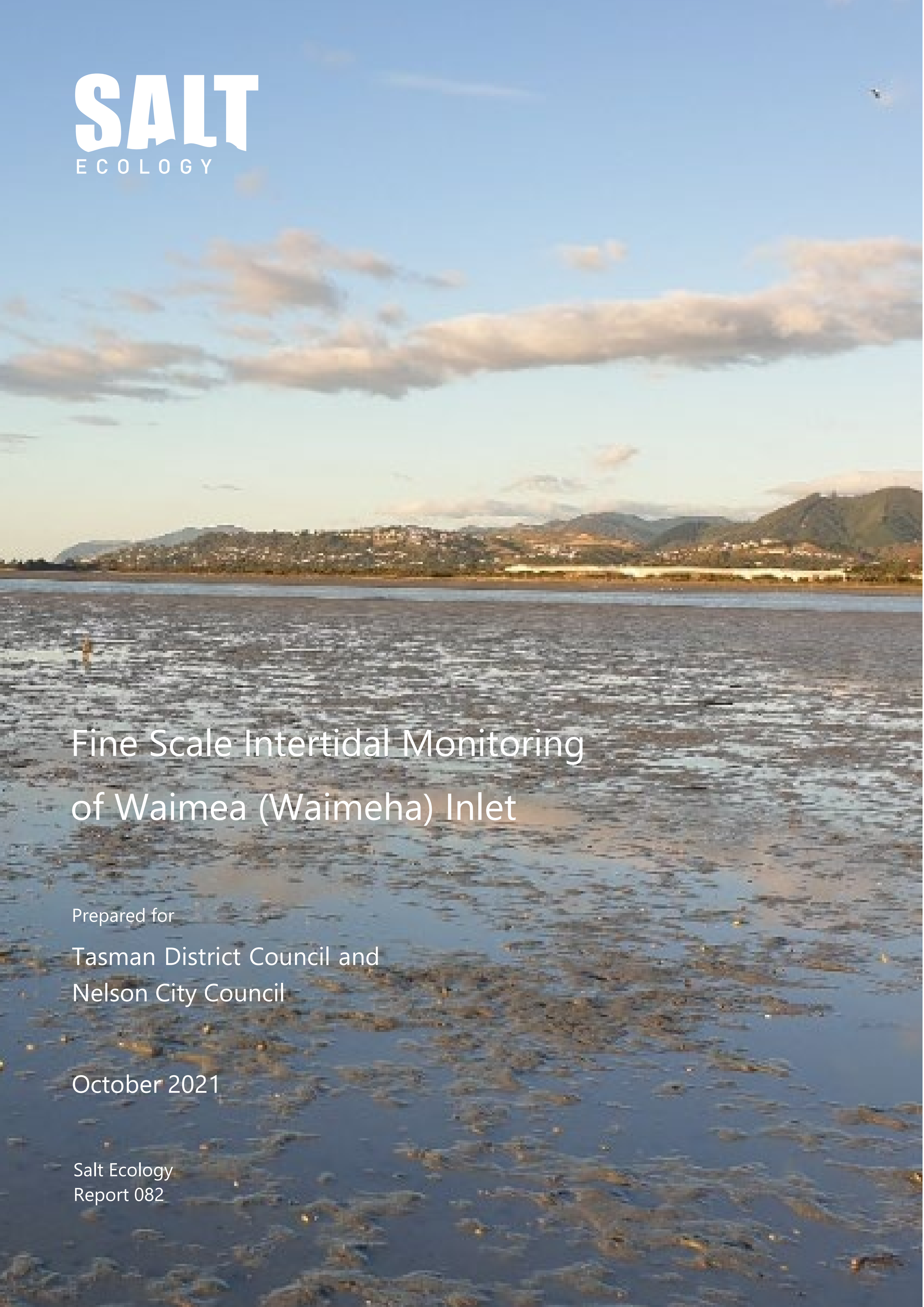

SALT
ECOLOGY

Fine Scale Intertidal Monitoring of Waimea (Waimeha) Inlet

Prepared for

Tasman District Council and Nelson City Council

October 2021

Salt Ecology
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SALT ECOLOGY

For the People
Mō ngā tāngata

Fine Scale Intertidal Monitoring of Waimea (Waimeha) Inlet

Prepared by

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for

Tasman District Council

and

Nelson City Council

October 2021

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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)
aRPD	Apparent redox potential discontinuity
As	Arsenic
Cd	Cadmium
Cr	Chromium
Cu	Copper
DGV	Default Guideline Value
ETI	Estuary Trophic Index
Hg	Mercury
NCC	Nelson City Council
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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SUMMARY

BACKGROUND

As part of its State of the Environment programme, Tasman District Council and Nelson City Council monitor the ecological condition of significant estuaries in their regions. This report describes ecological monitoring and sedimentation surveys conducted in Waimea (Waimeha) Inlet since 2001. The surveys largely follow the 'fine scale' approach described in New Zealand's National Estuary Monitoring Protocol (NEMP). Differences among monitoring sites, and temporal trends, are evaluated. Results are assessed against estuary 'health' rating criteria (see Table on following page), and discussed in the context of future monitoring, investigation and mitigation needs.

KEY FINDINGS

Sedimentation

- Sediment plate monitoring revealed variable erosion and accretion since the first baselines were established in 2008. Only three of the fourteen sedimentation monitoring sites were near fine scale monitoring sites, and these did not reveal patterns of excess sedimentation (i.e. exceeding the national guideline of 2mm/yr) that were attributable to muddy sediment inputs from the catchment.

Sediment quality

- Despite the low sedimentation, surface sediments (top 20mm), when last measured in 2016, four of the fine scale sites (Sites A-D in TDC's region) had mud contents exceeding the biologically relevant threshold of 25%, and were rated as 'poor' against estuary health criteria. These sites have exhibited a marked increase in mud content since 2001. Site E (established in 2019 in NCC's region) is sandy, with a mud content of ~10%.
- Sediments were generally unenriched, with low nutrient and total organic carbon levels, and no symptoms of strong anoxia (i.e. black sediment with a sulphide odour). The mud-dominated sediments of Site D were, in 2015 and 2016, covered in moderate growths of the opportunistic macroalga *Agarophyton chilense*. This species can thrive in nutrient-enriched and/or muddy conditions, and is present in localised hot spots around Waimea Inlet. However, estuary-wide modelled nutrient loads are considerably less that the threshold above which widespread nuisance growths are predicted to occur.
- Trace contaminant concentrations were very low relative to national sediment quality guideline values, except for nickel and chromium, which were elevated due to natural catchment sources.

Macrofauna

- Compared with other estuaries in the top of the South Island, four of the five fine scale sites had a relatively high macrofaunal richness and abundance, being second only to Nelson Haven. Among the dominant species are cockles, wedge shells, and other important prey species for birds, fish and rays.
- Relative to other New Zealand estuaries, a recently developed National Benthic Health Model (BHM) rated the impact from mud on Waimea Inlet macrofauna as 'moderate' to 'high'. The impact of trace metals (copper, lead, zinc) was rated by the BHM as 'good' or 'fair' when scaled against sediment quality thresholds that are more conservative than national guideline values.

Notwithstanding the BHM findings, the overall impression provided by the suite of indicators and their associated health ratings is that the main tidal flats of Waimea Inlet are in a reasonably healthy state ecologically. This situation has persisted despite the considerable historic modification of estuary margins, loss of salt marsh habitat, and land development in the catchment.

However, the present-day sediment load is high compared with the estimated natural state, and the elevated and steady increase in sediment mud content at Sites A-D over 15 years (2001-2016) appears to be indicative of an incremental degradation of habitat quality. In the event that this degradation has been ongoing since the last survey of these sites in 2016, there is a risk that the sediments will exceed the mud tolerance of key species, whose populations will eventually decline. Such an outcome could have flow on effects to the wider ecosystem.

Recent studies have highlighted agricultural land uses and exotic forest harvest as being key historic or potential contributors of sediment to the Waimea Inlet. It is important that these and other potential sources are managed so that the current state of the estuary is maintained or improved.

RECOMMENDATIONS

On the basis of the above findings, and the discussion in the report regarding further investigation and monitoring needs (see Section 5.2), the following is recommended for consideration by TDC and NCC:

1. Desktop assessment of catchment stressor sources and inputs based on: (i) assessment of land uses, including exotic forest harvest patterns; (ii) modelled mass loads of nutrients and sediments; and (iii) appraisal of freshwater (including stormwater) inputs and associated water quality data. A particular focus should be on links between catchment land use and muddy sediment inputs.
2. Investigation of estuary condition in the vicinity of point source inputs and/or where local issues have already been identified, including sampling key NEMP indicators and a suite of priority pollutants. The scope would be better determined after completion of the assessment in #1.
3. Undertake annual or biennial monitoring of sedimentation, sediment composition and enrichment status at representative sites. Also undertake another fine scale survey (reduced in terms of sites and sampling effort) when budgets allow, to evaluate ongoing degradation (due to muddy sediment inputs) since TDC Sites A-D were last sampled in 2016.
4. Depending on the outcomes of the above, the potential for implementation of mitigation strategies to reduce future impacts should be considered.

Summary of indicators against estuary condition criteria

Site	Year	Mud %	TOC %	TN mg/kg	TP mg/kg	aRPD mm	As mg/kg	Cd mg/kg	Cr mg/kg	Cu mg/kg	Hg mg/kg	Ni mg/kg	Pb mg/kg	Zn mg/kg	AMBI (0-7)
A	2001	31.9	0.57	633	441	30	-	0.100	69.3	10.3	-	65.1	4.2	44.2	2.8
	2006	33.8	0.75	468	458	20	-	< 0.100	48.6	7.9	-	64.8	6.4	34.7	3.0
	2011	42.5	1.00	380	480	10	6.0	< 0.090	55.0	9.3	< 0.05	70.0	7.9	39.0	3.0
	2014	42.7	0.54	700	437	15	5.2	0.025	51.7	9.8	0.03	74.0	7.4	40.0	2.9
	2015	37.4	0.35	633	463	10	-	-	-	-	-	-	-	-	2.5
	2016	44.4	0.49	700	463	10	-	-	-	-	-	-	-	-	3.1
B	2001	15.9	0.43	279	480	30	-	0.100	44.6	8.8	-	72.3	6.3	38.4	2.5
	2006	19.9	0.58	353	516	20	-	< 0.100	32.0	6.7	-	69.4	5.1	27.9	2.6
	2014	25.2	0.38	333*	493	20	5.6	0.016	31.7	7.4	0.02	75.3	5.6	32.0	2.7
C	2001	9.6	0.31	329	273	30	-	0.375	61.3	7.0	-	58.3	7.7	34.5	1.9
	2006	21.6	0.84	550	376	20	-	< 0.100	42.3	7.8	-	60.6	5.9	28.2	2.4
	2011	25.5	0.75	387	400	10	5.0	< 0.100	49.0	8.9	< 0.05	64.0	6.3	33.0	2.5
	2014	26.6	0.54	733	370	10	5.4	0.022	51.0	9.3	0.02	72.7	6.8	37.7	2.6
	2015	26.5	0.42	800	410	10	-	-	-	-	-	-	-	-	2.4
	2016	37.6	0.51	700	397	20	-	-	-	-	-	-	-	-	2.5
D	2001	40.5	0.85	783	539	30	-	0.475	95.2	12.3	-	94.2	11.3	50.2	2.2
	2006	33.4	0.89	487	509	20	-	< 0.100	55.1	9.4	-	89.2	6.4	34.5	2.4
	2014	50.1	0.62	700	530	10	6.3	0.026	58.3	10.4	0.02	95.3	7.0	41.0	2.1
	2015	62.6	0.67	967	637	5	-	-	-	-	-	-	-	-	2.0
	2016	66.8	0.98	1033	577	10	-	-	-	-	-	-	-	-	2.1
E	2019	10.1	0.17	< 500	310	30	4.5	0.014	33.0	5.4	< 0.02	49.3	5.2	30.7	1.6
	2020	9.9	0.18	< 500	280	29	4.2	0.013	32.3	5.6	< 0.02	58.3	5.2	33.7	1.2
	2021	10.3	0.18	< 500	303	24	4.7	0.014	35.7	6.0	< 0.02	58.0	5.2	33.7	2.0

** Sample mean includes values below lab detection limits*

< All values below lab detection limit

Condition rating key:

Very Good	Good	Fair	Poor

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. 2002b). 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 programmes of Tasman District Council (TDC) and Nelson City Council (NCC) have included NEMP broad scale and fine scale surveys in estuaries across the council regions. One of these estuaries is Waimea (Waimeha) Inlet (Fig. 1), which crosses the jurisdictional boundary of both TDC and NCC. It was one of the national estuaries sampled by Cawthron Institute in 2001 as part of the original development of the NEMP approach (Robertson et al. 2002a).

Since the 2001 baseline, repeat NEMP broad scale surveys have been undertaken on three occasions (Clark et al. 2008; Stevens & Robertson 2014; Stevens et al. 2020b). Similarly, repeat NEMP fine scale surveys have been conducted on four occasions in TDC's part of the estuary; in 2006 by Cawthron (Gillespie et al. 2007), then across consecutive years in 2014 (Robertson & Robertson 2014), 2015 and 2016 by Wriggle Coastal Management. Subsequently, three annual NEMP surveys (2019, 2020, 2021) were undertaken for NCC by Salt Ecology at a new site in the eastern estuary. In addition to the broad and fine scale surveys, in 2008 TDC commenced sediment plate monitoring with a total of 14 sites now established in the estuary. Sediment plate monitoring at the NCC site began in 2019.

Previous reports have summarised the above survey work up to and including the 2014 survey, with the data from the five fine scale surveys conducted since 2014 having been archived. In addition, although the sediment plate records were summarised in an appendix to the 2020 broad scale report, these data have not been extensively analysed nor subject to QA checks. Accordingly, to provide an understanding of long-term changes, 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 monitoring indicators. The NEMP data are supplemented with information from a subset of sites that were sampled in 2011 as part of a Bell Island regional sewerage discharge assessment (Gillespie et al. 2012). Collective findings are discussed in terms of estuary condition, and considered within the context of several historic studies that have been undertaken in Waimea Inlet. The management implications for Waimea Inlet are considered, as well as needs for ongoing monitoring and further investigation.

2. BACKGROUND TO WAIMEA INLET

Background information on Waimea Inlet was detailed in the 2020 broad scale survey report (Stevens et al. 2020b) and is recapped below along within the key findings from that work.

Waimea Inlet is one of the largest intertidally-dominated estuaries in the South Island. It covers an area of ~3462ha, and is defined as a well-flushed, shallow, intertidal-dominated lagoon type estuary (SIDE). The estuary comprises two main intertidal basins, each with side arms and embayments (some separated by causeways), and several islands. It discharges to Tasman Bay via two tidal entrances at either end of Rabbit Island. Residence time in the estuary is less than one day; most of the intertidal flat is perched high in the tidal range, meaning that the estuary almost completely drains at low tide leaving the intertidal area exposed for much of the tidal cycle.

The estuary has high human use and high ecological and cultural values. A comprehensive overview of estuary ecology is contained in Davidson and Moffat (1990). Waimea Inlet is recognised as an important nursery area for marine and freshwater fish, has extensive shellfish resources, and is considered to be of international importance for shorebirds (Schuckard & Melville 2019). While dominated by intertidal sand and mudflats, the well-flushed and often steeply incised estuary channels are deep and, particularly near the entrances, support a variety of cobble, gravel, sand, and biogenic (oyster, tubeworm, sponge garden) habitats (Asher et al. 2008; Stevens et al. 2020b).

Catchment land use (Table 1, Fig. 2) is dominated by indigenous and exotic forestry, and pasture. Catchment geology includes the Dun Mountain "mineral belt" region, which contains rock formations particularly high in metals such as nickel, chromium and copper (Robinson et al. 1996; Rattenbury et al. 1998). Freshwater flows from the catchment arise from Waimea River and a number of small streams. The Waimea River is the main freshwater inflow to the estuary; however, at least nine small streams also contribute, with the potential for localised impacts in the estuary (Gillespie et al. 2001). Monthly water quality monitoring is conducted in the Waimea River, with the nearest site being SH60 at Appleby, which is ~2.5km upstream. Results from water clarity monitoring (as an indicator of suspended fine sediment) place the river water in the best 25% of monitoring sites nationally (5-yr median clarity >6m; see: https://www.lawa.org.nz/explore-data/tasman-region/river-quality/waimea-river/waimea-at-sh60-appleby/. Concentrations of the nutrient total nitrogen (which can contribute to excess algal growth in estuaries) place the river water in the best 25-50% of monitoring sites. Point source anthropogenic contaminant inputs include treated sewage from Bell

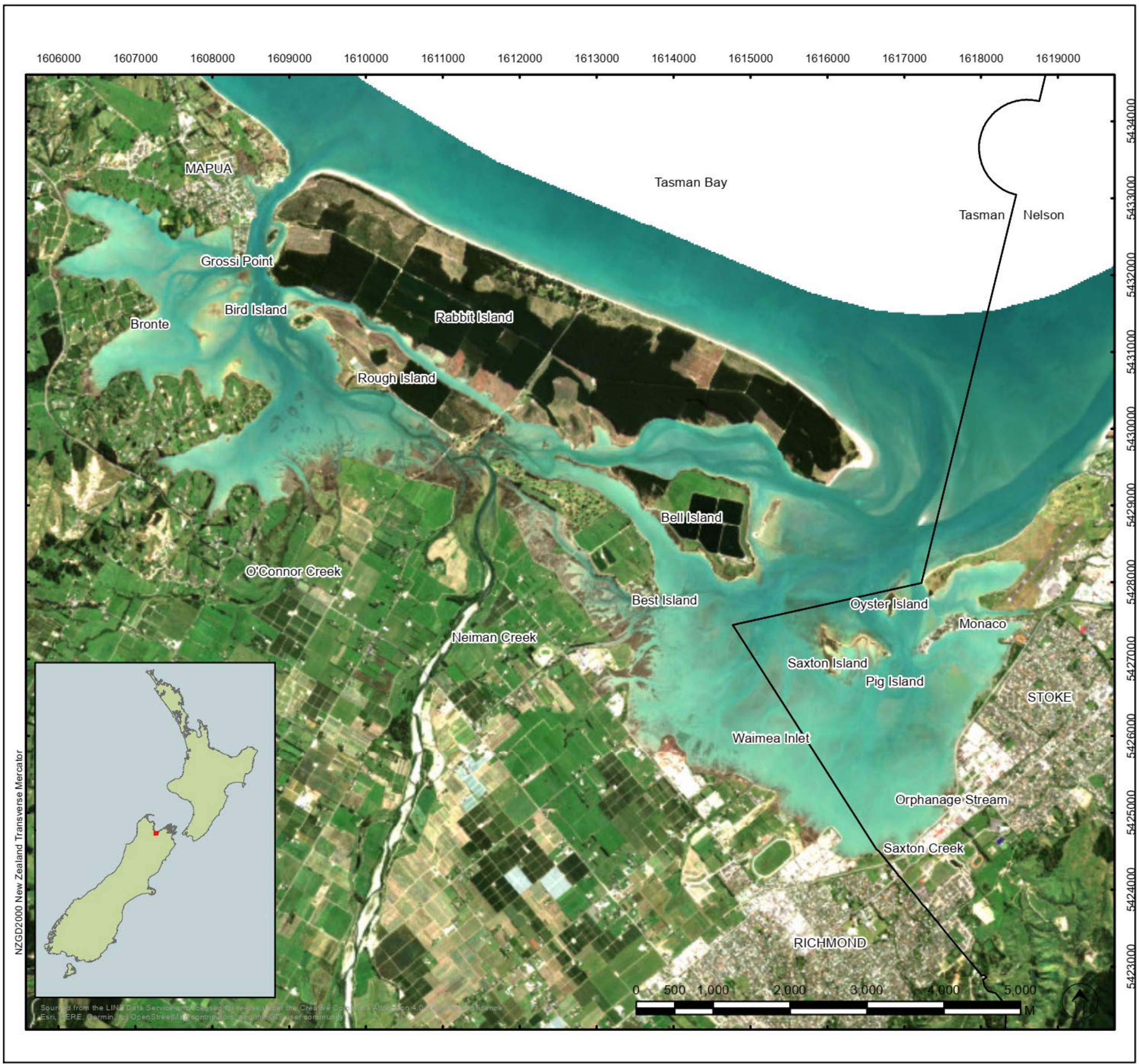


Fig. 1. Location of Waimea Inlet and places names referred to in text. Map sourced from Stevens et al. 2020)

Island and various stormwater inputs (Robertson et al. 2002a; Dunmore 2016). Parts of Rabbit and Bell islands are used for the land disposal of sewage sludge from the Bell Island oxidation ponds.

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

LCDB5 (2018) Class and Name		Ha	%
1	Built-up Area (settlement)	2,356.7	2.5
2	Urban Parkland/Open Space	602.6	0.6
5	Transport Infrastructure	115.1	0.1
6	Surface Mine or Dump	77.3	0.1
10	Sand or Gravel	28.3	0.03
15	Alpine Grass/Herbfield	396.9	0.4
16	Gravel or Rock	592.7	0.6
20	Lake or Pond	112.1	0.1
21	River	15.8	0.02
22	Estuarine Open Water	133.5	0.1
30	Short-rotation Cropland	888.1	0.9
33	Orchard, Vineyard Other Perennial Crop	2,689.9	2.8
40	High Producing Exotic Grassland	18,357.0	19.4
41	Low Producing Grassland	501.1	0.5
43	Tall Tussock Grassland	1,934.1	2
45	Herbaceous Freshwater Vegetation	6.2	0.01
46	Herbaceous Saline Vegetation	91.7	0.1
50	Fernland	67.1	0.1
51	Gorse and/or Broom	959.6	1
52	Manuka and/or Kanuka	2,769.7	2.9
54	Broadleaved Indigenous Hardwoods	2,171.9	2.3
55	Sub Alpine Shrubland	494.4	0.5
56	Mixed Exotic Shrubland	107.9	0.1
64	Forest - Harvested	4,681.5	5
68	Deciduous Hardwoods	198.6	0.2
69	Indigenous Forest	28,614.2	30.3
71	Exotic Forest	25,491.0	27
	Total	94,455	100

The recent broad scale report showed that the estuary has been extensively modified over many years, as follows:

- Much of the estuary margin is directly bordered by developed urban and rural land, roads, cycleways/walkways, causeways, and seawalls.
- Salt marsh covers ~10% of the intertidal, but is estimated to be <40% of its historic extent, with losses due to drainage, reclamation, margin development and channelisation.
- Historic seagrass cover is unknown, but a reduction of high-density beds of >60% is estimated to have occurred since the first records were made in 1990.
- Sediments are mud-dominated, especially in mid-upper estuary embayments. This contrasts the estimates made from deep sediment coring, which suggested the estuary was historically dominated by sand and shell/gravel, having little mud and plentiful populations of large shellfish (Stevens & Robertson 2011).
- The source of the mud-dominated sediment appears to be largely historical, with anecdotal reports of high inputs following the development of orchard land in the 1950's and 1960's. However, important ongoing sources include runoff from agricultural land and harvested exotic forest (Gibbs & Swales 2019).
- The estuary does not exhibit any widespread symptoms of excessive nutrient-enrichment (eutrophication). However, nuisance growths of the opportunistic red seaweed *Agarophyton chilense* were recorded from a few localised hotspots (~20ha) in 2020.

Seagrass near Saxton Creek

Soft mud in south east of the estuary

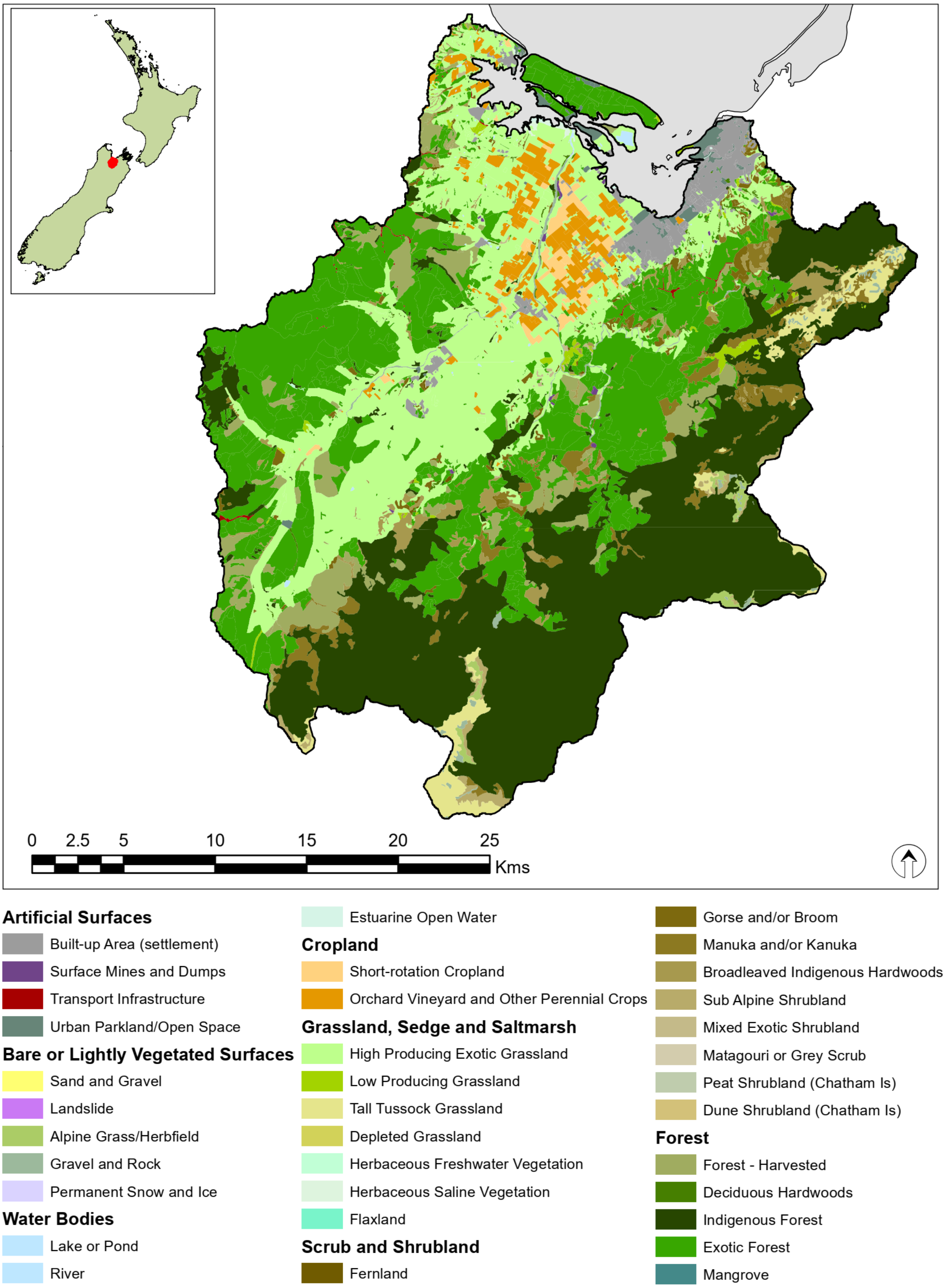


Fig. 2. Waimea Inlet and surrounding catchment land use classifications, LCDB5 2018 (source: Stevens et al. 2020).

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 WAIMEA FINE SCALE AND SEDIMENT PLATE SITES AND SAMPLING

The initial fine scale survey in March 2001 established four monitoring sites (A-D), with Site E added in April 2019. The fine scale sites were chosen to be representative of the dominant muddy-sand substrate within the estuary, with Sites B and C selected in firm substrate with <25% mud content and Sites A and D in soft substrate with >25% mud content. All sites are of the recommended NEMP dimensions of 30 x 60m. Fine scale site locations are shown in Fig. 3, along with the 14 locations where sediment plate monitoring has been conducted. Site boundaries and locations of sediment plates are marked with wooden pegs, with position data provided in Appendix 1. At Site E, sediment plates are along the boundary of the fine scale site itself.

Not all fine scale sites have been sampled in all years, and there is a timing offset between historic sampling in the TDC part of the estuary (2001-2016) and the most recent sampling in the NCC part (2019-2021). Table 2 summarises the sites sampled in different years since 2001, and also the sampling effort undertaken. Similarly, sediment plate sampling has been undertaken in a staged manner in the estuary, with the first plates installed at sites 1-8 in 2008, and additional plates installed in 2012 (sites 9 & 10), 2015 (sites 11 & 12) and 2019 (fine scale site E).

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.

Sediment plates consist of concrete pavers (typically 19cm x 23cm), with typically four plates installed at each of the 14 sites. Baseline depths (from the sediment surface to each buried plate) were measured by TDC staff at the time of plate installation, with most subsequent monitoring also conducted by TDC staff. To make measurements of sediment depth at each plate, a straight edge is typically placed over the plate position to average out any small-scale irregularities in surface topography. The depth to each plate is then measured in triplicate by vertically inserting a probe into the sediment until the plate was located.

Table 2. Summary of fine scale sampling years, effort and provider. Sample numbers (replicates) are shown for macrofauna and sediment, with sediment samples shown in brackets (3) for 2011 to 2021 indicating that 3 composite samples were collected.

Year	Date	A	B	C	D	E	Field	Sorting	Taxonomy	Sediment analysis
2001	March	12	12	12	12	-	Cawthron	Cawthron	Cawthron	Cawthron
2006	14-27 April	10	10	10	10	-	Cawthron	Cawthron	Cawthron	Cawthron
2011	Sept	4 (3)	-	4 (3)	-	-	Cawthron	Cawthron	Cawthron	Cawthron
2014	11 March	10 (3)	10 (3)	10 (3)	10 (3)	-	Wriggle	CMEC	CMEC	RJ Hill
2015	4 Feb	10 (3)	-	10 (3)	10 (3)	-	Wriggle	Wriggle	CMEC	RJ Hill
2016	17 March	10 (3)	-	10 (3)	10 (3)	-	Wriggle	Wriggle	CMEC	RJ Hill
2019	5 April	-	-	-	-	10 (3)	Salt	Salt	CMEC	RJ Hill
2020	5 March	-	-	-	-	10 (3)	Salt	Salt	CMEC	RJ Hill
2021	25 Jan	-	-	-	-	10 (3)	Salt	Salt	CMEC	RJ Hill

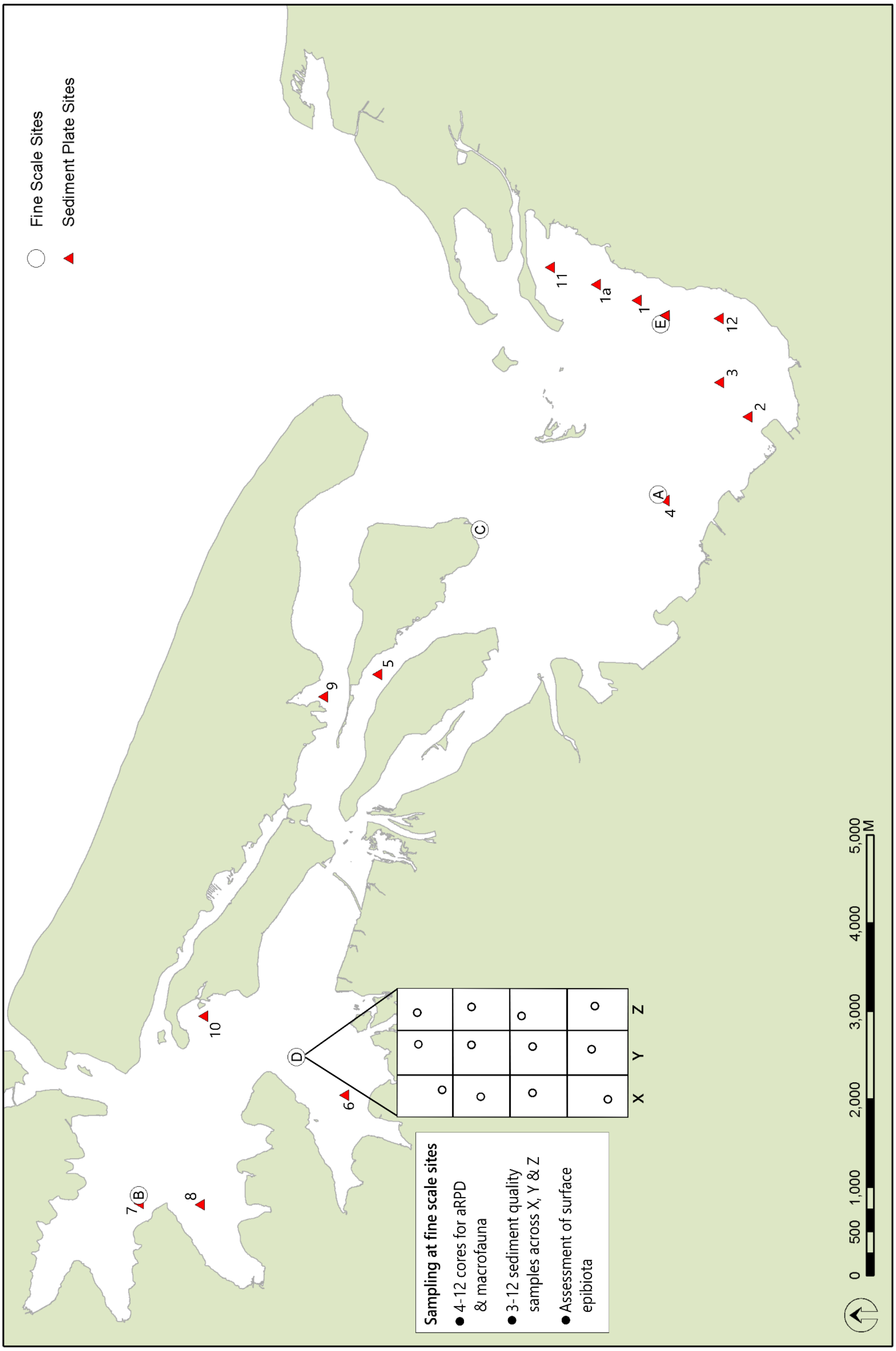


Fig. 3. Location of fine scale (A-E) and sediment plate monitoring sites. Schematic illustrates fine scale sampling scheme. Sampling effort has varied among surveys, but usually consists of one or more replicates taken randomly across each of three vertical columns (represented by X-Z) in the grid.

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

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 4-12 of these plots (see Table 2). 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). Note that although the 2011 survey generally followed the NEMP approach, sampling was undertaken in only 4 plots at Sites A and C.

A summary of the benthic indicators, the rationale for their inclusion, and the field sampling methods, is provided in Table 3. Although the sampling approach across all years has generally adhered to the NEMP, a recent review (Forrest & Stevens 2019) highlighted alterations and additions to early NEMP methods that have been introduced in most surveys conducted over the last 10 or more years. These modifications are reflected in the surveys conducted since 2014, as indicated in Table 3.

Sampling of sediments in earlier surveys (2001, 2007) consisted of discrete samples collected within each plot, but in later surveys involved collection of three composite sediment samples (each ~250g) that were pooled from sub-samples (to 20mm depth) collected across each of zones X, Y and Z (corresponding to replicates 1-3, 4-6 and 7-10, respectively; see Fig. 3). Samples have been analysed by either Cawthron (2001-2011) or RJ Hill Laboratories (2014-2021), and included most of 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); nutrients (total nitrogen, TN; total phosphorus, TP); and trace elements consisting of trace metals (cadmium, Cd; chromium, Cr; copper, Cu; mercury, Hg; lead, Pb; nickel, Ni; zinc, Zn) and the metalloid arsenic (As). 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 over 2001-2011, hence was estimated from %AFDW as: TOC = $(0.4 * \text{AFDW}) + 0.0025 * \text{AFDW}^2$.

Sediment oxygenation was assessed according to the depth of the apparent redox potential discontinuity (aRPD) (Table 3). The aRPD provides a subjective measure of the enrichment state of sediments according to the depth of 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 (sample effort as per Table 2) and gently washed through a 0.5mm sieve bag to remove fine sediment. The retained animals (i.e. macrofauna) were preserved in a dilution of either formalin or isopropyl alcohol. The macrofauna in each sample 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, epibiota (macroalgae and conspicuous surface-dwelling animals nominally >5mm body size) visible on the sediment surface at each site have been semi-quantitatively categorised since 2014 using 'SACFOR' abundance (animals) or percentage cover (macroalgae) ratings as shown in Table 4. 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 2014 as an alternative to the quantitative quadrat sampling specified in NEMP (used in 2001 & 2006), which is known to poorly characterise scarce or clumped species. For comparative purposes the quadrat data from the 2001-2011 surveys were expressed as SACFOR ratings.

Table 3. Summary of NEMP fine scale benthic indicators, rationale for their use, and sampling method. These methods were adopted by surveys conducted since 2014, with any meaningful departures from early surveys or the NEMP protocol described in footnotes.

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 of 4-12 plots (see Table 2), 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 4-12 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 (Table 4) 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 (Table 4) 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 and/or visual assessment of conspicuous growths (see note 4).

[1] For reasons of cost and low sample variance, since 2011 sediment quality has been assessed in 3 composite samples rather than 10 discrete samples as specified in the NEMP or 12 samples as in the 2001 survey.

[2] Arsenic and mercury, not originally included in the NEMP because of cost constraints, have been subsequently included as part of a standard RJ Hill trace element suite in later years (see Table 2).

[3] Assessment of epifauna, macroalgae has used SACFOR since 2014, 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.

Table 4. SACFOR ratings for assessing 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

Note: SACFOR epibiota assessment conducted since 2014 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).

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 2001. 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 among surveys, 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 2014 were made, based on limited retrospective taxonomic verification undertaken by NIWA on reference samples provided by CMEC (Appendix 3a).

The QA process could not be applied to the Cawthon samples collected over 2001 to 2011. 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) could be 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 for TDC and NCC (e.g. Forrest & Stevens 2021).

Collecting sediment macrofauna cores at Site E (top), and example of sieving (bottom)

Multivariate analysis of the macrofaunal community data were undertaken using methods detailed in previous reports 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 details 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 and Stevens 2021).
    - Calculation of Benthic Health Model (BHM) scores in relation to sediment mud and metals (copper, lead, zinc), 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 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 recently developed by Cawthron Institute (unpubl.), 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).

Table 5. ETI condition ratings used to characterise Waimea Inlet health for key indicators. See footnotes and other Salt Ecology reports (e.g. Forrest and 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.

4. KEY FINDINGS

4.1 GENERAL FEATURES OF FINE SCALE SITES

The sites are typical of the main intertidal fine sediment habitats present in Waimea Inlet ranging from relatively sandy (Site E) to mud-dominated (Site D). The photos on the following page illustrate the superficially uniform and relatively barren tidal flats. Shell hash across the sediment surface is present but not particularly conspicuous. 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.

4.2 SEDIMENT PLATES

Sediment plate raw data have been provided to NCC and TDC in electronic form. The summary Figure and Table in Appendix 4 reveal highly variable sediment accrual and erosion patterns across the sites. In terms of the present report, the sediment plate sites of most interest are those next to fine scale survey sites as follows:

- Plate Sites 4 and 7, next to fine scale Sites A and B, respectively: There has been no appreciable sedimentation at these sites.
- Fine scale E: Shows a net deposition of 4.4mm/yr which exceeds the guideline value for New Zealand estuaries of 2mm/yr. However, the accrual is attributable to the movement of sand as opposed to the deposition of catchment-derived muddy sediment.

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 were sandiest at Site E (~10% mud) and muddiest at Site D (~40-67% mud). Illustrative photos of the sediments from Site E are provided in Fig. 5. Of most interest is that sediment mud content has increased relatively steadily at Sites A-D over the 2001-2016 period.

To provide a visual comparison of sediment quality relative to the Table 5 condition ratings, Fig. 6 compares the mean percentage mud, total organic carbon (TOC) and total nitrogen (TN) from fine scale sites against the rating thresholds. Due to mud content exceeding 25%, sediments were consistently rated as 'poor' in all years at Site A and D. Based on the most recent sampling year Sites B and C were also in the 'poor' category. Site E is at the boundary of 'good' and 'fair' in the surveys conducted over 2019-2021, reflecting its relatively low sediment mud content (Fig. 6).

Fine scale Site A

Fine scale Site B

Fine scale Site C

Fine scale Site D

Fine scale Site E

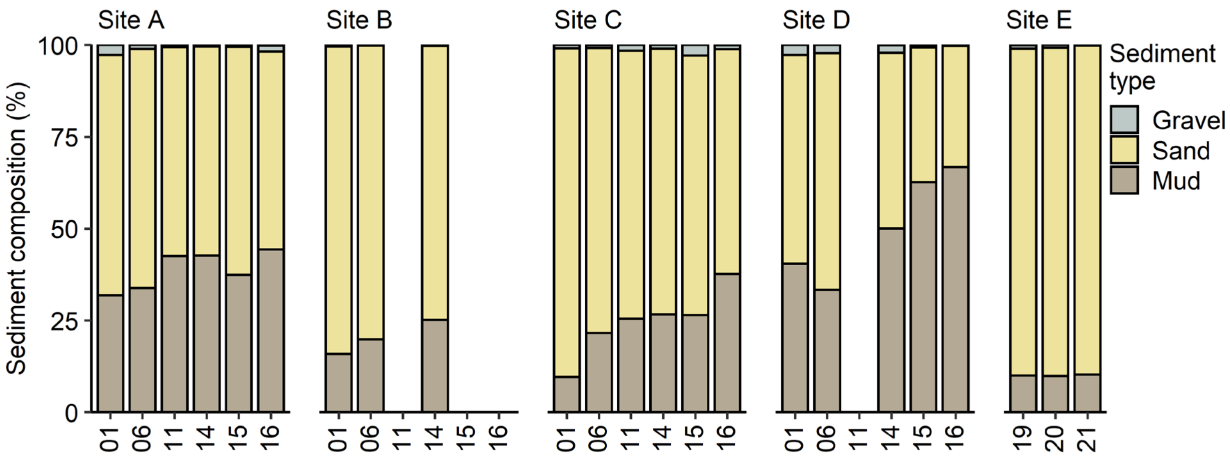


Fig. 4. Sediment particle grain size analysis showing percentage composition of mud (<63μm), sand (<2mm to ≥63μm) and gravel (≥2mm). See sampling replication in Table 2.

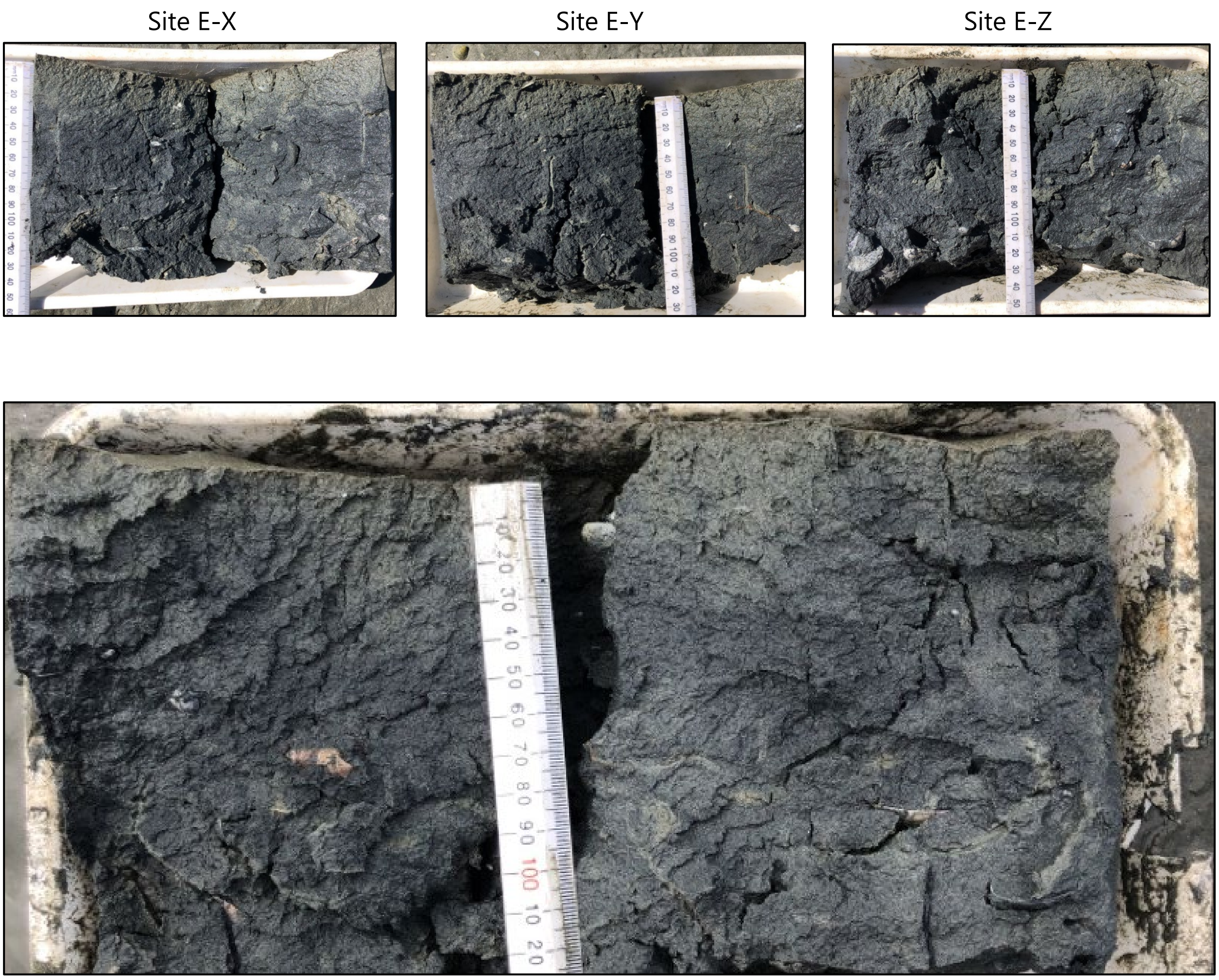


Fig. 5. Top: example sediment cores from Site E in 2020; Bottom: mixing of oxygenated and oxygen-depleted sediment in cores due to bioturbation, and enhanced oxygen penetration due to porous sandy sediment, can make the aRPD indistinct.

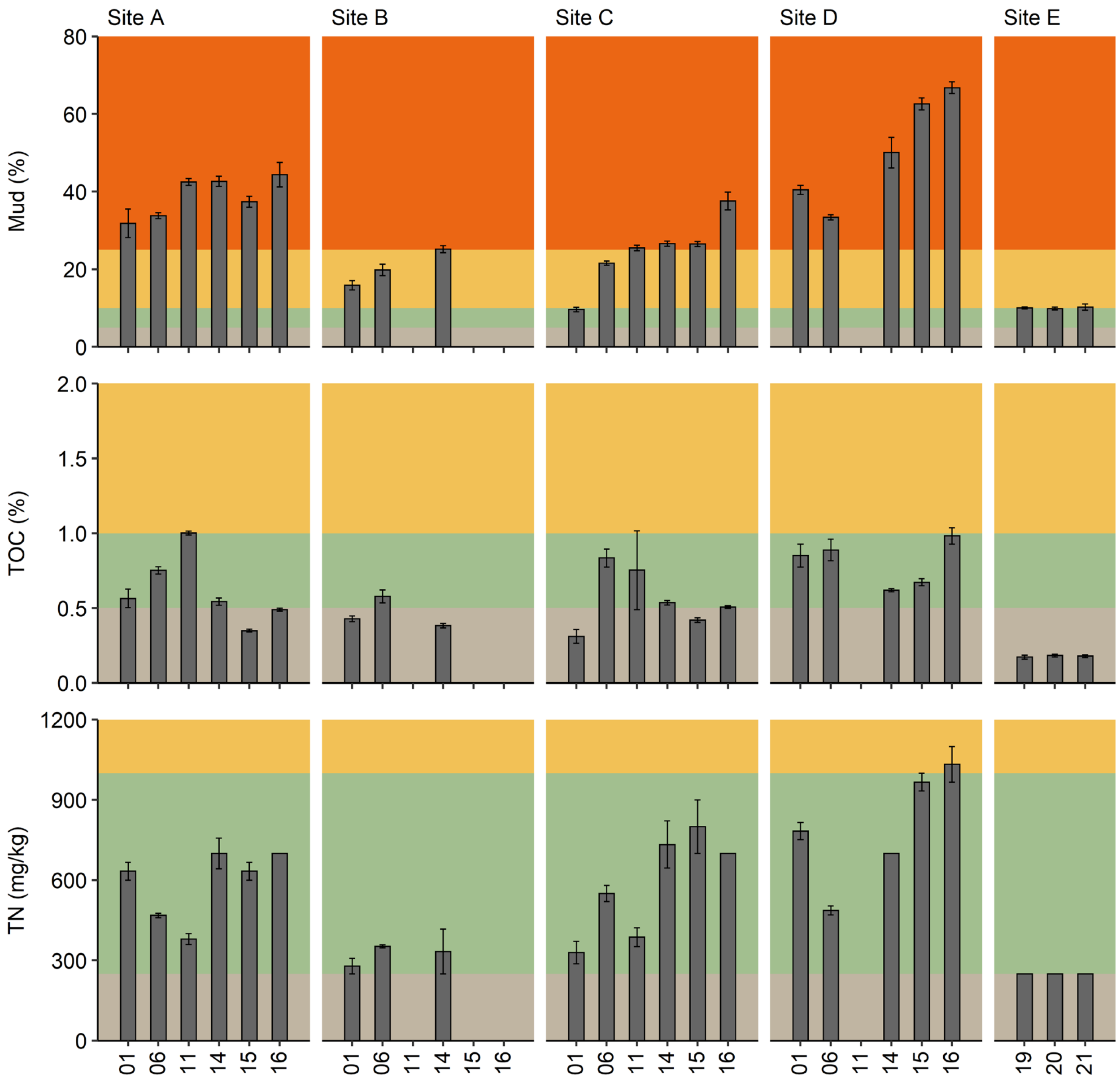


Fig. 6. Mean (±SE, n=3-12) sediment %mud, total organic carbon (TOC), and total nitrogen (TN) relative to condition ratings. Note that TOC in 2001-2011 was estimated from ash-free dry weight data.

Condition rating key:

Very Good	Good	Fair	Poor

Total organic carbon (TOC) and total nitrogen (TN) values were, in almost all instances, rated 'good' or 'very good', with lowest values at Site E reflecting the sandy nature of the sediments there. The man exception was the most recent (i.e. 2016) survey at Site D, where TOC and TN values were elevated into the 'fair' rating category, reflecting the increase in sediment mud content. It is difficult to discern any strong temporal trends from the data, in part reflecting that TOC values from 2001 to 2011 are an approximation based on conversion from sediment ash-free dry weight values (see Methods).

4.3.2 Redox status

No signs of excessive sediment enrichment were evident. The sandy to sandy-mud sediments at the fine scale sites appear sufficiently porous to enable water penetration into the sediment matrix, maintaining reasonably well-oxygenated conditions. There is an apparent trend for aRPD to become shallower at most sites over time, which in consistent with the increased sediment mud content providing a barrier to oxygenation (Fig. 7). For example, mean baseline values of aRPD were estimated at ~20-30mm in 2001 and 2006, but at Sites A, C and D the more recent values are

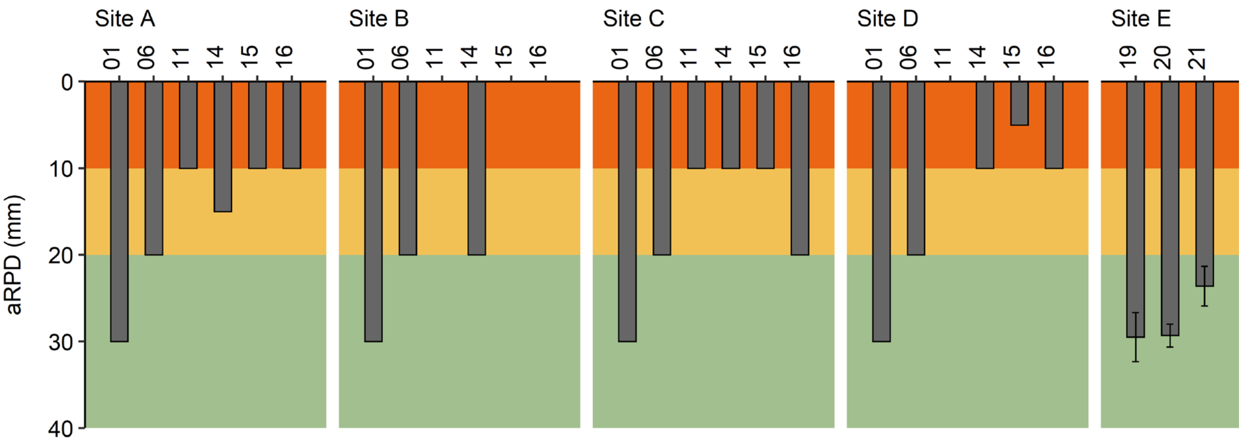


Fig. 7. Mean (±SE, n=3) aRPD relative to condition ratings. Values for 2001 to 2011 reported in Robertson and Roberston (2014) are assumed to have been estimated from photos in Gillespie et al. (2012). The absence of error bars for all values over 2001 to 2016 indicates that these are rough estimates of aRPD.

Condition rating key:

Very Good	Good	Fair	Poor

10-20mm. Some of this shallowing may reflect the subjective nature of the estimates rather than providing evidence for reduced sediment oxygenation. There is considerable judgement in assessing an exact depth for the aRPD, as it can be indistinct or show a gradual gradation of colour change. This lack of a distinct layering can reflect mixing within the sediment profile due to processes such as bioturbation (i.e. sediment turnover by macrofauna) (see photos in Fig. 5). Hence, while measurements are carried out by experienced field staff it should be acknowledged that there is inherent subjectivity in the aRPD assessment, thus some variability due to interpretation can be expected. However, the approach aims to assess gross meaningful shifts in aRPD, which indicate changes in sediment condition. Importantly, none of the sites showed evidence of black anoxic (and sulphide-smelling) sediments at (or within a few millimetres of) the sediment surface (nor high TOC or TN values) as would occur under strongly enriched conditions.

4.3.3 Trace contaminants

Plots of trace contaminants in relation to condition ratings are provided in Fig. 8 (see also Appendix 5). The main impression from Fig. 8 is that, with the exception of nickel (Ni) and chromium (Cr), 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.

By contrast, mean chromium concentrations were rated as 'fair' on one occasion (2001), whereas nickel concentrations were mainly rated as 'poor' as they exceeded GV-high values. These results will reflect natural inputs from the catchment; as noted in Section 2, the catchment has rock formations that are naturally high in nickel and chromium. The ecological implications are discussed in subsequent sections. Overall, despite the urbanised catchment around Waimea Inlet, there is no evidence of estuary-wide contamination from anthropogenic sources.

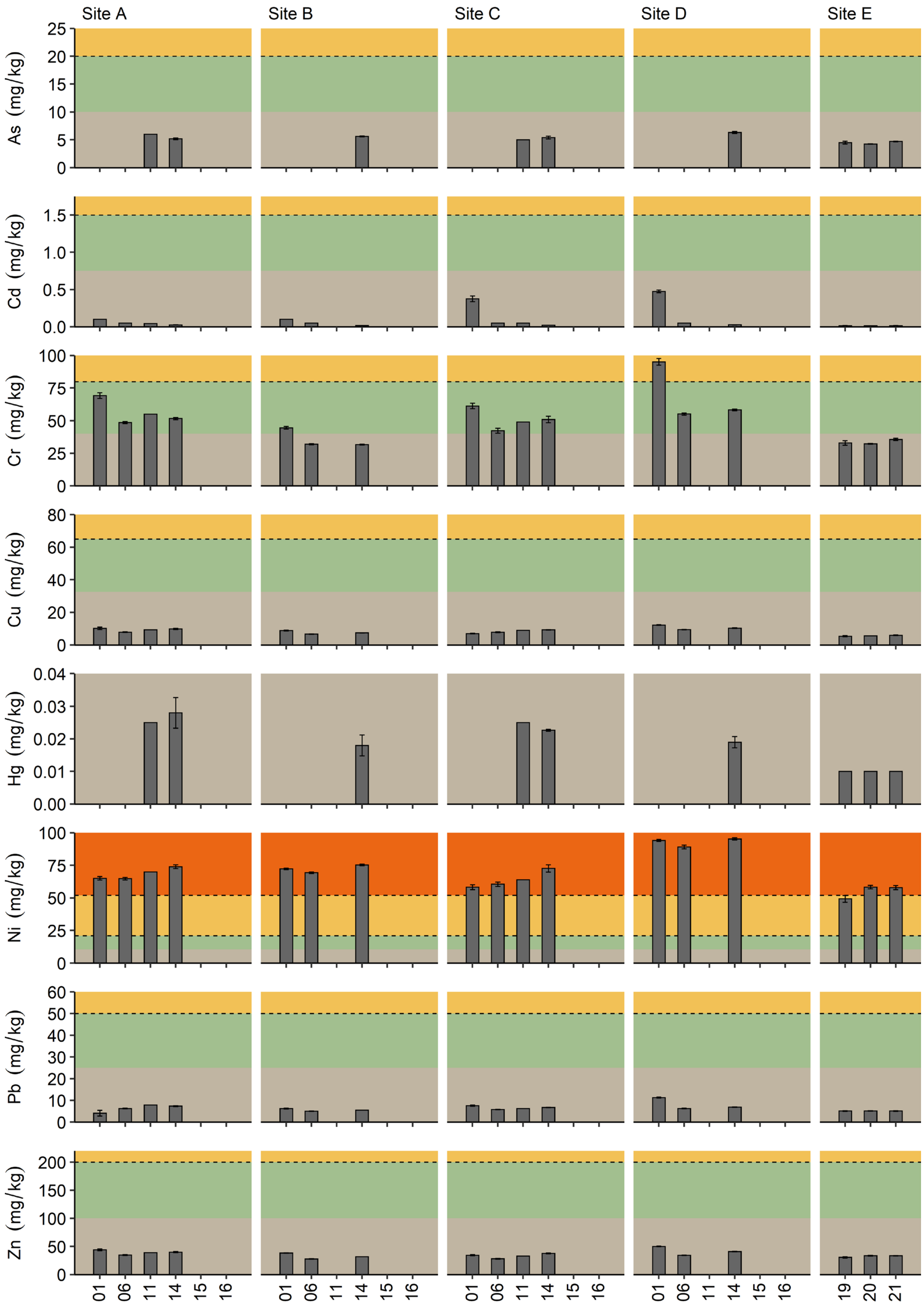


Fig. 8. Mean (±SE, n=3-12) trace element concentrations relative to condition ratings (rating key as per Fig. 7). Dotted line indicates national DGV for sediment quality.

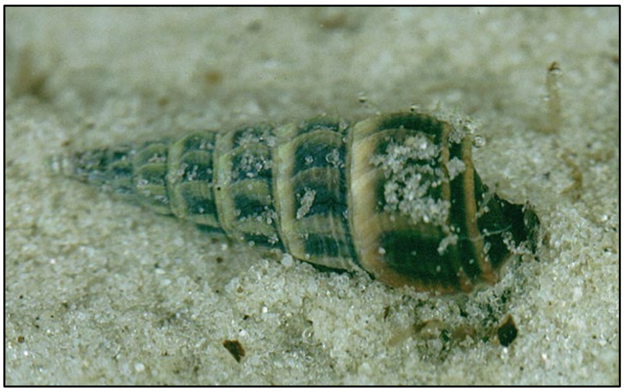
4.4 MACROFAUNA

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 four estuarine snail species and two species of common macroalgae; green 'sea lettuce' *Ulva* spp. and the red seaweed *Agarophyton chilense*. These two macroalgae were either absent or at a cover classed as rare (R, <1% cover) or occasional (O, 1-4% cover), except for Site D where the *Agarophyton* cover ranged from common (C, 10-19% cover) to abundant (A, 20-50%). The extensive *Agarophyton* coverage in 2016 equated to an estimated 2kg/m^2 of wet weight.

The most widespread and commonly occurring animals were the horn snail *Zeacumantus lutulentus*, and the mudflat topshell *Diloma subrostratum* (see photos). Both of these species had a higher abundance at Site D, where they were scored as common (C, 10-99/m^2) in two of the surveys. The mud whelk *Cominella glandiformis* was widespread but generally less abundant, and the mud snail *Amphibola crenata* was mainly associated with the relatively muddy conditions at Site D. 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 greatly 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.



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 4. Dash = absent. For 2012 data, SACFOR ratings were scaled from quadrat counts.

Species	Common name	Functional description	A 2006	A 2014	A 2015	A 2016	B 2006	B 2014	C 2006	C 2014	C 2015	C 2016	D 2006	D 2014	D 2015	D 2016	E 2019	E 2020	E 2021
Agarophyton chilense [1]	Red seaweed	Primary producer	ns	R	O	F	ns	R	ns	-	O	O	ns	C	A	A	-	-	R
Amphibola crenata	Mud snail	Microalgal and detrital grazer	-	R	O	-	-	-	-	O	-	-	O	C	F	-	-	-	-
Cominella glandiformis	Mud whelk	Carnivore and scavenger	O	R	F	-	-	R	O	R	O	F	F	R	O	F	-	-	O
Diloma subrostrata	Mudflat topshell	Grazer and deposit feeder	O	O	F	F	O	O	-	-		F	F	O	C	F	C	C	F
Ulva spp.	Sea lettuce	Primary producer	ns	-	-	-	ns	-	ns	-	O	-	ns	O	-	-	-	-	-
Zeacumantus lutulentus	Horn snail	Microalgal and detrital grazer	O	O	-	F	F	F	C	O	F	C	F	O	O	F	C	C	C

[1] *Agarophyton chilense* is the revised name for *Gracilaria chilensis.* ns = not sampled

4.4.2 Macrofauna cores

Raw data for sediment-dwelling macrofauna are provided in Appendix 6.

Main taxonomic groups and species

The species recorded represented 18 main taxonomic groups. The most well-represented in terms of species richness were polychaete worms and bivalve shellfish, with gastropod snails also reasonably species-rich and abundant (Fig. 9). However, polychaetes and bivalves were by far the most dominant in terms of organism abundances.

Richness, abundance and AMBI

A total of 67 species or higher taxa of sediment dwelling macrofauna were sampled by Cawthron over 2001-2011, compared with 53 described by CMEC over 2014-2021. Table 7 and Table 8 describe the most commonly-occurring species that were recorded.

Mean species richness ranged from ~6 to 16 taxa per core sample (Fig. 10a), with abundances being highly variable across sites and years (Fig. 10b). Site B was the most species-poor and impoverished overall. At Sites A to D, survey years 2014-2016 were more species-poor than earlier surveys, with lower organism abundances. The result may reflect differences among the providers that did the sample processing (see Table 2), but more likely reflects true temporal differences, possibly in response to increasing mud content over time.

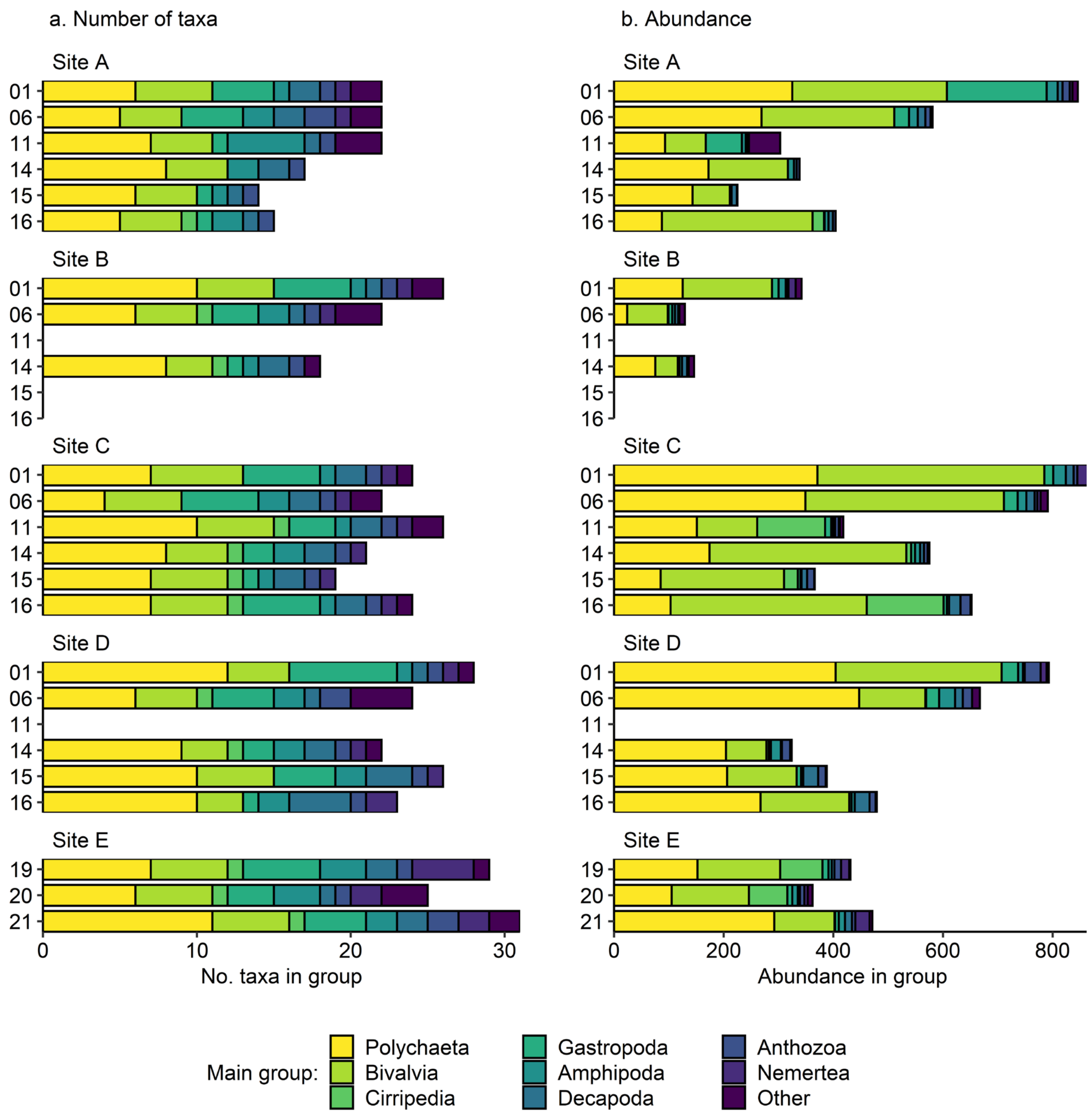


Fig. 9. Pooled data showing the contribution of main taxonomic groups to site richness and abundance.

Of interest is that despite the richness and abundance differences, values of the biological index AMBI were rated as indicative of 'good' or 'very good' estuary health. AMBI values were reasonably consistent across years (hence providers), but lower (i.e. indicative of higher relative health) in the sandier sediments of Site E and generally slightly higher at Site A (Fig. 11). There was no clear association between AMBI scores and the sediment grain size or trophic state variables described above (Pearson correlation, r = 0.19 TN to 0.40 TOC).

The low AMBI scores reflect a high prevalence of species or higher taxa classified as EG-I or EG-II (Fig. 12), being those eco-groups regarded as relatively sensitive to enrichment and other types of environmental pollution. For example, across the dataset there were 18 species classified as EG-I and 39 EG-II (Appendix 6). Some of the EG-II species were quite widely-occurring and abundant, such as cockles (*Austrovenus stutchburyi*), wedge shells *(Macomona liliana*), and spionid worms *Prionospio aucklandica* and *Boccardia* spp. (Table 7). Widely-occurring subdominant EG-II species include nutshells (*Linucula hartvigiana*). Many of these abundant macrofauna are known to be important prey items for birds, fish and rays.

Among the highly sensitive EG-I species the sunset shell *Hiatula* spp. was relatively abundant at Site E, reflecting the sandy sediment at that site. The EG-I taxa were otherwise represented by a suite of species whose abundances were low and whose occurrences were patchy. These taxa included the spionid worm *Aonides* (e.g. *Aonides trifida*), and shrimp-like mysids and cumaceans.

Among the more resilient species, those that were relatively abundant and widely-occurring among sites and surveys were:

- The capitellid worm, *Heteromastus filiformis* (EG-III), a generalist species that can thrive in disturbed conditions.
- Polychaete 'ragworms', *Nicon aestuariensis* (EG-III), which can tolerant strong freshwater influences.
- Mudflat anemones, *Anthopleura aureoradiata* (EG-III).
- The ubiquitous small bivalve *Arthritica* spp. (EG-IV).
- Various crabs, including hardy mud crabs, *Hemiplax hirtipes* (EG-V).
- Also abundant where shell hash was common (to enable attachment) were small filter-feeding estuarine barnacles, *Austrominius modestus*.

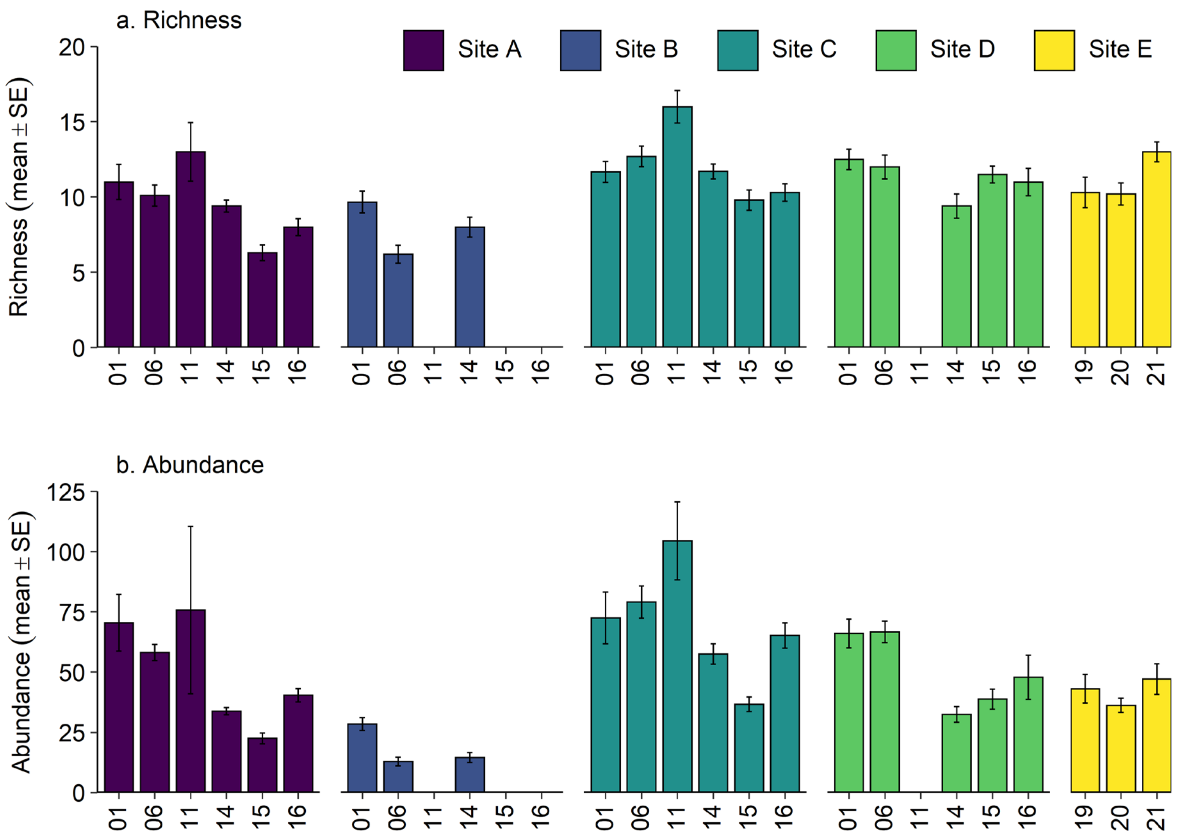


Fig. 10. Patterns (mean ± SE) in taxon richness and abundance per core sample.

Table 7. Sediment-dwelling species that comprised ≥5% of total abundance at any one site. The Table shows site abundances pooled across cores within each survey.

Main group	Taxa (eco-group)	Site A 2001	Site A 2006	Site A 2011	Site A 2014	Site A 2015	Site A 2016	Site B 2001	Site B 2006	Site B 2014	Site C 2001	Site C 2006	Site C 2011	Site C 2014	Site C 2015	Site C 2016	Site D 2001	Site D 2006	Site D 2014	Site D 2015	Site D 2016	Site E 2019	Site E 2020	Site E 2021
Bivalvia	*Arthritica* spp.[1] (EG-IV)	171	162	43	85	16	175	50	17	4	44	114	28	155	58	115	66	17	3	11	36	1	-	6
Bivalvia	*Austrovenus stutchburyi* (EG-II)	69	73	12	49	47	92	94	52	24	303	216	63	181	150	229	197	88	67	94	99	28	34	45
Bivalvia	*Hiatula* spp. (EG-I)	-	-	-	-	-	-	1	-	-	3	1	-	-	-	-	-	-	-	2	-	19	38	1
Bivalvia	*Macomona liliana* (EG-II)	35	6	4	9	3	7	16	4	13	31	21	3	10	7	11	30	13	4	16	27	100	67	55
Cirripedia	*Austrominius modestus* (no EG)	-	-	-	-	-	21	-	1	2	-	-	124	9	25	140	-	1	4	-	-	77	70	1
Gastropoda	*Potamopyrgus estuarinus* (EG-III)	155	-	66	-	-	-	5	-	-	-	-	-	-	-	-	9	8	-	-	-	-	-	-
Oligochaeta	Oligochaeta (EG-III)	-	1	52	-	-	-	-	-	-	-	-	5	-	-	-	-	-	-	-	-	-	-	-
Polychaeta	*Boccardia* spp. (EG-II)	7	-	3	8	27	35	1	-	1	-	-	5	10	14	5	12	-	2	28	28	1	1	18
Polychaeta	*Heteromastus filiformis* (EG-III)	155	30	-	60	5	-	34	1	18	111	49	56	42	6	16	214	239	90	13	48	7	1	83
Polychaeta	*Nicon aestuariensis* /Nereididae[2] (EG-III)	62	145	49	57	107	44	37	11	27	35	107	31	44	46	38	28	67	37	17	21	29	12	35
Polychaeta	*Prionospio aucklandica* (sp.)[3] (EG-II)	97	85	21	36	1	2	33	-	6	190	190	34	66	10	39	114	131	56	134	157	110	76	139
Polychaeta	*Scolecolepides benhami* (sp.)[3] (EG-IV)	-	8	4	9	3	6	14	9	14	-	3	-	5	9	5	-	2	3	9	7	3	-	4

[1] Assumed to be the same species identified by NIWA QA from other regional samples as *Arthritica* sp. 5.
[2] Nereididae were juveniles that were likely to have been *Nicon aestuariensis*.
[3] Assumed to be the same as *Prionospio* sp. and *Scolecolepides* sp., respectively, in Cawthron data.

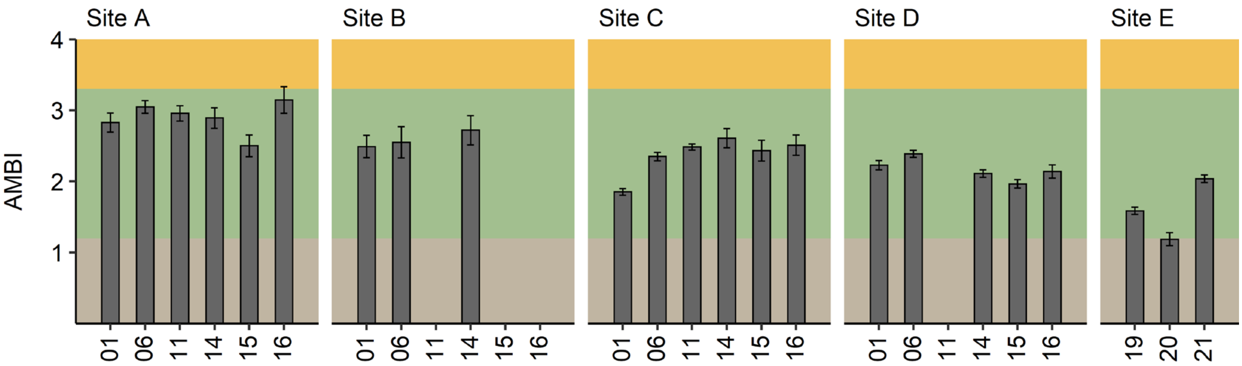


Fig. 11. Patterns (mean ± SE) in AMBI scores compared with condition rating criteria, with values indicating 'good' or 'very good' habitat conditions.

Condition rating key:

Very Good	Good	Fair	Poor

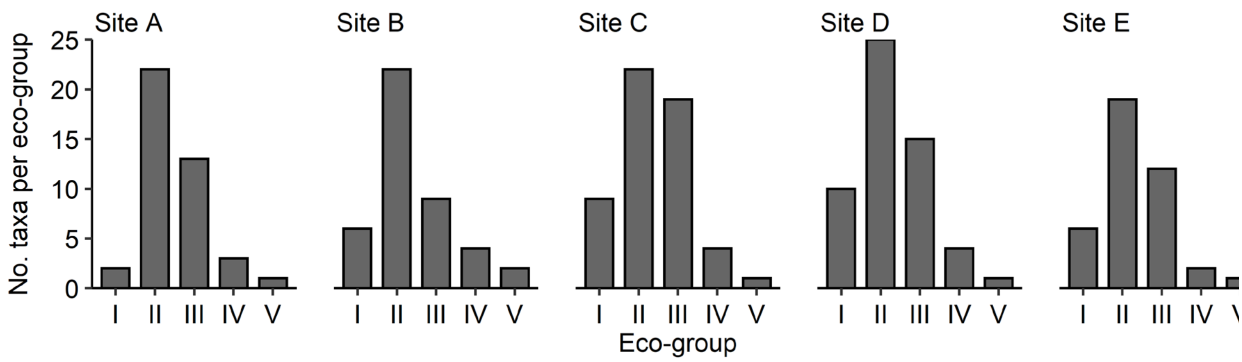
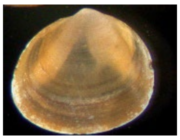
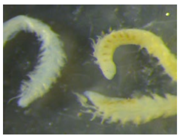


Fig. 12. Patterns in the number of taxa falling within eco-groups ranging from sensitive (EG-I) to tolerant (EG-V) to enrichment and other types of environmental pollution. For illustrative purposes, site data have been pooled across years.

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
Bivalvia, *Arthritica* spp.	A small sedentary deposit feeding bivalve that lives buried in the mud. Tolerant of muddy sediments and moderate levels of organic enrichment.	
Bivalvia, *Austrovenus stutchburyi*	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 diet of certain birds, rays and fish.	
Bivalvia, *Hiatula* spp.	Also known as sunset shells. A filter feeder most commonly associated with sandy sediments. An EG-I species considered sensitive to mud and enrichment.	
Bivalvia, *Macomona liliana*	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 diet of certain birds, rays and fish.	
Cirripedia, *Austrominius modestus*	Filter feeding estuarine barnacle, very common where there is shell material or other hard surfaces for attachment. Considered sensitive to muddy sediment.	
Gastropoda, *Potamopyrgus estuarinus*	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.	
Oligochaeta	Segmented worms in the same group as earthworms. Deposit feeder that is generally considered pollution or disturbance tolerant.	
Polychaeta, *Heteromastus filiformis*	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, *Nicon aestuariensis*/ Nereididae	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, Spionidae	The three dominant spionid worm species were as follows: *Boccardia* spp. (CMEC named *Boccardia accus*). This species prefers a low mud content but is found in a wide range of sand/mud habitats. It is considered very sensitive to organic enrichment. *Prionospio aucklandica*/sp., a species associated mainly with muddy sands, but occurs across a range of mud contents. Considered tolerant to organic enrichment although has an EG II classification. *Scolecolepides benhami*/sp. is a surface deposit feeder that commonly occurs in sandy/mud estuaries.	 

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 different providers (see Methods section and Appendix 3).

Fig. 13a illustrates a suite of sites that formed a central cluster, for which similarity in pairwise comparisons (measured by the Bray-Curtis index) was typically 65-70%. This result primarily reflects the reasonably similar suite of dominant and sub-dominant species across these sites as evident in Table 7 and illustrated by the vector overlay in Fig. 13a. The main outliers were:

- Site E which had a suite of species that thrive in the sandy sediments that characterise that site (illustrated in the bubble plot in Fig. 13b). Dominant among these species were wedge shells (*Macomona liliana*) and sunset shells (*Hiatula* spp.). Although Site E was sampled in a different time period to the other sites, we expect that habitat differences (rather than temporal change) are a more likely explanation for the its differentiation from other sites.
- Site B, which had the most impoverished macrofauna in terms of the richness and abundance of taxa (see Fig. 10). This was also the site with a sediment mud content that was intermediate between Site B and the other sites. It had the highest overall densities of the spionid worm *Scolecolepides benhami*.
- Most of the sites sampled over 2001-2011; although the Bray-Curtis similarity of each site with those in the main cluster was nonetheless relatively high at ~60%. Sites sampled in these years had a reduced mud content compared with later years at the same site, revealed by the steady temporal increase in mud described above. The pattern also coincides with a provider change after 2011, although the potential influence of this change on the nMDS analysis was reduced by the taxonomic aggregation undertaken.

The separation in the Fig. 13 ordination is in part driven by shifts in relative abundances of species, as well as compositional differences in the minor species. When the nMDS analysis was undertaken based on species presence only (i.e. ignoring abundances) all sites grouped at ~65% Bray-Curtis similarity. However, at a higher threshold of 70%, sites sampled in 2001 and 2006 did not group with each other, nor with the cluster formed by the remaining sites. This pattern suggests subtle temporal changes 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 changing environmental conditions, relationships between macrofaunal composition and sediment quality variables were explored. Using the BIO-ENV procedure in PRIMER, macrofaunal composition changes could not be related to any of the sediment quality variables, with only very weak correlations evident (Spearman rank correlation, ρ <0.27 for all variables). For non-metals, these weak correlations are evident by the short length of the vectors in Fig. 13b.

These results suggest that are other unmeasured factors that explain the observed spatial and temporal differences. 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 Waimea River, depth and location-related effects of wind-induced wave disturbance, as well as biological processes such as recruitment events and species interactions.

Despite the absence of strong correlations of macrofauna changes with sediment quality, Mud BHM values for Waimea Inlet equate to impact categories of 'moderate' (score 3 to <4) or 'high' (4 to <5) relative to other New Zealand estuaries (Fig. 14). The relative scale used for the Mud BHM has not been calibrated to provide guidance on the magnitude and significance of impacts in absolute terms, but the results nonetheless suggest that Waimea is relatively impacted in a national context, which is consistent with the 'fair' or 'poor' ratings against the New Zealand ETI values shown above in Fig. 6. However, there are site-to-site differences for which the BHM result is less clear. For example, some of the highest Mud BHM values were for Sites A and B, whose sediment mud content was moderate and much less than at Site D.

a. Main species or higher taxa

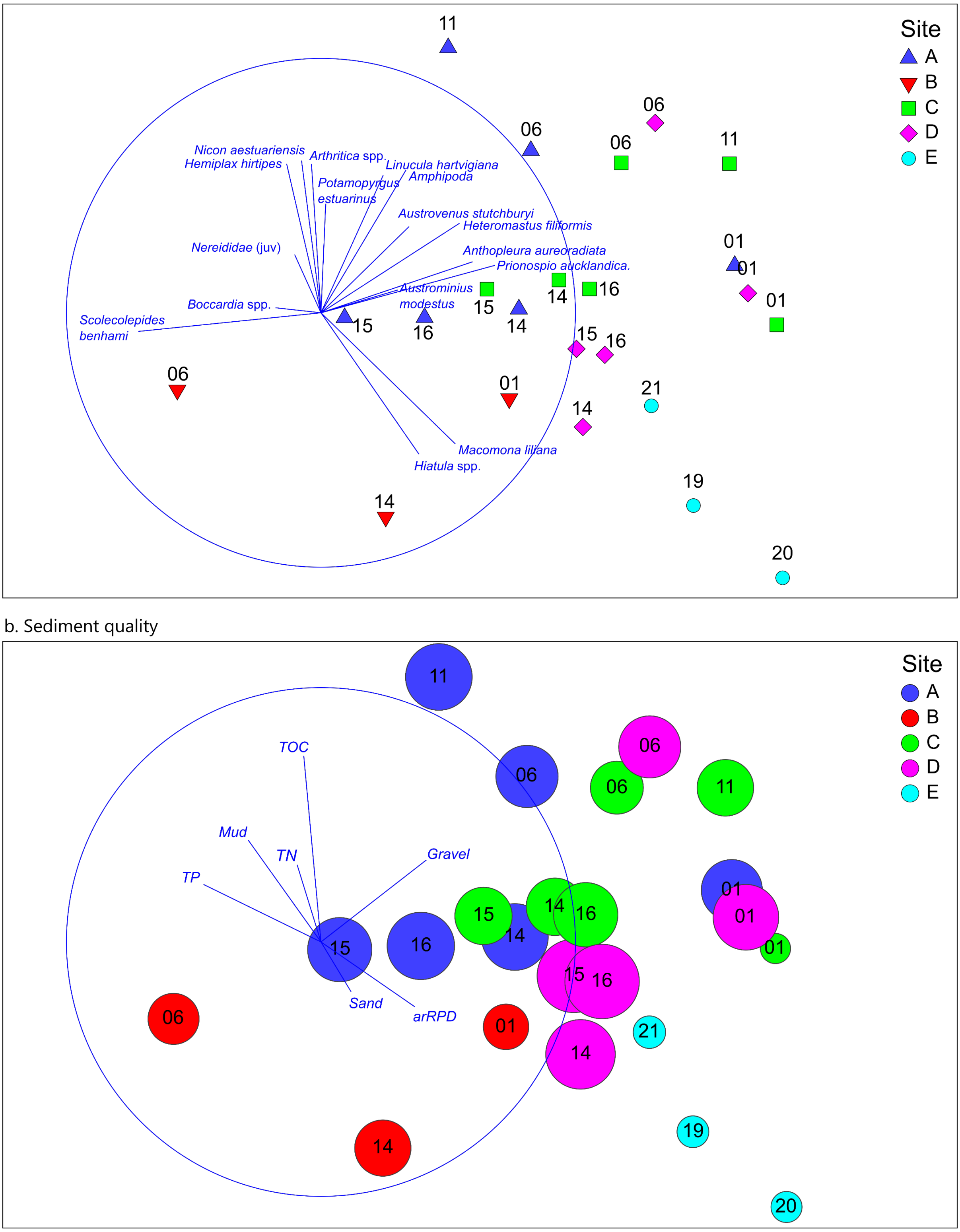


Fig. 13. Non-metric MDS ordination of macrofaunal core samples for data aggregated within each site, and subject to the taxonomic aggregation described in Appendix 3.

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; Bottom: vectors representing the most correlated sediment quality variables. In this case none of the variables were strongly associated with the ordination pattern. Bubble sizes are scaled to sediment % mud.

Similarly, Mud BHM scores were quite high for Site E, where the sediments are dominated by relatively clean and well-flushed sand. Nonetheless, it is of interest that within each of Sites A-D there is appears to be an increase in BHM scores over time, which is generally consistent with their increased sediment mud content.

Metals BHM scores ranged from 'good' to 'fair'. The latter appears counter-intuitive considering that concentrations for copper, lead, and zinc (on which the BHM is based) were very low relative to national sediment quality DGVs. The fair category encompasses the effect concentrations (FEC) guidelines derived by Hewitt et al. (2009) using data from Auckland estuaries (BHM scores = 4.1-4.5). These values represent the point at which 5% of all taxa are predicted to suffer a ≥ 50% decrease in abundance.

The fair category also includes the adjusted community hazardous concentration 5% value (cHC5) derived by Kwok et al. (2008) using field data from Hong Kong (BHM score = 4.7). This value represents the highest concentration of a metal at which no benthic organisms are expected to be affected adversely. As such, a classification of 'fair' in the Waimea Inlet context does not necessarily mean that the relevant metals (copper, lead, zinc) are exerting an adverse effect. Furthermore, as with ETI ratings, it is the temporal trends in BHM scores that are of most interest in the Waimea case. In this respect, the main point to note is that there is no consistent increase in Metals BHM scores over time.

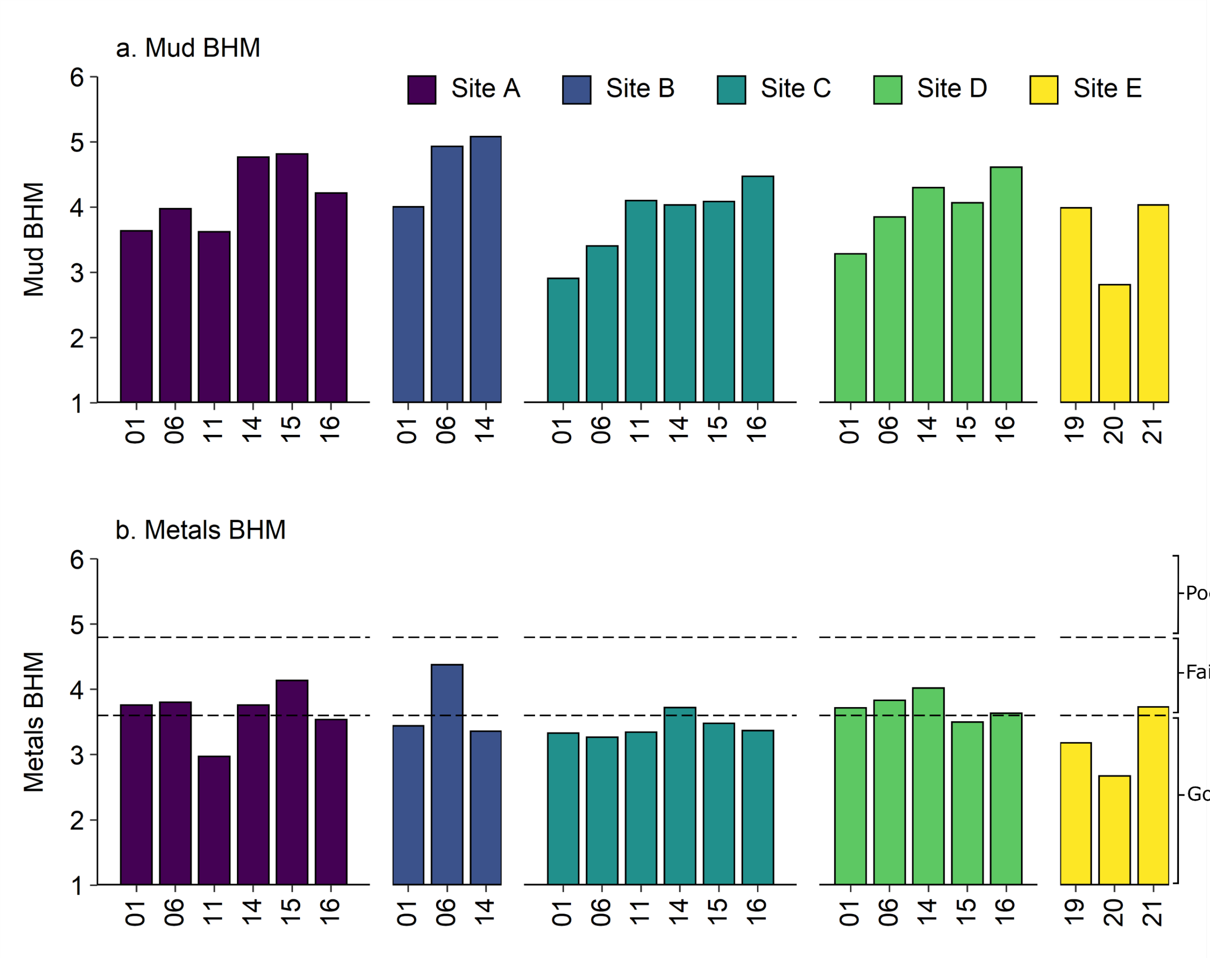


Fig. 14. Benthic health model (BHM) scores for mud and metals. The Mud BHM scores reflect a five-point scale from 1 ('very low') to 6 ('very high' impact relative to other New Zealand estuaries. Metals BHM scores are rated against an absolute scale from 'good' to 'poor' based on different sediment quality 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 Waimea Inlet since 2001, along with sedimentation monitoring undertaken since 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 condition rating criteria in Table 5.

Sediment mud content and sedimentation

Table 9 highlights the muddy nature of the sediments at Sites A-D, and the increase in mud content that has occurred since monitoring began in 2001. These four sites are currently rated as being in 'poor' condition due to mud content exceeding the biologically relevant threshold of 25%

Interestingly, in the three instances where sediment plates are next to fine scale sites, there was no logical concordance between muddiness and long-term or recent sedimentation patterns (Appendix 4). Sediment deposition was greatest at sandy Site E, but appears is attributable to a ridge of sand migrating onto the site

Table 9. Summary of condition scores of ecological health for each fine scale monitoring site, based on mean values of key indicators, and condition rating criteria in Table 5. Dash = missing value.

Site	Year	Mud %	TOC %	TN mg/kg	TP mg/kg	aRPD mm	As mg/kg	Cd mg/kg	Cr mg/kg	Cu mg/kg	Hg mg/kg	Ni mg/kg	Pb mg/kg	Zn mg/kg	AMBI (0-7)
A	2001	31.9	0.57	633	441	30	-	0.100	69.3	10.3	-	65.1	4.2	44.2	2.8
	2006	33.8	0.75	468	458	20	-	< 0.100	48.6	7.9	-	64.8	6.4	34.7	3.0
	2011	42.5	1.00	380	480	10	6.0	< 0.090	55.0	9.3	< 0.05	70.0	7.9	39.0	3.0
	2014	42.7	0.54	700	437	15	5.2	0.025	51.7	9.8	0.03	74.0	7.4	40.0	2.9
	2015	37.4	0.35	633	463	10	-	-	-	-	-	-	-	-	2.5
	2016	44.4	0.49	700	463	10	-	-	-	-	-	-	-	-	3.1
B	2001	15.9	0.43	279	480	30	-	0.100	44.6	8.8	-	72.3	6.3	38.4	2.5
	2006	19.9	0.58	353	516	20	-	< 0.100	32.0	6.7	-	69.4	5.1	27.9	2.6
	2014	25.2	0.38	333*	493	20	5.6	0.016	31.7	7.4	0.02	75.3	5.6	32.0	2.7
C	2001	9.6	0.31	329	273	30	-	0.375	61.3	7.0	-	58.3	7.7	34.5	1.9
	2006	21.6	0.84	550	376	20	-	< 0.100	42.3	7.8	-	60.6	5.9	28.2	2.4
	2011	25.5	0.75	387	400	10	5.0	< 0.100	49.0	8.9	< 0.05	64.0	6.3	33.0	2.5
	2014	26.6	0.54	733	370	10	5.4	0.022	51.0	9.3	0.02	72.7	6.8	37.7	2.6
	2015	26.5	0.42	800	410	10	-	-	-	-	-	-	-	-	2.4
	2016	37.6	0.51	700	397	20	-	-	-	-	-	-	-	-	2.5
D	2001	40.5	0.85	783	539	30	-	0.475	95.2	12.3	-	94.2	11.3	50.2	2.2
	2006	33.4	0.89	487	509	20	-	< 0.100	55.1	9.4	-	89.2	6.4	34.5	2.4
	2014	50.1	0.62	700	530	10	6.3	0.026	58.3	10.4	0.02	95.3	7.0	41.0	2.1
	2015	62.6	0.67	967	637	5	-	-	-	-	-	-	-	-	2.0
	2016	66.8	0.98	1033	577	10	-	-	-	-	-	-	-	-	2.1
E	2019	10.1	0.17	< 500	310	30	4.5	0.014	33.0	5.4	< 0.02	49.3	5.2	30.7	1.6
	2020	9.9	0.18	< 500	280	29	4.2	0.013	32.3	5.6	< 0.02	58.3	5.2	33.7	1.2
	2021	10.3	0.18	< 500	303	24	4.7	0.014	35.7	6.0	< 0.02	58.0	5.2	33.7	2.0

** Sample mean includes values below lab detection limits*

< All values below lab detection limit

Condition rating key:

Very Good	Good	Fair	Poor

and does not reflect the deposition of fine muds (LS, pers. obs.). At sediment plate Sites 4 and 7, which are next to muddier fine scale Sites A and B, respectively, there has been a negligible increase in sediment deposition despite an increase in muddiness. This is most likely due to interstitial spaces between coarser sediments infilling with fine muds over time, but with no overall change in sediment accrual.

Overall, the average net deposition across all sites in the estuary since 2008 was ~1mm/yr. This exceeds the historical (pre-European) rate of 0.5-0.7mm/yr recently estimated using ^{14}C dating methods (Gibbs & Swales 2019), but is less than the 1.7mm/yr sediment deposition estimated from coarse modelling of the ratio of current to natural state sediment loads (assuming 50% natural wetland sediment attenuation) (Stevens et al. 2020b). Despite low recent sedimentation, Waimea Inlet is becoming muddy and conceivably still expressing the effects of past periods of elevated sediment deposition. It is also important to recognise that sediment deposition most often occurs in large pulses, e.g. as a consequence of storm runoff or flooding, as opposed to gradual ongoing deposition, and that ongoing management of sediment sources is important, particularly for activities with cyclical disturbance like exotic forest harvesting.

Trophic status

Associated with the muddy sediment there appears to have been a shallowing of the aRPD, which is likely attributable to reduced oxygen penetration into the sediment matrix due to increased mud, but may in part reflect the coarse and subjective estimates that were made in most surveys. 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 none of the sites showed evidence of strong anoxia (e.g. black colour and strong sulphide-smell).

Superficially, the most degraded location was Site D, whose mud-dominated sediments were, in 2015 and 2016, covered in moderate growths of the opportunistic macroalga *Agarophyton chilense* (rated 'abundant', 20-50% cover). This species, known until recently as *Gracilaria chilensis*, can thrive in nutrient-enriched and/or muddy conditions, and in some New Zealand estuaries persistent dense growths have been symptomatic of the subsequent development of more widespread degradation (Stevens et al. 2020a). The recent broad scale survey report revealed an *Agarophyton* coverage of 30-90% in patches across the wider area around Site D, as well as a few other localised 'hot spots', but did not find evidence of widespread problems (Stevens et al. 2020b). The same report noted that modelled average nutrient loads to Waimea Inlet of $33mgN/m^2/d$, were considerably less that the threshold of ~$100mgN/m^2/d$ above which nuisance growths are commonly encountered in intertidally-dominated estuaries. However, it is possible that nutrient concentrations in localised streams or point source discharges may be sufficiently elevated to support nuisance growths, although data to assess this are currently lacking; the last comprehensive assessment was that undertaken by Gillespie et al. (2001).

Trace contaminants

Despite catchment development and urbanisation, concentrations of the measured anthropogenic contaminants on the main tidal flats were low. There are nonetheless elevated inputs around point and diffuse sources, such as trace metals from urban stormwater and formaldehyde from the MDF plant (Barter 2000). As noted above, the 'poor' rating for the trace element nickel, and moderately elevated chromium concentrations, are attributable to catchment geology and not anthropogenic sources. Elsewhere in the region, high concentrations of nickel have also been described, but no clear link to adverse ecological effects has been established (e.g. Forrest et al. 2007; Forrest & Stevens 2021). Concentrations of all other trace elements, including metals that are commonly elevated due to anthropogenic inputs, were very low, and often less than half of the national sediment quality guideline value for 'possible' ecological effects (ANZG 2018). As well as low trace element concentrations, analysis of 114 different priority pollutants (i.e. semi-volatile organic compounds, including pesticides) undertaken as part of the 2014 survey revealed all contaminants to be at concentrations less than method detection limits (Robertson & Robertson 2014).

Macrofauna and sediment quality

Despite the relatively high mud composition of some sites, there was nonetheless a moderately diverse and abundant macrofauna present. By comparison with other estuaries in the top of the South Island (Fig. 15), Waimea Inlet has relatively high macrofaunal richness and abundance values, second only to Nelson Haven. The obvious exception is Site B, for which species richness is low, although not the lowest of estuaries sampled regionally.

Due to very low Cu, Pb and Zn concentrations relative to the national DGV, it is of interest that the Benthic Health Model generally rated some Waimea sites as 'fair when evaluated against sediment quality guidelines that are more conservative that the DGVs. However, it is

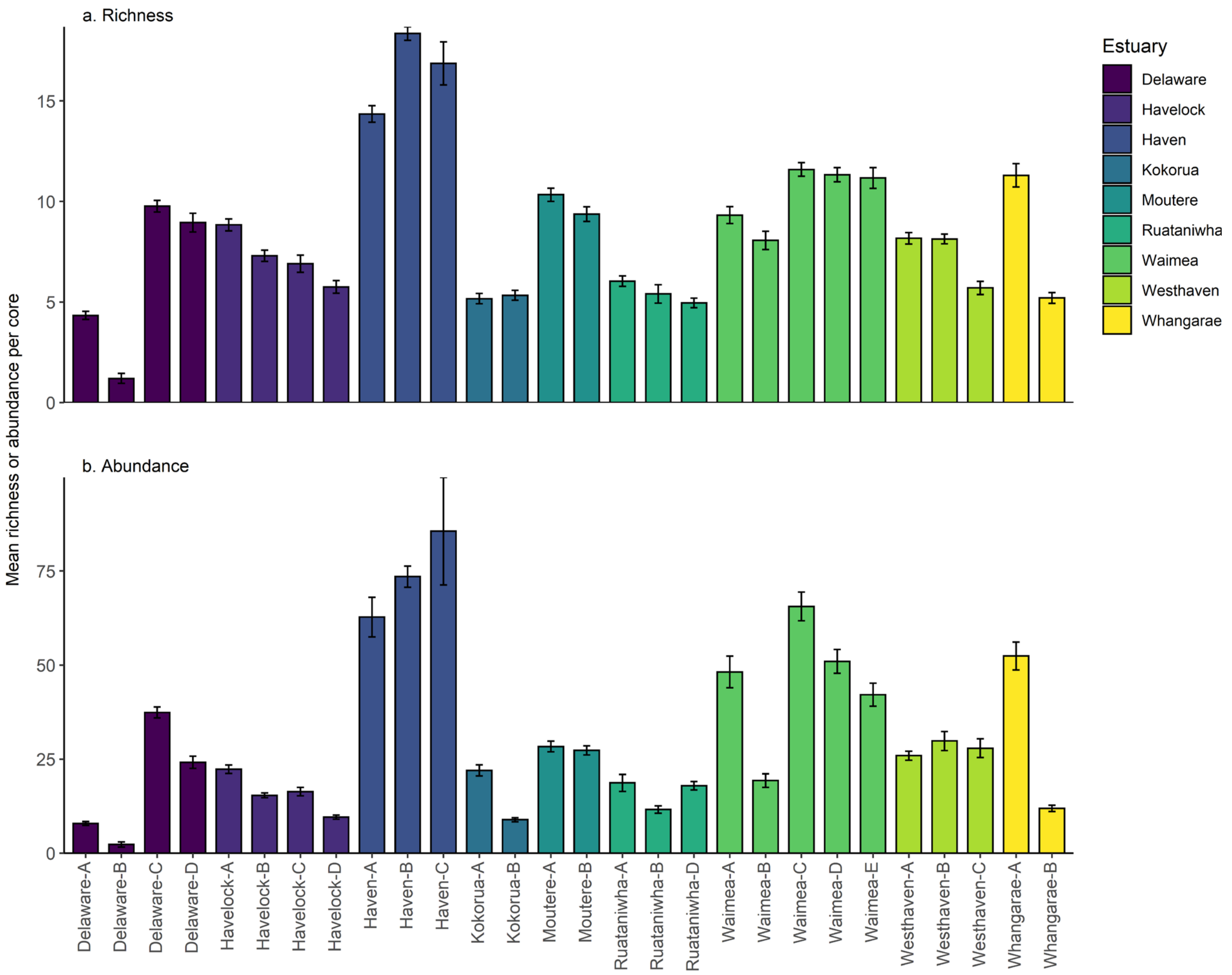


Fig. 15. Macrofauna richness and abundance summary (mean ±SE) based on NEMP monitoring in estuaries in the top of the South Island since 2014. For illustrative purposes, site-level data are averaged across multiple survey years in each location.

recognised that metals collectively can have a chronic impact at concentrations that are less that indicated by individual DGV thresholds (Hewitt et al. 2009). For example, the FEC threshold is based on concentrations of 6.5-9.3, 18.8-19.4, and 114-118mg/kg for coper, lead, and zinc, respectively. As indicated in Table 9, measured concentrations at Waimea fine scale sites are typically similar to, or lower than, these values- estuary averages for copper, lead, and zinc are 8.4, 6.5 and 36.1mg/kg, respectively, suggesting adverse impacts to macrofauna are unlikely. It is also of note that a total extractable analytical method is used for trace elements, which is likely to greatly overestimate the fraction that is bioavailable.

By contrast with the metals, Mud BHM scores within each site were consistent to some degree with the pattern of increasing muddiness at Sites A-D since 2001 (Table 9). However, differences in Mud BHM scores among sites were less intuitive. For example, the highest scores were associated with Site B, which is considerably less muddy than A, C and D. However, Site B was clearly the most impoverished of the five fine scale sites, having substantially reduced macrofaunal richness and abundances. This result may in part be due to higher hydrodynamic scouring of Site B relative to the other sites.

Of interest is that the other multivariate analyses did not reveal mud, nor in fact any of the measured sediment quality variables, to be clearly associated with the spatial and temporal changes observed in the macrofauna. This itself is surprising, given that sediment grain size composition, along with trophic state (enrichment) conditions, 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). This important role is the basis for inclusion of mud content and trophic state measures (i.e. TOC, TN, TP, aRPD) as key indicators in the ETI.

Overall the results and associated condition ratings indicate that the main tidal flats of Waimea Inlet are in a reasonably healthy condition ecologically. This situation has persisted despite the considerable historic modification of estuary margins, loss of salt marsh habitat, and considerable land development in the catchment. However, modelled estimates predict present-day sediment loads to be high compared with the natural state loads. Moreover, despite the absence of evidence for widespread sediment accretion, the gradual increase in sediment mud content at Sites A-D over 15 years (2001-2016) appears to be indicative of a relatively insidious degradation of habitat quality. In the event that this gradual accretion, or increase in mud content, has been ongoing since the last survey of these sites in 2016, there is a risk that the sediments will reach a point at which the mud tolerance of key species such as cockles and wedge shells (e.g. Robertson et al. 2015) 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. Recent studies have highlighted agricultural land uses and exotic forest harvest as being key historic or potential contributors of muddy sediment to Waimea Inlet (Gibbs & Woodward 2018; Gibbs & Swales 2019). It is important that these and other potential sources are monitored and managed so that the current state of the estuary is maintained or improved.

5.2 CONSIDERATIONS FOR FURTHER ASSESSMENT AND MONITORING

Given that this report represents a synthesis of data collected over two decades, TDC and NCC have asked us to consider the implications of the NEMP findings (including the recent broad scale survey) in terms of further needs for investigation and ongoing monitoring.

NEMP fine scale monitoring is valuable for understanding long-term ecological changes across the main body of an estuary, with broad scale monitoring helping track changes in the main habitats and identify eutrophication symptoms. There is certainly benefit in having long-term data that are collected using standardised approaches. Nonetheless, it is timely to consider whether monitoring needs to be modified or extended; for example, so that it is includes areas of Waimea Inlet that are most vulnerable to change from land use and other anthropogenic activities. Targeted monitoring and investigation in estuary hot spots (e.g. as identified in the broad scale survey) may better elucidate cause-effect linkages, and enable problems to be addressed before they become estuary-wide issues.

1. Investigative approaches

The fine scale survey highlights increasing sediment mud content over many years. Although symptoms of excess eutrophication were not evident at fine scale sites, the broad scale survey described some emerging hot spots of macroalgal growth and sediment enrichment. In addition, while the fine scale monitoring does not illustrate widespread anthropogenic contamination, there are sources of contaminants (e.g. urban stormwater) whose local influence is not well understood. In these respects the NEMP approach is limited in that it does not provide:

- An early warning of estuary degradation, as sampling sites are typically remote from point source influences.
- Insight into cause-effect linkages, such as sub-catchment influences and the origins of muddy sediment inputs.

Accordingly, to better understand the changing state of Waimea Inlet and its current pressures, we recommend that TDC and NCC consider the following:

- Desktop assessment of catchment stressor sources and inputs based on: (i) a detailed assessment of land uses, including forestry harvesting patterns; (ii) modelled mass loads of nutrients and sediments (e.g. Hicks et al. 2019); and (iii) appraisal of freshwater (including stormwater) inputs and associated water quality data.
- Targeted evaluation of estuary condition in the vicinity of point source inputs and/or where local issues have already been identified (or are identified from desktop assessment). Such an evaluation should use key indicators from NEMP fine scale and broad scale (as appropriate) approaches, as well as analysis of a full suite of priority pollutants (e.g. pesticides).

A particular focus should be on links between catchment land use and muddy sediment inputs. This component could include targeted investigations that complement sediment source tracking work recently undertaken for TDC (i.e. Gibbs & Woodward 2018; Gibbs & Swales 2019). Detailed assessment of recent, current and potential future catchment land use changes could be used to estimate potential changes in sediment loads. For example, it would be helpful to understand forest harvest schedules, given that ~27% of the catchment is in exotic forest (see Table 1 in Section 2), and harvesting is a practice that has already been identified as an important sediment source.

2. Ongoing NEMP 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 guidance was that reducing macrofauna sampling effort or frequency would affect the robustness of monitoring programmes. Current TDC and NCC monitoring is at a considerably lower level than this recommended optimum.

At present, NEMP fine scale monitoring is typically undertaken every 5 years by TDC and NCC, after first establishing a baseline for a given estuary. Sediment plate monitoring is typically undertaken annually, although Stevens et al. (2020b) recommended biennial monitoring for Waimea Inlet due to the low sedimentation rates measured. As there has been no fine scale monitoring at Waimea Sites A-D since 2016, but a steady increase in sediment muddiness, it would be timely to undertake a follow-up survey to determine whether there has been ongoing degradation over the last 5 years. Due to council budget constraints, we have considered the scope for reducing per survey effort and cost (bearing in mind the recommendations from NIWA guidance).TDC have also ask us to consider whether macrofaunal monitoring could be dropped from the SOE programme. In these respects we suggest the following:

Fine scale sites: Due to sites A-D showing similar temporal patterns, there appears to be some redundancy at present. We suggest retaining three representative sites for long-term monitoring as follows: Site A (TDC main body of estuary), Site D (TDC western arm) and Site E (NCC eastern arm).

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 collecting long-term data 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. 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: Macrofauna provide an important component of estuary assessment, and monitoring at 5-year intervals is reasonable. That said, if budget constraints mean that 5-yearly interval for fine scale surveys cannot be achieved, there should at least be regular monitoring of sediment quality, in particular to assess changes in sediment mud content and aRPD. A suggested sampling approach is to:

- Undertake another fine scale survey when the monitoring budget allows, incorporating the amendments above (reduced sites and sampling effort).
- Undertake annual to biennial sediment plate monitoring, and at each site (or a subset): (i) measure aRPD; (ii) subjectively assess sediment texture using NEMP broad scale methods; and (iii) collect a single composite sample (from each site, or a subset) for laboratory grain size analysis.

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.

5.3 RECOMMENDATIONS

This report has undertaken a synthesis of ecological monitoring data collected in Waimea Inlet since 2001, as part of SOE monitoring conducted by TDC and NCC. Although the estuary is in a reasonably healthy state, the gradual increase in sediment mud levels is a potential concern. 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 better link estuary state with drivers of change. Accordingly, to better understand Waimea Inlet and its current pressures, we recommend that TDC and NCC consider the following:

1. Desktop assessment of catchment stressor sources and inputs based on: (i) a detailed assessment of land uses, including exotic forest harvest patterns; (ii) modelled mass loads of nutrients and sediments (e.g. Hicks et al. 2019); and (iii) appraisal of freshwater (including stormwater) inputs and associated water quality data. A particular focus should be on links between catchment land use and muddy sediment inputs.

2. Investigation of estuary condition in the vicinity of point source inputs and/or where local issues have already been identified, including sampling key NEMP indicators and a suite of priority pollutants (e.g. pesticides). The scope would be better determined after completion of the assessment in #1.

3. Undertake annual or biennial monitoring of sedimentation, sediment composition and enrichment status at representative sites. Also undertake another fine scale survey (reduced in terms of sites and sampling effort) when budgets allow, to evaluate ongoing degradation (due to muddy sediment inputs) since TDC Sites A-D were last sampled in 2016.

4. 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.

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

Fine Scale Site A	NZTM_East	NZTM_North	Fine Scale Site C	NZTM_East	NZTM_North
A1	1615281	5426009	C1	1614862	5427973
A2	1615309	5426018	C2	1614921	5427984
A3	1615315	5425958	C3	1614918	5428014
A4	1615286	5425949	C4	1614859	5428002
Fine Scale Site B	NZTM_East	NZTM_North	Fine Scale Site D	NZTM_East	NZTM_North
B1	1607353	5431902	D1	1608898	5430059
B2	1607380	5431889	D2	1608927	5430051
B3	1607351	5431836	D3	1608939	5430108
B4	1607324	5431849	D4	1608909	5430117

Fine Scale Site E	NZTM_East	NZTM_North
E1	1617225	5425942
E2	1617249	54259234
E3	1617281	5425973
E4	1617258	5425992

All sites have four plates except for Site 1a, which has 16 plates. Monitoring at Site 1a was discontinued after the 2014 survey.

Site	Location	NZTM_East	NZTM_North	Baseline year	Install date
1a	Monaco South	1617685	5426685	2008	10/09/2008
1	T&G (ENZA)	1617507	5426223	2008	26/09/2008
2	Richmond Transfer Station	1616187	5424969	2008	11/09/2008
3	Reservoir Creek	1616577	5425289	2008	11/09/2008
4	Borck Creek	1615233	5425909	2008	11/09/2008
5	Bell-Best Island Traverse	1613263	5429166	2008	11/09/2008
6	Research Orchard-Hoddy	1608487	5429542	2008	12/09/2008
7	Bronte-Hoddy	1607242	5431180	2008	12/09/2008
8	Bronte-Mapua	1607256	5431872	2008	12/09/2008
9	Rabbit Island Boat Ramp	1613009	5429782	2012	25/09/2012
10	Rough Island Equestrian Area	1609380	5431142	2012	25/09/2012
11	Monaco North	1617883	5427206	2015	16/01/2015
12	Orphanage Stream	1617299	5425292	2015	16/01/2015

Appendix 2. RJ Hill analytical methods for sediments

Sample Type: Sediment			
Test	**Method Description**	**Default Detection Limit**	**Sample No**
Environmental Solids Sample Drying*	Air dried at 35°C Used for sample preparation. May contain a residual moisture content of 2-5%.	-	1-12
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%.	-	1-12
Dry Matter (Env)	Dried at 103°C for 4-22hr (removes 3-5% more water than air dry) , gravimetry. (Free water removed before analysis, non-soil objects such as sticks, leaves, grass and stones also removed). US EPA 3550.	0.10 g/100g as rcvd	13-16
Dry Matter for Grainsize samples (sieved as received)*	Drying for 16 hours at 103°C, gravimetry (Free water removed before analysis).	0.10 g/100g as rcvd	1-12
Total Recoverable digestion	Nitric / hydrochloric acid digestion. US EPA 200.2.	-	1-12
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	1-12
Total Nitrogen*	Catalytic Combustion (900°C, O2), separation, Thermal Conductivity Detector [Elementar Analyser].	0.05 g/100g dry wt	1-12
Total Organic Carbon*	Acid pretreatment to remove carbonates present followed by Catalytic Combustion (900°C, O2), separation, Thermal Conductivity Detector [Elementar Analyser].	0.05 g/100g dry wt	1-12
Heavy metals, trace As,Cd,Cr,Cu,Ni,Pb,Zn,Hg	Dried sample, <2mm fraction. Nitric/Hydrochloric acid digestion, ICP-MS, trace level.	0.010 - 0.8 mg/kg dry wt	1-12
Semivolatile Organic Compounds Trace in Soil by GC-MS	Sonication extraction, GC-MS analysis. Tested on as received sample. In-house based on US EPA 8270.	0.002 - 6 mg/kg dry wt	13-16
3 Grain Sizes Profile as received			
Fraction >/= 2 mm*	Wet sieving with dispersant, as received, 2.00 mm sieve, gravimetry.	0.1 g/100g dry wt	1-12
Fraction < 2 mm, >/= 63 µm*	Wet sieving using dispersant, as received, 2.00 mm and 63 µm sieves, gravimetry (calculation by difference).	0.1 g/100g dry wt	1-12
Fraction < 63 µm*	Wet sieving with dispersant, as received, 63 µm sieve, gravimetry (calculation by difference).	0.1 g/100g dry wt	1-12

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

A. Renaming of species undertaken within each dataset to ensure consistent species names were applied across years, which also followed the accepted names in the World Register of Marine Species. (Format: old name = new name).

Capitellethus zeylanicus = Notomastus zeylanicus,

Diloma subrostrata = Diloma subrostratum,

Elminius modestus = Austrominius modestus,

Haminoea zelandiae = Papawera zelandiae,

Helice crassa = Austrohelice crassa,

Hemipodus simplex = Hemipodia simplex,

Macrophthalmus hirtipes = Hemiplax hirtipes,

Notoacmea helmsi = Notoacmea elongata,

Nucula hartvigiana = Linucula hartvigiana,

Pectinaria australis = Lagis australis,

Perrierina turneri = Legrandina turneri,

Scoloplos cylindrifer = Leodamas cylindrifer,

Soletellina sp. = *Hiatula* spp.,

Trochodota dendyi = Taeniogyrus dendyi,

Tellina liliana = Macomona liliana.

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 2011.

a. Raw data

Year	2001	2006	2011	2014	2015	2016	2019	2020	2021
2001	100								
2006	45	100							
2011	30	35	100						
2014	21	23	28	100					
2015	25	22	27	65	100				
2016	25	25	27	61	68	100			
2019	23	23	21	44	46	58	100		
2020	23	20	18	38	44	47	54	100	
2021	25	25	30	56	63	63	50	60	100

b. Aggregated data

Year	2001	2006	2011	2014	2015	2016	2019	2020	2021
2001	100								
2006	54	100							
2011	46	41	100						
2014	45	46	46	100					
2015	49	38	48	68	100				
2016	46	47	48	63	65	100			
2019	45	40	40	54	56	71	100		
2020	41	39	32	41	46	50	60	100	
2021	48	49	50	57	63	63	59	63	100

C. Taxonomic aggregation 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).

Main group	Cawthron	CMEC	nMDS taxa	BHM taxa
Gastropoda	caw	cmec	*Amphibola crenata*	amphibola.crenata
Amphipoda	caw	NA	*Amphipoda*	amphipod.other
Anthozoa	caw	cmec	*Anthopleura aureoradiata*	anthopleura.hermaphroditica
Polychaeta	caw	NA	*Aonides* sp.	aonides
Polychaeta	caw	NA	*Armandia maculata*	armandia.maculata
Bivalvia	caw	NA	*Arthritica bifurca*	arthritica
Bivalvia	caw	cmec	*Austrovenus stutchburyi*	austrovenus.stutchburyi
Polychaeta	caw	NA	*Boccardia* sp.	polydorid.complex
Polychaeta	caw	NA	*Capitella capitata*	capitella.oligochaete
Gastropoda	caw	cmec	*Cominella glandiformis*	cominella.glandiformis
Gastropoda	caw	cmec	*Diloma subrostratum*	diloma
Polychaeta	caw	NA	*Eulalia microphylla*	phyllodocidae
Isopoda	caw	NA	*Isopoda Anthuroidea*	anthuroidea
Polychaeta	caw	NA	Glyceridae	glyceridae
Decapoda	caw	cmec	*Halicarcinus whitei*	halicarcinus
Gastropoda	caw	cmec	*Papawera zelandiae*	haminoea.zelandiae
Polychaeta	caw	NA	*Leodamas cylindrifer*	orbiniidae
Polychaeta	caw	cmec	*Heteromastus filiformis*	heteromastus.filiformis.baranatolla.lepte
Diptera	caw	NA	Diptera	NA
Bivalvia	caw	cmec	*Macomona liliana*	macomona.liliana
Decapoda	caw	cmec	*Hemiplax hirtipes*	austrohelice.hemigrapsus.hemiplax
Polychaeta	caw	NA	*Magelona papillicornis*	magelona
Polychaeta	caw	NA	Maldanidae	maldanidae
Gastropoda	caw	NA	*Micrelenchus tenebrosus*	cantharidus.micrelenchus
Decapoda	caw	NA	*Natantia* unid.	crustacea.unid
Nemertea	caw	NA	Nemertea	nemertea
Polychaeta	caw	cmec	Nereididae (juv)	nereididae
Polychaeta	caw	cmec	*Nicon aestuariensis*	nereididae
Gastropoda	caw	cmec	*Notoacmea* spp.	notoacmea
Bivalvia	caw	cmec	*Linucula hartvigiana*	linucula.hartvigiana
Bivalvia	caw	cmec	*Paphies australis*	paphies.australis
Polychaeta	caw	NA	*Lagis australis*	pectinariidae
Gastropoda	caw	NA	*Potamopyrgus estuarinus*	potamopyrgus
Polychaeta	caw	NA	*Prionospio* sp.	prionospio.other
Polychaeta	caw	NA	*Scolecolepides* sp.	scolecolepides
Sipuncula	caw	NA	Sipuncula	sipuncula
Bivalvia	caw	cmec	*Hiatula* spp.	hiatula
Polychaeta	caw	NA	*Sphaerosyllis hirsuta*	syllidae
Gastropoda	caw	cmec	*Zeacumantus lutulentus*	zeacumantus.lutulentus
Gastropoda	caw	NA	*Zeacumantus subcarinatus*	zeacumantus.subcarinatus
Amphipoda	caw	NA	Amphipoda A	amphipod.other
Amphipoda	caw	NA	Amphipoda B	amphipod.other
Cirripedia	caw	cmec	*Austrominius modestus*	NA
Decapoda	caw	NA	*Callianassa filholi*	biffarius.filholi
Copepoda	caw	NA	Copepoda	NA
Cumacea	caw	NA	Cumacea	cumacea
Gastropoda	caw	NA	*Diloma zelandicum*	diloma
Diptera	caw	NA	*Dolichopodidae larvae*	NA
Anthozoa	caw	NA	*Edwardsia* sp.	edwardsiidae

Main group	Cawthron	CMEC	nMDS taxa	BHM taxa
Decapoda	caw	cmec	*Austrohelice crassa*	austrohelice.hemigrapsus.hemiplax
Mysidacea	caw	NA	Mysidacea	mysida
Nematoda	caw	NA	Nematoda	NA
Oligochaeta	caw	NA	Oligochaeta	capitella.oligochaete
Polychaeta	caw	NA	Paraonidae	paraonidae.other
Polychaeta	caw	cmec	*Scolecolepides benhami*	scolecolepides
Amphipoda	caw	NA	Aoridae	amphipod.other
Polychaeta	caw	cmec	*Boccardia acus*	polydorid.complex
Amphipoda	caw	NA	Corophiidae	corophiidae
Bivalvia	caw	NA	*Crassostrea gigas*	crassostrea.gigas
Decapoda	caw	cmec	Decapod megalopa	NA
Insecta	caw	NA	*Hudsonema amabile*	NA
Amphipoda	caw	NA	Melitidae	amphipod.other
Insecta	caw	NA	Orthocladiinae	NA
Polychaeta	caw	cmec	*Owenia petersenae*	owenia.petersenae
Amphipoda	caw	NA	*Phoxocephalidae*	phoxocephalidae
Polychaeta	caw	NA	*Polydora* sp.	polydorid.complex
Polychaeta	caw	cmec	*Prionospio aucklandica*	prionospio.aucklandica
Amphipoda	NA	cmec	*Paracalliope novizealandiae*	paracalliopiidae
Bivalvia	NA	cmec	*Arthritica* sp. 1	arthritica
Polychaeta	NA	cmec	*Boccardia syrtis*	polydorid.complex
Diptera	NA	cmec	Diptera sp. 1	NA
Anthozoa	NA	cmec	*Edwardsia* sp. 1	edwardsiidae
Gastropoda	NA	cmec	*Epitonium tenellum*	epitonium.tenellum
Polychaeta	NA	cmec	*Glycera lamelliformis*	glyceridae
Polychaeta	NA	cmec	*Axiothella serrata*	maldanidae
Nemertea	NA	cmec	Nemertea sp. 1	nemertea
Polychaeta	NA	cmec	*Paradoneis* sp.	paraonidae.other
Amphipoda	NA	cmec	*Torridoharpinia hurleyi*	phoxocephalidae
Polychaeta	NA	cmec	*Disconatis accolus*	polynoidae
Bivalvia	NA	cmec	*Cyclomactra tristis*	cyclomactra
Decapoda	NA	cmec	*Halicarcinus varius*	halicarcinus
Bivalvia	NA	cmec	Mytilidae sp. 1	mytilidae.other
Nemertea	NA	cmec	Nemertea sp. 3	nemertea
Polychaeta	NA	cmec	*Phyllodocidae* sp. 1	phyllodocidae
Copepoda	NA	cmec	Copepoda sp. 2	NA
Decapoda	NA	cmec	*Palaemon affinis*	palaemon
Polychaeta	NA	cmec	*Perinereis vallata*	nereididae
Amphipoda	NA	cmec	Amphipoda sp. 5	amphipod.other
Amphipoda	NA	cmec	Amphipoda sp. 6	amphipod.other
Polychaeta	NA	cmec	*Magelona dakini*	magelona
Nemertea	NA	cmec	Nemertea sp. 2	nemertea
Nemertea	NA	cmec	Nemertea sp. 5	nemertea
Polychaeta	NA	cmec	*Aonides trifida*	aonides
Isopoda	NA	cmec	*Isocladus* sp.	isopod.other
Polychaeta	NA	cmec	*Orbinia papillosa*	orbiniidae
Holothuroidea	NA	cmec	*Taeniogyrus dendyi*	taeniogyrus.dendyi

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 over 2001-2011 compared with 2014-2021. 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).

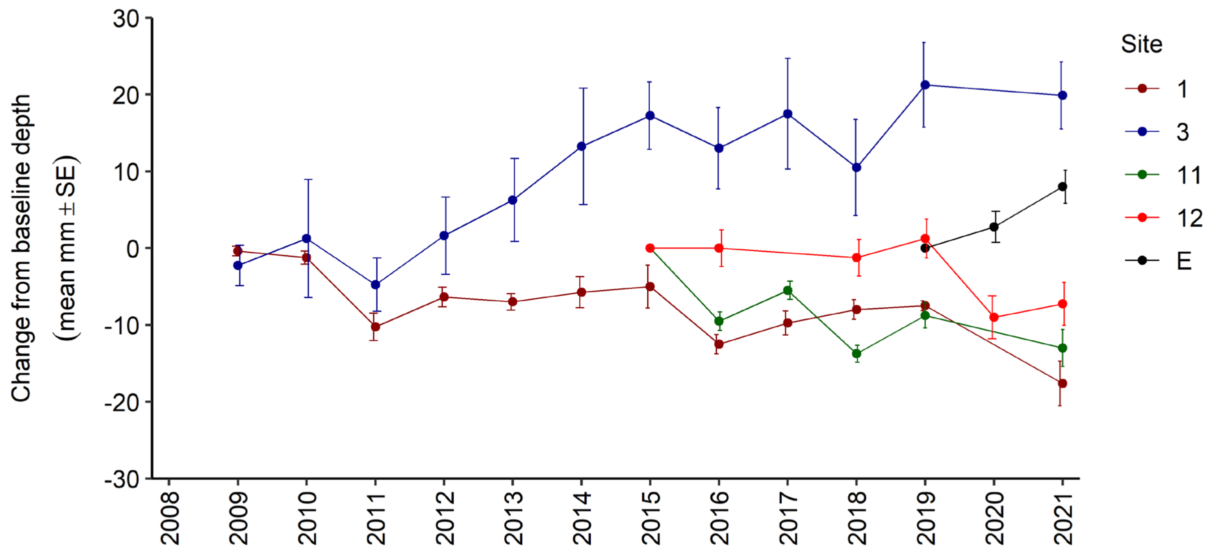
The Mud BHM tracks changes in health relative to increased mud content of the surface sediment as a surrogate for sediment accumulation rates. Mud BHM 'health' is defined by changes in benthic macroinvertebrate community structure observed along gradients of anthropogenic impact. This approach accounts 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).

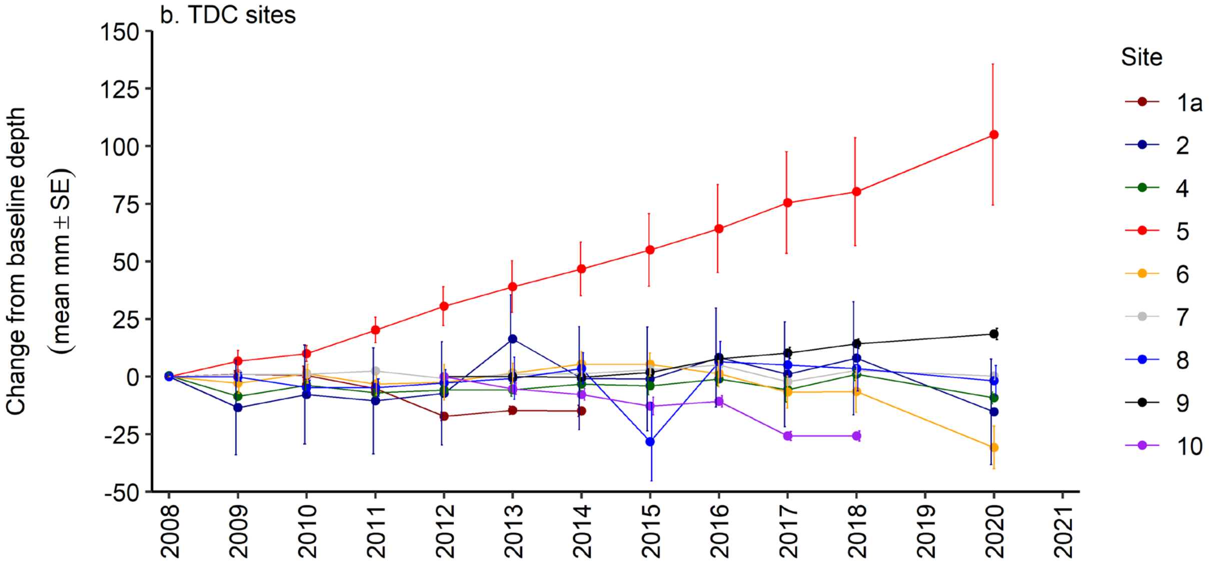
For Metals BHM scores, an absolute effects scale has recently been developed and is described by Clarke (2022, unpubl. Cawthron report). The absolute approach categorises sediment health as 'good', 'fair' or 'poor' when assessed against a suite of sediment quality guidelines that are more conservative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

For the present analysis, BHM scores were calculated by Dana Clarke 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





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. 'Poor' ratings (orange) exceed the national guideline value of 2mm/yr.

Council region	Site	Baseline date	Last sampling date	No years	Change from baseline depth (mm)	Annualised sedimentation (mm/yr since baseline)
NCC	1	26/09/2008	25/01/2021	12.3	-19.8	-1.6
NCC	3	26/09/2008	7/05/2021	12.6	19.0	1.5
NCC	11	16/01/2015	25/01/2021	6.0	-13.8	-2.3
NCC	12	16/01/2015	1/07/2020	6.0	-9.0	-1.5
NCC	E	5/04/2019	25/01/2021	1.8	8.0	4.4
TDC	1a	10/09/2008	9/09/2014	6.0	-14.9	-2.5
TDC	2	26/09/2008	1/07/2020	11.8	-15.2	-1.3
TDC	4	11/09/2008	1/07/2020	11.8	-9.3	-0.8
TDC	5	11/09/2008	1/07/2020	11.8	105.0	8.9
TDC	6	12/09/2008	1/07/2020	11.8	-30.8	-2.6
TDC	7	12/09/2008	1/07/2020	11.8	0.3	0.02
TDC	8	12/09/2008	1/07/2020	11.8	-1.8	-0.2
TDC	9	27/11/2012	1/07/2020	7.6	18.5	2.4
TDC	10	27/11/2012	3/09/2018	5.8	-25.8	-4.5

Appendix 5. Sediment quality raw data 2014-2021

Site	Year	Zone	Gravel %	Sand %	Mud %	TOC %	TN mg/kg	TP mg/kg	aRPD mm	As mg/kg	Cd mg/kg	Cr mg/kg	Cu mg/kg	Hg mg/kg	Ni mg/kg	Pb mg/kg	Zn mg/kg
A	2014	X	0.3	54.7	44.9	0.59	800	470	15	5.5	0.022	53	10.5	0.04	77	7.9	42
		Y	0.2	56.8	42.8	0.53	700	420	15	5.1	0.028	50	9.6	0.03	72	7.2	39
		Z	0.3	59.3	40.3	0.51	600	420	15	5	0.025	52	9.4	0.02	73	7.2	39
	2015	X	0.8	62.9	36.2	0.36	700	450	10	-	-	-	-	-	-	-	-
		Y	<0.1	59.7	40.2	0.36	600	500	10	-	-	-	-	-	-	-	-
		Z	0.4	63.8	35.8	0.33	600	440	10	-	-	-	-	-	-	-	-
	2016	X	1.8	50.3	47.9	0.5	700	450	10	-	-	-	-	-	-	-	-
		Y	1.3	60.5	38.1	0.47	700	470	10	-	-	-	-	-	-	-	-
		Z	2.1	50.6	47.1	0.5	700	470	10	-	-	-	-	-	-	-	-
B	2014	X	<0.1	75.8	24.1	0.41	500	480	20	5.5	0.013	32	7.4	0.01	76	5.6	32
		Y	<0.1	75.5	24.5	0.36	<500	490	20	5.6	0.017	31	7.3	0.02	74	5.6	32
		Z	0.4	72.4	27	0.38	<500	510	20	5.8	0.018	32	7.5	0.02	76	5.6	32
C	2014	X	2.1	70.1	27.9	0.54	600	340	10	4.9	0.022	46	9.1	0.02	67	6.9	36
		Y	0.4	74	25.7	0.51	900	390	10	5.6	0.022	53	9.5	0.02	75	6.8	39
		Z	0.4	73.2	26.3	0.56	700	380	10	5.7	0.023	54	9.3	0.02	76	6.7	38
	2015	X	4.2	68.1	27.7	0.43	700	410	10	-	-	-	-	-	-	-	-
		Y	3.3	71.1	25.5	0.44	1000	440	10	-	-	-	-	-	-	-	-
		Z	1.2	72.5	26.4	0.39	700	380	10	-	-	-	-	-	-	-	-
	2016	X	0.8	56.9	42.1	0.52	700	360	20	-	-	-	-	-	-	-	-
		Y	1.6	62.3	36.2	0.51	700	430	20	-	-	-	-	-	-	-	-
		Z	1	64.4	34.6	0.49	700	400	20	-	-	-	-	-	-	-	-
D	2014	X	0.4	41.8	57.6	0.63	700	510	10	6.7	0.03	59	10.7	0.02	95	7.1	42
		Y	1.7	53.8	44.5	0.6	700	560	10	6.2	0.025	59	10.3	0.02	97	7	41
		Z	4.2	47.8	48.1	0.63	700	520	10	6.1	0.024	57	10.2	0.02	94	6.8	40
	2015	X	0.9	39.1	60	0.64	1000	640	5	-	-	-	-	-	-	-	-
		Y	0.2	37.2	62.6	0.66	900	620	5	-	-	-	-	-	-	-	-
		Z	0.7	33.9	65.3	0.72	1000	650	5	-	-	-	-	-	-	-	-
	2016	X	0.4	35.8	63.8	0.98	1100	550	10	-	-	-	-	-	-	-	-
		Y	0.2	31.4	68.4	0.89	900	580	10	-	-	-	-	-	-	-	-
		Z	<0.1	31.7	68.2	1.08	1100	600	10	-	-	-	-	-	-	-	-
E	2019	X	0.2	89.6	10.2	0.2	<500	330	30.0 (20 to 35)	5	0.015	36	5.9	<0.02	54	5.4	33
		Y	1.8	87.8	10.4	0.16	<500	300	31.7 (25 to 40)	4.4	0.016	33	5.4	<0.02	49	5.1	30
		Z	0.8	89.5	9.6	0.16	<500	300	27.5 (15 to 40)	4.1	0.012	30	4.8	<0.02	45	5	29
	2020	X	0.8	89.7	9.4	0.17	<500	290	30.7 (25 to 35)	4.2	0.012	32	5.5	<0.02	57	5	32
		Y	0.6	89.9	9.6	0.2	<500	280	27.0 (25 to 28)	4.2	0.013	33	5.6	<0.02	61	5.2	35
		Z	0.7	88.7	10.6	0.18	<500	270	30.0 (24 to 35)	4.3	0.015	32	5.6	<0.02	57	5.3	34
	2021	X	<0.1	88	11.9	0.2	<500	300	24.7 (22 to 27)	4.8	0.012	36	6.3	<0.02	60	5.3	34
		Y	0.2	90.6	9.2	0.17	<500	300	14.3 (10 to 18)	4.7	0.015	34	5.7	<0.02	55	4.9	33
		Z	0.2	90.1	9.7	0.17	<500	310	29.8 (28 to 33)	4.6	0.015	37	6	<0.02	59	5.3	34
									DGV	20	1.5	80	65	0.15	21	50	200
									GV-high	70	10	370	270	1	52	220	410

Raw data for earlier surveys summarised in the main text is provided in the relevant reports by Cawthron (Gillespie et al. 2007; Gillespie et al. 2012), with 2001 data summarised in Robertson et al. (2002).

Appendix 6. Macrofauna data 2001-2021

Data are summed across cores within each site, hence site abundances depend on replication (described in main report Table 2). Raw core data have been provided electronically to NCC and TDC.

Main group	Taxa	Habitat	EG	01-A	01-B	01-C	01-D	06-A	06-B	06-C	06-D	11-A	11-C	14-A	14-B	14-C	14-D	15-A	15-C	15-D	16-A	16-C	16-D	19-E	20-E	21-E
Amphipoda	Amphipoda	Infauna	II	20	13	23	9																			
Amphipoda	Amphipoda A	Infauna	II					7	1	4	2	1														
Amphipoda	Amphipoda B	Infauna	II					9	4	12	27															
Amphipoda	Amphipoda sp. 5	Infauna	II																					2	1	
Amphipoda	Amphipoda sp. 6	Infauna	II																					2		
Amphipoda	Aoridae	Infauna	II									1														
Amphipoda	Corophiidae	Infauna	III									1	3													
Amphipoda	Melitidae	Infauna	II									1														
Amphipoda	Paracalliope novizealandiae	Infauna	II											3		3	3			2	2		1		1	1
Amphipoda	Phoxocephalidae	Infauna	II									3														
Amphipoda	Torridoharpinia hurleyi	Infauna	II											8	5	6	15	2	2	2	5	4	5	2	8	10
Anthozoa	Anthopleura aureoradiata	Epibiota	III	13	2	7	29	7		5	16	2	7	5		7	15	1	13	14	5	18	11	12	8	4
Anthozoa	Edwardsia sp.	Epibiota	II					2	1		1															
Anthozoa	Edwardsia sp. 1	Epibiota	II												3											2
Bivalvia	Arthritica bifurca	Infauna	IV	171	50	44	66	162	17	114	17	43	28													
Bivalvia	Arthritica sp. 1	Infauna	IV											85	4	155	3	16	58	11	175	115	36	1		6
Bivalvia	Austrovenus stutchburyi	Infauna	II	69	94	303	197	73	52	216	88	12	63	49	24	181	67	47	150	94	92	229	99	28	34	45
Bivalvia	Crassostrea gigas	Infauna	III										1													
Bivalvia	Cyclomactra tristis	Infauna	I															2								
Bivalvia	Hiatula spp.	Infauna	I		1	3				1										2				19	38	1
Bivalvia	Linucula hartvigiana	Infauna	II	6	2	32	10		1	10	3	15	15	2		13			9	4	1	2		3	1	3
Bivalvia	Macomona liliana	Infauna	II	35	16	31	30	6	4	21	13	4	3	9	13	10	4	3	7	16	7	11	27	100	67	55
Bivalvia	Mytilidae sp. 1	Infauna	III																1			1				
Bivalvia	Paphies australis	Infauna	II	1		1		1																	1	
Cirripedia	Austrominius modestus	Epibiota	NA						1		1		124		2	9	4		25		21	140		77	70	1
Copepoda	Copepoda	Infauna	II					1		3																
Copepoda	Copepoda sp. 2	Infauna	II																			1		2	1	1
Cumacea	Cumacea	Infauna	I							10	1															
Decapoda	Austrohelice crassa	Infauna	V						6						3											
Decapoda	Callianassa filholi	Infauna	I							1																
Decapoda	Decapod megalopa	Larva	NA										1				1		1							
Decapoda	Halicarcinus varius	Infauna	III																	7			1			
Decapoda	Halicarcinus whitei	Infauna	III	6			3	1						1		1				10		1	17	2	4	10
Decapoda	Hemiplax hirtipes	Infauna	V	3	3	2		13		14	14	4	3	4	6	6	1	10	9	10	8	20	7	3		3
Decapoda	Natantia unid.	Infauna	I			12																				
Decapoda	Palaemon affinis	Infauna	I																				2			
Diptera	Diptera	Larva	II		4																					
Diptera	Diptera sp. 1	Larva	II												10		1									
Diptera	Dolichopodidae larvae	Larva	II						5																	
Gastropoda	Amphibola crenata	Epibiota	III	24	1	6	4	5	1													1		2		
Gastropoda	Cominella glandiformis	Epibiota	III	2	3		3	13	3	6	10				1			1		3	1	2	4	1	1	2
Gastropoda	Diloma subrostratum	Epibiota	II	1		3	8										1			1		1		1	2	
Gastropoda	Diloma zelandicum	Epibiota	NA					2																		
Gastropoda	Epitonium tenellum	Epibiota	I													1								1		
Gastropoda	Micrelenchus tenebrosus	Epibiota	I				1			1			1													
Gastropoda	Notoacmea spp.	Epibiota	II			1	2	7		6			5			6	3		5	3		1				1
Gastropoda	Papawera zelandiae	Epibiota	I		1															1						1

Main group	Taxa	Habitat	EG	01-A	01-B	01-C	01-D	06-A	06-B	06-C	06-D	11-A	11-C	14-A	14-B	14-C	14-D	15-A	15-C	15-D	16-A	16-C	16-D	19-E	20-E	21-E
Gastropoda	Potamopyrgus estuarinus	Epibiota	III	155	5		9				8	66														
Gastropoda	Zeacumantus lutulentus	Epibiota	II		2	3	3		3	10	1		5									1		6	6	3
Gastropoda	Zeacumantus subcarinatus	Epibiota	I			3				2	5															
Holothuroidea	Taeniogyrus dendyi	Infauna	I																						7	4
Insecta	Hudsonema amabile	Larva	NA									1														
Insecta	Orthocladiinae	Larva	NA									4														
Isopoda	Isocladus sp.	Infauna	II																						1	
Isopoda	Isopoda Anthuroidea	Infauna	NA	2							2															
Mysidacea	Mysidacea	Infauna	I						3		6															
Nematoda	Nematoda	Infauna	II						2		5		1													
Nemertea	Nemertea	Infauna	III	5	13	19	11	2	1	6			2													
Nemertea	Nemertea sp. 1	Infauna	III													3	2			2		1	1	1	4	5
Nemertea	Nemertea sp. 2	Infauna	III																					1		
Nemertea	Nemertea sp. 3	Infauna	III																1				1	12	2	21
Nemertea	Nemertea sp. 5	Infauna	III																					1		
Oligochaeta	Oligochaeta	Infauna	III					1				52	5													
Polychaeta	Aonides sp.	Infauna	I		2	29	12				3															
Polychaeta	Aonides trifida	Infauna	I																						8	3
Polychaeta	Armandia maculata	Infauna	II		1		1																			
Polychaeta	Axiothella serrata	Infauna	II												5		10			3						
Polychaeta	Boccardia acus	Infauna	II									3	5	2				27	11	18	35	2	24			3
Polychaeta	Boccardia sp.	Infauna	II	7	1		12																			
Polychaeta	Boccardia syrtis	Infauna	II											6	1	10	2		3	10		3	4	1	1	15
Polychaeta	Capitella capitata	Infauna	IV				2						2													
Polychaeta	Disconatis accolus	Infauna	I												4		3									
Polychaeta	Eulalia microphylla	Infauna	II		1		1		1																	
Polychaeta	Glycera lamelliformis	Infauna	III											2		6	3			1			1			6
Polychaeta	Glyceridae	Infauna	III	1			1	1				15	20													
Polychaeta	Heteromastus filiformis	Infauna	III	155	34	111	214	30	1	49	239		56	60	18	42	90	5	6	13		16	48	7	1	83
Polychaeta	Lagis australis	Infauna	III		1		5					1	1													
Polychaeta	Leodamas cylindrifer	Infauna	I			3																				
Polychaeta	Magelona dakini	Infauna	III																					1		
Polychaeta	Magelona papillicornis	Infauna	III			2																				
Polychaeta	Maldanidae	Infauna	I	3	1		14		1		5															
Polychaeta	Nereididae (juv)	Infauna Juv	NA	62	37	35	27					36	24	41	25	34	33	78	31	8	40	34	20	29	12	30
Polychaeta	Nicon aestuariensis	Infauna	III				1	145	11	107	67	13	7	16	2	10	4	29	15	9	4	4	1			5
Polychaeta	Orbinia papillosa	Infauna	I																						7	2
Polychaeta	Owenia petersenae	Infauna	II										1													2
Polychaeta	Paradoneis sp.	Infauna	III													1							1			
Polychaeta	Paraonidae	Infauna	III						1																	
Polychaeta	Perinereis vallata	Infauna	II																				4	1		
Polychaeta	Phyllodocidae sp. 1	Infauna	II																	1						
Polychaeta	Polydora sp.	Infauna	III										1													
Polychaeta	Prionospio aucklandica	Infauna	II									21	34	36	6	66	56	1	10	134	2	39	157	110	76	139
Polychaeta	Prionospio sp.	Infauna	II	97	33	190	114	85		190	131															
Polychaeta	Scolecolepides benhami	Infauna	IV					8	9	3	2	4		9	14	5	3	3	9	9	6	5	7	3		4
Polychaeta	Scolecolepides sp.	Infauna	IV		14																					
Polychaeta	Sphaerosyllis hirsuta	Infauna	II			1																				
Sipuncula	Sipuncula	Infauna	II	8	7	6	4																			

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 Waimea 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 Waimea 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.

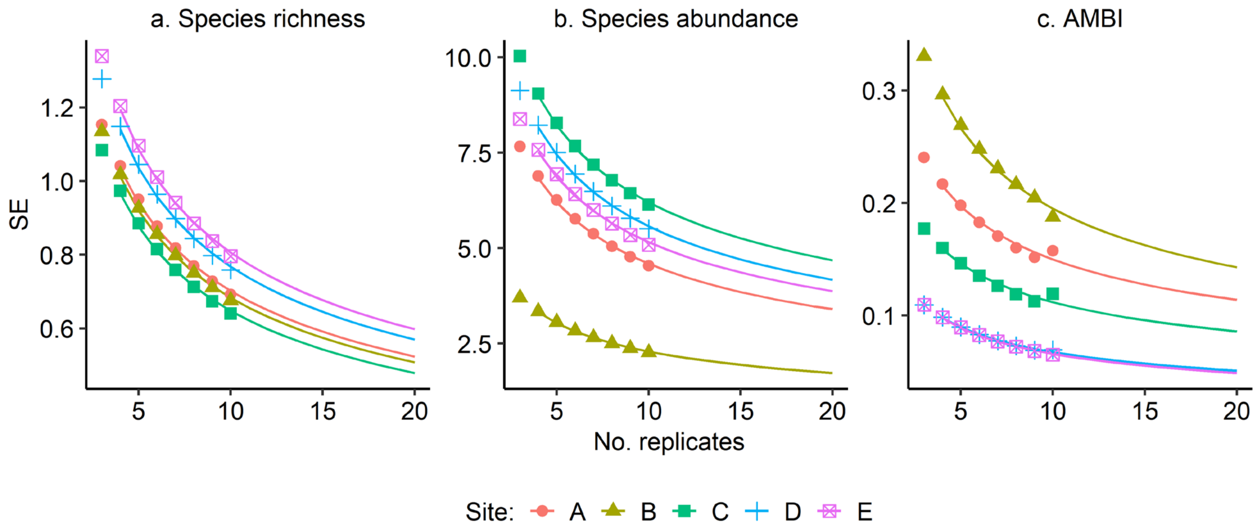


Figure A7.1. Standard error (SE) for Waimea Inlet species richness sample means plotted against the number of replicates, coloured by site. The markers show the SE of observed data, and the lines are simple power curve extrapolations. Note the differing scale of the y-axis, where SEs for species abundance (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 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 ~30%, 47% and 18% 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).

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 79% 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 77%, while increasing to 12 cores would increase it to 82%. However, 20 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 A7.1. 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 2021 and we took another 10 reps in 2022, a paired t-test for change in sample mean would suggest that an observed richness approximately less than 9.1 or greater than 16.9 (+/-29.7%), would not just be due to chance (alpha=0.05).

	No. reps								
Response	5	6	7	8	9	10	12	20*	58*
S	37.9	35.6	33.7	32.1	30.8	29.7	27.8	23.3	16.0
N	54.5	52.5	50.8	49.4	48.2	47.1	45.3	40.7	32.5
AMBI	26.0	23.6	21.8	20.3	19.1	18.1	16.5	12.6	7.3

* Note: the illustration of 20 and 58 reps was based on estimated species detection thresholds (of ~90% and 100%, respectively) described in Section A7.4.

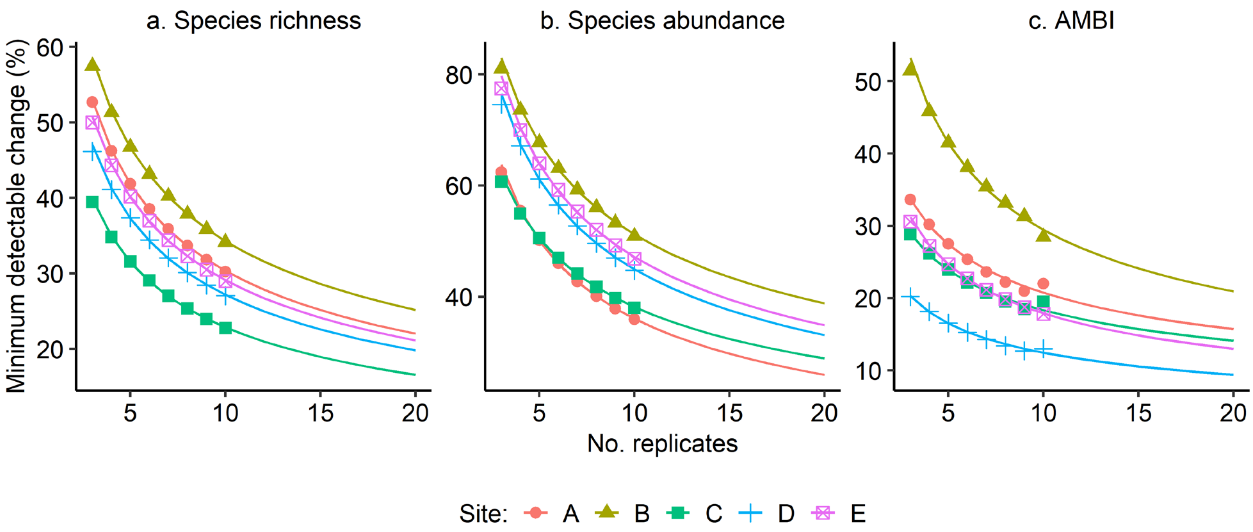


Figure A7.2. Minimum detectable change (%) in mean univariate responses plotted against the number of replicates each year, coloured by site. The markers show the detectable change (%) of observed data and the lines are simple power curve extrapolations. These data can be interpreted as minimum percentage 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.

Table A7.2. The average percentage of estimated total site richness captured over all sites and years at differing sampling effort. The columns showing 20 and 58* 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	20	58*
A	76.8	80.3	83.1	85.4	87.4	89.0	91.6	96.9	99.8 (38)
B	62.2	66.2	69.7	72.6	75.2	77.5	81.4	90.8	99.9 (56)
C	67.7	70.3	72.5	74.4	76.1	77.7	80.5	88.1	99.6 (69)
D	65.8	69.1	71.7	73.9	75.8	77.5	80.5	88.0	99.2 (60)
E	58.6	62.5	65.9	68.8	71.4	73.6	77.4	86.9	99.7 (65)
Average	66.0	70.0	73.0	75.0	77.0	79.0	82.0	90.1	99.6

* Note: an average of 58 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.

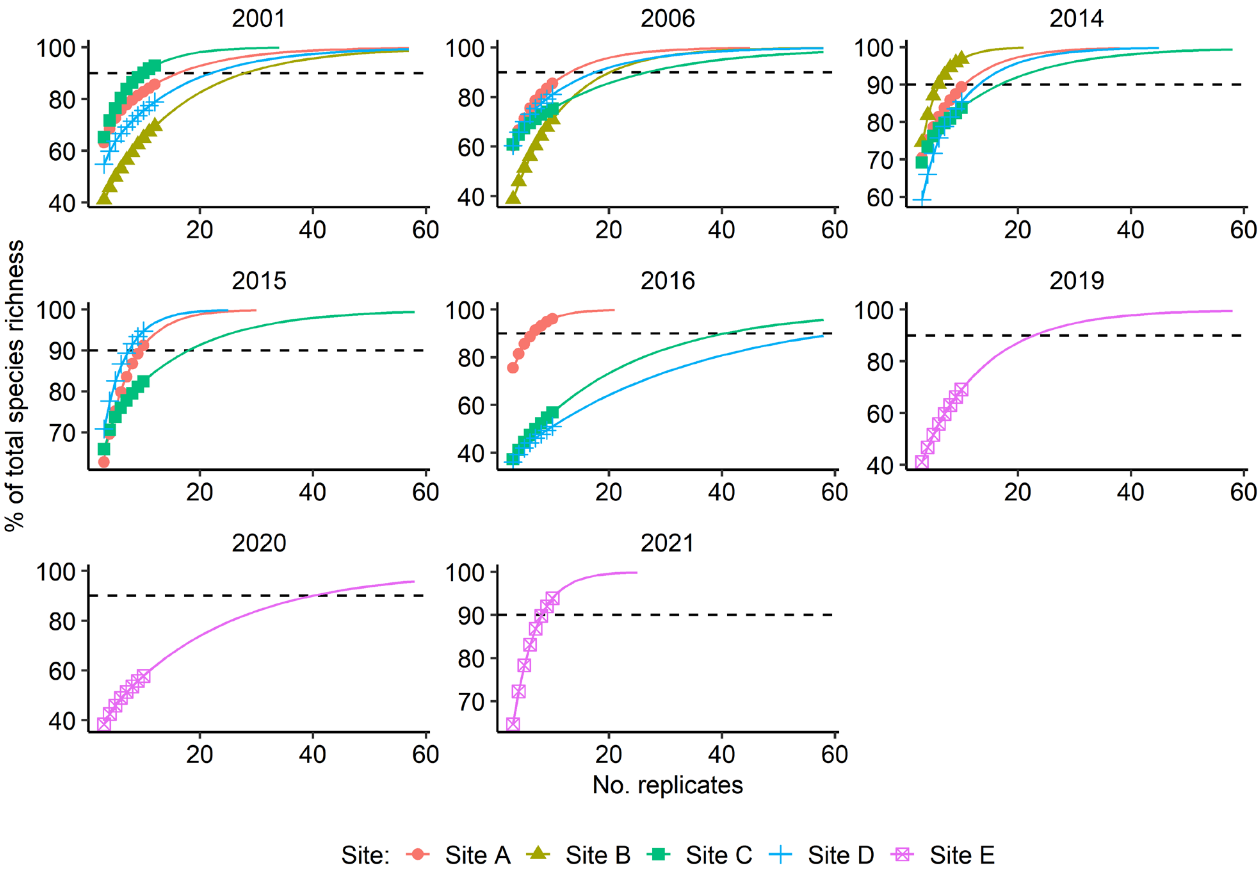


Figure A7.3. Percentage of total estimated richness at each site plotted against the number of replicates. Subplots correspond to sampling years. The points on the graph show % of total richness calculated from observed data and the lines are extrapolations towards the estimated 100% richness using the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

A7.5 References for Appendix 7

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