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BENTHIC EFFECTS MONITORING OF THE WAIROA DISTRICT COUNCIL MUNICIPAL WASTEWATER OUTFALL AT SITES IN THE LOWER WAIROA ESTUARY:

2017 SURVEY.

MAY 2018
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Benthic effects monitoring of the Wairoa District Council municipal wastewater outfall at sites in the Lower Wairoa River Estuary, Hawke's Bay: 2017 survey.

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Prepared for:

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TABLE OF CONTENTS

1. INTRODUCTION

Wairoa District Council (WDC) operates a municipal wastewater outfall 2.7km south of the Wairoa town centre near the mouth of the Wairoa River. The outfall services the needs of some 2,200 properties and around 4,400 residents in the reticulated urban area of the Wairoa township. The wastewater stream is largely from residential and light industry sources. However infiltration of the wastewater reticulation network by stormwater and groundwater during and after rainfall can significantly elevate flows. The only large industrial source of wastewater in the town is from the AFFCO freezing works which has its own onsite treatment units and outfall into the river at a site located approximately 2km upstream of the WDC outfall.

The reticulation network conveys wastewater to the Wairoa Wastewater Treatment Plant (WWWTP) which consists of a two pond aerobic and anaerobic decomposition system. Built in 1981 the plant incorporates coarse screening (<5mm), prior to influent entering the oxidation ponds. Pond 1 is a mixed primary pond which provides for facultative decomposition. The second stage of treatment occurs in a larger primarily anaerobic pond (pond 2), where some facultative decomposition occurs, but primarily pond 2 is described as a maturation pond.

From the elevated situation of the ponds at Pilot Hill the treated effluent exits the final stage anaerobic pond and is gravity fed to the outfall discharge port. The discharge port is located at or about E1982510 N5667389 (NZTM NZGD2000), in the sub-tidal area of the lower Wairoa River Estuary, approximately 150m from the shoreline (opposite the entrance to Fitzroy Street, Wairoa) and approximately 590 m north of the present location of the river mouth which opens into northern Hawke Bay (Figure 1).

The outfall is constructed of high density black polyethylene (internal Φ 300mm) and has in the past become buried in the sediment. In 2017 WDC attached a PVC riser to the outfall terminus to ensure blockages of the outfall were minimised. The outfall discharge port is simply the terminus of the outfall pipe. Effluent discharge occurs into the overlying water column, which during typical discharge conditions (i.e. during ebb tide beginning 30mins after high tide and between 6pm and 6am) varies in depth between 1 – 2m. The area immediately above the outfall terminus is termed the 'boil' given that the upwelling wastewater can at times be visible at the water's surface.

1.1 BACKGROUND

WDC currently holds consent (CD940404W), to discharge treated municipal wastewater into the Lower Wairoa River Estuary. Although the consent does not specifically address monitoring of the receiving environment three surveys, including the present one, have been carried out to assess the effect of treated wastewater discharge on macrobenthos and sediments around the outfall since 2007.

In April 2007 an assessment of ecological effects, incorporating high resolution dye dilution studies and a benthic survey and assessment of flatfish tissue was undertaken (Barter 2007; Smith 2007). In June of 2011 another benthic survey and assessment of effects on the receiving environment from the discharge was conducted (Smith 2011).

Prior to these studies the estuarine/riverine environment in the vicinity of the WDC outfall had been the subject of only a limited number of studies over the last 20 years. These include a coarse dye dilution study, and benthic and water quality surveys in 1996 at sites around the WWTP outfall (Larcombe 1996). In an attempt to disentangle the effects of upstream discharges from that of the WDC outfall a dye dilution study, benthic and water quality assessment around the AFFCO freezing works outfall was also conducted around the same time (Larcombe 1996).

Aside from the confounding effects of the AFFCO outfall on downstream assessments there are a number of other problems that have been identified by WDC with the outfall and associated WWTP and reticulation network. As mentioned above the outfall terminus has at times become covered in sediment and blocked, leading to the fitting of a short extension in May 2017. Another key issue is the occasional restriction of the estuary mouth, which can become partially or fully blocked heavily restricting river discharge into Hawke Bay. This generally occurs during heavy sea conditions when the shingle bar is built up particularly during easterly generated swell. Although the consent requires WDC to store effluent during these periods the WWTP storage capacity is often exceeded and thus effluent has to be discharged into the estuary during these 'restricted flow' conditions.

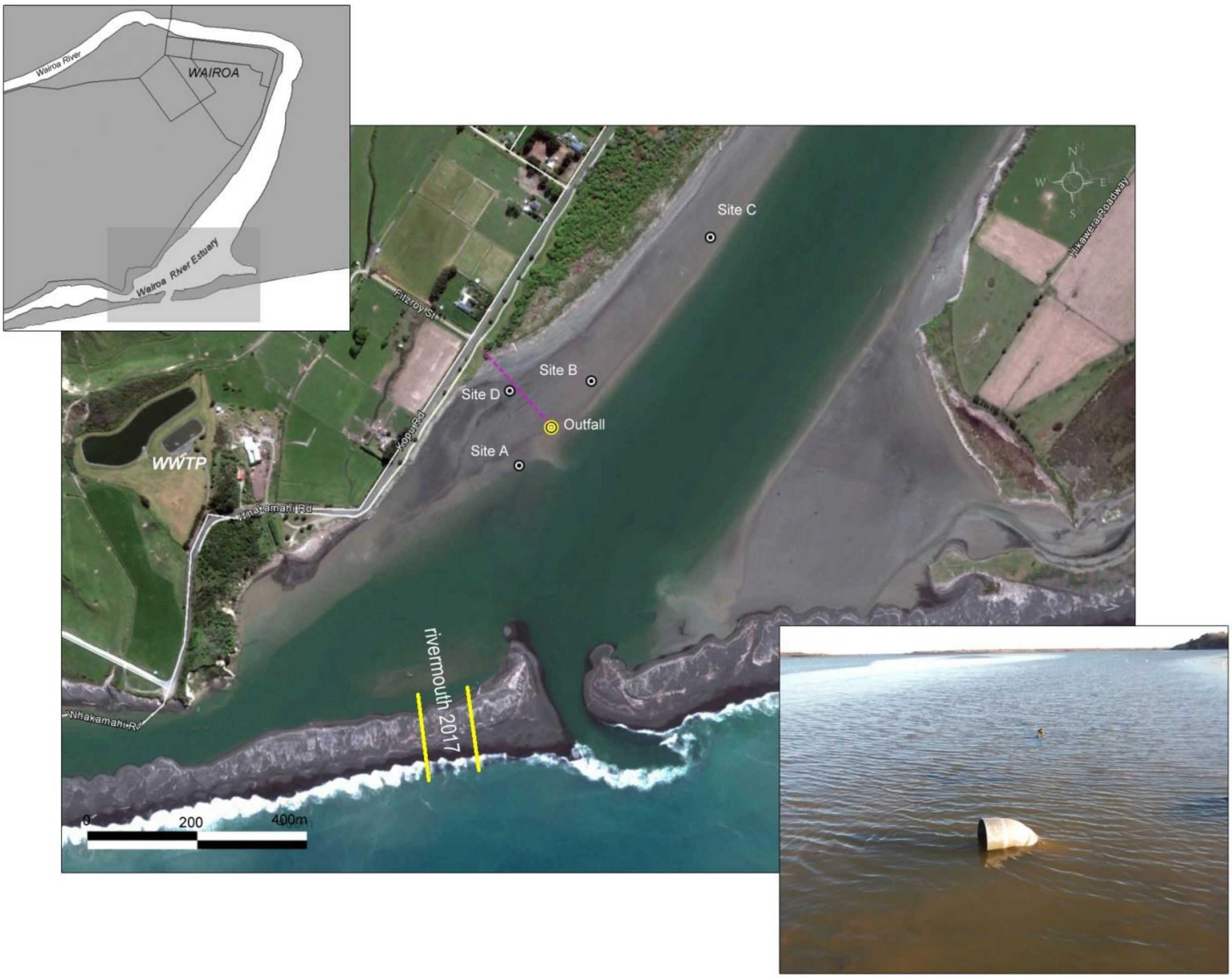


FIGURE 1: OVERVIEW OF THE LOWER WAIROA RIVER ESTUARY SHOWING THE LOCATIONS OF BENTHIC MONITORING SITES AROUND THE OUTFALL AND THE LOCATION OF THE RIVERMOUTH ON THE DAY OF SURVEY. (INSET: OUTFALL TERMINUS AT LOW TIDE ON THE DAY OF SURVEY).

The other key issue relates to the WWTP storage capacity being occasionally exceeded, largely as a result of infiltration of the aging reticulated wastewater network by stormwater and shallow groundwater. WDC records indicate a spike in flows during and immediately after rainfall events, with a study of infiltration indicating elevated flows of up to 2.4x normal up to six days following the cessation of rainfall (EAM 2011). This additional loading can overload the WWTP decreasing treatment efficacy and increasing the risk of flooding at the treatment ponds and consequently the potential for the discharge of treated wastewater into the estuary outside operational resource consent conditions. The potential effects on human health from a worst case scenario for outfall discharge, i.e. under 'restricted flow' conditions where the estuary mouth is partially or fully blocked, was undertaken as part of the dye study in 2007.

During this worst case scenario the treated wastewater plume does not disperse and remains close to the 'boil' with minimal transport, and effected only by wind. Dilutions of 5:1 at the 'boil' where typical while the plume moved only a maximum of 150m from the 'boil' (Barter 2007). Under these conditions there is a significant human health risk when using the lower estuary for contact recreation, especially if these conditions persist for an extended period of time. A second dye study under normal flow conditions (i.e. where the estuary mouth was not blocked) indicated wastewater dilutions were 5:1 at the 'boil', 50:1 at 125m downstream and 250:1 at 350m downstream (Barter 2007). Under these conditions, wastewater exits the estuary in a relatively short timeframe with the risk to human health moderate during discharge and less than minor at other times.

In terms of benthic effects, the 1996 survey found that the discharge had "no obvious effects on benthic biology or sediment quality" although nutrient enrichment of sediments was suggested as likely during 'restricted flow' conditions (Larcombe 1996). The 2007 survey found that "discharge effects on benthic infaunal communities around the outfall are evident, and may continue to alter composition over time" and that "sediments at sites around the outfall have declined in quality over time". The 2007 survey also included an examination of trace metal concentrations in flatfish tissue, and found that levels were well within food safety limits with no evidence of accumulation. Overall, it was considered the scale of effects from the discharge were within the assimilative capacity of the receiving environment and that the presence of a sensitive bivalve species, *pipi* (*Paphies australis*), at sites around the outfall suggested effects were not large enough to constitute an undue adverse effect. In the 2011 survey and assessment there was "…no indication of a discharge related effect, and results (sediment texture and chemistry) are similar to those observed at the upstream reference site, site C", while "assessment and analysis of the biota living in the sediments surrounding the outfall did not detect any significant differences in the community assemblages between sites suggesting that the outfall discharge has a limited influence on community structure".

1.2 THIS STUDY

To assess the effects of the treated wastewater discharge on the macrobenthos and physical environment at monitoring sites surrounding the outfall (Figure 1), WDC engaged Triplefin Environmental Consulting (Triplefin) to conduct a benthic survey of the receiving environment including the following components:

- Assessment of apparent redox depth of sediment cores, surficial sediment texture, organic content, trace elements including nutrient levels at outfall monitoring sites and at a suitable reference site.

- Assessment of benthic macroinfaunal communities at outfall monitoring sites and at a suitable reference site.

This report presents the findings of field survey work conducted in June 2017, subsequent data analyses, comparison to previous survey results and an assessment of effects. Previous surveys of the area around the existing outfall are referred to continuously throughout this report to provide a comparison to the present findings.

Where possible, the methods used in this survey were in keeping with those used in previous surveys. All methods used are fully detailed in the respective sections of this report.

1.3 RECEIVING ENVIRONMENT

The Wairoa River Estuary is commonly referred to as a barrier enclosed lagoon or drowned valley and is classified, according to the New Zealand estuary classification system (Hume, Snelder et al. 2007), as a category F estuary. Category F estuaries are characterised by a spit or shingle bar enclosing a large primary basin from which numerous arms lead off. This system gives rise to complex shoreline structure and supports extensive intertidal areas cut by deep channels. Wind mixing and wave driven re-suspension of sediments is generally minor given the limited fetch. Sediments tend to be muddier in the arms and sandier in the primary basin. The volume of river flow delivered over a tidal cycle is typically small compared to the total volume of the basin, and is generally less than the tidal volume entering the basin. Thus hydrodynamic processes tend to be dominated by the tides, meaning that these estuaries are moderately to poorly flushed.

In terms of wildlife and conservation values, the estuary and associated lagoons, the Ngamotu (western lagoon), and Whakamahi (eastern lagoon) are part of the nationally significant wetland lagoon system that stretches for 20km from the Whakamahi lagoon to the Whakaki lagoon (Seymour, Hogan et al. 1990). This system is the largest of its type in Hawke's Bay and supports endangered, threatened and rare species of wildlife. The expansive estuarine area enclosed within the unstable shingle bar at the mouth is approximately 200 Ha and includes extensive intertidal areas including large areas of permanent open water, ephemeral wetlands and intertidal mudflat habitat (approximately 81ha). This variety of habitat conditions consequently provide for recognised high biodiversity and Protected Natural Area (PNA) values (Seymour, Hogan et al. 1990). Formal protective designations include the Ngamotu Lagoon being a gazetted wildlife management reserve while the Whakamahi Lagoon is a conservation area managed by DOC (Seymour, Hogan et al. 1990).

Protected species occupying these habitats include the endangered white heron (*Egretta alba modesta*), the threatened banded dotterell (*Charadrus bicincta bicincta*) and the rare golden plover (*Pluvialis fulva*) (Seymour, Hogan et al. 1990). Migrant waders e.g. royal spoonbills (*Platelea regia*) also occur ephemerally in small numbers. The fisheries values of the estuary are also significant, particularly in terms of *Anguilla* sp. (eels), yellow-bellied flounder (*Rhombosolea leporina*), grey mullet (*Mugil cephalus*) and *Galaxias* sp (whitebait) especially inanga (*Galaxia maculata*).

Within the immediate surrounds of the outfall the environment is characterised as intertidal going to shallow subtidal mudflat, with the outfall located at or about Mean Low Water Springs (MLWS) and approximately 40m shoreward of the edge of the main channel.

Relevant literature that discusses the benthic habitat and epifauna of the area includes previous monitoring reports and data produced as part of the HBRC Estuarine State of the Environment (ESoE) monitoring program. Since 2010 the HBRC has monitored a site located approximately 600m upstream of the outfall for sediment and infaunal characteristics. These studies reveal a substratum predominantly comprised of sandy mud with some areas muddier and others sandier. Upstream of the outfall the intertidal flats are characterised by rippled muddy sand, while downstream of the outfall the substratum is muddier and rippling is absent. These sediments support a moderately diverse array of infaunal and epifaunal organisms, including polychaete worms, various bivalve and gastropod species, and small crustacea, particularly Corophid amphipods, and are indicative of a low energy, low salinity estuarine setting. Complex habitat is limited with the occasional large tree or tree limb held fast in the mud providing some complex 3D structure.

2. SEDIMENT CHARACTERISTICS

2.1 INTRODUCTION

Sediment characteristics can influence the distribution of benthic (bottom dwelling) invertebrates by affecting the ability of various species to burrow, build tubes or feed. In addition, demersal fish (fish that live on or near the bottom) are often associated with specific sediment types that reflect the habitats of their preferred prey. Both natural and anthropogenic factors affect the distribution, stability, quality and composition of sediments. Outfalls are one of many human derived (anthropogenic) factors that can directly influence the composition and distribution of sediments, and this occurs through the discharge of wastewater and subsequent deposition of a variety of compounds. In discharges from municipal wastewater treatment plants of the type WDC operate the most common effects result from input of fine sediment, predominantly as organic carbon, in the form of biosolids. Other contaminants of concern are volatile organic compounds, trace elements and bacterial and viral biomass. Moreover, the presence of outfall pipes or associated structures can alter the hydrodynamic regime in the immediate area surrounding the outfall.

This section presents a summary and analysis of sediment composition (grain size), organic matter and trace metal data collected in June 2017 in the vicinity of the WDC outfall. The aim is to assess the distribution and magnitude of any effects on sediments from the treated wastewater discharge.

2.2 METHODOLOGY

SAMPLING SITES

The location of sampling sites in the present survey were consistent with those sampled in previous surveys in 1996, 2007 and 2011, with the exception of site D, which has not been sampled previously and was included in the present survey (Figure 1, Appendix 1). Therefore, three "impact" sites were sampled, located in close proximity to the terminus of the outfall consisting of site A; approximately 100m south-west or downstream, site B; approximately 100m north-east or upstream of the outfall, site D 100m north-west, or inshore, of the outfall and a control, or "reference" site, site C; approximately 500m north-east of the outfall. These sites are deemed representative of the benthic environment surrounding the outfall while the pattern of their siting, allows adequate detection of the magnitude of outfall related effects in sediments.

METHODOLOGY

A handheld GPS unit was used to locate each site (± 3m). Samples were collected at low tide on the 29th June 2017. At each site, five haphazardly selected replicate sediment cores were collected using a PVC 60mm (internal Φ) x 150mm long corer. Cores were collected by pushing the corer into the sediment to a depth of 150mm and digging down 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 ejected onto a clean white tray and split vertically. Each core was visually assessed for the presence/absence of anoxic areas within the core and the apparent Redox Potential Discontinuity (aRPD) layer[1] measured. Cores were then photographed and the top 2cm of sediment from each half of the core placed into separate pre-labelled resealable plastic bags and immediately stored on ice. Each replicate sediment core was analysed for chemical composition, while the sediment texture samples from each site were composited, and a single sub sample prepared for analysis. The samples for organic matter assessment, trace elements and nutrients were analysed by Hill laboratories, Hamilton, while the sediment texture samples were analysed in-house. A summary of the analytical methods used are presented in Table 1.

[1]apparent Redox Potential Discontinuity (aRPD) layer - the brown coloured, oxygenated surface layer of sediments, distinct from the black anoxic layer beneath. Few macrobenthic species are able to live in anoxic sediments without some form of burrow, tube or respiratory siphon extended into the overlying sediments or water column.

Table 1: Summary of analytical methods used for sediment analyses

Parameter	Method	Description
Total recoverable elements As,Cd,Cr,Cu,Hg,Ni,Pb,Zn	Dry/sieve sample, Digestion US EPA 200.2	Total recoverable metals As,Cd,Cr,Cu,Hg,Ni,Pb,Zn
Texture	Wet sieving, gravimetric, Air drying 105°C overnight	Gravel >2mm Very Coarse Sand 1mm - 2mm Coarse Sand 500µm - 1mm Medium Sand 250µm - 500µm Fine Sand 125µm - 250µm Very Fine Sand 63µm - 125µm Silt and Clay <63µm
Total Recoverable P	USEPA 200.2	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, screen level.
Total N		Catalytic Combustion (900°C, O2), separation, Thermal Conductivity Detector [Elementar Analyser].
Total volatile solids (organic matter)	USEPA 2540 G 22nd ed. 2012	Ash - dried at 103°C. Ignition in muffle furnace 550°C, 1hr, gravimetric, TVS - calculation: 100 - Ash (dry wt).

DATA ANALYSIS

Trace element results were firstly compared against national sediment quality guidelines (ANZECC 2000) to assess relative toxicity and contamination status. These Interim Sediment Quality Guidelines (ISQG) consist of upper (ISQG-high) and lower (ISQG-low) thresholds above which biological effects can be expected. Where trace metal concentrations are below ISQG-low values then adverse biological effects are expected only on rare occasions. Trace metal concentrations falling between ISQG-low and ISQG-high are expected to cause adverse biological effects occasionally, while a result above the ISQG-high would be expected to cause adverse biological effects frequently. Currently there are no guidelines for assessing the effects of sediment-bound nutrient contaminants such as nitrogen or phosphorus, on the environment. If there are no obvious signs of enrichment at a site it may be difficult to assess a particular site for the effects of enrichment. Therefore concentrations of these contaminants were compared against other local and national reference sites.

In areas where anthropogenic influences are negligible, sediment trace element concentrations largely reflect the surrounding geology. Sediments formed under these conditions, contain trace elements at levels described as 'baseline/background concentrations. Therefore trace elements were also compared against a set of relevant trace metal "Regional Background Concentration" (RBC) levels (Strong 2005). The RBC's were derived from sediment trace metal levels at 17 study areas, comprising 67 sites, located throughout Hawke's Bay's estuarine, riverine and lagoon systems. The RBC's provide a relevant regional basis for assessing the impact of a particular activity on sediment quality.

Concentrations of certain elements, particularly metals, vary according to sediment properties[2]. Thus, normalization of metal concentrations is usually needed to differentiate between natural variability and anthropogenic input of contaminants. A commonly employed normalisation method for coastal marine and estuarine sediments is to normalise elemental constituents to the

[2] 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 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.

silt/clay ('fines') fraction of the sediment. Thus for spatial and temporal analyses of trace element levels data were normalised to the percentage of fines found at each site.

Spatial differences in sediment characteristics (texture, normalised TVS, normalised trace elements including P and N) between sites in the present survey were explored using one way ANOVA (StatSoft 2004). Results were reported as significantly different when p-values were <5% (i.e. testing at the 95% level of significance). The assumption of homogeneity of variance for ANOVA was checked using Levene's test. Tukey's HSD tests were used to assess differences between sites for each parameter. Results below detection limits were included in analyses at half detection limits in order to calculate means and standard errors.

To compare TVS data against the previous 1996 study the values for % organic carbon presented in the 1996 report had to be converted to % organic matter, or TVS, by multiplying by 1.724 (Metson, Saunders et al. 1971) before normalising.

Temporal analyses of sediment texture, organic matter content, and trace elements (within sites) were conducted using the non-parametric Mann-Kendall trend test (NIWA 2008). Trends in sediment characteristics were examined by computing a Mann-Kendall statistic, S, and associated p-value. Trends were considered to be significantly positive (i.e. increasing with time) or negative (i.e. decreasing with time) if the probability (two-sided p-value) of rejecting a correct hypothesis (in this case, no trend) was ≤ 0.05. Statistically significant trends were considered to be ecologically meaningful if the difference was more than 1% of the median value per annum.

For analyses of sediment texture the all sand fractions and the clay/silt fraction were tested while for analyses of trace elements, TVS, N & P the normalised levels for each site and year were calculated.

2.3 RESULTS

2.3.1 PRESENT SURVEY

Observations of the physical characteristics of benthic monitoring sites B, C and D were able to be made given they were uncovered at low tide, whereas site A remained covered by the tide to a depth of 0.6m throughout. Among sites B, C, and D the sediment surface had characteristic muddy sand ripples, oriented parallel to shore with the distance between peaks approximately 10cm.

SEDIMENT CORE PROFILES & REDOX STATUS

Photos of sediment core profiles showing aRDP depths are presented in Appendices 2 and 3. Cores were assessed for evidence of an outfall impact (e.g. colour, sheen, and odour) and the depth of the aRDP was measured.

At site A core profiles were fairly consistent with the aRDP layer depth very shallow among all cores (Table 2). In general site A cores were characterised by a muddy, organically rich, oxygenated layer of light brown sediment down to the aRDP layer depth. In the core photos, this layer is apparent as a light brown slurry at the top of the core that was not able to maintain its shape upon ejection of the core from the corer. Below the aRDP layer depth, sediments had a slightly blackened colour without sheen and a slight hydrogen sulphide odour was evident. Below the blackened layer sediments were dark grey coloured. These characteristics indicate sediments at site A are hypoxic, or poorly oxygenated, at shallow depths and potentially anoxic in places.

At site B core profiles were also consistent throughout the site with little variability in the aRDP layer depth (Table 2). Site B sediments were characterised as sandy, with a light grey slightly hypoxic zone down to the aRDP layer depth. No hydrogen sulphide odour was evident. Below this layer sediments were light brown in colour. These characteristics indicate moderate oxygenation to deeper layers of sediments.

At site D core profiles were more variable among replicates than sites A and B, though the average depth of the aRDP layer was similar to site B (Table 2). Despite the variability there were some general characteristics of core profiles noted, including the presence of a light grey layer of muddy sand at the surface, down to the aRDP layer depth. Below this sediments were variously light brown and light grey. However, among selected sites there was a distinctive blackened sediment layer, with light/dark grey sediments below. This layer occurred at various depths, and is likely the result of past accumulation of organic material at the surface that has been subsequently buried (e.g. sites D1, D2 and D5). No hydrogen sulphide odour was evident throughout the cores. These characteristics indicate moderate oxygenation of sediments to shallow depths with sediments below likely hypoxic.

At the upstream reference site C core profiles were similar among replicates with a fairly shallow aRDP depth. Despite surficial sediments being muddy and a shallow aRDP layer depth sediments were largely light or dark grey coloured with patches of blackened sediments in selected cores, e.g. C1 and C5. This was the only site where infaunal tubes were apparent, with each signalled by a thin rust coloured lining to the tube. There was no hydrogen sulphide odour detected among cores. These characters indicate sediments are poorly to moderately oxygenated.

Table 2: Mean depth of the redox potential discontinuity layer (RDPL) at WDC outfall monitoring sites (± 1 SE)

Site	RPDL depth (cm)	Sediment matrix
A	1.9 ± 0.1	Sandy mud
B	5.6 ± 0.4	Sand
C	2.6 ± 0.4	Sandy mud
D	5.7 ± 1.8	Muddy sand

Given the distinct differences in core profile characters among sites, it was not surprising that a one-way ANOVA comparing aRDP layer depths detected a significant difference among sites. However, post hoc Tukey's HSD tests did not detect which sites accounted for the difference. This suggests the strength of the overall among site difference was weak.

In general these results suggest that sites close to the outfall, and predominantly downstream and inshore, have reduced levels of oxygenation of sediments, which is likely a result of exposure to higher levels of deposited organic matter. The response to increased deposition of organic material differs among sites, with material at site D subjected to reworking and, or burial by currents, wave/tidal action and bioturbation, while at site A this material appears to be accumulating at the sediment surface.

SEDIMENT TEXTURE

Sediment texture data are detailed in Appendix 4. These show that surficial sediments at site A and reference site C were very similar in terms of composition with sites comprised primarily of silt/clay ('fines' or mud) (<63µm), with a secondary very fine sand fraction (63 – 125µm) and a subsidiary fine sand fraction (125 – 250µm)(Figure 2). In contrast site B was composed of sand, and primarily of fine sand, with a secondary very fine sand fraction and a small amount of fines. Compared to the other sites, sediments at site D were more evenly distributed among the various fractions, with approximately equal levels of fines and fine sand with the remainder very fine sand.

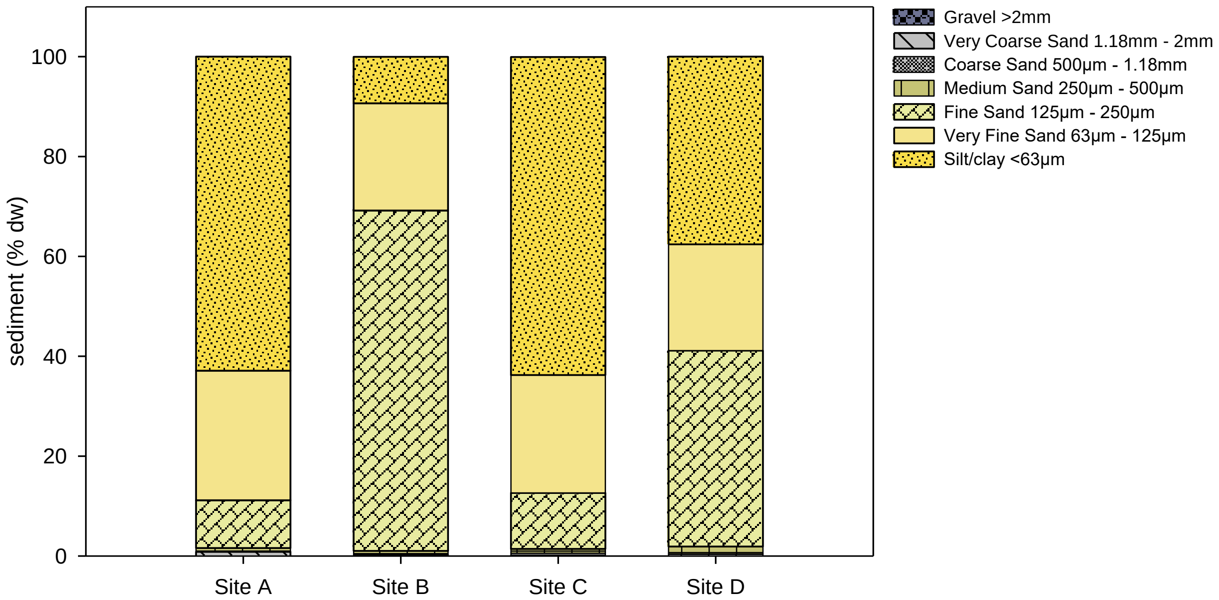


FIGURE 2: COMPARISON OF SEDIMENT TEXTURE AMONG WDC OUTFALL MONITORING SITES A, B, D ('IMPACT') AND C ('REFERENCE') DURING THE PRESENT (2017) SURVEY.

SEDIMENT TRACE ELEMENTS

The focus of this section is on concentrations of chemical elements: arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), mercury (Hg), nickel (Ni), lead (Pb) and zinc (Zn). These elements are often referred to as 'heavy metals.' However, given its ambiguity, e.g. arsenic is not regarded as a true metal, but a metalloid, the term 'trace elements' is used.

Concentrations of sediment trace elements are detailed in Appendix 4 and plotted by site in Figure 3. It should be noted that concentrations of Cd at site A (replicate A5) and site D replicates 3, 4, 5) were below laboratory detection limits of 0.1mg/kg. These detection limits were an order of magnitude higher than those for other samples. The reason for the increased detection limit was the requirement for a dilution as a result of matrix interference on Cd for these samples.

Comparison of levels to relevant national sediment quality guidelines (ANZECC 2000) showed that concentrations at all sites were well below ANZECC ISQG – Low sediment quality guidelines (Figure 3). At these levels the contaminant load at each site would rarely be expected to induce adverse biological effects, with all sites not considered to be contaminated. It is worth noting that the elevated level of Hg at site D is the result of an unusually high record in replicate D1 (0.67 mg/kg). The reason for this outlier is not clear.

It should also be noted that the ANZECC (2000) guidelines are designed to protect aquatic ecosystems rather than to protect human health. Although ISQG – Low values are lower than equivalent soil quality guidelines designed to protect human health, the aim of this study and the guidelines used in this study are related to the protection of aquatic ecosystems rather than human health.

The results were also compared to a set of relevant regional sediment background levels (RBC's) developed for estuarine/lagoon environments in Hawke's Bay (Strong 2005) (Figure 3). In general, concentrations of trace elements were below or within the confidence limits of background levels. The exception was Ni, which was slightly elevated at site A and C compared to the relevant Ni RBC.

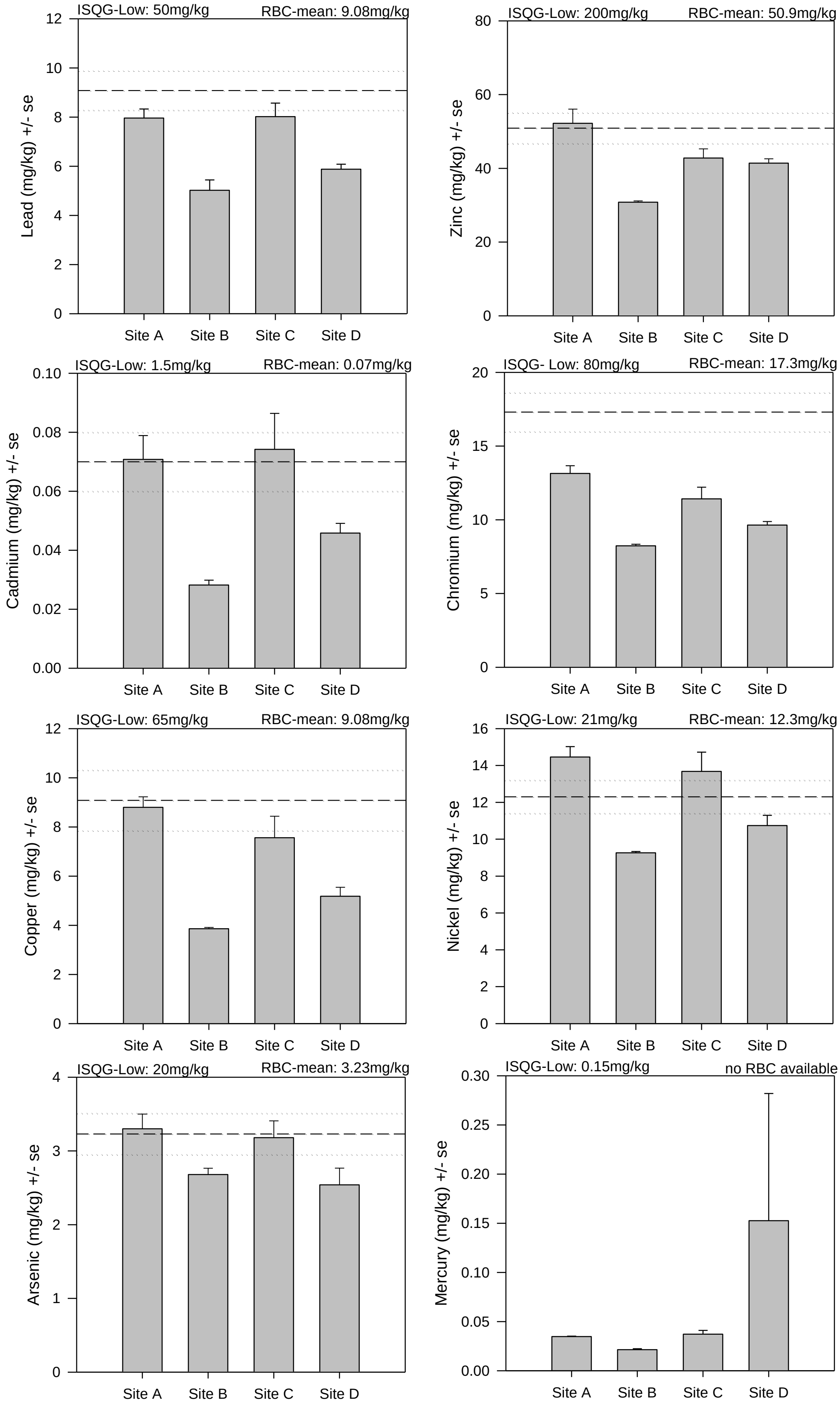


FIGURE 3: SEDIMENT TRACE ELEMENT CONCENTRATIONS AMONG WDC OUTFALL MONITORING SITES DURING THE PRESENT (2017) SURVEY. RELEVANT ANZECC ISQG TRIGGER LEVELS AND HAWKE'S BAY REGIONAL BACKGROUND CONCENTRATION (RBC) LEVELS OF TRACE ELEMENTS FOR ESTUARINE AND LAGOON SYSTEMS (± 95%CI) ALSO SHOWN FOR EACH RESPECTIVE PLOT (STRONG 2005). RESULTS EXPRESSED ON A DRY WEIGHT BASIS.

SEDIMENT TRACE ELEMENTS – COMPARISON BETWEEN SITES

Using normalised $\log_{10}$ transformed data, the comparison between sites estimated few significant differences, with the exception being site B which had significantly higher normalised levels of all trace elements compared to all other sites and site D which was significantly higher in Zn, Cr and Ni than sites A and C (Figure 3). These results however, are likely an artefact of the normalisation process wherein moderate trace element levels and a comparatively small 'fines' fraction result in high normalised levels. With this in mind it is worth noting that the two sites with very similar levels of fines, i.e. site A and site C, and for which comparison of normalised levels is meaningful, there was no significant differences detected. An alternative normalisation process that will provide for meaningful comparison among all sites is to conduct analyses on the sieved <63µm fraction, however it should be noted that these additional analyses will increase laboratory costs.

In general the results suggest sediments are not contaminated by trace elements, with levels immediately downstream of the outfall, i.e. at site A, no different to those found at the upstream reference site C. Moreover, given the effect on analyses from artefacts of the normalisation procedure, it is recommended that in future specific monitoring of trace element levels associated with the 63µm (fines) fraction to verify these observations is recommended.

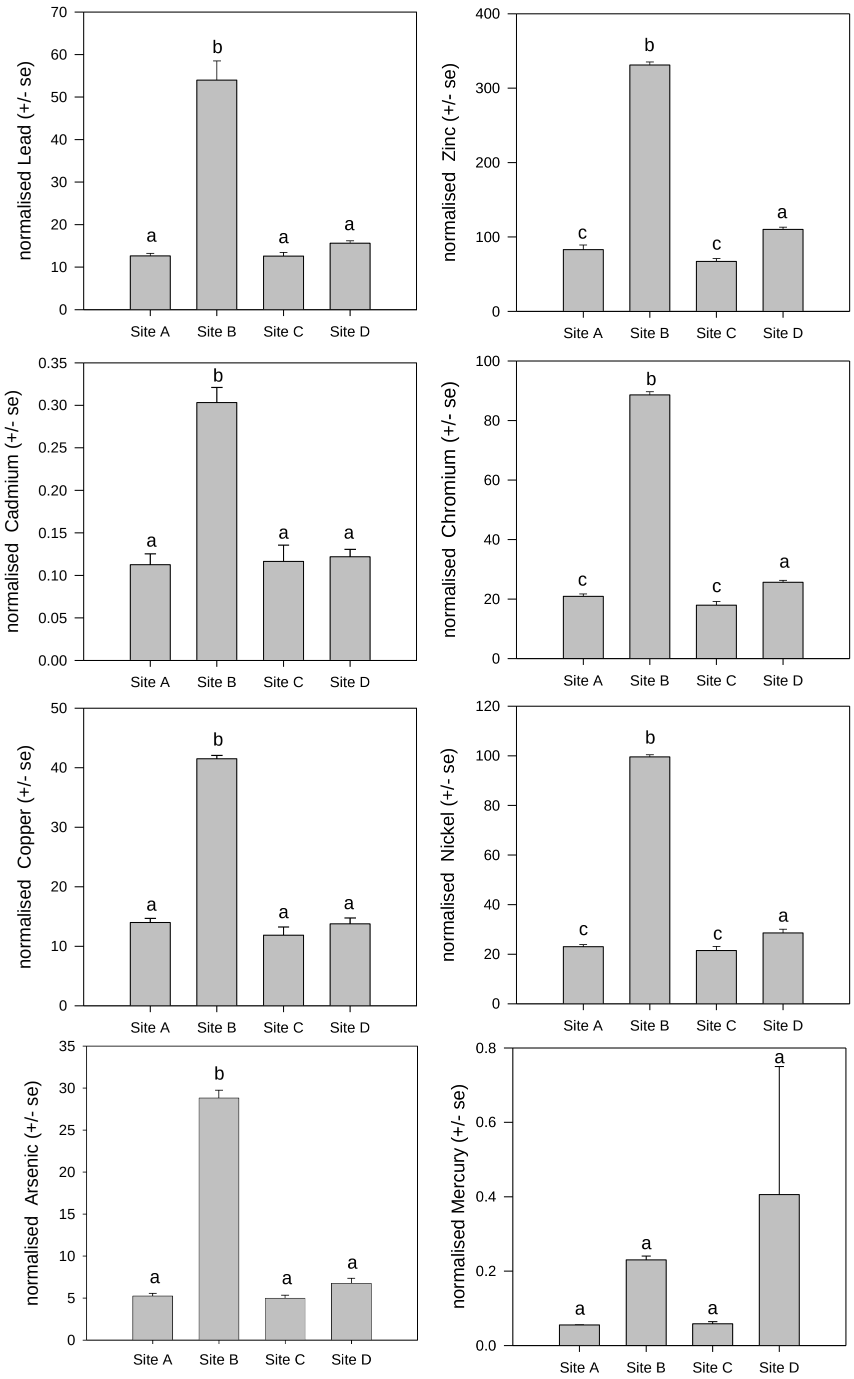


FIGURE 4: COMPARISON OF TRACE ELEMENT LEVELS NORMALISED TO 100% OF THE 'FINES' (SILT/CLAY OR MUD) CONTENT AMONG WDC OUTFALL MONITORING SITES DURING THE PRESENT SURVEY (2017). FOR A GIVEN ELEMENT BARS WITH THE SAME LETTER ARE **NOT** SIGNIFICANTLY DIFFERENT AT α = 0.05

SEDIMENT NUTRIENTS

Concentrations of sediment nutrient species nitrogen (N) and phosphorus (P) are detailed in Appendix 4 and plotted by site in Figure 5.

Although there are no New Zealand national sediment quality guidelines to compare against, from experience working in Hawke's Bay estuaries and waterways, it is typical to consider sites with phosphorus concentrations between 200-500mg/kg as low-moderately enriched, while levels between 500-1000mg/kg sites are considered enriched, and highly enriched for levels >1000mg/kg. In terms of total nitrogen, sites with levels between 0.05-0.2g/100g are viewed as low-moderately enriched while those with levels between 0.2-0.4g/100g are enriched and above 0.4g/100g, highly enriched.

These data therefore suggest that on average sediments at all sites are low – moderately enriched in both phosphorus and nitrogen. The effect of this level of enrichment on benthic communities is minor – moderate stress on sensitive organisms. A typical obvious sign of sediment nutrient enrichment is the proliferation of nuisance algae during summer, however as the survey was conducted during early winter (consistent with previous years) when benthic algal production is typically at its lowest no algae was expected and indeed none was evident. However, from experience in working in the Wairoa River estuary and associated tributaries during summer months, nuisance algal growth in the main basin and particularly in the area around the outfall has never been observed (pers. obs.).

Nutrient concentrations in sediment, similar to trace elements can vary depending on a variety of factors. These include the chemistry of overlying water e.g. pH, hardness and dissolved organic matter but chief among these is particle size and mineral composition of the sediment. To account for differences in sediment composition among sites, and on order to more accurately compare between sites sediment data were normalised to 100% of the silt/clay (mud/fines).

Using normalised $\log_{10}$ transformed data, the comparison between sites showed that, similar to the various trace elements, site B was significantly higher in both N and P compared to all other sites, while site D was significantly higher in P and lower in N than sites A and C, which were not significantly different to each other (Figure 6). As noted above the small amount of fines at site B has resulted in a high normalised level for N and P, however as also noted above the similar levels of fines at site A and site C make for a more meaningful comparison of normalised levels, and indeed there was no significant difference detected between these sites.

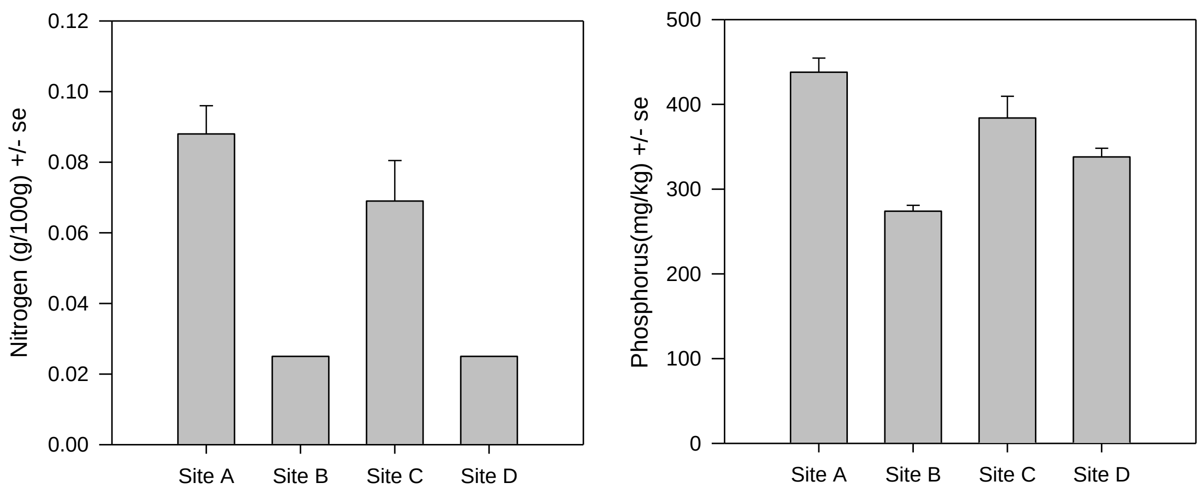


FIGURE 5: COMPARISON OF SEDIMENT NUTRIENT CONCENTRATIONS (TOTAL NITROGEN AND TOTAL RECOVERABLE PHOSPHORUS) AMONG WDC OUTFALL MONITORING SITES DURING THE PRESENT SURVEY. RESULTS EXPRESSED ON A DRY WEIGHT BASIS.

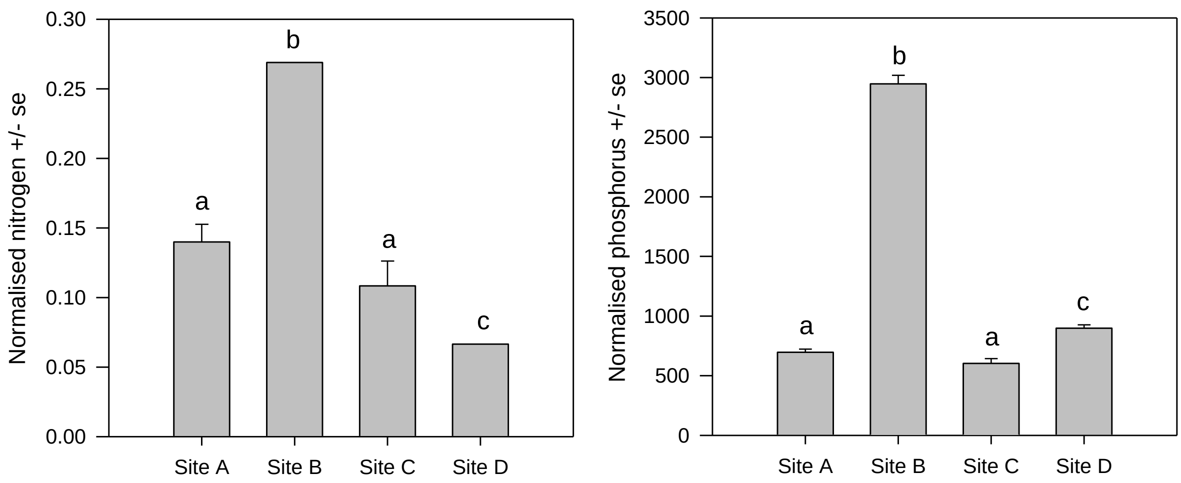


FIGURE 6: COMPARISON OF SEDIMENT NUTRIENT LEVELS NORMALISED TO 100% OF THE 'FINES' (SILT/CLAY OR MUD) CONTENT AMONG WDC OUTFALL MONITORING SITES DURING THE PRESENT SURVEY (2017). FOR A GIVEN ELEMENT BARS WITH THE SAME LETTER ARE **NOT** SIGNIFICANTLY DIFFERENT AT α = 0.05.

TOTAL VOLATILE SOLIDS

Total volatile solids (TVS), also referred to as Ash Free Dry Weight (AFDW) represents the amount of organic matter present in the sediments. Site averages are shown in Figure 7 and data are detailed in Appendix 4.

Interim thresholds for organic matter for a wide range of New Zealand estuaries have been developed as part of the national Estuarine Trophic Index (ETI) that provides recommended rating thresholds for a variety of estuarine trophic health indicators (Robertson, Stevens et al. 2016). However these have been developed using total organic carbon and not TVS. Therefore in order to compare results from monitoring sites in the present survey to the thresholds the thresholds were converted to TVS by multiplying by 1.724. Therefore sites with %TVS < 0.9 are placed within the 'A' band characterised as no stress caused by the indicator on any aquatic organisms, TVS of 0.9 – 1.724% is in the 'B' band characterised as minor stress on sensitive organisms caused by the indicator, 1.724 – 3.45% places a site in the 'C' band, characterised by moderate stress on a number of aquatic organisms caused by the indicator exceeding preference levels for some species and a risk of sensitive macroinvertebrate species being lost, while the lowest quality band is for sites with TVS > 3.45% and is classed as in the 'D' band, and characterised as significant, persistent stress on a range of aquatic organisms caused by the indicator exceeding tolerance levels with a likelihood of local extinctions of keystone species and loss of ecological integrity.

Examining TVS results from the present survey in the context of these ecological quality bands it is clear that site A would be rated in the 'D' band, sites B and C would be rated in the 'B' band and site D would be rated in the 'C' band (Figure 7).

As with trace elements and nutrients, organic matter in sediments is assumed to be higher at sites with a higher fines fraction, and therefore TVS data were also normalised to 100% of the fines fraction for each site in order to compare between sites. Comparing normalised TVS between sites it was apparent that all sites differed significantly from each other (Figure 8). As previously noted the high normalised level at site B is likely an artefact of the normalisation process given the small amount of fines compared to all other sites. Of the two sites with comparable levels of fines, i.e. site A and C, ANOVA estimated that site A was significantly higher than site C, with site D significantly higher than site C, but lower than site A. Because the amount of organic matter in sediments can have a significant influence on the depth of the aRDP layer depth, the presence of differences in normalised results between sites provides additional confidence in the results comparing aRDP layer depths.

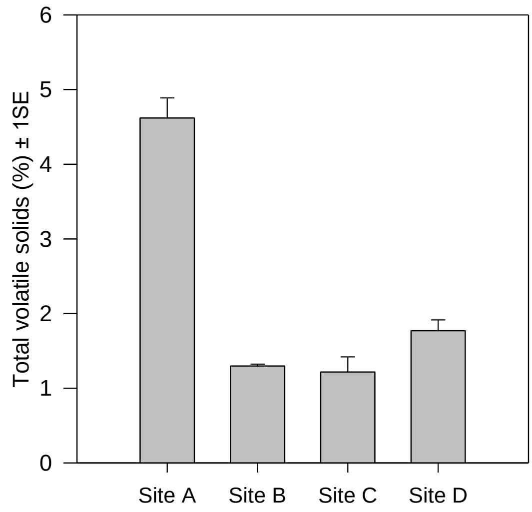


FIGURE 7: COMPARISON OF % SEDIMENT TVS (ORGANIC MATTER CONTENT) AMONG WDC OUTFALL MONITORING SITES A, B, D ('IMPACT') AND C ('REFERENCE') DURING THE PRESENT (2017) SURVEY.

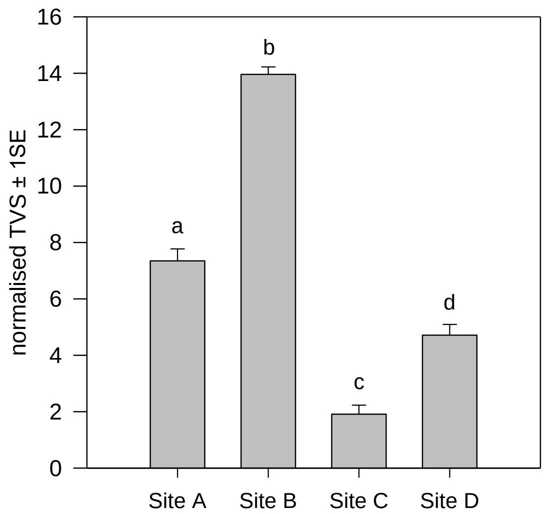


FIGURE 8: COMPARISON OF SEDIMENT TVS LEVELS NORMALISED TO 100% OF THE 'FINES' (SILT/CLAY OR MUD) CONTENT AMONG WDC OUTFALL MONITORING SITES DURING THE PRESENT SURVEY (2017).

2.3.2 TEMPORAL ANALYSES

SEDIMENT TEXTURE

The temporal variation of sediment texture among sites is shown in Figure 9. Given the present survey is the first time site D has been sampled it was not included in temporal analyses.

At site A the amount of fines increased between 1996 and 2011 with a slight reduction recorded in the present survey. At site B dominant fractions have tended to be more variable over time, with the results from the present survey more similar to those in 1996 and 2007 than the 2011 survey. Site C appears to have been the most consistent through time though there has been a change in the dominant fractions around the time of the 2011 survey, from sand (1996 and 2007) to fines (2011 and 2017).

To provide an estimate of whether these observations within sites, and over time, were significant or not, the data were subjected to Mann-Kendall trend testing (Jowett 2012). These tests estimated no significant trends for any of the fractions, and among any of the sites.

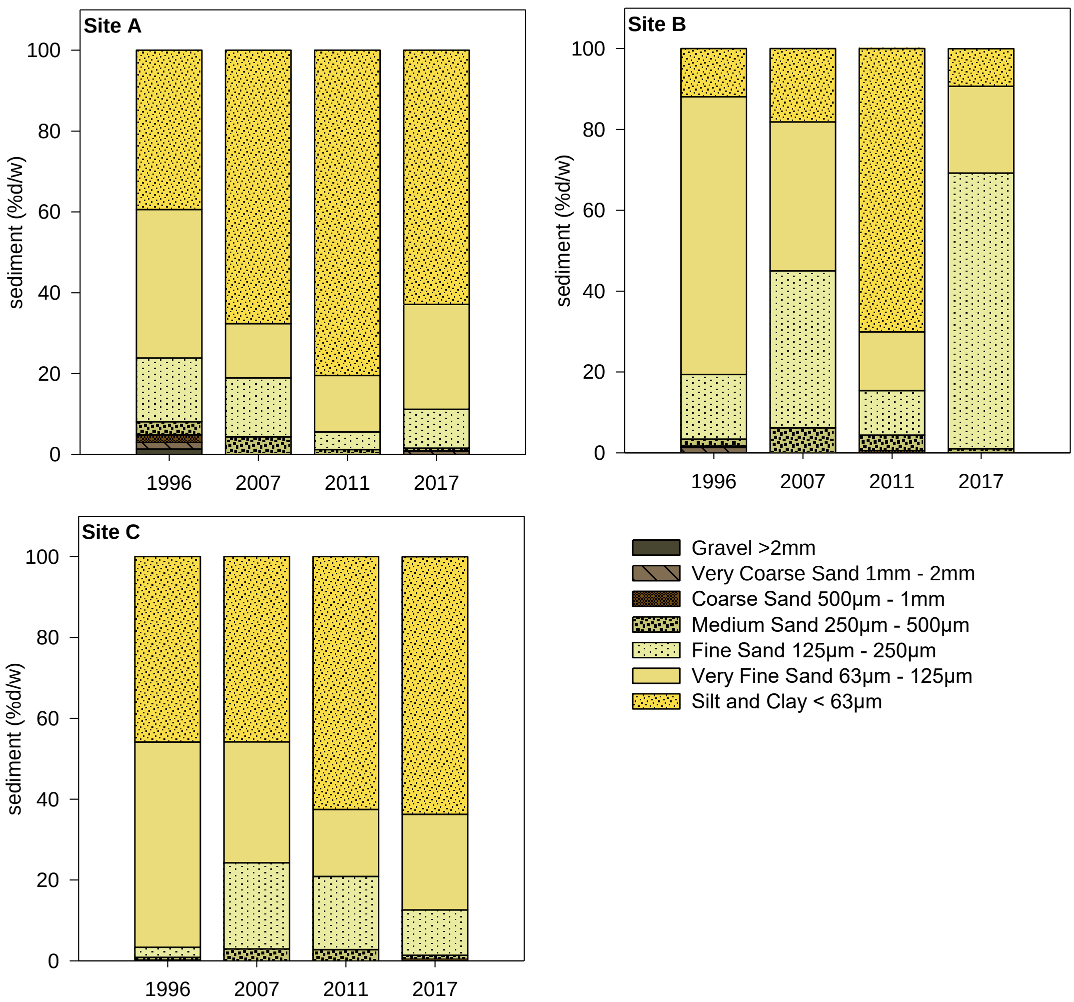


FIGURE 9: TEMPORAL COMPARISON OF SEDIMENT TEXTURE AMONG WDC OUTFALL MONITORING SITES DURING SURVEYS IN 1996, 2007, 2011, AND THE PRESENT (2017).

TRACE ELEMENTS

For an analysis of within site temporal variability, concentrations were normalised to 100% of the fines fraction, plotted (Figure 10), log_{10} transformed and subjected to Mann-Kendall trend testing. No analyses of sediment trace elements were conducted in the 1996 survey.

At site A, 6 statistically significant positive trends, i.e. increasing over time, were estimated, with As and Hg the only trace elements where a trend was not detected. The mean rate of increase among trace elements where positive trends were estimated was 3.6% ± 0.32 (1SE) per annum.

At site B, the large proportion of fines in the 2011 survey had an effect on normalised levels, and as a result no significant trends were detected.

At site C a total of 3 statistically significant trends were detected, and these were all negative, i.e. decreasing levels of normalised As, Cr, and Zn. The mean rate of decrease for these elements was estimated to be 2.55% ± 0.47 (1SE) per annum.

In summary, although normalised data has been used to undertake trend testing which has precluded a meaningful analysis of site B data, the more consistent sediment texture over time among sites A and C allowed a greater level of confidence in trend test results. These generally suggest sediment quality at site A, in terms of Cd, Cr, Cu, Ni, Pb and Zn is deteriorating over time. At the reference site C levels are relatively stable, if not slightly improving over time.

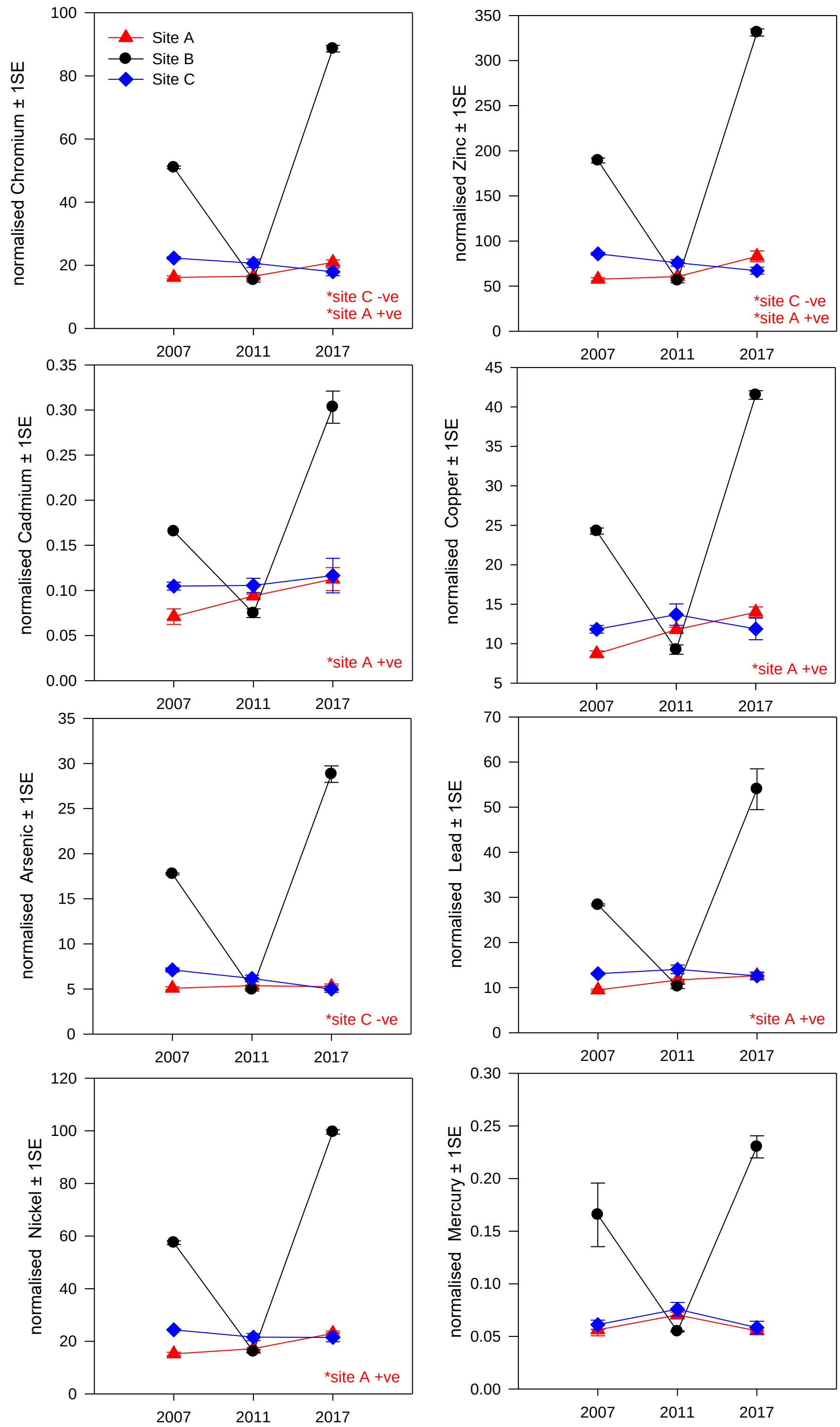


FIGURE 10: TEMPORAL COMPARISON OF TRACE ELEMENT LEVELS NORMALISED TO 100% OF THE 'FINES' (SILT/CLAY OR MUD) CONTENT AMONG WDC OUTFALL MONITORING SITES DURING SURVEYS IN 1996, 2007, 2011 AND THE PRESENT (2017). STATISTICALLY SIGNIFICANT TRENDS ARE MARKED WITH AN ASTERISK {*) WITH DIRECTION OF TREND; POSITIVE (+VE) OR NEGATIVE (–VE), ALSO SHOWN.

NUTRIENTS

NITROGEN

Examination of the inter-survey normalised nitrogen plot for site A reveals a slight increase in each successive survey and a low overall level of variability. Trend testing detected a significant positive trend, estimated to be 2.6% per annum (Figure 11). These results suggest that there has been a gradual increase in the normalised levels of N in sediments at site A over time.

A significant positive trend was also detected in normalised nitrogen at site B despite the extreme inter year variability. However, given the low concentrations of nitrogen in sediments at site B during the present survey, which were all at levels below the laboratory detection limit, and known limitation of normalisation to the fines fraction at sites with a small fines content the positive trend at site B should be viewed with some scepticism.

Temporal variability at site C was limited with levels fairly stable and no apparent trends detected.

PHOSPHORUS

Normalised inter-survey phosphorus levels at site A were similarly fairly stable with Mann-Kendall trend testing verifying the absence of any trend (Figure 11). Conversely normalised levels at site B exhibited a high degree of inter-survey variability, though in this case no trend was detected.

At reference site C there was an apparent slight decrease in each successive survey. This reduction over time was confirmed as a significant negative trend.

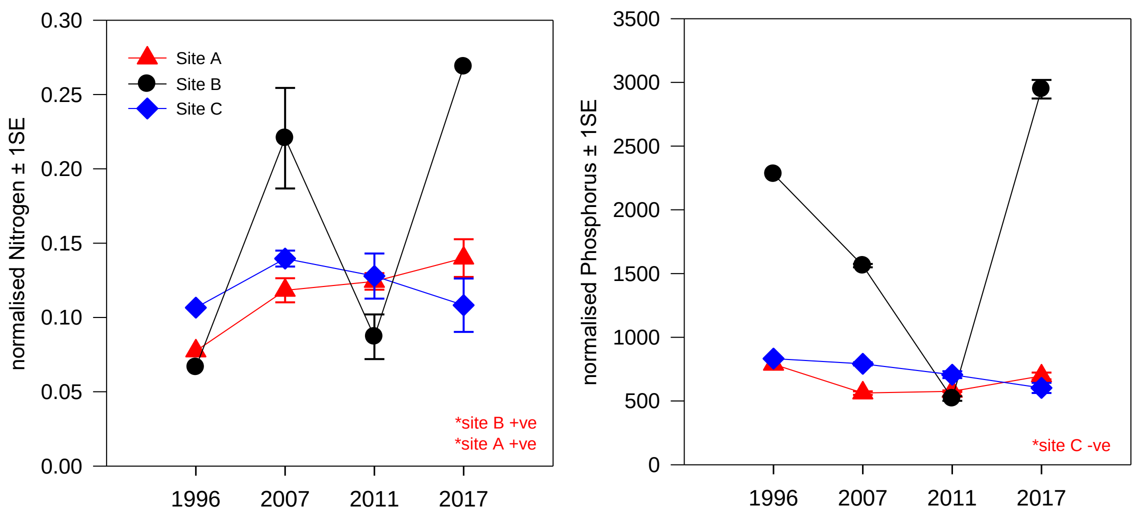


FIGURE 11: TEMPORAL COMPARISON OF SEDIMENT NUTRIENTS NITROGEN AND PHOSPHORUS NORMALISED TO 100% OF THE 'FINES' (SILT/CLAY OR MUD) CONTENT AMONG WDC OUTFALL MONITORING SITES DURING SURVEYS IN 1996, 2007, 2011 AND THE PRESENT (2017). STATISTICALLY SIGNIFICANT TRENDS ARE MARKED WITH AN ASTERISK {*) WITH DIRECTION OF TREND; POSITIVE (+VE) OR NEGATIVE (–VE), ALSO SHOWN.

TOTAL VOLATILE SOLIDS

The temporal variation of sediment TVS among sites is shown in the fines normalised plots of Figure 12.

These results show normalised TVS levels at site A increasing over time which was confirmed by trend testing that estimated a significant positive trend with levels increasing by 4.98% per annum. A significant positive trend was also estimated at site B while no trends were detected at site C.

Another interesting observation is the striking similarity between the pattern of organic matter results among sites and the pattern observed for total nitrogen (*see* Figure 11). The close resemblance between organic matter and nitrogen patterns in this system suggests that much of the nitrogen present is from the degradation of organic matter, i.e. much of the nitrogen is produced by bacteria involved in nitrogen fixation.

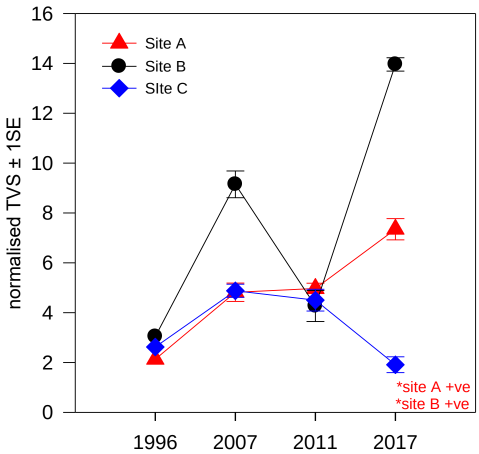


FIGURE 12: TEMPORAL COMPARISON OF TOTAL VOLATILE SOLIDS (ORGANIC MATTER CONTENT) NORMALISED TO 100% OF THE 'FINES' (SILT/CLAY OR MUD) CONTENT AMONG WDC OUTFALL MONITORING SITES DURING SURVEYS IN 1996, 2007, 2011 AND THE PRESENT (2017). STATISTICALLY SIGNIFICANT TRENDS ARE MARKED WITH AN ASTERISK {*} WITH DIRECTION OF TREND; POSITIVE (+VE) OR NEGATIVE (–VE), ALSO SHOWN.

2.4 SUMMARY

PRESENT SURVEY

- Sediment oxygenation, indicated by depth of the aRDP layer, at the upstream reference site C was moderate, at the site immediately upstream and inshore of the outfall, sites B and D respectively, oxygenation was moderate – good, whereas at the site located immediately downstream of the outfall, site A, sediments were hypoxic at a shallow depth and, in places, anoxia was developing and therefore oxygenation was poor.

- Sediment texture among sites A and C was similar and characterised primarily by mud, or 'fines'(<63µm) with a sand subsidiary while sites B and D were predominantly very fine/fine sand with site B more sandy and site D more muddy.

- Trace element levels at all sites were low with all results below ANZECC ISQG – Low sediment quality guidelines, and were in general within the range of average background concentrations of Hawke's Bay estuarine and lagoonal systems. Evidence of a significant outfall effect in terms of sediment trace elements was absent with normalised levels not significantly different between site C and site A.

- Concentrations of sediment nutrients, N and P, at all sites were within the moderate range among Hawke's Bay estuaries suggesting low-moderate enrichment of sediments. Normalised levels of N and P at site A where not significantly different to the comparable reference site C, indicating the absence of a significant outfall effect on sediment nutrient levels.

- Total volatile solids, or organic content, of sediments differed significantly among all sites, with the most polluted site being site A where levels are likely to induce persistent and significant stress on resident infauna and loss of ecological integrity. Levels at the inshore site D were rated as likely to induce moderate stress on infauna. The least polluted sites in terms of TVS were sites B and reference site C which were rated as likely to induce only minor stress on infauna with ecological integrity unlikely to impact ecological integrity.

TEMPORAL ANALYSES

- Sediment texture at site A and C were relatively more consistent over time, compared to site B, which returned to a predominantly sandy composition in the present survey compared to being predominantly mud in the previous 2011 survey. Trend testing did not provide any evidence of significant trends among any sites for any of the sediment fractions.

- Temporal trend testing of trace elements normalised to the fines fraction revealed 6 statistically significant positive trends, i.e. increasing levels, in Cd, Cr, Cu, Ni, Pb and Zn at site A, and 3 significant negative trends in As, Cd and Zn at reference site C. These results suggest a slight deterioration in sediment quality over time at site A in terms of these elements and a slight improvement at site C.

- Trend testing of sediment nitrogen levels estimated increasing trends at site A and B, though the result for site B is considered tenuous, and no trends at site C. In terms of phosphorus, levels among all sites were fairly stable, with the only significant trend among sites, a negative trend at site C.

- TVS at site A was estimated to be increasing over time, while at site C levels were stable with no trends detected. A positive trend was also estimated at site B, though this result is viewed with some scepticism. It was also noted that the levels of TVS were likely to be driving the sediment N levels given the similarity in variability over time of N and TVS.

3. BIOLOGICAL CHARACTERISTICS

Benthic infauna (species residing within the sediment matrix) form diverse communities that are important to the coastal and estuarine ecosystems. These animals serve vital functions in a wide variety of capacities, for example some species decompose organic matter, aiding nutrient cycling, while other species filter particulate matter from the water, affecting water clarity. Many species of benthic macroinfauna are also prey for fish and other organisms. Human activities such as wastewater disposal impact the benthos in a number of ways; including smothering, oxygen depletion, toxic contamination, and organic enrichment. Some macrofaunal species are highly sensitive to such effects and rarely occur in impacted areas, while other, more opportunistic species can thrive under altered conditions. Different species respond differently to environmental stress, so monitoring macrobenthic assemblages can help to identify anthropogenic impact. Also, since the animals in these assemblages are relatively stationary and long-lived, they integrate environmental conditions spatially and over time. Consequently, the assessment of benthic community structure is a major component of many marine monitoring programs, which document both existing conditions and trends over time.

This section presents analyses and interpretations of benthic macrofaunal data collected at outfall monitoring sites during June 2017 in the vicinity of the WDC outfall, with the aim to assess any effects on the benthic biology from the treated wastewater discharge.

3.1 METHODOLOGY

SAMPLING SITES

Biological sampling sites were the same as those described in section 2.2, Sediment Characteristics and are shown in Figure 1.

METHODOLOGY

Infaunal sampling was carried out at the same time as the respective sediment sample was collected. At each site five replicate infaunal cores were collected a 5m radius of the sites GPS position. Infaunal 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 150mm and digging down the outside of the core, placing a hand over the bottom, extracting the core and intact sample and ejecting the sample into a 0.5mm mesh bag, which was attached to the top of the core. The mesh bag was then detached from the core and the sediment in the sample was gently washed through the bag, leaving only the infauna. Samples were washed into labelled jars with 80% ethanol and fixed in same. After transporting samples back to the lab a few drops of Rose Bengal solution was added to each sample, and left for several hours to allow the biota to uptake the stain. Samples were then poured into shallow trays and all biological material carefully picked out. The material was then examined under a dissecting microscope, and fauna enumerated and identified to the lowest possible taxonomic group.

DATA ANALYSIS

Spatial differences in the number of individuals - N and diversity indices (collectively called biological summary indices) consisting of S - number of taxa, H' - Shannon-Weiner diversity index, J' - Pielou's evenness and d - Margalef's richness) between sites in the present survey were explored using one-way ANOVA (STATISTICA 7), with post hoc analysis of individual terms by Tukeys HSD tests. Levene's test was used to test the assumption of homogeneity of variance for parametric analyses.

Temporal analyses were conducted using the non-parametric Mann-Kendall trend test {NIWA, 2008 #90}. Trends in biological summary indices were examined by computing a Mann-Kendall statistic, *S*, and associated *p*-value. Trends were considered to be significantly positive (i.e. increasing with time) or negative (i.e. decreasing with time) if the probability (two-sided *p*-value) of rejecting a correct hypothesis (in this case, no trend) was ≤ 0.05. Statistically significant trends were considered to be ecologically meaningful if the difference was more than 1% of the median value per annum.

To assess the benthic community composition, both spatially and temporally,, data were analysed using a permutational multivariate analysis of variance (PERMANOVA) (Anderson 2005). This method of data analysis is regarded as a powerful way to test the significance of taxonomic compositional changes (Walters and Coen 2006).

The model was based on permutation of raw data for the fixed factor 'site' and or 'year'. Data were log(x+1) transformed before analysis, as this type of transformation scales down the effect of highly abundant species thus increasing the equitability of the dataset (variance standardisation). Data were also contrasted using non-metric multidimensional scaling (Kruskal and Wish 1978) ordination based on the Bray-Curtis distance matrix in PRIMER v5.

Additionally, major taxa contributing to the similarities of each site were identified using analysis of similarities (SIMPER) (Clarke and Warwick 1994; Clarke and Gorley 2001).

Although the size of the infaunal samples collected in the 1996 monitoring survey was $0.05m^2$, compared to $0.013m^2$ used in subsequent surveys, data were compared without scaling. The reason for not scaling results is that the number of species detected in a sample usually changes much more in relation to sample size or sampling intensity than the distribution of relative abundances (Huston 1997).

3.2 RESULTS

3.2.1 PRESENT SURVEY

BIOLOGICAL SUMMARY INDICES

SPECIES ABUNDANCE, DIVERSITY, RICHNESS AND EVENNESS

A complete list of benthic infaunal data from the present survey is included in Appendix 5. Biological summary indices are presented in Figure 13.

The most commonly encountered species among sites was the polychaete worm *Boccardia* sp. accounting for just under half of all individuals counted, though this species was absent from sites A and C. The second most common species was another polychaete worm, *Nicon aestuariensis,* (except for site D). The third most common species was the estuarine snail *Potamopyrgus estuarinus* (except for site A), followed by the amphipod *Paracorophium excavatum* (except for site D). The fifth most common species was the filter feeding bivalve, pipi, *Paphies australis* (except for site A). In general polychaete species dominated the abundance among all sites except for site D where molluscs were numerically dominant.

The number of taxa (S), taxa richness at each site was low, with a total of 12 species identified across all sites. Site B had the highest average diversity (3-7 species) however no significant differences were detected between sites.

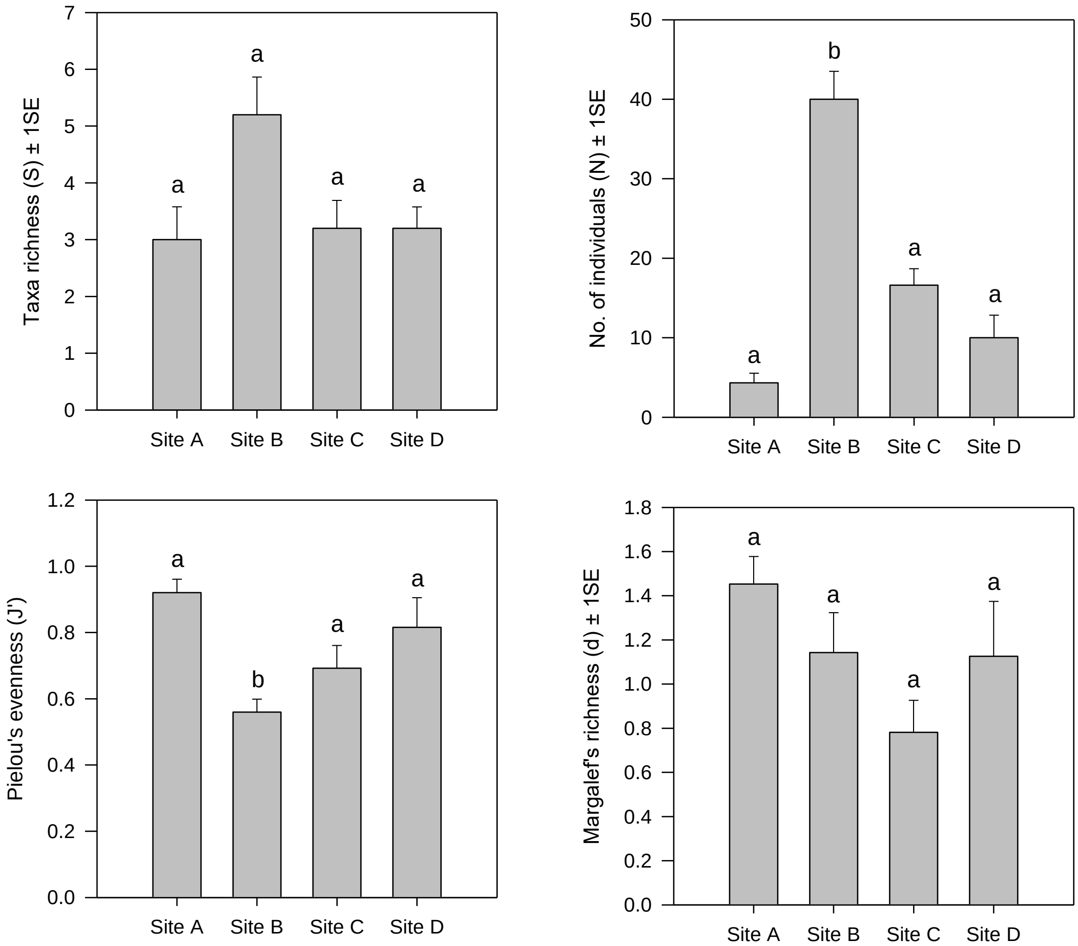
Number of individuals in each core, or abundance, (N) was low-moderate across all sites, with site B recording the highest average abundance that was estimated to be significantly higher than all other sites. Although the lowest abundance was recorded at site A there was no significant difference between sites A, C and D.

Margalef's Richness (d), is described as a measure of biodiversity based on the number of species adjusted for the number of individuals sampled. Values increase with the number of species and decrease with relative increases in number of individuals. The highest average score was at site A, followed by sites B and D and then site C. However, these differences were estimated as non-significant.

Pielou's evenness (J') is a measure of the similarity of the abundances of different species in a group or community, and the nearer values are to 1 the more even abundances are among species. In the present survey evenness at sites A, C and D was estimated as significantly higher than site B, likely as a result of the inconsistency in occurrence of some taxa throughout site B.

The Shannon diversity index (H') is a measure of the likelihood that the next individual will be the same species as the previous individual, the higher the number the more diverse the sample. In the present survey there were no significant differences estimated between sites.

In general, there were few statistically significant differences among sites for the various summary indices. Given the lack of power, i.e. significant differences, the magnitude of any outfall related effects on infaunal community characteristics, e.g. decreased diversity and evenness, at sites immediately around the outfall compared to the reference site is therefore slight.



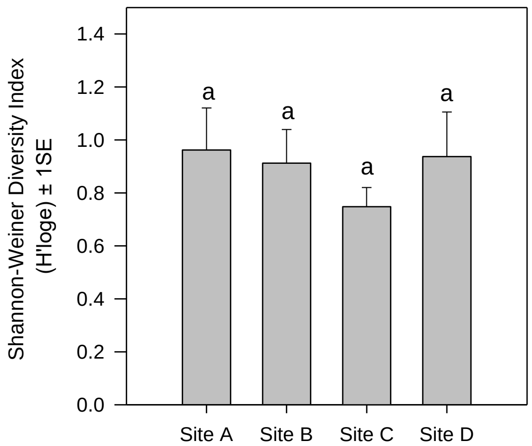


FIGURE 13: SPATIAL COMPARISON OF DIVERSITY INDICES OF BENTHIC MACROINFAUNAL COMMUNITIES INCLUDING TAXA RICHNESS (S), ABUNDANCE (N), MARGALEF'S RICHNESS (d), PIELOU'S EVENNESS (J') AND THE SHANNON-WEINER DIVERSITY INDEX (H') FROM THE PRESENT (2017) SURVEY AT MONITORING SITES AROUND THE WDC OUTFALL. FOR A GIVEN INDEX BARS WITH THE SAME LETTER ARE **NOT** SIGNIFICANTLY DIFFERENT AT α = 0.05

INFAUNAL COMMUNITY STRUCTURE

Multivariate analysis of infaunal data allows a comparison of community structure between sites, and years. Similarities in species abundance between sites and years are expressed on a two dimensional plane called a non-metric multi-dimensional scaling (nMDS) plot. The plot comparing infaunal communities between sites in the present survey (2017) shows distinct separation between all sites (Figure 14).

PERMANOVA results confirm this graphical separation observed in the plot as significant (*p*MC = 0.001) (Table 3). Pair-wise *a posteriori* comparisons among sites estimated that community structure at each site was significantly different to every other site.

Table 3: PERMANOVA results examining the effect of site on benthic macroinfauna during the present (2017) survey. All data were ln(x+1) transformed, and analysis was based on Bray-Curtis similarities. P (perm) indicates the permutational p-value, P(MC) indicates the Monte Carlo p-value.

Source	df	SS	Mean Square	F-Value	P (perm)	P (MC)
Site	3	31483.2	10494.4	12.1	0.001*	0.001*
Residual	16	13782.9	861.4			
Total	19	45266.2				

'*' indicates significant result

A SIMPER analysis was used to assist identification of species associations that account for the observed differences in community structure between sites and are shown in Table 4. The key species at site D was the estuarine snail, *Potamopyrgus estuarinus* which is in fact classed as epifauna, i.e. they reside at the sediment surface, and so are somewhat less affected by adverse sediment conditions than infauna. Among sites A and C and to a lesser extent at site B the estuarine omnivorous polychaete worm *Nicon aestuariensis* was key. This species is found in sediments with a wide range of fines content however in general its presence is strongly positively correlated with increasing fines. Another polychaete, the surface deposit feeding worm *Boccardia* spp. which typically occurs at sites with an intermediate level of fines, characterised the assemblage at site B and was of secondary importance at site D. Also of secondary importance in driving the assemblages at site A and B was the estuarine amphipod *Paracorophium excavatum*.

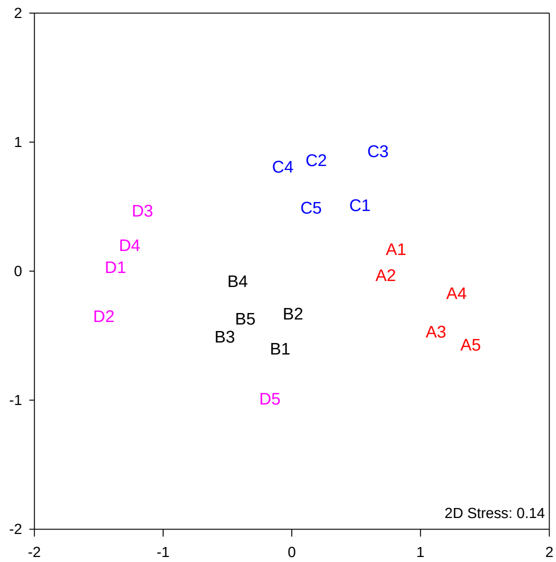


FIGURE 14: NON-METRIC MDS PLOT OF BENTHIC MACROINFAUNA DATA FROM THE PRESENT (2017) SURVEY AT MONITORING SITES AROUND THE WDC OUTFALL. DATA WERE LOG(X+1) TRANSFORMED PRIOR TO ANALYSIS AND GROUPINGS ARE BASED ON BRAY-CURTIS SIMILARITIES.

Table 4: Infaunal species that contribute most to the similarity among WDC outfall monitoring sites during the present (2017) survey. (SIMPER log(x+1) transformed data, PRIMER). Species at each site contributing to 90% of observed similarity listed.

Site	Species	Av. abund	Av. Sim	Sim/SD	Contrib %	Cum%
Site A (av. sim. 35.12%)	*Nicon aestuariensis*	1.8	26.7	1.09	76.04	76.04
	Paracorophium excavatum	0.6	8.41	0.61	23.96	100
Site B (av. sim. 47.22%)	*Boccardia* sp.	21	24.64	1.14	52.18	52.18
	Nicon aestuariensis	2.6	14.12	2.77	29.9	82.09
	Paracorophium excavatum	2	5.74	1.14	12.16	94.25
Site C (av. sim. 45.96%)	*Nicon aestuariensis*	10.6	38.22	2.25	83.15	83.15
	Potamopyrgus estuarinus	1.8	3.65	0.6	7.93	91.08
Site D (av. sim. 52.26%)	*Potamopyrgus estuarinus*	7	34.08	3.77	65.23	65.23
	Boccardia sp.	1.4	11.34	1.12	21.7	86.93
	Paphies australis	1	5.04	0.62	9.64	96.57

Given the similarity in the % fines between site A and reference site C and shared key species, i.e. *Nicon aestuariensis,* but significantly higher levels of organic matter content in sediments at site A, it is suggested that the assemblage is not majorly affected by the increased organic content. In general results suggest a spatially variable community structure at monitoring sites around the outfall that can be broadly explained by species preferences to sediment texture, and specifically fines content.

3.2.2 TEMPORAL ANALYSES

BIOLOGICAL SUMMARY INDICES

SPECIES ABUNDANCE, DIVERSITY, RICHNESS AND EVENNESS

For an analysis of temporal variability and detection of trends within sites, diversity indices were plotted (Figure 15), and subjected to Mann-Kendall trend testing.

Comparison of how the various summary indices have varied over time at reference site C provides a yardstick against which the temporal variation in summary indices at sites A and B could be assessed. Overall site A exhibited the highest degree of inter-survey variability across all indices, with multiple significant differences estimated between years among the various indices, this compared to site B where there were no significant differences detected between years in any of the indices and site C where the only significant inter-survey difference estimated was in N where the 1996 result was higher than all other results. As stress is generally considered to increase variability the increased variability at site A suggests this site is subject to higher levels of stress than sites B and C.

To assess whether or not stress has resulted in deterioration in benthic community health trend tests were conducted on within site data for the various indices. These tests revealed a number of significant trends including; significant negative (i.e. decreasing) trends at site A in both taxa richness, S, and abundance, N (Figure 15), with the mean rate of decrease for S estimated to be 5.3% per annum and for N 7.7% per annum. Similarly, at the reference site C significant negative trends were also detected for taxa richness and abundance, with mean rates of decrease estimated to be 2.5% and 22.4% per annum for S and N respectively. At site B a significant negative trend in N was also detected, with the mean decrease estimated to be 8.9% per annum. No other trends among the other indices were detected.

In general these results indicate the community at site A is subjected to increased levels of stress compared to upstream site B and reference site C, and that over time benthic community 'health' at all sites has deteriorated but is particularly pronounced at site A.

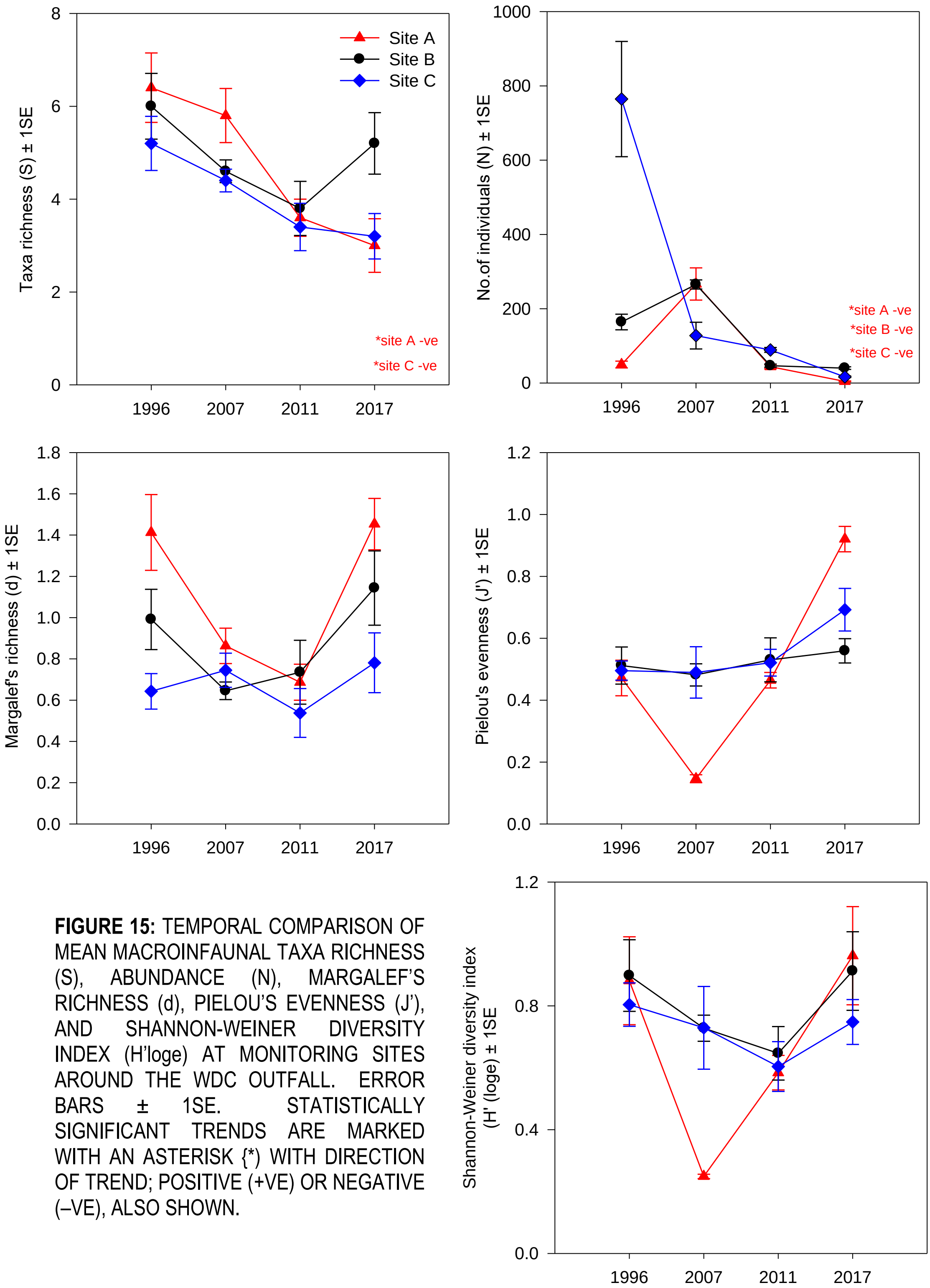


FIGURE 15: TEMPORAL COMPARISON OF MEAN MACROINFAUNAL TAXA RICHNESS (S), ABUNDANCE (N), MARGALEF'S RICHNESS (d), PIELOU'S EVENNESS (J'), AND SHANNON-WEINER DIVERSITY INDEX (H'loge) AT MONITORING SITES AROUND THE WDC OUTFALL. ERROR BARS ± 1SE. STATISTICALLY SIGNIFICANT TRENDS ARE MARKED WITH AN ASTERISK {*) WITH DIRECTION OF TREND; POSITIVE (+VE) OR NEGATIVE (–VE), ALSO SHOWN.

INFAUNAL COMMUNITY STRUCTURE

An nMDS plot illustrating the variation in community structure both temporally and spatially is shown in Figure 16.

Examining the nMDS plot it is apparent that the degree of temporal variability (i.e. distance between years, within sites) is greater than spatial variability (i.e. distance between sites within a survey year). This suggests temporal variability has a relatively stronger influence on community composition than spatial, or site, related effects. Also apparent is that temporal variability of sites A and B is relatively greater than at site C, which suggests these sites are subject to higher levels of stress than site C. The other observation of note is the reduced dispersion within sites and distance between site data during the 2007 and 2011 surveys compared to the 1996 and present (2017) surveys. This suggests that spatial effects, within and between sites in the 1996 survey, with the possible exception of site C, and 2017 surveys, was relatively more pronounced than in the 2007 and 2011 surveys.

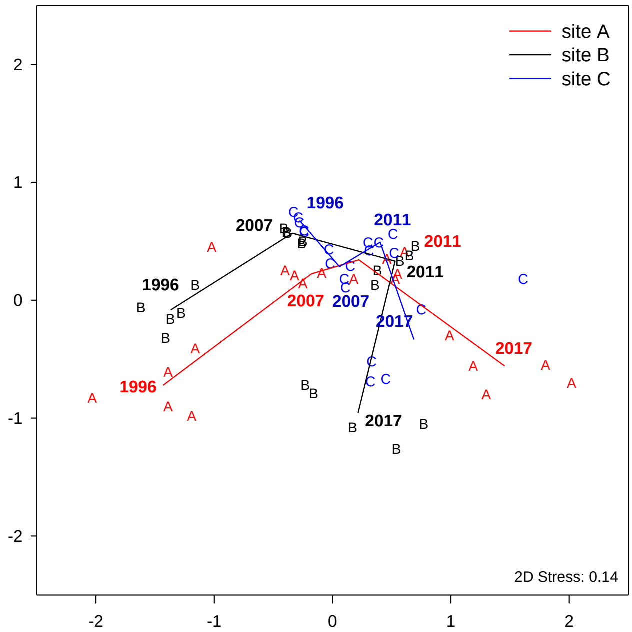


FIGURE 16: NON-METRIC MDS PLOT OF BENTHIC MACROINFAUNA DATA FROM THE 1996, 2007, 2011, AND PRESENT (2017) SURVEYS AT MONITORING SITES SURROUNDING THE WDC OUTFALL. DATA WERE TRANSFORMED LOG(X+1) PRIOR TO ANALYSIS AND GROUPINGS ARE BASED ON BRAY-CURTIS SIMILARITIES. LINES

The 2-factor PERMANOVA (examining all years, and sites) confirms that both site and year were significant factors estimating infaunal community structure (Table 5). However, evidence of a significant interaction term (year*site) indicates these differences were not consistent between either years or sites. To investigate the source of this inconsistency pair-wise *a posteriori* comparisons between years and sites (including within sites and years), were conducted. These showed that overall community structure did not differ significantly between site A and B (when all survey data were pooled) but differed significantly between all years (within sites). Between sites (within years) non-significant differences occurred between sites A and B and sites B and C (2011 survey only).

Table 5: PERMANOVA results examining the effect of site and year on infauna data. All data were ln(x+1) transformed, and analysis was based on Bray-Curtis dissimilarities. P (perm) indicates the permutational p-value, P(MC) indicates the Monte Carlo p-value. Significant results denoted by asterisk.

Source	*df*	SS	Mean Square	F-Value	P (perm)	P (MC)
Site	2	16237.2	8118.6	21.4	0.001*	0.001*
Year	3	58750.3	19583.4	51.7	0.001*	0.001*
Site x Year	6	21635.5	3605.9	9.5	0.001*	0.001*
Residual	48	18177.0	378.7			
Total	59	114800.2				

The important points to note are that there are distinctly different communities at each site with a significant level of natural variability from year to year. However, at sites A and B the communities are more similar to one another and the magnitude of temporal variation is relatively greater compared to site C. Thus it is suggested that as well as natural variability there are other factors operating at sites A and B that significantly influence community structure, e.g. wastewater discharge.

SIMPER analyses provide for a detailed examination of the underlying key species and species dynamics that account for the significant differences in assemblages between sites and over time (Table 6). From these analyses species that were key community drivers during the 1996 survey were *Scolelepis* sp. (polychaete), *Potamopyrgus estuarinus, Paracorophium excavatum, Chironomus zelandica,* and *Agalophamous macroura*. In 2007 key species were *P. excavatum, P. estuarinus, Nicon aestuariensis*, and *Scolecolepides spp.* In 2011 key species were *P. excavatum,* and *N. aestuariensis,* and as mentioned above in 2017 key species were *N. aestuariensis, P. excavatum, P. estuarinus* and *Boccardia* sp.

Table 6: List of infauna species that contribute most to the similarity among survey years (1996, 2007, 2011, 2017) at monitoring sites around the WDC outfall during the surveys. (SIMPER log(x+1) transformed data, PRIMER). Top 80% of contributing species listed.

Site	Year	Species	Av. abund	Av. Sim	Sim/SD	Contrib %	Cum%
A	1996 (av. sim. 70.99%)	*Scolelepis* sp.	39	37.02	10	52.16	52.16
		Potamopyrgus estuarinus	2.8	13.14	9.3	18.51	70.66
		Chironomus zelandica	2.2	5.91	1.07	8.33	79
		Agalophamous macroura	1.4	5	1.12	7.04	86.03
	2007 (av. sim. 78.34%)	*Paracorphium excavatum*	254.4	47.74	24.15	60.93	60.93
		Potamopyrgus estuarinus	5.4	12.49	7.26	15.94	76.87
		Nicon aestuariensis	2.2	8.97	3.2	11.46	88.33
	2011 (av. sim. 69.95 %)	*Paracorophium excavatum*	35.4	51.12	14.62	73.07	73.07
		Nicon aestuariensis	4.6	13.99	1.11	20	93.08
	2017 (av. sim. 56.85%)	*Nicon aestuariensis*	2	48.44	3.08	85.2	85.2
		Paracorophium excavatum	0.6	8.41	0.61	14.8	100
B	1996 (av. sim. 74%)	*Potamopyrgus estuarinus*	90.8	30.36	5.22	41.02	41.02
		Scolelepis sp.	59	27.51	7.7	37.18	78.2
		Paracorophium excavatum	6.6	5.16	0.97	6.97	85.17
	2007 (av. sim. 92.58%)	*Paracorphium excavatum*	202.2	38.74	40.17	41.84	41.84
		Scolecolepides sp.	42	26.71	15.86	28.85	70.69
		Potamopyrgus estuarinus	18.6	21.45	18.88	23.16	93.86
	2011 (av. sim. 78.3 %)	*Paracorophium excavatum*	36.8	48.3	10.02	61.69	61.69
		Nicon aestuariensis	7.6	28.12	7.24	35.92	97.61
	2017 (av. sim. 70.4%)	*Boccardia* sp.	28.4	41.51	8.62	58.96	58.96
		Nicon aestuariensis	3	13.29	2.68	18.88	77.85
		Paracorophium excavatum	2.4	10.23	6.55	14.54	92.38
C	1996 (av. sim. 90.05%)	*Paracorophium excavatum*	508.6	31.72	13	35.23	35.23
		Potamopyrgus estuarinus	220.6	27.54	23.1	30.59	65.81
		Scolecolepides sp.	21.2	16.45	23.09	18.27	84.08
	2007 (av. sim. 83.97%)	*Paracorophium excavatum*	105.2	36.95	7.64	44.01	44.01
		Nicon aestuariensis	14	23.93	9.34	28.5	72.51
		Potamopyrgus estuarinus	4.6	13.81	5.17	16.45	88.96
	2011 (av. sim. 83.34 %)	*Paracorophium excavatum*	72.2	48.06	11	57.67	57.67
		Nicon aestuariensis	12.2	27.91	9.75	33.49	91.16
	2017 (av. sim. 60.13%)	*Nicon aestuariensis*	11.8	50.1	8.26	83.32	83.32
		Potamopyrgus estuarinus	2.2	5.98	0.52	9.95	93.26

These results demonstrate a general shift in the polychaete assemblage, particularly at site A, from species that are typically associated with sandier sediments (e.g. *Scolelepis* sp. and *Agalophamous macroura*) to species more tolerant of fines (e.g. *N. aestuariensis* and *Scolecolepides* sp.). There has also been a marked reduction in the abundance of the amphipod *P. excavatum* at all sites over time.

3.3 SUMMARY

PRESENT SURVEY

- Dominant taxa encountered during the present survey included: the spionid polychaete worm, *Boccardia* sp., the nereid worm *Nicon aestuariensis*, the estuarine snail *Potamopyrgus estuarinus*, the amphipod *Paracorophium excavatum,* and pipi, *Paphies australis.*

- In general polychaete species dominated the abundance among all sites except for site D where molluscs were numerically dominant.

- Taxa richness (S) was low at all sites, with no significant differences estimated between sites. Similarly there were no differences among sites in Margalef's richness (d) or the Shannon-Weiner diversity index (H'). Abundance (N) was low-moderate among all sites, with site B estimated to have significantly more individuals than any other site. However the distribution of individuals or evenness (J') at sites A, C and D was significantly higher than site B.

- Given the few statistically significant differences among sites for the various summary indices, the magnitude of any outfall related effect on infaunal community characteristics at sites immediately around the outfall compared to the reference site is therefore slight.

- Infaunal community structure analyses however indicated a high level of spatial variability, with each sites assemblage significantly different to every other site.

- Given the similarity in the % fines between site A and reference site C and shared key species, i.e. *Nicon aestuariensis,* but significantly higher levels of organic matter content in sediments at site A, it is suggested that the assemblage is not majorly affected by the increased organic content.

- In general the spatially variable infaunal assemblage can be broadly explained by species preferences to sediment texture, and specifically fines content.

INTER SURVEY COMPARISON

- The highest degree of inter-survey variability across all indices and among sites was site A, suggesting this site is subject to higher levels of stress than sites B and C.

- Trend testing of abundance, and diversity indices indicated an ecologically significant decrease in taxa richness and abundance at both site A and reference site C. At site B an ecologically significant decrease in abundance was also detected.

- In general these results indicate the community at site A is subjected to increased levels of stress compared to upstream site B and reference site C, and that over time benthic community 'health' at all sites has deteriorated but is particularly pronounced at site A

- In terms of community structure, communities varied greatly both temporally and spatially though the factor 'year' (i.e. temporal variability) is the primary explanatory variable, followed by 'site' (i.e. spatial variability).

- Despite the variability, generally over time, the communities at sites A and B have been more similar to one another compared to site C and the magnitude of temporal variation is relatively greater at these sites compared to site C.

- Therefore it is suggested that as well as natural variability there are other factors at sites A and B that significantly influence community structure, e.g. wastewater discharge.

- The differences between years may be attributable to a shift in the key drivers of community composition from polychaetes typically associated with sandier sediments (e.g. *Scolelepis* sp. and *Agalophamous macroura*) to species more tolerant of fines (e.g. *N. aestuariensis* and *Scolecolepides* sp.). There has also been a marked reduction in the abundance of the amphipod *P. excavatum* at all sites over time.

5. DISCUSSION

The WDC municipal wastewater discharge via an outfall into the lower Wairoa River estuary is an existing activity subject to compliance with a number of resource consent conditions set out in resource consent CD940404W. Although the consent does not specifically require monitoring of the receiving environment, WDC have nonetheless undertaken monitoring to determine the magnitude of any effects occurring as a result of the discharge.

The approach used was an upstream/downstream comparison of physico-chemical sediment quality and biological communities and then how these sites compare to relevant national and regional guidelines, background levels, and biological indicators of health.

SEDIMENT CHARACTERISTICS

The results of previous dye dilution studies have shown that when the river mouth is unrestricted and discharge occurs on an ebb tide, wastewater exits the estuary in a relatively short timeframe (30 – 40 mins) (Barter 2007). In this situation discharge related effects may be expected at the monitoring site immediately downstream of the outfall only, i.e. site A. During periods of restriction at the mouth potentially all three monitoring sites around the outfall, i.e. sites A, B and D, could be exposed to the plume and be affected by the discharge. During the present survey, at two of the three sites surrounding the outfall (sites A and D), there was some evidence of the discharge affecting sediment characteristics, with the most pronounced effects evident at site A.

Potential negative effects on sediment characteristics from wastewater discharge include the accumulation of contaminants such as fine sediments, organic matter, trace elements and nutrients. In the present survey negative effects related to wastewater discharge were principally concerned with the deposition of organic matter, with current and past organic matter accumulation evident at site A and site D respectively. This resulted in reduced levels of sediment oxygenation at site A compared to the similarly comprised upstream reference site C. Moreover normalised levels of TVS at site A were 3.8x higher than the normalised levels at reference site C. At the levels of TVS, or organic matter, found at site A induction of persistent and significant stress on resident infauna and loss of ecological integrity is likely. While at site D it is suggested that the levels of TVS would likely induce moderate stress on infauna. These effects also appear to be compounding over time with a statistically significant and ecologically meaningful increasing trend in organic matter detected at site A.

Aside from the accumulation of organic matter at site A there was no evidence to suggest contamination by trace elements of sediments with no difference in normalised levels among site A and C and all sites well within ANZECC guidelines and comparable to Hawke's Bay estuarine and lagoonal background concentrations. Although adverse effects from these contaminants are unlikely at present there appears to have been a slight deterioration of sediment quality over time at site A, with Cd, Cr, Cu, Ni and Pb increasing over time.

In terms of nutrients, levels at all sites were within the moderately enriched range, but well within results observed for other Hawke's Bay estuaries (Madarasz-Smith 2016).

Based on these results, it is suggested that the discharge is likely to be having a persistent adverse effect on sediment quality from organic loading at site A immediately downstream from the discharge.

When assessing temporal and spatial trends in sediment contaminants it is standard practice to normalise contaminant concentrations to 100% of the mud/fine component to account for changes in sediment composition. It is possible that these results could be an artefact of this process. Direct monitoring of the fine fraction (63µm) is recommended to provide more certainty around the results obtained.

BIOLOGICAL CHARACTERISTICS

Given evidence of accumulation of organic matter at site A it may be expected that the infaunal community there responds in a manner that is different and perverse compared to other sites. Certainly the infaunal community structure during the present survey was spatially variable and significantly different among sites, with communities tending to reflect differences in sediment texture, and to some extent, level of organic matter. However, the temporal component of variability, or natural variability, was also significant and indeed had a stronger influence on community structure than that explained by site, or spatial, differences. Notwithstanding this finding the highest degree of inter-survey variability across all summary indices and among sites was site A, while the magnitude of variation in community structure, both within and between years, was relatively greater at sites immediately surrounding the outfall (particularly site A) compared to the reference site C. This suggests that despite significant year to year variation there is a discernible site effect with those sites immediately surrounding the outfall not as resilient to perturbations in the environment as reference site C.

In terms of infaunal summary indices, apart from significantly higher abundance at site B compared to all other sites, few other differences were detected among sites for any of the other indices. This suggests that the despite the differences in community structure and increased levels of stress at sites around the outfall, the magnitude of effects from organic matter accumulation on infaunal community characteristics is slight-minor. Moreover the dominance of polychaete species among all sites and the generally low numbers of filter feeding bivalve species suggests the infaunal assemblages among all sites is subjected to one or more stressors that have a greater influencing effect on community characteristics than the outfall discharge.

A comparison of biological summary indices from the present survey against other Hawke's Bay estuary sites (Table 7) does tend to support this view with scores for taxa richness and abundance in particular from the present survey in the lower half of the range, suggesting that the overall health of the Wairoa River estuary may be compromised. Further evidence of broader scale deterioration in the estuary infaunal health was provided by the detection of significant negative trends in taxa richness and abundance at site C and site A and abundance at site B.

Table 7: Summary of average abundance, taxa richness, Pielou's evenness, Margalef's richness and Shannon-Weiner diversity index scores at estuarine monitoring sites in the Hawke's Bay region from recent benthic surveys, including the present survey.

Estuary (site)	Abundance (N) ±1SE	Taxa richness (S) ±1SE	Margalef's richness (d) ±1SE	Pielou's evenness (J') ±1SE	S-W diversity index (H'loge) ±1SE
Wairoa (site A)[1]	4.3 ± 1.2	3 ± 0.6	1.45 ± 0.12	0.92 ± 0.04	0.96 ± 0.16
Wairoa (site B)[1]	40.0 ± 3.5	5.2 ± 0.7	1.14 ± 0.18	0.55 ± 0.04	0.91 ± 0.13
Wairoa (site C)[1]	16.6 ± 2.1	3.2 ± 0.5	0.78 ± 0.14	0.69 ± 0.07	0.74 ± 0.07
Wairoa (site D)[1]	10.0 ± 2.8	3.2 ± 0.4	1.12 ± 0.24	0.81 ± 0.09	0.93 ± 0.17
Ahuriri (site A)[2]	44.4 ± 7.9	9.4 ± 0.8	2.30 ± 0.16	0.84 ± 0.02	1.85 ± 0.08
Porangahau (site A)[2]	6.6 ± 2.1	3.1 ± 0.3	1.31 ± 0.14	0.88 ± 0.04	0.94 ± 0.08
Waitangi (site A)[2]	89 ± 12.7	4.8 ± 0.5	0.83 ± 0.09	0.43 ± 0.04	0.68 ± 0.09
Tukituki (site A)[2]	72.8 ± 13.4	6.2 ± 0.6	1.25 ± 0.18	0.76 ± 0.06	1.36 ± 0.12

[1]Present survey (2017), [2]Hawke's Bay Regional Council Estuarine Monitoring Programme - 2016 results.

There appears to be several factors influencing the biological characteristics of sites including both the discharge and other up catchment sources of contaminants, e.g. organic matter and fine sediments. In summary, ecological condition among all sites is poor-moderate, though evidence indicates infaunal communities around the outfall are responding negatively to wastewater discharge. The magnitude of this effect on infauna however is slight-minor, though there is also some evidence of a deterioration of infaunal characteristics of the Wairoa River estuary as a whole. In situations where the wastewater is able to exit the estuary in a timely manner the contribution of this effect to the suggested overall estuary deterioration is likely to be less than minor. At times of rivermouth restriction, and wastewater lingers in the primary basin, it is suggested that the discharge constitutes a significant adverse effect.

6. CONCLUSION

There is a persistent adverse effect on sediment quality from organic loading in the area 100m downstream of the outfall that results in an infaunal community that despite being subject to higher stress than upstream sites remains similar in terms of biological 'health' indices. The discharge therefore appears to be enriching the downstream area adjacent to the outfall in a manner in which adverse effects are minor. The contribution of this effect to deterioration of whole estuary health is therefore less than minor in circumstances where the wastewater is able to exit the estuary basin rapidly. Where wastewater is prevented from exiting the estuary basin in a timely manner then the effect from wastewater discharge likely significantly affects the receiving environment, with adverse effects considered more than minor.

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APPENDIX ONE

SAMPLING SITES

Table A1-1: Location of WDC outfall monitoring sites sampled during the present (2017) survey.

Site	Distance from outfall	Description	NZTM (NZGD2000)	
			East	North
A	100	Upstream "impact"	1982449	5667310
B	100	Downstream "impact"	1982583	5667455
C	500	Upstream "reference"	1982840	5667752
D	100	Inshore "impact"	1982432	5667454

APPENDIX TWO

SEDIMENT CORE PROFILES

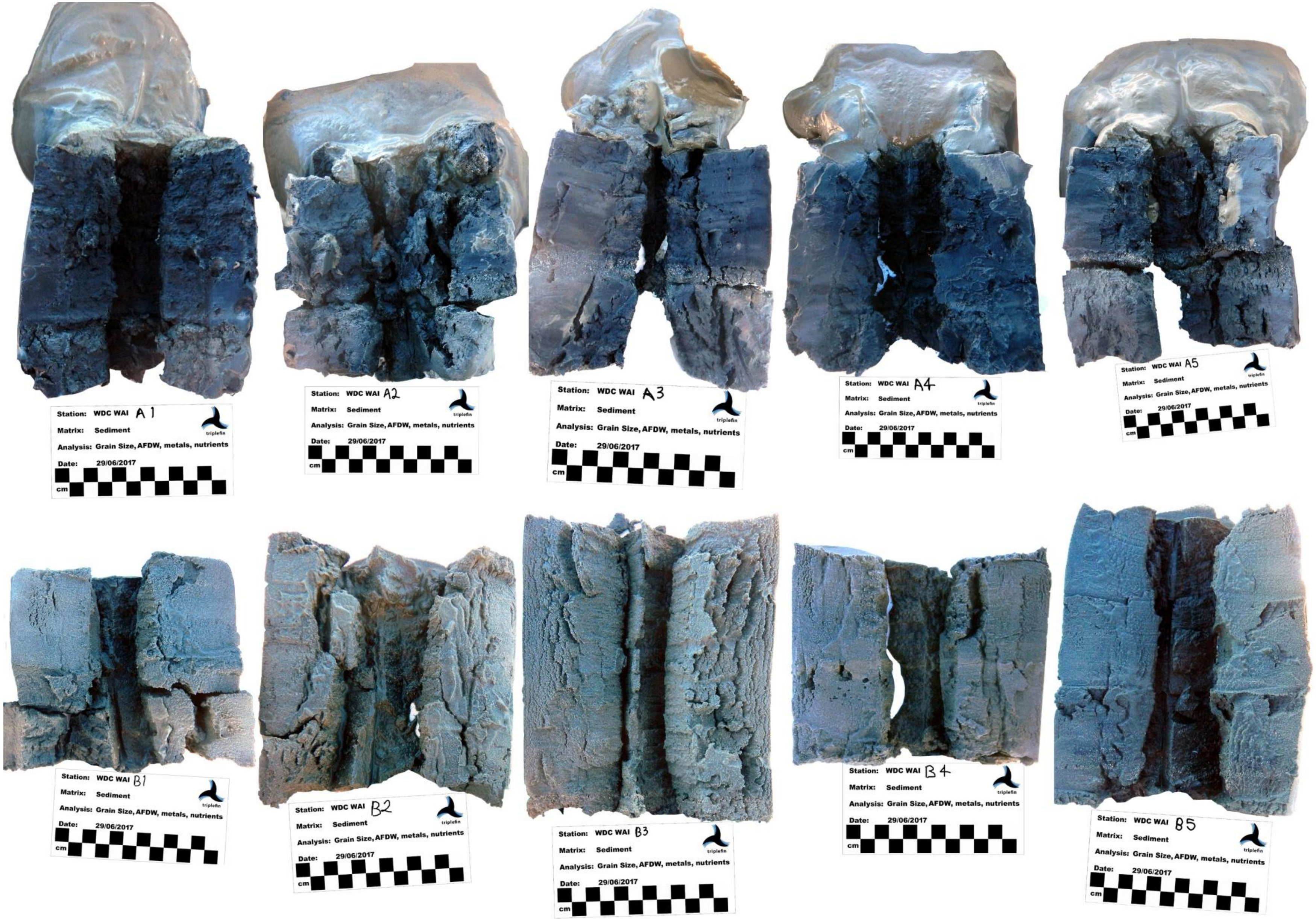

FIGURE A2-1: SEDIMENT CORE PROFILES OF THE WDC OUTFALL MONITORING SITES A (100M DOWNSTREAM) AND B (100M UPSTREAM).

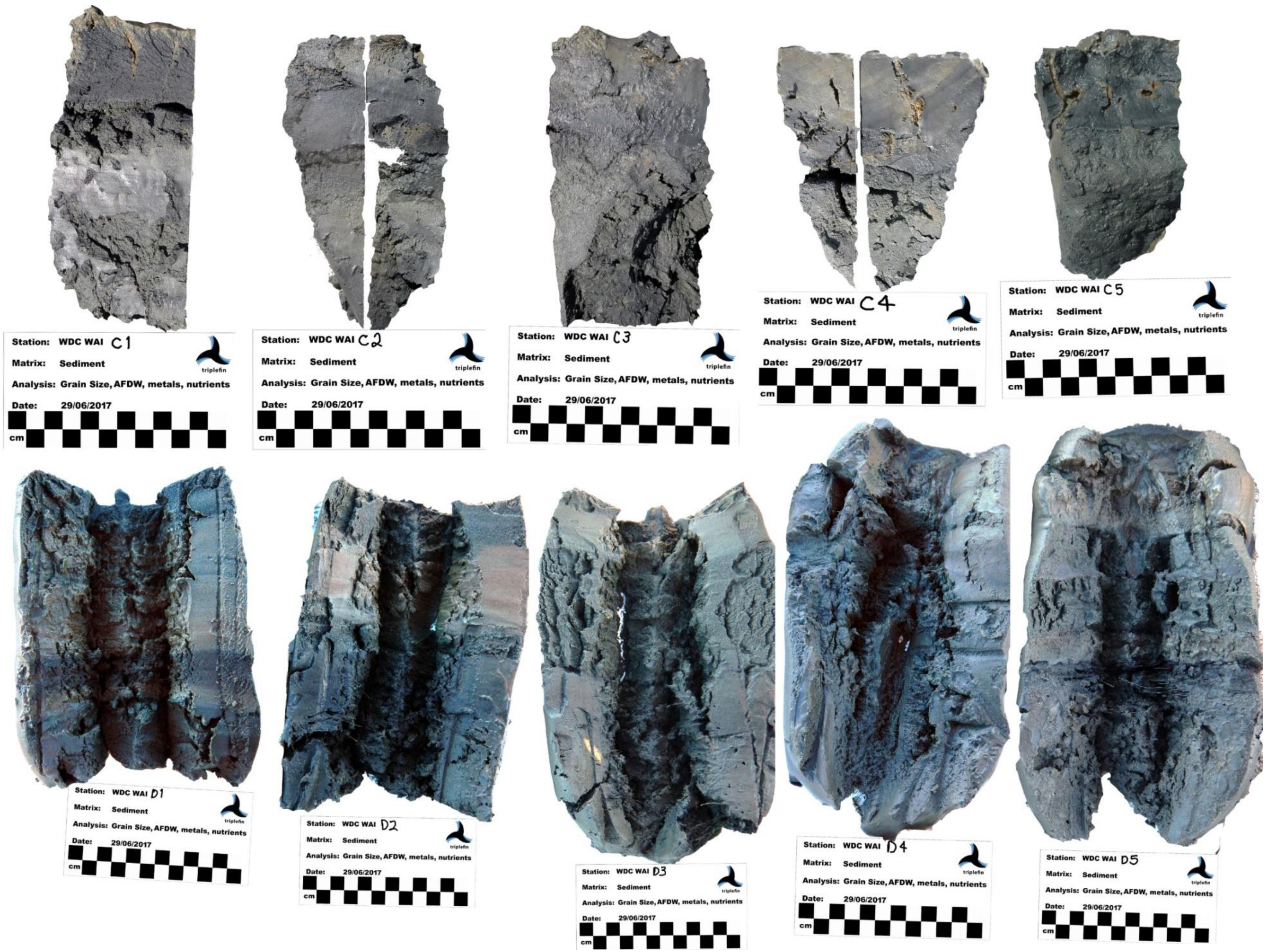

FIGURE A2-2: SEDIENT CORE PROFILES OF THE WDC OUTFALL MONITORING SITES C (500M UPSTREAM) AND D (100M INSHORE OF OUTFALL).

APPENDIX THREE

SITE APPARENT RDP LAYER DEPTHS

Table A3-1: Apparent Redox Discontinuity Potential layer depths.

Site	aRDP depth (cm)
WAI A1	1.8
WAI A2	2.3
WAI A3	2.1
WAI A4	1.6
WAI A5	1.9
WAI B1	6.4
WAI B2	6.0
WAI B3	5.9
WAI B4	4.2
WAI B5	5.5
WAI C1	2.9
WAI C2	2.6
WAI C3	3.3
WAI C4	3.2
WAI C5	1.0
WAI D1	1.8
WAI D2	2.1
WAI D3	11.5
WAI D4	6.9
WAI D5	6.3

.

APPENDIX FOUR

SEDIMENT TEXTURE, ORGANIC MATTER & TRACE ELEMENT DATA

Site	Clay and Silt (%d/w) <63µm	Very Fine Sand (%d/w) 63 - 125µm	Fine Sand (%d/w) 125 - 250µm	Medium Sand (%d/w) 250 - 500µm	Coarse Sand (%d/w) 500µm - 1mm	Very Coarse Sand (%d/w) 1 - 2mm	Gravel (%d/w) >2mm
WAI A	62.9	26	9.6	0.6	0.2	0.8	0
WAI B	9.3	21.5	68.2	0.8	0.1	0.2	0
WAI C	63.7	23.6	11.2	0.9	0.4	0.1	0.1
WAI D	37.6	21.3	39.2	1.5	0.1	0.4	0

Site	As mg/kg (dry wt)	Cd mg/kg (dry wt)	Cr mg/kg (dry wt)	Cu mg/kg (dry wt)	Hg mg/kg (dry wt)	Ni mg/kg (dry wt)	Pb mg/kg (dry wt)	Zn mg/kg (dry wt)	P mg/kg (dry wt)	N g/100g (dry wt)	TVS g/100g (dry wt)
WAI A1	3	0.098	14.8	10	0.036	16	8.8	61	480	0.11	4.8
WAI A2	3	0.076	12.9	8.5	0.034	14.8	7.7	47	450	0.09	5.2
WAI A3	3	0.061	11.5	7.4	0.034	12.6	6.7	40	380	0.06	3.6
WAI A4	4	0.069	13.1	9	0.035	14	8.1	55	430	0.09	4.8
WAI A5	3.5	<0.1	13.4	9.1	0.035	14.9	8.5	58	450	0.09	4.7
WAI B1	2.6	0.024	7.9	3.9	0.018	9.1	4.6	30	270	<0.05	1.36
WAI B2	2.7	0.027	8.3	3.7	0.021	9.1	4.6	31	300	<0.05	1.21
WAI B3	2.5	0.027	8.2	3.8	0.024	9.3	6.7	30	260	<0.05	1.31
WAI B4	2.6	0.029	8.3	4	0.022	9.3	4.5	31	270	<0.05	1.29
WAI B5	3	0.034	8.5	3.9	0.022	9.5	4.7	32	270	<0.05	1.32
WAI C1	4	0.11	13.3	9.5	0.04	15.8	9.3	49	440	0.09	1.64
WAI C2	3.3	0.077	11.9	8.3	0.045	14.6	8.5	45	410	0.08	1.41
WAI C3	2.7	0.034	8.5	4.3	0.024	9.7	6	34	290	<0.05	0.44
WAI C4	2.9	0.07	11.5	7.6	0.034	13.8	8.1	42	380	0.07	1.26
WAI C5	3	0.08	11.9	8.1	0.043	14.5	8.2	44	400	0.08	1.33
WAI D1	2	0.033	8.9	4	0.67	9	5.2	38	310	<0.05	1.43
WAI D2	3	0.046	9.4	5	0.019	11	5.7	41	340	<0.05	1.63
WAI D3	2.7	<0.1	10	5.9	0.024	11.7	6.1	44	360	<0.05	1.73
WAI D4	2	<0.1	9.6	5	0.03	10	6	40	320	<0.05	1.76
WAI D5	3	<0.1	10.3	6	0.02	12	6.4	44	360	<0.05	2.3

APPENDIX FIVE

INFAUNAL DATA

WDC OUTFALL 2017 SURVEY		WAI A					WAI B					WAI C					WAI D				
General group	**Taxa**	**1**	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Bivalvia	*Paphies australis*								2		1										
Bivalvia	*Austrovenus stutchburyi*						1		3				1				1	3		1	1
Gastropoda	*Potamopyrgus estuarinus*								2	13	2		1		6	4	3	2	9	17	
Nemertea	*Nemertea*													1							
Oligochaeta	Oligochaete		1								4						1	2			
Polychaeta: Nereididae	*Nicon aestuariensis*	3	3	1	2	1	3	6	1	2	3	11	17	10	12	9					1
Polychaeta: Spionidae	*Scolecolepides sp.*	1							1	1		1	3								
Polychaeta: Spionidae	*Boccardia spp.*						24	30	22	29	37						1	1	3	2	2
Decapoda	*Halicarcinus whitei*												2	1							
Decapoda	*Helice crassa*																				
Mysidacea	Mysid shrimp	1																			
Amphipoda	*Paracorphium excavatum*	1	1	1			1	1	6	2	2	3				1					
Trichoptera	Trichopteran						1														
	NO OF INDIVIDUALS	**6**	**5**	**2**	**2**	**1**	**30**	**37**	**37**	**47**	**49**	**15**	**24**	**12**	**18**	**14**	**6**	**8**	**12**	**20**	**4**
	NO OF TAXA	**4**	**3**	**2**	**1**	**1**	**5**	**3**	**7**	**5**	**6**	**3**	**5**	**3**	**2**	**3**	**4**	**4**	**2**	**3**	**3**

APPENDIX SIX
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