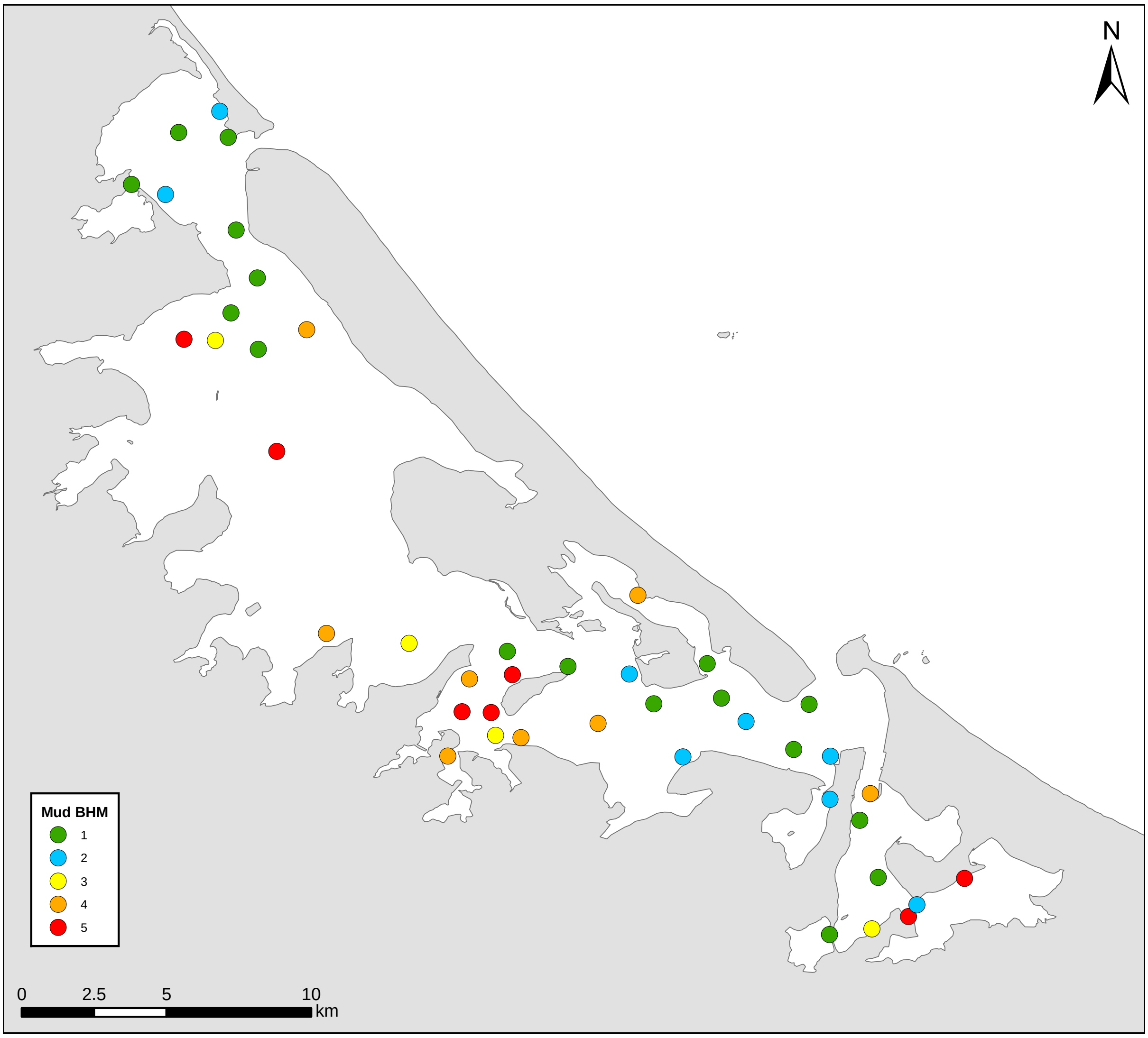

Figure 15. Map of Tauranga Harbour showing the allocation of 44 subtidal sites (circles) to Mud Benthic Health Model (BHM) groups based on canonical analysis of principal coordinates (CAP). Colours indicate the benthic health ranking, where green indicates low and red high mud effect, respectively.

Only 23% of sites were ranked in Metals BHM Group 3, suggesting that most of the benthic communities in the harbour are fairly healthy with regard to metal impacts. Subtidal areas impacted by metals were primarily located in the southern part of the harbour, near the urbanised town centre, and near the Te Puna sub-estuary and mid- harbour (Figure 16). Communities were 72% dissimilar between Group 1 and Group 3, with Group 3 communities characterised as having higher abundances of certain polychaetes (*Aricidea* sp., polydorids, *Heteromastus filiformis*, Exogoninae, Paraonidae), oligochaetes and amphipods (including Corophiidae) than sites in Group 1, but fewer pipi and para-syllid polychaetes (refer Appendix 7 for full results).

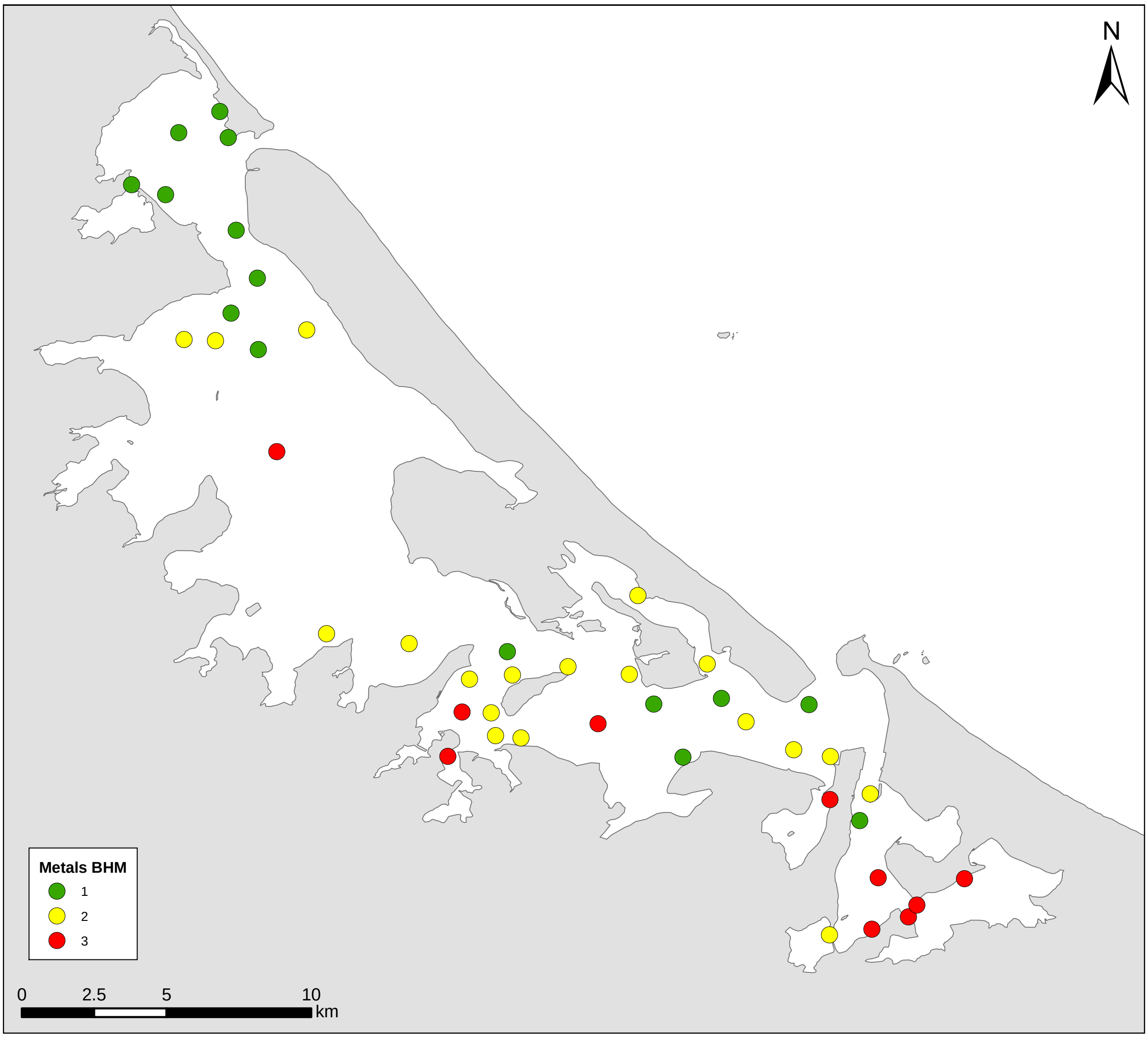

Figure 16. Map of Tauranga Harbour showing the allocation of 44 subtidal sites (circles) to Metals Benthic Health Model (BHM) groups based on canonical analysis of principal coordinates (CAP). Colours indicate the benthic health ranking, where a green ranking indicates low red high metals effect, respectively.

AMBI and RI-AMBI values for the 44 Tauranga Harbour sites ranged from 1.2–2.8 and 1.4–2.3, respectively, indicating that all subtidal sites in Tauranga Harbour would be classified in ecological group III (Table 12; refer Appendix 5 for full results). AMBI and RI-AMBI values and the Mud BHM CAP scores were moderately correlated (r = 0.62 and 0.72, respectively) but only a weak correlation was found between these values and the Metal BHM CAP scores (r = 0.30 and 0.44, respectively). Mean values for both AMBI and RI-AMBI generally increased with increasing BHM group for the mud and the metals BHMs; however, the Metals BHM could differentiate between smaller changes in community structure than either AMBI or RI-AMBI.

Table 12. Benthic Health Model (BHM) groups for the mud and metals BHMs for 44 subtidal sites in Tauranga Harbour, where higher group numbers indicate greater impact on benthic communities by the environmental gradient of interest. Average values (± standard error) for key variables are shown. Mud is reported as a percentage and metals are measured in mg/kg. n = number of sites in each group, Cu = copper, Pb = lead, Zn = zinc, N = total abundance per core, S = total number of taxa per site, AMBI = AZTI Marine Biotic Index (Borja et al., 2000), RI-AMBI = Richness Integrated AZTI Marine Biotic Index (B. P. Robertson et al., 2016).

Mud BHM								
Group	n			Mud	N	S	AMBI	RI-AMBI
1	17			3.5 (± 0.29)	82 (± 13.2)	26 (± 2.3)	1.6 (± 0.07)	1.6 (± 0.04)
2	8			5.3 (± 0.54)	212 (± 79.9)	37 (±1.7)	2.0 (± 0.11)	1.8 (± 0.06)
3	4			9.7 (± 2.12)	301 (± 94.6)	35 (± 2.8)	2.2 (± 0.08)	2.0 (± 0.08)
4	8			11.5 (± 2.63)	323 (± 73.1)	41 (± 3.0)	2.2 (± 0.13)	2.0 (± 0.08)
5	7			12.5 (± 1.33)	364 (± 62.7)	36 (± 2.5)	2.1 (± 0.11)	2.0 (± 0.08)
Metals BHM								
Group	Sites	Cu	Pb	Zn	N	S	AMBI	RI-AMBI
1	15	0.5 (± 0.04)	2.1 (± 0.15)	11.5 (± 1.16)	125 (± 47.0)	27 (± 2.2)	1.8 (± 0.09)	1.7 (± 0.05)
2	19	0.9 (± 0.10)	2.8 (± 0.18)	15.6 (± 1.2)	234 (± 39.0)	36 (± 2.0)	1.9 (± 0.09)	1.9 (± 0.05)
3	10	1.8 (± 0.33)	4.1 (0.48)	25.5 (± 2.05)	309 (± 62.9)	35 (± 3.3)	2.1 (± 0.15)	1.9 (± 0.09)

3.4.3 Model validation

The success of each model at identifying and predicting real and repeatable patterns in the data, was measured by its ability to correctly place the six validation sites onto the environmental gradient. The position of each validation site on each environmental gradient (mud and metals) based on their observed measured mud content and metal concentrations is shown in Table 13. There was a good spread of samples in the validation set in terms of mud content and concentrations of metals, with sites found in each of the mud and metals groups, except group 4 for mud.

Table 13. Observed values of validation sites in relation to the environmental gradients used in the Benthic Health Models (BHMs) and groups assigned by the BHMs.

Site	% mud	Mud BHM	PC1 metals	Metals BHM
2	3.3	1	-0.911	1
5	6.2	2	-0.214	1
15	11.3	3	0.275	2
27	5.1	2	0.238	2
36	5.6	2	0.0831	2
45	8.0	5	1.39	3
Modelled range	2.4-23.4		-1.3-1.65	

Both mud and metals BHMs were relatively good at predicting the positions of validation sites along the environmental gradients (Figure 17; Table 14). Figure 17 shows predicted vs observed values for the validation sites along the mud and metal gradients. For both mud and metals BHMs, most validation sites lie close to the 1:1 line, with a strong relationship between predicted and observed (R^2 = 0.90 and 0.90, respectively) and with slopes of the regressions lines close to 1 (0.70 and 0.73, respectively).

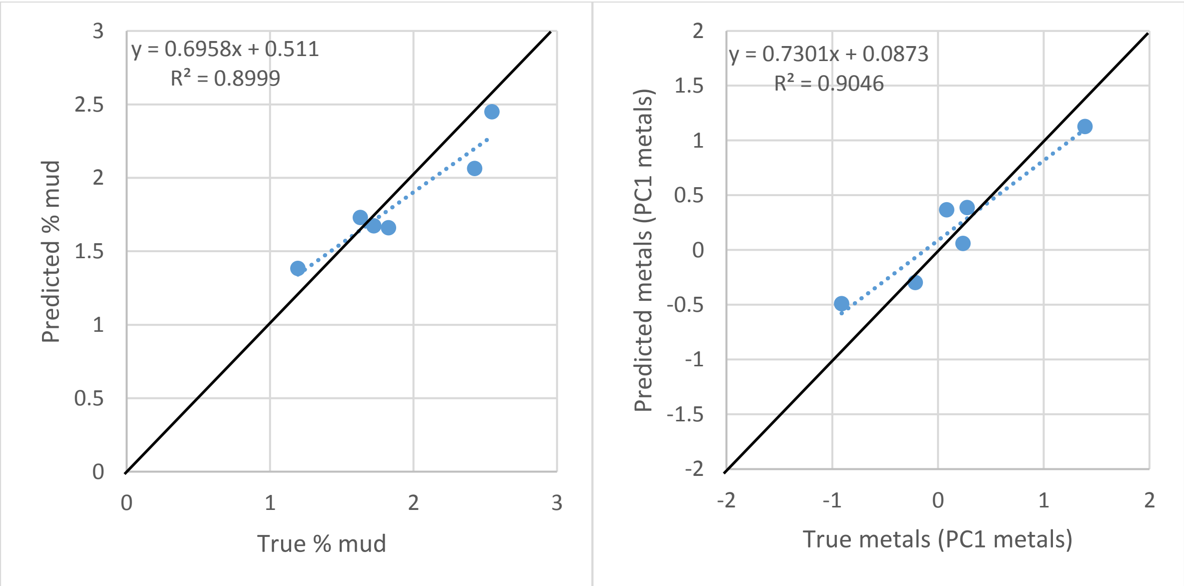

Figure 17. Linear regression between predicted values and observed values for validation sites for the mud (left) and metal (right) Benthic Health Models (BHMs). The black line on the plot has a slope of 1 and an intercept of zero (i.e. 1:1 line), where all points would lie if model predictions were perfect.

Table 14. Summary of validation success for the mud and metals Benthic Health Models (BHMs). SS_{RES} = residual sum of squares calculated as the sum of squared deviations of predicted values from the observed values (smaller is better), a = intercept, b = slope (closer to 1 is better), R^2 = strength of the relationship (coefficient of determination) between the predicted and observed values (closer to 1 is better).

Model	SS_{RES}	*a*	*b*	R^2
Mud BHM	0.22	0.5110	0.6958	0.90
Metals BHM	0.39	0.0873	0.7301	0.90

The Mud BHM showed moderate correlations with TOC and LOI (r = 0.72 and 0.67, respectively), which was expected given the high correlation between percentage mud and these two variables. Positive correlations were also found between Cu (r = 0.59), Pb (r = 0.46) and Zn (r = 0.44) and mean current velocity (r = 0.56). Weak correlations between Mud BHM CAP scores and salinity and depth were found, indicating that these were unlikely to be confounding factors (r = 0.31 and 0.26, respectively). For the Metals BHM, correlations with all variables were less than 0.69, with moderate positive correlations found with mud (r = 0.67), LOI (r = 0.57), and TOC (r = 0.60). Similar to the Mud BHM, salinity, depth and mean currents had weak correlations with Metals BHM CAP scores, thus are unlikely to be confounding factors (r = 0.03, 0.22 and 0.37, respectively).

The potential for interactions between mean current velocity and the mud and metals BHM CAP scores was further checked by overlying current velocity groups over the BHM plot. For the Mud BHM, most sites classified as current velocity Groups 4 and 5 were at the lower end of the CAP axis (Figure 18). This result, combined with the 4.1% overlap in explained variation identified by the DistLM, and the moderate correlation between current velocity and Mud BHM CAP scores, indicates that currents and mud are coupled in this system. No pattern was found for the Metals BHM, suggesting that currents are not driving patterns in the Metals BHM (Figure 19).

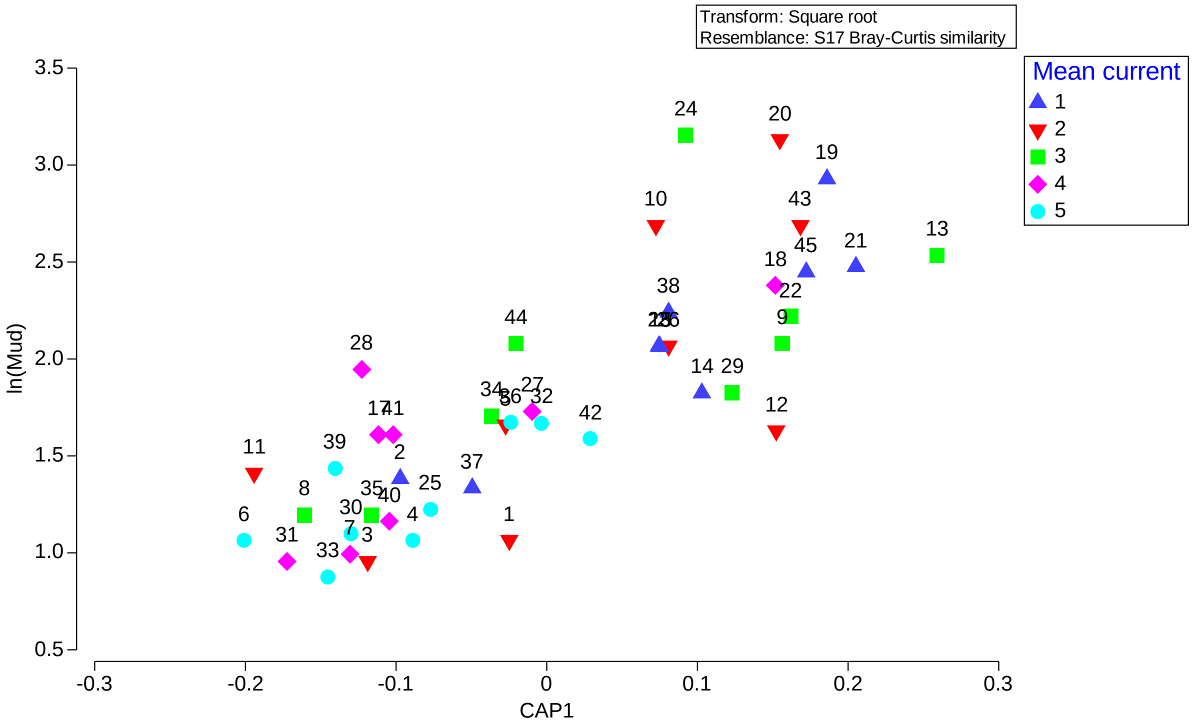

Figure 18. Canonical analysis of principal coordinates (CAP) model for mud (Mud BHM), based on data from 38 subtidal sites in Tauranga Harbour. Labels indicate site number and sites are colour coded according to mean current velocity groups, with Group 1 and 5 indicating low and high currents, respectively.

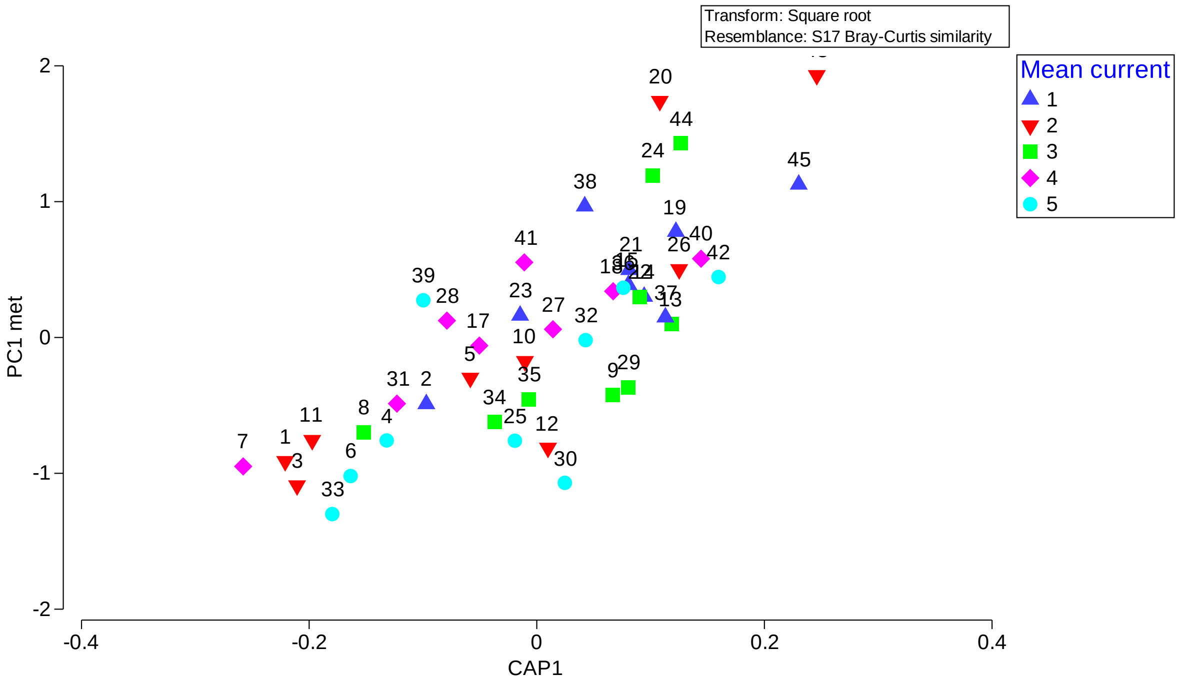

Figure 19. Canonical analysis of principal coordinates (CAP) model for metals (Metals BHM) based on data from 38 subtidal sites in Tauranga Harbour. Labels indicate site number and sites are colour coded according to mean currents with Group 1 indicating low currents and Group 5 indicating high currents.

A moderate positive correlation (r = 0.71) was found between the mud and metals BHM CAP scores, indicating a potential interaction between the two models. However, the relationship between the two subtidal BHMs showed a relatively high degree of variation (Figure 20).

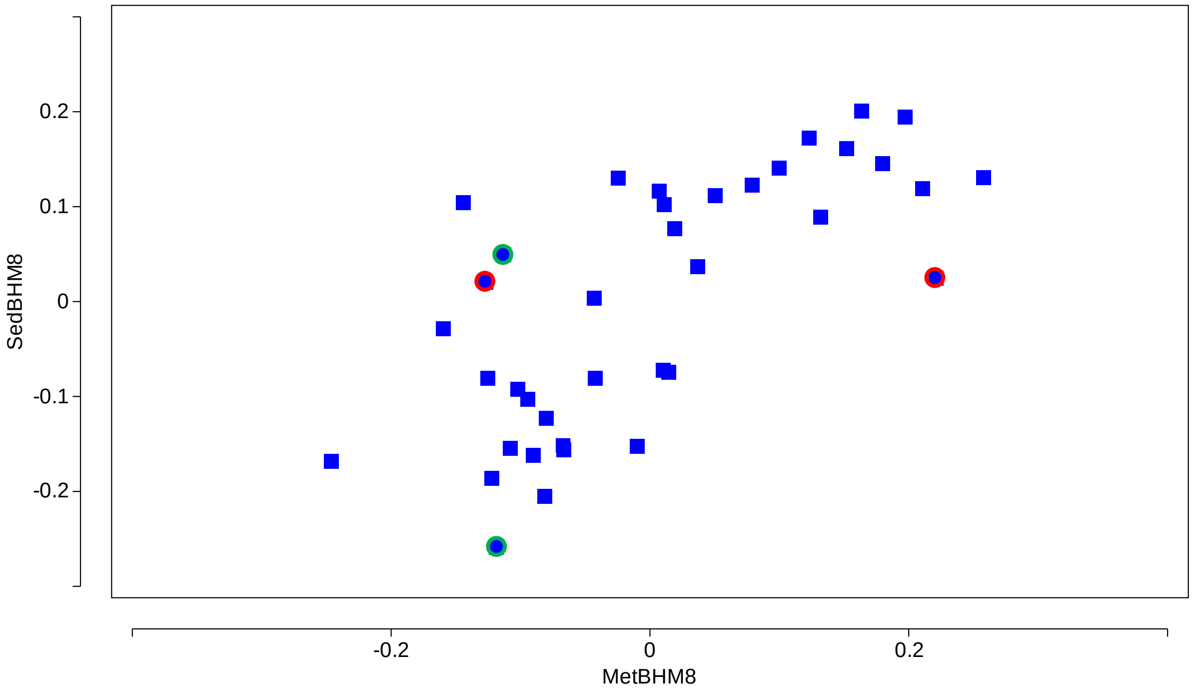

Figure 20. The relationship between the CAP scores from the mud (SedBHM8) and metals (MetBHM8) Benthic Health Models (BHMs) show a high degree of variation as evidenced by the wide range of corresponding values possible for a given CAP mud or CAP metal score (illustrated by the red and green circles, respectively).

4. Discussion

4.1. Water column physico-chemical variables

Measured water temperatures and salinities were generally in keeping with modelled summer predictions for Tauranga Harbour (Tay et al., 2013). The lack of observed water column stratification is consistent with other studies (Monahan, 2018; Pritchard, Gorman, & Hume, 2009) and most likely reflects well mixed shallow waters at the surveyed sites (most < 7.5 m depth). The salinity (21-23 PSU) observed at Site 1 near Pios Beach (Bowentown) was considerably lower than other sites and could be explained by freshwater discharge from the Waiau River or groundwater discharges, although these sources would need to be significant to reduce salinity to the measured level.

Concomitant with other hydrodynamic models of the harbour (Tay et al., 2013), predicted current velocities were relatively low over the tidal flats and sub-estuaries and greater in the channels. Highest velocities were predicted near the harbour entrances due to their constricted morphologies. Overall, subtidal sites were predicted to have relatively high current velocities, which were consistent with other studies (Spiers et al. 2009; Monahan 2018). These results indicate the model estimates were suitable for use in the development of the Benthic Health Models.

4.2. Sediment physico-chemical variables

Subtidal sediment physico-chemical results were compared with data collected during the 2011/12 intertidal survey of Tauranga Harbour as well as the National Estuary Dataset, which contains data from 409 intertidal estuarine sites across New Zealand (Berthelsen, Clark, et al., 2018; Appendix 8). Results were also evaluated against the Estuary Trophic Index (ETI) interim threshold bands, which are being developed for intertidal estuarine sediments (Robertson et al., 2016). While subtidal data and thresholds bands would have been preferable for comparison, there are currently no national-scale subtidal estuarine datasets or thresholds available for New Zealand.

Sediments in subtidal areas of Tauranga Harbour were found to be predominantly sandy with less mud than intertidal areas. Highest mud contents were found near the Te Puna sub-estuary and beach, which also had high intertidal mud content (Ellis et al., 2013). These areas have been previously reported to have high suspended sediments and turbidity (Scholes, 2015). Te Puna sub-estuary is partially enclosed by a spit at the entrance, making it susceptible to accumulation of sediments (Tay et al., 2013). Modelling of sediment loads identified the Te Puna sub-catchment as having relatively high sediment yields (Elliott, Parshotam, & Wadhwa, 2010). Site 20, off Te Puna beach, is located outside the sub-estuary in the central deep channel. As this was the deepest subtidal site surveyed (9.3 m depth), the high mud content may reflect a depositional hollow within the channel or its proximity to the Wairoa River. In general, sediment in the upper reaches of the channels (around Te Puna, Omokoroa, Rangataua Bay and the upper northern harbour) tended to contain higher mud content than sites closer to the main channels. Areas with relatively high gravel content were located around the Mt Maunganui main channel, extending into the Tauranga City Basin, and were generally associated with relatively high mean current velocities (0.3–0.6 m/s). Interim threshold bands for mud content developed for intertidal estuarine sediments (ETI; Robertson et al., 2016) suggest that subtidal portions of the harbour reflect no more than minor stress on sensitive organisms in most areas. Tauranga Harbour sediments were slightly sandier than the national median for intertidal estuarine sites with significantly less mud.

Like sediment mud content, the upper reaches of the channels tended have higher organic content and nutrient concentrations than sites closer to the main channels. High sediment nitrogen, phosphorus and organic content are indicators of enrichment and are generally closely associated with muddier sediments. Organic content in subtidal sediments was comparable to that measured in intertidal areas; however, nutrient concentrations were slightly lower, perhaps reflecting the lower mud content of subtidal sediments.

Bay of Plenty Regional Council water quality monitoring indicates that nitrogen is increasing within the harbour, likely from terrestrial sources, but phosphorus appears to be decreasing (Scholes, 2015). Most major point source discharges of nitrogen and phosphorus (such as sewage outfalls) were removed from the harbour in the early to mid-1990s (Sinner et al., 2011) but nutrients still enter the harbour through surface runoff and streams. The low residence times within Tauranga Harbour (Tay et al., 2013) result in rapid dilution of nutrients, which largely mitigate seabed enrichment effects in the central and outer regions of the harbour.

The Te Puna sub-estuary (Site 20) was found to have relatively high nutrient concentrations, which may be a result of the depositional environment created by the constricted entrance and high sub-catchment sediment yields (Elliott et al., 2010; Tay et al., 2013). Nutrient concentrations in the inner Te Puna sub-estuary were the highest recorded during the intertidal survey. However, subtidal sites close to waterways predicted to be delivering high nitrogen loads into the estuary (e.g. Wairoa River, Kopurererua Stream, Waiorohi Stream) did not have elevated sediment TN concentrations. Interim threshold bands for organic matter and nutrients developed for intertidal estuarine sediments (ETI; Robertson et al., 2016) suggest that subtidal portions of the harbour reflect no more than minor stress on sensitive organisms in most areas. On a national scale, maximum organic content in Tauranga Harbour sediments was in the upper quartile of concentrations normally found in intertidal estuarine site in New Zealand but nutrient concentrations were lower than the national median.

Like nutrients and organic matter, sediment contamination by metals can also be highly correlated with the percentage of mud content due to the adherence of chemicals to fine sediments and/or organic content (Green et al., 2001). Metal concentrations were lower in subtidal sediments than intertidal areas of the harbour, perhaps reflecting the lower mud content of subtidal sediments or greater distance from source inputs. Indeed, sites with the highest Cu, Pb and Zn concentrations were located either in the highly urbanised southern harbour or in areas of high mud deposition. Park et al. (2014) reported that the emerging spatial pattern of impact related to urban development in Tauranga Harbour appears very similar to that found around Auckland, where storm-water discharges of Zn and Cu are still accumulating in the settlement zone of estuaries. Acceptably low levels of Cu, Pb and Zn were found throughout Tauranga Harbour compared with ANZECC (2000) ISGQ trigger guidelines and the TELs (threshold effects level 18.7, 30.2 and 124 for Cu, Pb and Zn respectively) developed by MacDonald et al. (1996) and utilised by the Auckland Council. Comparison with other New Zealand intertidal estuarine sites showed Tauranga Harbour is relatively unpolluted with respect to metals, with all metals, except Pb, less than median national values.

Chlorophyll *a* was higher in subtidal sediments than intertidal sediments with highest concentrations in the main channel of the southern basin. This pattern of sediment chlorophyll *a* concentration is consistent with water chlorophyll *a* measurements, which showed high median chlorophyll *a* concentrations in the upper reaches of the southern harbour (Scholes, 2015). Sediment chlorophyll *a* was not correlated with sediment nutrient concentrations.

4.3. Benthic macrofauna

There have been very few comprehensive quantitative studies conducted on the subtidal communities of Tauranga Harbour, with Park and Donald (1994) carrying out the last harbour-wide survey in 1990/91. Direct comparison between the 1990/91 and 2016 subtidal datasets is difficult because macrofauna were only sieved on a 1 mm mesh during the former survey, biasing those samples towards larger organisms. However, many of the numerically dominant taxa observed in 1990/91 were also found in the harbour in 2016.

Consistent with Park and Donald's survey, pipi, often found in large beds around estuary mouths and harbour channels (Cook, 2010), were the most abundant bivalves encountered during the 2016 subtidal survey. Although primarily intertidal (Cook, 2010), nut shells and cockles were also relatively abundant in both subtidal surveys due to the shallow nature of the harbour. Similarly, wedge shells, which usually inhabit intertidal areas down to the low water mark (Cook, 2010), were occasionally seen in the subtidal.

The Asian date mussel (*Arcuatula senhousia*), a marine pest introduced to New Zealand in the 1970s (Cook, 2010), was not detected in Park and Donald's 1990/91 survey of the harbour, but was relatively abundant during the 2016 survey. Incursions of the Asian date mussel were identified in the Bay of Plenty region prior to 2013 and it is believed their current distribution is beyond what can be effectively managed with current control tools (Lass, 2015).

Park and Donald (1994) reported extensive benthic communities associated with scallop (*Pecten novaezelandiae*) and horse mussel (*Atrina zelandica*) beds. Although the 2016 subtidal survey was not designed to quantitatively measure large bivalves, no live scallops were observed during the recent survey and horse mussels were only recorded at four sites. These large bivalves stabilise the sediment and provide complex physical structure to soft sediment habitats, providing predation refuges and settlement substrate for epifauna, so the apparent decline in these taxa is concerning.

Subtidal benthic community structure was very different to that observed in intertidal portions of Tauranga Harbour, which reflects the restricted distribution of many taxa to subtidal or intertidal zones. Consistent with Park and Donald's survey, intertidal areas of the harbour had greater proportions of bivalves, sea anemones and sea cucumbers and gastropods than subtidal communities, which had greater proportions of polychaetes, sea stars and urchins. Unlike the 1990/91 survey, however, crustaceans were found in higher proportions in the intertidal relative to subtidal areas in more recent surveys. Species richness and abundances were generally higher in the subtidal than the intertidal.

4.4. Performance of the Benthic Health Models

We used multivariate ordination modelling approaches to identify key environmental gradients affecting the health of macrofaunal communities in Tauranga Harbour. Mud and metals were identified as key environmental gradients, i.e. variables affecting the ecology of the harbour. Therefore, two models were developed based on the variability in community structure using CAP analyses. The models generally reflected environmental gradients very well with the model for mud performing slightly better than the model for metals.

This study uses the term environmental gradient rather than anthropogenic stressor gradient because it is recognised that some environmental gradients may reflect natural processes and are not solely driven by anthropogenic activities. For example, estuaries naturally infill with sediment as they progress from the land to the sea. Upper reaches of estuaries will always have muddier areas, and this is not necessarily indicative of an unhealthy state. However, in New Zealand rates of accumulation of sediments have been accelerated as a result of anthropogenic land-based activities (Thrush et al., 2004); therefore, it is likely that mud inputs are causing stress to estuarine communities and the decline of a site's Mud CAP score over time would certainly warrant concern. It is important to note that reference to 'less impacted' and 'more impacted' is relative to the sampling sites used to construct the model. For example, a site ranked as being highly impacted by mud would have a high percentage of mud relative to other sites in the model; however, on a global or even national scale it might not necessarily be considered to be highly impacted.

Most sites were ranked in the lower BHM groups, suggesting Tauranga Harbour has fairly healthy subtidal communities with regard to mud and metal impacts. This is supported by values from other biotic indices used to assess the health of estuarine communities (AMBI, RI-AMBI), which indicated that all subtidal sites in Tauranga Harbour would be classified as having 'good' ecological status (defined as being 'slightly polluted' with 'unbalanced' benthic community health; Borja et al., 2003; Robertson et al., 2016). The Mud BHM CAP scores showed a stronger correlation with AMBI and RI-AMBI values than the Metals BHM CAP scores, most likely because many of the eco-groups used to calculate these indices were based on the response of taxa to mud (Berthelsen, Atalah, et al., 2018; Robertson et al., 2016).

Mean values for both AMBI and RI-AMBI generally increased with increasing BHM group for the mud and the metals BHMs, providing evidence that the BHMs are tracking benthic community health. However, as found in other studies (Ellis et al., 2015; Hewitt et al., 2005), the Metals BHM, for which non-hierarchal clustering methods were used to identify groupings, was able to differentiate between smaller changes in community structure than biotic indices that utilise more simplified community information (i.e. AMBI, RI-AMBI). This was not necessarily the case for the Mud BHM, as no indication of clustering structure was evident across the environmental gradient and groupings were determined by evenly splitting the environmental gradients rather than groupings based on distinct differences in community structure. Clear differences in relative abundance of certain taxa were found between BHM groups, with higher abundances of some relatively tolerant taxa (e.g. *Heteromastus filiformis*, oligochaetes; AZTI Marine Biotic Index, 2014; Robertson, Gardner, & Savage, 2015) in the 'more impacted' groups while 'less impacted' groups had higher abundances of more sensitive taxa (e.g. pipi; Robertson et al., 2015).

The accuracy of each CAP model at identifying and predicting real and repeatable patterns in the data, was measured by its ability to correctly place six validation sites onto the environmental gradient. Both the mud and metals BHMs were relatively good at predicting the positions of validation sites along the environmental gradients. Where sites deviated from the 1:1 line, they usually did so in a way that would lead to conservative remedial action in line with the precautionary principle (Figure 17). For example, Site 2 in the Metals BHM lies above the line and thus was predicted from the macrofaunal data to be more impacted than it actually was (i.e. to contain greater metal concentrations than it actually did). In contrast, the Mud BHM predicted Site 15 to be less impacted by mud content than it actually was. This latter kind of bias is more concerning, as it can have dire consequences from a management perspective. Thus, models that err on the side of caution should be preferred. Although it cannot be stated that this pattern of conservatism would necessarily be repeated with a new set of validation sites, the combination of accuracy and conservatism seen here does bode well for the use of these models in general.

Strong correlations between mud, TOC and LOI make it difficult to separate the effects of these three variables from each other. Changes in benthic community structure shown by the Mud BHM could be related to organic content, however, as mud explained the largest variability in the data (i.e. as demonstrated by the DistLM marginal tests) it is likely that most changes in community structure are primarily in response to mud. Model results also indicated that currents and mud are coupled in Tauranga Harbour's subtidal environment. We would expect coupling of these variables as areas with high currents tend to be sandier while those with lower currents tend to be depositional areas for fine sediments. However, given that mud consistently explained the most variation in the DistLMs, we can still be confident that changes in benthic communities shown by the Mud BHM are driven by changes in mud. TOC, LOI and current velocities were not considered to be confounding factors for the Metals BHM.

A moderate positive correlation ($r = 0.71$) was found between the mud and metals BHM CAP scores, indicating a potential interaction between the two models. Hewitt and Ellis (2010) also found moderate correlation ($r = 0.75$) between the mud and metals BHM CAP scores developed for Auckland estuaries. However, Hewitt et al. (2012) found that very rarely simultaneous changes in CAP scores relating to both their mud and metals models occurred, suggesting that, although there is some crossover between community structure found in response to high mud and high metals, the two effects could still be separated. The relatively high degree of variation between the two subtidal BHMs indicates a considerable amount of variation explained by the varying scores along the separate axes and suggests both mud content and metal contamination are important factors with neither being a replacement for the other (Hewitt & Ellis, 2010). However, the correlation between the two sets of CAP scores does suggest that when the health of a site is being assessed relative to changes in sediment characteristics and contaminant levels, a bivariate plot will be useful. Changes related to one axis would suggest a response to changes in that variable, while changes in both directions may be a result of responses to both factors and initiate a close inspection of which species are showing changes.

Intertidal BHMs have previously been developed for Tauranga Harbour in response to mud, nutrient and metal loading (Ellis et al., 2013). It is important to understand that the intertidal models were developed using a different dataset to the subtidal models, therefore, because the models are shown on relative scales, the intertidal and subtidal BHM CAP scores are not comparable to each other. A high Mud CAP score in the subtidal is not necessarily the same magnitude of effect as a high Mud CAP score in the intertidal. It was not possible to build a combined BHM for both the intertidal and subtidal sites because the community structure was too different between these two zones and it would be difficult to draw out responses associated only with mud or metals from the extreme physical differences between the two zones.

A drawback of the BHM approach is that currently BHMs have only been developed for two regions in New Zealand (Auckland and Tauranga) and may not be applicable elsewhere. As the BHM assessment of estuarine health is on a relative scale, sites can only be compared to other sites within that region, limiting the usefulness of this tool in a national context. Regional variations in species composition mean the model may not perform well in other locations, although the Auckland model has been successfully applied to estuaries in Northland and Southland (McCartain & Hewitt, 2016; NIWA unpbl. data; Parkes, Hewitt, & McCartain, 2016). In order to make the BHM more relevant for decision makers, a national BHM is being developed using the National Estuary Dataset (Berthelsen, Clark, et al., 2018), and it will allow intertidal estuary health to be assessed in a standardised way across all of New Zealand (Dana Clark PhD research). Once that model is developed, councils will be able to assess the health of new or existing sites without the need to fund the costly development of a model specific to their region. The community structure of sites and sampling times not in the existing model are compared to model data using the 'add new samples' routine in CAP, PERMANOVA add-on in the PRIMER software (Anderson et al., 2008). Eventually, once a greater number of observations from more sites and locations are obtained from across New Zealand, this model may be re-run and modified considering the new data.

4.5. Impacts of anthropogenic stressors on ecosystem health

In terms of mud, this study supports the general findings from previous research (Anderson, 2008; Ellis et al., 2017; Robertson et al., 2015; Thrush et al., 2003) of strong changes in benthic macrofauna distribution in relation to percentage mud, with important implications for assessing long-term responses of communities to habitat change. The export of terrestrial sediment into estuaries is a natural process; however, increasing rates of sediment delivery are causing sediment loading to become threat to coastal systems around the world (Thrush et al., 2004). For example, average sedimentation rates in Chesapeake Bay have increased by an order of magnitude since 1760, when land clearing activities were first initiated (Thrush et al., 2004). New Zealand has a naturally high potential for sediment deposition due to its steep catchments, erodible soils, high rainfall and short, flashy rivers. Additionally, large-scale native forest removal has occurred relatively recently and on a faster scale than many places overseas. Accordingly, sedimentation resulting from changes in land use is recognised as the leading catchment-derived threat to New Zealand's estuaries (MacDiarmid et al., 2012). Impacts of sedimentation include ecological effects associated with smothering of benthic organisms, decreases in water clarity, altered feeding behaviour and changes to the physical habitat (e.g. Ellis, Cummings, Hewitt, Thrush, & Norkko, 2002; Navarro, Iglesias, & Ortega, 1992; Norkko et al., 2002). These changes can result in reductions in diversity, abundance and the loss of functionally important species (Ellis, Nicholls, Craggs, Hofstra, & Hewitt, 2004).

Consistent with other research in New Zealand (Ellis et al., 2017; Hewitt, Anderson, Hickey, Kelly, & Thrush, 2009; Thrush, Hewitt, Hickey, & Kelly, 2008; Tremblay, Clark, Sinner, & Ellis, 2017), the current study identified changes in benthic communities associated with differing metal concentrations. Major terrestrial sources of anthropogenic metals to coastal areas are municipal and industrial discharges, mining and urban development. Urban stormwater in particular has been found to be a significant contemporary source of heavy metals (Barry, Taylor, & Birch, 2000). Metals can be essential for organisms as trace elements, however at higher concentrations they can become toxic (ANZECC, 2000). High exposure to heavy metals can cause physiological stress, reduced reproductive success, and outright mortality in associated invertebrates and fishes (Fleeger, Carman, & Nisbet, 2003; Gagnaire, Thomas-Guyon, & Renault, 2004; Nicholson, 1999; Peters, Gassman, Firman, Richmonds, & Power, 1997; Radford, Hutchinson, Burandt, & Raftos, 2000). Estuaries and coastal ecosystems are particularly vulnerable as they act as natural retention systems for metal contaminants.

Benthic community changes at subtidal sites in Tauranga Harbour were found across a relatively low-level metal gradient (maximums of 3.5, 6.4, 37 mg/kg Cu, Pb, Zn, respectively). Hewitt et al. (2009) also observed declines in estuarine infauna at relatively low metal concentrations (6.5–9.3 mg/kg Cu, 18.8–19.4 mg/kg Pb, 114-118 mg/kg Zn) and a strong gradient of community change across a low metal concentration contaminant gradient. Similarly, along the Norwegian coast, concentrations as low as 2.1 mg/kg Cu, 8.6 mg/kg Pb and 13 mg/kg Zn caused changes in soft sediment macrofaunal communities (Bjørgesæter & Gray, 2008). These studies suggest that current sediment quality guidelines, which have higher thresholds, may not adequately protect coastal ecosystems from the adverse effects of contaminants.

There is increasing evidence that sediment quality guidelines developed from reviews of laboratory dose-response experiments are higher than those derived from field surveys (Bjørgesæter & Gray, 2008; Leung et al., 2005). The present study used changes in macrofaunal community structure as an endpoint whereas less conservative mortality-based endpoints are more common in laboratory studies (Kwok et al., 2008). Additionally, field surveys incorporate the simultaneous effects of multiple stressors (other metals as well as non-metal stressors; Leung et al., 2005), which can result in some organisms showing increased responses to metal contamination (Fukunaga, Anderson, & Webster-Brown, 2011; Thrush et al., 2008). A review by Norwood et al. (2003) found that additive or more than additive responses (synergistic, potentiation, coalitive) were documented in 56% of studies addressing impacts from metal mixtures, suggesting that field studies such as this one, which assessed community responses to more than one metal (e.g. copper, lead and zinc), may observe responses at lower metal concentrations than single-contaminant studies. Field surveys also allow for regional differences and variability (Chapman, MacDonald, Kickham, & McKinnon, 2006; Long, Field, & MacDonald, 1998; Norwood et al., 2003) and the presence of different species, indirect effects arising from biological interactions or sub-lethal effects (Baert, De Laender, & Janssen, 2017; Calow, 1998; Fleeger et al., 2003; Rohr, Kerby, & Sih, 2006), and the differential susceptibility of life stages.

Excess nutrients can also cause changes in benthic communities via adverse eutrophication effects (e.g. Bricker, Rice, & Bricker, 2014; Gilbert et al., 2010; Smith, Tilman, & Nekola, 1999; Valiela et al., 1992). Indeed, nutrient loading was found to be a key environmental variable structuring intertidal communities in Tauranga Harbour (Ellis et al., 2013). Total nitrogen and total phosphorus were not found to be key variables explaining changes in subtidal benthic community structure, however, this cannot be conclusively established as the poor resolution of the total nitrogen data (ADL = 500 mg/kg) may have masked important patterns.

4.6. Conclusions

The 2016 survey was the first comprehensive quantitative survey of Tauranga Harbour's subtidal environment since 1990/91. The harbour's subtidal environment appeared to be in good condition with most sediment physico-chemical parameters lower than national median values. Upper reaches of the channels tended to have higher mud, organic matter and nutrient concentrations than closer to the main channels and metals were highest in the urbanised southern harbour or in areas of high mud deposition. Compared with 1990/91, fewer scallops and horse mussels were observed in the recent survey and the invasive Asian date mussel has become common. Overall, subtidal benthic communities appeared to be healthy with regard to mud and metal impacts. The BHM approach used in this study can be used as a management or monitoring tool where sites are repeatedly sampled over time and tracked to assess long term degradation or improvement in the ecology of an area. The development of a national BHM will allow estuary health to be assessed in a standardised way across the country without the need for costly model development by councils.

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7. Appendices

Appendix 1. Measured water column variables

Table A1. Water column variables measured in Tauranga Harbour 17 March to 4 May 2016. Currents were modelled using the mean over 14 days. Coordinates are supplied in NZGD 2000 New Zealand Transverse Mercator (NZTM).

Site	NZTME	NZTMN	Sampling date	Depth MSL (m)	Temperature (°C)		Salinity (PSU)		Currents (m/s)	
					Surface	Bottom	Surface	Bottom	Mean	Maximum
1	1863119.322	5849891.499	21/04/16	1.1	20.4	20.5	21.2	22.7	0.29	0.88
2	1863324.634	5848985.559	21/04/16	9.4	19.7	19.6	31.6	32.0	0.19	0.78
3	1861965.729	5849197.227	21/04/16	5.1	19.6	19.7	31.7	33.5	0.30	0.90
4	1860615.298	5847444.629	21/04/16	2.3	19.5	19.6	29.5	29.3	0.63	1.36
5	1861538.546	5847070.724	22/04/16	2.6	20.0	19.9	33.1	33.3	0.28	0.64
6	1863438.932	5845776.69	21/04/16	9.4	19.4	19.2	29.1	29.9	0.65	1.22
7	1863971.866	5844105.695	22/04/16	7.7	19.4	19.5	29.3	30.1	0.41	0.90
8	1863208.214	5842925.535	22/04/16	6.7	19.4	19.5	28.0	28.3	0.33	0.58
9	1861889.068	5842056.757	02/05/16	1.4	18.2	18.2	30.8	30.8	0.31	0.61
10	1862749.10	5841987.182	02/05/16	3.3	18.5	18.5	31.9	31.9	0.30	0.59
11	1863916.580	5841646.534	02/05/16	1.9	19.0	19.0	32.9	32.9	0.30	0.56
12	1865267.274	5842276.885	02/05/16	2.0	18.7	18.6	31.4	31.8	0.30	0.88
13	1864312.794	5838107.616	02/05/16	2.0	18.3	18.4	30.6	30.8	0.34	1.50
14	1865477.811	5831782.855	17/03/16	2.2	21.7	21.7	26.7	29.1	0.20	0.76
15	1867728.257	5831371.376	17/03/16	6.0	22.1	22.2	30.9	31.2	0.21	0.46
16	1868993.279	5832554.776	NA	3.5	NA	NA	NA	NA	0.39	0.69
17	1870415.586	5831000.541	17/03/16	4.0	22.3	22.2	31.6	32.4	0.46	0.85
18	1869343.694	5830086.139	22/03/16	6.2	21.5	21.4	26.6	27.8	0.40	0.85
19	1869104.936	5828958.37	22/03/16	5.8	21.4	21.5	28.2	28.7	0.24	0.54
20	1868665.521	5827445.795	31/03/16	1.5	22.1	21.6	30.2	30.6	0.29	0.69
21	1870527.158	5830195.930	22/03/16	3.4	21.4	21.4	29.0	29.2	0.25	0.52
22	1869904.856	5828906.876	22/03/16	7.0	21.4	21.4	29.2	29.3	0.33	0.60

Site	NZTME	NZTMN	Sampling date	Depth MSL (m)	Temperature (°C)		Salinity (PSU)		Currents (m/s)	
					Surface	Bottom	Surface	Bottom	Mean	Maximum
23	1869999.026	5828115.093	22/03/16	6.7	21.8	21.5	25.8	28.7	0.25	0.51
24	1870693.226	5828013.849	22/03/16	9.3	21.4	21.4	29.0	29.2	0.31	0.58
25	1872057.511	5830430.881	22/03/16	3.5	21.4	21.4	29.2	29.2	0.53	0.93
26	1872819.505	5828436.971	01/04/16	7.7	20.9	21.3	30.5	31.6	0.26	0.50
27	1873732.639	5830110.682	18/03/16	5.3	21.1	21.4	31.1	32.1	0.35	0.89
39	1879878.498	5824854.123	15/03/16	4.1	21.5	21.5	35.2	35.0	0.62	1.17
40	1880318.767	5822864.450	15/03/16	2.0	21.7	21.5	34.6	34.9	0.42	0.98
41	1878919.119	5820940.037	28/04/16	1.9	17.9	18.3	29.6	31.1	0.43	1.12
42	1880085.938	5821093.493	28/04/16	2.6	18.3	18.8	30.3	32.8	0.50	1.29
43	1881102.689	5821482.088	30/03/16	2.6	21.3	21.4	30.0	31.0	0.27	1.52
44	1881346.039	5821885.422	30/03/16	3.5	21.4	21.3	29.5	32.2	0.34	0.81
45	1882682.691	5822747.542	NA	4.6	NA	NA	NA	NA	0.22	0.62
			Minimum	1.1	17.8	18.2	21.2	22.7	0.15	0.46
			Mean	4.5	20.4	20.4	30.3	31.2	0.36	0.89
			Maximum	9.4	23.0	22.6	35.2	35.0	0.65	1.52

Appendix 2. Hydrodynamic model description

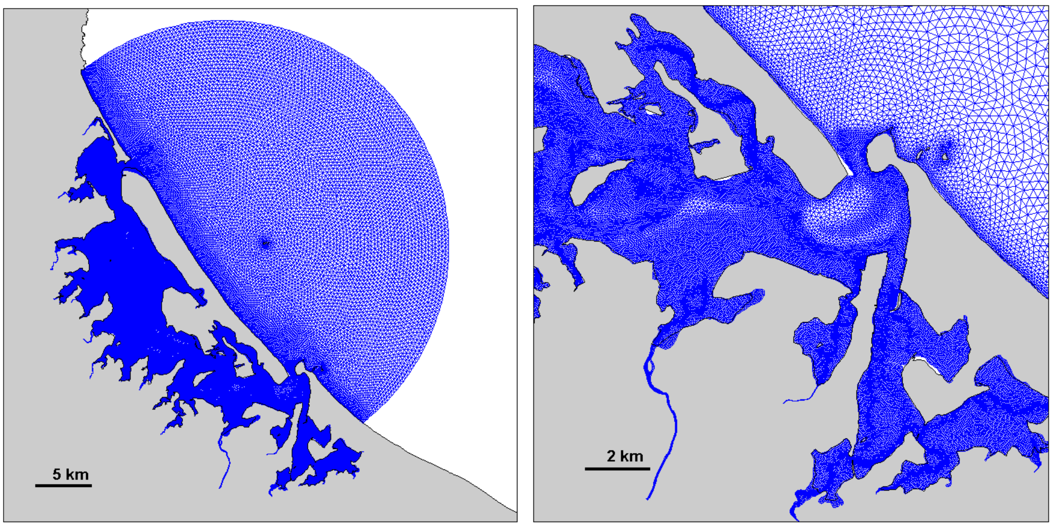
Authored by Ben Knight (Cawthron), Brett Beamsley and Remy Zynfogel (MetOcean Solutions Limited).

The Tauranga hydrodynamic model has been developed by MetOcean Solutions Limited (MOS) using the hydrodynamic model called the Semi-implicit Cross-scale Hydroscience Integrated System Model[7] (SCHISM; Zhang et al. 2016). SCHISM is a derivative work of a preceding 'SELFE' model (Zhang & Baptista 2008) and uses open-source code widely supported by a growing user community.

SCHISM is physically realistic, in that well-understood laws of motion and mass conservation are implemented. Therefore, water mass is generally conserved within the model, although it can be added or removed at open boundaries (e.g. through tidal motion at the ocean boundaries) and water is redistributed by incorporating information from the real-world system (e.g. bathymetric information, forcing by tides and wind). The model transports water and other constituents (e.g. salt, temperature, turbulence) through the use of triangular volumes (connected 3-D polyhedrons) of varying size and is described as an unstructured finite element model. For the model simulations conducted here, a two-dimensional model configuration, which assumes homogenous (well-mixed) water column properties, was used. This simulation methodology was selected to reduce the computational and storage requirements of the simulations, which sacrifice resolution in the vertical structure for high horizontal resolution. Additional information on the underlying methods employed in SCHISM can be found in the foundation papers Zhang & Baptista (2008) and Zhang et al. (2016).

Model domain and bathymetry

The model resolution was optimised to ensure replication of the salient hydrodynamic processes. The resolution ranged from about 1 km at the offshore boundary to ~ 20 m in shallow water and near the coast, with grid refinement near to coastal areas and within the estuary (Figure A2-1). Approximately 210,000 model elements are used to construct the model domain.



[7] http://ccrm.vims.edu/schismweb/

Figure A2-1. Model mesh employed for the hydrodynamic simulations for the whole model domain (left) and a close up of the southern entrance to Tauranga Harbour (right).

Local hydrographic soundings, LIDAR and fare sheet data were used to generate the model bathymetry (Figure A2-2). This was interpolated to the model grid using an inverse distance-weighted interpolation method (Shepard 1968).

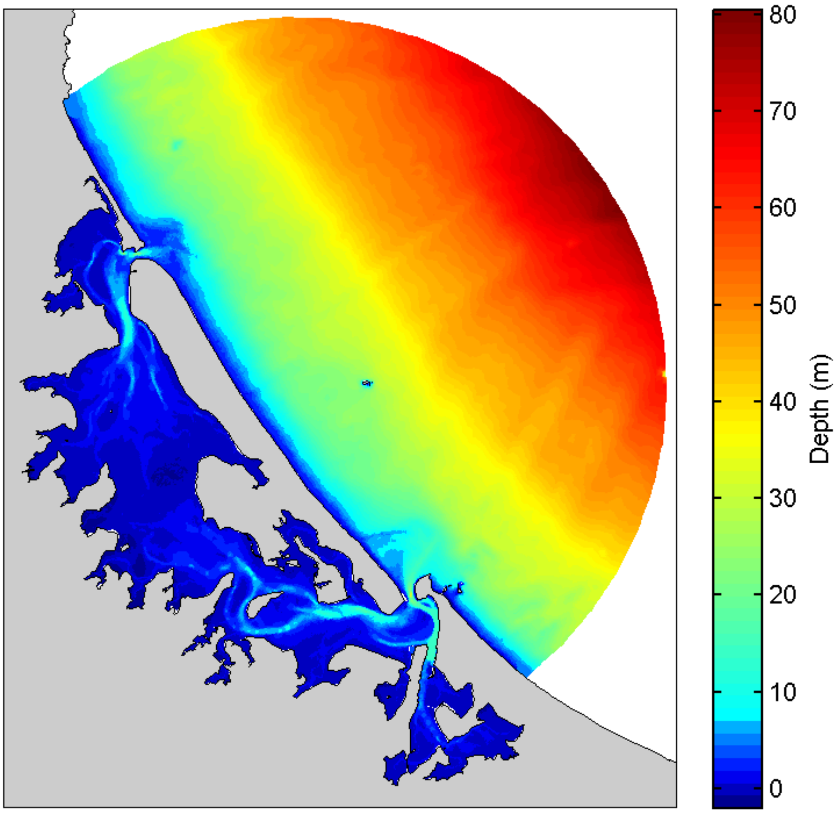


Figure A2-2. Model bathymetry employed for the hydrodynamic simulations for the whole model domain.

Model forcing

To drive flows within the model domain, energy and mass changes are introduced through the surface of the model (e.g. meteorological forcing such as rain, solar heating and wind) and open boundaries (e.g. river or ocean[8] forcing). The implementation of the forcing used in the model is discussed in the following sections.

Meteorological forcing

A 2-phase nesting approach using the Weather Research and Forecasting (WRF) atmospheric model was used to prescribe atmospheric forcing to the Tauranga SCHISM domain. A 12-km resolution WRF domain extending over New Zealand was nested within the Climate Forecast System Reanalysis (CFSR)

[8] The ocean boundary is visible as a semi-circular edge to the model in Figure A2-1 and Figure A2-2. River boundaries are small, 1 or 2 points and are located at the upper most point of the river.

data set from NOAA. The 12-km resolution WRF domain was used to prescribe full 3D atmospheric boundary conditions to a regional scale WRF domain with a 4-km resolution that extends over the Hauraki Gulf and surrounding area. This higher resolution model covers most of the Tauranga SCHISM model domain and hence a 4-km resolution WRF domain was used to prescribe full 3D atmospheric boundary conditions in the Tauranga SCHISM model[9]. The wind-speed from this hindcast has been validated at numerous sites around New Zealand; shown here are time series data from Auckland airport in January 2007 (Figure A2-3).

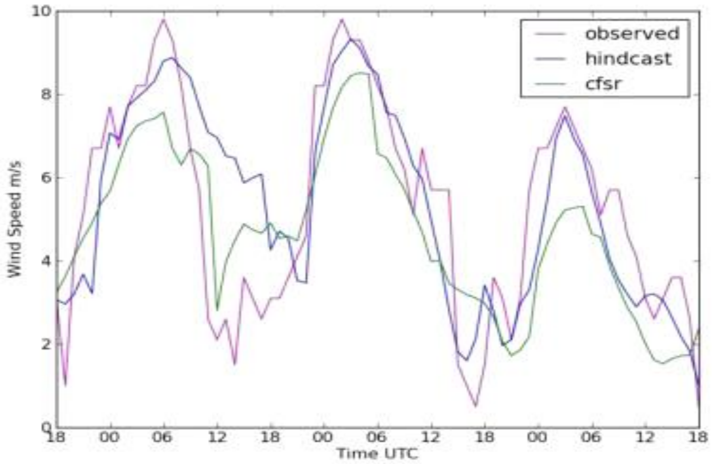


Figure A2-3. Comparison of both Climate Forecast System Reanalysis (CFSR) data and a high-resolution Weather Research and Forecasting (WRF) hindcast for Auckland Airport during a few days in January 2007.

Atmospheric forcing was specified with the following variables specified:

- Wind vector at 10 m elevation
- Near surface air temperature and humidity
- Precipitation rate
- Downwelling short and longwave radiation
- Pressure reduced to sea level.

The scale of the regional WRF model (4 km) means that some differences will exist between modelled and observed sea level winds in the region. These differences are likely to be most pronounced in areas with complex topography.

Ocean boundary forcing

A time-series of elevations, velocities, salinities and temperatures were interpolated at each offshore boundary node (Figure A2-3) and for each model time-step.

Elevations and velocities were defined using the combination of a New Zealand scale 2D Princeton Ocean Model (POM) tidal solution and a 3D implementation of a Regional Ocean Modelling System (ROMS)

[9] The eastern side of the Tauranga SCHISM model was modelled with 12-km meteorological data interpolated to a 4-km grid.

hindcast developed by MetOceans Solutions. Velocities were defined by combining logarithmically interpolated depth-averaged tidal velocities and residuals from the ROMS hindcast. Elevations were prescribed by combining tidal elevations from the POM solution and residual elevations from the ROMS hindcast.

Time-varying depth-averaged salinities and temperatures at each model node along the offshore open boundary were interpolated from the New Zealand scale ROMS hindcast dataset.

River boundary forcing

Time-varying fluxes from a total of 18 river sources were included in the modelling. River input salinity was set to 0.5 parts per thousand and a constant river temperature of 15 °C was defined for all rivers.

Model limitations

There are a number of limitations to any hydrodynamic model, therefore differences between actual currents and those simulated are possible. This can be due to differences associated with discretisation of the seafloor bathymetry, vertical structure of the water column properties and forcing parameters (e.g. wind, boundary residual flows etc.).

The 2D model simulation approach, while appropriate for periods where the water column is mixed within the estuary (e.g. during and following windy periods), will fail to reproduce 3D current flows which will at times dominate in the estuary. Such difference in flows between the surface and bottom waters within an estuary can be associated with density differences (i.e. salinity or temperature differences). Consequently, differences in simulated and real flows are possible. These are most likely to be most prevalent following periods of heavy rain (when freshwater is present in surface waters) and during warm calm weather (when surface heating can create thermal stratification). In addition, as tidal flows will dominate the flows over most of the estuary, any differences would be most noticeable during periods less affected by tidal flows and in areas away from the estuary entrances.

While these limitations can affect the accuracy of the model predictions, we consider that the model assumptions are unlikely to have a large effect on estimates of mean current speeds which has been the major focus of this aspect of the project.

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Appendix 3. Pearson correlation coefficients between variables considered during Benthic Health Model (BHM) development

	TP	TN	Chl *a*	Cu	Pb	Zn	LOI	Gravel	Sand	Mud	TOC	Salinity (S)	Salinity (B)	Depth	Day	Temp (S)	Temp (B)	PC1 met	Mean current	Max current
TP																				
TN	0.65																			
Chl *a*	0.19	0.07																		
Cu	0.67	0.54	0.02																	
Pb	0.64	0.46	0.01	**0.89**																
Zn	0.49	0.31	-0.02	**0.84**	**0.90**															
LOI	0.68	0.61	0.24	0.73	0.71	0.60														
Gravel	-0.10	0.00	-0.24	0.19	0.27	0.32	-0.14													
Sand	0.31	0.48	-0.19	0.69	0.70	0.69	0.50	0.68												
Mud	0.62	0.68	0.08	**0.80**	0.73	0.64	**0.86**	-0.14	0.60											
TOC	0.70	0.65	0.50	0.70	0.62	0.49	**0.86**	-0.25	0.36	**0.84**										
Salinity (S)	0.01	0.05	0.03	-0.08	-0.08	-0.11	-0.10	0.04	-0.07	-0.10	-0.13									
Salinity (B)	-0.01	0.01	0.08	0.07	0.07	0.04	0.20	-0.14	0.08	0.22	0.25	**-0.90**								
Depth	0.10	-0.09	-0.33	-0.19	0.00	-0.04	-0.11	-0.03	-0.06	-0.08	-0.27	-0.07	0.03							
Day	-0.19	-0.20	0.00	-0.24	-0.43	-0.38	-0.19	-0.13	-0.21	-0.18	-0.17	0.12	-0.05	-0.10						
Temp (S)	0.20	0.08	0.07	0.17	0.39	0.31	0.17	-0.02	0.09	0.22	0.23	0.06	-0.05	0.13	-0.54					
Temp (B)	0.20	0.07	0.04	0.18	0.40	0.35	0.16	0.03	0.12	0.21	0.20	0.02	-0.05	0.18	-0.58	**0.98**				
PC1 met	0.64	0.48	0.01	**0.97**	**0.96**	**0.93**	0.72	0.25	0.73	0.77	0.65	-0.09	0.07	-0.12	-0.33	0.27	0.28			
Mean current	-0.32	-0.17	-0.06	-0.38	-0.37	-0.30	-0.48	0.39	-0.04	-0.48	-0.44	0.23	-0.34	0.11	0.10	-0.08	-0.04	-0.37		
Max current	-0.21	0.03	-0.12	-0.11	-0.20	-0.15	-0.37	0.28	-0.02	-0.33	-0.28	0.18	-0.32	-0.14	0.08	-0.22	-0.19	-0.15	0.68	

Appendix 4. Taxa removed from dataset for Benthic Health Models (BHMs)

The table lists the taxa that were removed before modelling. Juveniles and larvae were removed because numbers are highly affected by recruitment events, meiofauna and non-infaunal taxa were removed because they are not likely to be well sampled using infauna cores.

Taxa name	Common name/taxa group	Life stage	Reason for exclusion
Acari	Arachnids (e.g. mites/ticks)	Adult	Not infauna
Ascidiacea	Ascidians	Adult	Not infauna
Bivalvia, unidentified	Shellfish	Juvenile	Juvenile
Brachyura	Crab	Juvenile	Juvenile
Bryozoa	Moss animals	Adult	Not infauna
Copepoda	Copepods	Adult	Meiofauna
Creediidae	Fish	Adult	Not infauna
Decapoda	Crustaceans (e.g. crabs)	Juvenile/larvae	Juvenile/larvae
Diasterope grisea	Ostracod	Adult	Meiofauna
Ephemeroptera	Mayfly	Adult	Not infauna
Epigonichthys hectori	Fish	Adult	Not infauna
Euphilomedes sp.	Ostracod	Adult	Meiofauna
Gobiesocidae	Fish	Adult	Not infauna
Hydrozoa	Hydroid	Adult	Not infauna
Insecta indeterminata	Insect	Adult	Not infauna
Leuroleberis zealandica	Ostracod	Adult	Meiofauna
Nematoda	Nematodes	Adult	Meiofauna
Nereididae	Polychaete worm	Juvenile	Juvenile

Appendix 5. Physico-chemical variables measured in Tauranga Harbour 15 March to 4 May 2016 and Benthic Health Model CAP scores and groups for each site.

A. Sediment grainsize, organics, nutrients and chlorophyll *a*

	Grain size (%)			Organics (%)		Nutrients (mg/kg)		Chl *a*
Site	**Mud**	**Sand**	**Gravel**	**TOC**	**LOI**	**TN**	**TP**	**(mg/kg)**
1	2.9	92.4	4.6	0.12	1.4	<500	98	13.0
2	3.3	96.3	0.2	0.09	2.2	<500	150	13.0
3	2.6	96.0	1.3	0.11	1.9	<500	152	23.2
4	2.9	87.6	9.3	0.10	1.9	<500	87	12.4
5	6.2	93.7	<0.1	0.32	4.2	500	167	31.8
6	2.9	84.5	12.6	0.08	1.4	<500	109	12.9
7	2.7	93.9	3.3	0.11	1.5	<500	119	22.1
8	3.3	94.4	2.4	0.13	1.8	<500	118	21.0
9	8.0	91.3	0.8	0.21	2.0	<500	114	14.8
10	14.7	83.9	1.5	0.30	3.2	700	141	22.7
11	4.1	89.0	6.9	0.10	1.8	<500	110	15.9
12	5.1	94.2	0.8	0.15	1.8	<500	79	12.6
13	12.6	86.9	0.5	0.36	3.3	600	132	16.3
14	6.2	93.4	0.4	0.32	3.2	<500	153	22.3
15	11.3	85.9	2.8	0.28	3.3	<500	154	16.5
16	4.0	92.1	3.9	0.22	2.1	<500	128	47.2
17	5.0	86.7	8.3	0.17	2.1	<500	115	56.3
18	10.8	87.9	1.2	0.23	2.7	<500	117	15.6
19	18.7	78.4	2.9	0.40	3.9	600	175	17.6
20	22.9	67.4	9.8	0.94	6.2	1200	340	19.2
21	11.9	86.0	2.0	0.31	3.1	<500	143	19.1
22	9.2	87.9	2.8	0.21	2.8	<500	147	14.0
23	7.9	89.8	2.2	0.16	3.6	<500	100	8.1
24	23.4	73.4	3.2	0.54	4.7	700	189	12.4
25	3.4	96.1	0.5	0.16	1.7	<500	114	39.4
26	7.9	91.6	0.5	0.24	3.3	<500	187	8.4
27	5.1	90.8	4.2	0.13	1.8	<500	111	32.8
28	7.0	80.2	12.8	0.22	2.7	500	180	10.2
29	6.2	91.5	2.4	0.25	1.8	500	115	19.1
30	3.0	97.0	<0.1	0.16	1.3	<500	136	34.7

	Grain size (%)			Organics (%)		Nutrients (mg/kg)		Chl *a*
Site	**Mud**	**Sand**	**Gravel**	**TOC**	**LOI**	**TN**	**TP**	**(mg/kg)**
31	2.6	85.0	12.5	<0.05	1.0	<500	81	2.0
32	5.3	89.9	5.0	0.23	2.4	500	182	22.8
33	2.4	93.3	4.3	<0.05	1.1	<500	111	2.2
34	5.5	91.5	3.0	0.19	2.1	<500	104	22.9
35	3.3	90.8	5.9	0.08	1.3	<500	134	9.1
36	5.6	81.8	12.7	0.11	2.2	<500	85	7.2
37	3.8	85.7	10.4	0.11	1.9	<500	91	19.5
38	9.4	89.9	0.7	0.23	2.9	<500	182	14.7
39	4.2	78.0	17.8	0.11	2.4	<500	108	15.2
40	3.2	87.5	9.4	0.07	1.7	<500	117	10.4
41	5.0	80.1	15.0	0.25	2.9	<500	152	4133
42	4.9	88.9	6.2	0.14	2.0	<500	117	14.1
43	14.7	78.5	6.9	0.40	3.0	500	183	11.2
44	8.0	79.6	12.4	0.23	2.6	<500	177	13.0
45	12.3	80.8	7.0	0.39	2.9	600	250	14.0
Minimum	**2.4**	**67.4**	**0.1**	**0.03**	**1.0**	**250**	**79**	**2.0**
Mean	**7.2**	**87.6**	**5.2**	**0.22**	**2.5**	**342**	**139**	**18.8**
Maximum	**23.4**	**97.0**	**17.8**	**0.94**	**6.2**	**1200**	**340**	**56.3**

B. Sediment metals

	Metals (mg/kg)							
	Cu	Pb	Zn	As	Cd	Cr	Hg	Ni
1	0.4	1.7	10.3	4.5	0.011	4.2	<0.010	0.8
2	0.4	1.9	9.2	3.9	<0.010	5.6	<0.010	1.1
3	0.4	1.6	7.7	5.0	<0.010	4.2	0.024	0.9
4	0.6	1.7	7.9	3.6	0.018	3.7	0.011	0.9
5	0.8	2.2	11.9	4.0	0.017	6.6	0.015	1.5
6	0.4	1.8	7.8	4.9	<0.010	3.4	<0.010	0.8
7	0.4	2.0	8.3	5.2	<0.010	3.7	<0.010	0.8
8	0.5	2.0	9.8	5.6	<0.010	4.0	0.013	0.9
9	0.7	2.0	10.4	3.6	0.017	4.7	0.014	1.1
10	0.8	2.4	11.9	4.6	0.020	6.0	0.017	1.4
11	0.5	2.0	8.8	5.9	<0.010	3.6	0.011	1.0
12	0.5	1.7	9.2	3.5	0.012	4.1	0.015	0.8
13	1.0	2.7	13.3	4.4	0.033	4.9	<0.010	1.4
14	1.0	3.2	17.0	6.1	0.044	4.5	0.017	1.4
15	0.9	3.2	18.7	6.0	0.051	4.6	0.019	1.3
16	0.4	2.3	18.0	7.0	<0.010	3.5	<0.010	0.9
17	0.6	2.8	19.4	6.2	0.016	4.3	<0.010	0.9
18	1.0	2.9	19.9	4.2	0.033	3.6	0.015	1.1
19	1.4	3.6	24.0	4.7	0.031	4.7	0.018	1.6
20	3.3	6.4	28.0	7.1	0.056	5.7	0.037	2.1
21	1.2	3.2	19.2	4.9	0.031	4.9	0.016	1.4
22	0.9	3.2	19.6	6.4	0.023	5.1	0.014	1.3
23	0.8	3.3	17.3	5.3	0.013	3.1	0.015	0.9
24	1.9	4.6	28.0	5.7	0.063	6.0	0.028	2.0
25	0.5	2.0	8.8	6.7	<0.010	2.6	0.011	0.9
26	1.0	3.7	22.0	6.4	0.026	5.7	0.015	1.5
27	0.8	3.3	20.0	4.9	0.018	4.4	<0.010	1.1
28	0.8	3.3	16.1	6.7	0.012	4.3	0.057	1.3
29	0.7	2.2	10.7	5.1	0.016	3.2	0.074	0.9
30	0.4	1.7	7.7	6.2	<0.010	2.9	0.036	0.8
31	0.5	2.1	14.1	3.4	<0.010	2.3	0.025	0.7
32	0.9	2.4	13.6	5.2	0.032	4.5	0.030	1.2

Metals (mg/kg)								
	Cu	**Pb**	**Zn**	**As**	**Cd**	**Cr**	**Hg**	**Ni**
33	0.3	1.4	8.4	4.2	<0.010	2.6	0.016	0.8
34	0.6	1.7	10.2	1.9	0.063	1.2	0.012	0.4
35	0.6	2.1	11.6	4.5	0.011	3.4	0.023	0.9
36	0.9	77.0	14.9	3.2	0.047	3.6	0.010	1.4
37	0.8	2.8	19.4	3.1	0.028	2.3	0.018	0.6
38	2.1	3.5	20.0	5.2	0.030	4.2	0.022	1.3
39	0.8	3.2	22.0	4.6	<0.010	2.0	0.027	0.7
40	1.0	3.8	25.0	4.1	0.010	1.7	0.024	0.8
41	1.4	3.0	18.2	2.7	0.025	1.5	0.016	0.6
42	1.1	2.1	28.0	3.0	0.022	1.4	0.025	0.5
43	3.5	6.4	37.0	3.6	0.062	3.5	0.041	1.3
44	2.2	5.5	31.0	4.9	0.027	2.9	0.026	1.1
45	2.7	4.2	27.0	4.3	0.061	3.0	0.030	1.2
Minimum	**0.3**	**1.4**	**7.7**	**1.9**	**0.005**	**1.2**	**0.005**	**0.4**
Mean	**1.0**	**4.5**	**16.5**	**4.8**	**0.023**	**3.8**	**0.019**	**1.1**
Maximum	**3.5**	**77.0**	**37.0**	**7.1**	**0.063**	**6.6**	**0.074**	**2.1**

C. Benthic Health Model CAP scores and groups and AMBI and RI-AMBI scores

Site	Mud BHM		Metals BHM			
	Score	Group	Score	Group	AMBI	RI-AMBI
1	0.025	2	0.221	1	2.3	1.9
2	0.097	1	0.097	1	1.9	1.8
3	0.119	1	0.211	1	2.2	2.0
4	0.089	1	0.132	1	1.9	1.7
5	0.027	2	0.058	1	2.0	1.8
6	0.201	1	0.164	1	1.8	1.6
7	0.131	1	0.258	1	1.6	1.6
8	0.161	1	0.152	1	1.6	1.5
9	-0.156	5	-0.067	2	2.2	2.0
10	-0.072	3	0.010	2	2.3	2.1
11	0.194	1	0.197	1	1.4	1.4
12	-0.152	4	-0.010	2	2.0	1.9
13	-0.259	5	-0.119	3	2.6	2.3
14	-0.103	4	-0.094	2	2.1	2.1
15	-0.075	3	-0.081	2	2.1	1.9
16	NA	NA	NA	NA	NA	NA
17	0.112	1	0.050	1	1.3	1.7
18	-0.152	4	-0.067	2	2.3	2.0
19	-0.186	5	-0.122	3	1.7	1.7
20	-0.155	4	-0.108	3	2.8	2.3
21	-0.205	5	-0.081	2	2.2	2.2
22	-0.162	5	-0.090	2	2.0	1.9
23	-0.074	3	0.015	2	2.2	1.9
24	-0.092	4	-0.102	2	2.3	2.0
25	0.077	1	0.019	2	1.2	1.5
26	-0.081	4	-0.125	3	1.8	1.7
27	0.010	2	-0.014	2	2.1	1.8
28	0.123	1	0.079	1	1.9	1.7
29	-0.123	4	-0.080	2	1.6	1.8
30	0.130	1	-0.025	2	1.5	1.6
31	0.172	1	0.123	1	1.7	1.7
32	0.004	2	-0.043	2	1.6	1.7

Site	Mud BHM		Metals BHM			
	Score	**Group**	**Score**	**Group**	**AMBI**	**RI-AMBI**
33	0.145	1	0.180	1	1.5	1.5
34	0.037	2	0.037	1	2.5	2.1
35	0.116	1	0.007	2	1.5	1.4
36	0.024	2	-0.076	2	1.9	1.7
37	0.049	2	-0.113	3	2.1	1.9
38	-0.081	4	-0.042	2	2.5	2.3
39	0.140	1	0.100	1	1.4	1.5
40	0.104	1	-0.144	3	1.3	1.5
41	0.102	1	0.011	2	1.2	1.6
42	-0.029	3	-0.160	3	2.5	2.2
43	-0.168	5	-0.246	3	2.0	2.0
44	0.020	2	-0.127	3	1.5	1.5
45	-0.172	5	-0.230	3	2.2	2.0
Min	**-**	**-**	**-**	**-**	**1.2**	**1.4**
Mean	**-**	**-**	**-**	**-**	**1.9**	**1.8**
Max	**-**	**-**	**-**	**-**	**2.8**	**2.3**

Appendix 6. Similarity percentage (SIMPER) results comparing Mud Benthic Health Model (BHM) groups using square-root transformed infauna abundances.

Similarities and differences between groups are shown to a 45% level. Differences are only shown between Group 1 and Group 5. Av. Abund = average abundance, Av. Sim = average similarity, Av. Diss = average dissimilarity, Sim/SD or Diss/SD = ratio of average contribution divided by standard deviation, Contrib. % = percent contribution, Cum. % = cumulative percent contribution.

Group 1
Average similarity: 30.91

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Oligochaeta	2.02	3.60	1.27	11.66	11.66
Nemertea	0.98	2.81	1.32	9.09	20.74
Nematoda	1.36	2.76	0.90	8.94	29.69
Exogoninae	0.91	1.97	1.11	6.36	36.05
Para-syllid	1.27	1.95	0.54	6.32	42.37
Hesionidae	1.23	1.89	0.57	6.11	48.47

Group 2
Average similarity: 40.35

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Oligochaeta	5.48	4.86	1.53	12.05	12.05
Exogoninae	2.64	3.27	0.97	8.10	20.15
Heteromastus filiformis	2.62	3.23	1.54	8.01	28.16
Paraonidae	2.14	2.81	1.20	6.97	35.12
Amphipoda	1.53	2.24	1.94	5.56	40.68
Nematoda	1.39	1.98	1.84	4.92	45.59

Group 3
Average similarity: 35.84

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Heteromastus filiformis	3.97	5.06	4.38	14.12	14.12
Aricidea sp.	4.45	4.40	1.33	12.29	26.41
Paraonidae	2.51	3.46	5.68	9.66	36.07
Oligochaeta	5.38	2.57	1.18	7.17	43.24
Exogoninae	2.38	2.19	2.86	6.10	49.34

Group 4
Average similarity: 49.15

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Polydorid	6.08	5.46	2.03	11.11	11.11
Oligochaeta	3.84	5.37	6.41	10.93	22.04
Heteromastus filiformis	4.52	5.02	1.57	10.21	32.25
Aricidea sp.	3.65	4.19	1.37	8.52	40.76
Paraonidae	3.29	3.70	1.56	7.53	48.29

Group 5
Average similarity: 54.25

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Aricidea sp.	7.69	10.09	3.76	18.59	18.59
Polydorid	7.60	7.25	2.34	13.36	31.95
Heteromastus filiformis	5.17	6.27	6.71	11.56	43.50
Paraonidae	3.78	3.66	2.38	6.74	50.25

Groups 1 & 5
Average dissimilarity = 75.85

Species	Group 1 Av.Abund	Group 5 Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Aricidea sp.	0.74	7.69	7.90	2.44	10.41	10.41
Polydorid	0.34	7.60	7.66	1.78	10.10	20.51
Corophiidae	0.45	5.34	4.89	0.99	6.45	26.96
Heteromastus filiformis	1.14	5.17	4.36	2.08	5.74	32.70
Paraonidae	0.66	3.78	3.28	1.75	4.33	37.03
Amphipoda	0.93	2.47	2.26	0.98	2.98	40.00
Oligochaeta	2.02	2.87	2.15	1.35	2.84	42.84
Pseudopolydora sp.	0.03	1.91	2.04	2.55	2.69	45.54
Paphies australis	1.82	0.00	2.03	0.52	2.68	48.22

Appendix 7. Similarity percentage (SIMPER) results comparing Metals Benthic Health Model (BHM) groups using square-root transformed infauna abundances.

Similarities and differences between groups are shown to a 45% level. Differences are only shown between Group 1 and Group 3. Av. Abund = average abundance, Av. Sim = average similarity, Av. Diss = average dissimilarity, Sim/SD or Diss/SD = ratio of average contribution divided by standard deviation, Contrib. % = percent contribution, Cum. % = cumulative percent contribution.

Group 1
Average similarity: 32.12

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Oligochaeta	3.07	3.71	1.23	11.55	11.55
Nemertea	1.15	3.14	2.30	9.76	21.31
Heteromastus filiformis	1.79	2.61	1.00	8.14	29.45
Nematoda	1.28	2.55	1.32	7.93	37.38
Amphipoda	1.14	2.09	1.12	6.49	43.87
Para-syllid	1.30	2.07	0.67	6.45	50.32

Group 2
Average similarity: 40.80

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Heteromastus filiformis	3.32	3.86	1.39	9.46	9.46
Oligochaeta	3.02	3.64	1.77	8.93	18.39
Polydorid	3.96	3.36	1.20	8.24	26.63
Aricidea sp.	3.11	3.19	1.03	7.83	34.46
Paraonidae	2.57	2.73	1.33	6.69	41.14
Cumacea	2.30	2.27	1.30	5.55	46.70

Group 3
Average similarity: 40.69

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Aricidea sp.	5.56	5.69	1.32	13.98	13.98
Exogoninae	3.44	5.34	2.04	13.11	27.09
Oligochaeta	4.69	4.57	1.70	11.23	38.32
Heteromastus filiformis	3.85	3.17	1.14	7.78	46.10
Polydorid	5.43	3.03	0.66	7.44	53.54

Groups 1 & 3
Average dissimilarity = 72.10

Species	Group 1 Av.Abund	Group 3 Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Aricidea sp.	0.75	5.56	5.69	1.46	7.89	7.89
Polydorid	0.32	5.43	5.55	0.94	7.70	15.59
Oligochaeta	3.07	4.69	4.65	0.97	6.44	22.03
Exogoninae	0.87	3.44	3.56	1.72	4.94	26.97
Heteromastus filiformis	1.79	3.85	3.45	1.27	4.79	31.76
Paphies australis	1.44	0.86	2.62	0.57	3.63	35.39
Corophiidae	0.20	2.85	2.59	0.67	3.59	38.98
Paraonidae	1.03	2.70	2.47	1.46	3.42	42.40
Amphipoda	1.14	1.83	1.82	1.00	2.52	44.92
Para-syllid	1.30	0.16	1.66	0.89	2.30	47.23

Appendix 8. Comparison of Tauranga subtidal and intertidal sediment physico-chemical variables with the National Estuary Dataset, which contains data from 409 intertidal estuarine sites across New Zealand (Berthelsen, Clark, et al., 2018).

National analytical detection limits (ADL) are shown in brackets, with Tauranga intertidal and subtidal values below ADL replaced by zero values to be consistent with the national dataset. Tauranga Harbour values higher than the national dataset are shown in bold. TOC = total organic content, LOI = loss on ignition, TN = total nitrogen, TP = total phosphorus, Cu = copper, Pb = lead, Zn = zinc, Cd = cadmium, Cr = chromium, Ni = nickel, NA = not available, n = sample size, min = minimum, max = maximum.

Variable	Dataset	n	Min	1st quartile	Median	3rd quartile	Max
Mud	Tauranga subtidal	45	**2.4**	3.3	5.3	9.2	23.4
(No ADL %)	Tauranga intertidal	75	**0.6**	3.9	9.3	17.5	76.4
	National dataset	817	0.0	4.3	12.6	26.9	99.9
Sand	Tauranga subtidal	45	**67.4**	**84.5**	**88.9**	**92.1**	97.0
(No ADL %)	Tauranga intertidal	75	**23.7**	**81.5**	**88.7**	**94.4**	100.0
	National dataset	817	0.1	73.1	87.4	95.7	100.0
TOC	Tauranga subtidal	45	0.0	0.1	0.2	0.3	0.9
(0.05%)	Tauranga intertidal	NA	NA	NA	NA	NA	NA
	National dataset	194	0.0	0.2	0.4	0.6	3.4
LOI	Tauranga subtidal	45	**1.0**	**1.8**	**2.2**	**3.0**	6.2
(0.04%)	Tauranga intertidal	75	**0.9**	**2.0**	**2.7**	**3.5**	10.0
	National dataset	563	0.4	1.1	1.9	2.9	16.3
TN	Tauranga subtidal	45	0	0	0	0	1200
(500 mg/kg)	Tauranga intertidal	75	0	0	0	585	1900
	National dataset	467	0	0	0	685	4133
TP	Tauranga subtidal	45	**79**	111	128	154	340
(40 mg/kg)	Tauranga intertidal	75	51	120	160	195	580
	National dataset	390	51	221	355	482	1837
Cu	Tauranga subtidal	45	0.0	0.0	0.0	0.0	3.5
(2 mg/kg)	Tauranga intertidal	75	0.0	0.0	0.0	0.0	6.1
	National dataset	435	0.0	2.2	5.4	8.9	38.0
Pb	Tauranga subtidal	45	**1.4**	2.0	2.6	3.3	6.4
(1 mg/kg)	Tauranga intertidal	75	**1.1**	1.7	2.5	3.4	13.0
	National dataset	435	0.0	2.6	5.3	9.4	140.0
Zn	Tauranga subtidal	45	**7.7**	10.2	16.1	20.0	37.0
(7.5 mg/kg)	Tauranga intertidal	75	0.0	10.3	15.0	21.0	55.0
	National dataset	435	0.0	20.0	38.4	55.1	231.0
Cd	Tauranga subtidal	45	0.0	0.0	0.0	0.0	0.0
(0.1 mg/kg)	Tauranga intertidal	NA	NA	NA	NA	NA	NA
	National dataset	311	0.0	0.0	0.0	0.0	1.1
Cr	Tauranga subtidal	45	0.0	3.0	3.7	4.6	6.6
(2 mg/kg)	Tauranga intertidal	NA	NA	NA	NA	NA	NA
	National dataset	311	2.3	7.2	10.8	18.3	104.3
Ni	Tauranga subtidal	45	0.0	0.0	0.0	0.0	2.1
(2 mg/kg)	Tauranga intertidal	NA	NA	NA	NA	NA	NA
	National dataset	311	0.0	4.8	7.4	10.3	95.3