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Assessment of effects of Wairoa District Council's existing intertidal sewage discharge on benthic sediment characteristics and ecology – Wairoa Estuary

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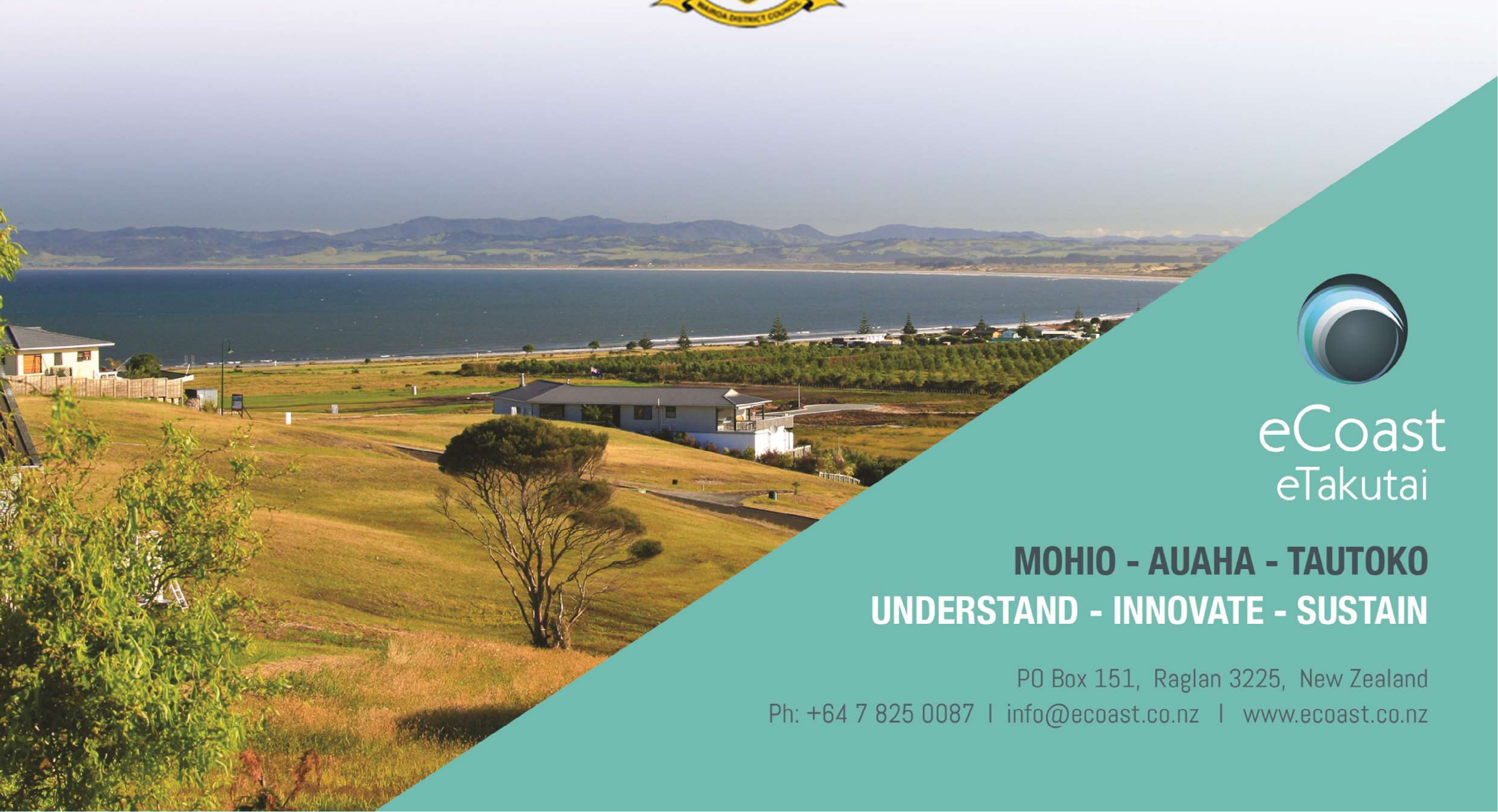

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Assessment of effects of Wairoa District Council's existing intertidal sewage discharge on benthic sediment characteristics and ecology – Wairoa Estuary

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Executive Summary

- The purpose of the study was to evaluate any effects on sediment geochemistry and infaunal biota (ecology) that could potentially be attributable to the Wairoa community wastewater discharge.
- A sediment geochemistry (contaminant) and ecological survey of the Wairoa Estuary was undertaken in April 2018 relating to re-consenting of treated sewage disposal via a subtidal outfall located near the estuary entrance. Historic surveys were conducted in 1996 (prior to the current consent); 2007 and 2011. The 2018 survey differed to historic surveys due to an increase in sampling sites; an increase from 3 to 10. Sediment geochemistry was also evaluated immediately adjacent to an overflow in the immediate intertidal zone along the western bank of the estuary.
- Sediment geochemistry characteristics were found to be highly variable across sampling sites in 2018, including across surveys (1996-2018). Sites with highest silt content > 60% in 2018 were G and H, closest to the entrance sites A, E and F located to the south of the discharge; and, Site C (reference). There was also a lack of a clear trend for both silt and organic content in relation to increasing or decreasing distance away from the outfall and thus no strong evidence that the outfall is impacting the immediate benthos. Sediments adjacent the intertidal overflow were found to have higher contaminants than the estuarine sites sampled exceeding ANZECC (2000) Interim Sediment Quality Guidelines low threshold values for lead.
- For sediment-bound contaminants and nutrients from estuarine samples there was generally no consistent pattern across sites, the exception being Site B (100 m to the north of the outfall) that generally had higher sediment bound contaminants than other sampling sites. However, all contaminant levels were below ANZECC (2000) Interim Sediment Quality Guidelines low threshold values and thus adverse ecological effects due to contamination would be unlikely.
- A total of 17 infaunal species were enumerated from benthic sampling. Species community composition was found to differ among survey sites with historical sites A, B, and C having similar species composition and higher species diversity compared to the new sites added to the programme. Numerically dominant species recorded at sites A, B, and C are typically considered to be synonymous with degraded/impacted environments (local nutrification). This was also true for the additional new sites, although abundances are lower than at Sites A-C and pipi were found to occur at much greater densities at sites further away from the outfall.

- Evaluating impacts of the outfall on benthic effects is generally difficult given the low species diversity and wider degraded nature of the lower Wairoa Estuary, as well as the initial monitoring only having a few sites likely within the zone of impact (which is evident due to the differences found at Sites A-C in comparison to the rest of the locations surveyed in 2018), and sparse temporal resolution; the 2018 survey was only the 4th since 1996.. The previous monitoring/sampling design was focussed on the mixing zone next to the outfall and so did not consider impacts on the wider estuarine environment. This assessment has attempted to address this issue with the addition of 7 new sites located at increasing distances away from the subtidal outfall.

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1 Introduction

1.1 Background

In 1999, a 20-year resource consent (CD940404W) was granted to Wairoa District Council (WBD), to discharge treated sewage into the lower Wairoa Estuary via a subtidal outfall. The maximum permitted discharge of sewage effluent as authorised by the resource consent is 5,400 m^3 d^{-1}. Additional conditions of the consent relate to the allowable timing of effluent discharge relative to tidal state, time of day and river mouth closure. The Wairoa wastewater treatment system requires a renewal consent by May 2019.

The outfall was constructed in 1981, concomitant with the Council Pilot Hill oxidation ponds. Effluent exits the final stage aerobic pond and is gravity fed to the outfall discharge port (LEI, 2015). The discharge port is positioned in the low intertidal zone, approximately 150 m from the western shoreline adjacent the Kopu Rd and Fitzroy Street intersection (Figure 1). Over the lifespan of the present consent there have been issues with overflow from pumping stations attributable to the very high rate of stormwater entry into the sewer network during heavy and/or prolonged rainfall events. Similarly, the rate of sewage disposal is restricted and the WDC has to store effluent when the mouth of the estuary is closed. However, when storage capability is exceeded effluent has to be discharged into the estuary during these "regulated flow" periods.

The WDC is not formally required under the existing consent conditions of CD940404W to undertake ecological and sediment contaminant monitoring surrounding the outfall; however, WDC has commissioned a variety of environmental studies as they relate to the sewage outfall and wider estuarine environment. With reference to the present study, these have included ecological surveys undertaken in 1996 (Larcombe 1996), 2007, 2011 and 2017 (EAM 2007, 2011; Triplefin, 2018) and a dye dilution study in 2007 (Barter 2007).

The Wairoa Estuary is a river mouth estuary approximately 9,700,000 m^3 in volume with two bar-built lagoons the Whakamahi and Ngamotu at its mobile entrance. The estuary marks the end point of Te Wairoa Hōpūpū Hōnengenenge Mātangi Rau awa which sits within a catchment area of 356,300 km^2 of which 264,547 km^2 is within the Hawke's Bay regional boundary – this is Hawke's Bay's largest catchment. The river is formed by the confluence of the Hangaroa and the Ruakituri rivers which meet at Te Reinga Falls, and flows 65 km to drain into the sea near Wairoa Township. Other rivers of note include the Waiau that meets the Wairoa River adjacent Frasertown approximately 10 km from the mouth. Main uses of the catchment include sheep and beef farming (34.5%) with indigenous forests accounting for approximately 36 %. Smaller horticultural practices are common to the area as well. The

Wairoa Township is a dominant feature of the lower river. Consented industrial activities associated with Affco Meat works (Wairoa) and Silver Fern Farms meat works (Frasertown) discharge waste into the Wairoa River/Estuary.

An early summary document of New Zealand's estuaries (McClay 1976) inferred that the environmental condition of the Wairoa Estuary had worsened between 1965 and 1976 and in 1976 was classified as moderately polluted. This is still likely the case today. Hawke's Bay Regional Council (HBRC) established formal benthic monitoring within the estuary in 2011 at site approximately 600 m upstream of the AFFCO outfall. Results from the monitoring suggest that sediment contaminants (trace metals, nutrients) are not problematic within the estuary, whereas impacts associated with sedimentation and recreational contact for the upper river adjacent Wairoa Water Ski Club were (HBRC, 2016).

The Wairoa Estuary comprising the lagoon, sandspit, and mudflats is a Department of Conservation Wildlife management reserve (Cheyne and Addenbrook, 2002) and is designated by HBRC as a significant conservation area due to its biodiversity values. Birds of significance associated with the estuary include Australasian Bittern (*Botaurus poiciloptilus)* Marsh Crake (*Porzana pusilla)* and Spotless Crake (*Porzana tabuensis)* and the Royal Spoonbill (*Platalea regia)*. The Wairoa Estuary, as for other estuarine areas within Hawke's Bay serves as a nursery ground for flounder, short and long-finned eel, and inanga. Both cockles and pipi occur within the estuary proper and fresh water mussels (kakahi) were also historically abundant (Haggitt and Wade 2016).

1.2 Purpose

This report documents the third ecological and sediment geochemistry/pollutant study since the granting of resource in 1999. The purpose of the study was to evaluate any effects on sediment geochemistry and infaunal biota (ecology) and) that could potentially attributable to the outfall together with a comparison of trends with previous surveys i.e., spanning from 1996-2011. A review of the findings of these earlier monitoring surveys was undertaken in 2018, which concluded that the monitoring design was insufficient to provide a useful indication of the magnitude and extent of the impacts of the outfall and the ecological health of the wider estuary (eCoast, 2018:A3D2). From this review it was recommended that additional ecological monitoring would benefit from an increase in sampling resolution and that a study of the hydrodynamics of the Wairoa Estuary mouth would also be warranted.

2 Methodology

For the most-part methodological procedures were kept consistent with that of previous 2007, 2011, 2017 EAM Ltd and Triplefin Ltd studies in terms of size of sample cores used and a resurvey of identical sampling sites. For the current survey, however, modifications were made to the number of sampling sites including sample replication for sediment geochemistry. The 1999, 2007, 2011 and 2017 studies sampled three sites two impact (Sites A and B) and one control (Site C) in accordance with a historical survey design (Larcombe, 1996). Specifically, Site A is located ~100m south of the outfall discharge point; Site B located ~100m north of the outfall discharge point; and Site C located ~ 500 m north of the outfall discharge point (Figure 1). Previous studies have classified Site C as a reference/control site, as it is considered to be outside of the "zone of influence" of the outfall. In 2018, an additional 7 sites (D-J) were added to the sampling programme for the current assessment in order to 1) evaluate potential impacts of the outfall at distances >100m away from the discharge point, and 2) provide a broader context on the present-day environmental state of the of the Wairoa Estuary (refer to Appendix 1 for site co-ordinates).

Figure 1 Site locations sampled within the lower Wairoa estuary in April 2018, including the approximate location of the wastewater outfall. Note sites A and B are considered as historical impacted sites and Site C is a historical control/reference site. Sites D-J were new to the monitoring programme in 2018.

2.1 Sediment geochemistry - RDPL and contaminant levels

Benthic sampling across the 10 sites took place at low tide on 19 April 2018. To maintain general consistency with previous sampling events a similar approach was employed for sample acquisition. A handheld Garmin eTrex GPS unit was used to locate each site (± 3m). At each site, three replicate sediment cores were collected using a PVC 60 mm (internal Φ) x 150 mm long corer (sleeve). Individual cores were sampled and retrieved by pushing the corer into the sediment to a depth of 150mm, digging around the outside of the corer and placing a hand over the bottom of the corer when extracting the core from the surrounding sediment in order to maintain the integrity of the core profile. Cores were then removed from the PVC sleeve intact onto a silver tray and split vertically (Figure 2A and 2B). Each core was then visually assessed for the presence/absence of anoxic layers and areas within the core and the redox potential discontinuity layer (RPDL) measured to +/- 1mm. The top 5 cm of each core was then combined to form a composite sample and placed into pre-labelled containers and transported on ice to Hill's laboratory (Hamilton) for analyses. Specific analyses (tests) included organic matter; particle size analysis by weight; Total Recoverable Phosphorus, Total Nitrogen, and trace metals As, Cd, Cr, Cu, Ni, Pb, Zn, and Hg. Additional replicate sediment samples were also taken adjacent to an overflow in the immediate intertidal at the base of the western bank adjacent the Kopu Rd/Fitzroy St intersection (Figure 2C).

Figure 2 A) Replicate sediment core; B) sediment core split in half to evaluate RDL layer; C) and D) overflow adjacent Kopu Rd/Fitzroy St intersection.

2.2 Ecological - Infaunal sampling

Infaunal samples (cores) were collected using a circular PVC 130mm (internal Φ) x 200mm long core (total area 0.013m^2). Samples were collected by pushing the core into the sediment to a depth of 150 mm and digging down the outside of the core, placing a hand over the bottom, extracting the core and intact sample. The sample was then removed from the sleeve and immediately washed into a 0.5 mm sieve. Biological material retained on the sieve was further washed into labelled (time, site #) jars containing 70% isopropyl alcohol in seawater. Rose Bengal solution was added to each sample, and left for several hours for the satin to be absorbed by the biological material. Species identification was undertaken at the Leigh Marine Laboratory, by a qualified technician. Briefly, individual samples were placed into sorting trays and biological material removed from inorganic debris. Biological material was then identified to the lowest possible taxonomic level using a Lecia MS5 dissecting microscope (35 ×

objective). Data were then compiled into MS Excel and checked for accuracy using QA/QC procedures.

2.3 Data analysis

2.3.1 Sediment RDPL and contaminant levels.

Results for sediment RPDL and contaminant levels are presented either graphically or in table format for comparative purposes. No formal statistics was run on the physical data, particularly temporal comparisons as in 2018 sediment samples were a composite (following the procedures of Robertson, 2002) as opposed to analysing replicate samples as was done for previous surveys. Comparisons to earlier studies and broad-scale trends are presented where appropriate.

2.3.2 Ecological data

Ecological data consisted of counts of different infaunal species and analysis was performed on two main datasets: 1) 2018 data that incorporated all 10 sites, and 2) the temporal dataset that consisted of sites A, B, and C for 1999-2007; 2011, 2017 and 2018 surveys. Data are primarily presented as either plots of diversity, abundance, or community ordinations. A combination of multivariate and univariate statistical tests was used to analyse various datasets with the majority of statistical analyses undertaken using PRIMER-E statistical software (Clarke and Warwick 2001) and associated routines, particularly PERMANOVA[1] (Anderson *et al*. 2008). In recent years PERMANOVA has become increasingly popular in the analysis of both univariate and multivariate datasets relating to environmental impact assessments.

2018 survey

2.3.2.1 Diversity

To evaluate patterns in diversity across the ten estuarine sampling sites evaluated in this study the Primer routine “Diverse” was employed to calculate species richness (S), Shannon-Wiener diversity (*H*’) and species evenness estimates (*J*’). Collectively these tests are used to

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[1] PERMANOVA (permutational multivariate analysis of variance) is used for the analysis of univariate or multivariate data in response to factors, groups or treatments in an experimental design.

describe how species within communities are comprised and distributed. All data were (log(x+1) transformed to ensure that commonly occurring species did not overly dominate the analysis. To test for differences in species richness across sites one-way PERMANOVA was run on Log (x+1) transformed data using a Euclidean distance measure (which is equivalent to traditional ANOVA (see Anderson *et al*. 2008) using 4999 permutations, α = 0.05.

2.3.2.2 Community composition

To test for differences in community composition across sites (Site) again one-way PERMANOVA was run on log(x+1) transformed data and a Bray-Curtis similarity measure using 4999 permutations α = 0.05. Principal coordinates analysis (PCO) (Anderson et al. 2008) was also utilised to support multivariate PERMANOVA results and visualise patterns among sites in multivariate space, again using log(x+1) transformed data and a Bray-Curtis similarity measure. To further compliment these analyses canonical analysis of principal coordinates (CAP, Anderson and Willis, 2003; Anderson et al., 2008) was undertaken. CAP is a constrained ordination method that is effectively a PCO followed by a traditional canonical discriminant analysis on an appropriate number of the PCO axes. It allows one to find an axis through the multivariate cloud that is best at discriminating group differences, if such differences do undeniably exist in the multivariate space. The P-value for the multivariate CAP test was obtained using 4999 permutations of the raw data. Both Pearson's correlation coefficients and one-way SIMPER analysis (Anderson et al. 2008) were also employed to help evaluate species responsible for any among- and between-site differences detected. To test for differences in dominant species across sites additional univariate PERMANOVA (as above) were undertaken.

The central univariate null hypothesis tested was: *there is no statistically significant difference in the response variable (species richness or individual species abundances) across sampling sites in 2018.*

The central multivariate null hypothesis tested was: *there is no statistically significant difference in infaunal assemblages across sampling sites in 2018.*

2.3.2.3 Temporal data

To evaluate trends and test for differences in community composition between sampling sites (A, B and C) among survey years (1996; 2007; 2011; 2017 and 2018) the same suite of multivariate analyses (above) were used. Two factor PERMANOVA was employed to evaluate changes between sites (Site) and Survey (1996-2018), whereas PCO and CAP were used to visualise any among site patterns through time in multivariate space. All analyses were run on log(x+1) transformed data and a Bray-Curtis similarity measure. Again, SIMPER analyses and Pearson's correlation coefficients were used to help discern main species responsible for driving and observed changes through time.

The central multivariate null hypothesis tested was: *there is no statistically significant difference in infaunal assemblages among sampling sites (A, B, C) across surveys (1996, 2007, 2011, 2017 and 2018) for sites A, B and C*. Note, the 2017 survey also incorporated new location Site D located 100 m north-west (inshore) of the outfall; however, this was excluded from the temporal analysis.

3 Results

3.1 Sediment geochemistry - RDPL and contaminant levels

Measurement of redox discontinuity layers (RDL) from sediment cores varied across sites in part in accordance with silt/mud content (Table 1). For potential impact sites A and B and the control site C redox discontinuity layers were evident in sample cores being on average around 30-35 mm deep across these sites, with a distinct anoxic layer (dark band) often present below the RDL layer. For the 7 new sampling sites equally the RDL layer varied between 18-56 mm, but was also indistinct at sampling sites G and H that had the highest silt/mud content. These sample cores also had a distinct H_2S odour associated with them.

Table 1 Average depth of the redox potential discontinuity layer (RDPL) from sediment at sampling sites within the Wairoa Estuary April 2018. Sites A and B are potential impact sites, whereas site C is a reference/control site. Sites D-J were added to the monitoring programme in 2018.

Site	Average RDL depth (mm) (+/- SD); n= 3 per site.
A	37.7 (2.5)
B	28.7 (10.9)
C	32.0 (3.6)
D	39.3 (17.0)
E	18.7 (6.0)
F	31.0 (3.6)
G	Indistinct
H	Indistinct
I	39.0 (11.3)
J	45.3 (9.5)

3.1.1 Sediment Texture

The change in composition of sediment grain sizes, or more notably, silt/clay content (<63µm) at Sites A, B and C can be seen in **Error! Reference source not found.** and Table 2. The greatest observable change is the large reduction of silt/clay content at Site B between 2011 and 2018, which dropped from approximately 70% to 19.26% respectively, and appears to have reverted back to 2007 proportions. Site A has also seen a drop in silt/clay content between 2011 and 2018 also, albeit slight (approximately 80% to 68.54% respectively), while Site C has remained relatively constant since 2011.

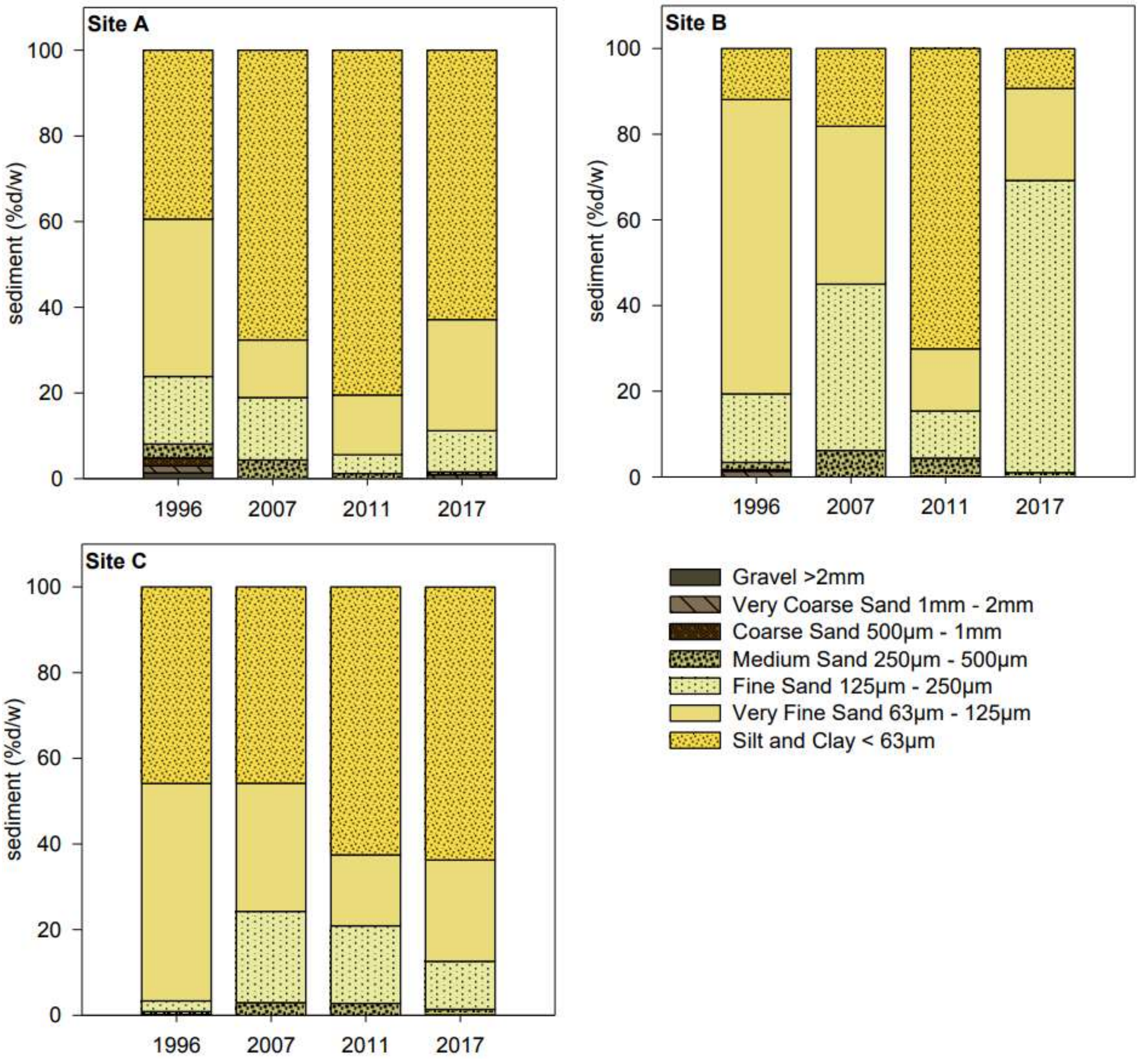

Figure 3. Sediment texture over time at sites A, B and C (from Larcombe, 1996 and EAM 2007,2011 and Triplefin 2017).

Table 2. 2018 silt/clay content results (<63µm).

Site	Silt fraction (%)
A	68.54
B	19.26
C	64.45
D	32.57
E	61.82
F	59.13
G	80.47
H	87.23
I	36.1
J	13.88
Overflow	10.19

3.1.2 Sediment Organic Content

Figure 4 and Table 3 below show the change in organic content over time at Sites A, B and C. It can be seen that since 2017 the total organic matter content at Site A has dropped from approximately 4.5% to 2.9%, Site B has remained stable at approximately 1.35% and Site C has seen an increase from approximately 1.8% to 2.4%. However, as levels of organic matter in sediments tend to be higher in those with higher silt fractions, results have been normalised to 100% of the silt/clay fraction for each site in order to more accurately compare between sites. Herein, 'normalised' results are referring to this adjustment.

Comparing normalised organic matter data over time, it appears that Site A has seen a drop between 2017 and 2018 from approximately 6.7% to 4.23% after steadily increasing between 1996 and 2017. Site B has fluctuated greatly over time showing large differences from 1996-2007, 2007-2011, 2011-2017 and now an observed normalised organic matter content decrease from approximately 14% to 7% since 2017. This can be attributed mostly to the large changes in silt/clay content that has occurred here between surveys. Normalised organic matter content at Site C steadily declined between 2007 and 2017 after an initial increase between 1996 and 2007 and the trend is now upward again with a change from approximately 2% in 2017 to 3.72% in 2018.

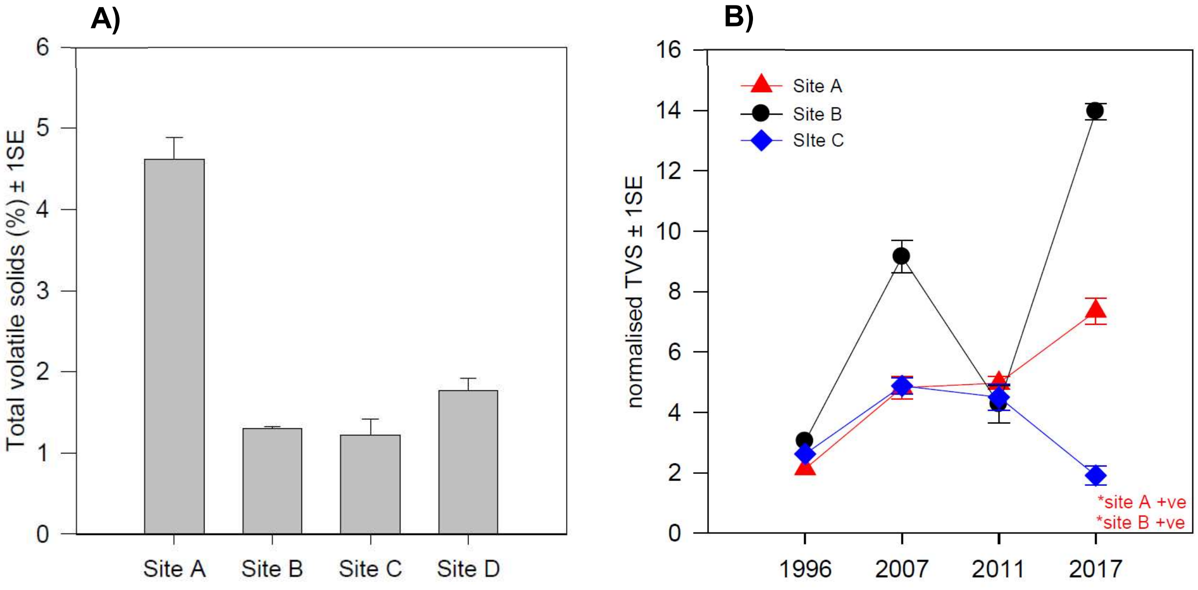

Figure 4. A) 2017 Total Volatile Solids content (organic matter) at Sites A, B and C (and D). B) Normalised organic matter content over time at Sites A, B and C. (from Larcombe, 1996 and EAM 2007,2011 and Triplefin 2017).

Table 3. 2018 total and normalised organic matter content results.

Site	Organic matter (%)	Normalised organic matter (%)
A	2.9	4.23
B	1.35	7
C	2.4	3.72
D	1.82	5.59
E	3.2	5.18
F	2.4	4.06
G	2.4	2.98
H	0.77	0.88
I	1.45	4.02
J	2.9	20.89
Overflow	0.99	9.72

The 2018 normalised results (Figure 4B) at Site J and the Overflow location are relatively high at 20.89% and 9.72% respectively, largely due to the low silt/clay content in the sediments at these locations.

3.1.3 Sediment Nutrients

Figure 5 and Table 4 below show the total nitrogen and phosphorus content in 2017 and 2018 respectively at Sites A, B and C. All three of these locations have seen a reduction in total nitrogen and phosphorus content over this period. All these reductions have been slight with the most notable change being that of total nitrogen at Site A, which has dropped from 0.09 g/100g to 0.05 g/100g.

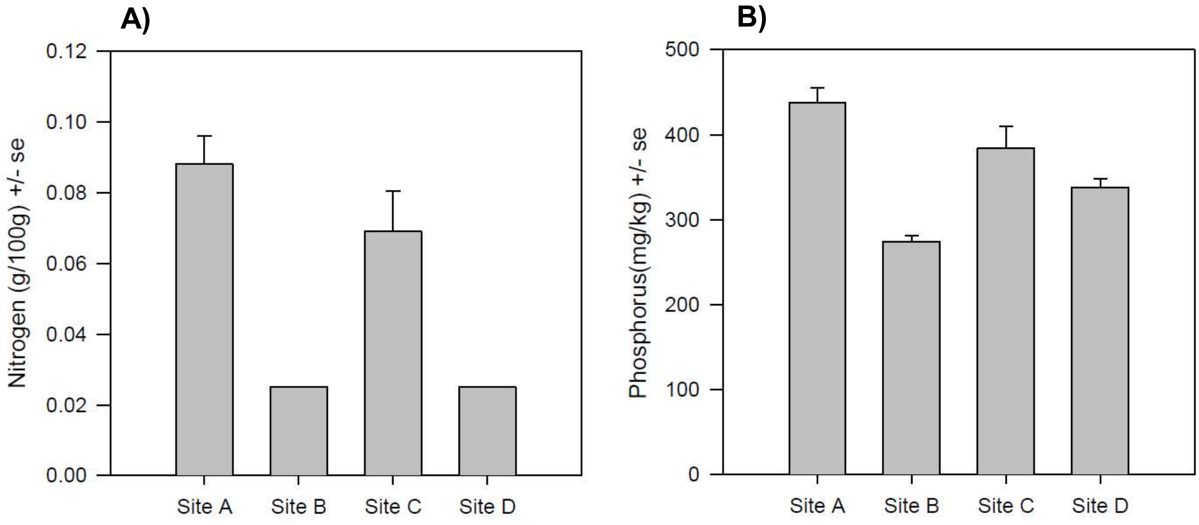

Figure 5. 2011 Mean total nutrient concentrations A) Nitrogen and B) Phosphorus at Sites A, B, C and D (from Triplefin 2018).

When compared to other Hawkes Bay and reference estuaries throughout New Zealand, total nitrogen and total phosphorus concentrations at Sites A-D are not significantly elevated, and lie in the mid-range of values (Table 4).

Figure 6 and Table 4 below show the change in normalised total nitrogen and phosphorus content over time at Sites A, B and C. Regarding the total normalised nitrogen content, Sites A and C show their lowest ever recorded levels dropping to 0.073 g/100g and 0.093 g/100g from approximately 0.14 g/100g and 0.11 g/100g in 2017, respectively. 2018 Site B content was below the detection limit of 0.05 g/100g and so no reliable comparisons can be made to historical data at this location.

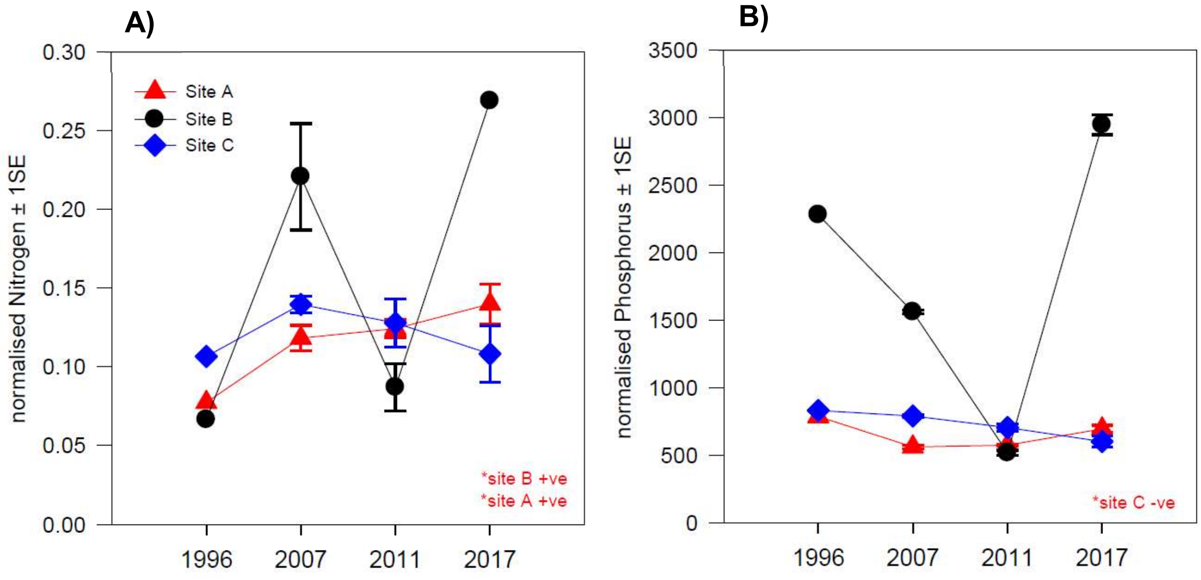

Figure 6.. Normalised nutrient concentrations over time A) Nitrogen and B) Phosphorus at Sites A, B and C.

With regard to the total normalised phosphorus content over time, Site A has decreased slightly from approximately 750 mg/kg to 598 mg/kg between 2017 -2018 whereas Site B has seen a dramatic decrease from approximately 3000 mg/kg in 2017 to 1558 mg/kg in 2018, almost returning to 2007 levels of around 1600 mg/kg. As in the case of organic content at Site B, this trend can be attributed mostly to the large changes in silt/clay content that has occurred here between surveys. In contrast, Site C has remained relatively stable at 559 mg/kg perhaps decreasing slightly from 2017 levels of approximately 600 mg/kg.

The 2018 normalised phosphorus results (Table 4 below) at Site J and the Overflow location are relatively high at 1873 mg/kg and 3729 mg/kg respectively, again, largely due to the low silt/clay content in the sediments at these locations.

Table 4. 2018 total and normalised nitrogen and phosphorus content results. Note: BDL = Below Detection Limit of 0.05 g/100g. Ranges of typical values at New Zealand estuary reference sites are included for comparison.

Site	Total Nitrogen (g/100g)	Normalised Total Nitrogen (g/100g)	Total Phosphorus (mg/kg)	Normalised Total Phosphorus (mg/kg)
A	0.05	0.073	410	598
B	BDL	BDL	300	1558
C	0.06	0.093	360	559
D	BDL	BDL	290	890
E	0.07	0.11	380	615
F	0.06	0.1	380	643
G	BDL	BDL	430	534
H	0.07	0.08	460	527
I	BDL	BDL	300	831
J	BDL	BDL	260	1873
Overflow	BDL	BDL	380	3729
Reference Sites				
Otamatea/Kaipara[1]	0.08 – 0.24	-	443 – 619	-
Ohiwa[1]	0.025 – 0.1	-	212 – 350	-
Ruataniwha[1]	0.025 – 0.07	-	330 – 580	-
Waimea[1]	0.025 – 0.1	-	243 – 562	-
Havelock[1]	0.007 – 0.09	-	241 – 433	-
Kaikorai[1]	0.15 – 0.21	-	728 – 913	-
Avon-Heathcote[1]	0.025 – 0.06	-	298 – 355	-
Ahuriri[2]	0.079 – 0.084	-	320 – 810	-

1. Robertson (2002) 2. Bennet (2006)

3.1.4 Sediment Trace Metals

Table 5 and Table 6 below shows the 2018 trace metal content results, as well as the ANZECC (2000) Interim Sediment Quality Guidelines low threshold values (ISQG-Low), above which biological effects can be expected. Only one ISQG-Low exceedance occurred in the 2018 trace metal results which was the Lead content at the Overflow location. This value was recorded at 124 mg/kg, nearly 2.5 times greater than the ISQG-Low threshold value of 50 mg/kg. It should be noted that following the trace metal analysis of the 2018 samples, the analyst's report indicated that the replicate analyses performed on this particular sample (as part of the Quality Assurance procedures) showed greater variation than would normally be expected which may reflect the heterogeneity of the sample. The replicate results were as follows: 133.813 mg/kg, 12.002 mg/kg, 38.262 mg/kg, 21.894 mg/kg, 412.715 mg/kg. The average of these results was that which was reported (124 mg/kg). Elevated levels of lead at this site are presumably resulting from reduced tidal mixing that this intertidal section of the estuary experiences, and potentially from stormwater run-off from the road dumped waste materials in this shoreline area.

Table 5. 2018 trace metal results and ANZECC (2000) ISQG-Low guidelines (mg/kg dry weight). Yellow shaded cell indicates an ISQG-Low exceedance.

Site	**Cadmium**	**Chromium**	**Copper**	**Nickel**	**Lead**	**Zinc**	**Arsenic**	**Mercury**
A	0.054	12	6.4	11.7	7.4	40	3.8	0.04
B	0.027	7.4	3.8	8.8	5.1	29	2.8	0.03
C	0.035	10.5	6.2	11.5	6.8	38	3.8	0.03
D	0.021	8.2	4	8.4	5.5	27	3.5	0.03
E	0.038	11.3	7.1	11.9	7.1	41	4.3	0.04
F	0.041	10.7	5.8	10.8	6.8	37	4.1	0.04
G	0.055	11.2	7.8	12.2	7.9	41	3.7	0.05
H	0.063	12.3	8.7	13.6	8	47	4.2	0.04
I	0.027	8.9	4.5	10	6	33	3.5	0.03
J	0.018	5.1	2.4	5.5	4	22	2.2	0.02
Overflow	0.051	7.9	10	8.6	124	50	4.9	0.06
ISQG-Low	1.5	80	65	21	50	200	20	0.15

Trace metals have been shown to preferentially adhere to fine sediments in the silt/clay fraction that have reactive surface properties. Therefore, differences in trace metal concentrations through time between sites may simply reflect differences in the proportion of sediments in this fraction. Normalising sediment contaminant data allows standardisation of sediment contaminants to sediment composition.

Table 6 and Figure 7 below show the change in normalised trace metal concentrations over time at Sites A, B and C. In addition, Table 6 contains a range of average values from New Zealand estuarine reference sites for comparison. Unfortunately, no trace metal data was available from the 1996 study to compare against.

In keeping with the trend seen in other analysed parameters, normalised trace metal concentrations vary greatly at Site B, again mostly due to the large changes in silt/clay content that has occurred here. Across all trace metals analysed, normalised concentrations at Site B have greatly increased since 2011 but not quite to the levels of 2007. Site C has seen reduced trace metal levels since 2011, however, most of these reductions are relatively slight with the exception of Cadmium which dropped from 0.11 mg/kg to 0.05 mg/kg over this period. Normalised trace metal concentrations at Site A have remained relatively constant since 2011.

Table 6 Normalised 2007, 2011 and 2018 results as well as a range of average values from New Zealand estuarine reference sites (mg/kg dry weight). ND = No Data.

Site	**Cadmium**	**Chromium**	**Copper**	**Nickel**	**Lead**	**Zinc**	**Arsenic**	**Mercury**
2018								
A	0.08	17.51	9.34	17.07	10.8	58.36	5.54	0.06
B	0.14	38.42	19.73	45.69	26.48	150.57	14.54	0.16
C	0.05	16.29	9.62	17.84	10.55	58.96	5.9	0.05
D	0.06	25.18	12.28	25.79	16.89	82.9	10.75	0.09
E	0.06	18.28	11.48	19.25	11.48	66.32	6.96	0.06
F	0.07	18.1	9.81	18.26	11.5	62.57	6.93	0.07
G	0.07	13.92	9.69	15.16	9.82	50.95	4.6	0.06
H	0.07	14.1	9.97	15.59	9.17	53.88	4.81	0.05
I	0.07	24.65	12.47	27.7	16.62	91.41	9.7	0.08
J	0.13	36.74	17.29	39.63	28.82	158.5	15.85	0.14
Overflow	0.5	77.53	98.14	84.4	1216.88	490.68	48.09	0.59
2011								
A	0.09	16.5	11.78	17.2	11.68	60.64	5.37	0.07
B	0.07	13.35	8.05	14.02	8.92	48.96	4.3	0.05
C	0.11	20.62	13.69	21.58	14.04	75.78	6.17	0.08
2007								
A	0.07	16.17	8.72	15.23	9.49	57.69	5.09	0.05
B	0.17	50.97	24.27	57.47	28.32	189.3	17.76	0.16
C	0.11	22.3	11.82	24.35	13.1	85.82	7.11	0.06
Reference Sites								
Ahuriri (site GPC)[2]	0.55	59.42	59.68	33.84	68.69	944.89	0.55	ND

Ahuriri (site PUR)[2]	0.36	45.21	35.95	28.06	44.86	463.89	0.36	ND
Ahuriri (site AHU)[2]	0.32	66.75	33.13	44.59	46.5	346.8	0.32	ND
Otamatea[3]	0.71	36.48	24.56	16.73	20.28	69.98	ND	ND
Ohiwa[3]	0.49	36.82	20.02	19.4	16.92	137.81	ND	ND
Ruataniwha[3]	1.09	260.87	77.17	148.91	51.09	407.61	ND	ND
Waimea[3]	1.22	275.97	39.18	295.92	30.2	170.61	ND	ND
Havelock[3]	1.57	255.49	56.02	138.74	29.32	225.13	ND	ND
Avon-Heathcote[3]	1.85	288.89	59.26	122.22	116.67	709.26	ND	ND
Kaikorai[3]	0.37	177.94	61.76	57.35	166.54	677.21	ND	ND
New River[3]	5.88	652.94	223.53	294.12	41.18	1005.9	ND	ND

1. Smith (2007) 2. Smith (2010) 3. Robertson (2002)

In comparison with other estuaries in Hawkes Bay and New Zealand, the levels of all trace metals, with the exception of Arsenic, at all sites excluding the overflow, are low to very low. The high levels of Arsenic at all sites through time in the Wairoa River Estuary when compared to the Ahuriri Estuary (Napier) is a reflection of the naturally elevated Arsenic content in the rocks comprising the Wairoa River basin.

Normalised 2018 trace metal levels at the Overflow location, as previously mentioned, are elevated for the most part due to the low fraction of silt/clay sediments (10.19%) recorded here.

Figure 7 Plots of normalised trace metal concentrations (mg/kg dry weight) over time at sites A, B and C in the lower Wairoa River Estuary (includes data from EAM 2007 and 2011; and Triplefin 2018).

3.2 Ecological - Infaunal sampling

2018 survey

A total of 17 infaunal species (Appendix 2) were enumerated from faunal cores across the 10 sites surveyed in 2018. Species richness was low to moderate across sites, with higher species richness recorded at sites A, B, C and E with >4 species core^{-1}, and lower species richness recorded at sites D and F-J (Figure 8). Differences in species richness were statistically significant among sites (PERMANOVA, Pseudo-*F*=5.95, *P*<0.001). Across-site patterns in species richness were largely mirrored by Shannon-Wiener diversity indices, whereas estimates of evenness were higher for sites with lower species richness. Sites with higher average number of individuals, typically >30 individuals core^{-1}, were predominantly those on situated in the western region of the estuary closest to the outfall i.e., sites A-E; whereas, core samples from other sites were generally <20 individuals core^{-1}.

Variation in infaunal community composition was statistically different among sites surveyed in 2018, a trend that is supported by PCO analysis depicting separation of individual sites across the PCO ordination that explained >50 % of the variation in the dataset (Figure 9a; Table 7). Group separation was characterised as follows. Replicate cores for sites A, B, C were grouped to the right of the ordination positively associated with PCO Axis 1 with replicate cores for sites D, E, and F generally negatively associated with PCO Axis 1, and site G exhibiting greater variation among sample cores along PCO Axis 2. The overall pattern depicted from PCO analysis is one of greater among-site variation in community composition with increasing distance away from the area of potential impact. A similar pattern was apparent when PCO analysis was conducted on group centroids (Figure 10). Supplementary multivariate analysis using the constrained ordination CAP (Figure 11) inferred that differences in multivariate space among sites were highly statistically significant δ^2= 0.859, *P* < 0.001. Group clustering depicting similar species compositions included: Group 1, (A-C); Group 2 (D, E, and I); and Group 3 (F and J).

Table 7 Results of PERMANOVA analysis testing for differences in infaunal community composition across 10 sites in 2018. Analysis was run on Log (x+1) abundance data and a Bray-Curtis similarity matrix using 4999 permutations. Statistically significant P-values

Source	df	SS	MS	Pseudo-F	P(perm)
Site	9	65790	7310	5.6723	***0.0002***
Res	40	51549	1288.7		
Total	49	1.17E+05			

Species responsible for among-site differences based on Pearson's correlation coefficients are presented in (Figure 9B). Sharing commonality with previous surveys and supported by SIMPER analysis, species responsible for among-site similarity between sites A, B and C were the amphipod (*Paracorphium excavatum)*, and the polychaetes (*Nicon aestuariensis* and *Scolelepis* sp.). The pipi (*Paphies australis)* and to a lesser degree (*Paracorphium excavatum)* were responsible for underpinning any similarities across the additional sites (D-J).

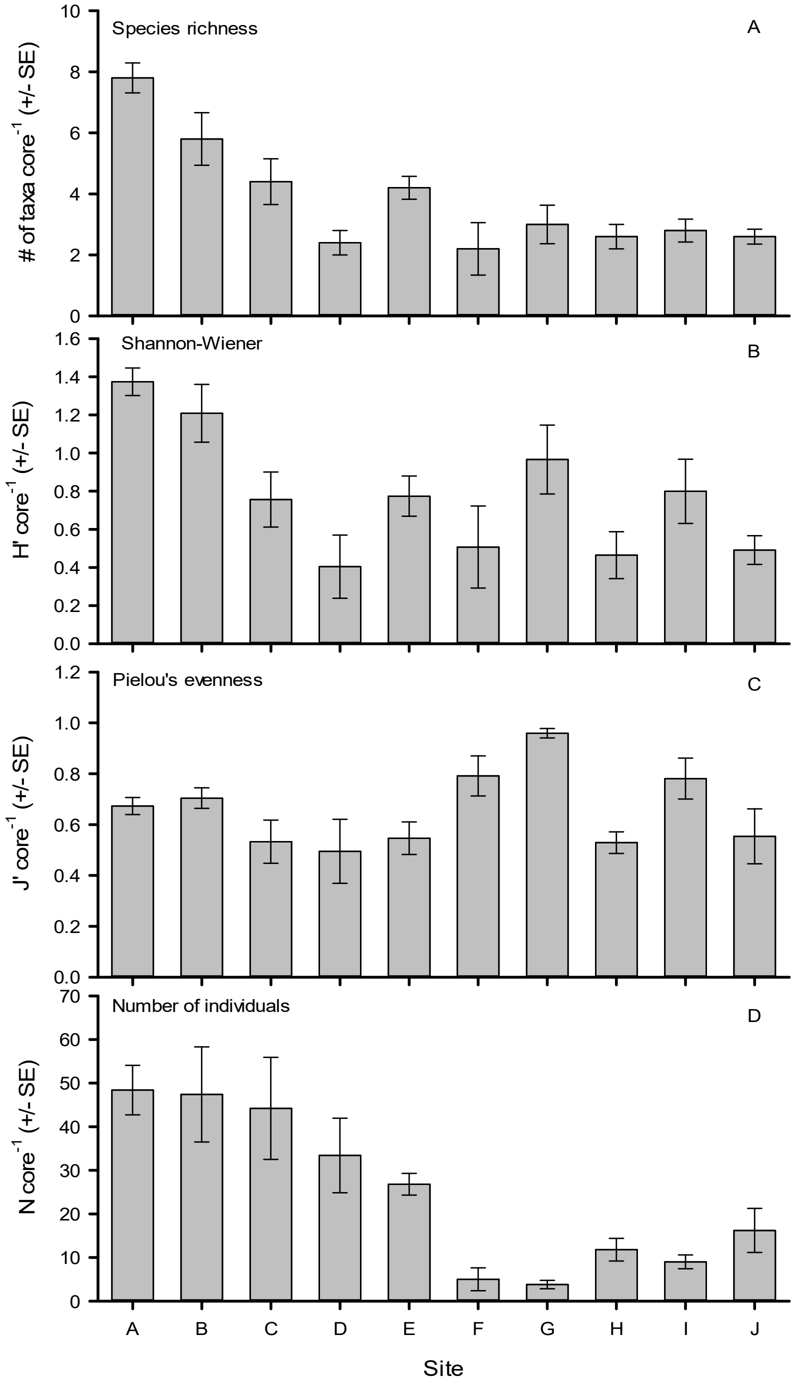

Figure 8 A) Species richness; B) Shannon-Wiener diversity; C) Species evenness; and D) number of individuals across sampling sites within the Wairoa Estuary in 2018. Data are mean values ± associated standard error.

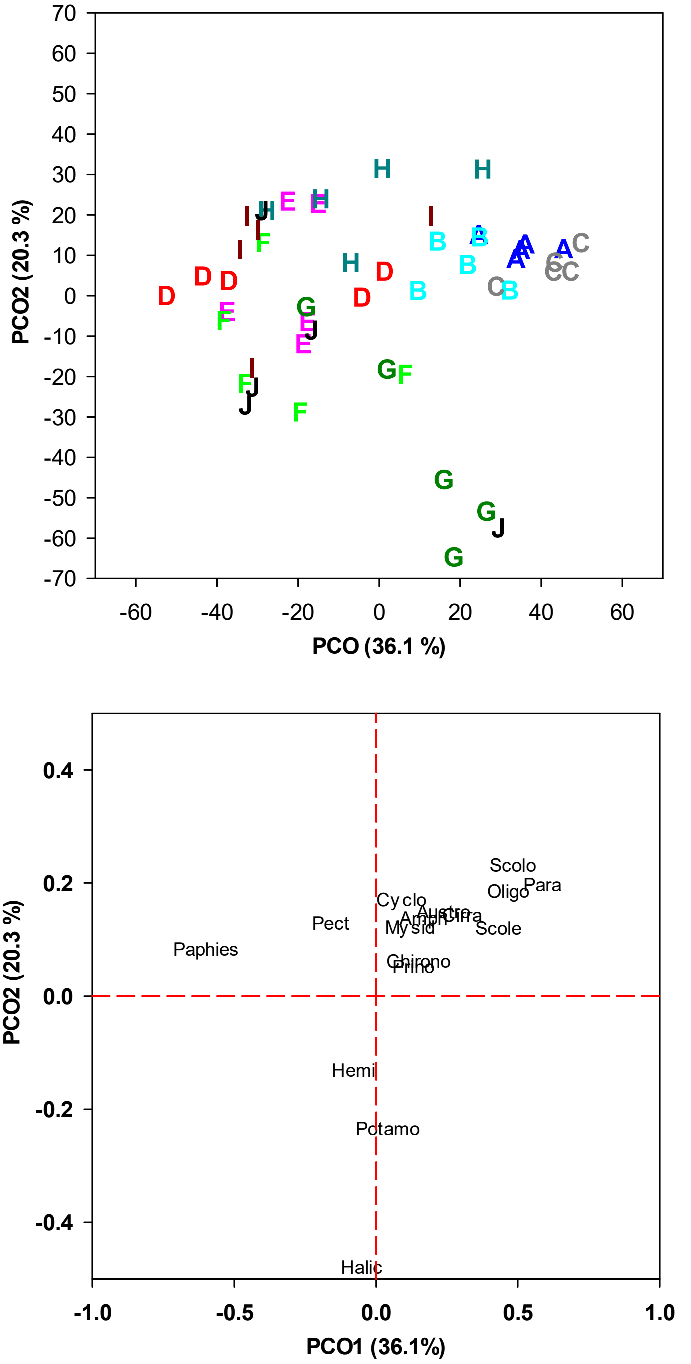

Figure 9 A) Principal coordinates ordination of 17 infaunal taxa derived from core sampling across 10 sites within the Wairoa Estuary (n=5 per site). Analysis was run on Log (x+1) transformed data and a Bray-Curtis similarity matrix. B) Bi-plots depict Pearson correlation coefficients between abundance and the two PCO axes.

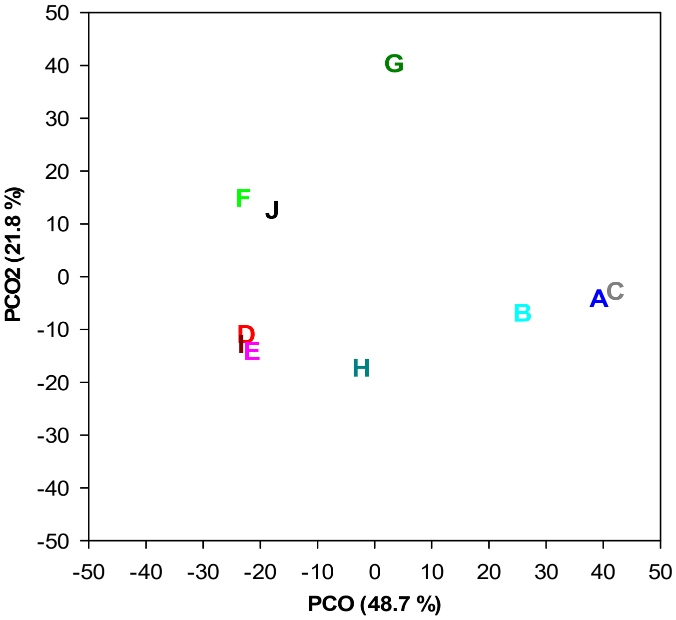

Figure 10. Principal coordinates ordination run on group centroids of 17 infaunal taxa derived from core sampling across 10 sites within the Wairoa Estuary (n=5 per site).

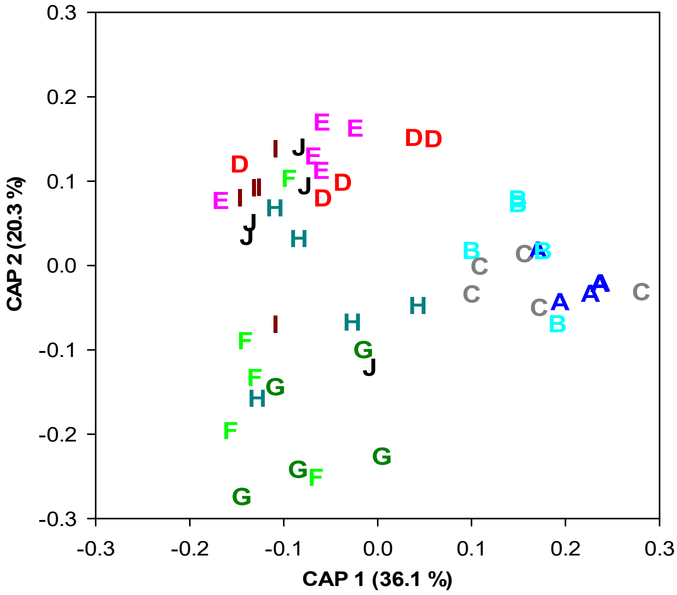

Figure 11. Canonical analysis of principal coordinates (CAP) ordination of 17 infaunal taxa derived from core sampling across 10 sites within the Wairoa Estuary (n=5 per site). Analysis was run on Log (x+1) transformed data and a Bray-Curtis similarity matrix.

Abundances of dominant infaunal species driving group similarities and dissimilarities across sites are presented in Figure 9. Of these, the bivalve/pipi (*Paphies australis)* were encountered at the majority of sampling sites attaining highest densities at sites D, E and J. However, irrespective of site-specific abundances, all pipi enumerated across sites were typically small <30 mm shell length. Of the dominant polychaetes encountered, *Scolelepis* sp was common to sites A and B, whereas *Nicon aestuariensis* was more widespread, but also attained highest abundances across sites A-C. This trend was also apparent for burrowing amphipod (*Paracorphium excavatum)*, although highest abundances of *Paracorphium excavatum* were apparent at reference site C. Differences in abundances for all species were statistically different across sites based on PERMANOVA (Refer to Appendix 3). Collectively based on the results of analyses performed on 2018 data there was strong rejection of the underlying null hypothesis of no differences in specific components (diversity, abundance community composition) across sampling sites.

3.3 Temporal analysis (1996-2018)

Species richness among sites A, B, and C has for all survey periods (1996; 2007; 2011; 2017 and 2018) (Figure 12) and irrespective of survey, species richness has been slightly higher at Site A and lowest at Site C, with Site B intermediate between the two (Figure 12). There was a trend for decreasing diversity between 1996 and 2011 surveys, with the 2018 survey notable for elevated diversity across the three sites. This largely reflects new species recorded in 2018 (refer to Appendix 3). Shannon-Wiener diversity equally varied across sites and surveys, being markedly reduced at Site A in 2007 which, in part, reflects the low evenness value for this site. Shannon-Wiener indices for survey years 1996 and 2011 were similar across sites, whereas for 2018 sites A, and B had higher values than C, a pattern attributable to higher sample evenness (Figure 12).

Temporal analysis (PERMANOVA) undertaken on infaunal data from the three sites A, B and C across the five sampling periods (1996; 2007; 2011; 2017; 2018) identified statistically significant differences for infaunal community composition among sites and surveys, with the statistically significant Survey×Site interaction indicative of heterogeneous changes (variation) in community composition among locations through time (Table 8). The variation in community composition through time is represented well in the PCO ordination (Figure 13A) that depicts strong serration across sites among surveys with a general movement from right to left across the ordination predominantly associated with PCO Axis 1. For the 2017 survey there was strong serration along PCO Axis 2. The PCO ordination encapsulates >55 % of the variation in the dataset, and thus is a robust reflection of the fluctuations in community composition through space and time. Among-site differences in community composition were greatest in 1996 and 2017 relative to other surveys (Figure 13B) that are generally characterised by less among-site heterogeneity. Between-survey differences were however, apparent for all sequential surveys; i.e., supplementary pair-wise analysis (PERMANOVA) indicated that all survey year comparisons were statistically different (P < ***0.0002***) from one another. This is supported further by constrained multivariate analysis using CAP (Figure 14), emphasising strong among-survey differences that were statistically significant δ^2= 0.546, P < ***0.0002*****,** which is supported by the CAP ordination. As identified in previous survey reports (Smith, 2007; 2011) differences in community composition between 1996 through to 2007 were primarily driven by changes (reductions) in the marine gastropod *Potamopyrgus antipodum* and amphipod *Paracorphium excavatum,* with differences between 2007 and 2011 due to both *Paracorphium excavatum* and *Nicon aestuariensis.* The major difference between the 2017 survey (Triplefin, 2018) and other surveys was due to the high numbers of the spionid *Boccardia* spp. Species such as *Potamopyrgus antipodum, Paracorphium excavatum* and *Nicon aestuariensis* were still present within the sample population in 2018 with differences between 2011, 2017 and 2018 (based on Pearson's correlation coefficients and SIMPER analysis) being driven by increases in the polychaete *Scolelepis* sp., slight decreases in *Paracorphium excavatum* and the occurrence of *Paphies australis*. Again, based on the results of analyses performed on temporal data there was strong rejection of the underlying null hypothesis of no differences in multi-species data among sampling sites through time.

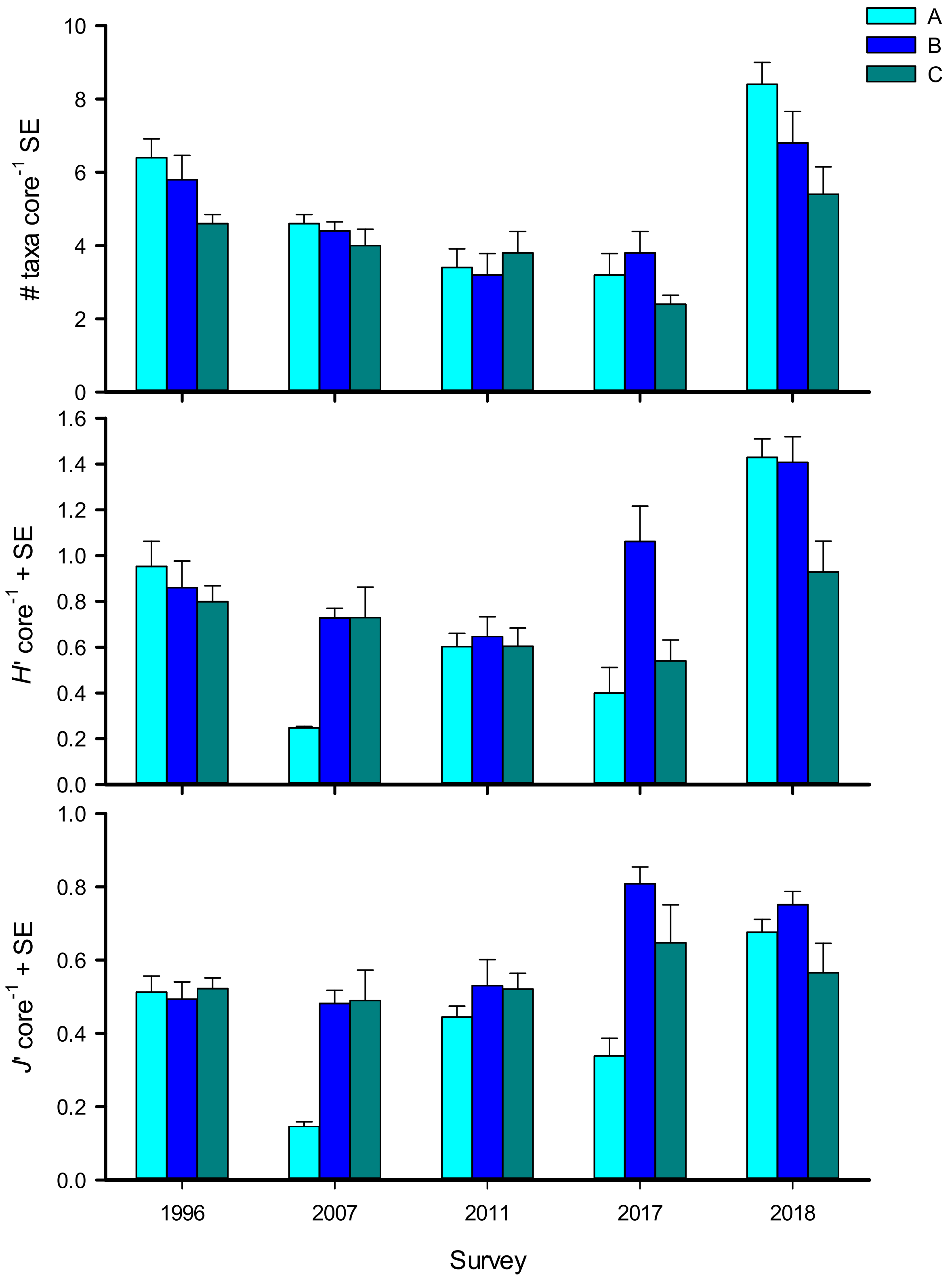

Figure 12 A) Species richness; B) Shannon-Wiener diversity; C) Species evenness; across sampling sites A, B and C within the Wairoa Estuary in 1996; 2007; 2010; 2017 and 2018. Data are mean values + associated standard error.

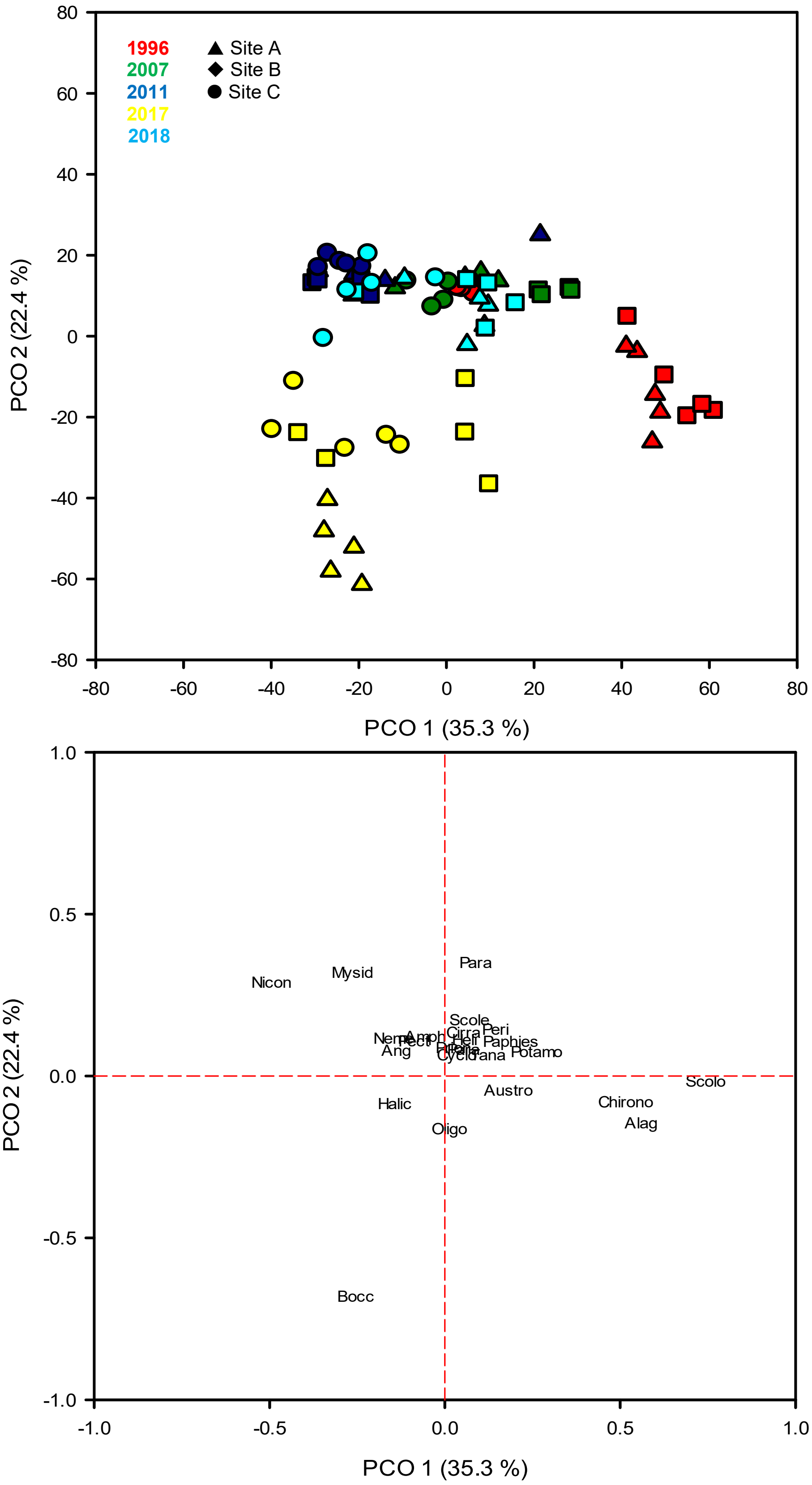

Figure 13 A) Principal coordinates ordination of 14 infaunal taxa derived from core sampling across 3 sites within the Wairoa Estuary (n=5 per site) for 1996; 2007; 2011; 2017 and 2018 surveys. Analysis was run on Log (x+1) transformed data and a Bray-Curtis similar.

Table 8 Results of PERMANOVA analysis testing for differences in infaunal community composition across 3 sites (A, B and C) and four surveys (1996; 2007; 2011; 2017 and 2018). Analysis was run on Log (x+1) abundance data and a Bray-Curtis similarity matrix with 4999 permutations. Statistically significant P-values at the 5% level are shown italicised and in bold.

Source	**df**	**SS**	**MS**	**Pseudo-F**	**P(perm)**
Site	2	16751	8375.3	19.342	***0.0002***
Survey	4	70908	17727	40.939	***0.0002***
SixSu	8	26013	3251.7	7.5095	***0.0002***
Residual	60	25980	433.01		
Total	74	1.40E+05			

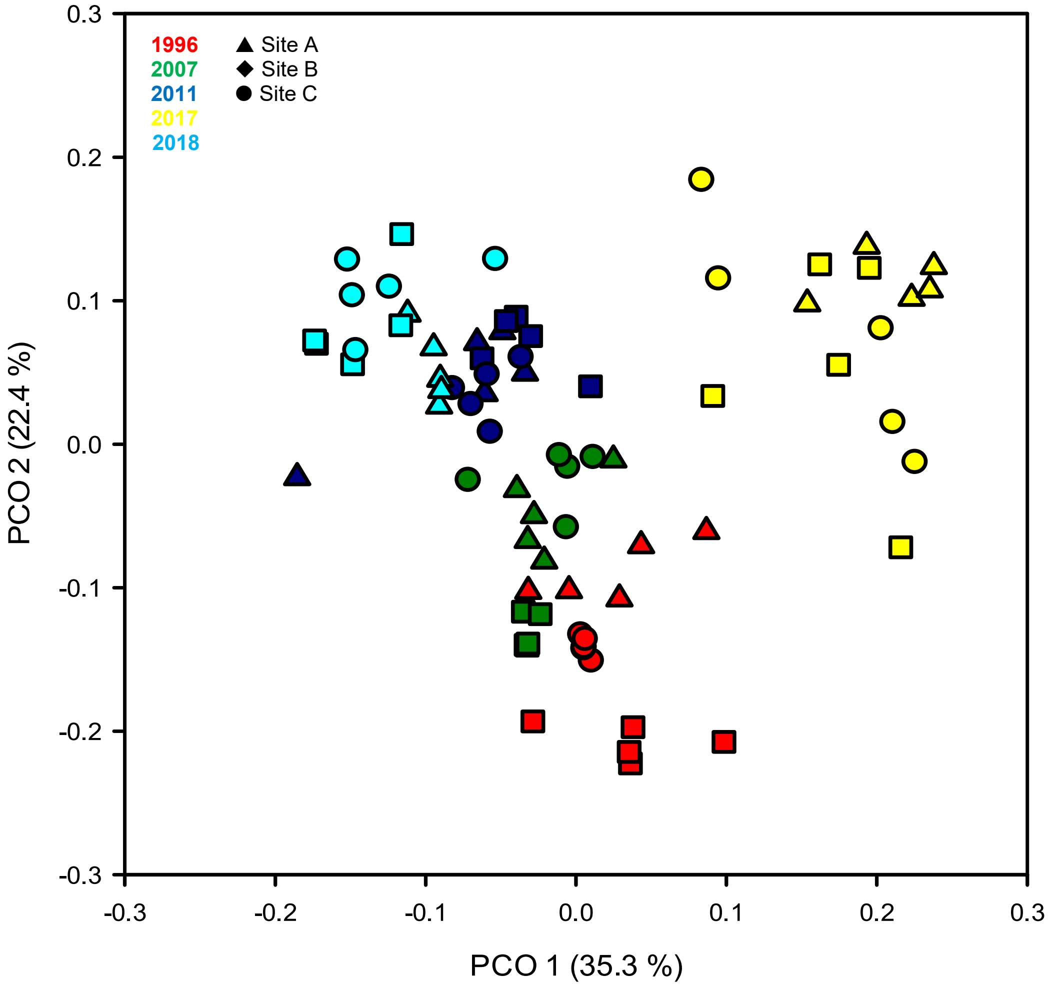

Figure 14 Canonical analysis of principal coordinates (CAP) of 14 infaunal taxa across 3 sites within the Wairoa Estuary derived from core sampling (n=5 per site)   Analysis was run on Log (x+1) transformed data and a Bray-Curtis similarity matrix.

4 Discussion

This report presents the findings from a sediment geochemistry (contaminant) and ecological surveys undertaken in April 2018 relating to re-consenting of treated sewage disposal into the Wairoa Estuary via an intertidal outfall. The survey incorporated the 3 historically sampled sites (A ~ 100m to the south of the outfall; B ~ 100m north of the outfall; and, C a control/reference site ~ 500m to the north of the outfall) and an additional 7 new sites in order to provide a more comprehensive survey of the lower Wairoa Estuary. Some concern was raised in the review of Mead (2018) with respect to the possibility of all historic sites (A-C) being "potentially" within the wider impact zone of the outfall especially in the absence of any comprehensive hydrodynamic modelling. While the reference "Site C" has provided useful information, supplementary sites are typically required in monitoring studies of this nature that span a gradient with increasing distance away from the outfall. The additional sites to the south of the existing estuary (D-G) and those along the eastern bank (H-J) were added to evaluate potential impacts of the outfall at distances >100m away from the discharge point and provide more contextual information on wider Wairoa Estuary environment. Another addition to the programme was the evaluation of sediment geochemistry immediately adjacent the overflow within the immediate intertidal adjacent to the Kopu Rd/Fitzroy St intersection. It must be emphasized that studies of this nature present merely a snap-shot in time and due to the infrequency of sampling and lack of spatial detailed studies linking cause and effect based on the measures presented here can be challenging.

4.1 Sediment geochemistry

Sediment geochemistry characteristics were highly variable across sampling sites in 2018, including across surveys (1996-2018). Sites with highest silt content >60% in 2018 were G and H closest to the entrance; sites A, E and F located to the south of the discharge; and, Site C (reference). Sites I and J were typically <30% silt content with site B having the lowest silt content at <19%. Based on historic surveys, Site B is notable for large fluctuations in silt content, hypothesised to reflect large variation in physical process for this part of the estuary i.e., transitioning between attrition and deposition relative to existing hydrodynamic characteristics at the time of sampling. This has also been reported for the HBRC State of the Environment (SoE) monitoring site approximately 600 m to the north of the AFFCO outfall. There was also a lack of a clear trend for both silt and organic content in relation to increasing or decreasing distance away from the outfall and thus no strong evidence that the outfall is impacting the geochemistry of the immediate benthos per se, a finding evident in previous surveys (Smith, 2007, 2012). However, the high silt content characteristic of many of the survey sites collectively highlights the wider degraded nature of the Wairoa Estuary. SoE monitoring undertaken at Wairoa by HBRC since 2011 has indicated a median mud content of 53.89 % within the Wairoa River upstream of the AFFCO outfall. This is at the top end of the estuaries it monitors; albeit mud content is reported as “highly variable” between surveys (e.g., ranging between 35% to 88%). Despite this, there is a trend is for increasing mud content with a wider perception that the estuary is likely to be sediment ‘stressed’ (HBRC, 2016).

Similarly, for sediment contaminants and nutrients there was generally no consistent pattern across sites, the exception being Site B (100 m to the north of the outfall) that consistently had higher sediment bound contaminants than other sampling sites. Temporal comparisons made with 2007 and 2011 surveys, indicated that 2018 contaminant levels were largely comparable to those measured in 2007; while there was also similarity with 2017, Site C showed increased organic matter, nitrogen and phosphorus. As contaminant levels were below ANZECC (2000) Interim Sediment Quality Guidelines low threshold values (ISQG-Low), it is anticipated that no adverse ecological effects would be expected as a result of the current discharge. Furthermore, sediment contaminant levels measured in this study were all comparable to levels recorded from SoE monitoring (HBRC, 2016). An ISQG-low exceedance was however, detected for lead immediately adjacent to the overflow opposite Fitzroy Street/Kopu Rd intersection. This location generally had higher levels for all contaminants relative to the other 10 estuarine-based sampling sites, presumably resulting from reduced tidal mixing that this intertidal section of the estuary experiences; these results are elevated for the most part due to the low fraction of silt/clay sediments (10.19%) recorded here, which skew the results. It is also notable that this overflow is predominantly a stormwater outlet from the road (which contribute to heavy metal contaminants), and historically there has been dumping of materials along the coastal margin.

4.2 Ecological characteristics

Irrespective of sampling sites evaluated, species diversity was low across sites, although there was a trend for significantly higher diversity and abundance of individuals occurring at sites A, B and C. The lower diversity at many of the new sampling sites do however, fall within comparable SoE monitoring diversity values (HBRC, 2016). Species composition and richness at sites A, B and C were found to be relatively similar, but also differed from the other sites, which is an important result as collectively these patterns could reflect outfall-related responses. Firstly, relative to other sampling sites species compositions at A and B are consistent with impacted environments, with site C also being influenced in a similar way.

Alternatively, if the outfall is not impacting the immediate benthos at these sites, then the species compositions could reflect a combination of influences that have a greater influence on this specific area of the estuary, i.e., from other stressors impacting the estuary including sedimentation, and additional discharges associated with storm-water. Numerically abundant species associated with sites A, B and C that characterised the infaunal communities at these sites included the amphipod *Paracorphium excavatum* and polychaetes *Scolelepis* sp. and *Nicon aestuariensis.* All are synonymous with degraded/muddy or impacted environments (see Hewitt *et al.* 2006; Rowden *et al*. 2012).

Previous monitoring reports (EAM, 2007, 2011; Triplefin, 2018) have suggested that the presence of species like pipi (*Paphies australis*), at sites around the outfall were evidence that any potential effects emanating from the outfall were not large enough to constitute an undue adverse effect. While pipi were encountered at the majority of sites in 2018 (including A, B and C), when the potential impact sites are evaluated against the new sites it is apparent that pipi numbers are significant lower at sites A, B and C, at least relative to sites E, F, G and H. This trend appears unrelated to silt content; however it must be stressed that all pipi enumerated were <30 mm in size, therefore are likely to be stressed at all sites where they are encountered. Again comparisons of trends detected here are consistent with those derived from SoE monitoring.

Temporal analysis done at the community level for sites A, B and C detected an increase in the diversity of species in 2018, which contrasts a trend of declining diversity between 1996 and 2017 surveys. By and large, the community composition has remained stable with the more abundant species present at the start of monitoring (1996) routinely being enumerated across surveys. Major shifts in community composition tend to reflect large-scale changes in 1 or 2 species such as decreasing abundances of mud snail (*Potamopyrgus antipodum)* between 1996 and 2007 surveys and the increase followed and decrease in the polychaete *Boccardia* spp between 2011 and 2018 surveys. Respectively. In low density, low diversity soft sediment benthic communities even modest shifts in 1 or 2 species can impact greatly on multivariate ordination results (Anderson *et al*. 2008) The increase in diversity in 2018 also reflects several new species being encountered at low abundance (Refer to Appendix 1 for species list).

4.3 Summary

The purpose of the study was to evaluate any effects on sediment geochemistry and infaunal biota (ecology) that could be potentially attributable to the Wairoa community wastewater discharge.

Comparing commonality with previous surveys indicates there was no obvious pattern that the outfall is negatively impacting sediment geochemistry, based on texture, organic matter and trace metal analyses.

Sediment geochemistry characteristics were found to be highly variable across sampling sites in 2018, including across surveys (1996-2018). Sites with highest silt content > 60% in 2018 were G and H, closest to the entrance sites A, E and F located to the south of the discharge; and, Site C (reference). There was also a lack of a clear trend for both silt and organic content in relation to increasing or decreasing distance away from the outfall and thus no strong evidence that the outfall is impacting the immediate benthos. Sediments adjacent the intertidal overflow were found to have higher contaminants than the estuarine sites sampled exceeding ANZECC (2000) Interim Sediment Quality Guidelines low threshold values for lead.

For sediment-bound contaminants and nutrients from estuarine samples there was generally no consistent pattern across sites, the exception being Site B (100 m to the north of the outfall) that generally had higher sediment bound contaminants than other sampling sites. However, all contaminant levels were below ANZECC (2000) Interim Sediment Quality Guidelines low threshold values and thus adverse ecological effects due to contamination would be unlikely.

Species community composition was found to differ among survey sites with historical sites A, B, and C having similar species composition and higher species diversity compared to the new sites added to the programme. Numerically dominant species recorded at sites A, B, and C are typically considered to be synonymous with degraded/impacted environments; in this case it is likely attributed to local nitrification and siltation. This was also true for the additional new sites, although abundances are lower than at Sites A-C and pipi were found to occur at much greater densities at sites further away from the outfall.

Evaluating impacts of the outfall on benthic effects is generally difficult given the low species diversity and wider degraded nature of the lower Wairoa estuary, as well as the initial monitoring only having a few sites within the zone of impact (which is evident due to the differences found at Sites A-C in comparison to the rest of the locations), poor temporal resolution (Mead, 2018); this survey is only the 4th since 1996. The previous monitoring/sampling design was focussed on the mixing zone next to the outfall and so did not consider impacts on the wider estuarine environment. This assessment has attempted to address this issue with the addition of 7 new sites located at increasing distances away from the subtidal outfall.

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Appendix A. **Site Coordinates**

Table A1 Sampling site co-ordinates

Site	Latitude	Longitude
A	39° 3'34.98"S	177°25'8.67"E
B	39° 3'30.06"S	177°25'14.09"E
C	39° 3'20.85"S	177°25'26.34"E
D	39° 3'37.42"S	177°25'6.07"E
E	39° 3'37.42"S	177°25'10.92"E
F	39° 3'42.16"S	177°25'2.24"E
G	39° 3'43.71"S	177°24'55.22"E
H	39° 3'41.57"S	177°25'21.86"E
I	39° 3'36.74"S	177°25'26.95"E
J	39° 3'15.05"S	177°25'45.16"E

Appendix B. **Infaunal Counts**

Table B1 Infaunal counts derived from core sampling across 10 sites within the Wairoa estuary. Paphies australis; Cyclomactra ovata; Austrovenus stutchburyi; Amphibola crenata; Potamopyygrus antipodum; Nicon aestuariensis; Scolelepis sp.; Scolecolepides benhami.

Site	*Paphies*	*Cyclomactra*	*Austrovenus*	*Amphibola*	*Potamopyrgus*	*Nicon*	*Scolelepis* sp	*Scolecolepides*
A	1	2	4	0	0	2	10	0
A	1	0	2	1	0	11	16	0
A	3	1	1	1	0	9	2	0
A	1	1	0	0	1	15	15	1
A	0	1	0	0	0	5	14	1
B	5	0	1	0	0	12	22	0
B	1	0	0	0	0	1	10	0
B	2	0	0	0	0	5	0	0
B	11	0	0	0	0	1	15	0
B	11	0	0	0	1	7	13	0
C	0	0	0	0	0	10	12	0
C	0	0	0	1	0	5	1	0
C	1	0	0	0	0	2	0	0
C	0	0	0	0	0	5	0	1
C	0	0	0	0	0	14	0	1
D	25	0	0	0	0	1	0	0
D	16	0	0	0	0	2	0	0
D	13	0	0	0	0	0	1	0
D	56	0	0	1	0	0	1	0
D	14	0	0	0	0	0	0	0
E	14	1	0	0	0	12	0	0
E	22	2	0	0	0	0	0	0
E	21	0	0	0	0	5	0	0
E	13	0	0	0	0	1	0	0
E	28	0	0	0	0	0	0	0
F	3	0	0	0	0	0	0	0
F	2	0	0	0	0	0	0	0
F	2	0	0	0	0	0	0	0
F	11	1	1	0	0	1	0	0
F	0	0	0	0	0	0	0	0
G	0	0	0	0	0	0	0	0
G	1	0	0	0	0	0	1	0
G	0	0	0	0	0	0	0	0
G	0	0	0	0	3	0	0	0
G	1	0	0	0	2	1	0	0
I	1	0	0	0	1	1	0	0
I	6	0	0	0	0	2	0	0
I	5	0	0	0	0	4	0	0
I	2	0	0	0	0	11	1	0
I	0	0	0	0	0	6	0	0
J	0	0	0	0	0	2	0	0
J	10	0	0	0	0	1	0	0
J	12	0	0	0	1	2	0	0
J	15	0	0	0	0	2	0	0
J	32	0	0	0	0	0	0	0
H	0	0	0	0	0	0	0	0
H	15	0	0	0	0	0	0	0
H	10	0	0	0	0	1	0	0
H	12	1	0	0	0	2	0	0
H	10	0	0	0	1	0	0	0

Table B2 continued Infaunal counts derived from core sampling across 10 sites within the Wairoa estuary. Pectinaria australis; Prionospio sp.; Paracorphium excavatum; Halicarcinus sp.; Hemiplax hirtipes.

Site	Oligochaete	Cirratulidae	*Pectinaria*	*Prionospio*	*Paracorphium*	Mysid	*Halicarcinus*	*Hemiplax*	Chironomid
A	1	0	0	0	28	0	0	0	1
A	0	1	0	1	30	0	0	0	0
A	0	0	0	0	34	1	0	0	0
A	1	0	0	3	12	0	0	0	0
A	1	0	0	0	6	0	0	0	0
B	0	1	0	2	44	2	0	0	0
B	0	0	0	0	20	1	0	0	0
B	0	0	0	0	20	0	0	0	0
B	0	1	1	0	15	0	1	0	0
B	1	0	0	0	10	0	0	0	0
C	2	1	0	0	50	0	0	0	0
C	1	0	0	1	21	1	0	0	0
C	0	0	0	0	66	1	0	0	0
C	0	0	0	0	19	0	0	0	0
C	0	0	0	0	5	0	0	0	0
D	0	0	0	0	14	0	0	0	0
D	0	0	0	0	23	0	0	0	0
D	0	0	0	0	0	0	0	0	0
D	0	0	0	0	0	0	0	0	0
D	0	0	0	0	0	0	0	0	0
E	0	0	1	0	0	0	0	1	0
E	0	0	1	0	2	2	0	0	0
E	0	0	2	0	0	0	0	0	0
E	0	0	1	0	2	0	0	0	0
E	0	0	0	1	0	1	0	1	0
F	0	0	0	1	1	0	0	0	0
F	0	0	0	0	0	0	0	0	0
F	0	0	0	0	0	0	1	0	0
F	0	0	0	1	0	0	0	0	0
F	0	0	0	0	0	0	0	0	0
G	0	0	0	0	1	0	1	0	0
G	0	0	0	0	0	0	0	0	0
G	0	0	0	0	2	0	0	1	0
G	0	0	0	1	1	1	1	0	0
G	0	0	0	0	1	0	0	0	0
I	0	0	0	0	0	2	0	0	0
I	0	0	0	0	0	0	0	0	0
I	0	0	0	0	0	2	0	0	0
I	0	0	0	0	0	0	0	0	0
I	0	0	0	0	0	1	0	0	0
J	0	0	0	1	0	0	0	0	0
J	0	0	0	1	0	0	0	0	0
J	0	0	0	0	0	0	0	0	0
J	0	0	0	0	0	0	0	0	0
J	0	0	0	1	1	0	0	0	0
H	0	0	0	0	2	0	0	0	0
H	0	0	0	0	1	0	1	0	0
H	0	0	0	0	2	0	0	0	0
H	0	0	0	0	0	0	0	0	0
H	0	0	0	0	1	0	0	0	0

Appendix C. **PERMOANOVA results for individual species**

Table C1 Results of PERMANOVA analysis testing for differences in individual species abundance across 10 sites in 2018. Analysis was run on Log (x+1) abundance data and a Bray-Curtis similarity matrix using 4999 permutations. Statistically significant *P*-values at the 5% level are shown italicised and in bold.

Scolelepis sp.

Source	df	SS	MS	Pseudo-F	P(perm)
Site	9	1045.7	116.19	9.2358	***0.0002***
Res	40	503.2	12.58		
Total	49	1548.9			

Nicon aestuariensis

Source	df	SS	MS	Pseudo-F	P(perm)
Site	9	25.465	2.8295	6.6405	***0.0002***
Res	40	17.044	0.4261		
Total	49	42.509			

Paphies australis

Source	df	SS	MS	Pseudo-F	P(perm)
Site	9	49.269	5.4743	9.0412	***0.0002***
Res	40	24.219	0.60549		
Total	49	73.488			

Paracorphium excavatum

Source	df	SS	MS	Pseudo-F	P(perm)
Site	9	76.251	8.4724	16.345	***0.0002***
Res	40	20.734	0.51836		
Total	49	96.986			