SALT ECOLOGY

Fine Scale Intertidal Monitoring of Moutere Inlet

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for

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For the environment
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SALT ECOLOGY

GLOSSARY

AMBI	AZTI Marine Biotic Index
ANZECC	Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000)
ANZG	Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2018)
BHM	Benthic Health Model
aRPD	Apparent Redox Potential Discontinuity
As	Arsenic
Cd	Cadmium
Cr	Chromium
Cu	Copper
DGV	Default Guideline Value (ANZG 2018)
ETI	Estuary Trophic Index
GV	Guideline Value (ANZG 2018)
Hg	Mercury
NEMP	National Estuary Monitoring Protocol
Ni	Nickel
Pb	Lead
SACFOR	Epibiota categories of Super-abundant, Abundant, Common, Frequent, Occasional, Rare
SOE	State of Environment (monitoring)
TDC	Tasman District Council
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphorus
Zn	Zinc

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BACKGROUND

As part of its State of the Environment programme, Tasman District Council monitors the ecological condition of significant estuaries in their region. This report describes ecological monitoring and sedimentation surveys conducted in Moutere Inlet between 2006 and 2015, based on an analysis of archived data. The surveys largely followed the 'fine scale' approach described in New Zealand's National Estuary Monitoring Protocol (NEMP). Differences among monitoring sites, and temporal trends at each, are evaluated. Results are assessed against estuary condition criteria (see Table below), and discussed in the context of future monitoring, investigation and mitigation needs.

KEY FINDINGS

Sedimentation

- Sediment plate monitoring at seven sites did not reveal significant levels of sedimentation. There has been variable erosion and accretion since the first baselines were established in 2008, with the highest long-term mean annual sedimentation of 0.54mm/yr being less than the 2mm/yr national guideline value.

Sediment quality

- Despite the low sedimentation, surface sediments at two fine scale monitoring sites showed a trend for an increase in mud content over the period 2013-2015 compared with an earlier survey in 2006. Sediment mud content was getting close to the biologically relevant threshold of 25% (at Site A) when last measured in 2015.
- Sediments had low nutrient and total organic carbon levels, but showed visual symptoms of mild enrichment. However, there was no evidence of the typical characteristics of strong enrichment and anoxia (i.e. black sediment with a sulphide odour).
- Trace element concentrations were very low relative to national sediment quality guideline values, except for nickel which was elevated due to natural catchment sources. A limited analysis (one sample at each fine scale site) of a suite of organochlorine pesticides in 2014 revealed ecologically significant concentrations of DDT; this is a banned pesticide that was once widely used in horticulture and agriculture.

Summary of condition scores of ecological health for each fine scale monitoring site, based on mean values of key indicators and ETI rating criteria. Dash = not measured. TP not rated. See glossary for analyte definitions.

Site	Year	Mud	TOC	TN	TP	aRPD	As	Cd	Cr	Cu	Ni	Pb	Zn	AMBI
		%	%	mg/kg	mg/kg	mm	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	na
A	2006	10.3	0.56	309	546	19	-	< 0.100	33.8	6.0	76.1	3.7	26.8	2.0
	2013	18.2	1.03	333*	520	10	-	0.022	36.7	7.5	87.0	4.6	34.0	1.7
	2014	14.4	0.34	< 500	503	25	4.3	-	-	-	-	-	-	1.8
	2015	24.5	0.28	< 500	550	10	-	-	-	-	-	-	-	1.8
B	2006	12.9	0.71	368	513	26	-	< 0.100	29.6	6.1	58.4	4.6	25.0	1.9
	2013	18.2	0.99	< 500	497	10	-	0.022	30.7	6.8	66.3	5.0	31.7	1.8
	2014	13.7	0.33	< 500	457	7	5.7	-	-	-	-	-	-	1.7
	2015	18.3	0.28	550*	520	5	-	-	-	-	-	-	-	1.7

** Sample mean includes values below lab detection limits*

< All values below lab detection limit

Condition rating key:

Very Good	Good	Fair	Poor

Macrofauna

- Compared with other estuaries in the top of the South Island, the two fine scale sites had moderate levels of macrofaunal richness and lower abundances. AMBI index scores were rated as 'good'; however, species richness and abundance at both sites over 2013-2015 had markedly declined since the 2006 survey. This change was associated with a decline in sensitive macrofauna groups, which was related to a temporal increase in sediment mud content.

There is a risk that soft-sediment habitats in some parts of the estuary will reach a point at which the mud tolerance of key species (e.g. cockles, wedge shells) is exceeded, and their populations eventually decline. Such an outcome could have flow on effects to the wider ecosystem, for example due to a decline in important prey items for birds and fish. Elsewhere in the estuary, broad scale habitat mapping in 2019 revealed areas that were very muddy, and showed symptoms of high nutrient and organic enrichment. These included locations with excessive growths of opportunistic macroalgae species that can thrive in enriched muddy habitats. A recent study has highlighted activities associated with exotic forestry land use (in particular forest harvest) as being a key contributor of sediment to the Moutere Inlet. It is important that these activities and other potential sediment sources are managed so that the current state of the estuary is maintained or improved.

RECOMMENDATIONS

Comprehensive recommendations for further monitoring and investigations are provided in the report, which can be summarised as follows:

1. Undertake a desktop assessment to understand changes in catchment land use, and how such changes may affect inputs of stressors to the estuary. Specific related needs include as assessment of past, present and potential future inputs of fine (muddy) sediment, and links to exotic forest harvesting patterns.
2. Conduct targeted synoptic assessments of estuary condition in the vicinity of point source inputs and/or where local issues have already been identified. Such assessments should be based on the typical suite of NEMP indicators with the addition of an analysis of a full suite of priority pollutants including DDT.
3. Install three new sediment plate sites: (i) one in the central basin area, which appears (from previous observations) to experience relatively high sedimentation; and (ii) one at each of the two fine scale sites. Increase sampling effort at each sediment plate site to include measures of sediment grain size and enrichment status, and undertake sediment plate monitoring annually.
4. Repeat the fine scale survey in 2022 to determine changes since 2015. A reduction in macrofauna and sediment sampling effort as recommended in the main report should be considered, but the lab analysis of sediments should be expanded to encompass a full suite of priority pollutants, including DDT. Depending on the outcomes of the above, the potential for implementation of mitigation strategies to reduce future impacts should be considered.

1. INTRODUCTION

Monitoring the ecological condition of estuarine habitats is critical to their management. Estuary monitoring is undertaken by most councils in New Zealand as part of their State of the Environment (SOE) programmes. The most widely-used monitoring framework is that outlined in New Zealand's National Estuary Monitoring Protocol (NEMP; Robertson et al. 2002). The NEMP is intended to provide resource managers nationally with a scientifically defensible, cost-effective and standardised approach for monitoring the ecological status of estuaries in their region. The NEMP approach involves two main types of survey:

- Broad scale mapping of estuarine intertidal habitats. This type of monitoring is typically undertaken every 5 to 10 years.
- Fine scale monitoring of estuarine biota and sediment quality. This type of monitoring is typically conducted at intervals of 5 years after initially establishing a baseline.

One of the key additional methods that has been put in place subsequent to the NEMP being developed is 'sediment plate' monitoring. This component typically involves an annual assessment of patterns of sediment accretion and erosion in estuaries, based on changes in sediment depth over buried concrete pavers. Sediment plate monitoring stations are often established at NEMP fine scale sites, or nearby, to provide additional information for interpreting long-term changes.

The SOE programme of Tasman District Council (TDC) has included NEMP broad scale and fine scale surveys in estuaries across the region. One of these estuaries is Moutere Inlet (Fig. 1). The first NEMP surveys were conducted in 2006 (Clark et al. 2006; Gillespie & Clark 2006), which built on knowledge gained from earlier ecological investigations of the inlet and assessment of impacts of Talley's factory discharges in the northern end at Port Motueka (e.g. Forrest & Cooke 1995; Gillespie et al. 1995; Barter & Forrest 2001). Since the 2006 baseline, repeat NEMP broad scale surveys were undertaken in 2013 (Stevens & Robertson 2013) and 2019 (Stevens et al. 2020), with fine scale surveys in 2013 (Robertson & Stevens 2013), 2014 and 2015. Sediment plate monitoring has been undertaken annually by TDC staff since 2008.

Previous reports have summarised the above fine scale survey work up to and including the 2013 survey, with the data from the two surveys conducted since 2013 having been archived. In addition, although the sediment plate records were summarised in the broad scale report, these data have not been extensively analysed nor subject to QA checks. Accordingly, Salt Ecology was contracted to collate the results of all fine scale and sediment plate surveys conducted to date. This report describes the analyses undertaken, and evaluates spatial and temporal changes in key

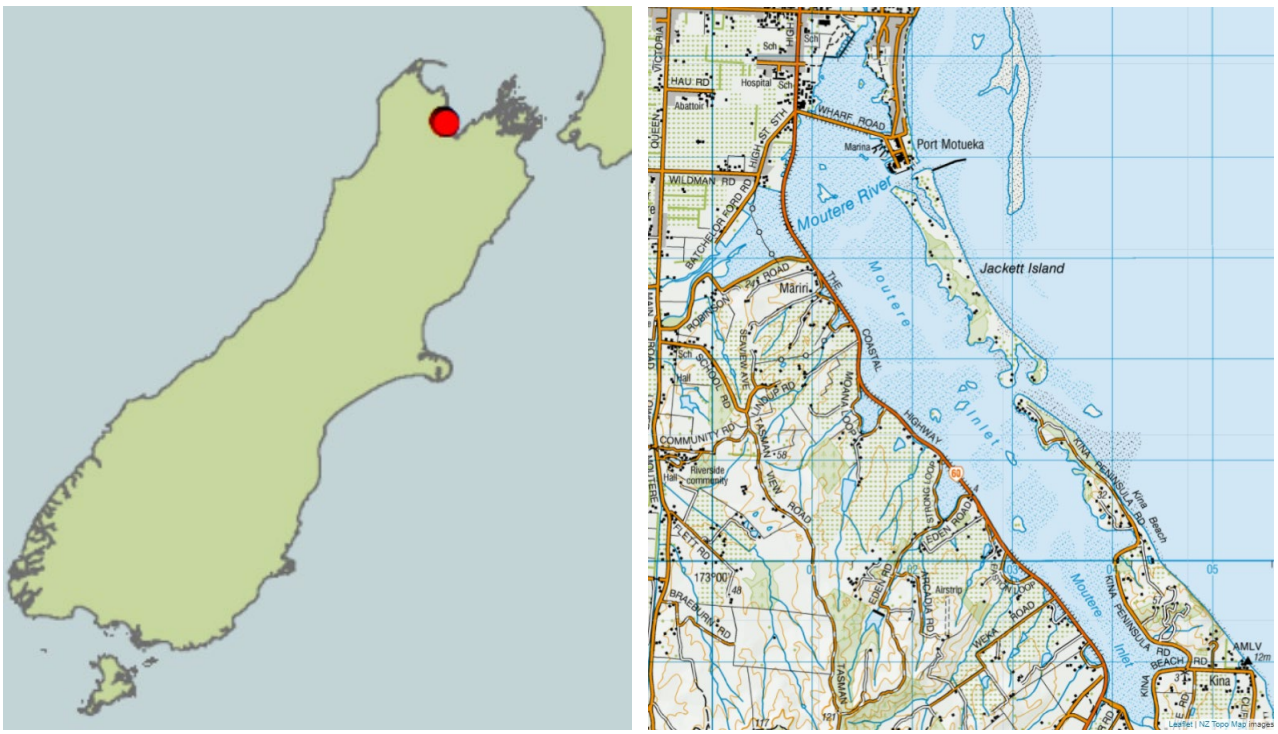


Fig. 1. Location of Moutere Inlet.

monitoring indicators. Findings are discussed in terms of estuary condition, and considered within the context of the historic studies that have been undertaken in Moutere Inlet. The management implications for Moutere Inlet are considered, as well as needs for ongoing monitoring and further investigation.

2. BACKGROUND TO MOUTERE INLET

The most recent broad scale survey report summarised background information on Moutere Inlet that was contained in previous reports. That information is presented below, and updated with the findings of the 2019 survey.

Moutere Inlet (Fig. 1) covers an area of 764ha, and is classified as a well-flushed, shallow, intertidally-dominated estuary (SIDE) located near Motueka. The estuary consists of one main basin with a tidal opening at each end of Jackett Island, and several tidal embayments separated from the main estuary basin by causeways. The estuary is shallow (mean depth ~2m) and almost completely drains at low tide. Intertidal habitats are characterised by wide sandflats and mudflats (many perched high in the tidal range), with steeply incised drainage channels, particularly near the entrances. These channels contain a variety of cobble, gravel, sand and biogenic (oyster, mussel, tubeworm) habitats, and support localised macroalgal growths. Although significantly reduced from their historical range, small patches of seagrass remain in the lower tidal reaches of the estuary, and salt marsh is present along the upper tidal margins.

The mean freshwater flow from the Moutere River in the northwestern corner of the estuary is quite low (<2m^3/s), with secondary inputs from several streams along the western side. Monthly water quality monitoring is conducted in the Moutere River, with the nearest site being ~2.5km upstream: see www.lawa.org.nz/explore-data/tasman-region/river-quality/moutere-river/moutere-at-riverside/. Results from water clarity monitoring (as an indicator of suspended fine sediment) place the river water in the best 25% of monitoring sites nationally. However, concentrations of the nutrient total nitrogen (which can contribute to excess algal growth in estuaries) place the river water in the worst 25-50% of monitoring sites.

The surrounding catchment (Fig. 2) is highly modified and dominated by pasture (53%), horticulture (15%), exotic forestry (12%) and built-up areas (2%), including the commercial port and marina located at Port Motueka. Native forest cover is low (2%) (Table 1).

Much of the terrestrial margin immediately adjacent to the estuary (70%) has been reclaimed or modified (by seawalls, roads, causeways), which has significantly displaced large areas of salt marsh, and also limits its ability to migrate inland in response to sea level rise. Stevens et al. (2020a) estimated that there has been a 45% reduction in salt marsh since 1947, with the remaining 83ha (~11% of the estuary area) dominated by rushland (55%) and herbfield (40%).

Despite historic changes the estuary remains valued for its aesthetic appeal, rich biodiversity, shellfish collection, swimming, waste assimilation, whitebaiting, fishing, boating, walking and scientific interest. It is recognised as a valuable nursery area for marine and freshwater fish and is regarded as a nationally important coastal area for birdlife. The key pressures have been identified as excessive areas of muddy sediment and increasing nutrient-related eutrophication. The latest broad scale report classified 31% of the intertidal area as mud-dominated habitat (sediment with >50% mud content). The report also described hot spots of nuisance macroalgae, with the extent of the estuary area exhibiting symptoms of excessive nutrient enrichment expanding from an estimated 0.1% of estuary area in 1947 to 4.1% (31ha) in 2019.

Table 1. Summary of catchment land cover (LCDB5 2018) for Moutere Inlet.

LCDB5 (2018) Class and Name		Ha	%
1	Built-up Area (settlement)	399	2.1
2	Urban Parkland/Open Space	56	0.3
5	Transport Infrastructure	69	0.4
6	Surface Mine or Dump	1	0.003
10	Sand or Gravel	24	0.1
20	Lake or Pond	76	0.4
21	River	4	0.02
22	Estuarine Open Water	48	0.3
30	Short-rotation Cropland	108	0.6
33	Orchard, Vineyard or Other Perennial Crop	2799	14.9
40	High Producing Exotic Grassland	9882	52.6
41	Low Producing Grassland	125	0.7
45	Herbaceous Freshwater Vegetation	9	0.05
46	Herbaceous Saline Vegetation	49	0.3
50	Fernland	0	0.002
51	Gorse and/or Broom	372	2.0
52	Manuka and/or Kanuka	74	0.4
54	Broadleaved Indigenous Hardwoods	104	0.6
56	Mixed Exotic Shrubland	29	0.2
64	Forest - Harvested	759	4.0
68	Deciduous Hardwoods	19	0.1
69	Indigenous Forest	369	2.0
71	Exotic Forest	3421	18.2
	Grand Total	18795	100

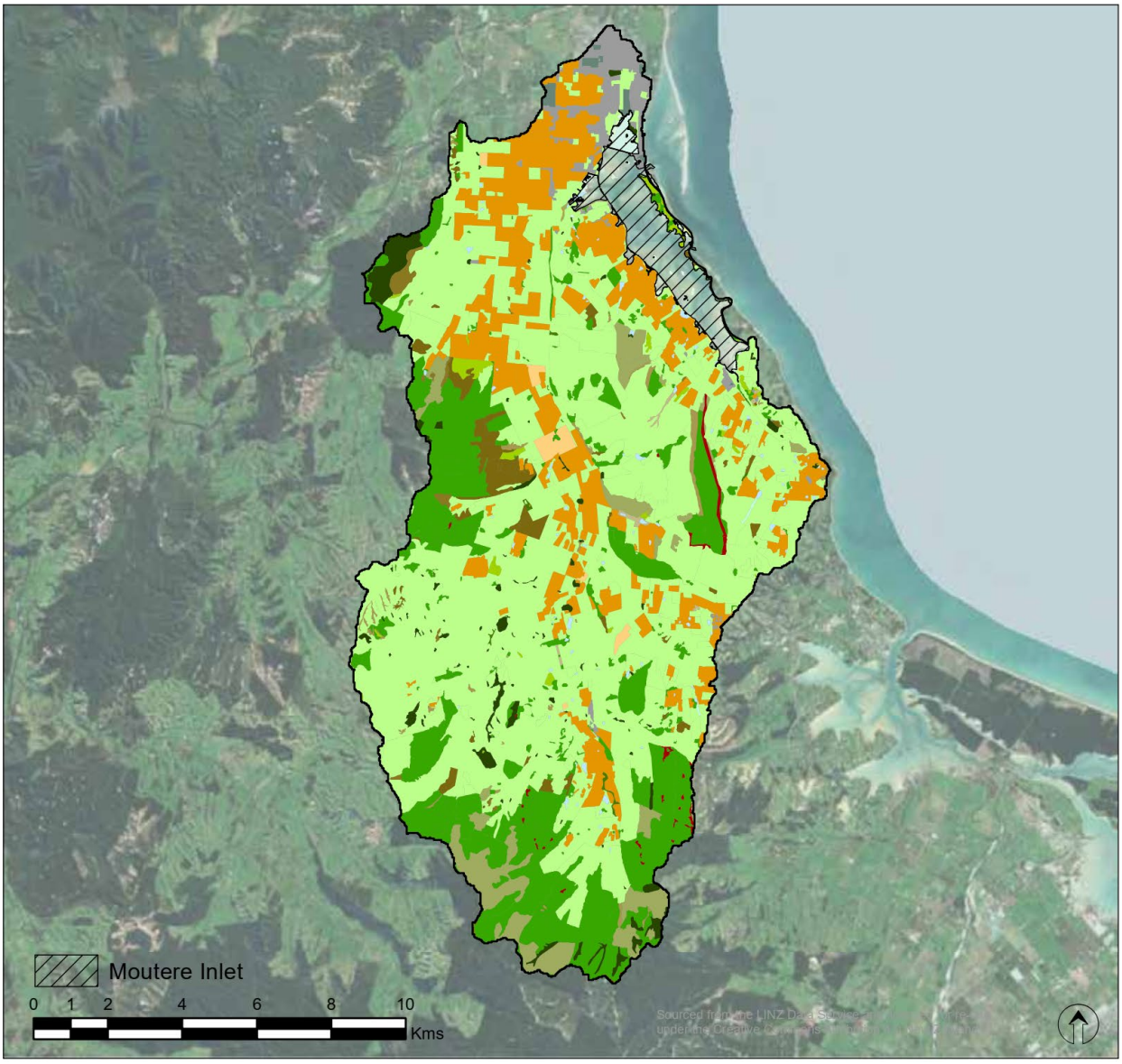


Fig. 2. Moutere Inlet (hatched area) and surrounding catchment land use classifications, LCDB5, 2018. Sourced from Stevens et al. (2020).

3. FINE SCALE METHODS

3.1 OVERVIEW OF NEMP FINE SCALE APPROACH

The NEMP advocates that fine scale monitoring is undertaken in soft sediment (sand/mud) habitat in the mid to low tidal range of priority estuaries, although seagrass habitats or highly enriched areas are sometimes included.

The environmental characteristics assessed in fine scale surveys incorporate a suite of common benthic indicators, including biological attributes such as the 'macrofaunal' assemblage and various physico-chemical characteristics (e.g. sediment mud content, trace metals, nutrients).

As well as the inclusion of sediment plate monitoring noted above, extensions to the original NEMP methodology that support the fine scale approach include the development of various metrics for assessing ecological condition according to prescribed criteria. These additional components are included in the present report.

3.2 MOUTERE FINE SCALE AND SEDIMENT PLATE SITES

The initial fine scale survey in March 2006 established two monitoring sites that were representative of the dominant muddy-sand substrate within the estuary. Both sites are of the recommended NEMP dimensions of 30 x 60m. Site locations are shown in Fig. 3, along with the seven locations where sediment plate monitoring is undertaken. Fine scale site boundaries and locations of sediment plates are marked with wooden pegs, with position data provided in Appendix 1. Note that the naming of sites used in this report follows the 2013 report, with Site A in the north of Moutere Inlet and Site B in the south. This is the reverse of the naming in the 2006 report. Table 2 summarises the sampling effort and provider undertaking each survey.

3.3 SEDIMENT PLATES AND SAMPLING

As well as providing a tool for understanding patterns of sediment accretion and erosion, sediment plate monitoring can aid interpretation of physical and biological changes at fine scale sites.

Four sediment plate sites were established by TDC staff in 2008, with three additional sites installed in 2013. The sediment plates consist of concrete pavers (19cm x 23cm), with four plates installed at each of site. TDC staff measured baseline depths (from the sediment surface to each buried plate) at the time of plate installation, and also undertook subsequent annual monitoring. To make measurements of sediment depth at each plate, a 2.5m long straight edge is placed over the plate position to average out any small-scale irregularities in surface topography. The depth to each plate is measured (at least in triplicate) by vertically inserting a probe into the sediment until it hits each plate, and measuring the penetration depth to the nearest mm.

Example of measuring sediment plate depth. A straight edge is used to account for small scale irregularities in the sediment surface. Depth is measured to the nearest millimeter (at least in triplicate) and recorded as an average per plate.

Table 2. Summary of fine scale sampling years, effort and provider. Replicate sample numbers are shown for macrofauna and sediment (sediment in brackets), indicating that for 2013 to 2015, three composite samples were collected for sediment. CMEC refers to Coastal Marine Ecology Consultants.

Year	Sampling dates	A	B	Field	Sorting	Taxonomy	Sediment analysis
2006	30-31 March	10 (10)	10 (10)	Cawthron	Cawthron	Cawthron	Cawthron
2013	21 March	10 (3)	10 (3)	Wriggle	Wriggle	CMEC	RJ Hill
2014	3 March	10 (3)	10 (3)	Wriggle	Wriggle	CMEC	RJ Hill
2015	5 February	10 (3)	10 (3)	Wriggle	Wriggle	CMEC	RJ Hill

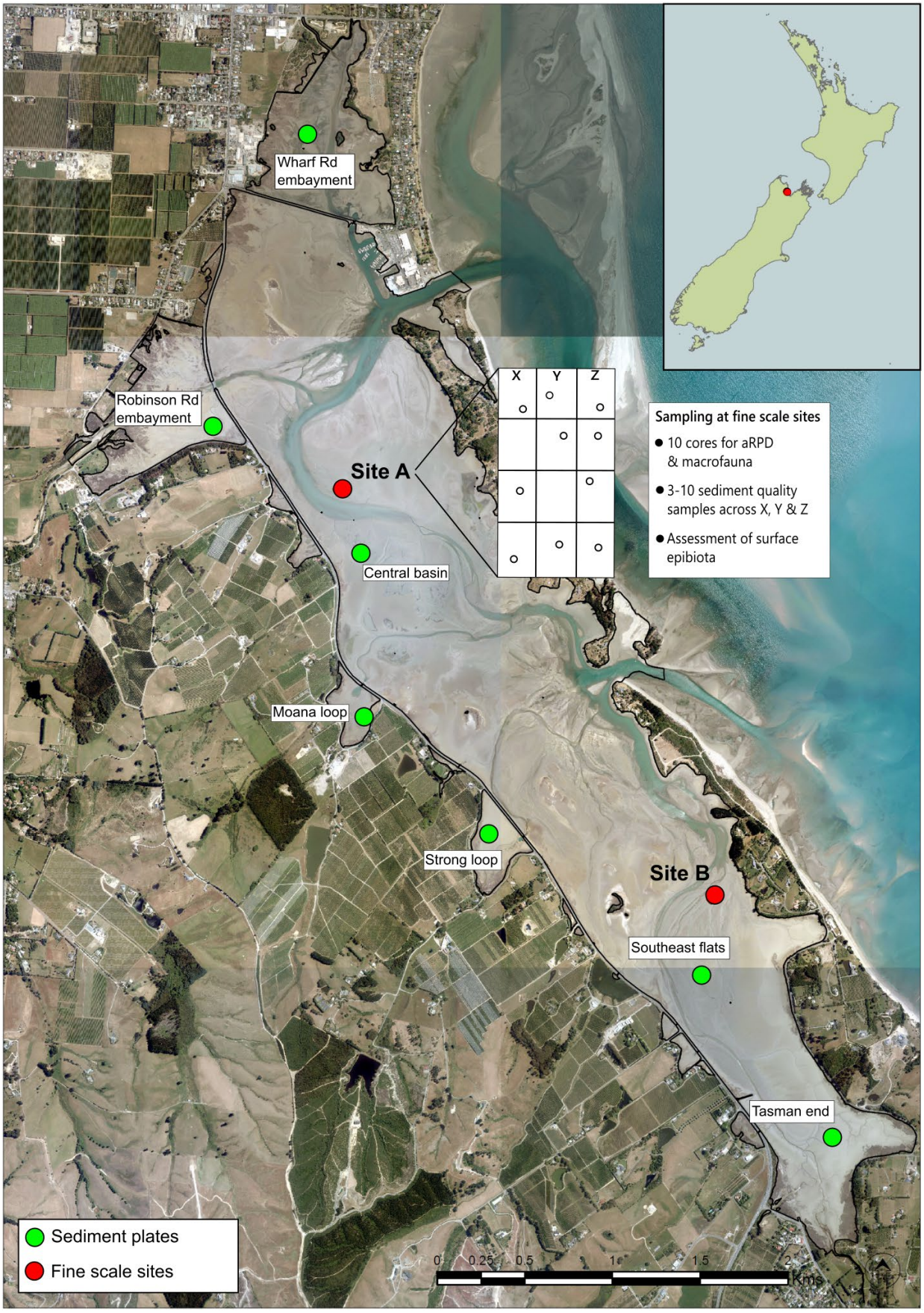


Fig. 3. Location of fine scale and sediment plate monitoring sites (A, B). Schematic illustrates fine scale sampling layout. Sampling consists of replicates taken randomly across cells in each of three vertical columns (represented by X-Z) in the grid.

3.4 FINE SCALE SAMPLING AND BENTHIC INDICATORS

Each fine scale site was divided into a 3 x 4 grid of 12 plots (see Fig. 3), with fine scale sampling for sediment indicators conducted in 10 of these plots. Fig. 3 illustrates the standard numbering sequence used in this report to describe the replicates at each site, and the designation of zones X, Y and Z (for compositing sediment samples; see below).

A summary of the benthic indicators, the rationale for their inclusion, and the field sampling methods, is provided in Table 4. Although the sampling approach across all years has generally adhered to the NEMP, alterations and additions to early NEMP methods have been introduced in most surveys conducted over the last 10 or more years. These modifications are reflected in the surveys conducted since 2013, as indicated in Table 4.

Sediments for physico-chemical analysis were collected as discrete samples (to ~20mm depth) within each plot in 2006, but in the later surveys the plot sub-samples were pooled within each of zones X, Y and Z (corresponding to replicates 1-3, 4-6 and 7-10, respectively; see Fig. 3) to provide three composite sediment samples (each ~250g). Samples were analysed by either Cawthron (2006) or RJ Hill Laboratories (2013-2015) and included the following analytes across all surveys: particle grain size in three categories (%mud <63µm, sand <2mm to ≥63µm, gravel ≥2mm); organic matter (either as % ash-free dry weight, AFDW, or total organic carbon, TOC); and nutrients (total nitrogen, TN; total phosphorus, TP). A suite of trace metals was measured in 2006 and 2013 (cadmium, Cd; chromium, Cr; copper, Cu; lead, Pb; nickel, Ni; zinc, Zn), with the metalloid arsenic (As) measured in 2014, along with a suite of organochlorine pesticides. Details of RJ Hill laboratory methods and detection limits are provided in Appendix 2, with Cawthron methods described in earlier reports. Note that %TOC was not measured in 2006, hence was estimated from %AFDW as: TOC = (0.4 * AFDW) + 0.0025 * AFDW^2.

Sediment oxygenation was assessed according to the approximate depth of the apparent Redox Potential Discontinuity (aRPD) (Table 4). The aRPD provides a subjective measure of the enrichment state of sediments according to the depth of the visible transition between oxygenated surface sediments (typically brown in colour) and deeper less oxygenated sediments (typically dark grey or black in colour).

To sample sediment-dwelling macrofauna, a large sediment core (130mm diameter, 150mm deep) was collected from each plot and gently washed through a 0.5mm sieve bag to remove fine sediment. The retained animals were preserved in a dilution of either formalin (2006) or isopropyl alcohol (2013-2015). The animals in each sample (macrofauna) were later picked out and identified to the lowest practical taxonomic level. The range of different macrofauna present (i.e. richness) and their abundance, are well-established indicators of ecological health in estuarine and marine soft sediments.

In addition to macrofaunal core sampling, the presence of epibiota (macroalgae and conspicuous surface-dwelling animals nominally >5mm body size) visible on the sediment surface were assessed at each site. In 2006 counts were made in 10 x 0.25m^2 quadrats, whereas in 2013-2015 epibiota were semi-quantitatively categorised using 'SACFOR' abundance (animals) or percentage cover (macroalgae) ratings as shown in Table 3. These ratings represent a scoring scheme simplified from established monitoring methods (MNCR 1990; Blyth-Skyrme et al. 2008).

The SACFOR method is suited to characterising intertidal epibiota with patchy or clumped distributions. It has been conducted since 2013 as an alternative to the quantitative quadrat sampling specified in NEMP, which is known to poorly characterise scarce or clumped species. For comparative purposes the quadrat data from the 2006 survey were expressed as SACFOR ratings.

Table 3. SACFOR ratings for site abundance and percent cover of epibiota and algae, respectively.

SACFOR category	Code	Density per m^2	Percent cover
Super abundant	S	> 1000	> 50
Abundant	A	100 - 999	20 - 50
Common	C	10 - 99	10 - 19
Frequent	F	2 - 9	5 - 9
Occasional	O	0.1 - 1	1 - 4
Rare	R	< 0.1	< 1

SACFOR epibiota assessment conducted since 2013 has not included infaunal species that may sometimes be visible on the sediment surface, but whose abundance cannot be reliably determined from surface observation (e.g. cockles).

Table 4. Summary of NEMP fine scale benthic indicators, rationale for their use, and sampling method. Any meaningful differences among surveys or with the NEMP protocol are described.

NEMP benthic indicators	General rationale	Sampling method
PHYSICAL AND CHEMICAL		
Sediment grain size	Indicates the relative proportion of fine-grained sediments that have accumulated.	1 x surface scrape to ~20mm sediment depth (see note 1).
Nutrients (nitrogen and phosphorus) and organic matter	Reflects the enrichment status of the estuary and potential for algal blooms and other symptoms of enrichment.	1 x surface scrape to ~20mm sediment depth (see note 1).
Trace metals (copper, chromium, cadmium, lead, nickel, zinc)	Common toxic contaminants generally associated with human activities.	1 x surface scrape to ~20mm sediment depth (see notes 1, 2).
Depth of apparent Redox Potential Discontinuity layer (aRPD)	Subjective time-integrated measure of the enrichment state of sediments according to the visual transition between oxygenated surface sediments and deeper deoxygenated black sediments. The aRPD can occur closer to the sediment surface as organic matter loading increases.	Extraction of a sediment core for each plot, split vertically, with depth of aRPD recorded in the field where visible.
BIOLOGICAL		
Macrofauna	The abundance, composition and diversity of macrofauna, especially the infauna living with the sediment, are commonly-used indicators of estuarine health.	1 x 130mm diameter sediment core to 150mm deep (0.013m^2 sample area, 2L core volume) for each of 10 plots, sieved to 0.5mm to retain macrofauna.
Epibiota (epifauna)	Abundance, composition and diversity of epifauna are commonly-used indicators of estuarine health.	Quadrat sampling in 2006 or SACFOR scale (see Table 3) since 2013 (see note 3).
Epibiota (macroalgae)	The composition and prevalence of macroalgae are indicators of nutrient enrichment.	Quadrat sampling in 2006 or SACFOR scale (see Table 3) since 2013 (see note 3).
Epibiota (microalgae)	The composition and prevalence of microalgae are indicators of nutrient enrichment.	Measurement of sediment chlorophyll-a as a biomass indicator (2006) and/or visual assessment of conspicuous growths (see note 4).

[1] For reasons of cost and low sample variance, since 2013 sediment quality has been assessed in 3 composite samples rather than 10 discrete samples as specified in the NEMP and collected in the 2006 survey.

[2] Arsenic and mercury were not originally included in the NEMP because of cost constraints, but have been included as part of a standard RJ Hill trace element suite in more recent years.

[3] Assessment of epifauna and macroalgae has used SACFOR since 2013, in favour of quadrat sampling outlined in NEMP and undertaken in earlier surveys. Quadrat sampling is subject to considerable within-site variation for epibiota that have clumped or patchy distributions.

[4] NEMP recommends taxonomic composition assessment for microalgae, but this is not typically undertaken due to unavailability of expertise and lack of demonstrated utility of microalgae as a routine indicator.

3.5 DATA RECORDING, QA/QC AND ANALYSIS

As indicated in Table 2, different providers have been involved in field work, sample processing and taxonomic or sediment analysis since 2006. As such, to ensure data comparability to the extent possible, various data filtering and QA procedures were undertaken as described below.

Rather than using previous data summaries, raw excel data sheets were obtained for all surveys and imported into the software R 4.0.5 (R Core Team 2021) and merged by common sample identification codes. All summaries of univariate responses (e.g. totals, means ± 1 standard error) were produced in R, including tabulated or graphical representations of data from sediment plates, laboratory sediment quality analyses, and macrofauna. Where results for sediment quality parameters were below analytical detection limits, averaging (if undertaken) used half of the detection limit value, according to convention.

Before macrofaunal analyses, the data were screened to remove species that were not regarded as a true part of the macrofaunal assemblage; these were planktonic life-stages and non-marine organisms (e.g. terrestrial beetles). To enable comparisons with future surveys, and other regional estuaries, cross-checks were made to ensure consistent naming of species and higher taxa to the extent feasible. For this purpose, the adopted name was that accepted by the World Register of Marine Species (WoRMS, www.marinespecies.org/). As appropriate, taxonomic naming revisions to CMEC data collected since 2013 were made, based on limited retrospective taxonomic verification undertaken by NIWA on reference samples (Appendix 3a).

The QA process could not be applied to the Cawthon samples collected in 2006. However, this situation does not negate comparison of species richness and abundance across years, but meant that taxonomic aggregation to common groups needed to be undertaken for multivariate analyses (see below). Similarly, scores for the biotic health index AMBI (Borja et al. 2000) were calculated and compared across years. AMBI scores are derived from the proportion of taxa falling into one of five eco-groups (EG) that reflect sensitivity to pollution (in particular eutrophication), ranging from sensitive (EG-I) to relatively resilient (EG-V). The approach used for AMBI calculation is described in previous Salt Ecology reports (e.g. Forrest & Stevens 2021).

Examples from other estuaries of collecting (top) and sieving (bottom) sediment macrofauna cores.

Multivariate analysis of the macrofaunal community data were undertaken using methods detailed in previous reports such as cited above. An initial Jaccard similarity analysis of the raw data (based on species presence and absence, irrespective of abundance) revealed temporal differences that were considered likely to reflect taxonomic inconsistencies between the surveys of Cawthron and CMEC (Appendix 3b). As such, before further macrofaunal community analysis, it was necessary to aggregate some of the species or taxa to higher groups (e.g. genus, family, phylum). Appendix 3c provides information on the taxonomic aggregation undertaken. Following this step, the main analyses undertaken were as follows (see detail in Appendix 3d):

- A non-metric multidimensional scaling (nMDS) ordination, based on pairwise Bray-Curtis similarity index scores among samples (data were square-root transformed) aggregated within each site and sampling year. This approach produced a plot that

could be used to visually assess macrofaunal community composition similarity among sites and survey years.

- Various approaches that aimed to help understand whether changes in macrofauna were related to the measured sediment quality variables, including:
  - Overlay vectors and bubble plots were used to visualise relationships between multivariate biological patterns and sediment quality data.
  - Use of an analytical procedure (Bio-Env) to evaluate the suite of sediment quality variables that were most closely correlated with the macrofauna similarity pattern (see Forrest & Stevens 2021).
  - Calculation of Benthic Health Model (BHM) scores in relation to sediment mud and metals (copper, lead, zinc) content, based on the national BHM described by Clark et al. (2020).

Calculation of BHM scores required a different species aggregation scheme to that described for the nMDS analysis above, as the method is prescriptive about the level of taxonomic resolution that is necessary (see Appendix 3c).

3.6 ASSESSMENT OF ESTUARY CONDITION

To supplement our analyses and interpretation of the data, results for all surveys were assessed within the context of established or developing estuarine health metrics ('condition ratings'), drawing on approaches from New Zealand and overseas (FGDC 2012; Townsend & Lohrer 2015; Robertson et al. 2016; ANZG 2018). These metrics assign different indicators to one of four rating bands, colour-coded as shown in Table 5. The origin and derivation of these metrics and most of the rating bands is also described in Forrest and Stevens (2021).

The ETI scoring categories described in Table 5 should be regarded only as a general guide to assist with interpretation of estuary condition. It is major spatio-temporal changes in the categories that are of most interest, rather than their subjective condition descriptors; i.e. descriptors such as 'poor' condition should be regarded more as a relative rather than absolute rating. For present purposes, our assessment of the multi-year data against the rating thresholds is

Table 5. ETI condition ratings used to characterise Moutere Inlet health for key indicators. See footnotes and other Salt Ecology reports (e.g. Forrest & Stevens 2021) for explanation of the origin or derivation of the different metrics. Benthic Health Model bands are not included in the Table as they are on a different scale (see Methods Section 3.6).

Indicator	Unit	Very good	Good	Fair	Poor
General indicators [1]					
Sedimentation rate[a]	mm/yr	< 0.5	≥0.5 to < 1	≥1 to < 2	≥ 2
Mud content[b]	%	< 5	5 to < 10	10 to < 25	≥ 25
aRPD depth[c]	mm	≥ 50	20 to < 50	10 to < 20	< 10
TN[b]	mg/kg	< 250	250 to < 1000	1000 to <	≥ 2000
TOC[b]	%	< 0.5	0.5 to < 1	1 to < 2	≥ 2
AMBI[b]	na	0 to 1.2	> 1.2 to 3.3	> 3.3 to 4.3	> 4.3
Trace elements [2]					
As	mg/kg	< 10	10 to < 20	20 to < 70	≥ 70
Cd	mg/kg	< 0.75	0.75 to <1.5	1.5 to < 10	≥ 10
Cr	mg/kg	< 40	40 to <80	80 to < 370	≥ 370
Cu	mg/kg	< 32.5	32.5 to <65	65 to < 270	≥ 270
Hg	mg/kg	< 0.075	0.075 to <0.15	0.15 to < 1	≥ 1
Ni	mg/kg	< 10.5	10.5 to <21	21 to < 52	≥ 52
Pb	mg/kg	< 25	25 to <50	50 to < 220	≥ 220
Zn	mg/kg	< 100	100 to <200	200 to < 410	≥ 410

[1] Ratings derived or modified from: [a]Townsend and Lohrer (2015), [b]Robertson et al. (2016) with modification for mud content described in text, [c]FGDC (2012).

[2] Trace element thresholds scaled in relation to ANZG (2018) as follows: Very good = < 0.5 x DGV; Good = 0.5 x DGV to < DGV; Fair = DGV to < GV-high; Poor = > GV-high. DGV = Default Guideline Value, GV-high = Guideline Value-high. These were formerly the ANZECC (2000) sediment quality guidelines whose exceedance roughly equates to the occurrence of 'possible' and 'probable' ecological effects, respectively.

based on site-level mean values for the different parameters.

In the case of the BHM scores, ETI rating bands have not been established as the method is relatively new. Instead the Mud BHM scores are rated according to Clark et al. (2020) against a five-point scale. The scale simply divides the possible BHM scores of 1-6 across even rating bands that reflect a 'very low' to 'very high' impact relative to other New Zealand estuaries as follows: 1 to <2 (very low), 2 to <3 (low), 3 to <4 (moderate), 4 to <5 (high) and 5 to 6 (very high). Metals BHM scores are rated against an absolute effects scale described by Clarke (2022, unpublished Cawthron report), which categorises sediment health as 'good', 'fair' or 'poor' when assessed against a suite of sediment quality guidelines that are more conservative than the DGV thresholds of ANZG (2018).

4. KEY FINDINGS

4.1 GENERAL FEATURES OF FINE SCALE SITES

The two sites are typical of the main intertidal habitats present in Moutere Inlet, being superficially uniform and relatively barren tidal flats consisting of sand-dominated sediment with variable amounts mud and of shell hash (see photos below). Pock marks and holes in the sediment surface reveal the presence of various burrowing organisms such as crabs, which play an important role in turning over the sediment ('bioturbation') and providing oxygenated water to deeper layers.

Fine scale Site A (2015). Source Wriggle Coastal Management.

Fine scale Site B (2015). Source Wriggle Coastal Management.

4.2 SEDIMENT PLATES

The summary Figure and Table in Appendix 4 reveal highly variable but low levels of sediment accrual across the sites, with marked recent erosion at the southeast basin and north embayment (Wharf Rd) sites. Maximum sedimentation was 0.54mm/yr (Moana loop), which is less than the 2mm/yr national guideline value. There appeared to be a large deposition event in the central basin between 2014 and 2015, followed by subsequent erosion (Appendix 4). Similarly, steady erosion of deposited mud at the SE Basin site has resulted in a highly fractured and lumpy sediment surface.

4.3 SEDIMENT CHARACTERISTICS

4.3.1 Sediment grain size, TOC and nutrients

Raw data on sediment characteristics are tabulated in Appendix 5. Laboratory analyses of sediment grain size highlighted the main habitat features described above. Fig. 4 shows sediments that are sand-dominated with a negligible gravel component, and a mean mud content ranging from ~10-25%. Illustrative photos of the sediments are provided in Fig. 6. Except for 2014, there is an overall trend of increasing mud from 2006 to 2015, in particular at Site A where mud content more than doubled over that period.

To provide a visual comparison of sediment quality relative to the Table 5 condition ratings, Fig. 5 compares the mean percentage mud, total organic carbon (TOC) and total nitrogen (TN) from fine scale sites against the rating thresholds. Due to average mud falling in the 10-25% range, sites were consistently rated as 'fair' in all years.

Total organic carbon (TOC) and total nitrogen (TN) values were, in almost all instances, rated 'good' or 'very good'. TOC was slightly elevated in 2013, but has reduced since then. TN was elevated at Site B in 2015, but nonetheless still at a low level (rated 'good'). These trophic state indicators provide no evidence for significant enrichment, or an increase in enrichment over time.

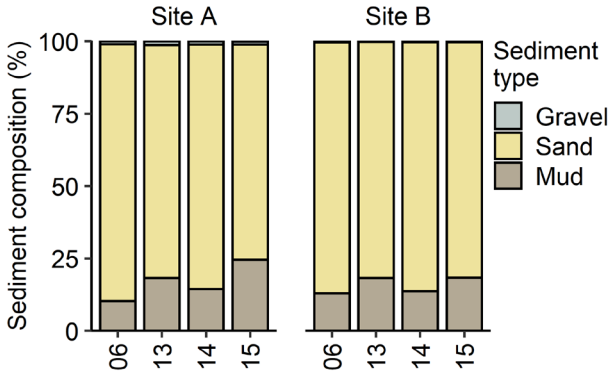


Fig. 4. Sediment particle grain size analysis showing percentage composition of mud (<63µm), sand (<2mm to ≥63µm) and gravel (≥2mm).

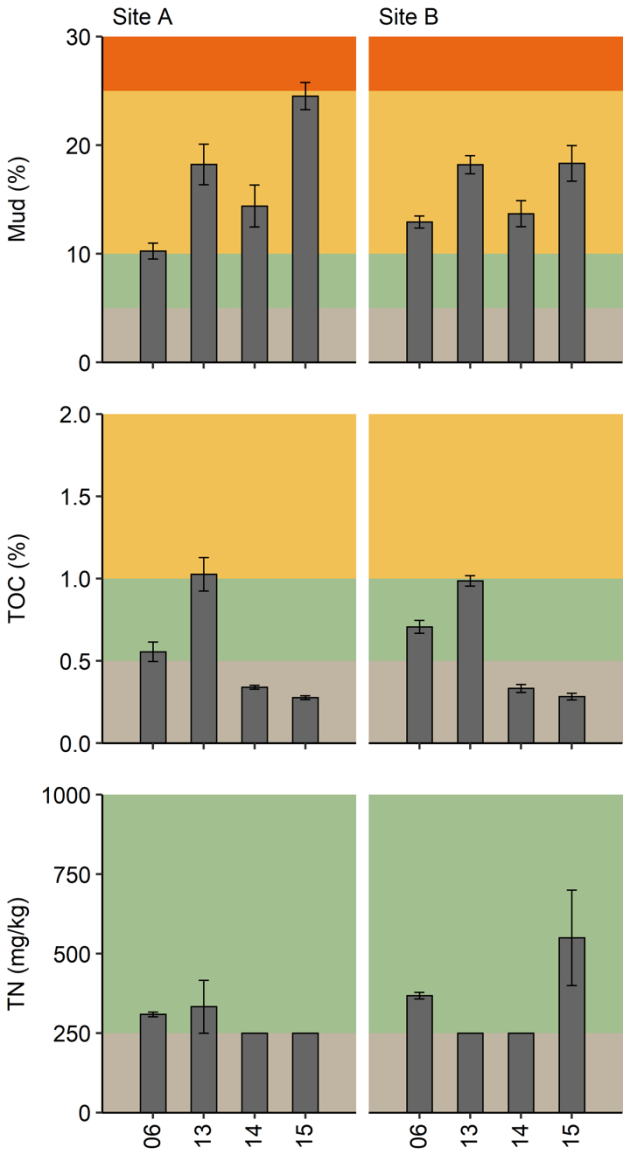


Fig. 5. Mean (±SE, n=3) sediment %mud, total organic carbon, and total nitrogen relative to condition ratings. Note that TOC for 2006 was estimated from ash-free dry weight data.

Condition rating key:

4.3.2 Redox status

There was an apparent trend for aRPD to become shallower at Site B over time, while at Site A the aRPD has been highly variable (Fig. 7). The measured values could be interpreted as indicating moderate enrichment, with aRPD depths ≤10mm rated as 'poor'. The apparent shallowing of the aRPD may be attributable to reduced oxygen penetration into the sediment matrix due to increased surface mud, as opposed to enrichment *per se*, given that TOC levels are

low and have reduced since 2013. However, the temporal pattern may also reflect sampling and measurement variation. For example. whereas aRPD values were measured (to the nearest mm) by Gillespie and Clark (2006), the 2013-2015 values appear to be 'ballpark' estimates. Unfortunately, there are no archived photographs of sediment cores from 2013-2015 that would have enabled retrospective assessment of aRPD depth for comparison with 2006.

This situation, combined with the inherently subjective nature of the aRPD assessment method (i.e. there is considerable judgement in assessing an exact depth of the aRPD), means that little weight can be placed on the temporal changes apparent in Fig. 7. Despite this, of most importance is that neither of the sites showed evidence of black anoxic (and sulphide-smelling) sediments at (or within a few millimetres of) the sediment surface, such as would occur under strongly enriched conditions.

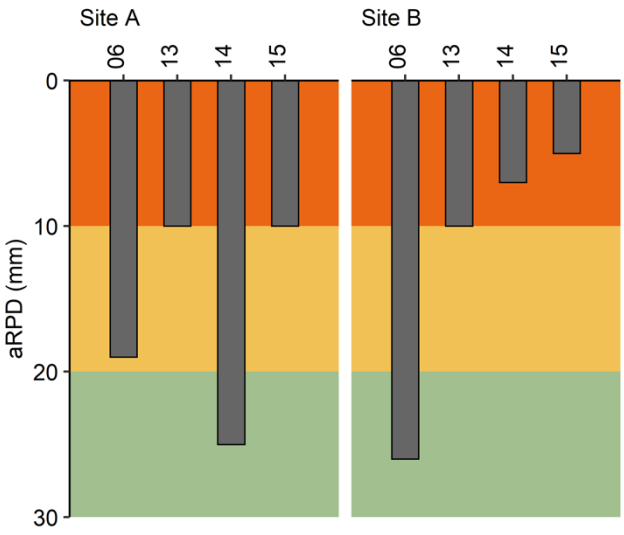


Fig. 7. aRPD values relative to condition ratings.

Condition rating key:

Very Good	Good	Fair	Poor

Site A

Site B

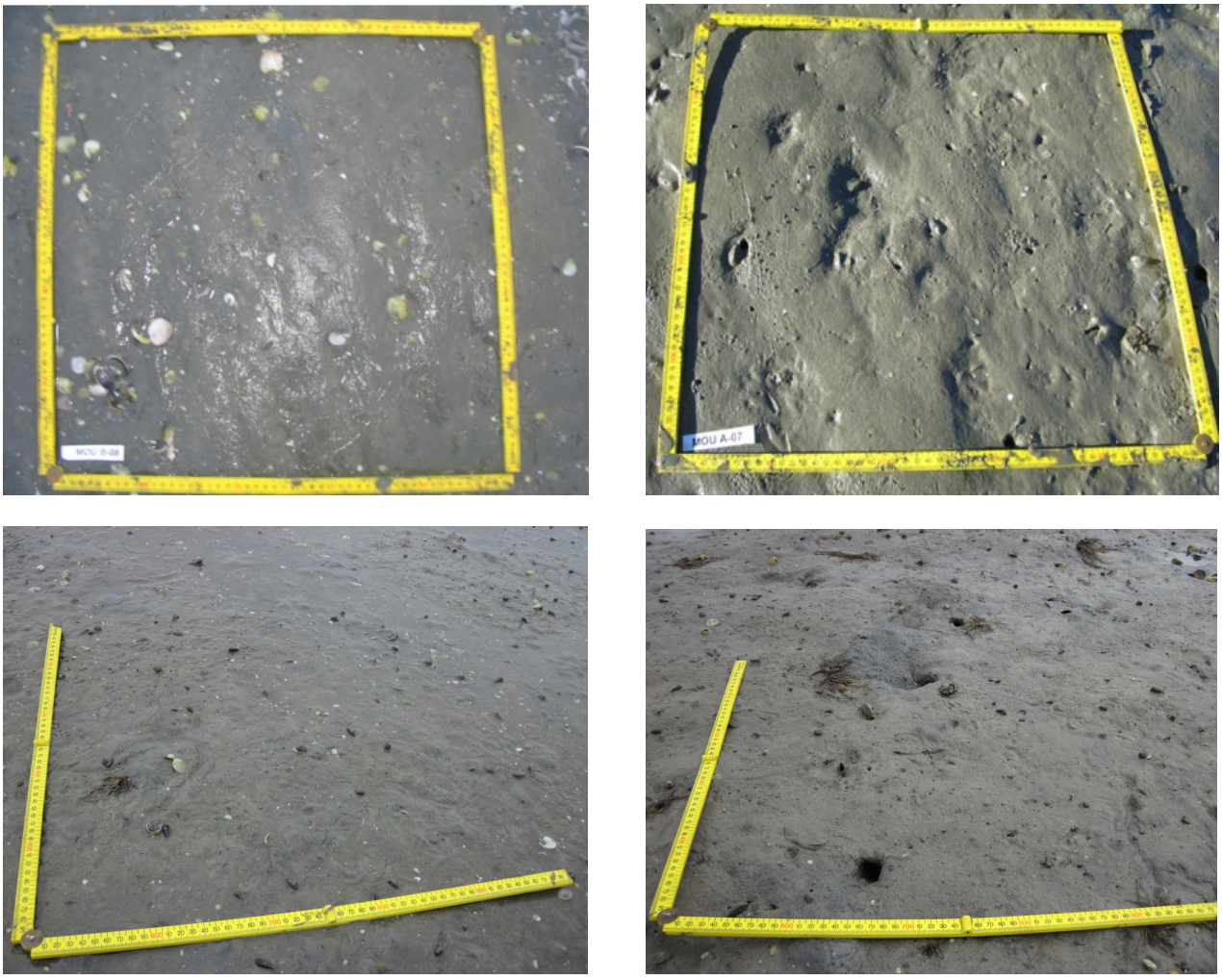


Fig. 6. Example of sediments from each site in 2006 (top) and 2013 (bottom). Sourced from Gillespie and Clark (2006) and Wriggle Coastal Management. Note 2006 site labels are opposite to those used in 2013-2015 and in the current report.

4.3.3 Trace contaminants

Plots of trace metal contaminants in relation to condition ratings are provided in Fig. 8 (see also Appendix 5).

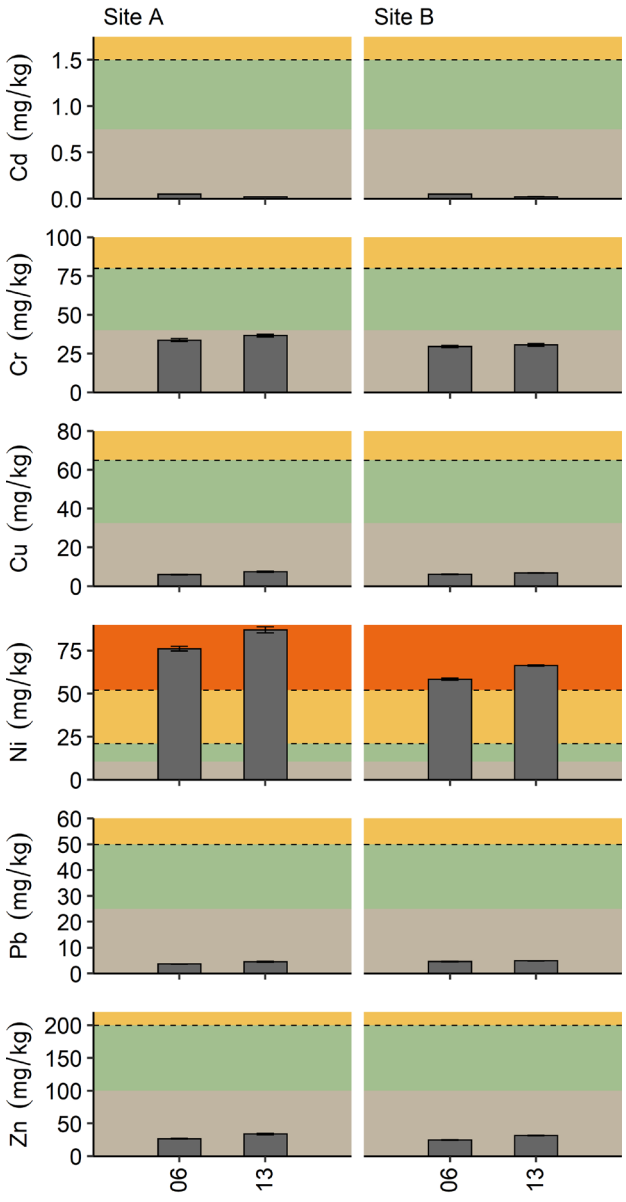


Fig. 8. Mean (±SE, n=3-12) trace element concentrations relative to condition ratings. Dotted line indicates national DGV for sediment quality.

Condition rating key:

Very Good	Good	Fair	Poor

The main impression from Fig. 8 is that, with the exception of nickel (Ni), trace element concentrations are very low and rated as 'very good', reflecting that they were less than half of the ANZG (2018) Default Guideline Value (DGV) for 'possible' ecological effects.

Mean nickel concentrations were rated as 'poor' on all sampling occasions, as they exceeded GV-high values. Similarly high nickel levels have been recorded in Waimea Inlet (Forrest et al. 2022) and in the nearshore subtidal sediments of western Tasman Bay (Forrest et al. 2007). Such findings reflect inputs from the catchment, due to naturally-occurring high concentrations of nickel and certain other trace contaminants in catchment soils (Rattenbury et al. 1998).

The only other contaminant monitoring undertaken as part of the NEMP surveys was in 2014, and consisted of sediment analysis for the metalloid arsenic (As), and for 114 organochlorine pesticides. Those analyses revealed arsenic to be 4-times lower than the DGV of 20mg/kg (Appendix 5), with all organochlorine pesticides except dichloro-diphenyl-trichloroethane (DDT) being less than method detection limits.

In the case of DDT, detectable concentrations of 4,4'-DDT, when normalised to 1% TOC (to enable comparison with national guidelines), equated to 5 and 3.9µg/kg at Sites A and B, respectively. Applying the convention of using half of the method detection limit value for constituents comprising total DDT (4,4'-DDT, 2,4'-DDT, and breakdown compounds DDD and DDE), the TOC-normalised concentrations equate to 12.4 and 11.4µg/kg at Sites A and B, respectively. These values exceed the DGV of 1.2µg/kg by a factor of ten, and are approximately twice the GV-high value of 5 µg/kg. The environmental significance of these results and potential sources of DDT are discussed in Section 5.1.

4.4 MACROFAUNA

4.4.1 Conspicuous surface epibiota

Results from the site-level assessment of surface-dwelling epibiota are compared across surveys in Table 6. Conspicuous epibiota consisted of three estuarine snail species and two species of common macroalgae, green 'sea lettuce' *Ulva* spp. and the red seaweed *Agarophyton chilense*. These two macroalgae were patchy across the sites, ranging from 'rare' (R, <1% cover) to 'frequent' (F, 5-9% cover).

The most widespread and commonly occurring snails were the horn snail *Zeacumantus lutulentus*, and the mudflat topshell *Diloma subrostratum* (see photos). Both of these species were rated as common (C, 10-99/m^2) on most sampling occasions. The mud whelk *Cominella glandiformis* (not recorded in 2006) was widespread but at relative low densities. As well as these

visible epibiota, crab holes, small burrows and mud casts also provided evidence of biological activity in the sediment.

Overall, epibiota density and cover varied somewhat among sites and surveys. This situation highlights their limited utility as a quantitative fine scale indicator, with the semi-quantitative SACFOR approach adequate for epibiota characterisation.

4.4.2 Macrofauna cores

Main taxonomic groups and species

The species recorded represented 13 main taxonomic groups. The most well-represented in terms of species richness were polychaete worms, with bivalve shellfish and gastropods (estuarine snails) also reasonably species-rich and abundant (Fig. 9).

Richness, abundance and AMBI

A total of 44 species or higher taxa of sediment dwelling macrofauna were sampled by Cawthron in 2006, compared with 45 described by CMEC over 2013-2015 (Appendix 6). Table 7 and Table 8 describes the most commonly occurring species or higher taxa that were recorded.

Mean species richness ranged from ~8 to 17 taxa per core sample (Fig. 10a). The most dramatic pattern evident in Fig. 10a,b is the marked decline in macrofaunal richness and abundance at both sites between 2006 and 2013, after which there was a gradual increase, perhaps indicating a recovery from disturbance between 2006 and 2013. Nonetheless, in 2015, richness and abundance values were still appreciably less than in 2006. These findings may in part reflect differences among the providers that did the work (see Table 2), but more likely reflect true temporal differences. As well as environmental disturbances (e.g. relating to sediment pulses), the differences may reflect natural ecological process (e.g. high macrofauna 'recruitment' prior to the 2006 survey). See discussion below.

The most widely occurring and abundant epibiota were horn snails, *Zeacumantus lutulentus* (top), and mudflat topshells, *Diloma subrostrata* (bottom). Images courtesy of Andrew Spurgeon (www.mollusca.co.nz).

Table 6. SACFOR scores for epibiota over the three surveys, based on the scale in Table 3. Dash = not recorded. For 2006 data, SACFOR ratings were scaled from quadrat counts.

Species	Common name	Functional description	A 2006	A 2013	A 2014	A 2015	B 2006	B 2013	B 2014	B 2015
Agarophyton chilense[1]	Red seaweed	Primary producer	-	R	O	O	-	O	O	O
Cominella glandiformis	Mud whelk	Carnivore and scavenger	-	R	O	O	-	R	O	O
Diloma subrostrata	Mudflat topshell	Grazer and deposit feeder	F	C	C	C	C	F	C	C
Ulva spp.	Sea lettuce	Primary producer	-	O	O	F	-	O	-	O
Zeacumantus lutulentus	Horn snail	Microalgal and detrital grazer	F	C	C	C	C	C	C	C

[1] *Agarophyton chilense* is the revised name for *Gracilaria chilensis*

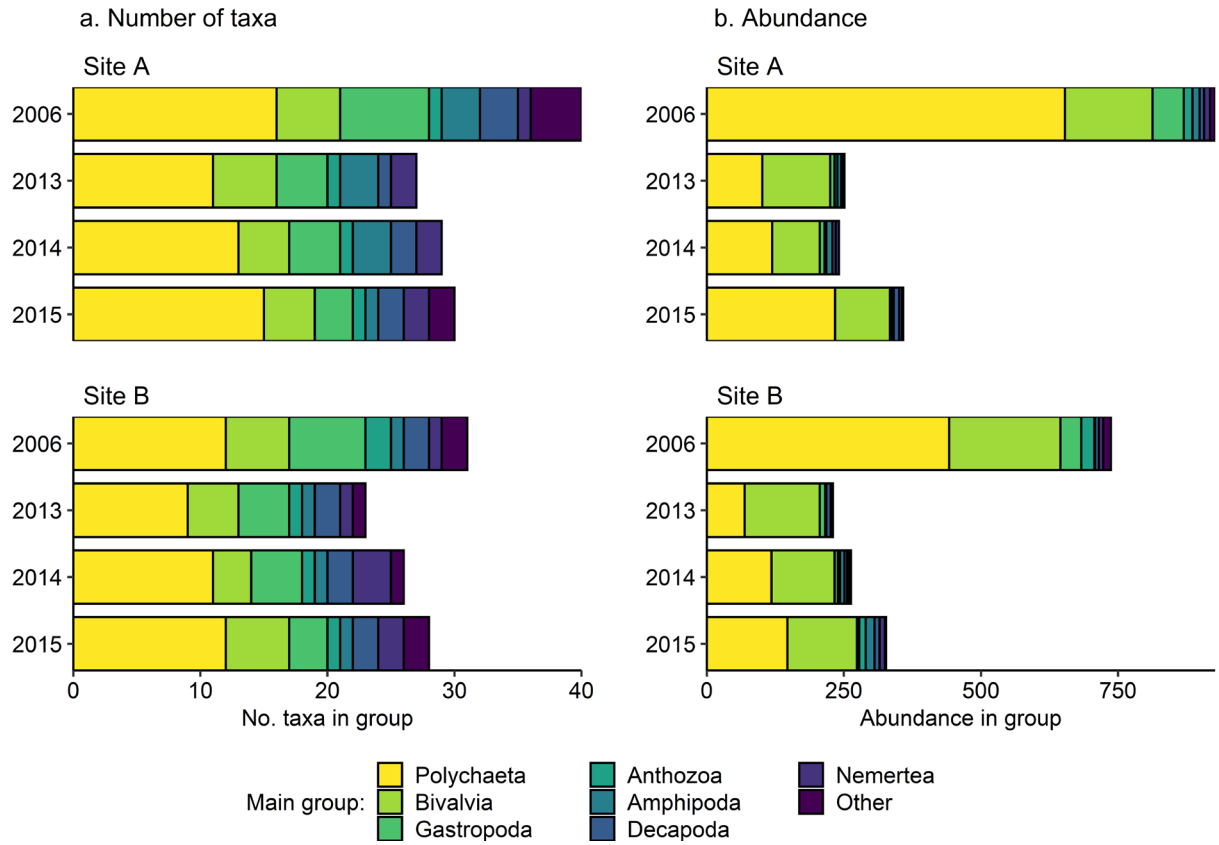


Fig. 9. Pooled data showing the contribution of main taxonomic groups to site richness and abundance.

Table 7. Sediment-dwelling species that comprised ≥5% of total abundance at any one site. The Table shows site abundances pooled across cores. The eco-group (EG) sensitivity on a scale from highly sensitive (I) to highly tolerant (V) is also indicated.

Main group	Taxa	EG	A 2006	A 2013	A 2014	A 2015	B 2006	B 2013	B 2014	B 2015
Amphipoda	Amphipoda various[1]	II	13	7	11	2	1	1	8	16
Bivalvia	*Austrovenus stutchburyi*	II	31	29	15	33	108	69	41	50
Bivalvia	*Linucula hartvigiana*	II	35	22	26	11	16	10	20	26
Bivalvia	*Macomona liliana*	II	81	71	44	55	67	57	54	48
Gastropoda	*Zeacumantus* spp.	I to II	40	3	3	-	24	4	1	1
Polychaeta	*Axiothella serrata*	II	-	7	12	25	-	2	3	1
Polychaeta	*Heteromastus filiformis*	III	75	6	13	23	177	17	9	25
Polychaeta	*Nicon aestuariensis*/ Nereididae (juv)[2]	III	30	6	8	14	-	6	18	11
Polychaeta	Paraonidae[3]	III	178	-	1	16	8	-	2	2
Polychaeta	*Prionospio aucklandica*[4]	II	333	68	67	112	205	38	67	86

[1] Amphipoda were mainly unnamed species, with the most prevalent identified by CMEC being the phoxocephalid *Torridoharpinia hurleyi*.

[2] Juvenile nereididae likely to be *Nicon aestuariensis*

[3] Cawthron paraonidae was identified by CMEC as a single species (*Paradoneis* sp.)

[4] *Prionospio aucklandica* is assumed the same species as Cawthron's *Prionospio* sp.

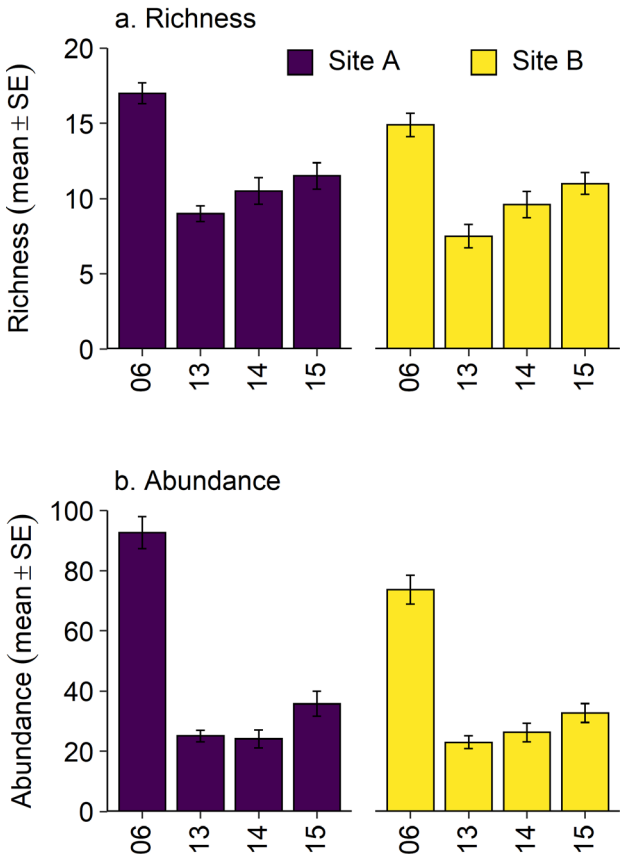


Fig. 10. Patterns (mean ± SE) in taxon richness and abundance per core sample.

Despite the richness and abundance patterns, values of the biological index AMBI were reasonably similar across years (hence providers) and rated as indicative of 'good' or 'very good' estuary health at all sites (Fig. 11).

The AMBI scores reflect a high prevalence of species or higher taxa classified as EG-II, being those eco-groups regarded as relatively sensitive to enrichment and other types of environmental pollution (Fig. 12). For example, across the dataset of 68 taxa with eco-groups assigned, 26 taxa were EG-II (Appendix 6). Some of the EG-II species were notably widely-occurring and abundant, such as cockles (*Austrovenus stutchburyi*), wedge shells *(Macomona liliana*), nut shells (*Linucula hartvigiana*) and the spionid worm *Prionospio aucklandica* (Table 7). Many of these abundant macrofauna are known to be important prey items for birds, fish and rays.

Of interest is that some of the EG-II taxa (notably paraonid worms) declined greatly in abundance after 2006. Similarly, there was a decline in highly sensitive EG-I taxa. There were 14 EG-1 taxa whose low abundances did not strongly influence the AMBI score. However, Fig. 12 reveals that the most marked presence-absence change from 2006 to 2013-15 was a reduction in the number of EG-I taxa. In fact, there was a 55% loss of EG-I taxa comparing 2006 with subsequent surveys.

The reasons for the apparent disappearance or abundance decline in these sensitive taxa may relate to environmental degradation. However, the situation is confounded by two other observations: (i) abundance declines (of similar magnitude) of moderately hardy taxa, notably EG-III paraonid worms, and the EG-III capitellid worm *Heteromastus filiformis* (which can thrive in degraded conditions); and (ii) the observation that the most hardy EG-IV and EG-V species (e.g. the small bivalve *Arthritica* sp. 1, and mud crabs *Austrohelice crassa* and *Hemiplax hirtipes*) were few in number, and did not become more prevalent over time (Appendix 6).

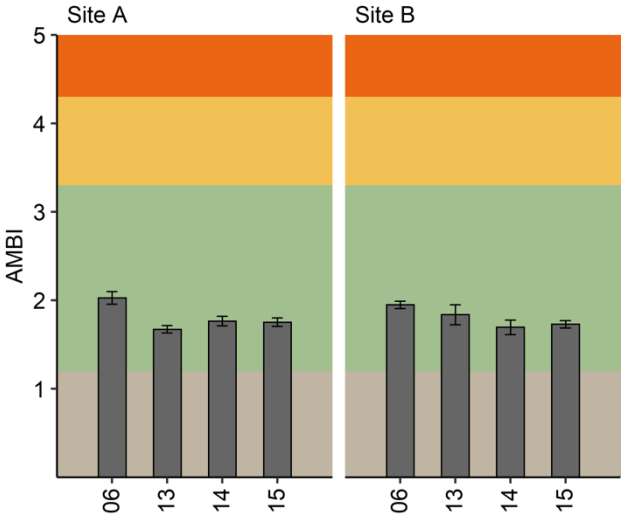
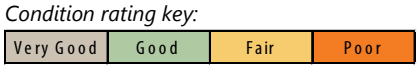


Fig. 11. Patterns (mean ± SE) in AMBI scores compared with condition rating criteria.



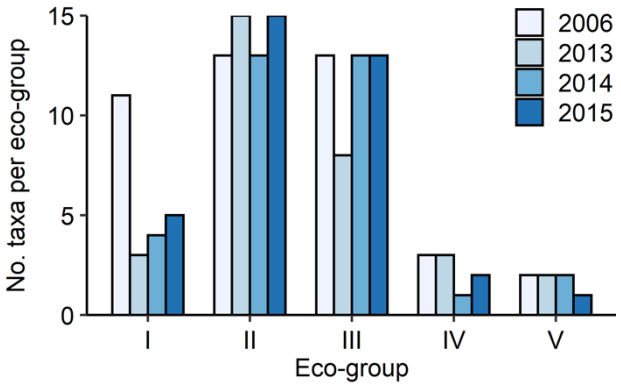
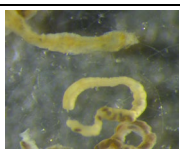
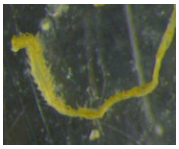


Fig. 12. Number of taxa within each of five eco-groups ranging from sensitive (EG-I) to relatively resilient (EG-V). Data are pooled within sites.

Table 8. Description of the sediment-dwelling species comprising ≥5% of total abundance at any one site. Some of the images are illustrative of the general group. See notes for Table 8.

Main group and species	Description	Image
Amphipoda, EG II	Shrimp-like crustaceans dominated by a species described by CMEC as a phoxocephalid species *Torridoharpinia hurleyi*. Considered to be tolerant of sedimentation and mud, although *T. hurleyi* is regarded as sensitive to enrichment. Probably important prey for birds and small fish.	
Bivalvia, *Austrovenus stutchburyi* EG II	Cockles are suspension feeding bivalves, living near the sediment surface. They can improve sediment oxygenation, increasing nutrient fluxes and influencing the type of macrofauna present. Sensitive to organic enrichment. Important in the diet of certain birds, rays and fish.	
Bivalvia, *Linucula hartvigiana* EG II	Small estuarine bivalve mollusc in the family Nuculidae, commonly called a nutshell. Can be very abundant in sand and mud sediments. Considered tolerant of excessive enrichment, despite EG II classification.	
Bivalvia, *Macomona liliana* EG II	A deposit feeding wedge shell. This species lives at depths of 5-10cm in the sediment and uses a long inhalant siphon to feed on surface deposits and/or particles in the water column. Important in the diet of certain birds, rays and fish.	
Gastropoda, *Zeacumantus* spp. EG II	Small estuarine snail, requiring brackish conditions for survival. Feeds on decomposing animal and plant matter, bacteria, and algae. Tolerant of muddy sediment and organic enrichment.	
Polychaeta, *Heteromastus filiformis* EG III	Small capitellid polychaete worm. A sub-surface, deposit-feeder that can thrive under conditions of moderate organic enrichment. Typically associated with muddy-sand substrate.	
Polychaeta, *Axiothella serrata* EG II	A deposit feeding maldanid 'bamboo' worm that is a common infaunal species on the sheltered flats of central New Zealand estuaries.	
Polychaeta, *Nicon aestuariensis*/ Nereididae EG III	Nereids are omnivores, with some of these being juveniles too small to identify accurately. *Nicon aestuariensis* is a deposit feeding species considered tolerant of freshwater influences.	
Polychaeta, Paraonidae, EG III	Likely to be CMEC *Paradoneis* sp. Common worm considered to be reasonably tolerant of muddy sediment and organic enrichment. Paraonids are considered to be deposit feeders, possibly selectively feeding on microscopic diatoms and protozoans.	
Polychaeta, *Prionospio aucklandica* & *Prionospio* sp. EG II	Deposit-feeding spionid worms are common in harbours and estuaries. *P. aucklandica* is associated mainly with muddy sands, but occurs across a range of mud contents (12 – 50 % optimum). Considered tolerant to organic enrichment despite EG II classification.	

Multivariate macrofauna patterns and association with sediment quality variables

In order to further explore the differences and similarities among sites and surveys in terms of the macrofaunal assemblage, the nMDS ordination in Fig. 13 places site-aggregated samples of similar composition close to each other in a 2-dimensional plot, with less similar samples being further apart. This analysis used species data aggregated (as necessary) to a higher taxonomic level, to enable comparison of datasets from 2006 with 2013-15 (see Methods section and Appendix 3).

Fig. 13a illustrates marked compositional changes over time, but especially between 2006 and subsequent surveys. SIMPER analysis revealed that this result was driven mainly by:

- Declines in abundance or absence of sensitive EG-I and II taxa post-2006. These taxa included sunset shells (*Hiatula* spp.), mysid shrimps, cumaceans, and polychaete worms (e.g. species of *Aonides* & *Prionospio*).
- Declines in abundance of relatively tolerant EG-III taxa, mainly polychaete worms *Heteromastus filiformis* and Paraonidae species.

Although the analysis forced a spatial separation of sites over 2013-15, the Bray-Curtis similarity index among this group was in fact quite high (70-75% compositional similarity). By contrast, macrofaunal composition in 2006 had only a 60% similarity with 2013-15. As species aggregation was undertaken to down-weight the potential influence of different taxonomic providers, the changes since 2006 can be attributed to shifts in species composition (especially among the minor species) that may be linked to a changing environment. That said, it is important to recognise that for minor species whose abundances are very low, there is an element of chance as to whether (or to what extent) they are detected by core sampling. Their apparent presence or absence may not be an accurate reflection of the true situation, and needs to be interpreted with caution.

In order to further explore whether spatial and temporal changes are linked to environmental conditions, relationships between macrofaunal composition and sediment quality variables were explored.

The BIO-ENV procedure in PRIMER revealed a moderate correlation (Spearman rank correlation, ρ = 0.41) between macrofaunal composition changes and increased sediment mud content post-2006. For all, other variables the association was weak (ρ < 0.18). Similarly, increased mud content provided the most plausible explanation (Pearson correlation r = 0.67) for the left-to-right separation of sites/years in Fig. 13. However, the bottom-to-top separation was more closely correlated with increasing total organic carbon values (TOC; Pearson correlation r = 0.85), with lowest TOC values in 2015.

Although these results suggest that increased mud, in particular, has played a role in the temporal change since 2006, there are likely to be many other factors that contribute. These factors could include processes that have differential effects across the estuary, such as intrusions of low salinity water and altered sedimentation (or hydrodynamics) during flood flows in the Moutere River, depth and location-related effects of wind-induced wave disturbance, as well as biological processes such as recruitment events and species interactions. The implications of the relatively high levels of DDT described above are unknown at this stage, as the results provide only a limited snapshot. Further assessment in relation to the latter issue is recommended (see Section 5).

Benthic health model (BHM) values for Moutere Inlet in relation to mud and 'metals' (i.e. copper, lead, zinc) are summarised in Fig. 14. The Mud BHM scores equate roughly to a moderate impact relative to other New Zealand estuaries included in the development of the method (see Appendix 3D). The scores are in general consistent with the 'fair' ratings against New Zealand Estuary Trophic Index (ETI) values (see Fig. 5). However, the BHM result does not indicate a clear change in impact due to mud since 2006, which is inconsistent with the other multivariate analyses described above.

Metals BHM scores are rated against an absolute effects scale described by Clarke (2022, unpublished Cawthron report) and are classified as being indicative of 'good' health, or around the transition between 'good' and 'fair'. As discussed above, metal concentrations were also very low (except for nickel, which is not included in the BHM) relative to national sediment quality guideline DGVs. These results suggest that the BHM indicator metals (copper, lead, zinc) are of no significant concern in Moutere Inlet.

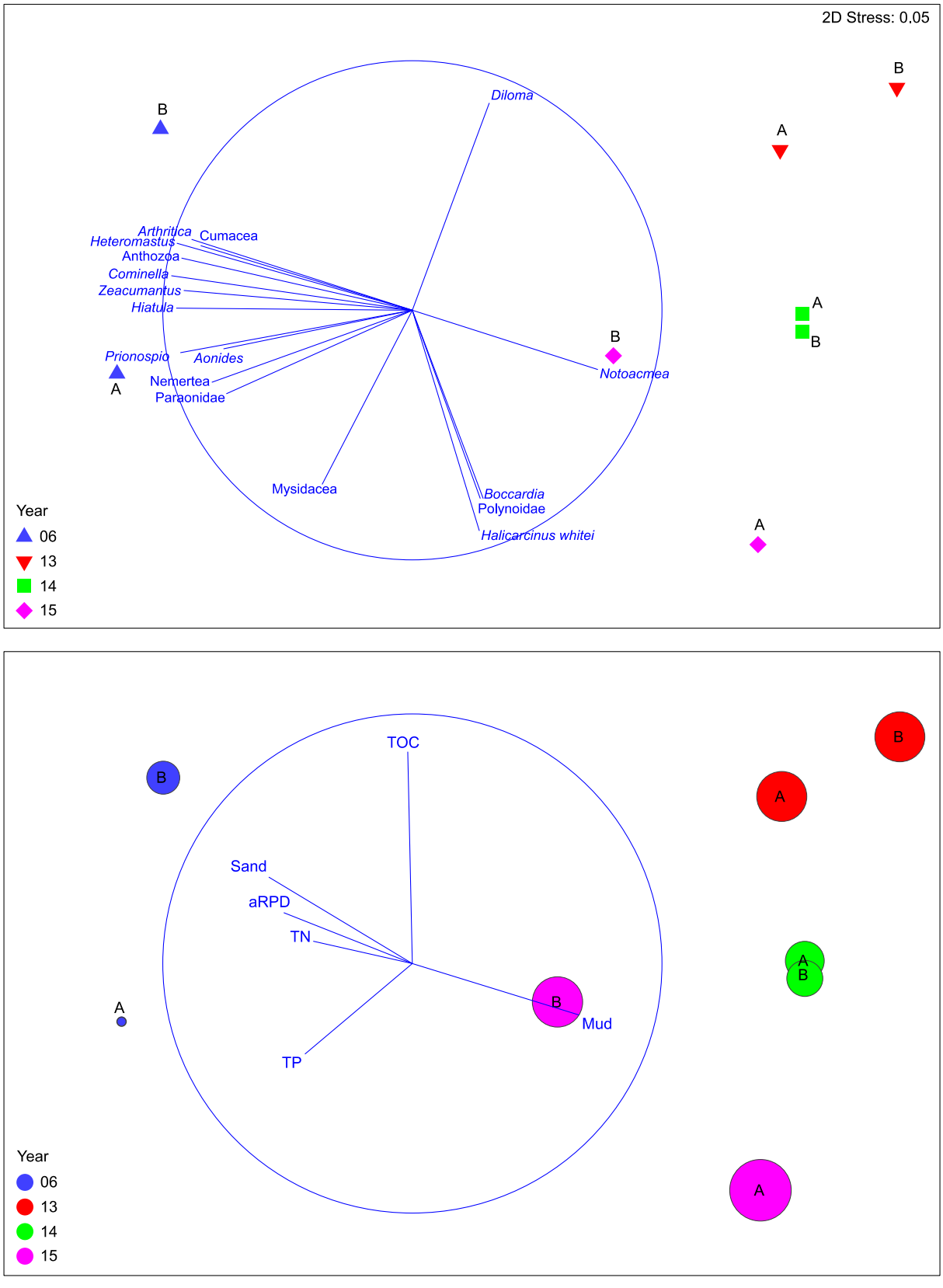


Fig. 13. Non-metric MDS ordination of macrofaunal core samples for data aggregated within each site, and subject to the taxonomic aggregation described in Appendix 3.

Sites are placed such that closer groups are more similar than distant groups in terms of macrofaunal composition. Top: vectors show direction and strength of association (length of line relative to circle) of the species or higher taxa that characterised each site and discriminated sites clusters from each other; Bottom: vectors representing the most correlated sediment quality variables. Bubble sizes are scaled to sediment % mud, which was the variable most strongly correlated.

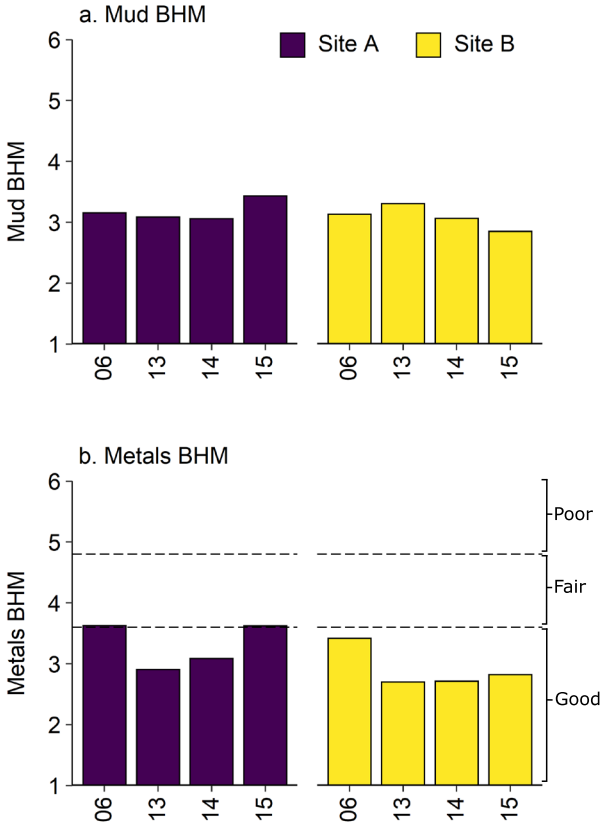


Fig. 14. Benthic health model (BHM) scores for mud and metals. The Mud BHM scores reflect a five-point scale from 1 ('very low') to 6 ('very high' impact relative to other New Zealand estuaries. Metals BHM scores are rated against an absolute scale from 'good' to 'poor' based on different sediment quality guidelines. Note that BHM scores are determined from macrofauna data, hence a metals BHM score can be calculated for 2015 and 2016 when metals analysis was not undertaken but macrofauna were collected.

5. SYNTHESIS AND RECOMMENDATIONS

5.1 SYNTHESIS OF KEY FINDINGS

This report has described the findings of ecological monitoring surveys conducted in Moutere Inlet between 2006 and 2015, along with annual sedimentation monitoring that started in 2008. The ecological surveys have largely followed the fine scale methods described in New Zealand's National Estuary Monitoring Protocol (NEMP). In Table 9, key physical and biological indicators are compared against the ETI condition rating criteria in Table 5.

Sedimentation and sediment quality

There has been no appreciable sediment accrual at the seven monitoring sites since the first plates were installed in 2008. The highest long-term annual rate of 0.54mm/yr (Appendix 4) is less than the 2mm/yr national guideline value, and less than the 0.9mm/yr calculated from NIWA's national estuary sediment load estimator (see Table 13 in Stevens et al. 2020a). However, Stevens et al. (2020a) note that some of the greatest observed deposition in the estuary in recent times has occurred in areas that are not currently monitored using sediment plates, in particular in the wider central basin area, with the one sediment plate site in this location recording a relatively large deposition event between 2014 and 2015 (Appendix 4).

Despite the negligible sedimentation at the monitoring sites, Table 9 highlights that there has nonetheless been an increase in sediment mud content at the two fine scale sites between 2006 and 2015. This result may be due to interstitial spaces among coarser sediments infilling with fine muds over time, despite no significant change in sediment depth. Although the sediment mud content at fine scale sites has not exceeded the biologically-relevant 25% threshold, it was approaching this threshold in 2015 at Site A.

Without any reduction in sediment inputs, it is conceivable that sediments at the fine scale sites have become increasingly muddy in the seven years since the last monitoring was undertaken. In fact, the broad scale habitat mapping undertaken in 2019 showed areas around Site A that were classified as mud-elevated (>25% mud) and mud-dominated (>50% mud). Across the estuary as a whole, sediments with mud content exceeding 25% covered ~38% of the intertidal area in 2019, which is a significant proportion of the estuary in national context (see Table 5 in Stevens et al. 2020a).

Table 9. Summary of condition scores of ecological health for each fine scale monitoring site, based on mean values of key indicators, and ETI rating criteria in Table 5. Dash = not measured. TP not rated.

Site	Year	Mud %	TOC %	TN mg/kg	TP mg/kg	aRPD mm	As mg/kg	Cd mg/kg	Cr mg/kg	Cu mg/kg	Ni mg/kg	Pb mg/kg	Zn mg/kg	AMBI na
A	2006	10.3	0.56	309	546	19	-	< 0.100	33.8	6.0	76.1	3.7	26.8	2.0
	2013	18.2	1.03	333*	520	10	-	0.022	36.7	7.5	87.0	4.6	34.0	1.7
	2014	14.4	0.34	< 500	503	25	4.3	-	-	-	-	-	-	1.8
	2015	24.5	0.28	< 500	550	10	-	-	-	-	-	-	-	1.8
B	2006	12.9	0.71	368	513	26	-	< 0.100	29.6	6.1	58.4	4.6	25.0	1.9
	2013	18.2	0.99	< 500	497	10	-	0.022	30.7	6.8	66.3	5.0	31.7	1.8
	2014	13.7	0.33	< 500	457	7	5.7	-	-	-	-	-	-	1.7
	2015	18.3	0.28	550*	520	5	-	-	-	-	-	-	-	1.7

** Sample mean includes values below lab detection limits*

< All values below lab detection limit

Condition rating key:

Very Good	Good	Fair	Poor

A study by Gibbs and Woodward (2018) estimated that almost 90% of the sediment in the estuary at the Moutere River mouth originated from plantation pine forest, most of which appears to be produced during forest harvest and for ~2-3 years afterwards. Such findings indicate a disproportionate contribution from forest logging to catchment sediment load, given the LCBD5 data in Table 1 (see Section 2) that only 4% of the catchment is harvested forest, with a further 18.2% classified as exotic forest (i.e. in production). Gibbs and Woodward (2018) also noted the potential for an historic catchment contribution of sediment over 2007-2008, due to conversion of pine forest to pasture. This period coincides with the marked change in macrofauna richness and composition between the 2006 to 2013 surveys. TDC have photographic evidence of considerable runnel erosion in these conversion areas, along with observations of sediment deposits in adjacent streams (pers. comm. Trevor James, TDC). Figures cited in the broad scale report indicate that 96% of the catchment-derived sediment entering the estuary is predicted to be trapped and retained within it.

Trophic status

There has been a temporal trend suggesting an apparent shallowing of the aRPD, which can be an indicator of increased sediment enrichment. This result may in part reflect the coarse and subjective aRPD estimates that were made in most surveys, but it is nonetheless plausible that increased sediment mud content has led to reduced oxygen penetration into the sediment matrix. Irrespective, the apparent shallowing of the aRPD does not appear to reflect increased sediment organic or nutrient enrichment *per se*. For example, TOC and nutrient levels are not particularly high, and neither of the fine scale sites showed evidence of strong anoxia (e.g. black colour and strong sulphide-smell). Similarly, neither site had extensive growths of opportunistic algae (see Table 6).

However, sediment anoxia and prolific growths of opportunistic macroalgae (referred to a 'High Enrichment Conditions', HECs) have previously been recorded elsewhere in the estuary, even though the modelled average nutrient load to Moutere Inlet is about half the threshold (~100mgN/m^2/d) above which nuisance growths are commonly encountered in intertidally-dominated estuaries (Stevens et al. 2020c). The occurrence of HECs in 2019 in Moutere Inlet was mainly associated with the area south of the Moutere River mouth, and also the Wharf Rd embayment; i.e. areas where dissolved (i.e. bioavailable) nutrients may be elevated (Moutere River mouth) and/or with

sheltered and flow restricted conditions that facilitate macroalgal blooms (i.e. Wharf Rd embayment).

Trace contaminants

DDT was recorded at concentrations exceeding the national sediment quality DGV, and would probably have exceeded the GV-high threshold for 'probable' ecological effects if laboratory detection limits had been low enough to quantify all DDT compounds and breakdown residues. DDT is a ubiquitous and persistent contaminant in New Zealand due to its historic use as an insecticide (e.g. to control grass grub and codling moth). It was banned in New Zealand in the 1970's due to its globally recognised environmental persistence and adverse impacts (Boul 1995). DDT adsorbs strongly to fine sediments and organic matter, with the main sources to Moutere Inet likely to be residues associated horticultural and agricultural soils to which DDT was historically applied (Gaw et al. 2006).

Given the relatively high DDT concentrations, it is worthwhile considering further sampling at fine scale sites and around catchment freshwater inflows to determine how widespread and persistent DDT is across the estuary, as well as the potential ecological implications of elevated levels. DDT is notorious for its bioconcentration in aquatic organisms (e.g. in filter feeders such as cockles) and biomagnification up the food chain; for example, via transfer to aquatic invertebrates and then to the fish and birds that eat them.

Other than DDT, there was no other evidence of widespread pollution with anthropogenic contaminants. As noted above, the 'poor' rating for the trace metal nickel is attributable to catchment geology rather than anthropogenic sources. All other trace metals, including those that can commonly be elevated due to anthropogenic inputs, were at very low concentrations, which were often less than half of the national sediment quality guideline value for 'possible' ecological effects (ANZG 2018). However, it cannot be discounted that localised hot-spots may exist around point sources (freshwater and stormwater inflows) due to urbanisation in the northern Inlet, and agricultural and horticultural development in the wider catchment. Potential historic and ongoing contaminant sources include trace metals and hydrocarbons in urban run-off, and various metals (cadmium, copper, lead, mercury zinc,) or metalloids (arsenic) associated with horticultural compounds (e.g. pesticides, fungicides) and/or fertiliser application (Gaw et al. 2006). The study of Gaw et al. (2006) showed that the highest concentrations of almost all of these contaminants in Tasman soils was associated with orcharding.

Macrofauna and Benthic Health Model

Despite the sediment mud content approaching the 25% threshold (especially at Site A), there was still a moderately diverse and abundant macrofauna present in 2013-15. Even though richness and abundance both greatly declined post-2006, Moutere Inlet in 2013-15 is still species-rich compared to several other estuaries in the top of the South Island (Fig. 15). However, organism abundances are fairly low in a regional context. Moreover, the apparent loss, or decline in abundance, of a range of sensitive species since 2006 may be a warning sign of a wider decline in the Inlet, and appears most strongly linked to sediment inputs, although this possibility needs to be further evaluated. Sediment mud content, along with trophic status, are recognised as strongly influencing macrofaunal composition in estuarine and coastal environments (Pearson & Rosenberg 1978; Cummings et al. 2003; Thrush et al. 2004; Robertson et al. 2015; Ellis et al. 2017). Whether the sediment-dwelling macrofaunal community has declined further since the last survey in 2015 can only be resolved by further monitoring.

The Mud BHM rated Moutere sites as 'moderate' across a gradient of 'impact' relative to other New Zealand estuaries. Mud BHM scores showed little temporal change, which was surprising considering the increase in mud and loss or reduction in sensitive eco-groups post-2006. For copper, lead and zinc, the Metals BHM scores were largely indicative of 'good' conditions when rated against sediment quality guideline thresholds that were far more conservative that the national ANZG (2018) values.

Overall the results and associated condition ratings indicate that the main tidal flats of Moutere Inlet have suffered degradation between 2006 and 2015, yet were still in a reasonably healthy condition ecologically at the time of the 2015 survey. However, the gradual increase in sediment mud content and decline of sensitive macrofauna appears to be indicative of a relatively insidious change in habitat quality, that may have worsened since the last survey in 2015. There is a risk that soft-sediment habitats in parts of the estuary will reach a point at which the mud tolerance of key species (e.g. cockles, wedge shells) is exceeded, and their populations eventually decline. Such an outcome could have flow on effects to the wider ecosystem, for example due to a decline in important prey items for birds and fish.

5.2 CONSIDERATIONS FOR FURTHER ASSESSMENT AND MONITORING

NEMP fine scale monitoring is valuable for understanding long term ecological changes at specific sites in an estuary, with broad scale monitoring helping track changes in the main habitats and identify areas of excessive mud deposition and/or eutrophication. There is benefit in having long-term data that are collected using standardised approaches, but there is also a need to consider what is required to address present and potential management needs. In terms of monitoring and investigative work, monitoring would ideally be extended to: (i) include areas of Moutere Inlet that are most vulnerable to change from land use and other anthropogenic activities; (ii) enable changes to be detected early, so that problems can be addressed before they become estuary-wide issues; and (iii) provide insight into cause-effect linkages, in particular between muddy sediment inputs and estuary condition.

1. Investigative approaches

To better understand the changing state of Moutere Inlet and its current pressures, we recommend that TDC consider the following:

- Undertake a desktop assessment to understand changes in catchment land use, and how such changes may affect inputs of stressors to the estuary. Specific related needs are as follows:
  - Evaluate sources of past, present and potential inputs of muddy sediment, building on the findings of Gibbs and Swales (2018). Initially, it would be of value to understand changes in catchment land use, and how such changes may affect sediment inputs to the estuary (e.g. it would be helpful to understand forest harvest schedules, given that ~27% of the catchment is in exotic forest; see Table 1).

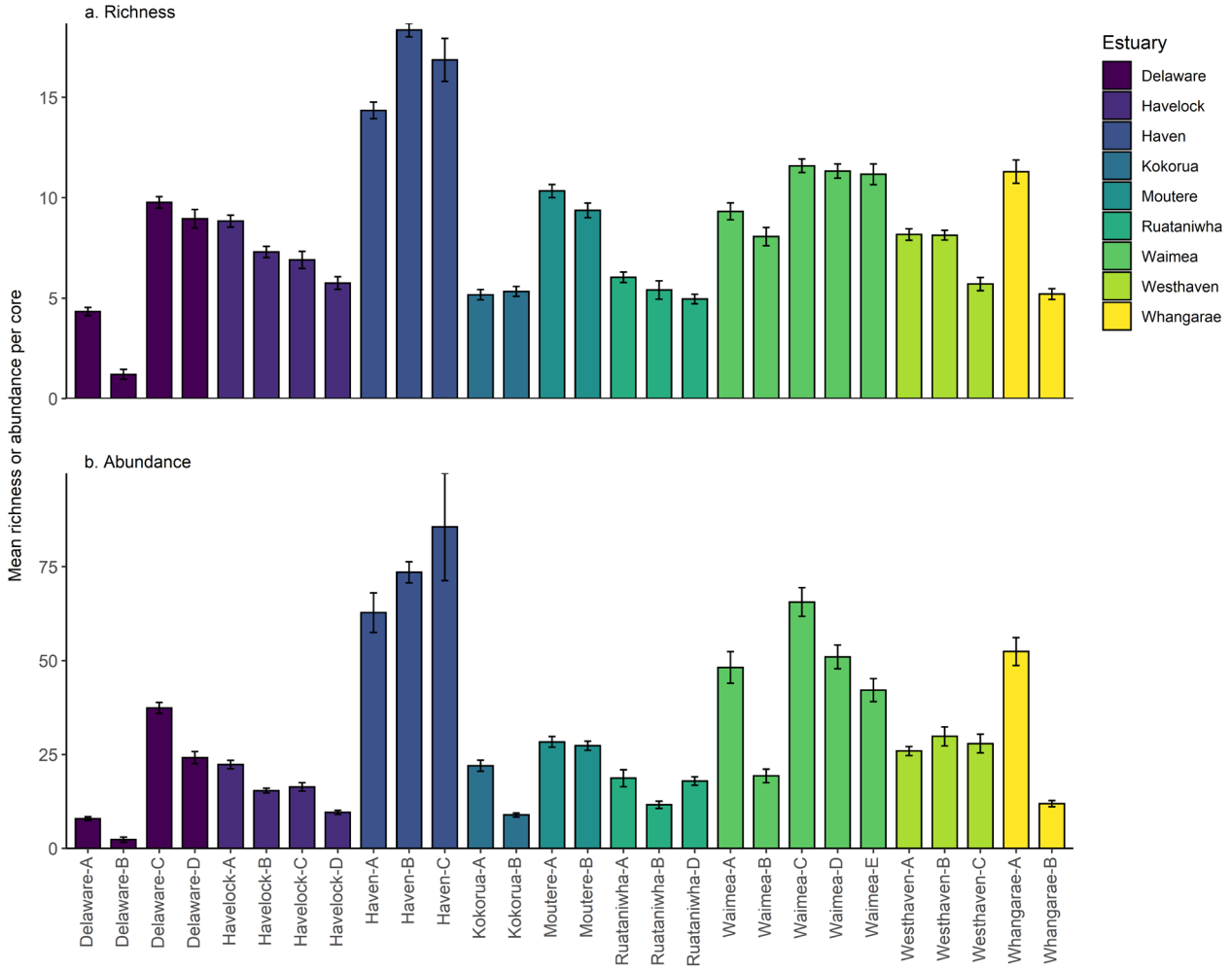


Fig. 15. Macrofauna richness and abundance summary (mean ±SE) based on NEMP monitoring in estuaries in the top of the South Island since 2014. For illustrative purposes, site-level data are averaged across multiple survey years in each location.

- As part of a focus on links between catchment land use and muddy sediment inputs, it would be appropriate to also consider improvements to the present sediment plate monitoring work (see specific recommendations below).

- Conduct targeted synoptic assessments of estuary condition in the vicinity of point source inputs and/or where local issues have already been identified (e.g. by the broad-scale survey). Such assessments should be based on key indicators described in fine scale and broad scale (as appropriate) reports, and encompass analysis of a full suite of priority pollutants including DDT. Findings from the desktop assessment will guide this component.

2. Ongoing fine scale and sedimentation monitoring

Recent guidance produced by NIWA (Hewitt 2021) recommended collecting 12 macrofauna reps per estuary site and conducting monitoring more than twice per year (up to 6 times is optimal to detect tipping points), with a time series of approximately 15 years needed for trend detection. The NIWA advice was that reducing macrofauna sampling effort or frequency would affect the robustness of monitoring programmes. Current TDC monitoring (as for many other councils) is considerably less than this recommended optimum. Despite this situation, TDC have asked us to consider whether efforts can be even further reduced, or in fact whether macrofauna can be dropped from the SOE programme.

At present, NEMP fine scale monitoring is typically undertaken every 5 years by TDC, after first establishing a baseline for a given estuary. Sediment plate monitoring is typically undertaken annually. As there has been no fine scale monitoring at the two Moutere sites since 2015, it would be timely to undertake a follow-up survey to determine whether there has been ongoing degradation over the last 7 years. Due to council budget constraints, we have considered the scope for reducing per survey effort and cost, for which we suggest the following:

Fine scale sites: The present two fine scale sites appear adequate for long-term monitoring purposes, although the number of sites would ideally be increased if the synoptic survey recommended above identifies other areas under pressure.

Fine scale indicators and sampling effort: All of the measured indicators contribute to the understanding of estuary health and temporal change. The relative cost of the macrofaunal component (currently 10 cores per site) is high; typically ranging from 40-45% of the total survey budget, depending on the organisation undertaking sample processing and taxonomy. A separate analysis (summarised in Appendix 7) suggests that replication of macrofauna could be reduced to nine samples, without any substantive loss of ability to detect long term changes. A reduction to <9 would make it difficult to distinguish temporal change from sampling variation (e.g. chance sampling of less common species). We would not recommend dropping macrofauna from the programme, as they are the main indicator for assessing biological responses to physico-chemical changes in the estuary. Also, as noted above, there is considerable benefit in having long-term data that are collected using standardised approaches.

By contrast with macrofauna, sediment quality indicators, except aRPD, tend to be less variable within sites and therefore subject to less sampling variation. For the purpose of tracking long term change, it would be sufficient to collect a single composite sample from within each site for lab analysis. That analysis should be expanded to include DDT and other priority pollutants. As aRPD is easily measured in the field, and can also be spatially variable, we recommend continuing to undertake replicate measurements (e.g. an aRPD measurement matching each macrofauna core).

Fine scale sampling design and sediment plates: Fine scale monitoring at 5-year intervals is reasonable for tracking long term change. In interim years, it would be desirable to also keep track of changes in sediment quality, in particular to monitor changes in sediment mud content and aRPD. A suggested sampling approach is to:

- Undertake another fine scale survey in 2022 that includes the amendments above (i.e. reduced sampling effort overall, but expand the lab analysis to include DDT and other priority pollutants).

- Continue annual sediment plate monitoring, and at each site measure/assess the following parameters in addition to sediment depth: (i) Measure aRPD; (ii) Subjectively assess sediment texture using NEMP broad scale methods; and (iii) Collect a single composite sample (from each site) for laboratory grain size analysis.

- Installing one additional sediment plate site in the central basin area where signs of sediment deposition were noted during the broad scale survey. It would also be of value to install plates at each of the fine scale sites, to help with interpretation of ecological changes.

5.3 RECOMMENDATIONS

This report has undertaken a synthesis of ecological monitoring data collected in Moutere Inlet since 2006 as part of SOE monitoring conducted by TDC.

Although the estuary is in a reasonably healthy state, the gradual increase in sediment mud levels is a potential concern, and high levels of DDT recorded in 2014 need to be further investigated. Furthermore, as fine scale monitoring has focused on the main tidal flats of the estuary, there is a need to better understand estuary state around point source inputs, and to link estuary state with drivers of change (in particular muddy sediment). Accordingly, to better understand the changing state of Moutere Inlet and its current pressures, we recommend that TDC consider the following:

1. Undertake a desktop assessment to understand changes in catchment land use, and how such changes may affect inputs of stressors to the estuary. A particular need is to understand past, present and future exotic forest harvest patterns and links with muddy sediment inputs.
2. Investigate estuary condition in the vicinity of point source inputs and/or where local issues have already been identified, involving sampling of key NEMP indicators and a suite of priority pollutants including DDT. The scope would be better determined after completion of the assessment in #1.
3. Install three new sediment plate sites: (i) one in the central basin area, which appears (from previous observations) to experience relatively high sedimentation; and (ii) one at each of the two fine scale sites. Increase sampling effort at each sediment plate site to include measures of sediment grain size and enrichment status, and undertake sediment plate monitoring annually.
4. Undertake further fine scale monitoring to determine whether there has been ongoing degradation at the two fine scale sites since 2015 when the last survey was conducted. A reduction in macrofauna and sediment sampling effort as recommended above should be considered, but the lab analysis of sediments should be expanded to encompass a full suite of priority pollutants, including DDT.
5. Depending on the outcomes of the above, the potential for implementation of mitigation strategies to reduce future impacts should be considered. Related to this are questions that may require considerable investment to resolve, and may therefore benefit from links with research providers. These questions include the practical changes in land use that are necessary to reduce sediment yield, and limits on sediment loads that will be necessary to lead to maintain or improve estuary condition.

6. REFERENCES CITED

Anderson MJ, Gorley RN, Clarke KR 2008. PERMANOVA+ for PRIMER: Guide to software and statistical methods. PRIMER-E, Plymouth, UK.

ANZG 2018. Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Governments and Australian State and Territory Governments, Canberra ACT, Australia. Available at https://www.waterquality.gov.au/anz-guidelines/guideline-values/default/sediment-quality-toxicants.

Barter P, Forrest B 2001. Environmental monitoring of the Talley's seafood factory discharges to Moutere Inlet630 36 p.

Blyth-Skyrme V, Lindenbaum C, Verling E, Van Landeghem K, Robinson K, Mackie A, Darbyshire T 2008. Broad-scale biotope mapping of potential reefs in the Irish Sea (north-west of Anglesey). JNCC Report No. 423, Joint Nature Conservation Committee. 210p.

Borja A, Franco J, Pérez V 2000. A Marine Biotic Index to Establish the Ecological Quality of Soft-Bottom Benthos Within European Estuarine and Coastal Environments. Marine Pollution Bulletin 40(12): 1100-1114.

Boul HL 1995. DDT residues in the environment—A review with a New Zealand perspective. New Zealand Journal of Agricultural Research 38(2): 257-277.

Clark DE, Hewitt JE, Pilditch CA, Ellis JI 2020. The development of a national approach to monitoring estuarine health based on multivariate analysis. Marine Pollution Bulletin 150: 110602.

Clark K, Stevens L, Gillespie P 2006. Broad scale habitat mapping of Moutere Inlet1037. 14 p.

Clarke KN, Gorley RN 2015. PRIMER v7: User Manual/Tutorial. PRIMER-E, Plymouth.

Clarke KR, Gorley RN, Somerfield PJ, Warwick RM 2014. Change in marine communities: an approach to statistical analysis and interpretation, 3rd edition. PRIMER-E, Plymouth, UK. 260p.

Cummings V, Thrush S, Hewitt J, Norkko A, Pickmere S 2003. Terrestrial deposits on intertidal sandflats: sediment characteristics as indicators of habitat suitability for recolonising macrofauna. Marine Ecology Progress Series 253: 39-54.

Ellis JI, Clark D, Atalah J, Jiang W, Taiapa C, Patterson M, Sinner J, Hewitt J 2017. Multiple stressor effects on marine infauna: responses of estuarine taxa and functional traits to sedimentation, nutrient and metal loading. Scientific Reports 7(1): 12013.

FGDC 2012. Coastal and Marine Ecological Classification Standard. Standard FGDC-STD-018-2012, Marine and Coastal Spatial Data Subcommittee, Federal Geographic Data Committee, June, 2012.. 343p. Available at: https://www.fgdc.gov/standards/projects/cmecs-folder/CMECS_Version_06-2012_FINAL.pdf.

Forrest B, Cooke A 1995. Discharges from Talleys Fisheries, Port Motueka: assessment of effluent quality and ecological effects285 25 p.

Forrest B, Gillespie PA, Cornelisen C, Rogers K 2007. Multiple indicators reveal river plume influence on sediments and benthos in a New Zealand coastal embayment. New Zealand Journal of Marine and Freshwater Research 41: 13-24.

Forrest BM, Stevens LM 2021. Fine scale intertidal monitoring of Nelson Haven. Salt Ecology Report 071, prepared for Nelson City Council, June 2021. 40p.

Forrest BM, Stevens LM, Rabel H 2021. Fine scale intertidal monitoring of Waimea (Waimeha) Inlet. Salt Ecology Report 082, prepared for Tasman District Council and Nelson City Council, October 2021. 50p.

Gaw SK, Wilkins AL, Kim ND, Palmer GT, Robinson P 2006. Trace element and ΣDDT concentrations in horticultural soils from the Tasman, Waikato and Auckland regions of New Zealand. Science of The Total Environment 355(1): 31-47.

Gibbs M, Woodward B 2018. Waimea and Moutere sediment sources by land use. NIWA Client Report No: 2018026HN, Prepared for Tasman District Council, February 2018. 63p.

Gillespie P, Clark K 2006. Moutere Inlet fine-scale benthic baseline 20061183. 18 p.

Gillespie P, Stark J, Asher R 1995. Moutere Inlet ecosystem investigation. Cawthron Report No. 273 prepared for Tasman District Council. 57p.

Hewitt J, McBride G, Pridmore R, Thrush S 1993. Patchy distributions: Optimising sample size. Environmental monitoring and assessment 27: 95-105.

Hewitt JE 2021. Monitoring estuaries in a changing world: Lessons for designing long-term monitoring programmes. Sustainable Seas, National Science Challenges, July 2021. 16p. .

Hewitt JE, Anderson MJ, Hickey CW, Kelly S, Thrush SF 2009. Enhancing the Ecological Significance of Sediment Contamination Guidelines through Integration with Community Analysis. Environmental Science & Technology 43(6): 2118-2123.

MacDonald DD, Carr RS, Calder FD, Long ER, Ingersoll CG 1996. Development and evaluation of sediment quality guidelines for Florida coastal waters. Ecotoxicology 5: 253-278.

MNCR 1990. Use of the Marine Nature Conservation Review SACFOR abundance scales. Joint Nature Conservation Committee. www. jncc.gov.uk/page-2684 (accessed 15 April 2019).

Pearson TH, Rosenberg R 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology Annual Review 16: 229-311.

R Core Team 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Rattenbury MS, Cooper RA, Johnston MR 1998. Geology of the Nelson area. Institute of Geological & Nuclear Sciences 1:250000 geological map 9. 1 sheet plus 67p. .

Robertson B, Gillespie P, Asher R, Frisk S, Keeley N, Hopkins G, Thompson S, Tuckey B 2002. Estuarine environmental assessment and monitoring: a national protocol part B: development of the monitoring protocol for New Zealand Estuaries. Appendices to the introduction, rationale and methodology. Sustainable Management Fund Contract No. 5096, Cawthron Institute, Nelson, New Zealand. 159p.

Robertson BM, Stevens LM 2013. Moutere Inlet fine scale monitoring 2012/13. Prepared for Tasman District Council. 25p.

Robertson BM, Stevens L, Robertson B, Zeldis J, Green M, Madarasz-Smith A, Plew D, Storey R, Hume T, Oliver M 2016. NZ Estuary Trophic Index Screening Tool 2: determining monitoring indicators and assessing estuary trophic state. Prepared for Envirolink Tools Project: Estuarine Trophic Index MBIE/NIWA Contract No: C01X1420. 68p.

Robertson BP, Gardner JPA, Savage C 2015. Macrobenthic–mud relations strengthen the foundation for benthic index development: A case study from shallow, temperate New Zealand estuaries. Ecological Indicators 58: 161-174.

Stevens LM, Robertson BM 2013. Moutere Inlet 2013 broad scale habitat mapping. Report prepared for Tasman District Council, September 2013. 31p.

Stevens LM, Scott-Simmonds T, Forrest BM 2020. Broad scale intertidal habitat mapping of Moutere Inlet, 2019. Salt Ecology Report 034 prepared for Tasman District Council. 52p.

Thrush S, Hewitt J, Cummings V, Ellis J, Hatton C, Lohrer A, Norkko A 2004. Muddy waters: elevating sediment input to coastal and estuarine habitats. Frontiers in Ecology and the Environment 2(6): 299-306.

Townsend M, Lohrer D 2015. ANZECC Guidance for Estuary Sedimentation. NIWA client report number HAM2015-096, prepared for Ministry for the Environment. 45p.

Appendix 1. GPS coordinates for fine scale sites (corners)

FINE SCALE SITE A

Corner	NZTM East	NZTM North
C1	1601499	5444688
C2	1601452	5444726
C3	1601471	5444749
C4	1601518	5444712

FINE SCALE SITE B

Corner	NZTM East	NZTM North
C1	1603574	5442393
C2	1603614	5442440
C3	1603636	5442420
C4	1603598	5442375

SEDIMENT PLATE SITES (provided by TDC)

Site	Plate	NZTM East	NZTM North
Moutere Rv mouth	NE	1600776	5445082
Moutere Rv mouth	NW	1600749	5445091
Moutere Rv mouth	SW	1600737	5445063
Moutere Rv mouth	SE	1600736	5445054
Moana Loop	NE	1601645	5443436
Moana Loop	NW	1601611	5443442
Moana Loop	SW	1601606	5443418
Moana Loop	SE	1601634	5443406
Strong Loop	NE	1602347	5442761
Strong Loop	NW	1602318	5442770
Strong Loop	SW	1602309	5442738
Strong Loop	SE	1602339	5442730
Tasman End	NE	1604315	5441044
Tasman End	NW	1604291	5441062
Tasman End	SW	1604279	5441035
Tasman End	SE	1604303	5441018
SE Basin	NE	1603550	5441953
SE Basin	NW	1603528	5441950
SE Basin	SW	1603532	5441929
SE Basin	SE	1603551	5441934
Central Basin	NW	1601599	5444366
Central Basin	SW	1601600	5444344
North Embayment	NE	1601322	5446758

Appendix 2. RJ Hill analytical methods for sediments, based on 2014 analysis that also included organochlorine pesticides.

SUMMARY OF METHODS

The following table(s) gives a brief description of the methods used to conduct the analyses for this job. The detection limits given below are those attainable in a relatively clean matrix. Detection limits may be higher for individual samples should insufficient sample be available, or if the matrix requires that dilutions be performed during analysis.

Sample Type: Sediment			
Test	**Method Description**	**Default Detection Limit**	**Sample No**
Environmental Solids Sample Preparation	Air dried at 35°C and sieved, <2mm fraction. Used for sample preparation. May contain a residual moisture content of 2-5%.	-	1-6
Organochlorine/nitro&phosphorus Pest.s Trace in Soils, GC-MS	Sonication extraction, GPC cleanup, GC-MS analysis. Tested on as received sample	0.0010 - 0.03 mg/kg dry wt	7-8
Dry Matter (Env)	Dried at 103°C for 4-22hr (removes 3-5% more water than air dry) , gravimetry. US EPA 3550. (Free water removed before analysis).	0.10 g/100g as rcvd	7-8
Dry Matter	Drying for 16 hours at 103°C, gravimetry (Free water removed before analysis).	0.10 g/100g as rcvd	1-6
Total Recoverable digestion	Nitric / hydrochloric acid digestion. US EPA 200.2.	-	1-6
Texture Marine (2 mm, 63 µm fractions) *		-	1-6
Fraction < 2 mm, >/= 63 µm*	Wet sieving, 2.00 mm and 63 µm sieves, gravimetry (calculation by difference).	0.1 g/100g dry wt	1-6
Fraction < 63 µm*	Wet sieving, 63 µm sieve, gravimetry (calculation by difference).	0.1 g/100g dry wt	1-6
Total Recoverable Arsenic	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, trace level. US EPA 200.2.	0.2 mg/kg dry wt	1, 4
Total Recoverable Phosphorus	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, screen level. US EPA 200.2.	40 mg/kg dry wt	1-6
Total Nitrogen*	Catalytic Combustion, separation, Thermal Conductivity Detector [Elementar Analyser].	0.05 g/100g dry wt	1-6
Total Organic Carbon*	Acid pretreatment to remove carbonates if present, neutralisation, Elementar Combustion Analyser.	0.05 g/100g dry wt	1-6

Appendix 3. Macrofauna renaming and taxonomic aggregation undertaken to ensure comparability of surveys for multivariate analyses

A. Renaming of species undertaken to ensure consistent species names were applied across years, which followed the accepted names in the World Register of Marine Species. (Format: old name = new name). This is a generic list that includes species in addition to those in Moutere Inlet.

Boccardia (Paraboccardia) acus = Boccardia acus,
Boccardia (Paraboccardia) syrtis = Boccardia syrtis,
decapod megalopa/juvenile = Decapod megalopa,
Callianassa filholi = Biffarius filholi,
Capitellethus zeylanicus = Notomastus zeylanicus,
Decapoda larvae unid. = Decapod megalopa,
Decapoda (larvae unid.) = Decapod megalopa,
decapod megalopa/juvenile = Decapod megalopa,
Decapoda larvae unid. = Decapod megalopa,
Diloma subrostrata = Diloma subrostratum,
Diloma zelandica = Diloma zelandicum,
Elminius modestus = Austrominius modestus,
Haminoea zelandiae = Papawera zelandiae,
Helice crassa = Austrohelice crassa,
Hemipodus simplex = Hemipodia simplex,
Hiatula nitida = Hiatula spp.,
Hiatula sp. 1 = Hiatula spp.,
Macrophthalmus hirtipes = Hemiplax hirtipes,
Nemertea (unidentifiable) = Nemertea,
Nereidae = Nereididae (juv),
Nereidae (juvenile) = Nereididae (juv),
Nereidae (unidentified juveniles) = Nereididae (juv),
Nereididae = Nereididae (juv),
Nereididae (juvenile) = Nereididae (juv),
Nereididae (unidentified juv) = Nereididae (juv),
Nereididae (unidentified juveniles) = Nereididae (juv),
Notoacmaea helmsi = Notoacmea spp.,
Notoacmaea spp. = Notoacmea spp.,
Notoacmaea sp. = Notoacmea spp.,
Notoacmea sp. = Notoacmea spp.,
Notoacmea helmsi = Notoacmea spp.,
Nucula hartvigiana = Linucula hartvigiana,
Pectinaria australis = Lagis australis,
Perrierina turneri = Legrandina turneri,
Scoloplos cylindrifer = Leodamas cylindrifer,
Soletellina sp. = Hiatula spp.,
Soletellina nitida = Hiatula spp.,
Spheromatidae = Sphaeromatidae,
Trochodota dendyi = Taeniogyrus dendyi,
Tellina liliana = Macomona liliana,
unidentified decapod megalopa = Decapod megalopa

B. Jaccard similarity coefficients of presence and absence data indicating percentage of taxa in common in pairwise comparisons of each year based on: a) raw data, and b) data after taxonomic aggregation (see part C below) to address uncertainty associated with a change in provider after 2006.

Note the low similarity of 2006 vs 2013-15 based on the raw data before taxonomic aggregation.

a. Raw data

	06	13	14
06			
13	18.75		
14	23.81	73.684	
15	22.727	60.465	61.364

b. Aggregated data

	06	13	14
06			
13	58.14		
14	61.905	75	
15	65.909	69.444	74.286

C. Taxonomic aggregation of Moutere Inlet data undertaken to enable multivariate analyses of data across years using nMDS ordination and national Benthic Health Model (BHM) methods. Calculation of BHM score also require omitting certain taxa (noted as NA in BHM taxa column) as prescribed by Clark et al. (2020).

Taxa	Cawthron 2006	CMEC 2013-15	nMDS taxa	BHM taxa
Amphipoda A	caw	NA	Amphipoda	amphipod.other
Amphipoda B	caw	NA	Amphipoda	amphipod.other
Amphipoda C	caw	NA	Amphipoda	amphipod.other
Amphipoda sp. 1	NA	cmec	Amphipoda	amphipod.other
Paracalliope novizealandiae	NA	cmec	Amphipoda	paracalliopiidae
Torridoharpinia hurleyi	NA	cmec	Amphipoda	phoxocephalidae
Anthopleura aureoradiata	caw	cmec	Anthozoa	anthopleura.hermaphroditica
Edwardsia sp.	caw	NA	Anthozoa	edwardsiidae
Arthritica bifurca	caw	NA	Arthritica sp.	arthritica
Arthritica sp. 1	NA	cmec	Arthritica sp.	arthritica
Austrovenus stutchburyi	caw	cmec	Austrovenus stutchburyi	austrovenus.stutchburyi
Hiatula spp.	caw	cmec	Hiatula spp.	hiatula
Linucula hartvigiana	caw	cmec	Linucula hartvigiana	linucula.hartvigiana
Macomona liliana	caw	cmec	Macomona liliana	macomona.liliana
Colurostylis lemurum	NA	cmec	Cumacea	cumacea
Cumacea	caw	NA	Cumacea	cumacea
Austrohelice crassa	caw	cmec	Austrohelice crassa	austrohelice.hemigrapsus.hemiplax
Biffarius filholi	caw	NA	Biffarius filholi	biffarius.filholi
Halicarcinus whitei	caw	cmec	Halicarcinus whitei	halicarcinus
Hemiplax hirtipes	caw	cmec	Hemiplax hirtipes	austrohelice.hemigrapsus.hemiplax
Diptera sp. 1	NA	cmec	Diptera	NA
Dolichopodidae larvae	caw	NA	Diptera	NA
Amphibola crenata	caw	cmec	Amphibola crenata	amphibola.crenata
Cominella glandiformis	caw	cmec	Cominella glandiformis	cominella.glandiformis
Diloma subrostratum	NA	cmec	Diloma spp.	diloma
Diloma zelandicum	caw	NA	Diloma spp.	diloma
Micrelenchus tenebrosus	caw	NA	Micrelenchus tenebrosus	cantharidus.micrelenchus
Notoacmea spp.	NA	cmec	Notoacmea spp.	notoacmea
Papawera zelandiae	caw	NA	Papawera zelandiae	haminoea.zelandiae
Zeacumantus lutulentus	caw	cmec	Zeacumantus spp.	zeacumantus.lutulentus
Zeacumantus subcarinatus	caw	NA	Zeacumantus spp.	zeacumantus.subcarinatus
Taeniogyrus dendyi	caw	NA	Taeniogyrus dendyi	taeniogyrus.dendyi
Mysidacea	caw	NA	Mysidacea	mysida
Tenagomysis sp.	NA	cmec	Mysidacea	mysida
Nematoda	caw	NA	Nematoda	NA
Nemertea	caw	NA	Nemertea	nemertea
Nemertea sp. 1	NA	cmec	Nemertea	nemertea
Nemertea sp. 2	NA	cmec	Nemertea	nemertea
Nemertea sp. 3	NA	cmec	Nemertea	nemertea
Nemertea sp. 4	NA	cmec	Nemertea	nemertea
Nemertea sp. 5	NA	cmec	Nemertea	nemertea
Oligochaeta	NA	cmec	Oligochaeta	capitella.oligochaete
Aglaophamus macroura	caw	cmec	Aglaophamus macroura	aglaophamus
Aonides sp.	caw	NA	Aonides sp.	aonides
Aonides trifida	NA	cmec	Aonides sp.	aonides
Armandia maculata	NA	cmec	Armandia maculata	armandia.maculata
Axiothella serrata	NA	cmec	Maldanidae	maldanidae
Boccardia acus	NA	cmec	Boccardia spp.	polydorid.complex
Boccardia sp.	caw	NA	Boccardia spp.	polydorid.complex
Boccardia syrtis	NA	cmec	Boccardia spp.	polydorid.complex

Appendix 3C (cont.)

Taxa	Cawthron 2006	CMEC 2013-15	nMDS taxa	BHM taxa
Capitella capitata	caw	NA	Capitella sp.	capitella.oligochaete
Capitella sp. 1	NA	cmec	Capitella sp.	capitella.oligochaete
Cirratulidae	caw	NA	Cirratulidae	cirratulidae
Disconatis accolus	NA	cmec	Polynoidae	polynoidae
Glycera lamelliformis	NA	cmec	Glyceridae	glyceridae
Glyceridae	caw	NA	Glyceridae	glyceridae
Heteromastus filiformis	caw	cmec	Heteromastus filiformis	heteromastus.filiformis.baranatolla.lepte
Lagis australis	caw	cmec	Lagis australis	pectinariidae
Magelona dakini	caw	NA	Magelona sp.	magelona
Magelona sp. 1	NA	cmec	Magelona sp.	magelona
Maldanidae	caw	NA	Maldanidae	maldanidae
Nereididae (juv)	NA	cmec	Nereididae	nereididae
Nicon aestuariensis	caw	cmec	Nereididae	nereididae
Orbinia papillosa	caw	cmec	Orbinia papillosa	orbiniidae
Owenia petersenae	NA	cmec	Owenia petersenae	owenia.petersenae
Paraonidae	caw	NA	Paraonidae	paraonidae.other
Paradoneis sp.	NA	cmec	Paraonidae	paraonidae.other
Polydora sp.	caw	NA	Polydora sp.	polydorid.complex
Prionospio aucklandica	NA	cmec	Prionospio sp.	prionospio.aucklandica
Prionospio sp.	caw	NA	Prionospio sp.	prionospio.other
Scolecolepides benhami	caw	cmec	Scolecolepides benhami	scolecolepides
Scolelepis sp.	caw	NA	Scolelepis sp.	scolelepis

D. Multivariate analysis methods

General analyses

Multivariate representation of the macrofaunal community data used the software package Primer v7.0.13 (Clarke et al. 2014). Patterns in similarity as a function of macrofaunal composition and abundance were assessed using an 'unconstrained' non-metric multidimensional scaling (nMDS) ordination plot, based on pairwise Bray-Curtis similarity index scores among samples aggregated within each site and sampling year. The purpose of sample aggregation was to smooth over the 'noise' associated with a core-level analysis, and enable the relationship to patterns in sediment quality variables to be better determined.

An initial Jaccard similarity analysis of the raw data (based on species presence and absence, irrespective of abundance) revealed temporal differences that were considered to potentially reflect taxonomic inconsistencies between the survey years (based on provider differences; see Appendix 3b above). To address this as part of the nMDS approach, it was necessary to aggregate some of the species or taxa to higher groups (e.g. genus, family, phylum), to minimise uncertainty associated with the macrofaunal identifications made in 2006 compared with 2013-2015. Appendix 3c above provides information on the taxonomic aggregation undertaken. Prior to analysis of the aggregated macrofaunal data, abundance values were square-root transformed to down-weight the influence on the ordination pattern of the most dominant species or higher taxa.

Overlay vectors and bubble plots were used to visualise relationships between multivariate biological patterns and sediment quality data, which were log(x+1)-transformed before analysis. Additionally, the Primer procedure Bio-Env was used to evaluate the suite of sediment quality variables that best explained the biological ordination pattern.

Benthic Health Model

The health of each site was assessed using recently developed National Benthic Health Models (BHMs; Clark et al. 2020). These models provide a health score, which indicates how healthy a site is with respect to stress from sedimentation (Mud BHM) and metal contamination (Metals BHM).

The Mud BHM tracks changes in health relative to increased mud content of the surface sediment as a surrogate for sediment accumulation rates. Mud BHM 'health' is defined by changes in benthic macroinvertebrate community structure observed along gradients of anthropogenic impact. This approach accounts for both acute effects and broader-scale degradation in community structure. Mud BHM scores are rated according to Clark et al. (2020) against a five-category scale. The scale simply divides the possible BHM scores of 1-6 across even rating bands that reflect a 'very low' to 'very high' impact relative to other New Zealand estuaries as follows: 1 to <2 (very low), 2 to <3 (low), 3 to <4 (moderate), 4 to <5 (high) and 5 to 6 (very high).

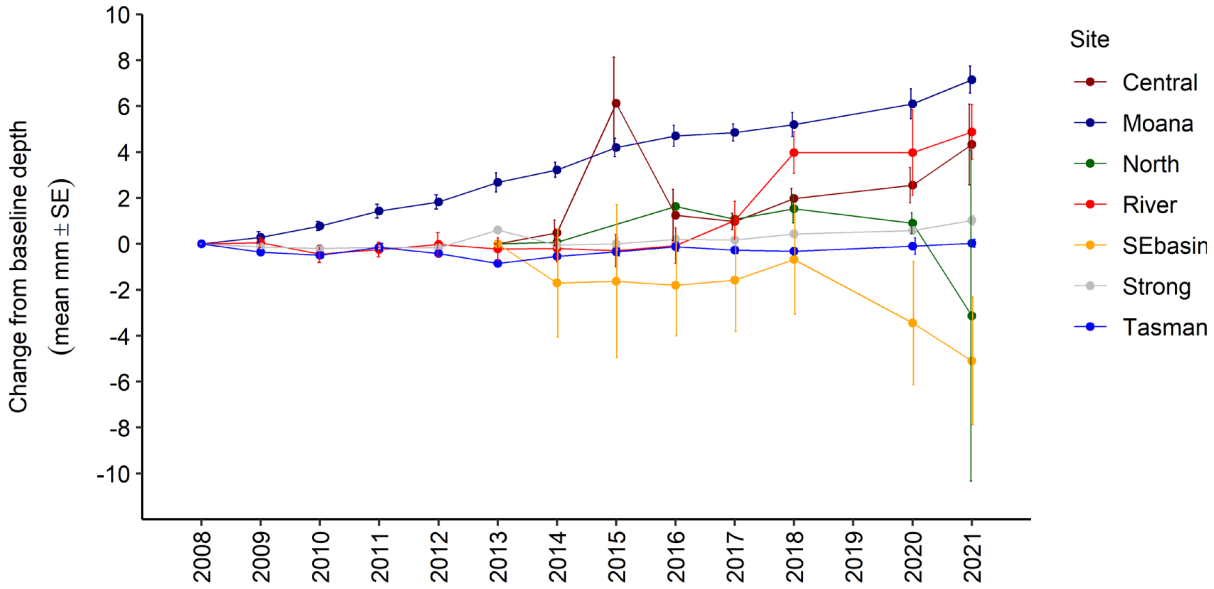
For Metals BHM scores, an absolute effects scale has recently been developed and is described by Clarke (2022, unpublished Cawthron report). The absolute approach categorises sediment health as 'good', 'fair' or 'poor' when assessed against a suite of sediment quality guidelines that are more conservative than the DGV thresholds of ANZG (2018).

For the present analysis, BHM scores were calculated by Dana Clark at Cawthron. Cawthron was provided with macroinvertebrate data standardised according to Clark et al. (2020), with replicates averaged by site for each year of sampling. Amphipods were not always identified to the level of taxonomic resolution required for BHMs. For most sites/times, the number of unidentified amphipods was low (<5 individuals). The influence that these unnamed amphipods may have on model scores was tested (data not shown) and deemed to be within the realm of natural variation.

BHM health scores were calculated following the methods of Clark et al. (2020) using PRIMER 7 (v 7.0.13) with the PERMANOVA+ add-on (Anderson et al. 2008; Clarke & Gorley 2015). The fit of the Mud BHM was assessed by plotting sediment mud content (log-transformed) against the Mud BHM scores to determine whether any sites/times fell outside of the model data points. The fit of the Metals BHM was assessed in the same manner using data from the site/times where sediment metal concentrations were available. Consistent with the Metals BHM, sediment metal concentrations were converted to a PC1 Metals gradient; a value that represents the combination of log-transformed copper, lead and zinc at each site. Mud and Metals BHM scores were then plotted at each site over time to explore changes in health over the last decade.

Appendix 4. Sediment plate summary data 2008-2021

For site locations see footnote to the Table below and Fig. 3 of main report. All sites established and data collected by Trevor James, TDC.



Mean change (± SE) in sediment depth over buried plates since the baseline was established. See Fig. 3 of main report for site locations.

Sedimentation data showing the average net change in sediment depth between the start and end of the monitoring period, and the average annual sedimentation rate across the period. Rating key as shown in Table 5 of main report (grey = very good, green = good). The national guideline value is 2mm/yr.

Site	Baseline date	Last sampling date	No years	Change from baseline depth (mm)	Annualised sedimentation (mm/yr since baseline)
Central basin	30/08/2013	30/11/2021	8.2	4.32	0.52
Moana Loop	25/09/2008	30/11/2021	13.2	7.15	0.54
North embayment*	30/08/2013	30/11/2021	8.2	-3.13	-0.38
Moutere River**	25/09/2008	30/11/2021	13.2	4.88	0.37
SE basin	30/08/2013	30/11/2021	8.2	-5.1	-0.62
Strong Loop	25/09/2008	30/11/2021	13.2	1.02	0.08
Tasman end	25/09/2008	30/11/2021	13.2	0.02	0.002

* North embayment site labelled as Wharf Rd embayment on Fig. 3 of report
** Moutere River site labelled as Robinson Rd embayment on Fig. 3 of report

Appendix 5. Sediment quality raw data 2013-2015

Raw data for 2006 in Cawthron report (Gillespie & Clark 2006). Organochlorine pesticides measured in 2014 only.

Site	Year	Zone	Gravel	Sand	Mud	TOC	TN	TP	aRPD	As	Cd	Cr	Cu	Ni	Pb	Zn
			%	%	%	%	mg/kg	mg/kg	mm	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
A	2013	X	1.8	76.3	21.9	1.23	<500	520	10	-	0.022	38	8	84	4.9	36
		Y	1.2	81.7	17.1	0.92	<500	540	10	-	0.022	35	7.4	87	4.5	34
		Z	1	83.3	15.7	0.93	500	500	10	-	0.021	37	7	90	4.3	32
	2014	X	1	80.7	18.2	0.34	<500	510	25	4.3	-	-	-	-	-	-
		Y	1.5	85.4	13.1	0.36	<500	500	-	-	-	-	-	-	-	-
		Z	0.7	87.4	11.9	0.32	<500	500	-	-	-	-	-	-	-	-
	2015	X	1.1	71.9	27	0.3	<500	580	10	-	-	-	-	-	-	-
		Y	1.6	74.6	23.7	0.27	<500	520	-	-	-	-	-	-	-	-
		Z	0.6	76.4	22.9	0.26	<500	550	-	-	-	-	-	-	-	-
B	2013	X	0.1	82.9	17	0.96	<500	480	10	-	0.021	31	6.5	66	4.9	31
		Y	0.2	82.1	17.8	1.05	<500	510	10	-	0.022	29	6.9	66	4.9	32
		Z	0.4	79.8	19.8	0.95	<500	500	10	-	0.023	32	7.1	67	5.1	32
	2014	X	0.3	86.7	13.1	0.3	<500	440	7	5.7	-	-	-	-	-	-
		Y	0.3	83.7	16	0.38	<500	470	-	-	-	-	-	-	-	-
		Z	0.5	87.5	12	0.32	<500	460	-	-	-	-	-	-	-	-
	2015	X	<0.1	79.5	20.4	0.25	<500	500	5	-	-	-	-	-	-	-
		Y	0.3	84.7	15.1	0.28	700	520	-	-	-	-	-	-	-	-
		Z	0.3	80	19.5	0.32	700	540	-	-	-	-	-	-	-	-
									DGV	20	1.5	80	65	21	50	200
									GV-high	70	10	370	270	52	220	410

Appendix 5 (cont.)

Organochlorine Pesticides Trace in Sediment	Sample Name:	Mout A 03-Mar-2014	Mout B 03-Mar-2014
Aldrin	mg/kg dry wt	< 0.0010	< 0.0010
alpha-BHC	mg/kg dry wt	< 0.0010	< 0.0010
beta-BHC	mg/kg dry wt	< 0.0010	< 0.0010
delta-BHC	mg/kg dry wt	< 0.0010	< 0.0010
gamma-BHC (Lindane)	mg/kg dry wt	< 0.0010	< 0.0010
cis-Chlordane	mg/kg dry wt	< 0.0010	< 0.0010
trans-Chlordane	mg/kg dry wt	< 0.0010	< 0.0010
2,4'-DDD	mg/kg dry wt	< 0.0010	< 0.0010
4,4'-DDD	mg/kg dry wt	< 0.0010	< 0.0010
2,4'-DDE	mg/kg dry wt	< 0.0010	< 0.0010
4,4'-DDE	mg/kg dry wt	< 0.0010	< 0.0010
2,4'-DDT	mg/kg dry wt	< 0.0010	< 0.0010
4,4'-DDT	mg/kg dry wt	0.0017	0.0013
Dieldrin	mg/kg dry wt	< 0.0010	< 0.0010
Endosulfan I	mg/kg dry wt	< 0.0010	< 0.0010
Endosulfan II	mg/kg dry wt	< 0.0010	< 0.0010
Endosulfan sulphate	mg/kg dry wt	< 0.0010	< 0.0010
Endrin	mg/kg dry wt	< 0.0010	< 0.0010
Endrin aldehyde	mg/kg dry wt	< 0.0010	< 0.0010
Endrin ketone	mg/kg dry wt	< 0.0010	< 0.0010
Heptachlor	mg/kg dry wt	< 0.0010	< 0.0010
Heptachlor epoxide	mg/kg dry wt	< 0.0010	< 0.0010
Hexachlorobenzene	mg/kg dry wt	< 0.0010	< 0.0010
Methoxychlor	mg/kg dry wt	< 0.0010	< 0.0010
Total Chlordane [(cis+trans)*100/42]	mg/kg dry wt	< 0.002	< 0.002
Organonitro&phosphorus Pesticides Trace in MR Soil by GCMS			
Acetochlor	mg/kg dry wt	< 0.009	< 0.008
Alachlor	mg/kg dry wt	< 0.006	< 0.006
Atrazine	mg/kg dry wt	< 0.009	< 0.008
Atrazine-desethyl	mg/kg dry wt	< 0.009	< 0.008
Atrazine-desisopropyl	mg/kg dry wt	< 0.017	< 0.016
Azaconazole	mg/kg dry wt	< 0.005	< 0.004
Azinphos-methyl	mg/kg dry wt	< 0.017	< 0.016
Benalaxyl	mg/kg dry wt	< 0.005	< 0.004
Bitertanol	mg/kg dry wt	< 0.017	< 0.016
Bromacil	mg/kg dry wt	< 0.009	< 0.008
Bromopropylate	mg/kg dry wt	< 0.009	< 0.008
Butachlor	mg/kg dry wt	< 0.009	< 0.008
Captan	mg/kg dry wt	< 0.017	< 0.016
Carbaryl	mg/kg dry wt	< 0.009	< 0.008
Carbofuran	mg/kg dry wt	< 0.009	< 0.008
Chlorfluazuron	mg/kg dry wt	< 0.009	< 0.008
Chlorothalonil	mg/kg dry wt	< 0.009	< 0.008
Chlorpyrifos	mg/kg dry wt	< 0.009	< 0.008
Chlorpyrifos-methyl	mg/kg dry wt	< 0.009	< 0.008
Chlortoluron	mg/kg dry wt	< 0.017	< 0.016
Cyanazine	mg/kg dry wt	< 0.009	< 0.008
Cyfluthrin	mg/kg dry wt	< 0.009	< 0.008
Cyhalothrin	mg/kg dry wt	< 0.009	< 0.008
Cypermethrin	mg/kg dry wt	< 0.017	< 0.016
Deltamethrin (including Tralomethrin)	mg/kg dry wt	< 0.009	< 0.008
Diazinon	mg/kg dry wt	< 0.005	< 0.004
Dichlofluanid	mg/kg dry wt	< 0.009	< 0.008
Dichloran	mg/kg dry wt	< 0.03	< 0.03
Dichlorvos	mg/kg dry wt	< 0.010	< 0.010
Difenoconazole	mg/kg dry wt	< 0.012	< 0.012
Dimethoate	mg/kg dry wt	< 0.017	< 0.016
Diphenylamine	mg/kg dry wt	< 0.017	< 0.016

Appendix 5 (cont.)

Organochlorine Pesticides Trace in Sediment	Sample Name:	Mout A 03-Mar-2014	Mout B 03-Mar-2014
Diuron	mg/kg dry wt	< 0.009	< 0.008
Fenpropimorph	mg/kg dry wt	< 0.009	< 0.008
Fluazifop-butyl	mg/kg dry wt	< 0.009	< 0.008
Fluometuron	mg/kg dry wt	< 0.009	< 0.008
Flusilazole	mg/kg dry wt	< 0.009	< 0.008
Fluvalinate	mg/kg dry wt	< 0.006	< 0.006
Furalaxyl	mg/kg dry wt	< 0.005	< 0.004
Haloxyfop-methyl	mg/kg dry wt	< 0.009	< 0.008
Hexaconazole	mg/kg dry wt	< 0.009	< 0.008
Hexazinone	mg/kg dry wt	< 0.005	< 0.004
IPBC (3-Iodo-2-propynyl-n-butylcarbamate)	mg/kg dry wt	< 0.05	< 0.04
Kresoxim-methyl	mg/kg dry wt	< 0.005	< 0.004
Linuron	mg/kg dry wt	< 0.009	< 0.008
Malathion	mg/kg dry wt	< 0.009	< 0.008
Metalaxyl	mg/kg dry wt	< 0.009	< 0.008
Methamidophos	mg/kg dry wt	< 0.05	< 0.04
Metolachlor	mg/kg dry wt	< 0.006	< 0.006
Metribuzin	mg/kg dry wt	< 0.009	< 0.008
Molinate	mg/kg dry wt	< 0.017	< 0.016
Myclobutanil	mg/kg dry wt	< 0.009	< 0.008
Naled	mg/kg dry wt	< 0.05	< 0.04
Norflurazon	mg/kg dry wt	< 0.017	< 0.016
Oxadiazon	mg/kg dry wt	< 0.009	< 0.008
Oxyfluorfen	mg/kg dry wt	< 0.005	< 0.004
Paclobutrazol	mg/kg dry wt	< 0.009	< 0.008
Parathion-ethyl	mg/kg dry wt	< 0.009	< 0.008
Parathion-methyl	mg/kg dry wt	< 0.009	< 0.008
Pendimethalin	mg/kg dry wt	< 0.009	< 0.008
Permethrin	mg/kg dry wt	< 0.003	< 0.003
Pirimicarb	mg/kg dry wt	< 0.009	< 0.008
Pirimiphos-methyl	mg/kg dry wt	< 0.009	< 0.008
Prochloraz	mg/kg dry wt	< 0.05	< 0.04
Procymidone	mg/kg dry wt	< 0.009	< 0.008
Prometryn	mg/kg dry wt	< 0.005	< 0.004
Propachlor	mg/kg dry wt	< 0.009	< 0.008
Propanil	mg/kg dry wt	< 0.03	< 0.03
Propazine	mg/kg dry wt	< 0.005	< 0.004
Propiconazole	mg/kg dry wt	< 0.006	< 0.006
Pyriproxyfen	mg/kg dry wt	< 0.009	< 0.008
Quizalofop-ethyl	mg/kg dry wt	< 0.009	< 0.008
Simazine	mg/kg dry wt	< 0.009	< 0.008
Simetryn	mg/kg dry wt	< 0.009	< 0.008
Sulfentrazone	mg/kg dry wt	< 0.05	< 0.04
TCMTB [2-(thiocyanomethylthio)benzothiazole,Busan]	mg/kg dry wt	< 0.017	< 0.016
Tebuconazole	mg/kg dry wt	< 0.009	< 0.008
Terbacil	mg/kg dry wt	< 0.009	< 0.008
Terbumeton	mg/kg dry wt	< 0.009	< 0.008
Terbuthylazine	mg/kg dry wt	< 0.005	< 0.004
Terbuthylazine-desethyl	mg/kg dry wt	< 0.009	< 0.008
Terbutryn	mg/kg dry wt	< 0.009	< 0.008
Thiabendazole	mg/kg dry wt	< 0.05	< 0.04
Thiobencarb	mg/kg dry wt	< 0.009	< 0.008
Tolylfluanid	mg/kg dry wt	< 0.005	< 0.004
Triazophos	mg/kg dry wt	< 0.009	< 0.008
Trifluralin	mg/kg dry wt	< 0.009	< 0.008
Vinclozolin	mg/kg dry wt	< 0.009	< 0.008

Appendix 6. Macrofauna data 2013-2015

Raw core data have been provided electronically to TDC. Taxa list for 2006 in Cawthron report (Gillespie & Clark 2006).

2013 Data

Main group	Taxa	Habitat	EG	13A1	13A2	13A3	13A4	13A5	13A6	13A7	13A8	13A9	13A10	13B1	13B2	13B3	13B4	13B5	13B6	13B7	13B8	13B9	13B10
Amphipoda	Amphipoda A	Infauna	II																				
Amphipoda	Amphipoda B	Infauna	II																				
Amphipoda	Amphipoda C	Infauna	II																				
Amphipoda	Amphipoda sp. 1	Infauna	II				1																
Amphipoda	Paracalliope novizealandiae	Infauna	II										1										
Amphipoda	Torridoharpinia hurleyi	Infauna	II	2			1			1	1					1							
Anthozoa	Anthopleura aureoradiata	Epibiota	III			1		1			3					1							
Anthozoa	Edwardsia sp.	Epibiota	II																				
Bivalvia	Arthritica bifurca	Infauna	IV																				
Bivalvia	Arthritica sp. 1	Infauna	IV						1														
Bivalvia	Austrovenus stutchburyi	Infauna	II	6		6	4	1	2	4	2	2	2	4	2	6	5	11	9	5	12	10	5
Bivalvia	Hiatula spp.	Infauna	I										1								1		
Bivalvia	Linucula hartvigiana	Infauna	II	1	1	3	2	5	4	3	1	1	1		2	1		2		3	1	1	
Bivalvia	Macomona liliana	Infauna	II	7	10	7	6	6	7	10	6	5	7	5	4	5	6	4	6	8	10	3	6
Cumacea	Colurostylis lemurum	Infauna	I																				
Cumacea	Cumacea	Infauna	I																				
Decapoda	Austrohelice crassa	Infauna	V																				2
Decapoda	Callianassa filholi	Infauna	I																				
Decapoda	Halicarcinus whitei	Infauna	III																				
Decapoda	Hemiplax hirtipes	Infauna	V					1					1			2	2					1	1
Diptera	Diptera sp. 1	Larva	II																		1		
Diptera	Dolichopodidae larvae	Larva	II																				
Gastropoda	Amphibola crenata	Epibiota	III																				
Gastropoda	Cominella glandiformis	Epibiota	III	2										1		1							
Gastropoda	Diloma subrostratum	Epibiota	II		1	1									1	1		1					
Gastropoda	Diloma zelandicum	Epibiota	NA																				
Gastropoda	Micrelenchus tenebrosus	Epibiota	I																				
Gastropoda	Notoacmea spp.	Epibiota	II					1							1								
Gastropoda	Papawera zelandiae	Epibiota	I																				
Gastropoda	Zeacumantus lutulentus	Epibiota	II			2					1			1			1	2					
Gastropoda	Zeacumantus subcarinatus	Epibiota	I																				
Holothuroidea	Trochodota dendyi	Infauna	NA																				
Mysidacea	Mysidacea	Infauna	I																				
Mysidacea	Tenagomysis sp.	Infauna	II																				
Nematoda	Nematoda	Infauna	II																				
Nemertea	Nemertea	Infauna	III																				
Nemertea	Nemertea sp. 1	Infauna	III		1																		
Nemertea	Nemertea sp. 2	Infauna	III							1		1	1										
Nemertea	Nemertea sp. 3	Infauna	III											1	1	1							
Nemertea	Nemertea sp. 4	Infauna	III																				
Nemertea	Nemertea sp. 5	Infauna	III																				
Oligochaeta	Oligochaeta	Infauna	III																				
Polychaeta	Aglaophamus macroura	Infauna	II																				
Polychaeta	Aonides sp.	Infauna	I																				
Polychaeta	Aonides trifida	Infauna	I							1			1			1							
Polychaeta	Armandia maculata	Infauna	II					1															
Polychaeta	Axiothella serrata	Infauna	II	1		1	1			1	1		2						1		1		
Polychaeta	Boccardia acus	Infauna	II																				
Polychaeta	Boccardia sp.	Infauna	II																				
Polychaeta	Boccardia syrtis	Infauna	II				1		1			1					1						
Polychaeta	Capitella capitata	Infauna	IV																				
Polychaeta	Capitella sp. 1	Infauna	IV									1											
Polychaeta	Cirratulidae	Infauna	III																				
Polychaeta	Disconatis accolus	Infauna	I																				
Polychaeta	Glycera lamelliformis	Infauna	III	1							1	1											
Polychaeta	Glyceridae	Infauna	III																				
Polychaeta	Heteromastus filiformis	Infauna	III		1			2					3	2	3		4		3		1	1	3
Polychaeta	Lagis australis	Infauna	III																				
Polychaeta	Magelona dakini	Infauna	III														1				1		
Polychaeta	Maldanidae	Infauna	I																				
Polychaeta	Nereididae (juv)	Infauna Juv	NA	1		1		2	1				1			1		2		1	1	1	
Polychaeta	Nicon aestuariensis	Infauna	III																				
Polychaeta	Orbinia papillosa	Infauna	I													1							
Polychaeta	Owenia petersenae	Infauna	II	1	1																		
Polychaeta	Paradoneis sp.	Infauna	III																				
Polychaeta	Paraonidae	Infauna	III																				
Polychaeta	Polydora sp.	Infauna	III																				
Polychaeta	Prionospio aucklandica	Infauna	II	3	1	15	13	2	3	7	7	10	7	3	1	14	7	1	7	4	1		
Polychaeta	Prionospio sp.	Infauna	II																				
Polychaeta	Scolecolepides benhami	Infauna	IV	1			1																1
Polychaeta	Scolelepis sp.	Infauna	I																				

Appendix 6 (cont.)

2014 Data

Main group	Taxa	Habitat	EG	14A1	14A2	14A3	14A4	14A5	14A6	14A7	14A8	14A9	14A10	14B1	14B2	14B3	14B4	14B5	14B6	14B7	14B8	14B9	14B10
Amphipoda	Amphipoda A	Infauna	II																				
Amphipoda	Amphipoda B	Infauna	II																				
Amphipoda	Amphipoda C	Infauna	II																				
Amphipoda	Amphipoda sp. 1	Infauna	II		1					1		1											
Amphipoda	Paracalliope novizealandiae	Infauna	II			1																	
Amphipoda	Torridoharpinia hurleyi	Infauna	II	1	2	1	1			1		1					1	1	1	2		1	2
Anthozoa	Anthopleura aureoradiata	Epibiota	III	2						1							1		1	1			1
Anthozoa	Edwardsia sp.	Epibiota	II																				
Bivalvia	Arthritica bifurca	Infauna	IV																				
Bivalvia	Arthritica sp. 1	Infauna	IV																				
Bivalvia	Austrovenus stutchburyi	Infauna	II	2	2	1		2	1	1	3	2	1	5	4	4	6		6	3	3	2	8
Bivalvia	Hiatula spp.	Infauna	I								2												
Bivalvia	Linucula hartvigiana	Infauna	II	2	4		3	2	2	3	5	1	4	1	2		5		4	2	2	1	3
Bivalvia	Macomona liliana	Infauna	II	6	1	2	4	7	5	3	4	7	5	6	3	6	7	2	4	6	6	8	6
Cumacea	Colurostylis lemurum	Infauna	I																				
Cumacea	Cumacea	Infauna	I																				
Decapoda	Austrohelice crassa	Infauna	V							1													
Decapoda	Callianassa filholi	Infauna	I																				
Decapoda	Halicarcinus whitei	Infauna	III	2	1			2											2	1			1
Decapoda	Hemiplax hirtipes	Infauna	V																			1	
Diptera	Diptera sp. 1	Larva	II															1			1		1
Diptera	Dolichopodidae larvae	Larva	II																				
Gastropoda	Amphibola crenata	Epibiota	III																				
Gastropoda	Cominella glandiformis	Epibiota	III	1	1							1								1			1
Gastropoda	Diloma subrostratum	Epibiota	II										1										1
Gastropoda	Diloma zelandicum	Epibiota	NA																				
Gastropoda	Micrelenchus tenebrosus	Epibiota	I																				
Gastropoda	Notoacmea spp.	Epibiota	II			1					1						1			1			
Gastropoda	Papawera zelandiae	Epibiota	I																				
Gastropoda	Zeacumantus lutulentus	Epibiota	II		1								2										1
Gastropoda	Zeacumantus subcarinatus	Epibiota	I																				
Holothuroidea	Trochodota dendyi	Infauna	NA																				
Mysidacea	Mysidacea	Infauna	I																				
Mysidacea	Tenagomysis sp.	Infauna	II																				
Nematoda	Nematoda	Infauna	II																				
Nemertea	Nemertea	Infauna	III																				
Nemertea	Nemertea sp. 1	Infauna	III	1			1			1		2						1	1				
Nemertea	Nemertea sp. 2	Infauna	III									1											
Nemertea	Nemertea sp. 3	Infauna	III																			1	
Nemertea	Nemertea sp. 4	Infauna	III																		1		
Nemertea	Nemertea sp. 5	Infauna	III																				
Oligochaeta	Oligochaeta	Infauna	III																				
Polychaeta	Aglaophamus macroura	Infauna	II																				
Polychaeta	Aonides sp.	Infauna	I																				
Polychaeta	Aonides trifida	Infauna	I								1				1	2		1			1		
Polychaeta	Armandia maculata	Infauna	II																				
Polychaeta	Axiothella serrata	Infauna	II	3	1	1			2	1		1	3		1			1			1		
Polychaeta	Boccardia acus	Infauna	II																				
Polychaeta	Boccardia sp.	Infauna	II																				
Polychaeta	Boccardia syrtis	Infauna	II				1			2			1	1		1			1				
Polychaeta	Capitella capitata	Infauna	IV																				
Polychaeta	Capitella sp. 1	Infauna	IV																				
Polychaeta	Cirratulidae	Infauna	III																				
Polychaeta	Disconatis accolus	Infauna	I						1												1		
Polychaeta	Glycera lamelliformis	Infauna	III	1	1	1	1			1	1			2					1				1
Polychaeta	Glyceridae	Infauna	III																				
Polychaeta	Heteromastus filiformis	Infauna	III	3	1						7		2		2		1	1	1		4		
Polychaeta	Lagis australis	Infauna	III				1																
Polychaeta	Magelona dakini	Infauna	III		1		1			1													
Polychaeta	Maldanidae	Infauna	I																				
Polychaeta	Nereididae (juv)	Infauna	Juv	NA	1	1	1	1		1	1		1			4	2	4	1	3	2			2
Polychaeta	Nicon aestuariensis	Infauna	III					1															
Polychaeta	Orbinia papillosa	Infauna	I																			1	1
Polychaeta	Owenia petersenae	Infauna	II																				
Polychaeta	Paradoneis sp.	Infauna	III								1									1		1	
Polychaeta	Paraonidae	Infauna	III																				
Polychaeta	Polydora sp.	Infauna	III																				
Polychaeta	Prionospio aucklandica	Infauna	II	4	6	9	2	3	3	6	21	1	12	4	1	3	6		13	21	8	9	2
Polychaeta	Prionospio sp.	Infauna	II																				
Polychaeta	Scolecolepides benhami	Infauna	IV			1						1						2			1		1
Polychaeta	Scolelepis sp.	Infauna	I																				

Appendix 6 (cont.)

2015 Data

Main group	Taxa	Habitat	EG	15A1	15A2	15A3	15A4	15A5	15A6	15A7	15A8	15A9	15A10	15B1	15B2	15B3	15B4	15B5	15B6	15B7	15B8	15B9	15B10
Amphipoda	Amphipoda A	Infauna	II																				
Amphipoda	Amphipoda B	Infauna	II																				
Amphipoda	Amphipoda C	Infauna	II																				
Amphipoda	Amphipoda sp. 1	Infauna	II																				
Amphipoda	Paracalliope novizealandiae	Infauna	II																				
Amphipoda	Torridoharpinia hurleyi	Infauna	II			1		1						1	2	2	2	4	2	2		1	
Anthozoa	Anthopleura aureoradiata	Epibiota	III	1											2	1		1	5				3
Anthozoa	Edwardsia sp.	Epibiota	II																				
Bivalvia	Arthritica bifurca	Infauna	IV																				
Bivalvia	Arthritica sp. 1	Infauna	IV														1						
Bivalvia	Austrovenus stutchburyi	Infauna	II	5	5	3	4	3	6	2	2		3	5	1	7	3	8	4	6	7	5	4
Bivalvia	Hiatula spp.	Infauna	I	1												2							
Bivalvia	Linucula hartvigiana	Infauna	II	2	2		2	1					4			3	3	2	4	4	1	6	3
Bivalvia	Macomona liliana	Infauna	II	6	5	3	4	7	7	7	5	5	6	5	3	4	8	5	9	4	3		7
Cumacea	Colurostylis lemurum	Infauna	I												1								
Cumacea	Cumacea	Infauna	I																				
Decapoda	Austrohelice crassa	Infauna	V																				
Decapoda	Callianassa filholi	Infauna	I																				
Decapoda	Halicarcinus whitei	Infauna	III				2		2	2			1			3	1			1	1		
Decapoda	Hemiplax hirtipes	Infauna	V			1		1				1					1		1				1
Diptera	Diptera sp. 1	Larva	II					1															
Diptera	Dolichopodidae larvae	Larva	II																				
Gastropoda	Amphibola crenata	Epibiota	III					1															
Gastropoda	Cominella glandiformis	Epibiota	III									1	1										
Gastropoda	Diloma subrostratum	Epibiota	II																				1
Gastropoda	Diloma zelandicum	Epibiota	NA																				
Gastropoda	Micrelenchus tenebrosus	Epibiota	I																				
Gastropoda	Notoacmea spp.	Epibiota	II						1									1	1				
Gastropoda	Papawera zelandiae	Epibiota	I																				
Gastropoda	Zeacumantus lutulentus	Epibiota	II														1						
Gastropoda	Zeacumantus subcarinatus	Epibiota	I																				
Holothuroidea	Trochodota dendyi	Infauna	NA																				
Mysidacea	Mysidacea	Infauna	I																				
Mysidacea	Tenagomysis sp.	Infauna	II		1																		
Nematoda	Nematoda	Infauna	II																				
Nemertea	Nemertea	Infauna	III																				
Nemertea	Nemertea sp. 1	Infauna	III			1								1	1		1		3	1		1	1
Nemertea	Nemertea sp. 2	Infauna	III	1	1			1				1											
Nemertea	Nemertea sp. 3	Infauna	III																				
Nemertea	Nemertea sp. 4	Infauna	III																				
Nemertea	Nemertea sp. 5	Infauna	III													1							
Oligochaeta	Oligochaeta	Infauna	III													1							
Polychaeta	Aglaophamus macroura	Infauna	II														1			2			
Polychaeta	Aonides sp.	Infauna	I																				
Polychaeta	Aonides trifida	Infauna	I							1				1				1	1				
Polychaeta	Armandia maculata	Infauna	II																				
Polychaeta	Axiothella serrata	Infauna	II	2	2	2	5	1	7	4		1	1			1							
Polychaeta	Boccardia acus	Infauna	II	2	4	2			3				2								1		1
Polychaeta	Boccardia sp.	Infauna	II																				
Polychaeta	Boccardia syrtis	Infauna	II	2	4		2		1	1		1			2		1	1			1		
Polychaeta	Capitella capitata	Infauna	IV																				
Polychaeta	Capitella sp. 1	Infauna	IV																				
Polychaeta	Cirratulidae	Infauna	III																				
Polychaeta	Disconatis accolus	Infauna	I	1			1		1	2	1									1			
Polychaeta	Glycera lamelliformis	Infauna	III				1											1					1
Polychaeta	Glyceridae	Infauna	III																				
Polychaeta	Heteromastus filiformis	Infauna	III	3	1	3		4	5	5	1	1		3	3		3	2	4	3	6	1	
Polychaeta	Lagis australis	Infauna	III																				
Polychaeta	Magelona dakini	Infauna	III		1		1		1	1													
Polychaeta	Maldanidae	Infauna	I																				
Polychaeta	Nereididae (juv)	Infauna Juv	NA	2			2		1	1	2	1		1	1	1		1	1	2		3	1
Polychaeta	Nicon aestuariensis	Infauna	III		1	1							3										
Polychaeta	Orbinia papillosa	Infauna	I					2	1				1			3		1		1	1		
Polychaeta	Owenia petersenae	Infauna	II						2	1													
Polychaeta	Paradoneis sp.	Infauna	III	5	1	1	1	7				1			1	1							
Polychaeta	Paraonidae	Infauna	III																				
Polychaeta	Polydora sp.	Infauna	III																				
Polychaeta	Prionospio aucklandica	Infauna	II	15	16	16	9	15	13	17		7	4	6	3	14	8	14	13	5	12	2	9
Polychaeta	Prionospio sp.	Infauna	II																				
Polychaeta	Scolecolepides benhami	Infauna	IV		1																		
Polychaeta	Scolelepis sp.	Infauna	I																				

Appendix 7. Macrofauna sampling optimisation

Summary

The current NEMP protocol specifying 10 macrofauna cores per site may not be optimal for statistical testing, and complete characterisation of the species pool. However, given the cost of macrofauna sample processing, and in light of the long-term dataset that has been developed for Moutere Inlet, it is not considered necessary to increase the number of cores beyond 10. In fact, reducing sampling to 9 cores would have a minor effect on ability to detect change and have the benefit of reduced taxonomy costs. Collection of 9 cores would also cater for a simplified 3x3 field sampling grid, compared with the present situation in which cores are taken from 10 random plots out of 12 available (i.e. reflecting a 3x4 grid).

A7.1. Background

The National Estuarine Monitoring Protocol (NEMP) recommended collecting 10 macrofauna core samples per site (reps) based on an analysis of a national dataset in 2002 (Robertson et al. 2002). This average sampling effort appeared to have been biased upwards slightly by sediment chemistry indicators, with the recommended number of reps specifically for species richness (S) reported as 7-8, and for abundance (N) 8-9. NIWA have released a recent guidance document recommending collection of 12 reps twice yearly for macrofaunal sampling (Hewitt 2021), based on long term work in Manukau Harbour.

The purpose of this document is to reassess macrofauna sampling requirements for Moutere Inlet considering:

- The NEMP approach, which was based on the coefficient of variation (CV) in univariate responses as a function of increasing sampling effort, using pooled estuary reps.
- An approach based on power analysis that reflects previous NIWA work (Hewitt et al. 1993; Hewitt 2021) and considers the levels of minimum detectable change in three univariate responses analysed in the report (S, N, AMBI).
- An approach based on species detection, which considers the percentage of the 'true' estimated pool of species that is captured by different levels of sampling effort. This approach is particularly relevant to multivariate analysis, for which knowledge of species detection provides insight into whether assessed differences in ecological communities among sites or times are true differences or are potentially biased by under-sampling of less common species.

There are additional more recent and sophisticated approaches that could be explored, including change detection in trends, multivariate approaches, and multilevel occupancy modelling, but going to this level of analysis would justify a standalone technical report and was beyond present scope.

A7.2 Description of NEMP approach

The NEMP approach was to model the coefficient of variation (CV) as a function of increasing reps, using pooled estuary reps, then determine a cost-benefit-point (CBP) whereby further increases in sample size yielded insubstantial returns (Robertson et al. 2002). The CBPs were used to assess levels of detectable change, sometimes referred to as statistical power. CV is the sample standard deviation divided by the sample mean, and a relative measure that could be compared across sites, estuaries, or even indicators. However, the value of using this statistic for determining optimal sample size lies solely in the sample estimate standard deviation, where increasing reps should decrease this measure of variation, given certain assumptions and bias corrections.

An improvement in the NEMP approach would be to consider standard error (SE), which is standard deviation divided by the square root of sample size. This was the approach taken by Hewitt et al. (1993) to optimize the trade-off between accuracy and cost for species abundance monitoring in Manukau Harbour. Figure A7.1 plots the change in SE of the 3 univariates responses (S, N, AMBI) in relation to sampling effort, with power curve extrapolations used to estimate SE beyond the number of actual samples taken. The graphs show the diminishing returns arising from sampling beyond the current effort of 10 reps. Of course, the specific responses are site and time dependent, which is smoothed over by the averaging in Fig. A7.1.

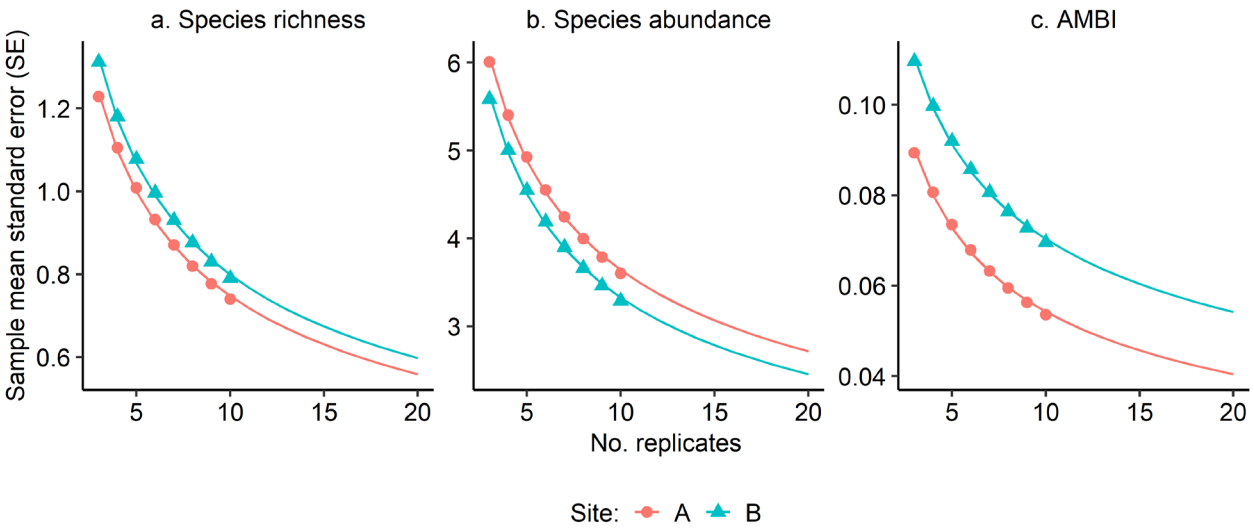


Figure A7.1. Standard error (SE) for Moutere Inlet species richness sample means plotted against the number of replicates, coloured by site. The markers show the SE of observed data, and the lines are simple power curve extrapolations. Note the differing scale of the y-axis, where SEs for species abundance (b) are much higher than that of species richness (a) and AMBI (c).

A7.3 Power analysis of univariate responses

Power analysis considers the 'effect size' that a certain statistical test could detect given differing data variance and sampling effort. This approach is of most interest for statistical tests of inter-year or inter-site differences in mean macrofauna responses. Figure A7.2 plots the average minimum detectable percentage change for each of the 3 macrofauna response variables as a function of sampling effort. Minimum detectable change is calculated as the change required for paired t-tests to signify a non-zero change in the sampling mean at each site from year to year, with type I and II error rates thresholds of 0.05 and 0.20 (Champely 2020). A summary of results is in Table A7.1.

These results are very similar to Figure A7.1, revealing that AMBI responses have the least variation rep-to-rep on average (i.e. changes in the AMBI response can be detected with the least sampling effort), followed by S and N. At the current level of NEMP sampling using 10 macrofaunal reps, changes in sample means of S, N and AMBI of ~28%, 37% and 14% could be detected. Increasing this number to 12 reps (as recommended by NIWA twice yearly for seasonality and change in trend detection, Hewitt 2021) does not appreciably improve accuracy. Similarly, a decrease in effort to 9 reps has very little effect in terms of loss of information. Reducing effort to 9 reps would have the benefit of reducing sample processing costs by 10% and enable sampling with a 3x3m grid. This grid configuration would simplify field sampling compared with the present situation in which cores are taken from 10 random plots out of 12 available (i.e. reflecting a 3x4 grid).

Table A7.1. Minimum detectable change (%) in sample mean (averaged across years and sites) under standard statistical testing conditions. i.e., if average richness was 13 from 10 reps at a site in year 1 and we took another 10 reps at year 2, a paired t-test for change in sample mean would suggest that an observed richness approximately less than 9.3 or greater than 16.7 (+/-28.3%), would not just be due to chance (alpha=0.05).

	No. reps								
Response	5	6	7	8	9	10	12	41*	84*
S	38.5	35.6	33.2	31.3	29.7	28.3	26.5	15.9	11.8
N	50.5	46.6	43.5	40.9	38.8	36.9	34.5	20.5	15.1
AMBI	18.1	16.8	15.8	14.9	14.2	13.5	12.7	7.8	5.9

* Note: the illustration of 41 and 84 reps was based on estimated species detection thresholds (of ~90% and 100%, respectively) described in Section A7.4.

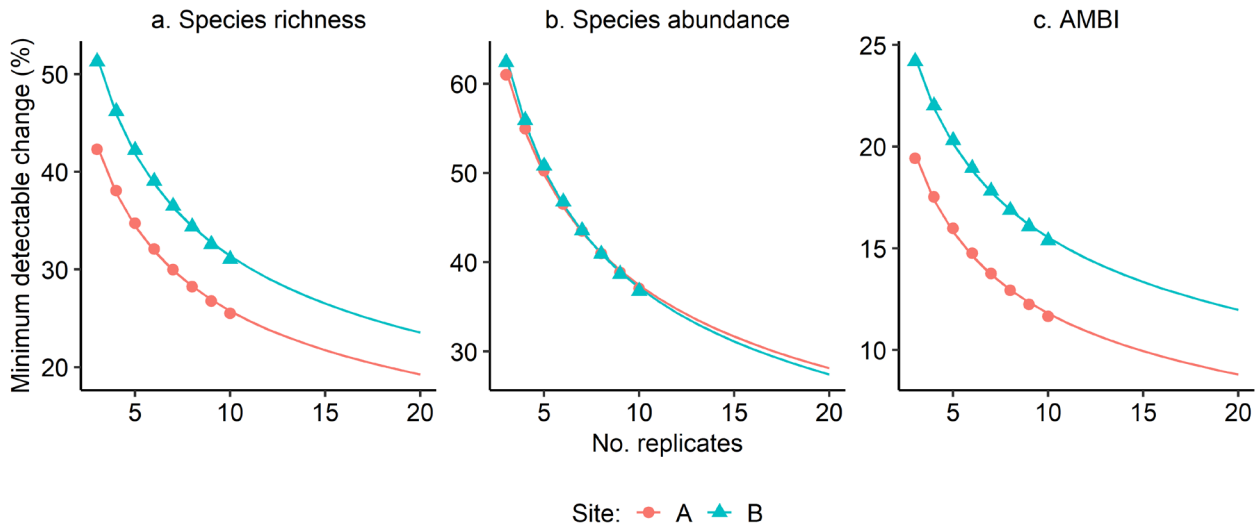


Figure A7.2. Minimum detectable change (%) in mean univariate responses plotted against the number of replicates each year, coloured by site. The markers show the detectable change (%) of observed data and the lines are simple power curve extrapolations. These data can be interpreted as minimum percentage change required for a paired t-test to indicate this difference would not just be due to chance (alpha=0.05), i.e. a change in sample mean significantly greater than zero.

A7.4 Species detection

The final approach considered was extrapolation of rarefaction curves, which is a permutation-based approach that describes the cumulative number of species detected with an increase in sampling effort. Typically such curves approach an asymptote, reflecting diminishing returns as sampling effort increases. Various techniques can be used to model the number of total species number where this asymptote is reached, which is the estimate of 'true' total species richness. This approach enables a CBP to be chosen based on the desired percentage of the estimated true total richness to be captured by a sampling programme. Achieving 100% species detection is unlikely to be practically attainable, due to the chance sampling of uncommon/rare species.

For present purposes several total species richness estimators were used and compared, with the Chao1 estimator from the iNEXT R package chosen as the most appropriate (Chao et al. 2014; Hsieh et al. 2020; R Core Team 2021). Table A7.2 suggests that under the current 10 core NEMP protocol only about 67% of total site richness is being detected on average at each site each year. Reducing sampling to 9 reps would decrease this figure to about 65%, while increasing to 12 cores would increase it to 71%. However, 41 or more reps might be needed to capture 90% of total site richness.

Figure A7.3 plots this data for each site-year and shows that returns in species richness for increasing sampling effort do not diminish as quickly as they do for SE (Figure A7.1) and minimum detectable % change (Figure A7.2). The differences between these species detection results and those of the more traditional statistical approaches above highlights the value in comparing multiple measures of sampling efficacy when determining a CBP.

Table A7.2. The average percentage of estimated total site richness captured over all sites and years at differing sampling effort. The columns showing 41 and 84* reps indicate the effort required to capture approximately 90 and almost 100% of estimated total richness in any given site-year.

	No. reps								
Site	5	6	7	8	9	10	12	41	84*
A	48.7	52.2	55.1	57.7	60	62.2	65.9	87	97.9 (91)
B	57	60.8	64.1	66.9	69.4	71.7	75.6	94.4	99.8 (76)
Average	52.8	56.5	59.6	62.3	64.7	66.9	70.8	90.7	98.8 (84)

* Note: an average of 84 reps was needed to reach almost 100% of total site richness; some sites reached this with more or fewer reps than others (Figure A7.3). The total number of reps needed for ~100% detection is shown in brackets.

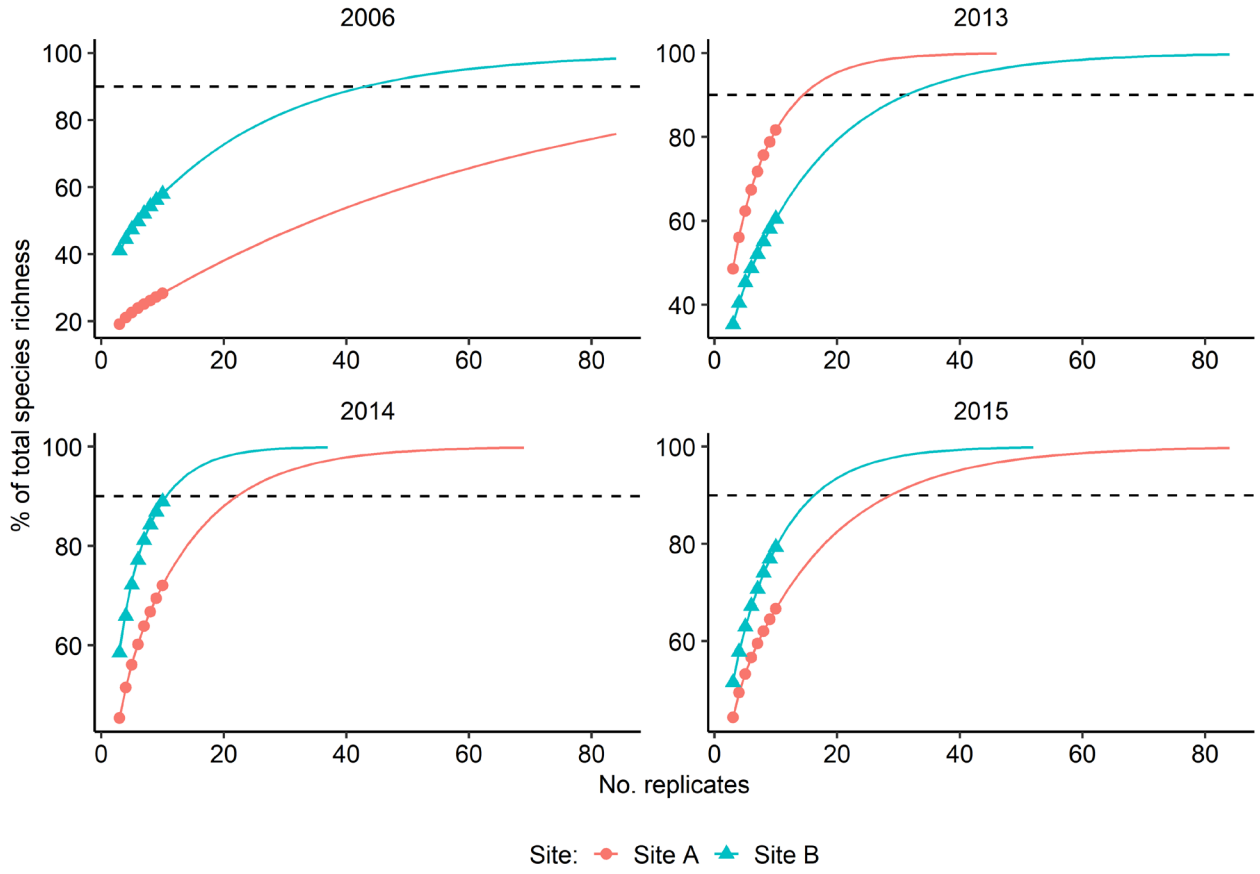


Figure A7.3. Percentage of total estimated richness at each site plotted against the number of replicates. Subplots correspond to sampling years. The points on the graph show % of total richness calculated from observed data and the lines are extrapolations towards the estimated 100% richness using the iNEXT package in R (Hsieh et al. 2020, R Core Team 2021). The dashed horizontal line indicates an estimated 90% of species detected.

References

Champely S 2020. pwr: Basic Functions for Power Analysis. R package version 1.3-0. https://CRAN.R-project.org/package=pwr.

Chao A, Gotelli NJ, Hsieh TC, Sander EL, Ma KH, Colwell RK, Ellison AM 2014. Rarefaction and extrapolation with Hill numbers: a framework for sampling and estimation in species diversity studies. Ecological Monographs 84: 45-67.

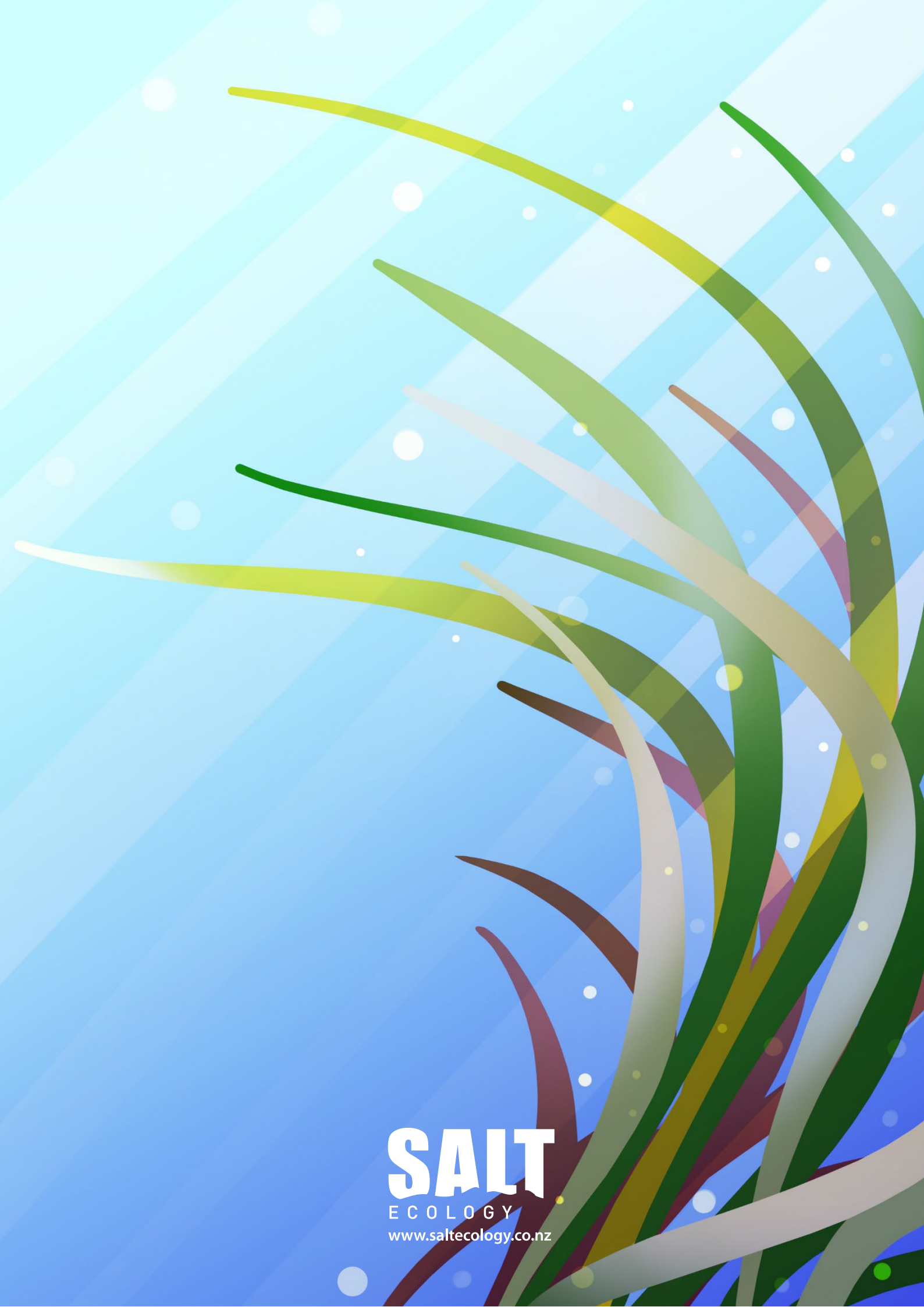
Hewitt J, McBride G, Pridmore R, Thrush S 1993. Patchy distributions: Optimising sample size. Environmental monitoring and assessment 27: 95-105.

Hewitt JE 2021. Monitoring estuaries in a changing world: Lessons for designing long-term monitoring programmes. Sustainable Seas, National Science Challenges, July 2021. 16p.

Hsieh TC, Ma KH, Chao A 2020. iNEXT: iNterpolation and EXTrapolation for species diversity. R package version 2.0.20 in R. http://chao.stat.nthu.edu.tw/wordpress/software-download/).

R Core Team 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Robertson B, Gillespie P, Asher R, Frisk S, Keeley N, Hopkins G, Thompson S, Tuckey B 2002. Estuarine environmental assessment and monitoring: a national protocol part B: development of the monitoring protocol for New Zealand Estuaries. Appendices to the introduction, rationale and methodology. Sustainable Management Fund Contract No. 5096, Cawthron Institute, Nelson, New Zealand. 159p.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