Impact of a Flood Event on the Sediment Quality and Ecology of Delaware (Wakapuaka) Inlet

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Cover and back photo: East Arm of Delaware (Wakapuaka) Inlet, April 2023.

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For the People
Mō ngā tāngata

Impact of a Flood Event on the Sediment Quality and Ecology of Delaware (Wakapuaka) Inlet

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for

Nelson City Council

October 2023

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For the environment
Mō te taiao

GLOSSARY

AMBI	AZTI Marine Biotic Index
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)
TN	Total Nitrogen
TOC	Total Organic Carbon
TP	Total phosphorus
Zn	Zinc

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SUMMARY

BACKGROUND

This report describes the findings of four ecological surveys, and associated sedimentation monitoring, conducted in Delaware (Wakapuaka) Inlet near Nelson since 2019. The report focuses on a comparison of three 'baseline' years (2019-2021) with investigations conducted following a regionally significant flood in August 2022. The ecological surveys have followed the fine scale methods described in New Zealand's National Estuary Monitoring Protocol (NEMP). Post-flood impacts were assessed in the context of estuary condition rating criteria (see Table below), and considerations for future investigations and event-based monitoring are discussed.

KEY FINDINGS

- The most conspicuous impact of the flood was the widespread deposition of muddy sediments across the estuary. At the three main monitoring sites, cumulative sedimentation over a period of 3-5 years has exceeded the guideline value of 2mm/yr for New Zealand estuaries. At the two sites closest to the main catchment freshwater input to the estuary, the accrual depth of muddy sediment immediately post-flood was ~20-30mm greater than the preceding baseline survey.
- Surface sediments at all monitoring sites had a greatly increased mud content post-flood. The change was most pronounced at Site C in the Cable Bay arm of the estuary, where sediment mud content increased from a baseline of ~20% pre-flood to >90% immediately post flood, but was still 38% (i.e., almost double the baseline value) around eight months post-flood in April 2023.
- Except for the increased sediment mud content, sediment quality did not otherwise greatly deteriorate due to the flood event, and in fact has not changed appreciably since an earlier survey conducted in 2009. Overall sediment quality, in terms of trophic state indicators and trace contaminants, remains high (condition ratings of 'good' or 'very good' in the Table below). Note that the elevated content of the trace metal nickel shown in the Table below is a regional phenomenon that reflects natural inputs due to catchment mineralogy.
- Given the increased sedimentation and sediment mud content associated with the flood, we expected to see concomitant changes in the biota. However, there was evidence for only a small decline in condition, and a small change in the composition of sediment-dwelling macrofauna, some of which at Site C (Cable Bay arm) appeared most likely attributable to the flood-related deposition of muddy sediment. However, key species such as cockles remain abundant at sites where they already occurred.


Pre-flood (left) vs post-flood (right) condition at Site C in the Cable Bay arm.

Notwithstanding the issue of muddy sediment inputs, the overall impression provided by the results and associated condition ratings is that Delaware Inlet is in a reasonably healthy condition. Any significant ecological impacts at the monitoring sites from the August 2022 flood did either not occur in the first instance, or the ecological communities have largely recovered between the time of the flood and the comprehensive survey eight months later. However, it is also noted that the monitoring sites were not located in the parts of the estuary worst-affected by muddy sediment deposition. The implications for future event-based monitoring are discussed, with key challenges being council resourcing (staff and budget) constraints, and the absence of baseline data from the most susceptible parts of estuaries. A suggested way forward is proposed, which is reflected in the recommendations below.

RECOMMENDATIONS

On the basis of the findings and discussion in this report, recommendations for Delaware Inlet are as follows:

- Evaluate and maintain records of catchment land use changes to determine current and potential future sediment sources to the estuary, and investigate options to reduce inputs. As part of this work, consider whether the extent of land slips following the August 2022 flood is linked to particular land use types, and evaluate the benefits of actively planting priority slip areas.
- Continue annual sediment plate monitoring and sediment grain size analysis, to track recovery from flood-related impacts.
- Develop and implement a comprehensive estuary-wide programme for assessing sedimentation and sediment grain size, building on the programme already put in place by NCC, and potentially expanding to include broad-scale sedimentation mapping technologies (e.g., LiDAR).
- Undertake comprehensive NEMP fine scale ecological and sediment quality monitoring at an ongoing minimum of five-yearly intervals, using the current methods.
- Develop an event-based monitoring protocol and response pathway to address lags in data collection, so that worst-case effects are captured.

Summary of condition scores of ecological health for fine scale monitoring sites, based on mean values of key indicators, and rating criteria in Table 3. Sedimentation rate (Sed rate) is the average annual rate since the baseline year. TP not rated. See Glossary for definition of indicators.

Site	Year	Sed rate	Mud	aRPD	TN	TP	TOC	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn	AMBI
		mm/yr	%	mm	mg/kg	mg/kg	%	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	na
A	2019	-	84.6	20	933	613	0.85	6.2	0.032	54.0	17.3	0.01*	24.7	5.9	53.7	1.5
	2020	-	84.2	10	800	677	0.82	5.8	0.036	55.0	19.3	0.02	26.7	6.2	59.7	1.6
	2021	-	82.1	10	733	563	0.74	5.5	0.032	51.3	16.6	0.01*	23.0	5.8	50.7	1.8
	2023	6.3	87.3	16	933	583	0.99	6.5	0.047	54.0	24.0	0.03	24.0	7.0	55.7	2.2
C	2019	-	21.0	29	< 500	567	0.18	4.8	0.023	53.7	10.0	< 0.02	20.0	3.3	50.3	2.4
	2020	-	20.7	32	< 500	640	0.23	5.0	0.025	54.0	12.4	< 0.02	23.0	3.8	59.3	1.9
	2021	-	18.3	29	< 500	543	0.21	5.0	0.023	53.0	10.8	< 0.02	19.7	3.7	51.0	2.0
	2023	2.7	38.4	22	450*	587	0.44	6.2	0.036	50.3	16.8	0.01*	19.9	5.5	53.0	2.7
D	2020	-	4.0	24	< 500	640	0.16	4.7	0.020	53.3	10.7	< 0.02	23.3	2.7	56.7	1.4
	2021	-	4.7	28	< 500	543	0.12	4.6	0.020	52.0	9.2	< 0.02	19.5	2.6	47.3	1.3
	2023	4.9	8.8	36	< 500	600	0.11*	5.1	0.022	54.7	10.7	< 0.02	20.4	3.0	49.3	1.7

** Sample mean includes values below lab detection limits*

< All values below lab detection limit

Very Good	Good	Fair	Poor

1. INTRODUCTION

Estuary monitoring is undertaken by most councils in New Zealand as part of their State of the Environment (SOE) programmes. Many monitoring programmes focus on the effects of catchment derived muddy sediment, which is regarded as one of the key drivers of ecological health in New Zealand estuaries (Cummings et al. 2003; Robertson et al. 2015; Berthelsen et al. 2018; Clark et al. 2021). The most widely-used monitoring framework is that outlined in New Zealand's National Estuary Monitoring Protocol (NEMP; Robertson et al. 2002). 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, and coupled with monitoring to track changes in sediment mud content.

Nelson City Council's (NCC's) estuarine monitoring to date has included NEMP surveys, as well as sediment plate monitoring, in all of the main estuaries in the region. In Wakapuaka/Delaware Inlet (hereafter Delaware Inlet; Fig. 1), the most recent broad scale monitoring was undertaken in 2018 (Stevens & Forrest 2019), with three fine scale surveys undertaken in 2019, 2020, and 2021 (Forrest & Stevens 2021), which built on an earlier survey undertaken by Cawthron Institute in 2009 (Gillespie et al. 2009). The more recent fine scale monitoring is supported by sediment plate measurements that were periodically made between March 2018 and September 2022.

The September 2022 sediment plate monitoring was conducted following a major flood event in Nelson in August, and revealed significant deposition of muddy sediments (>30cm deep in places) in parts of Delaware Inlet (Stevens & Roberts 2022). Due to the observed physical impacts of the flood event, NCC had concerns regarding the ecological implications. To this end, it was recognised that the 2019-2021 fine scale monitoring provided a potentially useful baseline against which post-flood ecological effects could be assessed.

This report describes a post-flood assessment of the fine scale sites, which was conducted for NCC as part of an Envirolink Medium Advice Grant in partnership with the University of Waikato. The ecological condition of the fine scale sites post-flood is compared with conditions during the 2019-2021 surveys. Management implications and ongoing needs for event-based monitoring are discussed.

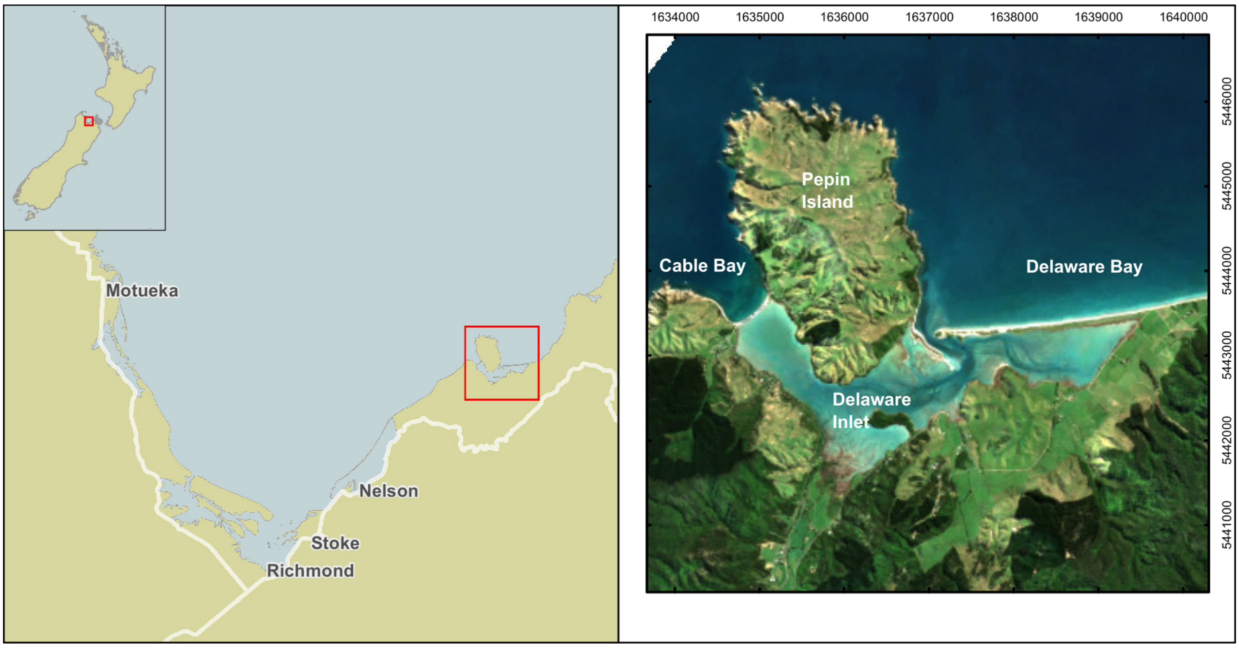


Fig. 1. Location of Delaware Inlet.

2. BACKGROUND TO DELAWARE INLET AND THE AUGUST 2022 FLOOD

Delaware Inlet covers an area of 355ha, and is classified as a shallow intertidal-dominated tidal lagoon type estuary (Plew et al. 2018). It is well-flushed and seawater-dominated, with a single tidal opening east of Pepin Island, and extensive intertidal arms located to the west near Cable Bay and to the east near Delaware Bay (Fig. 1). Monitoring conducted up to 2021 showed that estuary substrates were dominated by sandy sediments in the eastern arm, with muddier sediments in western and southern areas. Within these broad sediment types the estuary contains complex intertidal habitats with a variety of other substrates, including cobble, gravel, oyster reef, and shell banks, and a moderate cover of both salt marsh and seagrass (Gillespie et al. 2011; Stevens & Forrest 2019; Forrest & Stevens 2021).

Delaware Inlet at low tide viewed towards the northeast, with the western Cable Bay arm to the left.

The surrounding catchment of ~8,515ha is relatively steep, with land cover data (LCDB5 2018) revealing that it is extensively modified, with 34.8% in exotic plantation forest (standing plus harvested) and 18.5% being in pastoral land uses (Table 1). Despite the modification of the catchment, Delaware Inlet has been previously described as a 'relatively pristine' high-value estuary, which is considered an important nursery area for marine and freshwater fish, and birds (Gillespie 2009; Gillespie et al. 2011). Gillespie (2009) considered the relatively natural functional qualities of the Inlet, as historically described by Gillespie & MacKenzie (1981), to have been largely maintained.

Nonetheless, muddy sediment inputs from catchment runoff have been identified as the most significant ongoing issue for the estuary. The findings of most recent fine scale (Forrest & Stevens 2021) and broad scale (Stevens & Forrest 2019) surveys showed elevated sedimentation rates in the southern estuary near the Wakapuaka River, along with an increased prevalence of soft, muddy sediment.

Muddy sediment deposition was greatly exacerbated by the August 2022 flood. That event was one of the most significant on record, with more than one metre of rain falling over four days, leading to a state of emergency being declared in Nelson. The event resulted in more than 550 landslips across the region, including many in the Delaware Inlet catchment. Sediment plate monitoring and a synoptic estuary-wide assessment conducted 2-weeks post flood showed very high levels of muddy sediment deposition in the eastern, western and southern sections of the estuary, including in the general vicinity of the fine scale and sediment plate sites (Stevens & Roberts 2022). This situation provided a unique opportunity to consider sediment accumulation and its effects in the context of the pre-flood baseline.

Table 1. Catchment land use area and % based on LCDB5 (2018).

LCDB class and name		Ha	%
1	Built-up Area (settlement)	5.0	0.1
5	Transport Infrastructure	16.0	0.2
6	Surface Mine or Dump	3.8	0.0
10	Sand or Gravel	2.5	0.0
12	Landslide	1.8	0.0
16	Gravel and Rock	2.5	0.0
40	High Producing Exotic Grassland	1437.5	16.9
41	Low Producing Grassland	135.7	1.6
46	Herbaceous Saline Vegetation	26.0	0.3
50	Fernland	18.6	0.2
51	Gorse and/or Broom	428.1	5.0
52	Manuka and/or Kanuka	594.3	7.0
54	Broadleaved Indigenous Hardwoods	308.3	3.6
56	Mixed Exotic Shrubland	13.8	0.2
64	Forest - Harvested	140.9	1.7
68	Deciduous Hardwoods	4.7	0.1
69	Indigenous Forest	2237.2	26.3
71	Exotic Forest	2816.3	33.1
Grand Total		8515	100.0
Total densely vegetated area[1]		6588	77.4

1. LCDB5 classes 45-71

3. SURVEY METHODS

The survey methods are detailed in Appendix 1 and summarised below.

3.1 FINE SCALE AND SEDIMENT PLATE SITES

The initial fine scale survey by Gillespie et al. (2009) established three fine scale monitoring sites (A-C) shown in Fig. 2. In April 2019, sediment plates were installed at each of these sites, and at two additional sites in the upper east (BS) and western (CS) estuary. The fine scale sites were initially chosen to be representative of the range of substrates present across Delaware Inlet. Subsequent surveys revealed that Site B was ecologically impoverished due to its location in an area of shifting mobile sand (Forrest & Stevens 2019). As such, monitoring at Site B was discontinued in 2019, and replaced with Site D, where monitoring began in January 2020. Fig. 2 shows a schematic of the layout and sampling approach for fine scale and sediment plate monitoring. Appendix 2 provides GPS positions and other location information.

3.2 SEDIMENT PLATES

Sediment plates consisted of concrete pavers (19cm x 23cm). On 27 March 2018, four plates were installed along a 30m transect at each of the sites shown in Fig. 2, except for Site D where plates were installed on 8 January 2020.

At the time of each measurement (see Appendix 1), a single composite sediment sample was collected from next to the sediment plates, and sent to Hill Laboratories for particle grain size analysis (mud, sand and gravel fractions; see Table A2 of Appendix 1). As the sediment plate measurements are ongoing and undertaken at least annually, the grain size measurements provide a simple means of tracking ongoing changes in sediment muddiness.

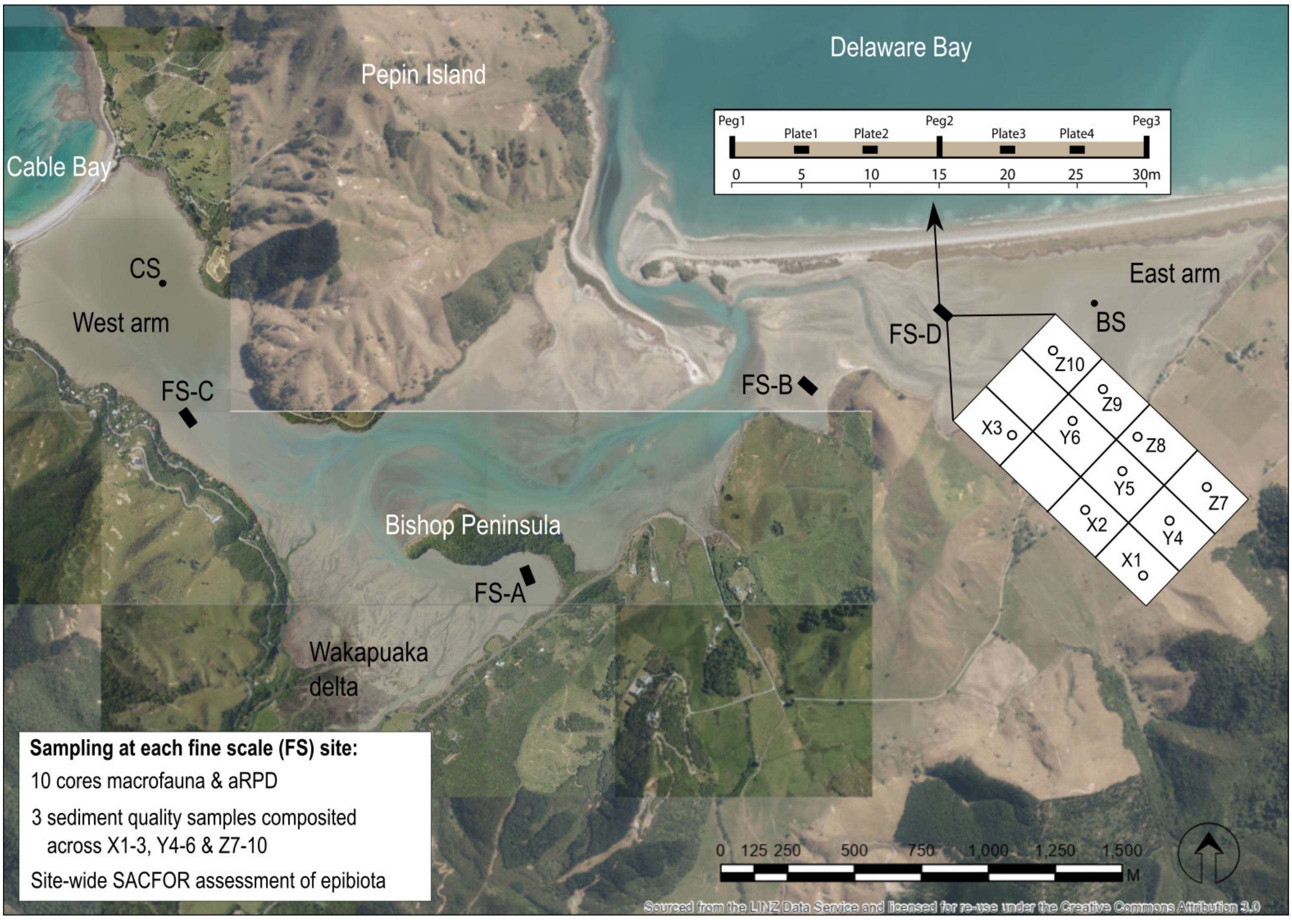


Fig. 2. Location of sites used for ongoing fine scale (FS) monitoring (A, C, D) and additional sites used for ongoing monitoring of sediment plates only (B, BS, CS). Note that Site B was discontinued as a fine scale site in 2019, and substituted with Site D. However, Site B has been retained for sediment plate monitoring. The schematic depicts the sediment core sample collection, and the sediment plate measurements. Appendix 1 provides sampling design and method details. Appendix 2 provides GPS positions and other location information.

3.3 FINE SCALE SAMPLING AND INDICATORS

As depicted in Fig. 2, each fine scale site was divided into a 3 x 4 grid of 12 plots, with sampling conducted in 10 of these plots. A summary of the NEMP indicators, the rationale for their inclusion, and the field sampling methods, is provided in Table 2. Although the general sampling approach closely follows the original NEMP, several alterations and additions to early NEMP methods have been introduced over the last 10 or more years, including for the Delaware Inlet surveys. The key sampling elements are summarised below.

Sediment quality: NEMP Indicators included sediment mud content, oxygenation status (measured as the apparent Redox Potential Discontinuity depth; aRPD), nutrients and organic content, and selected trace contaminants. Sediment aRPD was measured in the field. For the other variables, three composite samples (each composited from 3-4 sub-samples) were collected, and sent to Hill Laboratories for analysis.

Where sediment quality results included values less than laboratory method detection limits, half of the detection limit value was used for data averaging, according to standard convention.

Table 2. Summary of fine scale indicators, rationale for their use, and sampling method. The main departures from the NEMP are described in footnotes.

Indicator	General rationale	Sampling method
Physical and chemical		
Sediment grain size	Indicates the relative proportion of fine-grained sediments that have accumulated.	Composited surface scrape to 20mm sediment depth.
Nutrients (nitrogen and phosphorus), organic matter & total sulfur	Reflects the enrichment status of the estuary and potential for algal blooms and other symptoms of enrichment.	Surface scrape to 20mm sediment depth. Organic matter measured as Total Organic Carbon (TOC) (note 1).
Trace elements (arsenic copper, chromium, cadmium, lead, mercury, nickel, zinc)	Common toxic contaminants generally associated with human activities. High concentrations may indicate a need to investigate other anthropogenic inputs, e.g., pesticides, hydrocarbons.	Surface scrape to 20mm sediment depth (note 2).
Substrate oxygenation (apparent Redox Potential Discontinuity depth; aRPD)	Measures the enrichment/trophic state of sediments according to the depth of the aRPD. This is the visual transition between brown oxygenated surface sediments and deeper less oxygenated black sediments. The aRPD can occur closer to the sediment surface as organic matter loading or sediment mud content increase.	Sediment core, split vertically, with average depth of aRPD recorded in the field where visible.
Biological		
Macrofauna	Abundance, composition and diversity of infauna living with the sediment are commonly-used indicators of estuarine health.	130mm diameter sediment core to 150mm depth (0.013m^2 sample area, 2L core volume), sieved to 0.5mm to retain macrofauna.
Epibiota (epifauna)	Abundance, composition and diversity of epifauna are commonly-used indicators of estuarine health.	Abundance based on SACFOR in Appendix 1, Table B3 (note 3).
Epibiota (macroalgae)	The composition and prevalence of macroalgae are indicators of nutrient enrichment.	Percent cover based on SACFOR in Appendix 1, Table B3 (note 3).
Epibiota (microalgae)	The prevalence of microalgae is an indicator of nutrient enrichment.	Visual assessment of conspicuous growths based on SACFOR in Appendix 1, Table B3 (notes 3, 4).

[1] Since the NEMP was published, Total Organic Carbon (TOC) has become available as a routine low-cost analysis which provides a more direct and reliable measure than the NEMP recommendation of converting Ash Free Dry Weight (AFDW) to TOC.

[2] Arsenic and mercury are not specified in the NEMP, but can be included in the trace element suite by the analytical laboratory.

[3] Assessment of epifauna, macroalgae and microalgae uses the 'SACFOR' approach: S = super abundant, A = abundant, C = common, F = frequent, O = occasional, R = rare (see Appendix 1). SACFOR was used instead of the quadrat sampling outlined in the NEMP. 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 clumped or patchy distributions and the lack of demonstrated utility of microalgae as a routine indicator.

Biota: To characterise the dominant ecological features of the fine scale sites, we used qualitative field methods ('SACFOR'; see Appendix 1) to estimate the abundance or percent cover of conspicuous surface-dwelling snails, macroalgae, microalgae and seagrass. In addition, quantitative sampling was undertaken of macrofauna, which are small organisms that live within or on the sediment matrix.

Macrofauna were sampled using sediment cores (130mm diameter, 150mm deep, ~2L volume), which were sieved through a 0.5mm mesh to remove mud and sand. The composition of the sieved core samples in terms of macrofauna species (or higher taxa) and their abundance, was determined by Gary Stephenson at Coastal Marine Ecology Consultants. Quality assurance checks on voucher specimens were made by expert taxonomists at NIWA.

Macrofauna analyses included the following:

- Derivation of richness and abundance, which are simple measures that describe the number of different species present in a sample (i.e., richness), and total organism abundances, respectively.
- Calculation of 'AMBI' scores. The AMBI is an international biotic health index (Borja et al. 2000) whose calculation is based on the proportion of macrofauna falling into one of five eco-groups (EG) ranging from relatively sensitive species (EG-I) to relatively hardy ones (EG-V).
- Multivariate analysis methods were used to assess changes macrofauna community composition.
- Correlation-based univariate and multivariate approaches were used to relate macrofaunal composition to changes in sediment quality and sedimentation.

3.4 ASSESSMENT OF ESTUARY CONDITION

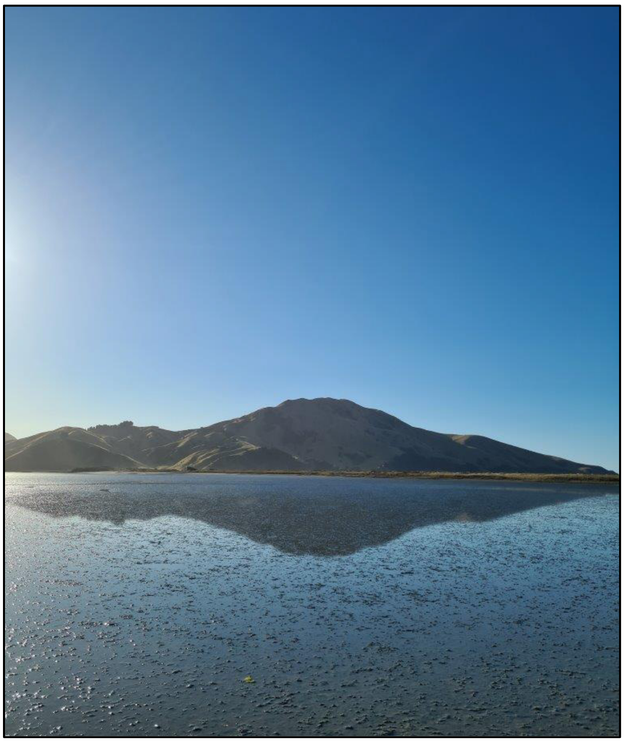
In addition to the authors' expert interpretation of the data, results are assessed against established or developing estuarine health metrics ('condition ratings'), drawing on approaches from New Zealand and overseas. These metrics assign different indicators to one of four colour-coded 'health status' bands shown in Table 3.

Collecting sediment macrofauna cores and measuring sediment plate depth at Site C.

Table 3. Condition ratings for assessing estuary health. See Glossary for definitions.

Indicator	Unit	Very good	Good	Fair	Poor
Sediment quality and macrofauna					
Mud content[1]	%	< 5	5 to < 10	10 to < 25	≥ 25
aRPD depth[2]	mm	≥ 50	20 to < 50	10 to < 20	< 10
TN[1]	mg/kg	< 250	250 to < 1000	1000 to < 2000	≥ 2000
TP		Requires development			
TOC[1]	%	< 0.5	0.5 to < 1	1 to < 2	≥ 2
TS		Requires development			
Macrofauna AMBI[1]	na	0 to 1.2	> 1.2 to 3.3	> 3.3 to 4.3	≥ 4.3
Sediment trace contaminants[3]					
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
Sedimentation					
Sedimentation rate[4]	mm/yr	< 0.5	≥0.5 to < 1	≥1 to < 2	≥ 2

1. Ratings from Robertson et al. (2016).
2. aRPD based on FGDC (2012).
3. 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.
4. Sedimentation rate adapted from Townsend and Lohrer (2015).


Delaware Inlet eastern flats


Cable Bay and western arm

4. KEY FINDINGS

4.1 GENERAL FEATURES OF FINE SCALE SITES

The fine scale sites are typical of the main habitats across Delaware Inlet. None of the sites have seagrass, although beds are present in the wider estuary. The sites range from very soft mud at Site A to relatively firm muddy sand at Site D. The sequence of photos on the next page shows the visual changes over the three baseline surveys (2019-2021), as well as site photos taken soon after the August 2022 flood event, and again in April 2023. The photos taken ~2 weeks post-flood show obvious muddy sediment deposition at all sites, but it was most conspicuous at Site C in the western arm.

4.2 SEDIMENT PLATES

Sedimentation rate

Sediment plate data collected up to April 2023 are summarised in Fig. 3 and Table 4. Raw data are provided in Appendix 3. Fig. 3 shows the mean change in sediment depth at the time of each survey, expressed as change since the baseline date of plate installation.

Sedimentation patterns across the sites have been highly variable. The cumulative sedimentation has exceeded the DGV of 2mm/yr at the three fine scale sites used for ongoing monitoring (A, C, D) but not at the other sites where sediment plate monitoring is undertaken (Table 4).

Fig. 3 shows a strong effect of the August 2022 flood at Sites A and C which are closest to the Wakapuaka River delta, with around 20-30mm of sediment deposition between January and September 2022. There was a weaker sedimentation effect at BS and CS (Fig. 3). Whereas sediment eroded from three of the sites after the initial flood deposition, there has been sediment retention or slight accrual at Sites A, D and B. Subsequent monitoring will help to ascertain ongoing post-flood changes. For example, it is conceivable that the sediment that accrued at Site A post-flood will not subsequently erode, as this site is clearly in a depositional area where soft-sediments accumulate.

Table 4. Sediment plate monitoring data summary.

Site	Interval (years)	Net change in April 2023 relative to baseline (mm)	Mean annual sedimentation (mm/yr)[1]
A	5.0	31.4	6.3
B	5.1	-6.5	-1.3
BS	5.1	6.4	1.3
C	5.0	13.6	2.7
CS	5.0	-0.3	-0.1
D	3.2	15.8	4.9

1. Sedimentation is the mean annual rate since the baseline year, which is an interval of ~5-yrs except at Site D.

Very Good	Good	Fair	Poor

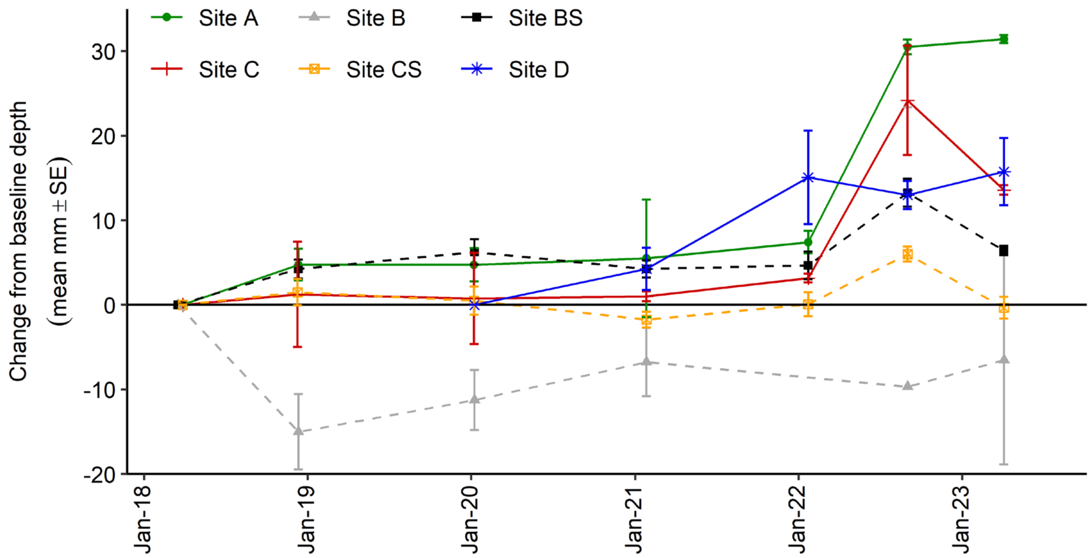
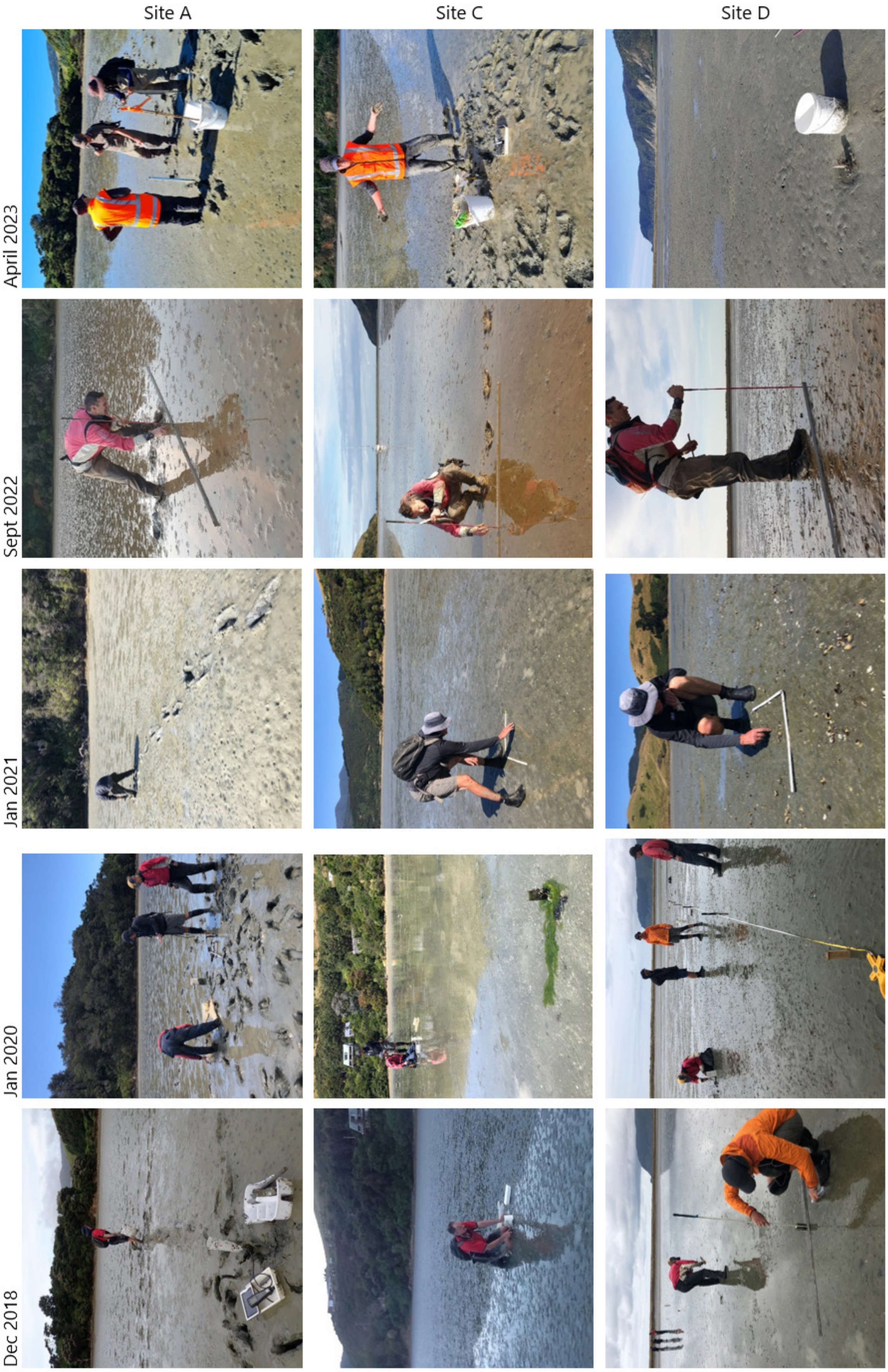


Fig. 3. Time series of mean change (± SE, n=4) in sediment depth over buried sediment plates since the baseline was established. As well as fine scale sites A, C and D, data are shown for Sites B, BS and CS, where additional sedimentation monitoring is undertaken. A conspicuous increase in sedimentation (i.e., increase in sediment depth) is evident due to the August 2022 flood.



Photos of fine scale sites over time, showing sediment inundation following the August 2022 flood.

Sediment plate grain size changes

The grain size data from the sediment plate monitoring sites show two main patterns (Fig. 4):

- A pronounced increase in sediment mud content at all sites from before to after the flood. The effect was particularly strong at the previously sand-dominated upper-estuary sites (BS, C, CS) but weaker at lower estuary sites closer to the main tidal channel.
- A very high mud content at Site A, compared to all other sites where sediments consist of muddy-sand (with the exception of the post-flood survey in September 2022). As noted above, Site A appears to be a depositional area for muddy sediment.

The magnitude of the mud increase was moderately correlated with sedimentation rate (Pearson r^2=0.50, p<0.01, Appendix 6). For example, there was a large increase in mud at Site C where the increased sedimentation due to the flood was particularly pronounced. These results highlight that the sedimentation was attributable to the deposition of catchment-derived sediment, rather than resuspension and redistribution of previously-deposited sediments. In April 2023, approximately eight-months after the flood, sediment mud content had abated at all sites by comparison with September 2022. However, mud levels at upper estuary Sites BS, C and CS were still clearly greater than the baseline values represented by the pre-flood data.

Muddy sediment inundation at Site C, two weeks post-flood.

In April 2023, almost eight months post-flood, sediments at Site C were still considerably muddier than the pre-flood baseline.

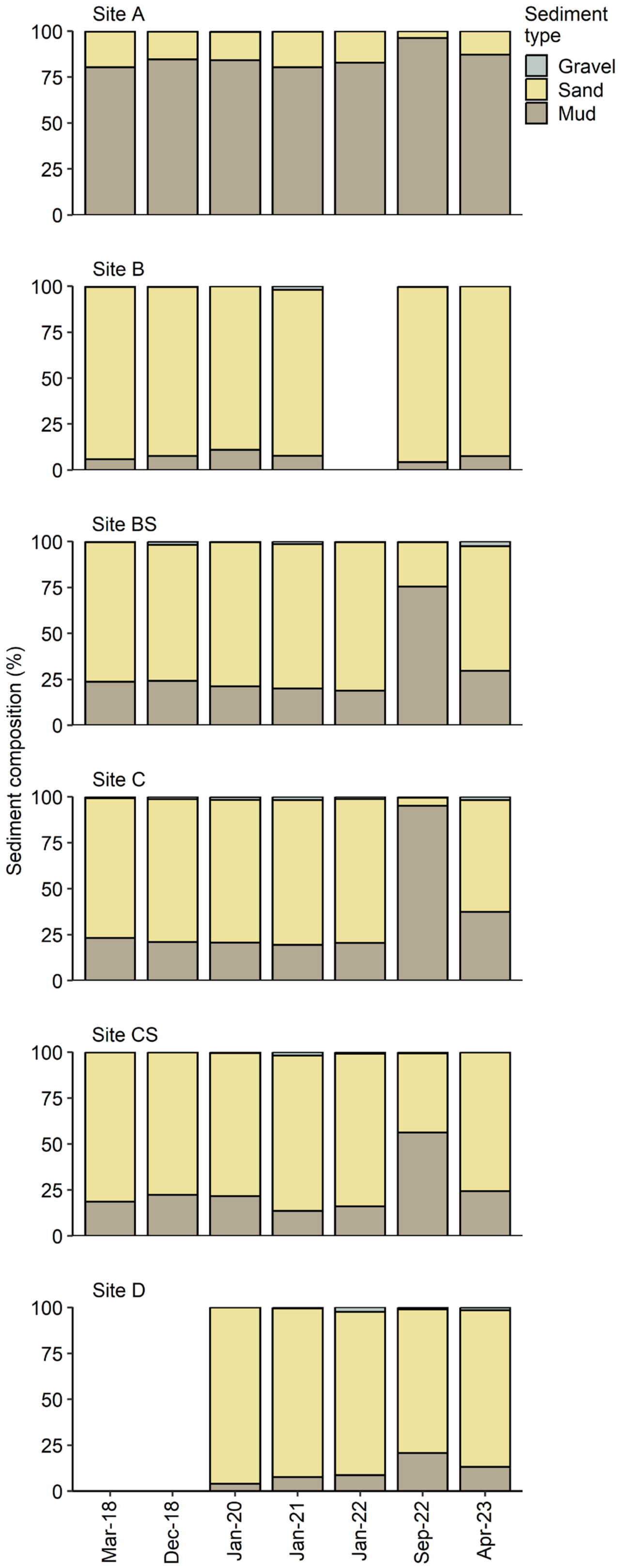


Fig. 4. Sediment particle grain size analysis showing percentage composition of mud (<63µm), sand (<2mm to ≥63µm) and gravel (≥2mm) from single composite samples collected next to sediment plates. Missing values are surveys where measurements were not made.

4.3 FINE SCALE SITES

4.3.1 Sediment grain size, TOC and nutrients

Composite sediment sample raw data for fine scale sites are tabulated in Appendix 4. Analyses of sediment grain size at these sites highlighted the same general patterns described above for the sediment plates. Although comprehensive sampling was not undertaken immediately post-flood, core photos taken at that time (see below) clearly show the deposits of catchment derived silt and clay that cover the sediment surface. In April 2023, eight-months post-flood, the mean mud content at fine scale sites C and D was still around twice that of the baseline (Fig. 5). The proportional difference in mud at Site A in April 2023 was less, reflecting that this site was already mud-dominated under baseline conditions.

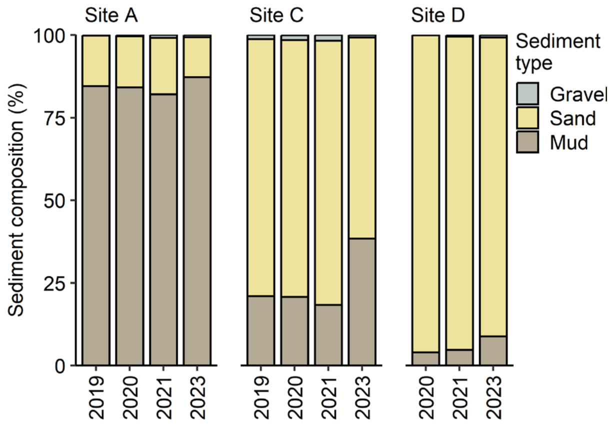
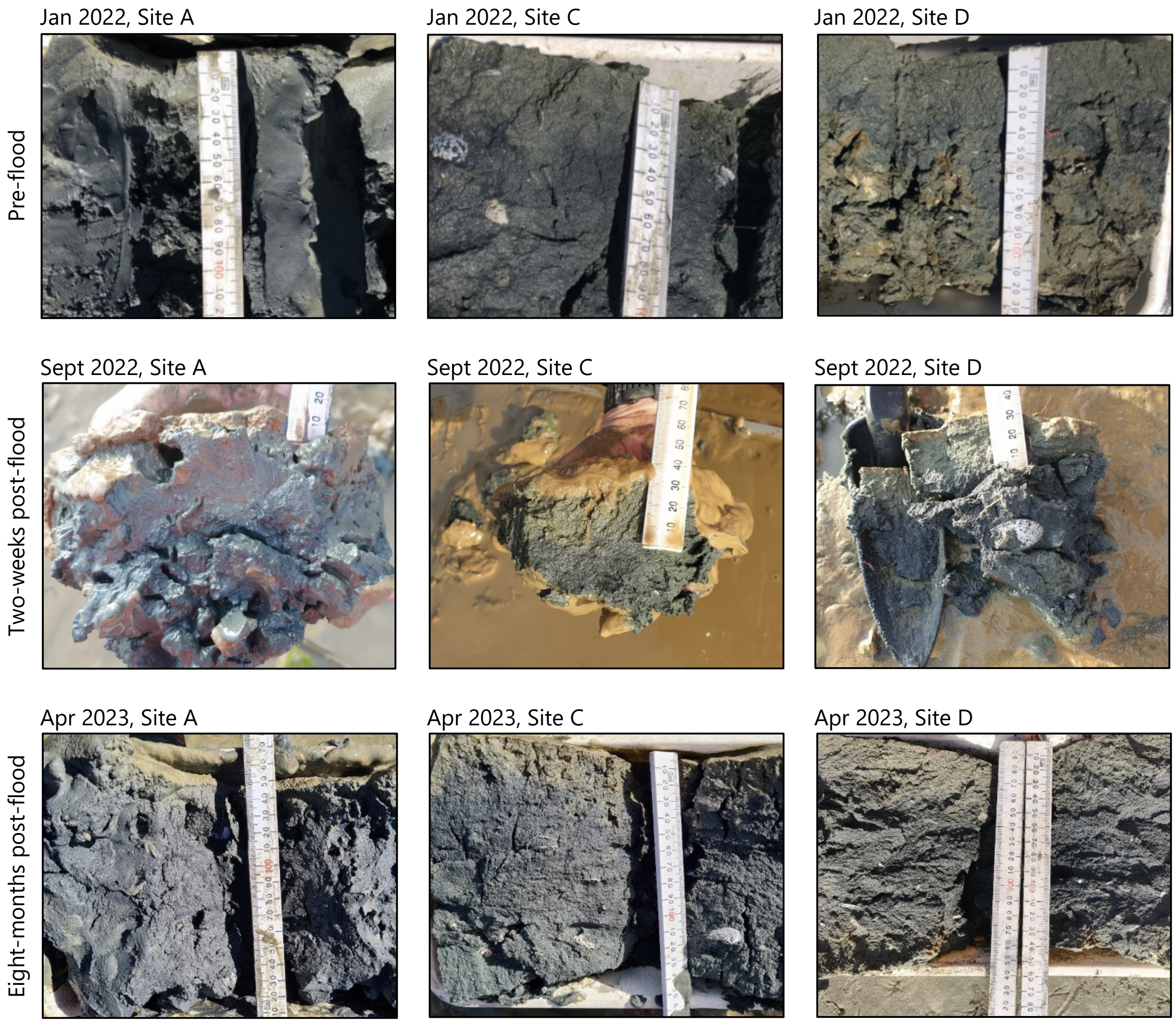


Fig. 5. Sediment particle grain size analysis showing percentage composition of mud (<63μm), sand (<2mm to ≥63μm) and gravel (≥2mm) from composite samples (n=3) at fine scale sites.



Temporal comparisons of fine scale site sediment cores from before to after the flood event.

To provide a visual impression of sediment quality relative to the Table 3 condition ratings, Fig. 6 compares the mean percentage mud, total organic carbon (TOC) and total nitrogen (TN) from composite samples (fine scale sites only) against the rating thresholds.

For mud content, whereas Site A was rated 'poor' under baseline conditions, Site C moved from a rating of 'fair' pre-flood to 'poor' post-flood in April 2023. This change reflects a shift from a mean content of ~18-21% under baseline conditions to ~38% post-flood. The 'poor' rating change reflects that the 25% threshold was exceeded.

TOC and TN values have been consistently low at the fine scale sites, with mean values rated as 'good' or 'very good'. The increase post-flood is a reflection of the increased in sediment mud content. Note that TN at Sites B and C is generally less than the routine method detection limit of 250mg/kg.

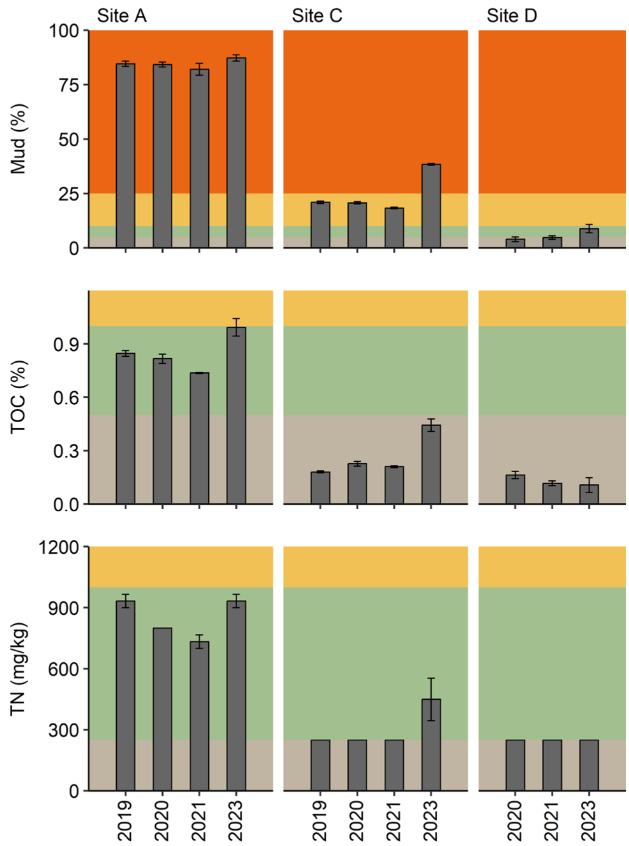


Fig. 6. Mean (±SE, n=3) sediment %mud, total organic carbon, and total nitrogen relative to condition ratings. Except for 2023, TN values at Sites C and D were less than routine laboratory method detection limits.

Very Good	Good	Fair	Poor

4.3.2 Oxygenation status

No signs of excessive sediment enrichment were evident. The aRPD depth was shallowest at Site A, which likely reflects that the high mud content reduces water penetration and oxygenation of the sediment matrix (Fig. 7). In general the aRPD has varied widely over time, but there was no consistent change attributable to the flood. The post-flood aRPD depth shallowed at Site C, which is consistent with the increase surface mud content, however, it was deeper at Sites A and D despite the increased mud content at those sites.

In part these results may reflect measurement variance due to the subjective nature of the aRPD estimates (Gerwing et al. 2013). Furthermore, aRPD depths were at times indistinct, with a sometimes poorly-defined oxygen-depleted zone evident beneath the sediment surface. In some cores there was mixing within the sediment profile due to bioturbation (i.e., sediment turnover by macrofauna). Hence, while measurements are carried out by experienced field staff, this range of factors means that some variability due to interpretation can be expected. Importantly, however, none of the sites showed evidence of black, anoxic (and sulphide-smelling) sediments at (or within a few millimetres of) the sediment surface, as would typically occur under strongly enriched conditions.

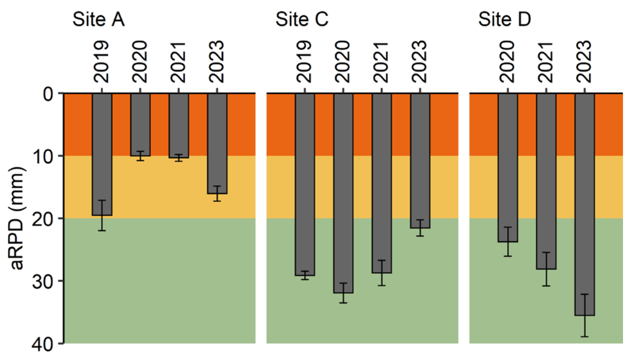
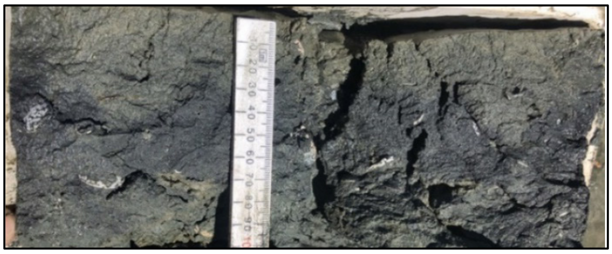


Fig. 7. Mean (±SE, n=10) aRPD depth relative to condition ratings.

Very Good	Good	Fair	Poor



Bioturbation by shellfish, worms and other invertebrates can lead to mixing of sediment and make the aRPD depth indistinct or variable.

4.3.3 Trace contaminants

Most trace elements were rated as 'very good' as concentrations were less than half of the ANZG (2018) DGV (Fig. 8), which is the value "*…below which there is a low risk of unacceptable effects…*". Nickel and chromium were slightly elevated (rated 'fair' and 'good', respectively), which is a reflection of natural sources due to the geology of catchment soils (Robinson et al. 1996).

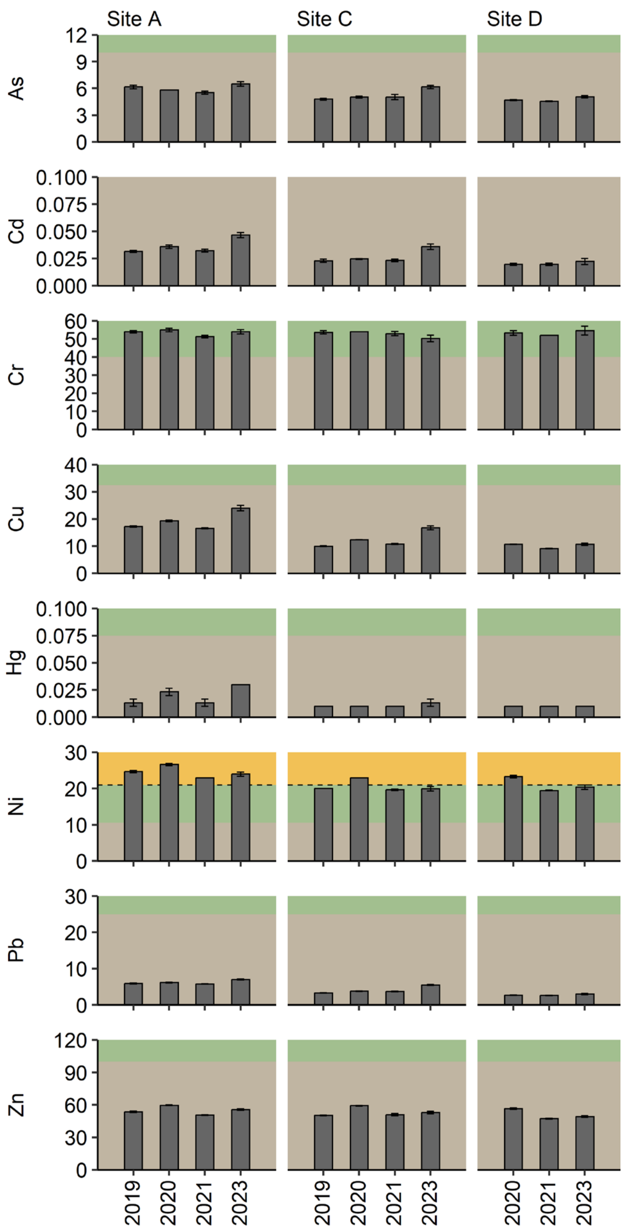


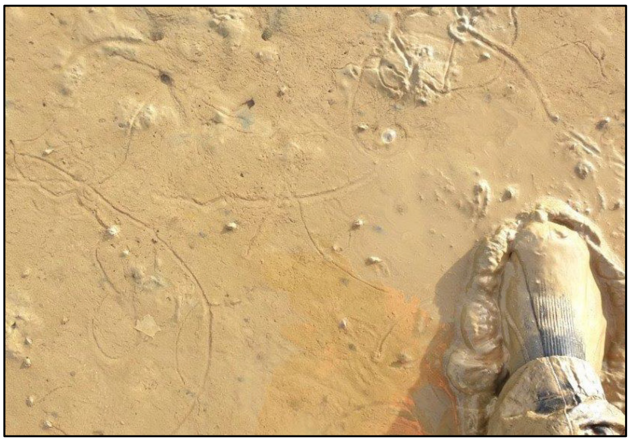
Fig. 8. Mean (±SE, n=3) trace element concentrations relative to condition ratings. ANZG (2018) sediment quality Default Guideline Values are represented by the boundary (dotted line) between 'good' and 'fair' condition. Elevated nickel (Ni) and chromium (Cr) reflect natural catchment sources.

Very Good	Good	Fair	Poor

4.4 MACROFAUNA

4.4.1 Conspicuous surface epibiota

Surface-dwelling epibiota are shown in Table 5. Conspicuous epibiota found in one or more surveys consisted of three estuarine snail species, and two species of common macroalgae, namely green 'sea lettuce' *Ulva* spp. and the red seaweed *Agarophyton* spp. (formerly called *Gracilaria*). Epibiota density (snails) and percent cover (macroalgae) has varied greatly among sites and surveys, but none of the temporal changes can be unequivocally linked to flood impacts. For example, macroalgae were absent from all sites in April 2023, but were not always present in the pre-flood surveys. Similarly, densities of the different estuarine snail species have varied widely among sites and over time, with the abundance of horn snails (*Zeacumantus* spp.) in fact being higher in 2023 than in previous surveys. The absence of an obvious effect is in contrast to the impression gained when surveying Delaware Inlet immediately post-flood (KR, pers. obs., see photos below), and from a survey of Nelson Haven estuary (5-months post-flood), in which impacts on epibiota appeared to be quite pronounced.



Muddy sediment inundation of estuary snails (top) and cockles (bottom) in September 2022, around two-weeks post-flood.

Table 5. SACFOR scores for epibiota over the four surveys, based on the scale in Appendix 1 (Section A5). Dash (-) = absent. Site D was not sampled in 2019.

Species	Common name	Functional description	Site A 2019	Site A 2020	Site A 2021	Site A 2023	Site C 2019	Site C 2020	Site C 2021	Site C 2023	Site D 2020	Site D 2021	Site D 2023
Snails													
Amphibola crenata	Mud snail	Microalgal grazer	-	O	O	O	-	R	-	-	-	-	-
Cominella glandiformis	Mud whelk	Carnivore and scavenger	-	R	-	-	C	O	O	R	R	O	R
Diloma subrostratum	Mudflat topshell	Grazer and deposit feeder	-	-	-	-	-	R	O	R	C	C	F
Zeacumantus spp.	Horn snail	Microalgal and detrital grazer	-	R	-	-	C	F	O	A	O	F	C
Macroalgae													
Agarophyton spp.[1]	Red seaweed	Primary producer	-	-	O	-	-	R	O	-	-	-	-
Ulva spp.	Sea lettuce	Primary producer	-	-	O	-	-	O	O	-	R	R	-

1. *Agarophyton* spp. is the revised name for *Gracilaria*, and in New Zealand can consist of more than one species.
S=Super-abundant, A=Abundant, C=Common, F=Frequent, O=Occasional, R=Rare

4.4.2 Macrofauna cores

Summary data for sediment-dwelling macrofauna are provided in Appendix 5. Table 6 and Table 7 describes the main species or higher taxa that were recorded.

A total of 42 taxa have been sampled from Delaware Inlet over the four surveys. In 2023, 35 taxa were recorded, compared with the pre-flood low of 22 (in 2019) and a high of 34 (in 2020). The earlier survey of Gillespie et al. (2009) recorded 33 taxa.

The muddy sediments at Site A were impoverished in terms of the range of taxa present and their abundances (Fig. 9). Mean species richness at Site A was ~4-5 taxa per survey, with mean abundances of <10 individuals. Site C has the greatest species richness and abundance values. From the temporal patterns evident in Fig. 9, there is no clear or consistent evidence of a flood effect on richness and abundance. For example, whereas mean species richness at Site C in 2023 was less than during the baseline, it was slightly greater at Site A, and similar but variable across years at Site D.

The species present over 2019-2023 represented 11 main taxonomic groups. The most well-represented in terms of both richness and abundance were polychaete worms and bivalve shellfish, with variable representation from shrimp-like amphipods, and decapod crabs (Table 7). The prevalence of the different groups was highly variable among sites, with no clear or consistent temporal pattern that pointed to a change post-flood

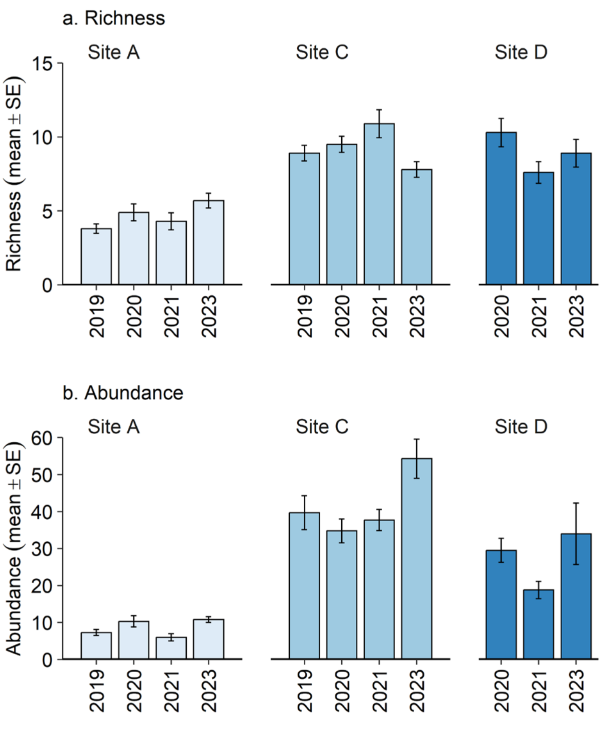


Fig. 9. Patterns in mean (±SE, n=5-10) taxon richness and abundance per core sample.

Table 6. Description of the sediment-dwelling macrofauna taxa comprising >10% of total abundance at any one site. Images from NIWA. Pink colour is due to a Rose Bengal stain used in the identification process.

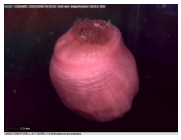
Main group	Description	Image
Amphipoda, *Torridoharpinia hurleyi*	Amphipods are shrimp-like crustaceans. This species contributes significantly to sediment turnover through its burrowing activities. It is an important prey item for birds and small fish. The adjacent image is illustrative.	
Anthozoa, *Anthopleura hermaphroditica*	Mud-flat anemone. This is a predatory species, living attached to cockles or broken shells. Grows up to 10mm. It is considered intolerant of high turbidity.	
Bivalvia, *Arthritica* sp. 5	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. Small cockles important in diet of some wading birds.	
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.	
Decapoda, *Hemiplax hirtipes*	Deposit feeding stalk-eyed mud crab, endemic to New Zealand. Can be common in wet areas at the mid to low water level. Makes extensive burrows in the mud.	
Polychaeta, *Axiothella serrata*	A deposit feeding maldanid 'bamboo' worm that is a common infaunal species on the sheltered flats of central New Zealand estuaries.	
Polychaeta, *Boccardia syrtis*	A small surface deposit-feeding spionid. Found in a wide range of sand/mud habitats. Lives in flexible tubes constructed of fine sediment grains, and can form dense mats on the sediment surface. Sensitive to organic enrichment.	
Polychaeta, Nereididae	Nereididae are a type of 'ragworm'. These were mainly unidentified juveniles, but included a few *Nicon aestuariensis* and *Perinereis vallata*. *Nicon aestuariensis* is a deposit feeding omnivorous worm that is tolerant of freshwater.	
Polychaeta, *Paradoneis lyra*	Common paraonid worm considered to be reasonably tolerant of muddy sediment and organic enrichment. Paraonids are considered to be deposit feeders, possibly selectively feeding on microscopic diatoms and protozoans.	
Polychaeta, *Prionospio aucklandica*	Deposit-feeding spionid worm, common in harbours and estuaries. *P. aucklandica* is associated mainly with muddy sands, but occurs across a range of mud contents and is rated as EG-III. Considered tolerant to organic enrichment.	

Table 7. Sediment-dwelling macrofauna taxa that comprised >10% of total abundance at any one site. The Table shows site abundances pooled across cores within each survey. Shading is used to distinguish pre-flood abundances (grey) from post-flood (white) at each site. Eco-groups (EG) from sensitive (EG-I) to resilient (EG-V) are shown.

Main group	Taxa	EG	Site A				Site C				Site D		
			Pre-flood			Post	Pre-flood			Post	Pre-flood		Post
			2019	2020	2021	2023	2019	2020	2021	2023	2020	2021	2023
Amphipoda	*Torridoharpinia hurleyi*	I	36	44	14	7	13	29	37	2	13	13	4
Anthozoa	*Anthopleura hermaphroditica*	III					1				26	29	37
Bivalvia	*Arthritica* sp. 5	III			6				2	1			
Bivalvia	*Austrovenus stutchburyi*	II	5		1	3	43	50	69	54	35	28	101
Bivalvia	*Macomona liliana*	II			2	1	43	49	49	29	91	57	50
Decapoda	*Hemiplax hirtipes*	III	17	10	3	18	4	7	8	8		2	2
Polychaeta	*Axiothella serrata*	II	1				33	57	29		17	10	2
Polychaeta	*Boccardia syrtis*	II		7	4	27		1	1				
Polychaeta	Nereididae	III	4	2	12	15	7	6	12	26	3	2	11
Polychaeta	*Paradoneis lyra*	III					186	47	42	139			
Polychaeta	*Prionospio aucklandica*	III		21	2	24	39	63	79	257	16	6	70

The biological index AMBI was slightly greater in the 2023 post-flood survey compared with the baseline years, suggesting a slight degradation in condition (Fig. 10). Nonetheless, values were rated as 'good' across all sites and surveys, and were not markedly different among sites despite their contrasting sediment characteristics. The AMBI results overall reflect a high prevalence of relative 'sensitive' taxa classified in eco-group (EG) I and II, with relatively few hardy taxa in EG-IV and V (Fig. 11, Table 6). That said, the post-flood survey indicated a loss of EG-I taxa in favour of an increased prevalence of moderately hardy EG-III taxa, especially at Sites A and C.

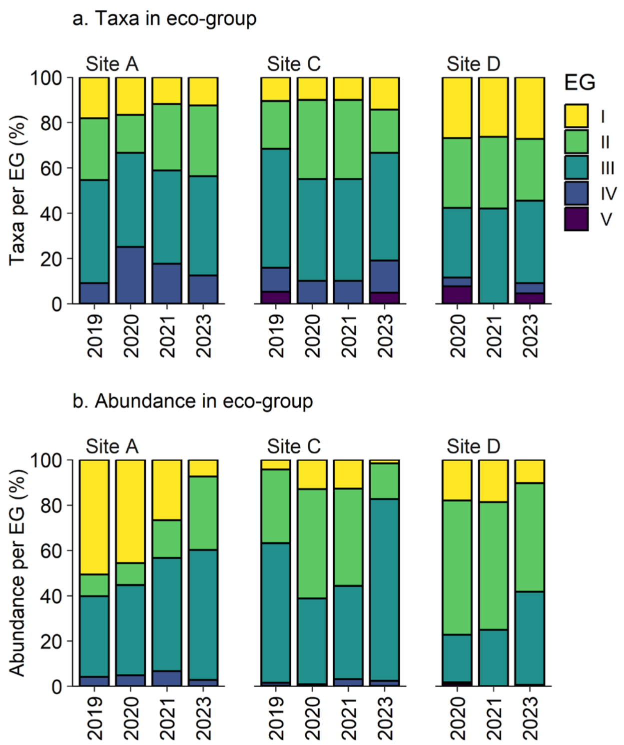


Fig. 11. Contribution to site richness and abundances of species within eco-groups ranging from sensitive (EG-I) to resilient (EG-V).

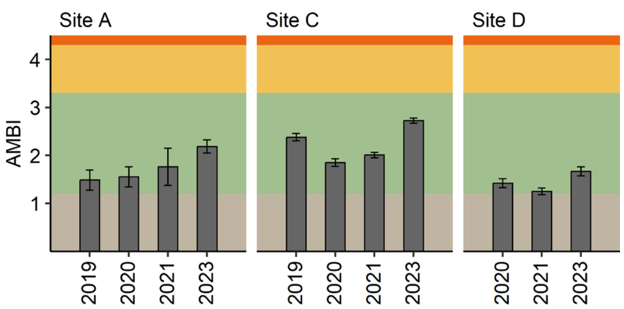


Fig. 10. Patterns in mean (±SE, n=5-10) AMBI scores relative to condition ratings.

From Table 7, this trend can be seen in the increased abundances in 2023 of EG-III polychaete worms including *Boccardia syrtis* (Site A), various nereids (all sites), *Paradoneis lyra* (Site C) and the spionid worm *Prionospio aucklandica* (all sites). Simultaneously EG-I species such as the amphipod *Torridoharpinia hurleyi* declined in abundance at all sites. In fact the initially high and subsequent declining contribution of EG-I taxa at Site A, as shown in Fig. 11, reflects the decline in abundance of *Torridoharpinia hurleyi*. The EG-I rating for this species is based on the international score assigned to the amphipod group to which it belongs (i.e., Phoxocephalidae). However, this New Zealand species can clearly tolerate a wide range of sediment mud values, suggesting that it is far more resilient than the EG-I designation suggests (see below).

Cockles (*Austrovenus stutchburyi*) did not appear to be negatively impacted by the flood. Very few cockles have been sampled at Site A but this species has been consistently abundant at Sites C and D (Table 7). At Site C, densities were similar pre- vs post-flood, and comprised generally small individuals (~5-9mm shell length). At Site D cockle densities increased 3-4 times post-flood, but size data provided by the taxonomist showed a decrease in mean shell length; from ~13-18mm pre-flood to 7mm post-flood. Hence, there appears to have been a post-flood cockle recruitment event at Site D.

4.4.3 Macrofauna responses to environmental drivers of change

Whereas the physical changes from the flood were obvious in terms of the increased deposition of muddy sediment, the biological changes described above were relatively subtle. Below we further explore the pre- vs post-flood macrofauna changes and consider whether they are linked to the physical effects of the flood event. For this purpose, the further analyses considered changes in macrofauna in relation to sedimentation rate and sediment quality. The sediment quality indicators chosen were a subset of variables for which a cause-effect association was considered plausible, namely mud and sand content, aRPD, TOC, TP (as a proxy for TN, which was often less than method detection limits) and nickel (Ni). Nickel was the only trace element included, as detectable concentrations of the other trace analytes were low relative to DGVs.

Richness, abundance and AMBI

Considered across all sites and surveys, increasing sediment mud content was associated with a significant decline in macrofauna richness (r^2=-0.89, P<0.001) and abundance (r^2=-0.69, P<0.05), although had little effect on AMBI scores (Appendix 6). However, when changes were considered within each site there was an inconsistent response of richness (Fig. 12a) and abundance (Fig. 12b) to changes in mud content, but AMBI values increased consistently across sites (i.e., representing a decline in condition). Statistical models that were used to explore changes in macrofauna response in relation to sediment mud content revealed a significant increase in AMBI scores in response to the flood event, which was associated with the increase in sediment mud content.

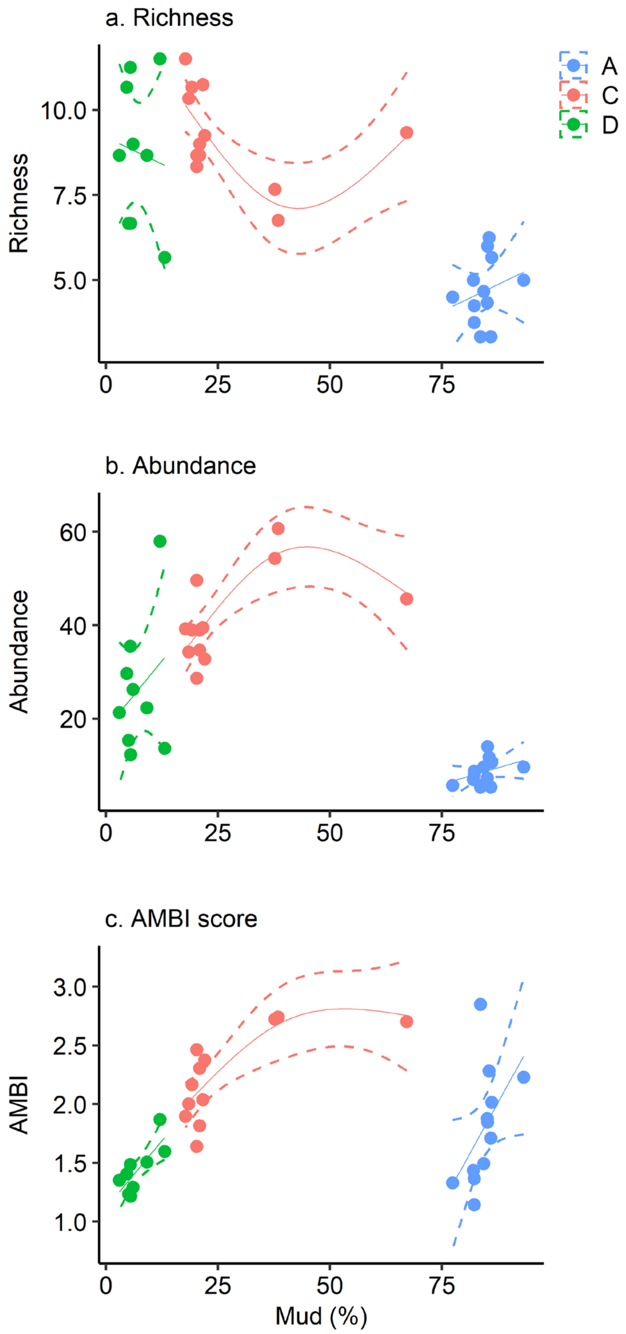


Fig. 12. Relationships between sediment mud content and macrofauna indices. Smoothing lines for each site are fitted with a 95% confidence interval (dashed). Values are based on 3 composite samples per site for mud (see schematic in Fig. 2).

To explore changes in macrofauna composition in relation to environmental factors, the nMDS ordination in Fig. 13 places sites and years of similar macrofauna composition close to each other in a 2-dimensional plot, with less similar years being further apart.

Fig. 13 reveals distinct differences in macrofauna composition among the three sites, which reflect a combination of species suites that are unique to each site, as well as shifts in abundance of the dominant taxa such as described above and in Table 7. For example:

- Site A: The small mud-tolerant bivalve *Theora lubrica* was unique to the site, and mud crabs (*Hemiplax hirtipes*) were relatively abundant.
- Site C: The polychaete worm *Paradoneis lyra* was relatively abundant and was not recorded at the other sites.
- Site D: Had several species unique to the site, the most abundance of which were mudflat anemones (*Anthopleura hermaphroditica*) and the polychaete worm *Aonides trifida*).

These types of differences were borne out in an analysis using PERMANOVA (Anderson et al. 2008), which highlighted the obvious (i.e., from Fig. 13) significant overall composition differences among sites (Pseudo-F=17.51, p<0.001) and also an overall contrast in community similarity pre- vs post-flood (Pseudo-F=6.52, p<0.001). When pre- vs post-flood contrasts were considered within each site significant differences were revealed at Site A (t=1.72, p=0.013) and Site C (t=2.77, p=0.008) but not Site D (t=1.37, p=0.10). These results are illustrated in Fig. 13a, whereby the dotted ellipses enclose groups of sites that are more similar to each other than sites in other groups. For example, it can be seen that in the 2023 post-flood survey macrofauna composition at Site C was distinct from the cluster formed by the baseline years, but at Site D was similar across all years.

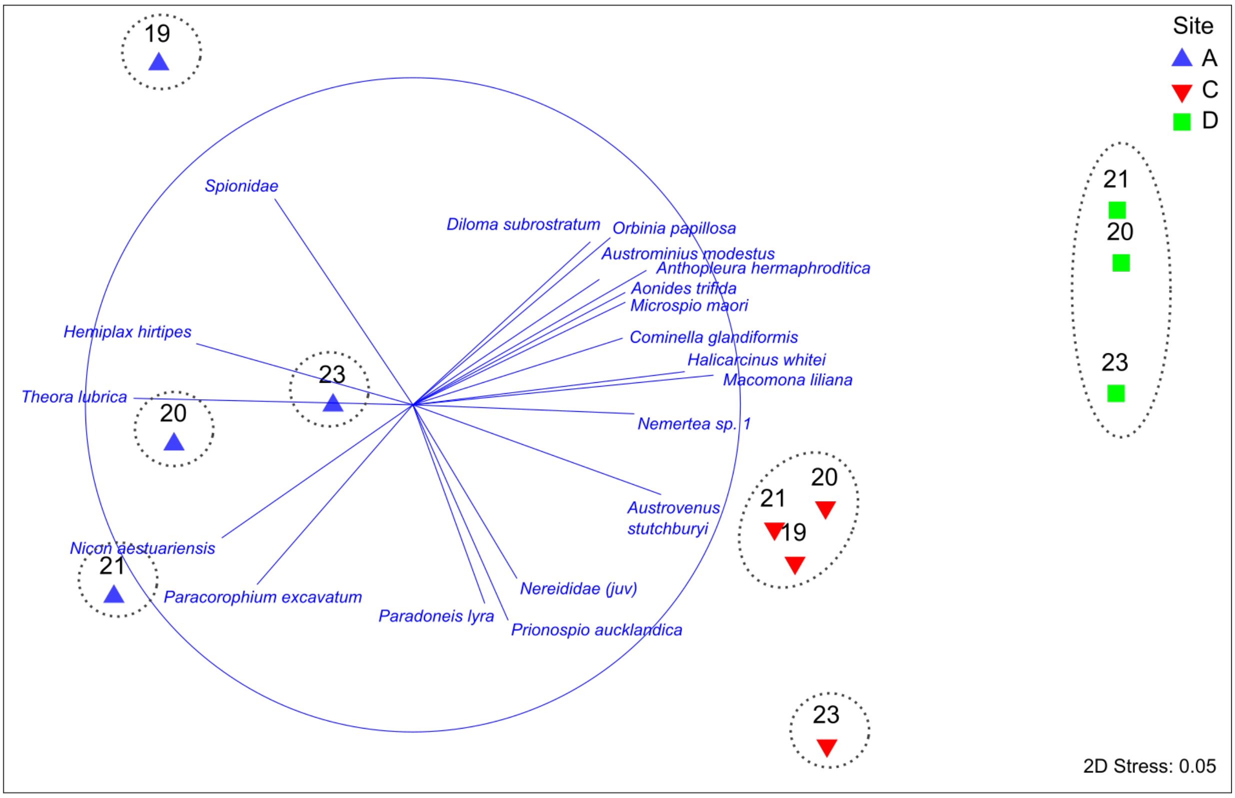
In an overall analysis using the BEST routine in PRIMER (Clarke & Gorley 2015), sediment mud content was highlighted as having by far the most significant association with macrofaunal composition (Spearman rank correlation=0.862 site-level & 0.786 zone-level), which is illustrated by the scaled bubbles for % mud in Fig. 13b. Sediment oxygenation (aRPD) and nickel also showed moderate correlations with macrofauna, which likely reflects their association with mud content (Appendix 6). For example, there was a negative effect of mud on aRPD depth (i.e., the aRPD shallowed with increased mud) reflecting the reduced capacity for oxygenation of the sediment matrix.

Despite the strong spatial effect due to mud, the extent to which the temporal change pre- vs post-flood is explained by the increased sediment mud content is less clear. To explore this question, a more forensic analysis in Appendix 6 shows within-site MDS plots and BEST results, illustrating correlations between the temporal change in macrofauna composition and sediment quality. The results show a very strong relationship between temporal change in macrofauna and sediment mud content at Site C, but at Sites A and D none of the sediment quality variables was associated with macrofauna composition changes.

Even at Site C, some of the species' changes that were correlated with a mud effect also occurred at the other sites, and reflected changes in abundances of species that appear tolerant of a wide range of sediment types. For example, Table 7 shows several species for which Site C showed a post-flood abundance increase (e.g., nereid polychaetes, *Prionospio aucklandica*) or decrease (e.g., *Torridoharpinia hurleyi*), but the changes were mirrored at the other sites. Furthermore, plots of the distribution of these key species in relation to sediment mud content reveal that most are tolerant of a wide mud range (Fig. 14). Among the dominant species, the only one whose absence from Site C (and decline at Site D) post-flood is consistent with a mud impact is the polychaete *Axiothella serrata* (aka bamboo worm). This species tends to be uncommon as mud content increases beyond ~25% (Fig. 14), which was roughly the threshold of pre- vs post-flood change at Site C.

Site C muddy sediment, eight-months post-flood in April 2023.

a. Species overlay



b. Sediment quality overlay (bubbles scaled to %Mud)

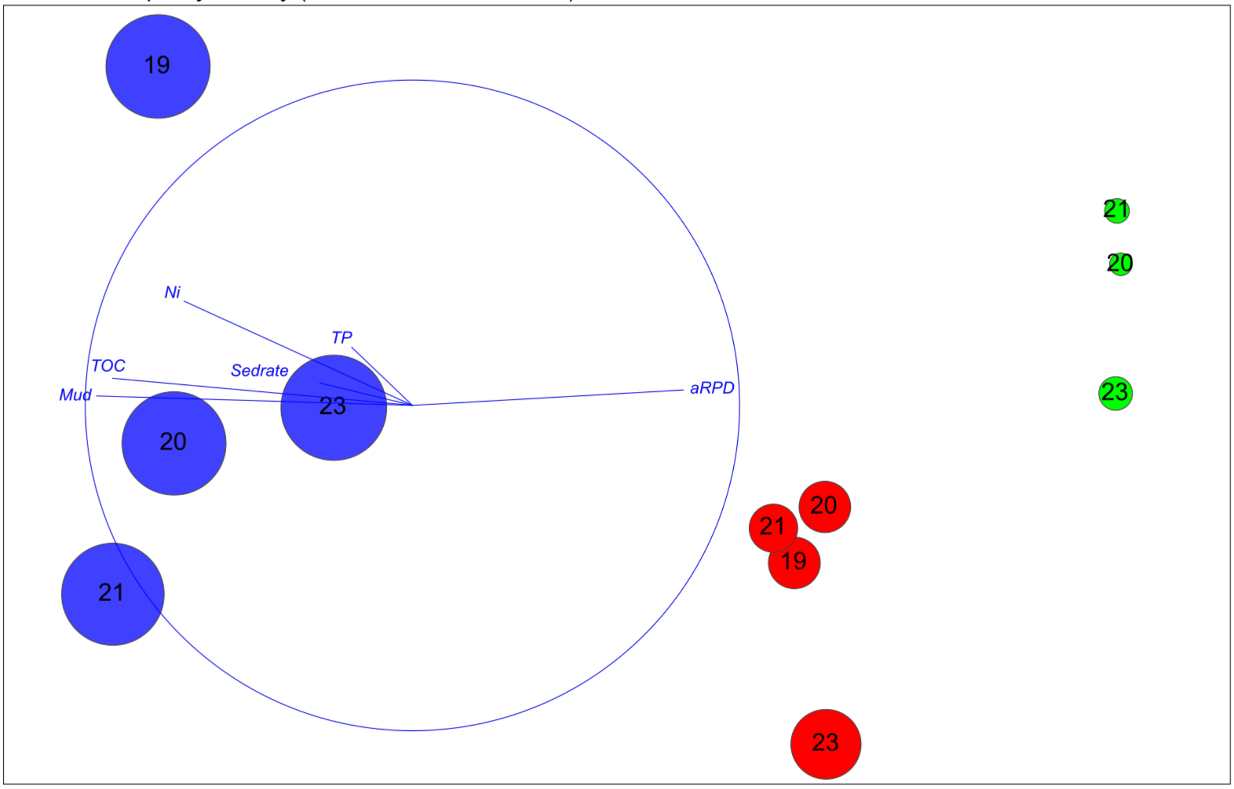


Fig. 13. Non-metric MDS ordination of macrofaunal core samples for data aggregated within each site and survey.

Sites and surveys closer to each other are more similar than distant ones in terms of macrofaunal composition. A 'stress' value of 0.05 indicates that a 2-dimensional plot provides a reliable representation of differences. The vectors show the direction and strength of association (length of line relative to circle) of the taxa (top) and environmental variables (bottom) that most strongly influenced the pattern of site-survey differences. Dotted ellipses in the top panel enclose sites with a high similarity (70%) based on the Bray-Curtis measure. Bubble sizes in the bottom panel representing the contrast of decreasing mud from the highest values at Site A (blue bubbles on left) to the lowest at Site D (green bubbles on right).

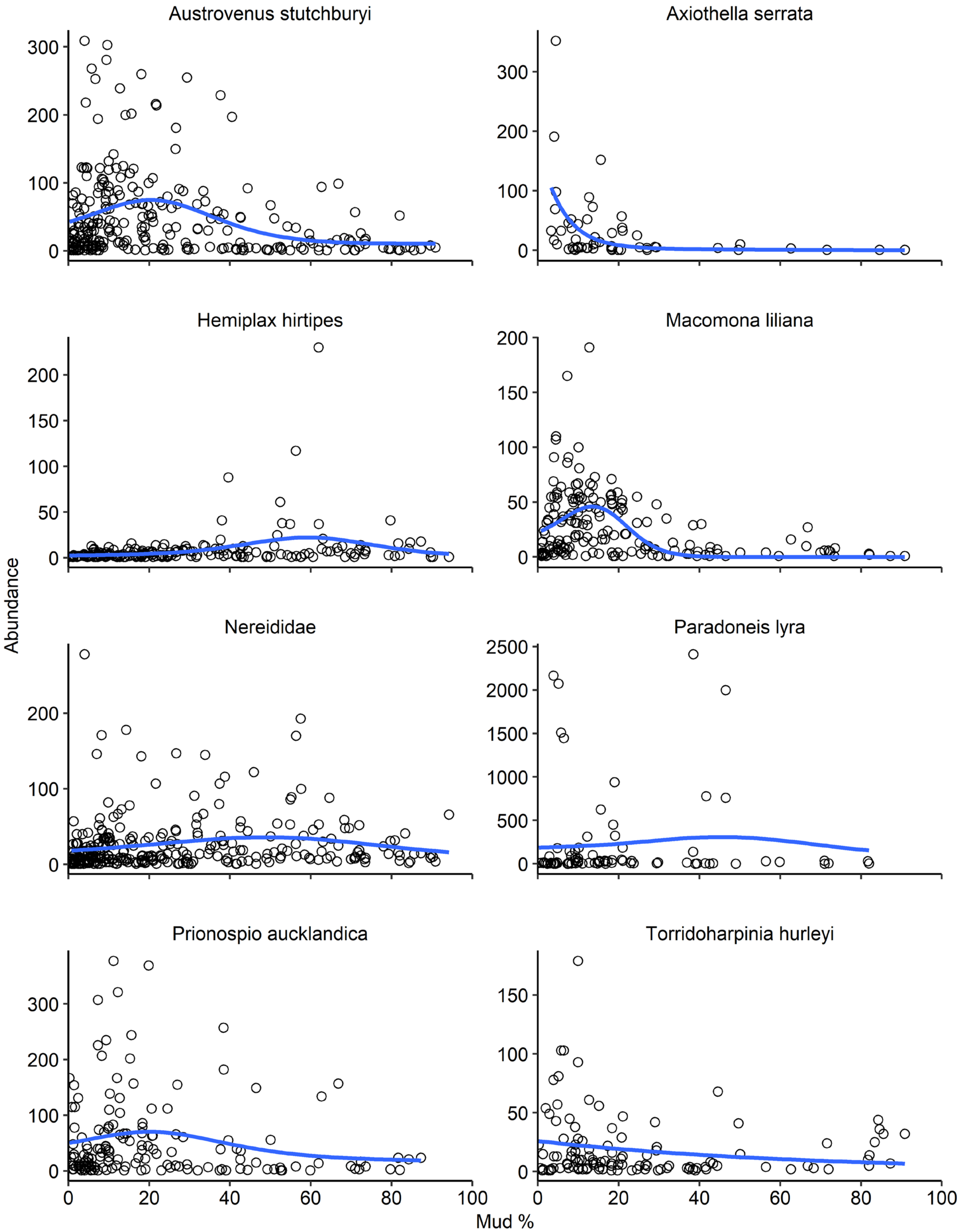


Fig. 14. Distribution of dominant Delaware Inlet macrofauna in relation to sediment mud content. Derived from Salt Ecology data for multiple surveys at 91 sites in 34 New Zealand estuaries. Average site-survey abundances are represented by the points, with log-normal smoothing lines showing the relationship with % mud.

5. SYNTHESIS AND RECOMMENDATIONS

5.1 SYNTHESIS OF KEY FINDINGS

This report has described the findings of four ecological surveys and associated sedimentation monitoring conducted in Delaware Inlet, Nelson, since 2019, focusing on a comparison of three 'baseline' years (2019-2021) with investigations conducted following a regionally significant flood in August 2022. The ecological surveys have largely followed the fine scale methods described in New Zealand's National Estuary Monitoring Protocol (NEMP). In Table 8, key physical and biological indicators are compared against the condition rating criteria shown in Table 3.

Table 8 highlights the marked increase in post-flood sediment deposition at Sites A and C, with a clear increase in sediment mud content at all three monitoring sites. As noted in the results, the concomitant increase in sediment mud content suggests that the recent deposition reflects new catchment-derived sediment due to the flood, rather than resuspension and redistribution of existing sediments on the tidal flats.

Although sediment mud content had abated at all sites by the time of the April 2023 survey (i.e., almost eight months after the flood), the levels at upper estuary sites were still clearly greater than the baseline values represented by the pre-flood data. Except for the increased sediment mud content, sediment quality did not otherwise greatly deteriorate due to the flood event, and in fact has not changed appreciably since an earlier survey described by Gillespie et al. (2009). Overall sediment quality, in terms of trophic state indicators and trace contaminants, remains high (condition ratings of 'good' or 'very good' in Table 8). As already noted, the 'fair' rating for the trace metal nickel is due to inputs from natural sources in the catchment, with implications previously discussed by Forrest and Stevens (2021).

Given the increased sedimentation and sediment mud content associated with the flood, we expected to see concomitant changes in the biota. In this respect, the surface-dwelling epibiota (snails and seaweeds) was not clearly impacted at the time of the survey. As noted above, this situation contrasts the impression gained when surveying Delaware Inlet immediately post-flood. It also contrasts the findings of an estuary survey in Nelson Haven conducted in January 2023 (five-months post-flood), which revealed a complete loss of most epibiota and a reduction in the cover of high-value seagrass habitat (Forrest et al. 2023). In the case of Delaware, it appears that the epibiota has largely recovered from any flood-related effects that may have initially occurred.

Impacts on sediment dwelling macrofauna that could be clearly attributed to the physical effects of the flood event were also quite subtle. There was a small increase in AMBI scores, potentially reflecting a slight degradation in ecological quality, and a reduction in the range of species present at Site C. Furthermore, pre- vs post-flood differences in macrofauna composition were

Table 8. Summary of condition scores of ecological health for fine scale monitoring sites, based on mean values of key indicators, and rating criteria in Table 3. Sedimentation rate (Sed rate) is the average annual rate since the baseline year. TP not rated. See Glossary for definition of indicators.

Site	Year	Sed rate mm/yr	Mud %	aRPD mm	TN mg/kg	TP mg/kg	TOC %	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 na
A	2019	-	84.6	20	933	613	0.85	6.2	0.032	54.0	17.3	0.01*	24.7	5.9	53.7	1.5
	2020	-	84.2	10	800	677	0.82	5.8	0.036	55.0	19.3	0.02	26.7	6.2	59.7	1.6
	2021	-	82.1	10	733	563	0.74	5.5	0.032	51.3	16.6	0.01*	23.0	5.8	50.7	1.8
	2023	6.3	87.3	16	933	583	0.99	6.5	0.047	54.0	24.0	0.03	24.0	7.0	55.7	2.2
C	2019	-	21.0	29	< 500	567	0.18	4.8	0.023	53.7	10.0	< 0.02	20.0	3.3	50.3	2.4
	2020	-	20.7	32	< 500	640	0.23	5.0	0.025	54.0	12.4	< 0.02	23.0	3.8	59.3	1.9
	2021	-	18.3	29	< 500	543	0.21	5.0	0.023	53.0	10.8	< 0.02	19.7	3.7	51.0	2.0
	2023	2.7	38.4	22	450*	587	0.44	6.2	0.036	50.3	16.8	0.01*	19.9	5.5	53.0	2.7
D	2020	-	4.0	24	< 500	640	0.16	4.7	0.020	53.3	10.7	< 0.02	23.3	2.7	56.7	1.4
	2021	-	4.7	28	< 500	543	0.12	4.6	0.020	52.0	9.2	< 0.02	19.5	2.6	47.3	1.3
	2023	4.9	8.8	36	< 500	600	0.11*	5.1	0.022	54.7	10.7	< 0.02	20.4	3.0	49.3	1.7

** Sample mean includes values below lab detection limits*

< All values below lab detection limit

Very Good	Good	Fair	Poor

evident at all sites, but could only be linked to increased sediment mud content at Site C. These results contrast experimental studies of terrigenous sediment deposition in New Zealand estuaries, which have shown pronounced impacts on macrofauna within 1-2 weeks, resulting from as little as 3-5mm of simulated sediment deposition (Norkko et al. 2002; Lohrer et al. 2004; Rodil et al. 2011).

In the Delaware Inlet case, it is possible that the macrofauna community may have undergone some degree of recovery from flood-related effects, given that ~8-months had passed between the event and the survey described here. Recovery processes include recruitment processes, and the potential for migration from deeper sediment layers (Wheatcroft 2006). However, it seems unlikely that complete recovery would have occurred, especially given that the physical effects of the flood-related sedimentation were still clearly evident in April 2023. At least two other New Zealand studies have described incomplete recovery of the macrofauna from experimental inundation with sediment after durations of ~7 months (Thrush et al. 2003) and 13 months (Norkko et al. 2002). However, recovery may be relatively fast (e.g., a few months) in locations subject to physical processes that erode and remove the deposited sediment (Norkko et al. 2002). For example, Site D in Delaware Inlet likely experiences the flushing effects of an adjacent tidal channel that maintains the sand-dominated sediments at that site, and has likely minimised the extent and impact of flood-related sediment deposition, or enabled rapid recovery.

Other factors may also be important in affecting the composition of the macrofauna and/or its resilience to sediment deposition. These factors could include differing salinity conditions due variable distances from freshwater inputs, site-specific differences in wind-wave disturbance, and existing sediment mud content. For example, Site A was already very muddy under baseline conditions, hence further mud deposition would not necessarily be expected to severely impact the biota already present. By contrast, the mud content at Site C was around 20% pre-flood, peaked at ~95% (at the sediment plate site) immediately post-flood, and was still elevated (~38%) at the time of the April 2023 survey. Hence the sediment mud content at Site C has persisted for at least eight months at a level above the 25% threshold beyond which marked biological changes can occur (Robertson et al. 2015; Ward & Roberts 2021), which is the basis for the fair/poor boundary in the Table 3 condition ratings.

Consistent with these physical changes at Site C is the apparent loss of bamboo worms, which were relatively abundant under baseline conditions and appear quite sensitive to muddy sediment. However, it is evident from Fig. 14 that most of the other dominant species at each site have been recorded nationally across a wide range in sediment mud content (see Fig. 14) and, as such, are likely to be relatively resilient to deposition-related impacts. That said, the optimal mud range for many species is likely to be far more restricted that their overall tolerance range (Robertson et al. 2015; Ellis et al. 2017).

The absence of an estuary-wide effect on macrofauna that can unequivocally be linked to muddy sediment deposition raises the question as to whether the pre- vs post-flood shifts in macrofauna reflect some unmeasured effect of the flood event, or are due to other factors. For example, a change in surface sediment conditions (i.e., the surface 20mm that is sampled according to NEMP methods) may not reflect the sediment grain size across the full depth profile (150mm) that macrofauna cores are taken. Average sediment mud content across the core profile will be less than indicated by surface sampling.

5.2 MANAGEMENT AND MONITORING CONSIDERATIONS

5.2.1 Management

The ecological effects of the marked physical changes at the fine scale sites are clearly relatively subtle. Unfortunately, however, it is likely that the worst-case effects of the flood event were not captured by the monitoring programme. Stevens and Roberts (2022) mapped the areas of greatest sediment deposition immediately post-flood (Fig. 15). A comparison of Fig. 15 with locations of sediment plates and fine scale sites in Fig. 2 shows that the greatest deposition (>30cm in places) occurred in locations outside the monitoring sites. The ecological effects in these areas are unknown, and there are no baseline data that would enable comparison.

It is also unknown at this stage whether deposited sediment will disperse from these worst-affected areas. However, NCC are conducting estuary-wide monitoring of sediment depth changes in some areas of high deposition, to address this matter. In addition, preliminary data from broad scale habitat mapping undertaken alongside the fine scale survey in April 2023 shows that, while the muddy sediment extent is quantitatively greater than in 2018, there has been a qualitative reduction in extent since the September 2022 post-flood assessment. Mud persists in the areas highlighted in red in Fig. 15. While not completely gone, muds have dispersed from the main tidal flats (i.e., yellow lines of Fig. 15) since the flood.

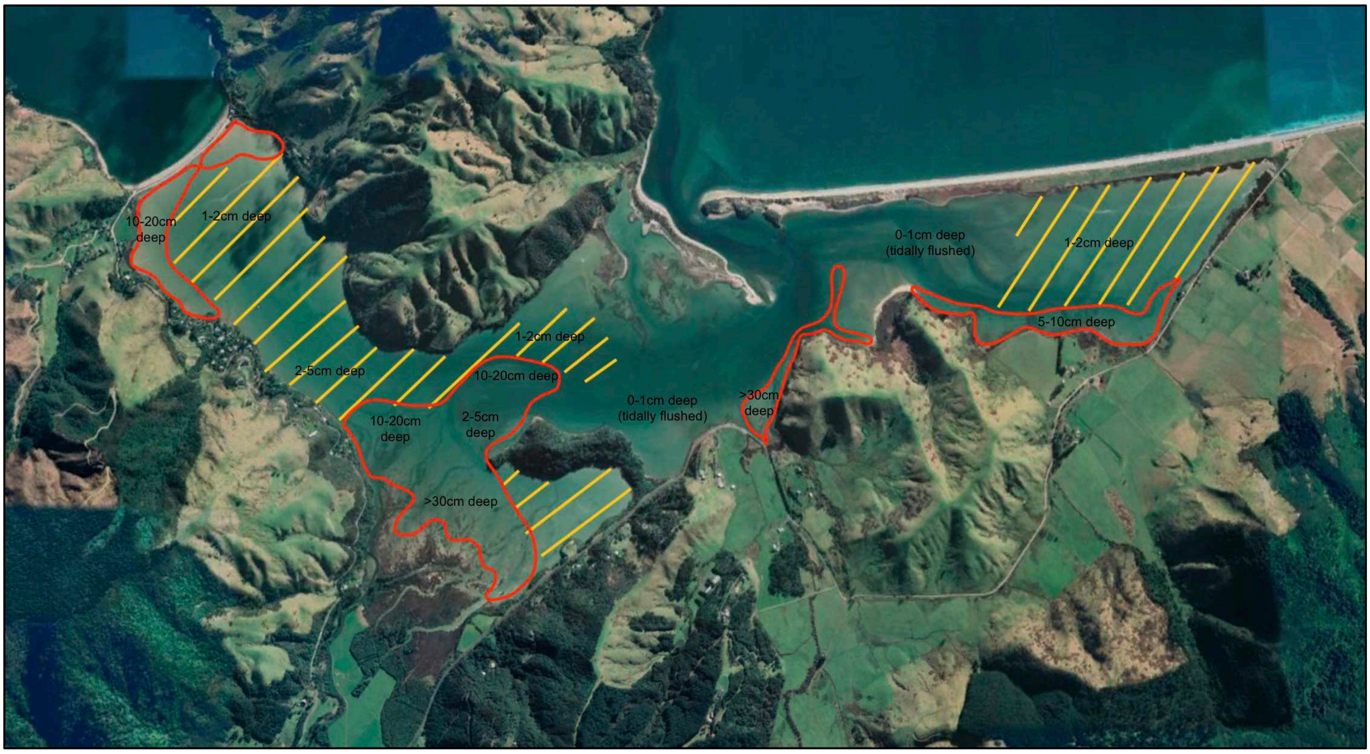


Fig. 15. General location and depth of post flood mud sediment deposits in Delaware Inlet, 1 September 2022. Source: Stevens and Roberts (2022). The red polygons show that areas of greatest sediment did not include locations where fine scale ecological or sediment plate monitoring is undertaken (see Fig. 2 for monitoring locations).

Despite the absence of longer-term post-flood monitoring, available data suggest that the estuary has considerable capacity to retain catchment-derived sediment. For example, modelled sediment loads for Delaware Inlet predict a sediment trap efficiency of 89% and estuary average deposition of 2.4mm/yr (Hicks et al. 2019). As such, a significant portion of future muddy sediment inputs to the estuary, especially from flood events, are likely to be retained. Muddy sediment may therefore have a cumulative effect on sediment quality and estuary biota in Delaware Inlet, which leads to an increase in mud extent and an eventual shift in ecological condition.

Management of ongoing sediment sources is therefore essential, so that the ecological health of the estuary is maintained or improved. The 2018 catchment data shown in Table 1 reveal that ~53% of the Delaware Inlet catchment is in land-uses that are known to generate a high fine-sediment run-off to waterways, in particular pasture (~18% of catchment area) and exotic plantation forestry (~35%). The 2018 data indicated only 1.4% of the catchment being harvested forest. This figure is likely to have changed as there has been extensive harvesting of parts of the catchment in the last few years.

Plantation forestry can be a particularly significant source of muddy sediment during forest harvest and for a few years after, when it can contribute a disproportionately high sediment load per catchment hectare (e.g., Gibbs & Woodward 2018). In addition, aerial imagery captured after the August 2022 flood shows extensive land slips in the Delaware Inlet catchment. As well as the event being a potentially significant contributor to the muddy sediment deposition described in this report, the bare and unstable land created by the slips may be an ongoing catchment sediment source after future rain events (Donovan 2022).

Given the above factors, it is timely for NCC to further consider mitigation options for catchment land uses that could lead to fine-sediment load increases, especially considering that delivery to the estuary is clearly exacerbated by intense rainfall events. Understanding future forest harvest schedules, and opportunities to mitigate harvest-related sediment inputs will be a key component to consider, and there may opportunities to mitigate ongoing slip inputs by active planting to stabilise slip areas.

5.2.2 Monitoring and investigations

Given the scale of sedimentation experienced in Delaware Inlet, there would be merit in the further investigations and monitoring described below.

Sources of muddy sediment

As an extension of a current NCC project to evaluate the number and area of land slips in the catchment, NCC could consider whether there are any predictors of land instability (e.g., slope, land-use; Donovan 2022). This type of approach potentially provides a simple means of elucidating the slip-related contribution from different land-use types, helping to direct and prioritise management actions to mitigate future inputs.

To understand sediment source contributions there are also more sophisticated approaches that could be considered, such as compound-specific stable isotope analyses. However, the presence of slip derived material that includes deeper sediment layers is likely to confound source attribution with this type of method.

Sedimentation monitoring

A longer time series of annual data collected during sediment plate monitoring (i.e., sedimentation rate and sediment mud content) will help to further elucidate present-day patterns and whether there is a further abatement of post-flood changes. NCC's broader-scale monitoring of sediment depth changes in areas of high deposition in the Delaware western and eastern arms will also provide some quantification of the physical recovery of the system (Stevens & Roberts 2022). That said, the NCC monitoring did not start until December 2022 in the western arm, and April 2023 in the eastern arm, hence is likely to have missed the worst-case effects and the initial phases of recovery. Unfortunately the immediate post-flood survey by Stevens and Roberts (2022) was deliberately broad scale and semi-quantitative, hence does not provide a quantitative benchmark against which NCC monitoring can be compared.

Ecological monitoring

In addition to the annual sediment plate monitoring, NCC's routine SOE monitoring for Delaware Inlet consists of broad scale habitat mapping and fine scale ecological surveys at intervals of ~5 years, reflecting the frequency of long-term monitoring recommended by the NEMP. Recent guidance produced by NIWA (Hewitt 2021) recommends fine scale monitoring is conducted twice a year as a minimum, with a time series of approximately 15 years needed for trend detection. This monitoring frequency for NCC is constrained by budgets and other monitoring priorities. As such, for Delaware Inlet, we suggest that NCC at least keep to their 5-yearly schedule. Given the absence of major flood-related ecological effects, there is no strong justification for a follow-up post-flood ecological assessment at shorter intervals.

Considerations for future event-based monitoring

In the case of Delaware Inlet, the delay in monitoring ecological effects and implementing the estuary-wide sediment assessment have meant that the worst-case effects of the flood event have been missed. An additional issue is that the fine scale and sedimentation sites for which quantitative pre-flood data were available were not in the worst-affected parts of the estuary. This situation highlights some key challenges for event-based monitoring:

<u>Timeliness of response:</u> The synoptic post-flood survey was implemented within two weeks of the event, and sediment plates were measured at the same time. A more comprehensive response involving quantitative technical studies (including the ecological survey described here, and the wider NCC sedimentation monitoring), were beyond 'business as usual' for the council, and could not be undertaken in a timely manner due to staffing constraints and unavailability of budget to outsource 'reactive' studies.

<u>Absence of a quantitative baseline in locations worst affected:</u> Due to budget constraints, NCC's estuary SOE programme provides a bare minimum approach that is intended to capture long-term changes in the main habitats of estuaries. The forensic nature of the NEMP fine scale approach and sediment plate method means that NCC budgets (as is also the case for most other councils) enable only a few sites to be set up in each estuary. Based on NEMP recommendations, the fine scale sites are deliberately positioned on the main tidal flats of each estuary, and are not necessarily in locations subject to the greatest pressures. The NEMP broad scale approach goes some way to addressing these limitations, but is based on subjective mapping of surface features only.

<u>Potential improvements:</u> The above issues suggest that there is a need for a modified monitoring approach that blends estuary wide assessment and quantitative sampling methods, and includes locations under pressure. Moreover, the approach needs to be implemented for little time and cost, so that resource budget and staffing constraints are not limiting factors. For the purpose of monitoring the physical effects of sedimentation, these outcomes could be achieved by:

- Quantitative measurement of sediment depth and sediment mud content at fixed points (e.g., pegged transects) in obvious depositional areas (e.g., the muddiest parts of an estuary) and river outflow deltas, as well as across main tidal flats. This scattergun approach would be quick to undertake (e.g., 1-2 low tides) after initial set-up, and not require specialist expertise.
- To complement point sampling methods, broad scale sedimentation mapping technologies such as laser-based Light Detection and Ranging (LiDAR), bathymetric surveys and Real-time Kinematic Positioning (RTK; a GPS-based approach) could be used to track long-term of event-based changes in sediment depth (e.g.,Townsend & Lohrer 2015).

The above approaches would complement each other to provide an estuary-wide overview of sediment depth and grain size, which would enhance the existing NEMP and sediment plate approaches.

Overcoming the ecological issues to capture worst-case event effects is more problematic due to analytical costs, especially for macrofauna taxonomy. In addition, as highlighted in this report, the links between large-scale events and ecological change are not always clear. For example, the magnitude of ecological change will likely depend on both the existing habitat type and the magnitude of physical impact. As noted in the present report, for instance, Site A may not be particularly susceptible to long term ecological impacts from large deposition events because it already consists soft-mud habitat. Part of the solution to addressing the above issues, and enabling monitoring to be targeted at susceptible parts of estuaries, would be to apply modelling approaches that predict and define depositional areas. An example of this type of approach is the Coastal Receiving Environment Scenario Tool (CREST) model recently developed for Auckland Council.

5.3 RECOMMENDATIONS

On the basis of the findings and discussion in this report, recommendations for Delaware Inlet are as follows:

- Evaluate and maintain records of catchment land use changes to determine current and potential future sediment sources to the estuary, and investigate options to reduce inputs. As part of this work, consider whether the extent of land slips following the August 2022 flood is linked to particular land use types, and evaluate the benefits of actively planting priority slip areas.
- Continue annual sediment plate monitoring and sediment grain size analysis, to track recovery from flood-related impacts.
- Develop and implement a comprehensive estuary-wide programme for assessing sedimentation and sediment grain size, building on the programme already put in place by NCC, and potentially expanding to include broad-scale sedimentation mapping technologies (e.g., LiDAR).
- Undertake comprehensive NEMP fine scale ecological and sediment quality monitoring at an ongoing minimum of five-yearly intervals, using the current methods.
- Develop an event-based monitoring protocol and response pathway to address lags in data collection, so that worst-case effects are captured.

Fine (post-flood) sediments deposited over firm substrates in the eastern arm.

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APPENDIX 1. FINE SCALE SAMPLING METHODS

Mapping the main habitats in an estuary using the NEMP broad scale approach provides a basis for identifying representative areas to sample sediment quality and associated biota using the NEMP fine scale approach.

This Appendix details the fine scale survey approach used by Salt Ecology for assessing intertidal estuary condition. This is a generic approach that follows the NEMP methodology except as described below. Any deviation from the NEMP that is site-specific for a given estuary is described in the main report. For example, the NEMP recommends fine scale sites be 30m x 60m in area and set-up in unvegetated mud/sand habitats in the mid-tidal range. However, site dimensions may be smaller sue to habitat availability, sites are sometimes set-up in vegetated seagrass or macroalgal habitats, and may be higher than mid-tide elevation where estuary flats are 'perched' high in the tidal zone.

A commonly-used addition to the NEMP fine scale approach is to install sediment 'plates' (buried concrete pavers) at fine scale sites, or the wider estuary environs, as a means of monitoring sedimentation. This approach involves monitoring temporal change in the depth of sediment that occurs over each plate, which indicates whether sediment is accumulating (sediment depth increases) or eroding (sediment depth decreases). As well as providing insight into estuary sedimentation processes, the sediment plate method provides supporting data that assists in the interpretation of changes occurring at fine scale sites.

The NEMP fine scale sampling approach is described in Section A below, with the additional sediment plate monitoring component described in Section B. General approaches to data recording, QA/QC and analysis are described in Section C.

A. FINE SCALE METHOD DESCRIPTION

A1. Sampling design and indicators

A summary of fine scale sediment and biota indicators, the rationale for their use, and field sampling methods, is provided in Table A1. As per the NEMP, each fine scale site is divided into a 3 x 4 grid of 12 plots and sampling is conducted in 10 of these plots. Fig. A1 shows the standard numbering sequence for replicate plots (1-10) and the indicator sampling approach that is used by Salt Ecology. Although the approach closely follows the NEMP, alterations and additions to early NEMP methods have been introduced over the last 10 or more years. Salt Ecology has adopted these modifications as described in footnotes to the Table. The general approach can be summarised as follows:

- From each plot, a discrete macrofauna sample core is collected and sediment oxygenation is assessed according to the depth of the apparent Redox Potential Discontinuity (aRPD).
- Sediment samples for laboratory analysis are also collected from each plot, but for instead of analysing discrete samples (as specified in the NEMP) three composite samples are analysed, consisting of subsamples pooled across each of plots (X1-4, Y4-6 & Z7-10).

The fine scale methods are detailed in subsequent sections.

A2. Sediment quality sampling and laboratory analyses

The three composite sediment samples collected from each site are ~500g in weight, with the sub-samples that make up each composite collected to 20mm depth using a trowel. Samples are stored on ice and sent to Hill Laboratories for analysis of: particle grain size in three categories (%mud <63μm, sand <2mm to ≥63μm, gravel ≥2mm); organic matter (total organic carbon, TOC); nutrients (total nitrogen, TN; total phosphorus, TP); and trace contaminants (arsenic, As; cadmium, Cd; chromium, Cr; copper, Cu; mercury, Hg; lead, Pb; nickel, Ni; zinc, Zn). Details of laboratory methods and detection limits are provided in Table A2.

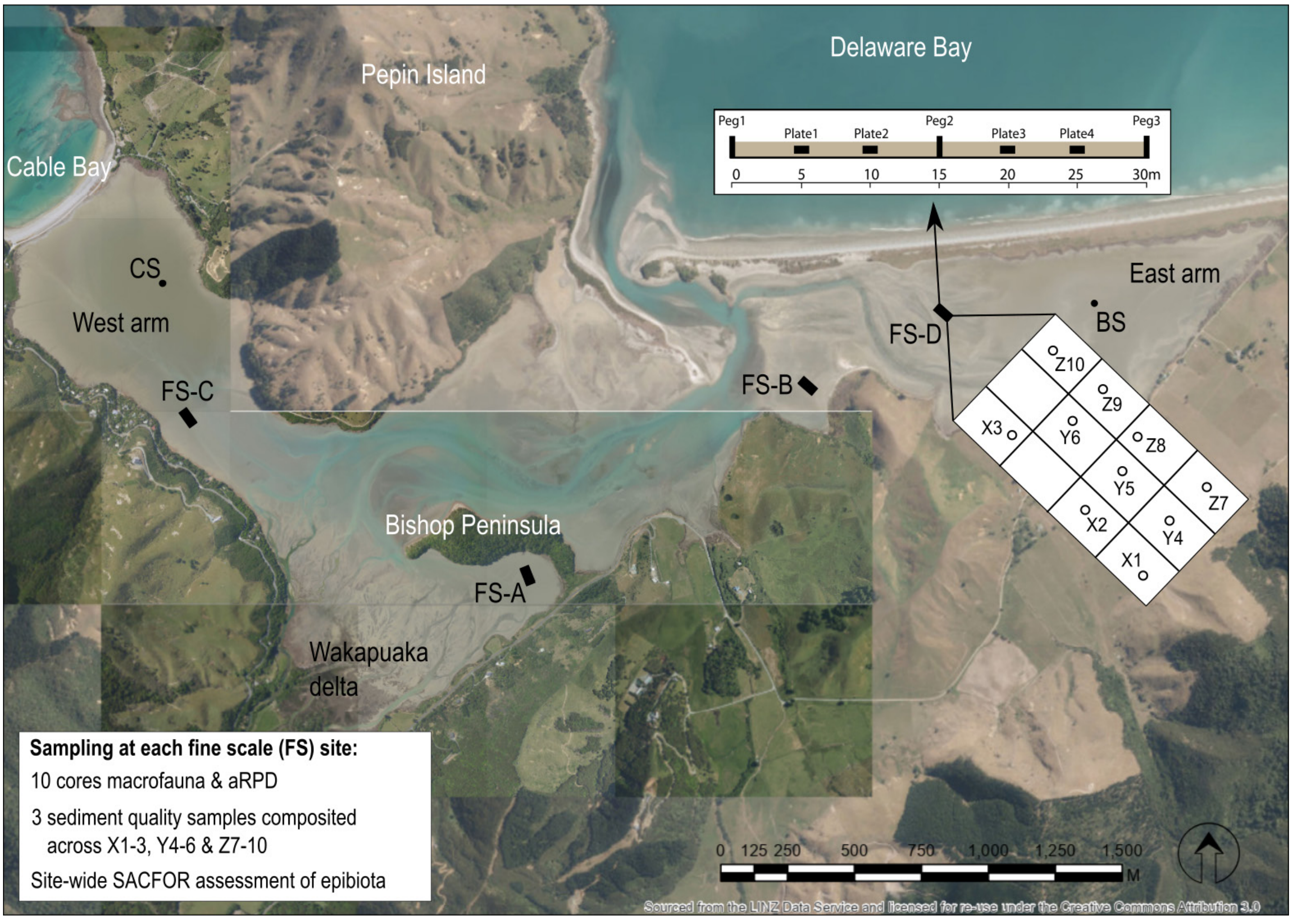
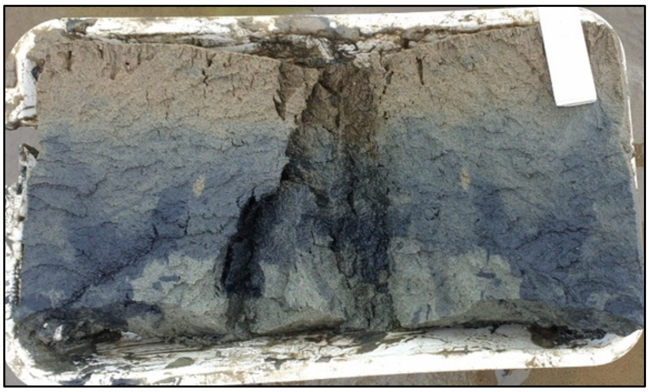


Fig. A1. Location of sites used for ongoing fine scale (FS) monitoring (A, C, D) and additional sites used for ongoing monitoring of sediment plates only (B, BS, CS). Note that Site B was discontinued as a fine scale site in 2019, and substituted with Site D. However, Site B has been retained for sediment plate monitoring. The schematic depicts the sediment core sample collection, and the sediment plate measurements.

A3. Field sediment oxygenation assessment

The aRPD depth (see Table A1) is used to assess the trophic status (i.e., extent of excessive organic or nutrient enrichment) of soft sediment. The aRPD provides an easily measured, time-integrated, and relatively stable indicator of sediment enrichment and oxygenation conditions (Rosenberg et al. 2001; Gerwing et al. 2013). Sediments are considered to have poor oxygenation if the aRPD is consistently <10mm deep and shows clear signs of organic enrichment, indicated by a distinct colour change to grey or black in the sediments. Extremely enriched sediments typically have an intense black sediment profile with aRPD at the surface, emit a rotten egg smell of hydrogen sulfide, and may have surface growths of sulfur oxidising bacteria.

Salt Ecology assesses mean aRPD depth (to the nearest mm) after extracting a large sediment core (130mm diameter, 150mm deep, ~2L volume) from each of the 10 plots, placing it on a tray, and splitting it vertically. Representative split cores are also photographed.



Example of aRPD profile.

Table A1. Summary of NEMP sediment quality and biota indicators, rationale for their use, and sampling method. Any significant departures from the NEMP are described in footnotes.

Indicator	General rationale	Sampling method
Physical and chemical		
Sediment grain size	Indicates the relative proportion of fine-grained sediments that have accumulated.	Composited surface scrape to 20mm sediment depth.
Nutrients (nitrogen and phosphorus), organic matter & total sulfur	Reflects the enrichment status of the estuary and potential for algal blooms and other symptoms of enrichment.	Surface scrape to 20mm sediment depth. Organic matter measured as Total Organic Carbon (TOC) (note 1).
Trace elements (arsenic copper, chromium, cadmium, lead, mercury, nickel, zinc)	Common toxic contaminants generally associated with human activities. High concentrations may indicate a need to investigate other anthropogenic inputs, e.g., pesticides, hydrocarbons.	Surface scrape to 20mm sediment depth (note 2).
Substrate oxygenation (apparent Redox Potential Discontinuity depth; aRPD)	Measures the enrichment/trophic state of sediments according to the depth of the aRPD. The aRPD can occur closer to the sediment surface as organic matter loading or sediment mud content increase.	Sediment core, split vertically, with average depth of aRPD recorded in the field where visible. The aRPD depth represents the visual transition between brown oxygenated surface sediments and deeper less oxygenated black sediments.
Biological		
Macrofauna	Abundance, composition and diversity of infauna living with the sediment are commonly-used indicators of estuarine health.	130mm diameter sediment core to 150mm depth (0.013m^2 sample area, 2L core volume), sieved to 0.5mm to retain macrofauna.
Epibiota (epifauna)	Abundance, composition and diversity of epifauna are commonly-used indicators of estuarine health.	Abundance based on SACFOR in Appendix 1, Table B3 (note 3).
Epibiota (macroalgae)	The composition and prevalence of macroalgae are indicators of nutrient enrichment.	Percent cover based on SACFOR in Appendix 1, Table B3 (note 3).
Epibiota (microalgae)	The prevalence of microalgae is an indicator of nutrient enrichment.	Visual assessment of conspicuous growths based on SACFOR in Appendix 1, Table B3 (notes 3, 4).

[1] Since the NEMP was published, Total Organic Carbon (TOC) has become available as a routine low-cost analysis which provides a more direct and reliable measure than the NEMP recommendation of converting Ash Free Dry Weight (AFDW) to TOC.

[2] Arsenic and mercury are not specified in the NEMP, but can be included in the trace element suite by the analytical laboratory.

[3] Assessment of epifauna, macroalgae and microalgae uses the SACFOR approach instead of the quadrat sampling outlined in the NEMP. 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 clumped or patchy distributions and the lack of demonstrated utility of microalgae as a routine indicator.

A4. Biological sampling: sediment-dwelling macrofauna

To sample sediment-dwelling macrofauna in each of the 10 plots, a large sediment core (130mm diameter, 150mm depth, ~2L volume) is collected, and placed in a 0.5mm mesh sieve bag, which is gently washed in seawater to remove fine sediment. The retained animals are preserved in a mixture of ~75% isopropyl alcohol and 25% seawater for later sorting and taxonomic identification by a skilled taxonomic laboratory (e.g., NIWA). The types of animals present in each sample, as well as the range of different species (i.e., richness) and their abundance, are well-established indicators of ecological health in estuarine and marine soft sediments.

Table A2. Analytical methods and detection limits for sediment samples used by Hill Laboratories.

Sample Type: Sediment			
Test	**Method Description**	**Default Detection Limit**	**Sample No**
Individual Tests			
Environmental Solids Sample Drying*	Air dried at 35°C Used for sample preparation. May contain a residual moisture content of 2-5%.	-	1-9
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%.	-	1-9
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	10-12
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	1-9
Total Recoverable digestion	Nitric / hydrochloric acid digestion. US EPA 200.2.	-	1-9
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	1-9
Total Nitrogen*	Catalytic Combustion (900°C, O2), separation, Thermal Conductivity Detector [Elementar Analyser].	0.05 g/100g dry wt	1-9
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	1-9
Heavy metal, trace level As,Cd,Cr,Cu,Ni,Pb,Zn	Dried sample, <2mm fraction. Nitric/Hydrochloric acid digestion, ICP-MS, trace level.	0.010 - 0.8 mg/kg dry wt	1-9
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.	0.10 - 6 mg/kg dry wt	10-12
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	1-9
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	1-9
Fraction < 63 µm*	Wet sieving with dispersant, as received, 63 µm sieve, gravimetry (calculation by difference).	0.1 g/100g dry wt	1-9

A5. Biological sampling: surface-dwelling epibiota

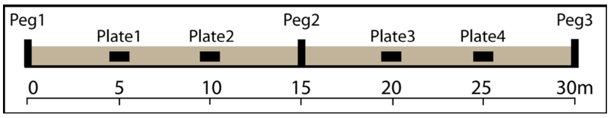
In addition to macrofaunal core sampling, epibiota (macroalgae and conspicuous surface-dwelling animals nominally >5mm body size) visible on the sediment surface at each site are semi-quantitatively categorised using 'SACFOR' abundance (animals) or percentage cover (macroalgae) ratings shown in the Table inset. These ratings represent a scoring scheme simplified from established monitoring methods (MNCR 1990; Blyth-Skyrme et al. 2008).

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

The SACFOR method is ideally suited to characterise intertidal epibiota with patchy or clumped distributions. It was conducted as an alternative to the quantitative quadrat sampling specified in the NEMP, which is known to poorly characterise scarce or clumped species. Note that our epibiota assessment does not include infaunal species that may be visible on the sediment surface, but whose abundance cannot be reliably determined from surface observation (e.g., cockles). Nor does it include very small organisms such as the estuarine snail *Potamopyrgus* spp.

B. SEDIMENT PLATES

The sediment plate method involves burying and levelling four (typically) concrete 'plates' (pavers, 19cm x 23cm) along a transect at each site, with pavers spaced between 2m and 5m apart. Plates are typically buried ~100m deep, and transect start, middle and end points marked with wooden pegs to enable relocation. At the time of baseline plate installation and on each subsequent sampling occasion, plate depth is measured by placing a 2m straight edge over each plate position to average out small-scale irregularities in surface topography, with the depth to each plate from the base on the straight edge measured by vertically inserting a probe into the sediment. Depth is measured to the nearest millimetre, with triplicate measures taken for each plate and averaged. Routine sediment plate measurements are made annually, and sometimes in response to event-related sediment inputs (e.g., after flooding). At the time of sampling, a single composite sediment sample is collected to 20mm depth for particle grain size analysis (see Section A2) and aRPD is usually also measured (see Section A3).



Example sediment plate array from Peg 1 (see Fig. A1) , in this case representing sediment plates installed along a 30m upstream boundary of a fine scale site


Measuring a sediment plate using a probe and straight edge.

C. DATA RECORDING, QA/QC AND ANALYSIS

All sediment and macrofaunal samples sent to analytical laboratories are tracked using standard Chain of Custody forms, and results are transferred electronically from the laboratory to avoid transcription errors. Field measurements (aRPD, sediment plate depth) and site metadata are recorded electronically in templates custom-built using Fulcrum app software (www.fulcrumapp.com). Pre-specified data entry constraints in the app (e.g., with minimum or maximum values for each data type) minimise the risk of erroneous data recording.

Excel sheets that contain the above data are imported into the software R 4.2.3 (R Core Team 2023) and assigned sample identification codes. All summaries of univariate responses (e.g., sediment analyte concentrations, macrofauna abundances, sediment plate depths) are produced in R, including tabulated or graphical representations of the data. Specific further data handling and analysis approaches for the different data types are describe below.

1. Sediment plates

Sediment plate data are compiled to display: (i) cumulative change in sediment depth since baseline plate installation; (ii) annual sedimentation rate, which involves an adjustment to annualise the plate depth at the time of each survey to 12 months; and (iii) longer-term sedimentation rate (e.g., 5-yr or overall average).

2. Sediment quality

Where sediment quality data include values less than analytical detection limits, half of the detection limit value is used for data averaging, according to standard convention.

3. Macrofauna

Sediment-dwelling macrofauna data preparation and analysis involves multiple steps, as follows:

- The data are screened to remove species that are not regarded as a true part of the macrofaunal assemblage; these are planktonic life-stages and non-marine organisms (e.g., freshwater drift).

- To facilitate comparisons with among surveys and other estuaries, cross-checks are made to ensure consistent naming of species and higher taxa. For this purpose, the adopted name is that accepted by the World Register of Marine Species (WoRMS, www.marinespecies.org).
- Macrofauna response variables are derived which include richness and abundance by species and higher taxonomic groupings, and calculation of scores for the biotic health index AMBI (Borja et al. 2000; Borja et al. 2019).
- AMBI scores reflect the proportion of taxa falling into one of five eco-groups (EG) that reflect sensitivity to pollution, ranging from relatively sensitive (EG-I) to relatively resilient (EG-V), and their calculation involves the following steps:
  - To meet the criteria for AMBI calculation, macrofauna data are reduced to a subset that includes only adult 'infauna' (those organisms living within the sediment matrix), which involves removing surface dwelling epibiota and any juvenile organisms.
  - AMBI scores are calculated based on standard international eco-group classifications where possible (http://ambi.azti.es). To reduce the number of taxa with unassigned eco-groups, international data are supplemented as appropriate with more recent eco-group classifications for New Zealand (Keeley et al. 2012; Robertson et al. 2016; Robertson 2018).
  - AMBI scores are not calculated for macrofauna cores that do not meet operational limits defined by Borja et al. (2012), in terms of the percentage of unassigned taxa (>20%), or low sample richness (<3 taxa) or abundances (<6 individuals).
- Multivariate analyses of macrofaunal community data are undertaken using the software package Primer v7.0.13 (Clarke et al. 2014), with the following being the main elements:
  - Prior to multivariate analysis, macrofaunal abundance data are transformed (e.g., square root) to down-weight the influence of the dominant species or higher taxa.
  - Patterns in site similarity as a function of macrofaunal composition and abundance are assessed using an 'unconstrained' non-metric multidimensional scaling (nMDS) ordination plot, based on pairwise Bray-Curtis similarity index scores among samples.
  - Overlay vectors and bubble plots on the nMDS are used to visualise relationships between multivariate biological patterns and sediment quality data.
  - Other Primer or PERMANOVA (Anderson et al. 2008) procedures (e.g., Bio-Env, DistLM) are used to evaluate the suite of sediment quality variables that best explain the macrofauna ordination pattern.

D. METHODS REFERENCES

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APPENDIX 2. GPS COORDINATES FOR FINE SCALE SITES AND SEDIMENT PLATES

FINE SCALE SITES

Site	NZTM East	NZTM North
A	1636650	5442096
C	1635390	5442679
D	1638231	5443005

SEDIMENT PLATES

Site	Plate	NZTM East	NZTM North
A	1	1636652	5442086
	2	1636649	5442084
	3	1636639	5442079
	4	1636634	5442077
B	1	1637689	5442791
	2	1637692	5442794
	3	1637701	5442799
	4	1637701	5442805
C	1	1635335	5442716
	2	1635338	5442720
	3	1635346	5442725
	4	1635352	5442728
D	1	1638235	5443009
	2	1638238	5443012
	3	1638246	5443018
	4	1638249	5443021
BS	1	1638748	5443121
	2	1638750	5443126
	3	1638751	5443136
	4	1638752	5443141
CS	1	1635278	5443207
	2	1635273	5443205
	3	1635266	5443198
	4	1635262	5443196

APPENDIX 3. SEDIMENTATION AND GRAIN SIZE DATA FOR SEDIMENT PLATES

The first (baseline) depth was measured at the time of plate installation (Jan 2020 Site D, otherwise March 2018). Mud, sand and gravel values are based on a single sample composited from subsamples taken next to each plate.

Date	Site	Plate 1	Plate 2	Plate 3	Plate 4	Mud	Sand	Gravel	aRPD
2018-03-27	A	40	28	46	37	80.3	19.5	0.2	-
2018-12-11	A	40	34	50	46	84.6	15.2	0.2	10
2020-01-07	A	39	34	52	45	84.2	15.4	0.3	3
2021-01-26	A	46	50	34	43	80.4	19.4	0.2	3
2022-01-22	A	44	34	56	47	82.8	17.1	0.2	4
2022-09-01	A	69	61	76	67	96.3	3.7	<0.1	15
2023-04-04	A	71	60	79	68	87.2	12.8	<0.1	15
2018-03-17	B	77	63	81	86	5.8	93.9	0.3	-
2018-12-11	B	67	41	58	81	7.6	92.0	0.4	30
2020-01-08	B	65	47	65	85	11.0	88.9	0.1	-
2021-01-26	B	69	52	68	91	7.7	90.4	1.9	5
2022-09-01	B	115	91	82	-20	4.3	95.4	0.2	95
2023-04-04	B	97	69	66	49	7.5	92.4	0.1	40
2018-03-17	BS	63	58	62	75	23.7	75.9	0.5	-
2018-12-11	BS	65	61	69	80	24.2	74.0	1.7	22
2020-01-08	BS	66	63	72	82	21.2	78.4	0.4	25
2021-01-26	BS	65	62	69	79	20.0	78.6	1.4	20
2022-01-22	BS	64	61	70	81	18.9	80.7	0.4	25
2022-09-01	BS	74	68	79	90	75.5	24.2	0.3	40
2023-04-04	BS	68	64	69	82	29.6	67.8	2.6	20
2018-03-28	C	55	34	48	54	23.1	76.2	0.7	-
2018-12-09	C	53	51	35	57	21.0	77.8	1.3	25
2020-01-07	C	53	49	37	55	20.7	77.7	1.5	32
2021-01-26	C	55	36	48	56	19.4	78.8	1.8	30
2022-01-22	C	59	37	50	58	20.4	78.5	1.1	25
2022-09-01	C	88	71	57	72	95.2	4.4	0.4	30
2023-04-04	C	67	49	61	68	37.4	60.8	1.8	25
2018-03-28	CS	46	45	42	36	18.7	81.1	0.2	-
2018-12-09	CS	49	42	44	40	22.2	77.6	0.2	30
2020-01-07	CS	43	45	47	36	21.6	77.9	0.5	40
2021-01-26	CS	44	42	43	33	13.6	84.5	1.8	25
2022-01-22	CS	44	43	45	38	16.1	83.1	0.8	30
2022-09-01	CS	52	49	50	41	56.2	43.1	0.7	30
2023-04-04	CS	47	47	42	32	24.3	75.5	0.2	30
2020-01-08	D	63	55	53	63	4.0	96.0	0.1	35
2021-01-26	D	67	66	56	62	7.7	91.8	0.5	25
2022-01-22	D	89	77	62	66	8.7	88.9	2.5	35
2022-09-01	D	79	71	64	72	20.7	78.2	1.1	35
2023-04-04	D	88	69	71	69	13.2	85.2	1.6	30

APPENDIX 4. SEDIMENT QUALITY RAW DATA

Values based on a composite sample within each of Zone X (reps X1-3), Y (reps Y4-6) and Z (reps Z7-10), except for aRPD for which the mean and range is shown for 10 replicates.

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	2019	X	0.2	14	85.7	0.83	900	600	15.0 (10 to 25)	5.9	0.030	53	16.9	<0.02	24	5.7	53
		Y	<0.1	14	85.9	0.88	1000	650	25.0 (15 to 30)	6.6	0.032	55	17.7	0.02	25	6.1	55
		Z	0.2	17.6	82.2	0.83	900	590	18.8 (15 to 25)	6.0	0.033	54	17.2	<0.02	25	6.0	53
	2020	X	0.2	13.5	86.2	0.82	800	660	10	5.8	0.039	56	19.8	0.02	27	6.3	60
		Y	0.3	15.4	84.3	0.86	800	680	8.3 (10 to 5)	5.8	0.034	56	19.4	0.03	27	6.2	60
		Z	0.4	17.4	82.2	0.77	800	690	11.3 (10 to 15)	5.8	0.035	53	18.8	0.02	26	6.0	59
	2021	X	0.1	13.1	86.8	0.73	700	550	10	5.2	0.031	50	16.7	<0.02	23	5.8	50
		Y	0.3	17.7	82	0.74	800	570	11.7 (10 to 15)	5.7	0.031	52	16.3	<0.02	23	5.9	51
		Z	2.1	20.5	77.4	0.74	700	570	9.5 (10 to 8)	5.7	0.035	52	16.8	0.02	23	5.7	51
	2023	X	<0.1	9.8	90.2	1.08	1000	580	18.3 (17 to 20)	6.7	0.051	56	26.0	0.03	25	7.3	57
		Y	1.8	12.1	86.1	0.99	900	590	15.0 (10 to 20)	6.0	0.043	52	23.0	0.03	23	6.8	55
		Z	0.1	14.4	85.5	0.91	900	580	15.0 (10 to 20)	6.8	0.046	54	23.0	0.03	24	7.0	55
C	2019	X	0.9	78.4	20.7	0.18	<500	570	30.7 (28 to 32)	4.6	0.024	54	10.1	<0.02	20	3.3	50
		Y	1.9	78	20.2	0.17	<500	560	28.7 (28 to 30)	4.9	0.025	55	9.7	<0.02	20	3.2	50
		Z	1	77	22	0.19	<500	570	28.3 (25 to 30)	4.9	0.020	52	10.3	<0.02	20	3.4	51
	2020	X	1.5	78.8	19.7	0.24	<500	630	29.3 (25 to 33)	4.8	0.025	54	12.3	<0.02	23	3.7	59
		Y	2	77.1	20.9	0.2	<500	660	33.0 (28 to 43)	5.1	0.025	54	12.4	<0.02	23	3.8	59
		Z	1	77.3	21.6	0.24	<500	630	33.0 (31 to 36)	5.2	0.024	54	12.5	<0.02	23	3.9	60
	2021	X	1.6	79.5	18.9	0.22	<500	540	24.3 (20 to 30)	4.5	0.024	53	10.6	<0.02	20	3.6	50
		Y	1.6	80	18.4	0.21	<500	540	31.7 (26 to 36)	5.5	0.025	55	11.2	<0.02	20	3.7	53
		Z	2	80.3	17.7	0.2	<500	550	29.8 (19 to 36)	5.1	0.021	51	10.6	<0.02	19	3.8	50
	2023	X	0.9	60.1	39.1	0.51	500	570	21.7 (15 to 25)	6.4	0.039	51	17.6	0.02	20	5.7	53
		Y	0.6	61.7	37.7	0.43	600	600	20.0 (15 to 25)	6.3	0.038	53	17.4	<0.02	21	5.6	55
		Z	0.7	60.9	38.4	0.39	<500	590	22.5 (20 to 25)	5.8	0.031	47	15.3	<0.02	19	5.2	51
D	2020	X	<0.1	98.1	1.9	0.2	<500	630	28.3 (20 to 35)	4.6	0.018	56	10.6	<0.02	24	2.6	58
		Y	<0.1	95.4	4.6	0.13	<500	640	26.3 (20 to 36)	4.7	0.021	52	10.6	<0.02	23	2.7	56
		Z	0.2	94.4	5.4	0.16	<500	650	18.3 (15 to 20)	4.8	0.020	52	10.9	<0.02	23	2.8	56
	2021	X	<0.1	96.8	3.2	0.09	<500	530	33.7 (33 to 35)	4.6	0.018	52	8.9	<0.02	20	2.5	48
		Y	0.2	94.8	5	0.13	<500	550	33.7 (30 to 36)	4.6	0.019	52	9.3	<0.02	20	2.7	47
		Z	1	92.9	6	0.13	<500	550	19.8 (10 to 27)	4.5	0.022	52	9.3	<0.02	19	2.7	47
	2023	X	<0.1	94.5	5.4	<0.13	<500	600	46.7 (35 to 60)	5.2	0.019	58	10.6	<0.02	21	2.9	50
		Y	0.2	90.7	9.1	<0.13	<500	600	35.0 (30 to 40)	4.8	0.020	50	10.0	<0.02	19	2.8	48
		Z	1.8	86.3	12	0.19	<500	600	27.5 (25 to 30)	5.2	0.028	56	11.6	<0.02	21	3.3	50
									DGV	20	1.5	80	65	0.15	21	50	200
									GV-high	70	10	370	270	1	52	220	410

APPENDIX 5. MACROFAUNA CORE DATA 2019-2023

Date summered across cores for each site and year. EG = eco-group.

Main group	Taxa	EG	A19	A20	A21	A23	C19	C20	C21	C23	D20	D21	D23
Amphipoda	Paracalliope novizealandiae	I	1	3	2	1	4	16	11	6	1	1	4
Amphipoda	Paracorophium excavatum	IV		1	1					1			
Amphipoda	Torridoharpinia hurleyi	I	36	44	14	7	13	29	37	2	13	13	4
Anthozoa	Anthopleura hermaphroditica	III					1				26	29	37
Bivalvia	Arthritica sp. 5	III			6				2	1			
Bivalvia	Austrovenus stutchburyi	II	5		1	3	43	50	69	54	35	28	101
Bivalvia	Hiatula spp.	I									4		1
Bivalvia	Macomona liliana	II			2	1	43	49	49	29	91	57	50
Bivalvia	Paphies australis	II			1								
Bivalvia	Theora lubrica	III	2	6	4	4							
Cirripedia	Austrominius modestus	II						1			25	6	7
Decapoda	Alpheus socialis	II				2				1			
Decapoda	Decapod megalopa	II	1										
Decapoda	Halicarcinus whitei	III					3	3	3		3	3	5
Decapoda	Hemiplax hirtipes	III	17	10	3	18	4	7	8	8		2	2
Gastropoda	Amphibola crenata	III			2								
Gastropoda	Cominella glandiformis	III		2			1		1	1	3	3	6
Gastropoda	Diloma subrostratum	II									2	3	
Gastropoda	Notoacmea spp.	II							1		1		2
Gastropoda	Zeacumantus lutulentus	II						1	5	1	2	2	1
Holothuroidea	Taeniogyrus dendyi	I									1		
Nemertea	Nemertea	III					1	1			2		
Nemertea	Nemertea sp. 1	III				1		1		2	2	1	3
Nemertea	Nemertea sp. 3	III						2	8	1	7	1	6
Oligochaeta	Oligochaeta	V					1			1	1		
Polychaeta	Aonides trifida	I									22	7	21
Polychaeta	Axiothella serrata	II	1				33	57	29		17	10	2
Polychaeta	Boccardia syrtis	II		7	4	27		1	1				
Polychaeta	Capitella sp. 1	V									2		1
Polychaeta	Glyceridae (juv)	II		3	2	2	10	9	8		2		
Polychaeta	Heteromastus filiformis	IV	3	2	2	2	4	1	8	10			
Polychaeta	Lagis australis	III			1		3	2		1			
Polychaeta	Magelona dakini	I								1			
Polychaeta	Microspio maori	I									2	1	3
Polychaeta	Nereididae (juv)	III	1			11	4	6	6	23	3	2	11
Polychaeta	Nicon aestuariensis	III	3	2	12	2	3		6	3			
Polychaeta	Orbinia papillosa	I									10	13	2
Polychaeta	Paradoneis lyra	III					186	47	42	139			
Polychaeta	Perinereis vallata	III				2							
Polychaeta	Prionospio aucklandica	III		21	2	24	39	63	79	257	16	6	70
Polychaeta	Scolecolepides benhami	IV		2	1	1	1	2	4	1	2		1
Polychaeta	Spionidae	III	3										

APPENDIX 6. MACROFAUNA AND ENVIRONMENTAL DRIVERS

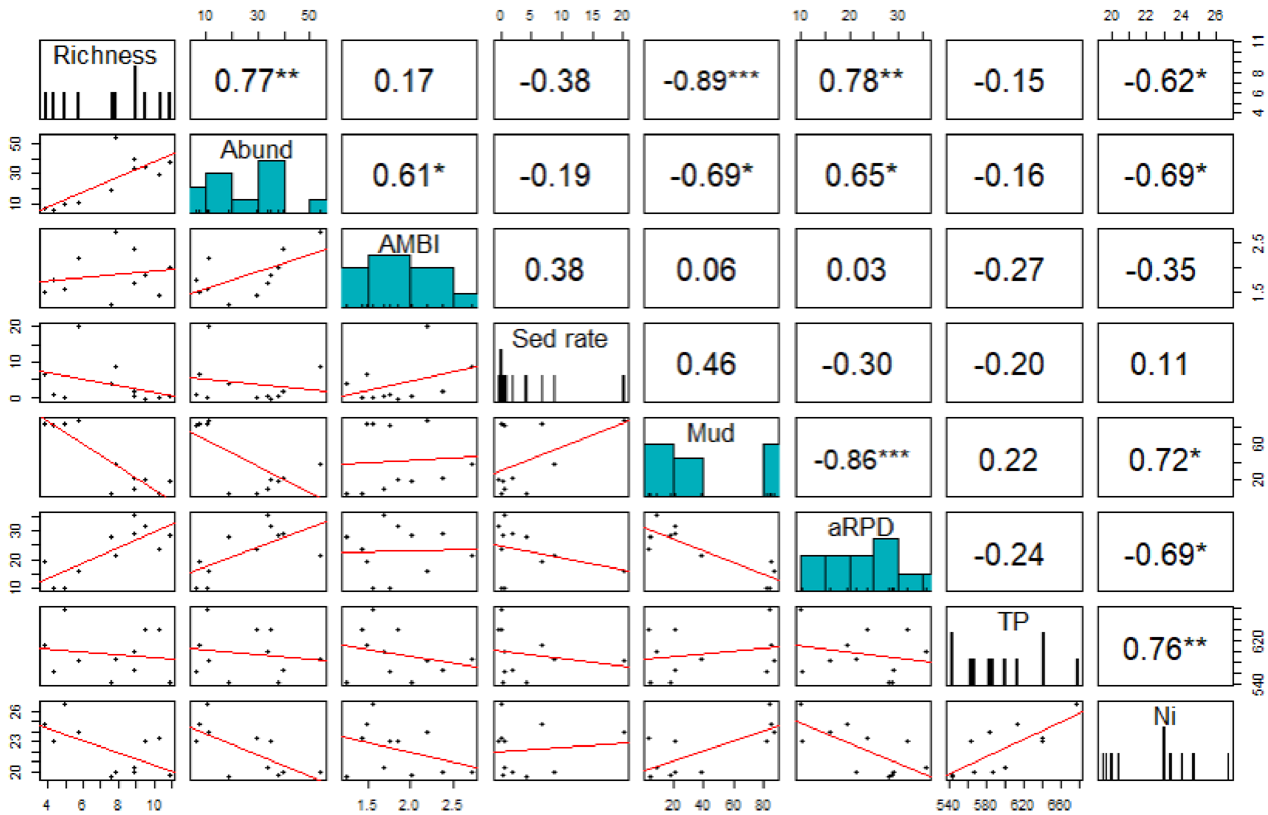
A. Reduced dataset for site-level analysis.

Sedimentation (Sed) rate expressed as annualised change from the preceding survey, except for 2023 which is benchmarked to pre-flood data from January 2022.

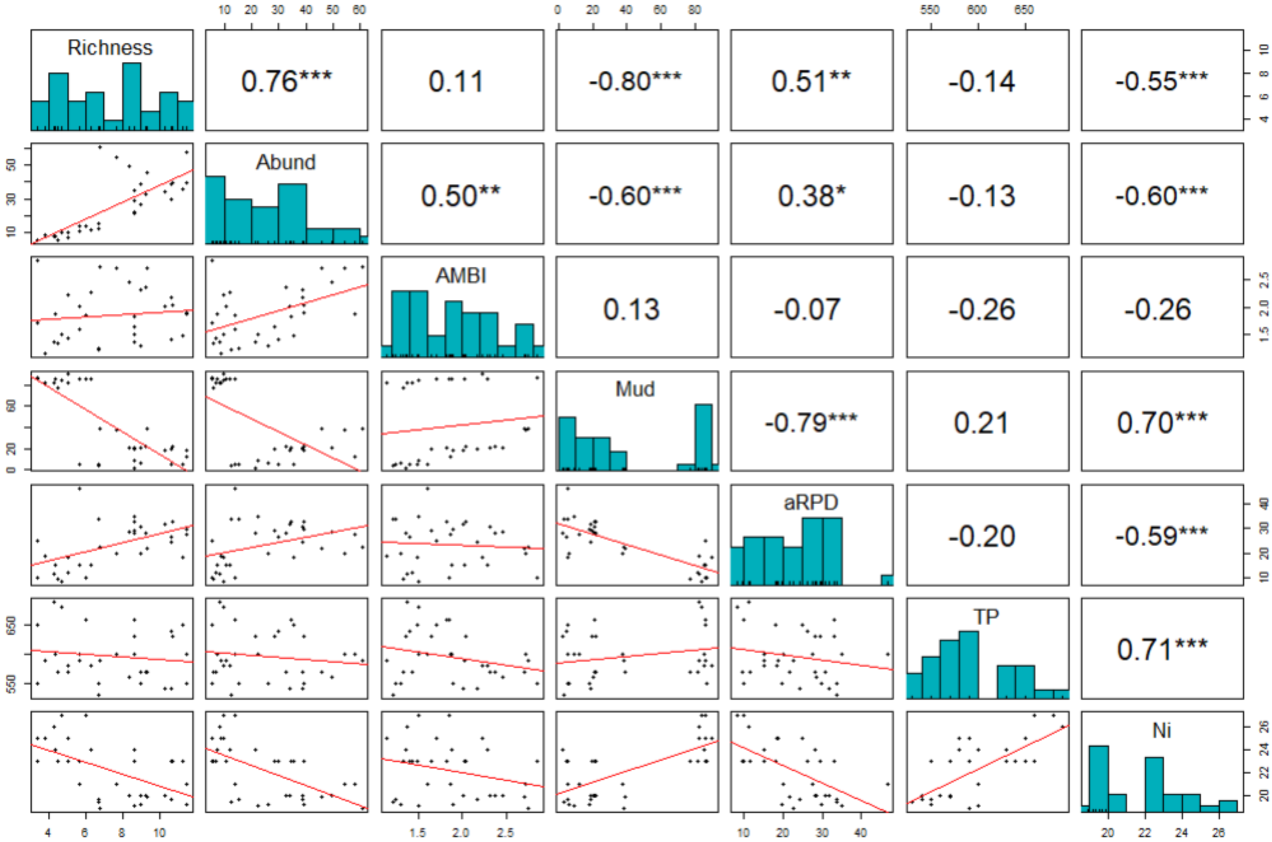
Year	Site	Richness	Abund	AMBI	Sed rate	Mud	Sand	aRPD	TP	Ni
2019	A	3.8	7.3	1.5	6.7	84.6	15.2	19.5	613	25
2020	A	4.9	10.3	1.6	0.0	84.2	15.4	10.0	677	27
2021	A	4.3	6.0	1.8	0.7	82.1	17.1	10.3	563	23
2023	A	5.7	10.8	2.2	20.0	87.3	12.1	16.0	583	24
2019	C	8.9	39.7	2.4	1.8	21.0	77.8	29.1	567	20
2020	C	9.5	34.8	1.9	-0.5	20.7	77.7	31.9	640	23
2021	C	10.9	37.7	2.0	0.2	18.3	79.9	28.7	543	20
2023	C	7.8	54.3	2.7	8.7	38.4	60.9	21.5	587	20
2020	D	10.3	29.5	1.4	0.0	4.0	96.0	23.7	640	23
2021	D	7.6	18.8	1.3	4.0	4.7	94.8	28.1	543	20
2023	D	8.9	34.0	1.7	0.6	8.8	90.5	35.5	600	20

B. Pearson correlation analysis of univariate macrofauna responses (richness, abundance, AMBI) with sediment quality and sedimentation (site-level only) variables.

(i) Fine scale site-level analysis. Data averaged across zone as per Table in A above. In a separate analysis of the site-level data for all surveys where sedimentation rate was measured (i.e. in addition to fine scale survey years), there was a moderate positive association with sediment % mud (Pearson r^2=0.50, p<0.01).



Fine scale zone-level analysis: Sediment quality data are from Appendix 4, with macrofauna index values (richness, abundance, AMBI) derived by averaging across replicates with each of zone X (cores X1-X3), Y (cores Y4-Y6) and Z (cores Z7-Z10) to match the composite sediment samples (see schematic in Fig. 2 of main report). Sedimentation rate not applicable as it is a site attribute.



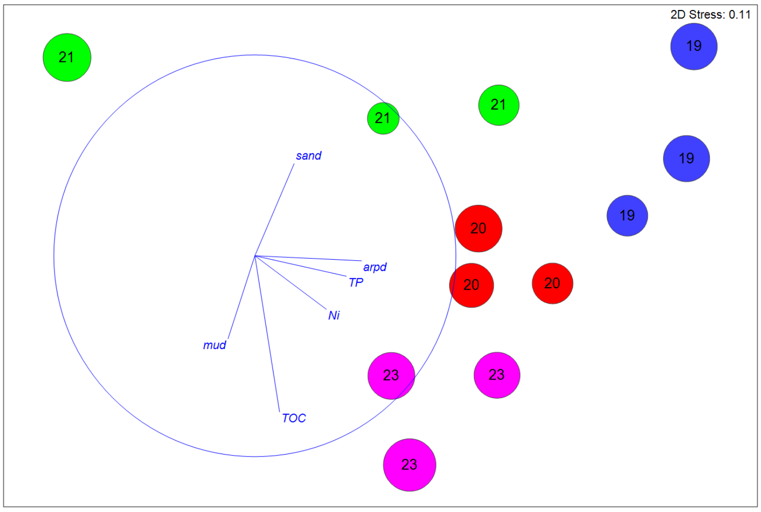
C. Multivariate macrofauna-environment matching

PRIMER BEST procedure: Excluding highly correlated variables. Variable and Spearman correlation as per following table:

Variable	Site-level correlation	Zone-level correlation
Mud (correlated with sand & TOC)	0.862	0.786
aRPD	0.526	0.428
Ni	0.338	0.311
Sedimentation rate	0.035	na
TP	-0.141	0.009

D. Within-site MDS plots based on macrofauna 'zone' data that match the composite sediment samples. Bubbles are scaled to % mud relative to the range within each site. Vectors illustrate direction and strength of association with sediment quality. BEST results shown Spearman rank correlation coefficients between sediment quality and macrofauna composition.

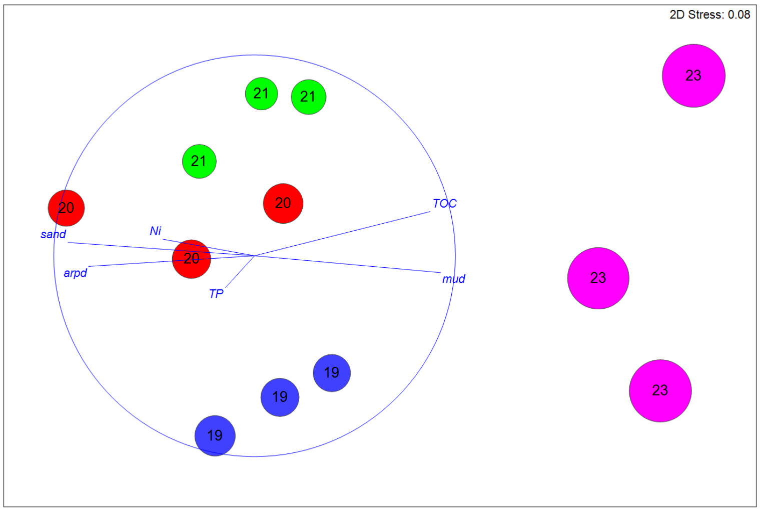
Site A



BEST results

Corr.	Variable
0.299	TOC
0.172	aRPD
0.155	Mud
0.011	TP
-0.091	Ni

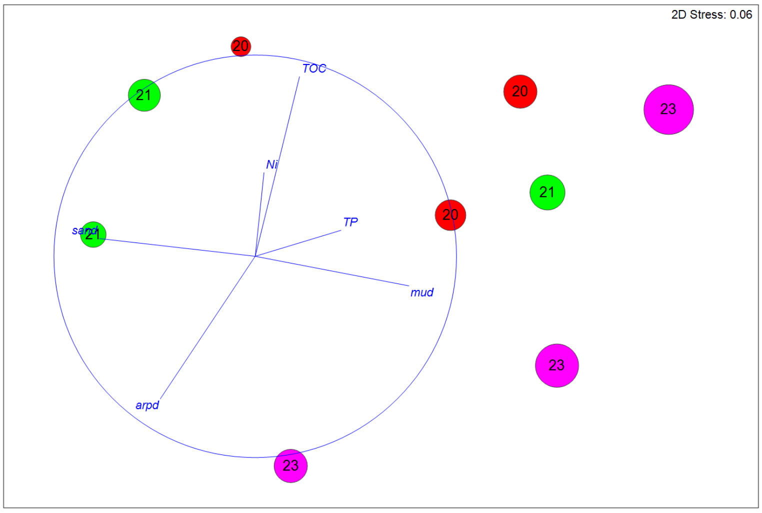
Site C



BEST results

Corr.	Variable
0.769	Mud
0.761	TOC
0.533	aRPD
0.082	Ni
-0.056	TP

Site D



BEST results

Corr.	Variable
0.257	TOC
0.239	Mud
0.138	aRPD
-0.035	TP
-0.185	Ni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