Estuarine Condition and Macro-benthic Communities in Te Tāhuna o Rangataua, Te Awanui, Tauranga Harbour.

by

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Abstract

Te Tāhuna o Rangataua is a semi-enclosed, shallow sub estuarine embayment in the southern most end of the Tauranga Harbour. In 1974, a large portion of the sub estuary was reclaimed for the siting of the Te Maunga Wastewater Treatment Plant (WWTP) oxidation ponds. When the ponds were created, important habitats were destroyed, including valuable marshlands in the upper intertidal fringe. As well as this, raw sewage was temporarily pumped into the intertidal area. Since 1974, the WWTP has been upgraded and the treatment ponds now act as a sludge settling pond, receiving treated waste water from the WWTP, which then moves through man-made wetlands and is discharged offshore. Seepages from the ponds are known to occur within Te Tāhuna o Rangataua and within these areas biological activity is limited.

To study the effects of such anthropogenic stressors on a sheltered estuarine environment, a fine scale multi–disciplinary study was undertaken. The objectives within this study set out to develop an understanding of the cumulative impacts within the estuarine area, with a focus on benthic biodiversity in response to environmental condition. The study aimed to identify key environmental parameters which may be driving change in biodiversity and to identify key benthic invertebrate species which may be driving difference in community composition within the sediments.

A gradient sampling design was adopted due to the lack of relevant control locations, with replicate samples taken at distances from the impact area. The design sampled the benthos along a gradient of distance to assess any changes in environmental condition and associated infauna, taking into account tidal influences. Sampling methods followed the Estuarine Monitoring Protocol which involved sampling sediments for environmental variables and benthic invertebrates.

A Principal Co-ordinate Analysis of benthic invertebrates found a change in community composition along a gradient of distance from the impact site. Taking into account the confounding factor of intertidal zonation, hydrodynamics and geomorphology of the area, the change in community composition observed was discussed in the light of influences of the WWTP together with other prevailing factors. It is suggested complex hydrodynamic processes are occurring within the

area. This is attributed to the various terrestrial inputs into the area, coupled with the flat and shallow nature of Te Tāhuna o Rangataua and being located at distance from the Tauranga Harbour entrance. These factors will result in longer residence times, accumulating sediments and pollutants from numerous sources. The depositional endpoints of sediments and pollutants therefore significantly influence associated benthic community structure.

Similarity percentages analysis was undertaken to assess patterns in species assemblage distribution. This highlighted key species that characterised different areas of the sub estuary. The dominant bivalve *Macomona liliana* was found to characterize composition at distances further away from the impact site and opportunistic taxa such as amphipods and nereid polychaete worms, dominated composition at distances in the upper intertidal fringe (closest to the impact site). Contaminants, nutrients and sedimentation were investigated within the area. A Principal Co-ordinate Analysis found that total phosphorus and chlorophyll-α were significant variables influencing the biophysical site character closer to the impact site. These measurements are indicative of eutrophic conditions or the sediment becoming increasingly anoxic, which suggests a degraded environment closest to the shoreline and WWT ponds.

Large bioturbators important to ecosystem functioning such as *M. liliana* and *A. stutchburyi*, were found to be displaced in the mid intertidal and high tidal area, adjacent to the WWT ponds. The absence of such organisms and apparent loss of fauna may be attributed to increased sedimentation from terrestrial sources, nutrient and organic enrichment. This study suggests that sedimentation in the upper intertidal fringe is additionally leading to a slow spread of mangroves and increased mud in the area creating an unfavourable habitat. Nutrient enrichment from wastewater seepages and freshwater inputs, would accrue microbenthic algae in high numbers, eventually adding to the detrital pool and promoting organic enrichment.

This study, the first comprehensive assessment of the habitat since the treatment facility was installed, aimed to identify the biophysical responses to pollutants and presence of the WWT ponds, along with other anthropogenic inputs which may be influencing biodiversity within the area. The study highlights the need to assess the interacting effects of all possible anthropogenic influences to understand the impact of any one factor. In addition, the study identified a change in benthic community composition along a gradient of distance from the shore and the Treatment Plant, a pattern that appears to coincide with environmental condition suggestive of eutrophic conditions within areas closest to the WWT ponds. Such conditions have been noted by previous studies to support opportunistic and pollution tolerant taxa.

Finally, the study also highlights the complex nature of estuarine environments and the need to incorporate a multidisciplinary approach to environmental assessment, taking into account hydrodynamics, biophysical components, geomorphology and ecology. Estuarine management and monitoring is constantly evolving as scientists are becoming increasingly aware of the complexities involved in understanding contaminant and pollutant effects over large spatial and temporal scales and against a backdrop of natural variability.

Acknowledgements

Much like the connectedness of an estuary to land and sea, I've come to learn the importance and value of connectivity within our lives. Without connection with others we can become stagnant, without the ebb and flow of other humans to carry us forward, towards good things, towards better understandings; we may get lost.

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Chapter 1

General Introduction

1.1 Rationale

Estuarine environments are vitally important for a multitude of reasons, many of which are not well recognised by the wider public. New Zealand is, by its very nature, a coastal entity, having an extensive shoreline of approximately 18000 km and including more than 400 estuaries (Thrush *et al.*, 2013). As a transitional zone between land and sea, the role an estuary plays in connecting environments is perhaps its most important feature. In addition, estuaries form uniquely distinct environments for an abundance of resident flora and fauna. Estuarine biota are specialised in coping with extreme biophysical fluctuations in an environment dominated by tidal flows and affected by large salinity, nutrient and sediment changes, due to input sources from both the coastal sea and land. As such, estuaries provide many environmental services involved in promoting and sustaining ecological health. They provide a buffer between land and the open ocean fulfilling functions in water quality enhancement and prevention of land erosion (Madarasz, 2006).

Habitats within estuarine environments play pivotal and often interconnected roles in the life cycles of many organisms and because of this they can be highly productive and support high levels of diversity. Examples of ecologically important habitats within estuaries include subtidal seagrass beds, marshland areas forming in the upper intertidal fringe of an area and intertidal sandflats. These are ecologically valuable habitats and provide feeding resources for birds, as well as shelter for roosting and breeding and nursery grounds and protection for valuable fish stocks (both freshwater and marine including migratory species, Thrush, *et al.*, 2013). As estuaries are by nature influenced by tidal and sedimentary regimes, the intertidal zone often makes up the largest proportion of the estuarine environment. Niche partitioning for a multitude of marine invertebrates occurs over a number of spatial (and temporal) scales adding to the highly diverse nature of these ecosystems (Madarasz, 2006).

Marine invertebrates play fundamental roles in ecosystem functioning and habitat-structuring within an estuary, with their behaviour altering physical and chemical dynamics within the sediment and at the sediment-water interface. The ecosystem services an estuary provides are intrinsically linked to its biodiversity, with biodiversity encompassing all forms of life and its functioning role within an environment (Thrush, *et al.*, 2013)

Estuarine environments are highly influenced by tidal movements and surrounding catchment character, with each sub-estuary and associated habitats being enclosed by land to differing degrees. In-coming marine tidal water is diluted by freshwater inputs from rivers and terrestrial runoff. The resultant high variability of habitats and ecosystems found from estuary to estuary can make defining an estuarine environment difficult (Thrush, *et al.*, 2013). The formation and continued changes an estuary undergoes are determined by a number of factors. The size and shape of an estuary is dependent on sediment accumulation and the nature in which an estuary in-fills is controlled by the interaction of stream/river processes, tidal inundation and waves (Hume & Swales, 2003).

Habitat formation is highly variable and dependent on the geological history of an area and sediment transport processes (Nybakken & Bertness, 2005). Freshwater entering estuaries can contain substantial amounts of suspended particles, including sediment from erosion of surrounding catchments and stream and river banks. Finer sediments are easily transported and have a huge influence on sediment condition, proportions of sand to mud and water clarity of their receiving environments (Thrush, *et al.*, 2013).

Estuaries can be a source or a sink for land and freshwater derived nutrients, sediment, contaminants and pollutants, trapping these locally or transporting them to other environments. The primary fate and accumulation of terrestrial inputs, as well as the exchange of sediments, contaminants and nutrients between an estuary and the coastal ocean is highly influenced by the hydrodynamics of an area, including freshwater flow, wind and tidal actions (Tay *et al.*, 2013).

Many types of contaminants including heavy trace metals, are able to bind to finer sediments (Thrush, *et al.*, 2013). Estuaries are depositional endpoints for the freshwater drainage network and because of this, within New Zealand, they are coastal environments considered most at risk, as accumulation of contaminants from surrounding catchment areas can reach toxic levels, degrade the environment and enter the food web (Madarasz, 2006).

Flora and fauna found within estuarine environments have adapted to survive in sometimes harsh physical and chemical conditions. Tidal cycles will leave organisms exposed for prolonged periods of time and they must contend with extreme fluctuations in salinity, temperature and oxygen availability. They must also have mechanisms to avoid desiccation and predation (Madarasz, 2006). With the added pressure of anthropogenic disturbance to an already harsh environment, change in environmental conditions may occur at rates which many organisms do not have the capability of adapting.

An estuary's placement within the environment makes it vulnerable to negative impacts from a variety of sources related to land or coastal ocean use. Human activities such as de-forestation, agricultural practices and rapid urbanisation over the years have had major influences over estuarine biodiversity and functioning (Hume & Swales, 2003). Throughout history humans have also used coastal areas to dump waste, discharging pollutants from urban, commercial and industrial communities (Carpenter *et al.*, 1998). Due to the rapid growth of human populations all over the world, there is an increase in pollutants entering waterways, degrading and changing water quality and coastal environments.

Degradation of water resources can be seen in the loss of natural systems and species associated with these systems, together with the services that these ecosystems provide (Carpenter, *et al.*, 1998). Waste water discharge from urban areas, either treated or non-treated, is a prime source of contamination and pollution, which includes excess nutrients such as nitrogen and phosphorus and faecal-borne bacteria. The problems that can arise from wastewater discharge, as well as from inputs from terrestrial and freshwater sources can include sedimentation, eutrophication and toxicity to sediments, organisms and the water column.

Given the importance of estuarine ecosystems and the threats to them as outlined above, the research presented here is aimed at understanding changes in estuarine productivity and function in response to anthropogenic stressors, with a focus on effects of land alteration, nutrient enrichment, inputs of pollutants and sedimentation to macro-benthic assemblages.

Understanding the underlying mechanisms at play within areas of ecological and cultural importance, such as estuarine environments within New Zealand, requires an integrated and whole system approach. This research will provide a baseline and support expanded work in cultural health assessments within Tauranga Harbour, which aim to restore and enhance coastal environments and their resources of importance to iwi/hapū, through a better knowledge of these ecosystems and the degradation processes that affect them.

1.2 Estuarine assessment and evaluation

A standardised Estuary Monitoring Protocol (EMP) was developed by Robertson *et al.* (2002), which assesses an estuary's physical and biological characteristics. The EMP provides an evaluation with which to prioritise estuaries that may be under threat and require further management. The EMP is designed to enable appropriate investigation of the key issues which can affect estuarine health condition. Due to the risks many New Zealand estuaries are under from contaminant impacts, the development of management techniques to assess estuaries have become a major resource management priority (Robertson, *et al.*, 2002).

An important aspect of the EMP is developing a protocol that managers can follow with relative ease and to assist in characterising estuaries within their region. The EMP encourages managers to become familiar with their estuaries, identify knowledge gaps and identify significant values within their estuaries and identify potential threats to these values. In doing this, managers may be able to prioritise estuary monitoring based on these findings (Robertson, *et al.*, 2002). The value of an estuary is considered to increase with the following characteristics;

- Increase in size, in terms of an area providing resources.
- Estuarine areas with a broader array of intertidal habitats (e.g. marshland areas) which promote intertidal biodiversity
- Estuarine areas with a broader array of subtidal habitats (e.g. seagrass beds) promoting subtidal diversity
- Increased flushing time of an estuary, described as the time for freshwater inputs to be flushed from an area. A shorter flushing time will reduce risk of contaminant accumulation

- Presence of coastal habitats acting as buffers (e.g. saltmarshes and mangroves). Areas where coastal habitats have been removed or reclaimed have lower ecological value, due to loss of feeding and nursery grounds and ability to filter contaminants and sediments.

- A presence of fish and shellfish resources.

Estuaries are also assessed based on their cultural, recreational and commercial use. Areas with higher value are those which are important for the following uses:

- An important habitat for fisheries, migratory birds with a high wetland and wildlife value

- A significant resource used by the community for food gathering, water sports etc.

- And area of high cultural value for local tangata whenua, in particular if used as a traditional food-gathering site

- Is economically important for commercial use e.g. fish or shellfish harvesting.

The risk of an estuary is based on characteristics and activities which affect an area including:

- Urban, industrial, forestry and agricultural land use of surrounding catchments, with modified catchments posing the greatest risk due to contaminant and sediment runoff

- Estuaries where margins have been reclaimed or altered, which are at higher risk of adverse effects

- Point source or diffuse wastewater discharges to an area from municipal, industrial or agricultural sources

- Risk of accidental spills or discharges to an area.

Characteristics which indicate that an estuary is impacted include:

- Excess macro or micro algal blooms

- Presence of invasive species

- Water clarity or odour problems due to effluent, decomposing algae or anaerobic sediments.

- Unsuitability to swim or wade in an area

- Faecal borne contamination or toxicity problems

- Solid waste

1.3 Thesis Structure

In order to address the objectives of this study, research is organised into the following components:

Chapter 1: Study Area and Ecological Literature Review

- Estuarine ecology is explained in a wider context, with attention placed on anthropogenic stressors which can inhibit estuarine health.
- Detailed description of the ecological and environmental background of Te Tāhuna o Rangataua, Tauranga Harbour, to gain an understanding of environmental changes that may have occurred over time. Attention is placed on historical growth of surrounding catchment areas and any anthropogenic stressors which may have contributed to ecological change up until the present.

Chapter 2: Methodology of the Ecological Survey

- Outlines the methods for data collection, following the Estuarine Monitoring Protocol developed by Robertson et al in 2002. Specific methods for sample processing and statistical methods undertaken are described.

Chapter 3: Results of the Ecological Survey

- A detailed description of preliminary and statistical results from the ecological survey
- Macrcrofaunal identification and distribution is examined
- Environmental parameters are examined, both as indicators of change in environmental condition and their association with change in community composition.

Chapter 4: Discussion Part I: Macrobenthic community structure

- Benthic invertebrates will be discussed, focusing on their roles as biological indicators in soft-sediment environments as well as examining the effects of tidal zonation to species distribution.
- Specific species response to environmental gradients will be assessed.

Chapter 5: Discussion Part II: Environmental condition

- Environmental variables will be discussed, with attention placed on the influences of hydrodynamics, freshwater and land inputs and geomorphology to environmental condition.
- Contaminants, nutrients and sedimentation are discussed, focusing on their influences to benthic community dynamics.

Chapter 6: Estuarine ecology, environmental management and concluding remarks.

- Present and future biodiversity and ecosystem functioning is discussed, with reference to species and habitat displacement within the area in responses to anthropogenic disturbance.
- The environmental condition of Te Tāhuna o Rangataua is discussed in relation to the broader context of ecosystem functioning and environmental management.

1.4 Area of Study

The estuarine area of this study is located within the Tauranga Harbour, a large harbour located on the northeast coast in the northern Bay of Plenty in the North Island of New Zealand (Park, 2011). The Tauranga Harbour is a complex system supporting a rapidly growing urban precinct with New Zealand's largest commercial port.

Tauranga Harbour is one of the largest estuaries in New Zealand (Tay*, et al.*, 2013) comprising of northern and southern basins, with large intertidal and flats separating the two (Ellis *et al.*, 2013) (Figure 1.1). Tauranga Harbour has two entrances, Bowentown and Mount Maunganui, and covers an approximate total area of 1275 km^2 (Bradley *et al.*, 2004). Much of the harbour is shallow and intertidal and is protected by a barrier island (Matakana Island) (Park, 2011). The northern basin drains via the Bowentown entrance and the southern basin drains via the Mount Maunganui entrance.

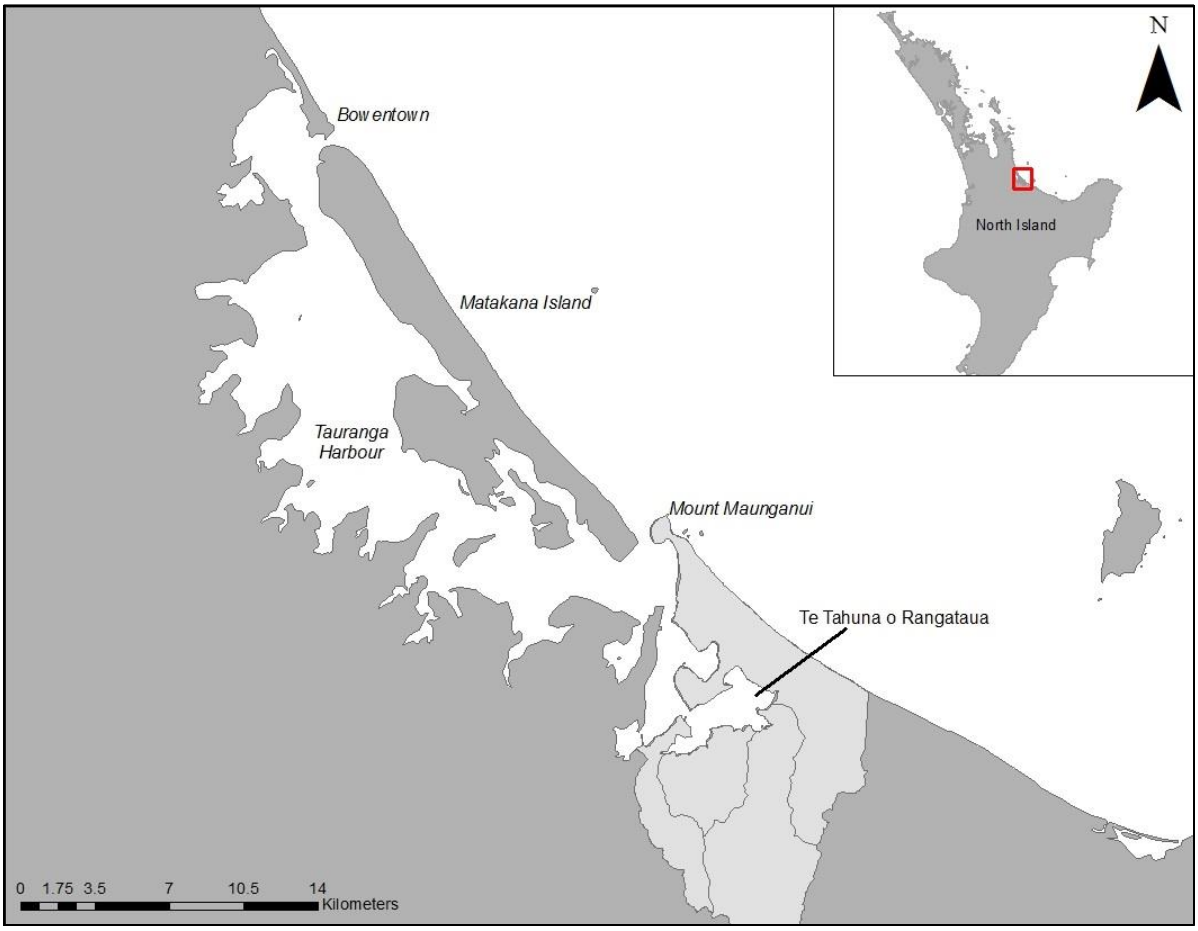

Figure 1.1: Map of Tauranga Harbour

Over many years, land surrounding the Tauranga Harbour has been continuously colonized due to the many services that the coastal area and open oceans have provided. The catchment area of Tauranga Harbour is well developed with extensive horticultural and agricultural use (Bradley, *et al.*, 2004). The city of Tauranga is located in the southern end of the harbour, which includes the Mount Maunganui area, and supports a large residential population (around 120,000 (Park, 2011)), which is increasing. The Mount Maunganui area is at the mouth of the southern entrance of the harbour and this area has been increasingly developed for port facilities. The port was established in 1873 and currently handles one of the highest cargo tonnages in New Zealand (Bradley, *et al.*, 2004). Due to the concentration of anthropogenic activities within the southern basin of the Tauranga Harbour, estuarine environments within this area receive increasing inputs of pollutants and run off from urban, commercial and industrial sources over the years (Park, 2011).

Within the Tauranga area there are two waste water treatment plants (WWTPs): Chapel Street and Te Maunga. The Te Maunga WWTP provides treatment for wastes from Mount Maunganui and Papamoa areas, whilst also receiving treated wastewater from the Chapel Street Plant.

Te Tāhuna o Rangataua is a semi-enclosed shallow and sheltered estuary located within the southern basin of Tauranga Harbour lying adjacent to the Te Maunga Wastewater Sewerage Facility and associated treatment ponds (Figure 1.2) and is the focus of this research.

1.5 Te Tāhuna o Rangataua

Te Tāhuna o Rangataua is the easternmost arm of the southern basin of Tauranga Harbour. It is a shallow, semi-enclosed and well-protected inlet and is located a significant distance from the harbour entrance (Figure 1.1). The Te Tāhuna o Rangataua area consists predominantly of intertidal sand and mud flats. The surrounding landscape is dominated by the Mangatawa and Papamoa Hills, with a few tributaries feeding into the area, including the Waitao, Kaitemako and Rocky streams. The hills are largely open farm land with some creeping urban development. The area also receives stormwater discharges from the Mangatawa Drain, flowing in from the north and draining State Highway 2 and the Papamoa Hills catchment area (Figure 1.2).

1.5.1.1 Historical Background of Te Tāhuna o Rangataua

Te Tāhuna o Rangataua and its surrounding landscape features are rich in Maori history. Nga Potiki are the main tangata whenua of Papamoa and the areas around Rangataua and they are associated with the iwi of Ngai Te Rangi. Te Tāhuna o Rangataua and the Te Maunga Waste Water treatment ponds are located within the traditional rohe of Nga Potiki, with Nga Potiki having recognised customary authority over that area (Bradley, *et al.*, 2004).

The disposal of sewage effluent into the Tauranga Harbour has been an ongoing concern for Maori, dating back to the 1920s. The issue of discharging sewage to marine environments have remained an issue of national concern to Maori, with reoccurring treaty claims being lodged with the Waitangi Tribunal regarding this activity (Bradley, *et al.*, 2004).

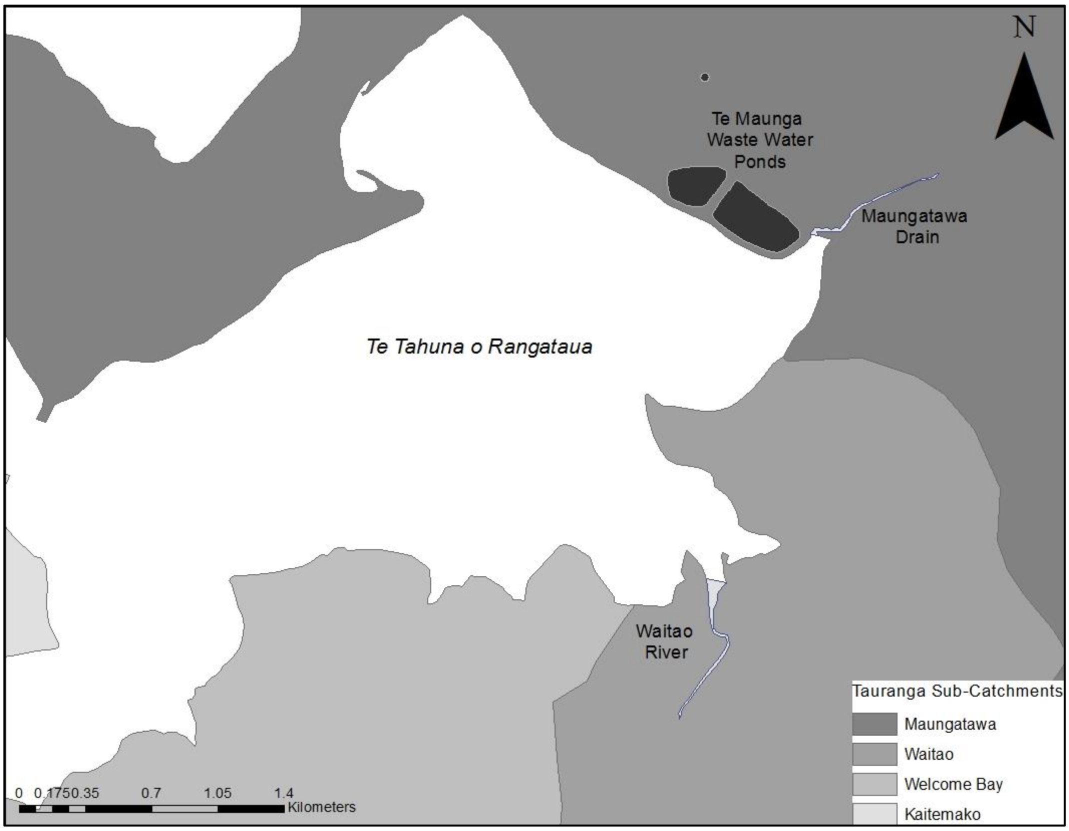

Figure 1.2: Map of Te Tāhuna o Rangataua, surrounding catchments, Waitao River and the Mangatawa Drain.

In 1969, the Mount Maunganui Borough Council (which in later years was amalgamated with the Tauranga City Council) proposed to establish a sewerage scheme which involved constructing oxidation ponds to treat effluent, within the Rangataua area. This required substantial reclamation of the area and involved effluent being temporarily discharged into Te Tāhuna o Rangataua (Stokes, 1980)

Although the Mount Maunganui Borough planned to provide a sewerage system, the proposal was met with much opposition by locals, which involved nine different groups, including Maori committees and conservation groups. Disposal of effluent into Te Tāhuna o Rangataua and the construction of the ponds raised many concerns. Potential effects included the reduced health and ecology of different environments, such as mud flats, mangroves and salt marshes, the interruption of drainage patterns and the suffocation of species and habitats from sediment and soil disturbance. Of particular concern was the proposed action for the temporary discharge of effluent for at least 5 years into the Rangataua area until completion of the Waste Water Treatment Facility, which included a discharge pipe extending 500 m out to sea in the Papamoa area (Stokes, 1980).

Issues were also raised concerning effluent seepages from the ponds into Te Tāhuna o Rangataua. There were concerns of contamination to fish and shellfish resources, as well as how far reaching those effects may extend within the harbour. There were doubts about the effectiveness of treating the wastewater within the ponds and worry over the bacterial contamination of waterways (Bradley, *et al.*, 2004)

The waste water scheme was commissioned in 1979 to collect, treat and dispose of sewage from the Mount Maunganui and Papamoa areas. It comprised two oxidation ponds covering approximately 23.5 hectares of reclaimed land at the edge of Te Tāhuna o Rangataua and involved temporarily discharging treated effluent to the harbour while the outfall pipe was constructed (Stokes, 1980). The ponds were originally constructed by building an embankment on the harbour bed and were not lined, but relied on the purported self-sealing properties of fine sediment and sewage sludge and the particle grain size of underlying sediments having low hydraulic conductivity. Since the placement of the ponds, seepages into Te Tāhuna o Rangataua from the ponds have however continuously occurred from several sites and are monitored as part of the Te Maunga Wastewater consent (Bradley, *et al.*, 2004).

In 1989 the Tauranga and Mt Maunganui sewerage systems became connected and managed together. In following years the council made changes to The Te Maunga sewerage scheme, changing the treatment of the effluent by adding aeration and clarifier treatment systems before the wastewater entered the sludge ponds. Previously, after pre-screening of the sewage, it was taken to the oxidation ponds straight away, to undergo aerobic degradation and sludge maturation and settling. Currently the ponds act as an oxidation pond, sludge settling pond/lagoon along with the surrounding land near the Mangatawa Drain being converted into manmade wetlands. The man-made wetlands were added to the plant, after consultation with Maori, to further filter the treated effluent before discharging it to sea (Bradley, *et al.*, 2004).

1.6 The physical and biological characteristics of Te Tāhuna o Rangataua.

The approximate area of Te Tāhuna o Rangataua is reported to be 6.30 km^2 which is a relatively large estuary in the southern basin of the Tauranga Harbour (Park, 2003). From the EMP's assessment criteria, an area which is greater than 25 km^2 and has a flushing time of less than three days is considered high in value.

1.6.1 Hydrodynamics

Marine water enters the sheltered embayment of Te Tāhuna o Rangataua with each tide. Wave energy is slight due to the estuary's distance from the harbour entrance and the shallow depth. An estuary, such as Te Tāhuna o Rangataua, can be a source or a sink for land and freshwater derived nutrients, sediment, contaminants and pollutants, trapping these locally or transporting them to other environments. The primary fate and exchange of inputs between an estuary and open ocean is highly influenced by the hydrodynamics of an area. Knowledge of hydrology drivers, which include circulation patterns, residence times and vertical structure, is important in understanding the health of an estuarine system which is impacted by anthropogenic stressors (Tay, *et al.*, 2013). As the estuary has only a small mouth for intertidal exchange, terrestrial run-off and freshwater inputs are influential to physical characteristics of the area.

Incoming and outgoing tidal action is not a simple act of seawater flowing in and out of an estuary. The speed and direction with which tidal currents enter an estuarine environment is influenced by many variables. These include the depth and narrowness of the entrance and the presence of physical structures such as sand banks and shoreline contours, which may alter and deflect tidal inflow and outflow. This may create small distinctive hydrodynamic features within an intertidal area, such as fronts or eddies, which in turn may promote concentrations of organisms in these prolonged or fleeting pockets of water. Hydrodynamics of an area are one of many physical features which influence levels of biodiversity within estuaries and are highly variable through space and time (Thrush, *et al.*, 2013).

A study by Tay *et al.* (2013) described hydrodynamic features of estuaries within the southern basin of the Tauranga Harbour, using a numerical model of harbour dynamics. Hydrodynamic features are said to be influenced by freshwater flow, tidal and wind-induced circulation. A 5-year model run was undertaken to study the spatial variation of tidal amplitude (the average difference between water levels at high and low tide). Tidal flow is constricted through the narrow main harbour entrance. For the Rangataua area, tidal amplitude was modelled to be between 0.64 and 0.65 m (Tay, *et al.*, 2013)

Within the study, the residence times (characterized as 'the average time that a parcel of water spends within a specified region of interest before it departs') were predicted for estuaries in the Southern Basin of the Tauranga Harbour. Residence times were found to be higher in sub-estuaries with shallow intertidal morphology and constricted entrances, such as Te Tāhuna o Rangataua. Within Te Tāhuna o Rangataua, residence times were predicted to be approximately five days longer than in the main channel of the Tauranga Harbour but this length of time could be significantly shortened with increase in wind and storm events. It was predicted that with no wind action, the average residence time within Rangataua Bay could be up to seven days and with storm wind, the average residence time was estimated to be five days. The reduction in residence time can be attributed to the influence of wind on freshwater inputs, causing substantial increases in flushing of streams. In winter, when freshwater flow is higher, residence times may be reduced even further during storm events (Tay, *et al.*, 2013).

The hydrodynamic study by Tay et al (2013) suggested that longer residence times in semi-enclosed embayments would have a great impact on how quickly sediments and contaminants will be flushed out of sub-compartments into the harbour and areas, such as Te Tāhuna o Rangataua, are susceptible to accumulation of sediments and contaminants.

1.6.2 State of the environment pre-reclamation

An environmental impact report was written by Bioresearches (Larcombe, 1974) in 1974, regarding reclamation of up to 176 acres (0.70 km^2) of land within the intertidal area of Te Tāhuna o Rangataua for the siting of oxidation ponds. The report investigated the ecological value of the area to be reclaimed and what negative impacts or losses may occur from the oxidation ponds and waste water. The reclaimed area was described as entirely intertidal apart from a drainage channel from the Mangatawa Drain. The upper intertidal fringe was low–lying, consisting of marsh and scrub protecting intertidal sand flats. Sand had also accumulated along the eastern side forming a small beach like area. Surface sediments were described as fine sands which were clean and firm (Larcombe, 1974).

1.6.2.1 Flora and fauna

The environmental impact report (Larcombe, 1974) aimed to highlight faunal and floral diversity within a range of habitats that were prominent (or considered ecologically important) in the area to be reclaimed. Twelve samples of invertebrate fauna were taken and flora was described. Sample stations for fauna were chosen taking into account ecological variation in the area. Within the area, the brackish water ecosystems appeared to thrive, beginning with the build-up of fine sediment creating a partially terrestrial environment. A band of vegetation at the upper intertidal fringe, where the ponds are now situated, once consisted of a few major species and plants such as rushes (*Leptocarpus simplex, Juncus martimus*), sedges (*Cladium* sp.) and mangroves (*Avicennia resinifera*).

These primary producers would have played important ecological roles in the area by way of buffering the shoreline, protecting from erosion and serving as biochemical filters off terrestrial runoff (Nybakken & Bertness, 2005). The marsh area trapped terrestrial derived fine sediments, preventing them from entering the harbour and adversely effecting water and sediment quality upon settlement. The habitat would have also provided nursery grounds for many species, including crustaceans and shellfish such as titiko (*Amphibola crenata*). Another ecologically important primary producer, seagrass (*Zostera* sp.), was also present in small patches at the high intertidal level.

In the Larcombe (1974) report a dominant biological feature of the area was the dense population of titiko, especially within the marshy area in which the oxidation ponds now lie. Once dominated by vegetation, the area provided favourable conditions for titiko larval settlement. Within the sediments of the marsh area, high amounts of microbial processing of organic and inorganic wastes would have occurred (Nybakken & Bertness, 2005). The accumulation of fine sediments and detritus within the area provided a rich food supply and the living vegetation would have supplied shelter and protection.

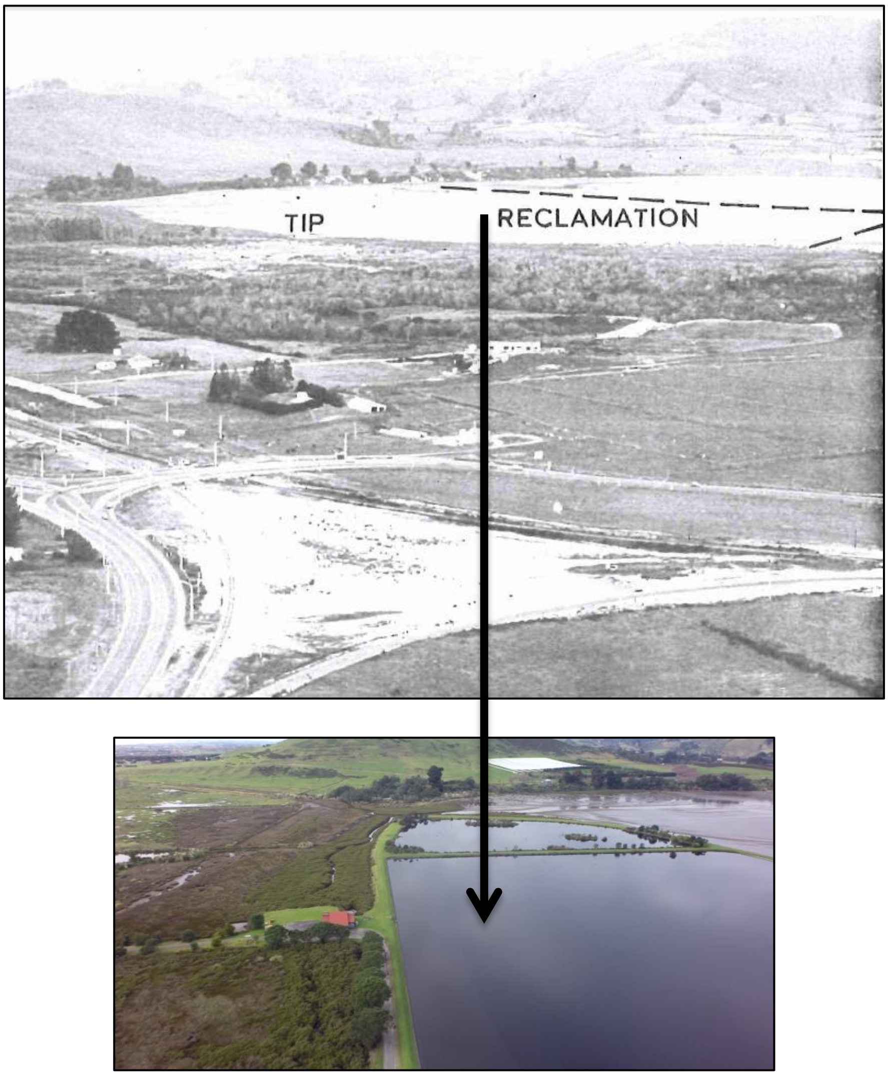

Figure 1.3: Te Tāhuna o Rangataua pre-reclamation in 1974 (from Larcombe (1974)) and in its current state, with Wastewater Treatment ponds (Picture taken 2014).

It was reported that there were a large number of juvenile flatfish in the low tidal channels and that they would have fed in the area of reclamation. Edible shellfish found within the area were titiko, cockle (*Chione stutchburyi*), trough shell (*Mactra ovata*) and wedge shell (*Macamona liliana*), although it was unknown whether cockles, wedge shell or trough shell were utilized as a food source (Larcombe, 1974).

1.6.2.2 Species abundance and distribution in 1974

Within the area to be reclaimed, 12 stations were sampled for fauna. Titiko were found to be abundant (more than 20 animals found per square metre) across the majority of the stations, with one station showing a count of between 300- 600 animals. Young titiko were found in high densities and the adult population were found to be between 4 to 8 years old in the area. The majority of titiko on the open sand flats were found to be of edible size (above 22 mm), while no cockles of edible size were sampled in the area.

The mud whelk (*Cominella glandiformis*) was found to be common (2-20 animals per square metre) across all but two stations. The estuarine limpet (*Notoacmea helmsi*) was rare across all stations, as were the mudflat topshell snails (*Zediloma subrostrata*). Horn snails (*Zeacumantus lutulentus*) were found to be abundant and common across stations, with the highest count being 400 within a square metre at station 12. Cockles (*Chione stutchburyi*) were found to be common (10-100 animals per square metre) at six stations, occasional (1-10 animals per square metre) at one station and rare (less than one animal per square metre) at five stations. Wedge shells (*Macamona liliana*) were found to be abundant (more than 20 animals per square metre) at two stations, common (5-20 animals per square metre) at six stations and rare at four stations. Trough shells (*Mactra ovata*) were found to be common (1-5 animals per square metre) at seven stations and rare at five stations. Of the crustaceans found, mud crabs (*Helice crassa*) were found to be common (5-20 animals per square metre) at three stations, occasional (1-5) at one station and rare at six stations. Two stations showed a high abundance of mud crabs, with the highest count being 640 per square metre. The stalk eyed crab (*Hemiplex hirtipes*) was found to be abundant at one station, common at three stations, occasional at three stations and rare at five stations.

Polychaete worms were also counted. *Lumbrinereis* sp. was found to be abundant (more than 100 animals per square metre) at seven stations, with the highest count being 500+. *Lumbrinereis* sp. was found to be common (10-100) at one station and rare at four stations. A Nereididae polychaete (*Nicon aestuariensis)* was found to be common at six stations and rare at six stations. A few other polychaete worms were found, including *Prionospio* sp. and *Scolelepsis* sp.

1.6.2.3 Ecological value of the area

In 1974, before reclamation had occurred, the ecological investigation summarised that the area considered for placement of oxidation ponds was healthy and thriving both ecologically and biologically (Larcombe, 1974). It was expected that with time, accumulation of fine sediments in the upper intertidal fringe would drive a slow spread of the marshy area, adding ecological value to the region. In terms of pollution, the refuse tip located near the marshland was introducing foreign material to the tidal area and a small amount of nutrient enrichment was occurring due to storm water entering the harbour from the Mangatawa Drain.

Overall however, the intertidal area was considered "exceptionally clean", with minimal polluting influence. Three types of habitat (salt marsh, mangrove marsh and open sand flats) found in the area proposed for reclamation exhibited high faunal and floral diversity and were considered ecologically stable. Biomass was high, with a large proportion of biological activity occurring in the marsh zone and a dense population of titiko on the sand flats.

The marsh area provided many environmental services, being a highly productive ecosystem. The deposition of organic detritus supplied food to deposit and filter feeders both locally and with tidal transport, to the Welcome Bay area in general, highlighting the importance of the marsh land for many food chains.

The investigation concluded that reclamation would "destroy the present ecology", with the ecological value of the area being particularly high when compared with other embayments within the Tauranga Harbour and other northern New Zealand harbours. The report highlighted that impacts to ecology would be influenced by several factors including interference with tidal action and currents. Change in tidal movement would have an effect on rates of siltation and larval and food transport. Leaching from the ponds of toxic material would also have negative effects, though the report states if construction of the ponds was done correctly, with "impervious walls and stone capping material on the intertidal zone" then pollution from the ponds would not be an issue, although it is now known that seepages from the ponds do occur into the bay and have done so for many years.

This small qualitative study gives only a brief overview of the ecology of the area and would not be able to fully assess the broad scale effects of the reclamation and siting of the oxidation ponds to ecosystem functioning. The cumulative effects that have occurred over the years could be far reaching as a result of both removing a large proportion of an important habitat and due to the pollution from the oxidation ponds into the surrounding area.

1.6.3 Treated Wastewater Seepages to Te Tāhuna o Rangataua

As a condition of the Tauranga City Council (TCC) Te Maunga Waste Water consent, annual assessment of treated waste water seepages into the bay is conducted (Gibbons-Davies, 2013). The condition states:

"In the month of February each year, at or near low tide, the permit holder shall undertake an inspection of the intertidal sand flats n a band extending 100 m seaward of the ponds. The aim of the inspection is to identify any indicator organisms or unusual biological features that could indicate the presence of leakage from the ponds."

The most recent TCC consent report stated that seepages can be identified by a green, grey black discolouration and an increased wetness in low tide, which has become known from previous surveys (Gibbons-Davies, 2013). At identified seepage sites, field measurements are made which estimate the flow rate of the seepage and measure water for salinity, temperature and dissolved oxygen. Water samples are taken to be analysed for faecal coliform bacteria, amonia-N, nitrate-N and dissolved reactive phosphorus. One seepage, labelled W6 (*see* Figure 1.4 map of seepages), from the old oxidation ponds has been present since monitoring began in 1987 and was identified again during the 2013 assessment.

The seepage is located in the adjacent tidal area to the pond in the western direction and the report states that there is clear visible evidence of the seepage area at low tide, identified by grey/green discolouration of the sediment, wetness and absence of fauna, including burrowing mud crabs. A sample of the area determined the seepage to be 77% freshwater and faecal coliforms were slightly elevated at 130 MPN/100 mls. Figure 1.5 shows the quantity of seepages from 2009-2013 within the seepage site W6. The area in which the seepage is visibly evident has increased since 2012, which was estimated to affect an area of 54m^2. In 2011 there was a marked increase in area affected but this was then found to decrease in 2012. In 2013, the total area that was affected from a visual assessment was 63m^2. Within this area, abundance and diversity of organisms is reported to be limited (Gibbons-Davies, 2013).

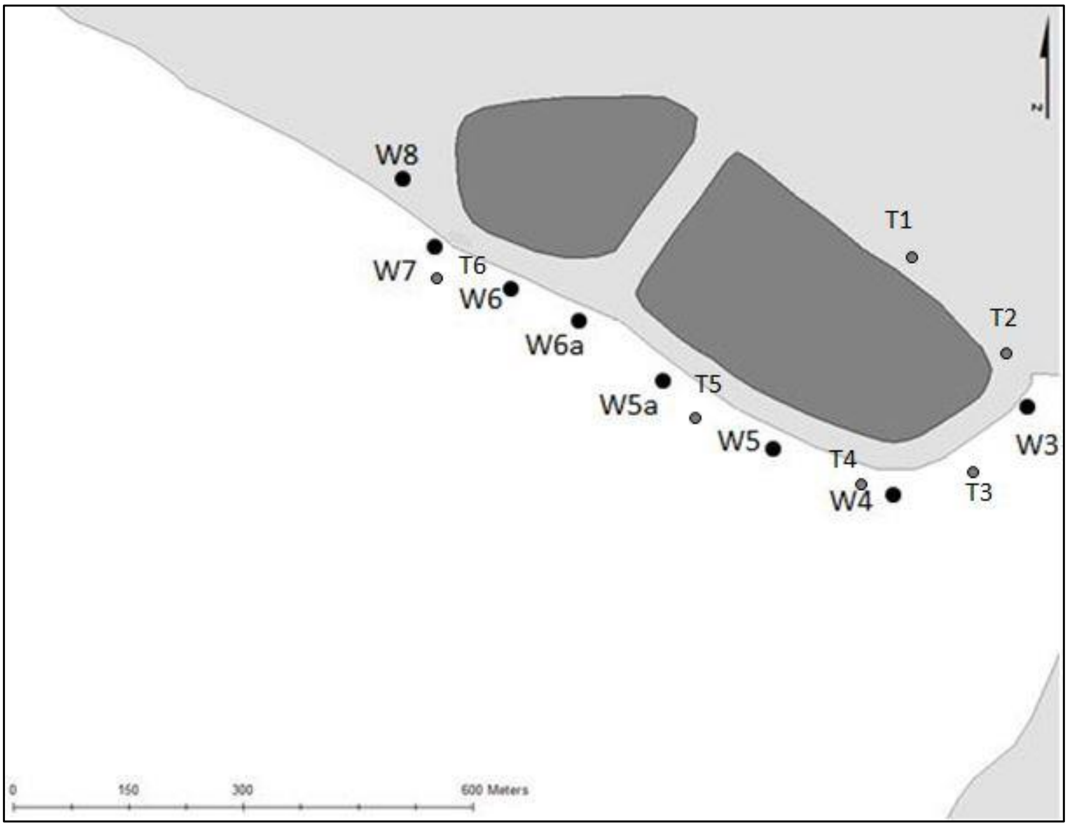

Figure 1.4: Location sites of titiko sampling (T1 – T 6) and known seepages (W3 – W8) from the ponds into Te Tāhuna o Rangataua since 1987 (Gibbons-Davies, 2013)

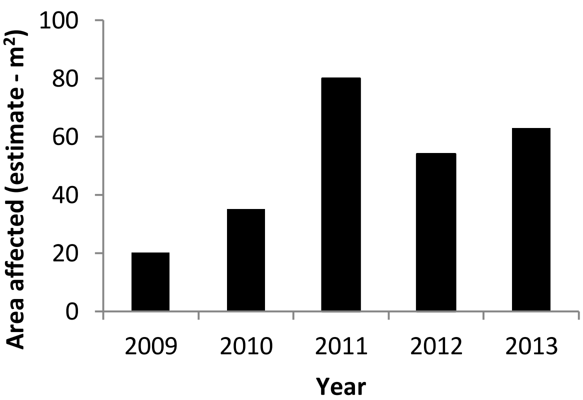

Figure 1.5: Affected estuarine area within Te Tāhuna o Rangataua from seepage site W6, from 2009 to 2013. Data taken from TCC monitoring report (Gibbons-Davies 2013).

As part of the seepages consent conditions, titiko (*Amphibola crenata*) numbers must be surveyed annually and have been surveyed at the same sampling locations since 1987. Six sites have been sampled since 1978, labelled T1, T2, T3, T4, T5 and T6. These sites are placed within the same vicinity in which the seepages are present (Figure 1.4). Sites T1 and T2 were located on the landward side of the oxidation ponds and since 1987 have become colonised with mangroves. Titiko have disappeared from these sites since 1996 and are no longer sampled. The remaining four sites are located at high tide marks seaward from the ponds. Titiko were found to be the dominant shellfish in the four remaining sites, though their distribution is patchy. Compared with surveys in previous years, the report states that titiko have showed a trend towards increasing at site T5 since 2006, but have declined in abundance at sites T3 and T4 since 2006, and have showed a trend towards decreasing at site T6 since 2009.

Site T6 is the closest site to the W6 seepage area and is shown to have the lowest density count of all remaining sites from the 2013 survey. Over the period of the surveys an overall loss of titiko abundance in the area is indicated. The report by Gibbons-Davies (2013) concludes there is no visual evidence of the seepage effect in water quality and fauna outside of the area, though testing of sediment and shellfish is not undertaken farther afield from the seepage site. Although the report states that the seepages are not likely to have a significant effect on overall estuarine water quality, the time series data indicates that there is a high chance of the affected area increasing, further limiting biological activity and ecosystem functioning in future years.

1.6.4 Catchment land use and sedimentation

In a study of sedimentation within the Tauranga Harbour by Hume et al (2009), the primary fate of fine sediment, characterized as the "primary transport pathways for terrigenous fine sediment that result in deposition" was mapped for each sub-catchment within the Tauranga Harbour. The information obtained in this study was used to identify catchment areas of highest priority for management purposes in relation to catchment run-off, by linking "sub-estuary effects" to "sub-catchment causes". The study looked at appropriate measures that could be taken to reduce sedimentation within a catchment and/or estuary in the future. The findings within this study are based on model predictions by using catchment, hydrodynamic and sediment models, analysing sediment yields, harbour bed sediment mapping and measurements of tides, waves and sedimentation in the harbour.

The principle source of sediment deposition to the Te Tāhuna o Rangataua was found to be from the Waitao sub-catchment (*see* Figure 1.2 for catchment areas), while secondary sources to the area are from the Kaitemako sub-catchment and the Papamoa sub-catchment (in which the Te Maunga Wastewater treatment plant is located). Te Tāhuna o Rangataua and the Welcome Bay estuary (a semi-enclosed embayment which extends out toward the south-west from Rangataua) share common sediment sources. Sedimentation rates are found to be elevated in areas such as Welcome Bay and Te Tāhuna o Rangataua, where sediment can be trapped along the fringes of larger embayments and in sheltered areas at river mouths (Hume *et al.*, 2009), such as the area in which the Waitao Stream enters Rangataua Bay. Dispersal of sediments to the ocean is greatly reduced in the upper northern reaches of Te Tāhuna o Rangataua, which is sheltered and acts as a sediment trap, with mangrove coverage which is estimated to increase with time. The central area of Te Tāhuna o Rangataua is considered to be well flushed, with fine particles being transported away. The Waitao sub-catchment area is approximately 43 km^2 and has a steep upper catchment in which an active quarry occurs.

Table 1.1: The area, land use breakdown and yearly sediment load of sub-catchments directly surrounding Rangataua Bay. Sediment load is an overall value from every sub-catchment (inclusive of sub-catchments other than Waitao and Papamoa) which has been identified as a fine sediment source to the area. Catchment land use proportions are from 2001. Data from Hume et al (2009).

	Subcatchment	
	Papamoa	Waitao
Area (km^2)	11.8	43.3
Proportion of fine sediment run-off that eventuates in Rangataua Bay from subcatchment	82%	17%
Land use Breakdown (%)		
Bush, scrub and native forest	1.3	35.9
Exotic forest	1	17.5
Urban and roads	26.3	2
Urban earthworks	1.4	0.1
Orchard and cropland	10.3	4
Pasture	55.6	38.4
Other	4.1	2.1
Total sediment load to Rangataua Bay (tonnes/yearly)	11,241	

The yearly sediment load for Te Tāhuna o Rangataua comes from several sub-catchment sources including Waitao, Waimapu, Papamoa, Kaitemako, Wairoa and Kopurererua (Table 1.1). Waitao, Waimapu and Papamoa sub-catchments supply the highest proportions in percentage of fine sediments deposited in the area (63%, 23.5% and 6% respectively). Of the total proportions of fine-sediment run-off from the Papamoa sub-catchment, 82% ends up in the upper fringe of Te Tāhuna o Rangataua alone. Of the total proportions of sediment run-off from the Waitao sub-catchment, 67% is lost to the ocean, 17% eventuates in the central area of Rangataua and 10% eventuates in the upper intertidal fringe.

Between 2001 and 2003, a survey by Environment Bay of Plenty (EBoP, now Bay of Plenty Regional Council, BoPRC) assessed sediment accumulation and contamination of sub-estuaries within the Tauranga Harbour (Park, 2003). Rate of sedimentation is related to catchment size, type and use. Contamination by different compounds or metals that accumulate in the sediment is dependent on land use of the area (Park, 2003). At each site within the study, a 5 m radius circle was marked out and sampled, to assess particle size, nutrients and contaminants. Nutrients, pesticides and herbicides are commonly attributed to agricultural use and as a large portion of the Rangataua catchment area is pasture, these types of contaminants are likely to be received into the bay.

Approximately half of the sediment samples collected for the EBoP 2003 survey were measured for total nitrogen, phosphorus and organic carbon. Across all sites in the Tauranga Harbour, total nitrogen values ranged from 0.025 to 0.22 g/100 g with a median of 0.06 and total phosphorus values ranged from 54 to563 mg/kg with a median of 157. Of the sites analysed within Te Tāhuna o Rangataua, the results were variable, with the study showing low and average values for both nitrogen and phosphorus, as well as showing a few high values for nitrogen.

Sediment results are compared with the Australian and New Zealand Environment and Conservation Council (ANZECC) 2000 guidelines (the interim sediment quality guidelines). The ANZECC values are used to guide decisions regarding conservation and management options. The low value is a level at which sub-lethal effects may occur to the most sensitive species and the high value is a trigger level to which remedial actions should occur due to the potential for toxicity to species and the environment (Park, 2009).

Metals and other organic contaminants were found in two sampling sites within Te Tāhuna o Rangataua, sites 5 and 26, which were both located in the upper reaches of Te Tāhuna o Rangataua (Figure 1.6). Site 5 and site 26 both showed traces of metals, site 5 showed pesticide contamination and site 26 found pesticides, TPH, Polycyclic aromatic hydrocarbons (PAH's) and Polychlorinated biphenyls. Across all sites within the Tauranga Harbour, Site 26 in Te Tāhuna o Rangataua was one of two sites with the highest extractable metals content in mud fraction of the sediment for copper, lead and zinc. Of these contaminants however, none exceeded ANZECC 2000 interim sediment quality guideline limits. Park (2003) states that there is potential for a large range of pesticide compounds to be found within sediment samples but in the case of this survey, if any sites were undergoing any adverse toxicological effects then they should have been detected.

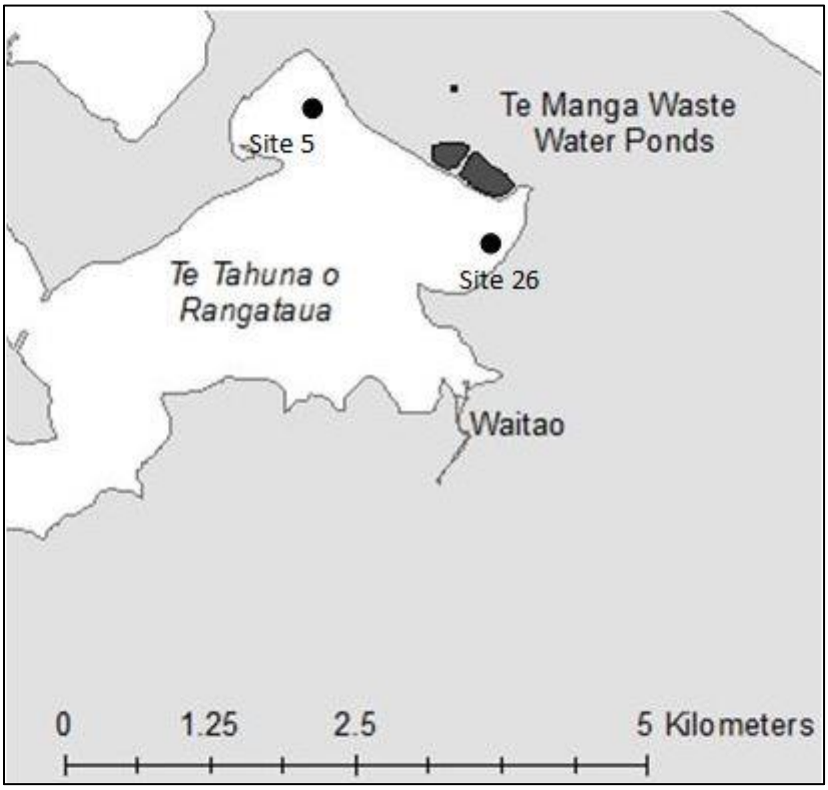

Figure 1.6. Site location of contaminant sampling sites of Te Tāhuna o Rangataua within the Tauranga Harbour sediment survey (Park, 2003).

Once contaminants enter an ecosystem they have the potential to accumulate to toxic levels. The chemical structure and bioavailability of toxins within an environment can lead to a multitude of negative effects and in some cases they may disperse large distances from the source of contamination (Park, 2003). Sedimentation may also have negative impacts to species abundance and diversity in an area. Deposition of fine silts may create a muddier and shallow environment, alter the physico-chemical nature of the sediment and water column and lead to loss of habitats for important species, whilst increasing unproductive mangrove growth.

1.6.5 Freshwater inputs and terrestrial run-off

Within the area of Te Tāhuna o Rangataua, terrestrial sources of nutrients in the upper intertidal area (in addition to the oxidation ponds) include:

- Agricultural, urban and road run-off via the Mangatawa Drain and all other tributaries entering the bay
- Agricultural run-off from the intensive agricultural area at Matapihi
- The now closed Te Maunga Tip adjacent to the ponds

In 2004, the Rangataua area was described as having moderately elevated nutrient levels (Bradley, *et al.*, 2004).

Freshwater inflow is an important key component in estuarine water quality and environmental condition (Tay, *et al.*, 2013). Some of the many freshwater tributaries that are associated with Te Tāhuna o Rangataua include the Rocky Stream, Kaitemako Stream and the Waitao Stream. A water quality survey prepared by Scholes and McIntosh (2009) for BOPRC, monitored water quality during the period 2001-2008 at 12 sites, including at both Rocky Stream and Waitao Stream.

1.6.5.1 Rocky Stream

The Rocky Stream is one of the few tributaries feeding the sub-estuary within the upper intertidal area, which begins in the Papamoa Hills, flows a few kilometres along State Highway 2 where it then turns west and flows into the eastern extreme of Te Tāhuna o Rangataua. The monitoring site for Rocky Stream was at the Mangatawa Bridge, which is located upstream from the Te Maunga Waste Water Treatment Plant. Water quality is influenced by catchment land use, which comprises of forest and pasture with some horticulture.

Suspended solids (g/m^3), turbidity, total nitrogen and total phosphorus were all found to decrease each year, though TN concentration (2.3 g/m^3) was found to be higher in this stream compared with others in the Tauranga area, which was attributed to high plant biomass and slow stream flow. Dissolved oxygen was found to decrease each year by 1.86% per year and the DO concentrations were found to be very low, especially compared with other streams. Turbidity was found to be high at the Rocky Stream site and hence water clarity found to be low. It was found that on average, *Escherichia coli* (cfu/100 mL), with a mean count of 690 during the period 2001-2008, increased by 1.97% a year, Enterococci (cfu/100 mL), with a mean count of 396, increased by 0.33% a year and faecal coliforms (cfu/100 mL), with a mean count of 740, increased by 1.25% a year. Rocky Stream was found to have the second highest value of total phosphorous of all streams monitored.

Scholes and McIntosh (2009) state that streams with higher TP levels tended to have a large proportion of phosphorous in particulate form which was due to suspended solids entering the streams. Urban and road construction around the Rocky Stream over the years would have influenced suspended solids and turbidity, with turbidity levels over the ANZECC trigger value for lowland rivers with slightly disturbed ecosystems.

1.6.5.2 Waitao Stream

The Waitao Stream is the major stream system influencing Te Tāhuna o Rangataua, flowing north into the area (Figure 1.2), with some small tributaries flowing through Papamoa before coming together and entering the bay. Scholes and McIntyre (2009) found that there was a decrease of nutrients within the stream over time and suggested that this was due to better rural practices, such as better effluent disposal and management and an increase in lifestyle properties. Dissolved oxygen was found to increase at the Waitao monitoring site. Suspended solids, turbidity and colour were all found to increase as well and a correlation between colour and indicator bacteria suggested that stock were responsible for the colour increase.

The Waitao Stream was among the few streams with the highest mean *E. coli* levels (730 cfu/100 mL increasing at 0.52% per year) of the streams in Tauranga, with *E. coli* levels found to exceed water quality guidelines (> a median of 126/100 ml (ANZECC, 2000)). Enterococci (cfu/100 mL), which increased by 0.92% per year, was found to have a mean count of 282 and faecal coliforms (cfu/100 mL), which increased by 0.21% a year, was found to have a mean count of 835 between 2001 and 2008 (Scholes & McIntosh, 2009). The study suggested that the Waitao Stream should be monitored at an increased frequency. Fine sediments, nutrients and contaminants discharged from both the Waitao and Rocky Stream catchments will eventuate in Te Tāhuna o Rangataua.

1.6.5.3 The Mangatawa Drain

The Mangatawa Drain was built in the 1950s, to divert storm water flow that previously flowed north into farm land that is now an urban area of Papamoa. The drain is approximately 2500 m long and is a vegetated, earth-lined channel that flows northwest along state highway two, veers southwest past Mangatawa Lane and discharges into Te Tāhuna o Rangataua, draining State Highway 2 and the Papamoa Hills catchment area. The channel is shaded by steep banks with minimal riparian vegetation, consisting of grasses, weeds and an occasional tree. Pre-European occupation of the area in which the drain now lies was suggested to be extensive wetland and an important habitat for birds and eels (Seabourne, 2009).

The Papamoa Hills is the major catchment contributing to discharge from the Mangatawa Drain. The southern area of the sub-catchment flows directly into the drain, discharging into Te Tāhuna o Rangataua without undergoing any form of treatment. Storm-water received from other areas flows through wetland treatment ponds (which are located near the waste-water treatment ponds and Rangataua shoreline) before being discharged to Te Tāhuna o Rangataua. (Seabourne, 2009).

A joint application and Assessment of Environmental Effects (AEE) was submitted by NZ Transport Agency and Tauranga City Council in 2008 which involved reconstruction works regarding the Mangatawa Drain (Seabourne, 2009). As part of the AEE, Bioresearches conducted an ecological survey in 2008 of some habitats within Te Tāhuna o Rangataua. The bay receives storm water from the Mangatawa Drain and a storm water treatment pond located between state highway two and the Te Tāhuna o Rangataua shoreline.

From the ecological report, it was found that sediments near the mouth of the drain were relatively course sands and a small proportion of silt and clay. In the low intertidal area of the Bay, there is a population of mangroves, with the pneumatophores creating a habitat for barnacles (*Chaemosipho columna*). Other invertebrates found living in the course sands close to the outfall included a small population of mud crabs (*Helice crassa*), occasional titiko (*Amphibola crenata*), turret shell (*Zeacumantus lutlentus*), topshell (*Diloma substrata*) and mud whelks (*Cominella sp*.). Estuarine worms and amphipods were also detected due to burrowing holes, though it was found that no edible shellfish such as pipis or cockles were present in the area. It would appear from the report that there is limited macrofaunal life in the area closest to the storm water drains, though the report only relays qualitative results for the distribution of species found.

Sediment quality samples were taken from four sites in the area, analysing metallic, organic and nutrient concentrations. Arsenic, cadmium, chromium, copper, lead, nickel and zinc were all found in very low concentrations, well below the Australian and New Zealand Environment and Conservation Council (ANZECC, 2000) sediment quality guidelines.

Total nitrogen and total phosphorus concentrations were analysed as a reflection of nutrient levels. Total nitrogen was found to vary between the four sites, with an elevated concentration in site 2, which was located in the main channel as the drain discharged into the bay. Total phosphorus concentrations were found to decrease on a gradient of distance from the mouth of Mangatawa Drain.

The results suggested that nutrient levels may be slightly elevated in the Bay due to the storm water discharge; however they do not appear to show serious nutrient input to the upper reaches of Tauranga Harbour. From the ecological survey included in the AEE by Seabourne (2009), it was concluded that concentrations of metallic and organic contaminants from the Mangatawa Drain and the surrounding catchments are not entering Te Tāhuna o Rangataua at significant levels. Although slightly elevated levels of nutrients were found, the report concluded that these nutrient inputs were not having significant adverse effects to the ecology of Te Tāhuna o Rangataua.

1.6.6 State of the environment (Ellis, *et al.*, 2013)

A broad scale survey of the Tauranga Harbour was conducted between December 2011 and January 2012 (Ellis, et al., 2013) of which three sites were located in the survey area of the current study within Te Tāhuna o Rangataua (Figure 1.7). Sediment properties and fauna were collected and analysed to assess the health of the area. The muddiness of the sediments was found to be moderate (10-20% silt and clay (<63µm)) (Hume, *et al.*, 2009) for two sites (72 and 73), organic content (AFDW) was found to be less than 2.5%, TN ranged between 180 and 640 mg/kg and TP ranged from 190-93 mg/kg, with the lowest values for nutrients being found closest the upper intertidal fringe and wastewater treatment ponds. Chl-α was found to be in the range of 9000-10,000 µg/kg, which was moderate to slightly elevated compared to other estuaries within the Harbour. From the faunal data collection, it was found that two of the sites (73 and 74) had a low number species (10-20, within a site area of 100 m^2), while site 72 had a higher species count.

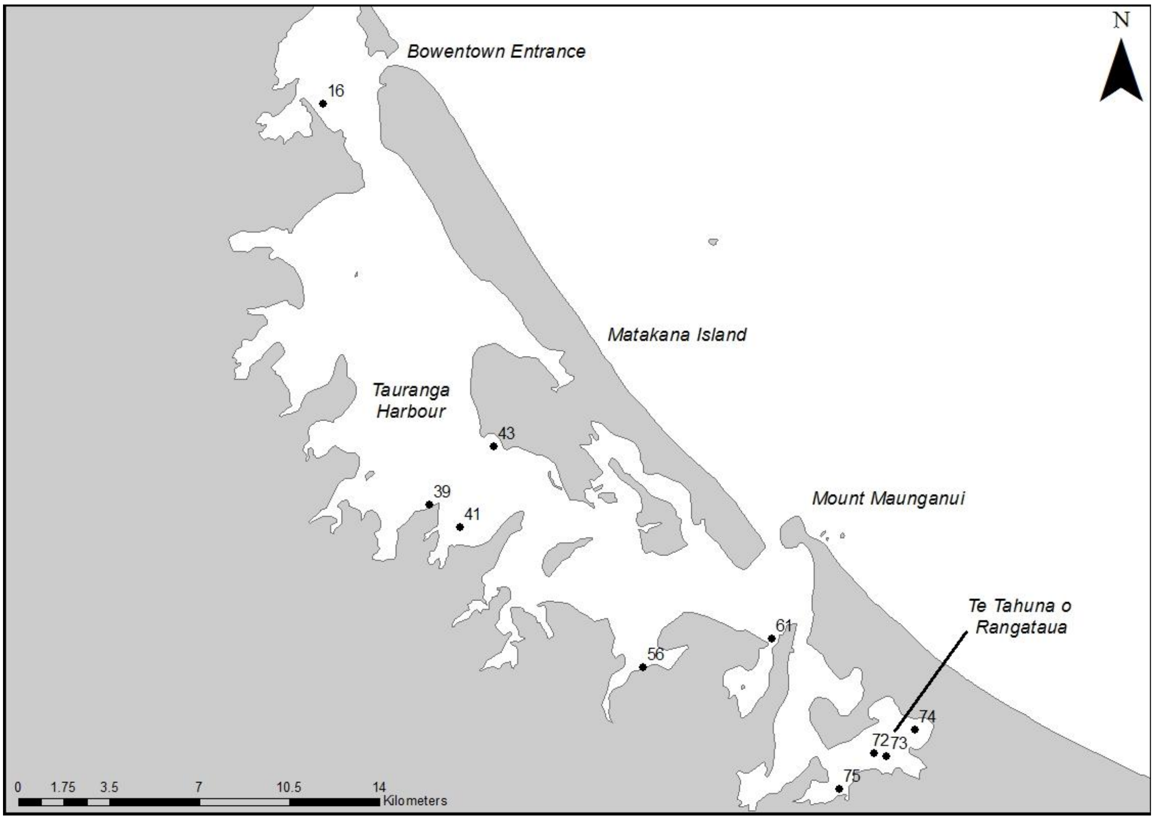

Figure 1.7: Sites within the Tauranga Harbour broad scale survey (Ellis, *et al.*, 2013).

Species richness was found to decrease closer to the upper intertidal fringe, while average abundance of organisms (across 10 replicates within a site, replicate area of 10 m^2) was found to increase at the site closest to the shoreline (Site 74) (see map of site locations in proximity to shoreline).

Ordination modelling approaches were applied to the data collected in the survey to rank the health of each site, based on predicted responses to sedimentation (%

mud content), nutrients (TP), chl-α and contaminants (Cu, Pb). These variables were highlighted to be key anthropogenic stressors and were important in explaining community variation in the harbour. A Benthic Health Model (BHM) was built, relating ecological gradients (which represented key environmental stressors) with community composition to assess the health of each site. Within the BHM, a ranking of one indicated a low effect (healthy) and five indicated a high effect (impacted).

Under the BHM for sedimentation, sites within Te Tāhuna o Rangataua fell within differing categories, with sites 72, 73 and 74 falling in categories 3, 4 and 2, respectively. This indicated that sites 72 and 73 were undergoing moderate to high rates of sedimentation which may be adversely affecting communities. Under the BHM for nutrients, sites within Te Tāhuna o Rangataua fell under category 1 (73 and 74) and 3 (72), indicating low levels of nutrient enrichment in the area, reflecting a relatively health site. Under the BHM for contamination, sites within the Rangataua area fell into category 3 (73 and 74) and 4 (72), indicating that moderate levels of copper, lead and zinc may be entering the area (relative to other estuaries within the Tauranga Harbour) and could have negative implications to ecological health (Ellis, *et al.*, 2013).

1.6.7 Concluding Remarks on Previous Work of Relevance

It appears from the collection of studies of the area, from 1974 to the present, that the ecological character in the direct vicinity of the wastewater treatment ponds has been changed significantly from the original habitat with a concomitant loss in biodiversity, indicating a decrease in overall ecological value to the area. Once an important area for shellfish gathering (in particular the mud snail titiko), since the siting of the oxidation ponds and discharge of sewage to the area, the upper areas are no longer utilized for this purpose. Reports indicate an overall loss of animals in the upper tidal fringe, which could be attributed to a number of impacts the area has seen over the years.

Current hydrodynamics appear to indicate the area has a relatively long flushing time, which may increase accumulation of sediments and pollutants in the area. The area appears to have limited shellfish resources, though the area is perceived to have high cultural significance by local tangata whenua. The area is impacted by a medium extent of urban industrial use, a medium to high extent of agricultural use and a medium to high extent of forestry use. This use would lead to moderate contaminant run-off into the area, via freshwater inputs and the Mangatawa Drain. Historically, the area has received extensive discharges, inclusive of raw sewage. Currently the area receives treated wastewater seepages, with only a minor risk of accidental spill over from the wastewater treatment ponds into the area (Bradley, *et al.*, 2004). The high extent of margin alteration would have reduced ecological value to the area. A moderate extent of modification to hydrodynamic characteristics would have occurred, due to reclamation and siting of oxidation ponds.

Te Tāhuna o Rangataua is an estuarine area with existing and potential adverse effects when considered under the EMPs assessment of degraded estuaries which require State of Environment Monitoring (Robertson, *et al.*, 2002). This is also reflected in reports of the area, which have been summarised within the current chapter.

1.7 Aims and objectives

This research comprises part of a wider Te Maunga Wastewater Management Project currently being undertaken by Manaaki taha Moana Research Team, under Manaaki Te Awanui. This study fits into the ecological survey component of the Wastewater Project, which focusses purely on estuarine condition and environmental impacts within Te Tāhuna o Rangataua.

The aims and objectives of this Master’s research is based on developing an understanding of any cumulative impacts that may be occurring within the estuarine area adjacent to the Te Maunga Waste Water Treatment Plant and its treatment ponds. Through a fine-scale ecological monitoring approach, this study aims to collect information regarding the following key objectives;

- Assessment of the ecological health of an estuarine area adjacent to the Te Maunga Wastewater treatment ponds within Te Tāhuna o Rangatua by following National Estuarine Monitoring Protocol (Robertson, *et al.*, 2002).
- Investigate cumulative effects that may be affecting biodiversity within the study area.

- Identifying key environmental parameters which may be driving change in biodiversity
- Identifying key benthic invertebrate species which drive or characterise variation in community composition

This study will be the first of its magnitude within Te Tāhuna o Rangataua, incorporating a multidisciplinary approach with respect to hydrology, biophysical components and ecology. This research aims to provide an in depth and comprehensive survey of an estuarine area of cultural and ecological importance which may add to the knowledge of estuarine ecosystem processing and may be applicable to the wider Tauranga Harbour in assessing effects of anthropogenic stressors on estuarine condition and biodiversity.

Chapter 2

Methods

The following chapter will explain in detail the methods undertaken within this study, to meet the objectives and aims outlined in Chapter One. This includes descriptions of:

- site choice and collection of samples for the fine scale ecological survey undertaken within the Te Tāhuna o Rangataua, based on the Estuarine Monitoring Protocol.
- the rationale for gradient style of sampling undertaken in order to examine possible effects of the Te Maunga Wastewater treatment ponds.
- the processes undertaken for univariate and multivariate statistical analysis of ecological data using statistical software PRIMER v.6 and STATISTICA

2.1 Ecological Survey Methodology

A standardised Estuary Monitoring Protocol was developed by Robertson et al in 2002 to assess an estuary based on benthic community composition and sediment properties. The collection and processing of samples undertaken within this study is consistent with the EMP.

The fine-scale monitoring programme incorporates measurements of environmental characteristics which reflect the health condition of an estuary. The methodology is used to measure the spatial variation and interactions of a suite of commonly measured indicators which include the following;

- indicators of sediment nutrient and organic enrichment (total nitrogen, total phosphorus, organic content)
- Sediment concentrations of heavy trace metals and other pollutants
- Microbenthic algal coverage (Chlorophyll α)
- Macrofauna community structure (epifauna and infauna)
- Sediment grain size distribution (gravel, sand, silt/clay)

Within an estuarine environment, tidal flushing has a large dilution effect within the water column, with often rapid water exchange (Updegraff *et al.*, 1977), therefore sampling of the benthos may be much more useful when determining environmental health. When sediment properties are altered, associated benthic communities can be profoundly affected (Roper, 1990). Sediments may also give better insight into contaminant levels of an area, with accumulation over time and therefor sampling of benthic characteristics are considered the most appropriate methods for estuarine monitoring. Sampling of the intertidal area was also considered the best option, based on the reasoning that intertidal areas are functionally important, they are the most accessible, data can be obtained quickly and cost-effectively and they are vulnerable to anthropogenic stressors (Robertson, *et al.*, 2002).

Previous studies of estuarine or soft sediment environments within New Zealand have also followed the EMP (Madarasz, 2006; Gillespie *et al.*, 2007; Robertson & Stevens, 2010, 2011; Ellis, *et al.*, 2013; Robertson & Stevens, 2013) to assess impacts of anthropogenic stressors. The protocol promotes robust, scientifically defensible methodology for monitoring (Madarasz, 2006). The development of the EMP involved baseline surveys of fine-scale benthic characteristics across estuarine sites representative of environments found across New Zealand. These surveys provided the backbone to which future estuarine monitoring could follow and compare (Gillespie, *et al.*, 2007).

The EMP is considered a "living document" in which changes and further information are updated periodically, with continuously expanding data sets being able to improve the value of assessment and monitoring (Robertson, *et al.*, 2002). For baselines studies such as the study undertaken within Te Tāhuna o Rangataua, interpretation of the data involves using the results to place the estuary into context both nationally and internationally to assess the current state of the area. Results are compared to relevant guidelines, such as the Australia and New Zealand Environment and Conservation Council (ANZECC) sediment quality guidelines and results from other estuaries (Robertson, *et al.*, 2002).

A gradient style sampling scheme is adopted for this study, to identify any adverse impacts to the estuarine area directly adjacent to and with distances from the Wastewater Treatment Ponds, in the intertidal area of Te Tāhuna o Rangataua.

2.1.1 Site selection

Sampling was undertaken over the summer period in January and February 2014. Digital mapping of Te Tāhuna o Rangataua was undertaken by Manaaki taha Moana and 30 sites were selected to show homogeneity at a range of intervals (over a representative range of intertidal zones) and varied distances seawards, from the wastewater treatment ponds. Using GPS coordinates, sites were selected along three transect lines (10 sites per transect) at varying intervals (5, 15, 30, 50, 100, 200, 300, 400, 500 and 600 m), beginning at the upper intertidal fringe, closest to the WWT ponds and extending out across the intertidal flats toward the mouth of the bay (Figure 2.1).

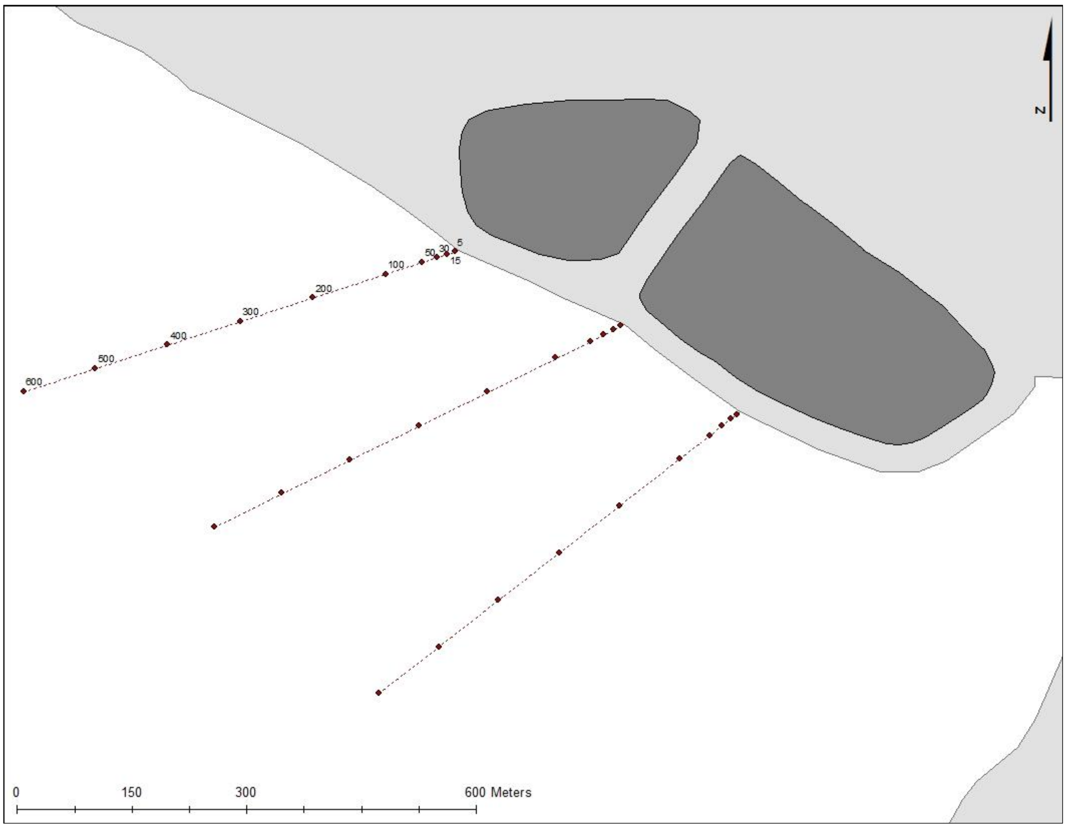

Figure 2.1: Site selection along transects within Te Tāhuna o Rangataua Bay.

At each site (n=30), a 25x10 m grid of ten 'replicate' plots (5 x 5 m) was marked out. Within each plot, a replicate position was randomly chosen and an infaunal core was taken (one core sample per plot, ten core samples per site) to quantify benthic macro-fauna. Surficial sediment core samples were collected (one core per plot, ten cores per site, which were then composited together to form one sample per site) to quantify associated sediment characteristics. Therefore, each site (25 x 10 m) yielded 10 macro-faunal samples and one composite sediment sample. Across all sites (n=30) within the study, 30 sediment samples and 300 benthic invertebrate samples were collected. Within each plot (5 x 5 m), sampling points were randomly generated, therefore maintaining independence within the sampling. The site grid (Figure 2.2) was designed to partition off separate areas to force a broader scale sampling effort.

Figure 2.2: Sampling design of Te Tāhuna o Rangataua. Adapted from Robertson et al (2002)

2.1.2 Sediment composition and quality

Sediment core samples were 20 mm in diameter, and reached a depth of 20 mm into the sediment. The top 20 mm of the sediment was sampled, as this is considered to be sedimentologically recent and is the most active layer, likely to reflect current environmental condition and characteristics of the benthos (Pridmore *et al.*, 1990). Samples were placed into containers, frozen and sent to Hills Laboratories for analysing.

Sediment samples were analysed for a variety of sediment characteristics including organic matter (ash free dry weight, AFDW), total recoverable arsenic (As), copper (Cu), lead (Pb), mercury (Hg), zinc (Zn), total nitrogen (TN), total phosphorus (TP), chlorophyll-α (chl-α) and particle size distribution (% mud, sand, gravel). Sediment grain size can be an indicator of the relative contaminant retention capacity, as contaminants can be retained at higher levels in finer sediments (Madarasz, 2006). Heavy metal analyses provide a low cost preliminary assessment of toxicity within sediments and are a starting point for examining possible contamination and accumulation throughout the food web (Robertson & Stevens, 2011). Sediment chl-α concentrations are a measurement of microscopic algae that can often grow in dense green to orange films on the sediment surface, taking advantage of excessive nutrients where conditions are unfavourable for macro-algal growth (Robertson, *et al.*, 2002).

Concentrations of contaminants within the sediment are compared with the Australia and New Zealand Environment and Conservation Council (ANZECC, 2000) sediment quality guidelines. The guidelines present two threshold levels (low and high) at which concentrations of contaminants may have biological effects. The lower threshold level indicates a concentration which may have a possible effect, while the higher threshold level indicates a probable biological effect. If concentrations exceed the higher trigger value, than more intensive assessment, monitoring or action must come into effect (Gillespie *et al.*, 2012a).

2.1.3 Benthic invertebrate sampling

To quantify benthic community composition at each site, samples of benthic macrofauna living within the sediment (infauna, *e.g.* worms, shellfish) were collected (sieved using a mesh size of 500 µm). A core sample, 130 mm in diameter and 150 mm in length, was taken from within each of the 10 plots in a grid, (photograph of infaunal core can be seen in Figure 2.3). The corer was manually driven into the sediment to approximately 15cm depth, removed with the core contents intact and inverted into a labelled plastic bag. The contents of the core were then washed through a 500 µm sieve mesh using seawater, the animals were retained and preserved in ethanol (diluted to ~70% with seawater) and identified to the lowest taxonomic resolution possible. Care was taken not to trample areas within a site which were being sampled.

Figure 2.3: Photographs of sampling procedure. Clockwise from top left: collecting sediment cores, collection of infaunal cores, example of infaunal core.

Figure 2.4: Photographs of processing infaunal cores: sieving of cores and sorting of infauna.

2.2 Rationale for gradient sampling design

When trying to detect anthropogenic disturbances to ecosystems, one of the hardest challenges an ecologist may come across is separating natural variability from changes in the environment that are brought about by human activities or pollution. Over the years there has been a steady development of sampling and experimental designs within coastal and estuarine environments. These sampling methods focus greatly on trying to detect disturbances within natural variability and having some level of control when dealing with many confounding variables (Ellis & Schneider, 1997).

The relative strength and weaknesses of various ecological monitoring or sampling designs to detect impacts have been evaluated by Ellis and Schneider (1997) beginning with an overview of the progression of such designs over the years. A simple Before-After design, which usually samples one site before and after an impact, is a design to monitor changes in a natural system before and after an environmental disturbance. The design usually involves two sampling periods and takes measurements of as many habitats, organisms and physical and chemical variables as can be afforded, to describe the environment (Morrisey, 1993). This design may be potentially flawed due to the lack of a reference site which is not impacted by anthropogenic disturbance, to control for natural variability.

A Before-After Control-Impact (BACI) design takes this further, by sampling a reference site as well as the site of impact both before and after the impact event (at the ‘impact site”), to control for temporal variability within an environment that may result in changes not attributable to the anthropogenic disturbance being investigated. A repeated BACI, also known as a Multiple-BACI of MBACI, goes further again and involves more intensive sampling over space and time, undergoing many random sampling periods before during and after a disturbance, at the site of impact and in a reference area. This allows for a robust and replicated suite of time series data (Ellis & Schneider, 1997). Ellis and Schneider (1997) discuss the merits of asymmetrical designs proposed by Underwood (1991)and moving beyond BACI. Underwood (1991,1992) highlights that between any two sites that may be chosen for sampling there may be an intrinsic pattern of difference due to any number of physical processes changing abundance of organisms, even if it has nothing to with an impact or human activity (Ellis & Schneider, 1997). Underwood (1991) emphasizes the importance of several randomly selected spatial controls in an experimental design, with designs lacking these resulting in "detection of environmental impacts not being possible against a background of differential variability in numbers of organisms from site to site".

Ellis and Schneider (1997) highlight that all of these designs deal with assessing an impacted area versus one or more control areas, and while these designs are ideal when dealing with an impacted area with known boundaries, they may not be as efficient at assessing impacts in an area in which it is unknown how far reaching a disturbance or pollution spreads. It is suggested that a gradient style design may be more appropriate, sampling along a gradient of distance from the source of the impact. This theory is tested by Ellis and Schneider (1997) by comparing the gradient design (sampling with distance from a disturbance) with a two strata, Control-Impact design (randomized sampling within impact and control areas) and the sensitivity of the two sampling models was evaluated. A data set from the Ekofisk oil platform in the North Sea was used to evaluate each design. For a gradient design all of the distances sampled can be analysed by regression and effects are analysed as a function of distance. For the Control-Impact design, sites are grouped in a block design, grouping sites next to the disturbance as impacted sites and comparing these with sites farthest away as the controls, using ANOVA.

It was found in the results that the gradient design was more powerful and sensitive than the block design, with the regression analysis finding a significant difference in abundance ($p=0.0172$) and the ANOVA block design, with the same data set of impacted and control sites, being less sensitive at detecting differences ($p=0.0340$). A gradient style sampling design may be appropriate, in lieu of a control-impact design, especially when assessing impacts of pollution from a point source. Advantages of gradient style sampling include avoiding having to choose a random control site, especially in another location in which it would not be known if the area is comparable to an impacted site with respect to physical processes (Ellis & Schneider, 1997). Along with this, an impact may not be detected in a disturbed site because a comparable decline or change in faunal abundance (if using community composition as an indicator of disturbance) happens to occur by chance in a control site (Underwood, 1992). Underwood (1992) advises that even between two areas in which there is no known impacts or disturbances, there is no reason to expect that they will undergo the same changes in community composition through time and space.

Results produced from a gradient design enable environmental and biological changes to be assessed as a function of distance. In this respect, results may be much easier to interpret and present to a wider audience (Ellis & Schneider, 1997), which is important given that environmental impact studies are often undertaken as part of a monitoring program, or required by council as part of consent conditions.

It is with this rationale that the current study employs the gradient style sampling procedures to assess the impacts, if any, of the wastewater treatment ponds and seepages with distance from the source of the impact. Sites at farthest distances are suggested to act as reference areas. Within Tauranga Harbour, choosing a reference site would be difficult due to the confounding factors of differing morphology and hydrodynamic regimes influencing variability between estuarine inlets, along with differing anthropogenic stressors between estuaries within the Harbour.

2.3 Statistical analyses

The statistical methods follow previous studies that deal with multivariate community data and associated environmental variables within an estuarine or soft sediment environment (Robertson, *et al.*, 2002; Anderson *et al.*, 2004; Bennett *et al.*, 2006). Appropriate univariate and multivariate statistical procedures were undertaken to aid in descriptions and interpretations of spatial and temporal variability within community composition (Bennett, *et al.*, 2006).
Relationships with environmental characteristics were also investigated to ultimately help describe ecological processes occurring within the area and to highlight any variables that are having a marked influence on benthic community structure.

Clarke and Warwick (2001) highlight that community data, such as the data collected for this study, is highly multivariate and subject to high statistical noise, therefore the data must be analysed 'en masse' to draw out biological structure and patterns in relation to the environment. Standard parametric modelling based on assumptions of normality is not statistically viable when analysing multivariate species data. Species are often absent from many samples, leading to a predominance of zeroes within the data, as well as this, the counts of species across the data can be highly variable (Clarke & Warwick, 2001). Multivariate analysis was undertaken using PERMANOVA+, an add-on to the PRIMER v6 computer program (Anderson *et al.*, 2008). PERMANOVA calculates p-values using permutations, rather than using p-values under normal standard distribution (Atalah & Crowe, 2012) and is specifically designed to deal with multivariate ecological data.

2.1.4 Species Data

Raw macro-faunal data was entered into the statistical software PRIMER v6 and the replicates were summed to the site level to reduce noise and variability (Anderson, *et al.*, 2004). Univariate analyses was first undertaken, collapsing the full set of species counts to describe species diversity at each site, reducing the complexity of the multivariate data (Clarke & Warwick, 2001). Diversity indices were calculated using the DIVERSE function in PRIMER v6. Univariate measures included number of species (S; total number of species at each site), number of individuals (N; total number of individuals at each site) and Shannon-Weiner diversity index (H'; A diversity measure which produces a single number to describe different types and abundances of animals present in a collection (Gillespie *et al.*, 2012b)). Smaller index values are produced for sites or communities with only one or a few species and higher values are produced for sites containing many species, with each species having fewer individuals (Bennett, *et al.*, 2006). This data was plotted in graphs in excel to visualise a low-dimensional picture of macro-faunal communities within each group (Clarke & Warwick, 2001).

Macro-faunal data was then transformed to the fourth root, a moderate transformation (chosen from square root, fourth root, log (+1) and presence/ absence; in order of severity), which down weights quantitatively common species and gives more weight to rare species, at a medium degree of severity (Clarke & Warwick, 2001). Transformation is primarily a biological decision, based on the raw data and how species may influence results. For example, a large group of opportunistic species may be found by chance capture, which would skew the results and may not be representative of true community composition in the area.

A Bray-Curtis dissimilarity matrix is a semi-metric index which is suggested to provide a meaningful measure of dissimilarity when dealing with analyses of multivariate species data and ecological community composition. The Bray-Curtis operates at the species level, so similarities for each group can be obtained for each species. A Bray-Curtis matrix uses permutations, unlike Euclidean distance (which is the *natural distance* between any two data points in space) and does not need to meet underlying assumptions of normality required for a standard approach to linear modelling (Clarke & Warwick, 2001). A dissimilarity matrix is a starting point in constructing ordinations, with dissimilarities between samples being transformed into distances between samples in a "map" style configuration (Clarke & Warwick, 2001). Large dissimilarities between community structure (samples that have few species in common or different levels of abundance) should reflect larger "distances" between samples on the map, while smaller dissimilarities would lead to clustering of samples (in this case, species) or a depiction of closer "distances" on the map.

2.1.5 Ordination techniques

The Bray-Curtis dissimilarity matrix was applied to the transformed macrofaunal data (Clarke & Warwick, 2001; Bennett, *et al.*, 2006) with the selection criteria chosen being a AICc stepwise selection procedure. An unconstrained non-metric MDS ordination was performed based on the Bray-Curtis matrix and distance, as the descriptor variable, to show patterns in community composition based on geographical distance within the area of study. An MDS method (along with other useful multivariate methods) place samples on a map in two or three dimensions, based on the relative order (ranking) of distances between samples and matching similarities or dissimilarities taken from an underlying matrix. Representation by an ordination is appropriate when looking at community patterns that are responding to continuous ecological gradients (Clarke & Warwick, 2001). The success of an ordination in terms of appropriately representing faunal data is measured by a stress co-efficient. The stress value for the MDS was found to be 0.15, which gives a potentially useful picture but not an excellent representation (e.g. Stress<0.05). Clarke and Warwick (2001) suggest that “a cross-check of any conclusions should be made against those from an alternative technique”.

Principal Co-ordinates Ordination

An unconstrained PCO (Principal Co-ordinates Ordination) was then performed, which is a metric ordination, like a PCA, but can be based on dissimilarity matrices such as Bray-Curtis, which makes it more flexible than an ordinary PCA, although the final projection is onto a low dimensional space, like that of a PCA (Clarke & Warwick, 2001; Anderson, *et al.*, 2008). For this data, the PCO was considered a better representation in an ordination, of species driving community composition. In practice, a MDS and PCO will tend to give similar results when based on the same matrix. Of more importance to the resulting patterns found for either an MDS or PCO ordination, is the treatment of the raw data in terms of transformations, normalisations or underlying matrices used. The PCO configuration was then overlaid with associated vector plots of correlated taxa (correlations found to be > 0.4).

2.1.6 SIMPER (Similarity percentage) analyses

A SIMPER (similarity percentage) analysis was then performed based on the Bray-Curtis dissimilarity matrix. A Bray-Curtis dissimilarity matrix is used to look at within-group similarities and between group dissimilarities, to identify which species are most important in creating the observed differences between communities at each site (Atalah & Crowe, 2012). Similarity percentages are produced, to distinguish species driving variation between and within groups. Percentage contributions from each species are listed in decreasing order of such contributions.

This analysis highlights key indicator species which are responsible for dissimilarities in community composition between groups characterized by distance. Output tables first show the species which contribute to highest percentage of variation in a group and then the following tables show species in order of highest percentage adding to variation between groups. A second SIMPER analyses was undertaken, looking at specific distances placed into two groups; close and far. The ‘close’ groups comprised of sites at distances 5 and 15 and the "far" groups consisted of sites at distances 500 and 600 m. This analysis was undertaken to investigate similarities between groups close together and groups far away together with dissimilarities between both groups, in order to detect species which may be driving similarities and dissimilarities in community composition at each end of the transects within the study.

2.1.7 Environmental Data

Environmental data, which consisted of sediment characteristics (organic matter, grain size, Heavy trace metals, nutrients (TP, TN) and chl-α) was then submitted to analyses. Samples collected for this data were composite replicates for each site (n=30) and all data was normalised to be analysed, as the environmental variables had different units of measurement which were incomparable when kept in raw data form. The coefficients associated with each variable, that are produced from normalising the data, will scale appropriately to adjust for disparity between variables (Clarke & Warwick, 2001). A draftsman plot was then produced, a simple matrix of all pairwise combination of variables, to detect correlations among the predictor variables. The draftsman plot calculates Pearson correlation coefficients between all pairs of variables.

2.1.8 DISTLM (Distance based linear modelling)

The relationship between species data and environmental variables was analysed using a nonparametric multiple regression model, DISTLM (Anderson, *et al.*, 2004). The DISTLM (distance based linear modelling) was performed on transformed species data (as described by Bray-Curtis dissimilarities), against predictor variables (normalised environmental variables), along with distance as a predictor variable. The DISTLM is used to describe relationships between species data and predictor variables by partitioning variance in a multivariate data cloud, based on dissimilarity matrices, in a multiple regression model.

The DISTLM model produced marginal results, which display the percentage of each individual predictor variables contribution to variation (ignoring all other variables), within the species data. The predictor variables were subjected to a step wise selection procedure (Anderson, *et al.*, 2004), choosing AICc criterion, which was considered the best criterion as it corrects for small sample sizes, with the step-wise including one variable at a time and evaluating the model repeatedly. The sequential test fits the variables in the best order, but looks at them together as a model.

The draftsman plot was examined for evidence of collinearity and variables which had the highest correlations were identified. A key variable that correlated with many other variables and in particular, was highly correlated with Organic Matter, was Pb (>0.95). This variable was forced out of the DISTLM and the analysis was run again. Variables that strongly correlate can have the issue of multi-collinearity, strong correlations among the X or predictor variables. If two explanatory variables show high co-linearity, then one may act as a proxy for the other, they effectively contain the same information and are "redundant" within the DISTLM model (Anderson, *et al*., 2008)

To visualise patterns from the multivariate data, the DISTLM model was plotted in a dbRDA (distance-based redundancy linear analysis) ordination, which uses principal coordinate analysis, fitting values on a plot for a given model (McArdle & Anderson, 2001). The model is fitted into ordination by doing an eigen-analysis of the fitted data cloud, a constrained ordination to find linear combinations of the predictor variables which explain the greatest variation in the data cloud. This will explicitly investigate the relationship between environmental variables and community composition (Anderson, *et al.*, 2004).

2.1.9 Principal Co-ordinates Analysis (PCA) for Environmental data

The PCA technique studies linear relationships among variables and results in a two dimensional ordination plot of relationships. A PCA is not a good technique to be used for species data as it is implicitly based on Euclidean distances which are not well-suited to the unique variability and predominance of zero values found in species data. A PCA is however a good technique for analysis of environmental variables on their own. Environmental data is often much smaller than species data. As well as this, it is continuously scaled and the data can be normalised so that they have comparable, dimensionless units of measurement enabling them to fit standard correlation coefficients (Clarke & Warwick, 2001). Because of this, Euclidean distances are an appropriate measure to describe relationships between environmental variables.

The PCA provides eigenvector weights, a linear system of equations, which provide coefficients for a linear combination of the original variables that will produce the PC scores (Anderson *et al.*, 2006). The higher the value of the eigenvector co-efficient (in both a negative and positive direction) the more important the variable is in explaining variance. A PCA ordination was plotted for the normalised environmental data. The PCA showed the distribution of distance in relation to environmental variables driving differences at each site along PC axes. The plot was overlaid with associated vector plots of correlated environmental variables (Pearson correlation > 0.4).

2.1.10 PCA comparing Environmental data with 74 estuarine sites within the Tauranga Harbour.

Raw data was obtained from A Broad Scale Ecological Survey of the Tauranga Harbour, undertaken by Manaaki Taha Moana and Cawthron Institution in 2011/12 (Ellis, *et al.*, 2013). Sampling was undertaken in the mid to late summer period, which is the time period suggested within the EMP in which estuarine monitoring should be undertaken, to minimise interference due to seasonal variation when comparing values between estuaries (Robertson, *et al.*, 2002).

This survey examined 75 sites from estuarine areas throughout the Tauranga Harbour, to understand the roles of various anthropogenic stressors to ecological health of the harbour. Collection of samples mirrored the sampling methods undertaken within this ecological survey, following the EMP (Robertson, *et al.*, 2002). Environmental data (organic matter (AFDW%), grain size %, TN (mg/kg), Cu (mg/kg), Pb (mg/kg), TP (mg/kg), Zn (mg/kg) chl α (ug/kg)) from 74 sites across the harbour was analysed together with the 30 sites from Rangataua Bay and a PCA ordination was produced, using the software STATISTICA. Site 48 within the broad scale survey was an outlier and therefore omitted from the analysis, which had exceedingly high levels of nutrients and mud content.

2.4 Limitations

The gradient style sampling has come into question regarding the lack of a control or reference site, separate from the area of interest. Within the Tauranga Harbour and in particular the southern basin, most estuaries are receiving inputs from any number of sources, taking this into account, along with differences in geomorphology and hydrodynamics across estuaries, choosing a relevant control or reference sight would have been difficult if not impossible. When incorporating a reference site into a study, anthropogenic impacts that may be influencing a control site, along with differing morphology and hydrodynamics, would also have to be examined. However, a reference site would have added valuable information to the present study. A reference site would have arguably removed the confounding factor of tidal elevation and allow for the separation of change in assemblages due to tidal inundation from other environmental variables. A reference site would also allow for comparison of community composition against a backdrop of comparable environmental variables, which may have produced trends not apparent within this study.

Estuarine sites across Tauranga Harbour have, however, been compared with sites from this study in order to place Te Tāhuna o Rangatua within a wider context of environmental condition.

Chapter 3

Results

The results section will begin by first focusing on the environmental data and trends found within the data and then present the species univariate and multivariate analyses. The environmental data and benthic invertebrate data will then be bought together and analysed using Distance Based Linear modelling (multiple regression analyses). A PCA ordination was also undertaken, looking at Te Tāhuna o Rangtaua compared with other estuaries within the Tauranga harbour.

3.1 Environmental Data

Sediment properties that were analysed include organic matter (ash free dry weight, AFDW), total recoverable arsenic (As), copper (Cu), lead (Pb), mercury (Hg), zinc (Zn), total nitrogen (TN), total phosphorus (TP), chlorophyll-α (chl-α) and sediment grain size. Mercury (Hg) was found to be below the default detection limit (0.10 mg/kg dry weight) and was therefore omitted from any further analysis. Site-specific details of physical variables can be found in Appendix 1.

3.1.1 Sediment grain size and organic content

Grain size profiling gives a good indication of the muddiness of an area and may reflect rate of sedimentation occurring (Robertson & Stevens, 2013). Across all 30 sites within the survey, Site 1 along transect A (5 m from shoreline (Fig 2.1)) was found to have the highest organic content (4.9% (AFDW)) and the second highest percentage of mud content (20.2% silt and clay). Values across all sites for organic content ranged from 1.09 - 4.9 % ($\bar{x}$ =1.96 %).

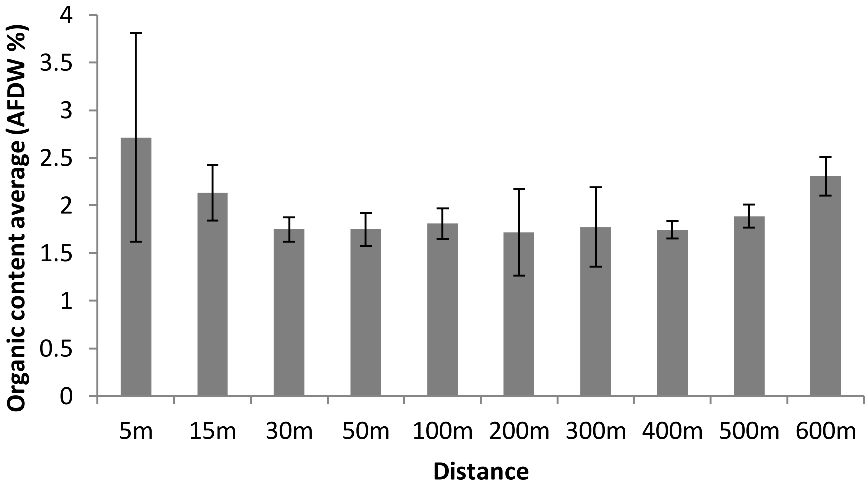

Figure 3.1: Average of organic content (% AFDW) across three transects (A, B and C) at each distance (n=3), with error bars (±SE), Te Tāhuna o Rangataua.

In previous reports of estuarine areas, organic content has been given a condition rating of 1< 2% = moderate, 2< 3.5% =high and >3.5% very high (Robertson & Robertson, 2014). In general, organic content was found to be moderate or high across all sites, with the exception of Site 1 on transect A (Figure 2.1). This indicates that organic enrichment is occurring in some areas within Te Tāhuna ō Rangataua.

From grain size profiling analysis, the percentage of mud content (silt and clay, <63 µm) was variable across sites, with some sites having moderate to high mud content, indicating that inputs of fine sediments are occurring in the area. As noted by Hume, *et al.* (2009), present day mud content is scaled as high (>20%), moderate (10-20%) and low (<10%) within estuarine areas. The highest percentage of mud content (clay and silt; 22.8%) was found at site 6.5 along transect C (300 m from the shoreline, Figure 2.1). Values across all sites ranged from 22.8% - 3.9% ($\bar{x}$ =9.2%).

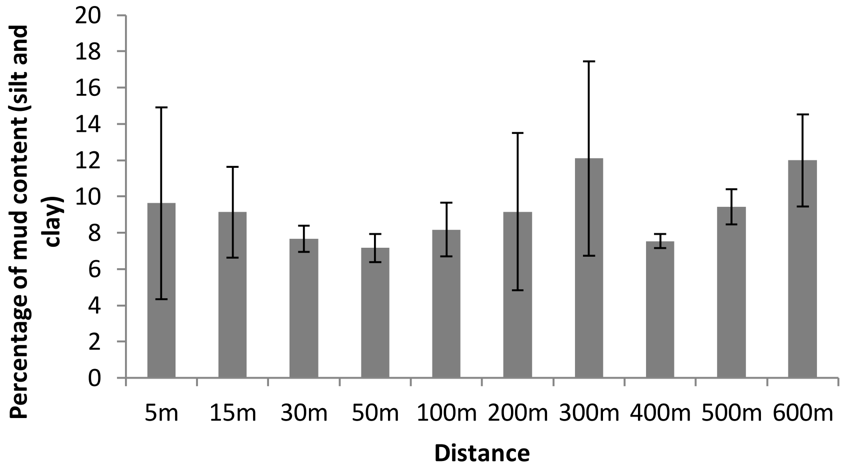

Figure 3.2 Average mud content percentages across three transects (A, B and C) at each distance (n=3), with error bars (±SE), Te Tāhuna o Rangataua.

Percentage of mud content (Figure 3.2) does not show a pattern along a gradient of distance and mud content reaches an average of 12% at distances 400 and 600 m along transect lines.

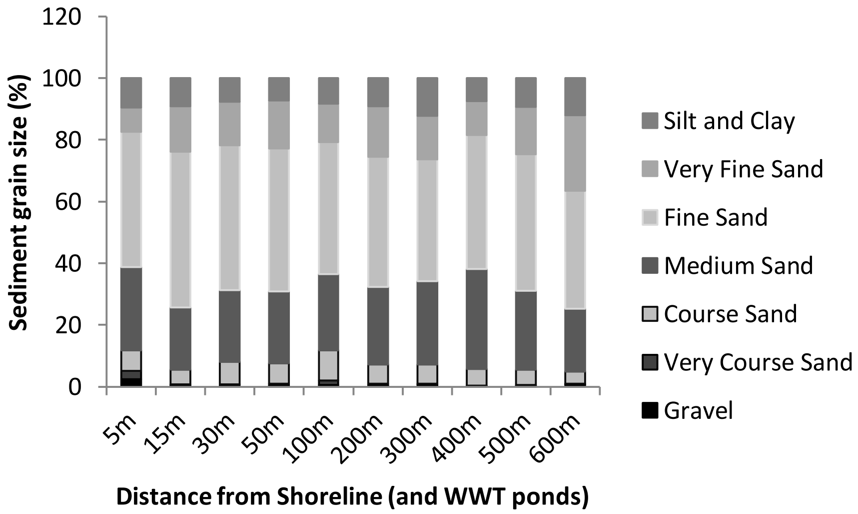

Figure 3.3 Average grain size percentages across three transects (A,B and C) at each distance (n=3) within the intertidal area of Te Tāhuna o Rangataua.

Across all sites, fine sand (fraction < 250 μm, >/= 125 μm) appears to be the most dominant grain size (indicating sediment grain composition within the intertidal area is moderately fine to silty), ranging from 56% - 21.6% ($\bar{x}$ =43.8%). The proportions of sediment classes are relatively consistent across all sites and distances offshore.

3.1.2 Nutrients

Within New Zealand estuaries, total nitrogen (TN) values 1000– 2000 mg/kg are considered moderately elevated, while values 250–1000 mg/kg are low. Total phosphorus (TP) values 100–300 mg/kg indicate low levels and >300–500 mg/kg are moderate (Robertson & Robertson, 2014).

TN concentrations were found to be moderate to slightly elevated within the area, while TP concentrations were found to have moderate to low concentrations across sites. TN concentrations across sites ranged from 200 to 1500 mg/kg ($\bar{x}$ =600 mg/kg). TP concentrations across sites ranged from 90 to 470 mg/kg ($\bar{x}$ = 159.6 mg/kg). The highest concentrations of TN and TP and the highest percentage of organic content were found at one site, Site 1 on transect A (5 m from the shoreline, *see* Figure 2.1).

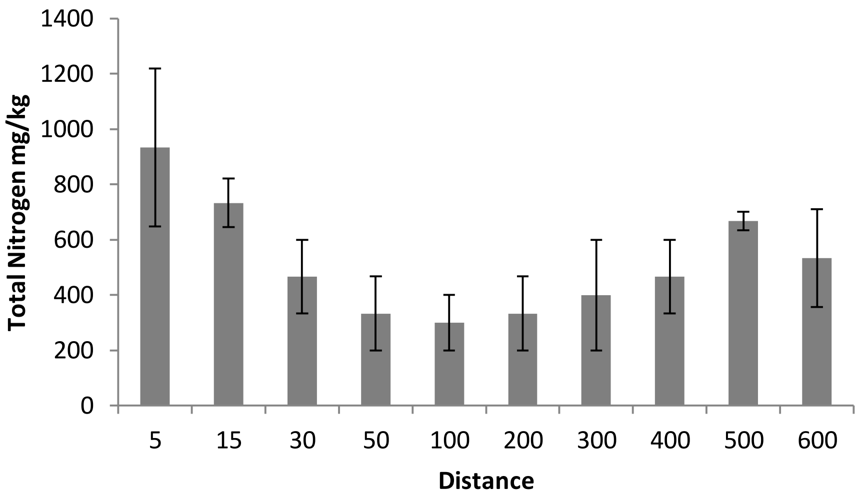

Figure 3.4 Average concentrations of total nitrogen (mg/kg) across three transects (A, B and C) at each distance (n=3), with error bars (±SE), Te Tāhuna o Rangataua.

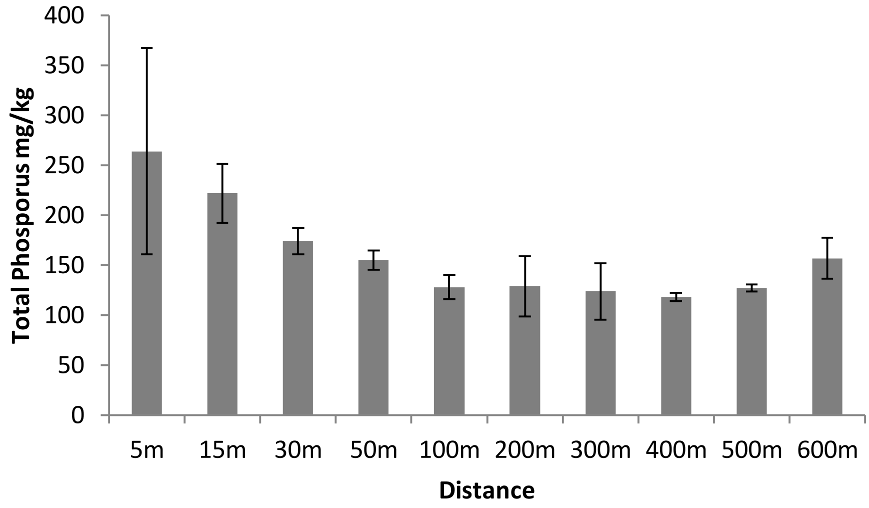

Figure 3.5 Average concentrations of total phosphorus (mg/kg) across three transects (A, B and C) at each distance (n=3), with error bars (±SE), Te Tāhuna o Rangataua.

TN and TP both are both elevated closest to the shoreline (5 m) and WWT ponds and then decreased with distance, though TN shows a gradual increase between 200 and 500 m.

3.1.3 Chlorophyll-α

Chl-α concentrations were found to be particularly high across most sites within the area, with values ranging from 49100 µg/kg to 12800 µg/kg ($\bar{x}$= 23,900 µg/kg) when compared to estuaries within the broad scale survey of Tauranga Harbour ((Ellis, *et al.*, 2013) *see* Table 3.13 for chl-α values across the Tauranga region). The highest concentration of ch-α was found at site 1, on transect C (5 m from shoreline (Figure 2.1, *see* Appendix 1)) and the lowest concentration was found at site 6.5 on transect C (300 m from the shoreline).

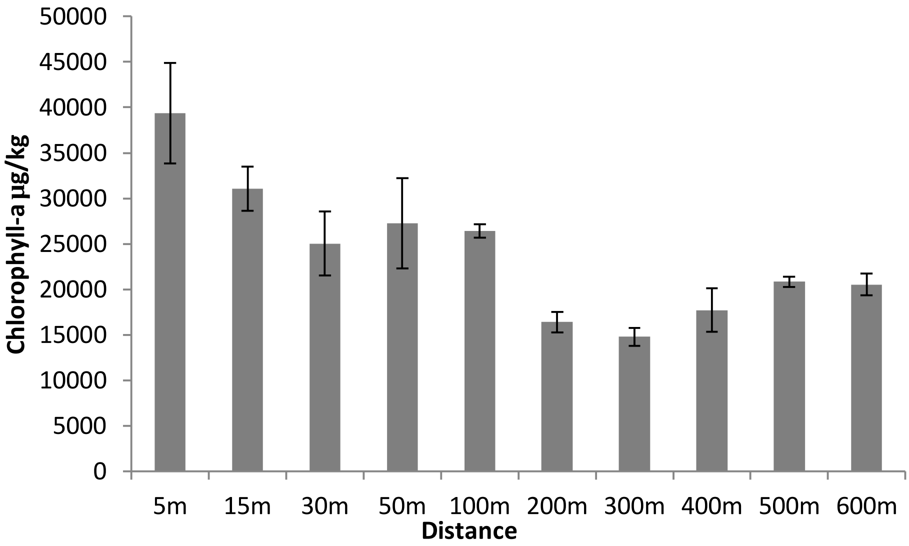

Figure 3.6: Average of concentrations of Chloropyhll-α (mg/kg) across three transects (A, B and C) at each distance (n=3), with error bars (±SE), Te Tāhuna o Rangataua.

From Figure 3.6, Chlorophyll-α concentrations are highest closest to the wastewater treatment ponds and then decrease with distance, as was the trend found for nutrients.

3.1.4 Heavy Trace Metals

Heavy trace metals detected within the sediment were all found to fall below the Australian and New Zealand Environment and Conservation Council (ANZECC, 2000) Interim Sediment Quality Guidelines (Table 3.1).

Table 3.1: ANZECC (2000) Interim Sediment Quality Guideline trigger values for trace metals analysed within Te Tāhuna o Rangataua

ANZECC mg/kg		
Trace Metal	ISQG-Low (Trigger value)	ISQG-High
Copper	65	270
Lead	50	220
Mercury	0.15	1
Zinc	200	410
Arsenic	20	70

Lead and zinc were found at higher concentration levels than other heavy metals. Pb concentrations ranged from 5.4 mg/kg to 1.2 mg/kg ($\bar{x}$ = 2.1 mg/kg), with the highest concentrations also being found at site 1 on transect A (*see* Appendix 1 for all environmental data). Zn concentrations across sites ranged from 5 mg/kg to 31 mg/kg ($\bar{x}$ = 13), with the highest concentration found at site 6.5 on transect C (Figure 2.1). Copper concentrations were low across sites (many sites did not reach default detection limits) and ranged from 0.5-4mg/kg ($\bar{x}$ = 0.53 mg/kg), with the highest concentration being found at Site 1 on transect A (5 m from shoreline). Arsenic (As) concentrations were also found to be very low and ranged from 0.5-3 mg/kg ($\bar{x}$ = 1.3 mg/kg).

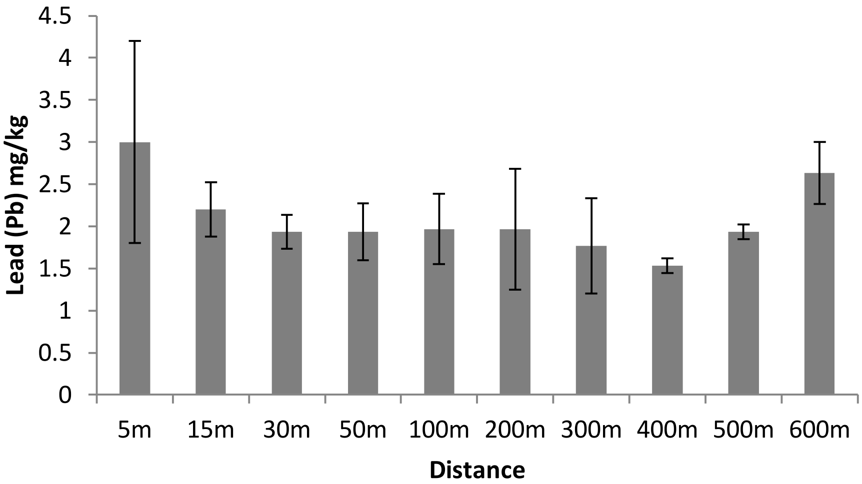

Figure 3.7: Average of concentrations of Lead (Pb) across three transects (A, B and C) at each distance, with error bars (±SE), Te Tāhuna o Rangataua .

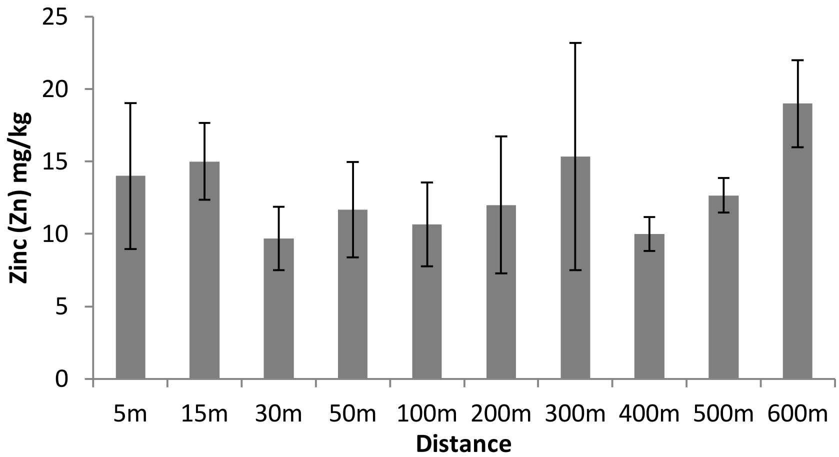

Figure 3.8: Average of concentrations of Zinc (Zn) across three transects (A, B and C) at each distance, with error bars (±SE), Te Tāhuna o Rangataua.

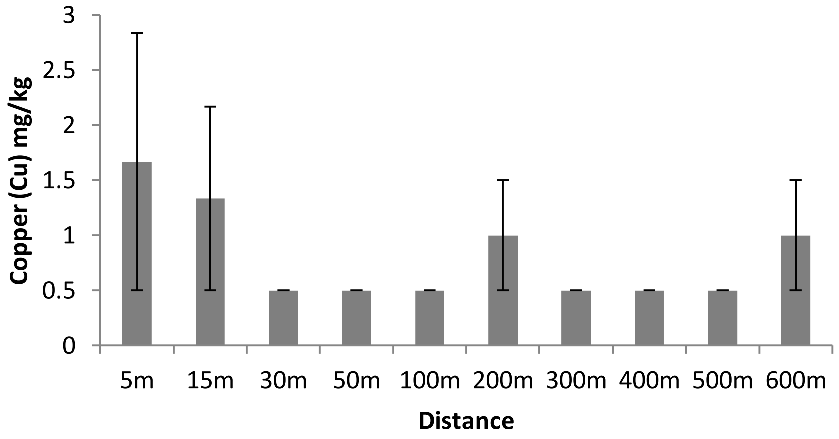

Figure 3.9: Average of concentrations of Copper (Cu) across three transects (A, B and C) at each distance (n=3), with error bars (±SE), Te Tāhuna o Rangataua.

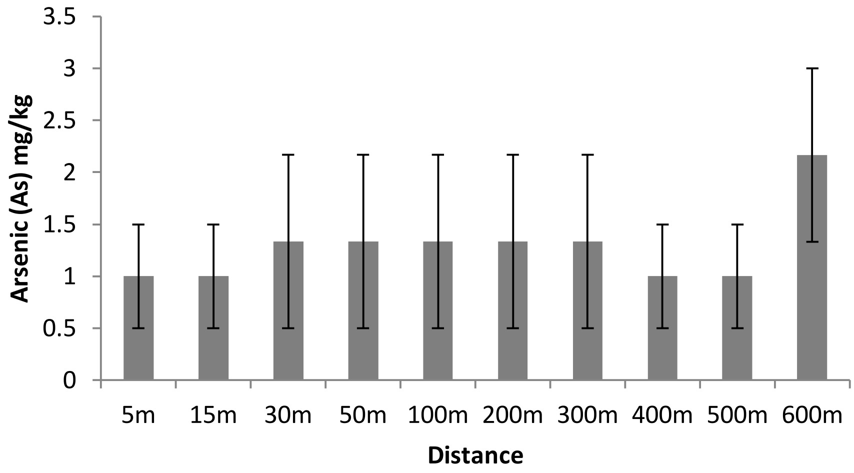

Figure 3.10: Average of concentrations of Arsenic (mg/kg) across three transects (A, B and C) at each distance (n=3), with error bars (±SE), Te Tāhuna o Rangataua.

None of the metals followed a decreasing trend with a gradient of distance, however, copper and lead were slightly elevated closest to the treatment ponds. Lead appeared to be slightly elevated, decreases slightly with distance, then increasing again towards the mouth of the bay. Zinc appears to follow the same trend as percentage of mud content, indicating that accumulation of zinc may be occurring within finer sediments.

3.2 Species Data

A total of 33 macro-faunal taxa were identified from the 297 replicate samples (that were processed. A total of 300 samples were collected but three of the macrofaunal replicate samples (transect A- rep 2, transect A- rep 3 and transect C-rep 7) were mislabelled and were not able to be included in the data set.

3.2.1 Diversity measures

For calculation of diversity measures, macrofaunal data was summed to site level, to reduce noise and variability and mirror the composite sediment samples collected (n=30). The full table of diversity data can be found in Appendix 2. Total abundance (N, number of individual fauna per site (each site = 250 m^2) ranged from 58 to 753 ($\bar{x}$=356). The number of species (S) found across sites (an area of 250^2) ranged from 10 to 26 ($\bar{x}$=18.9). Site 8 on transect C (600 m from shoreline, Figure 2.1) had both the highest species and total abundance count of all sites. The gastropod *Zeacumantus lutulentus* was found to be the most commonly occurring animal across all sites, while polychaete worms *Scoloplos cylindrifer* and Nereididae (juveniles) were also found in high numbers across sites (*see* Appendix 3).

Table 3.2 All taxon, descriptions and feeding modes (retained from a 5 mm mesh) found within the ecological survey of Te Tāhuna o Rangataua.

Taxon	Description	Feeding group	Taxon	Description	Feeding group
Amphibola crenata	Gastropod (mud snail)	Grazer	*Helice crassa*	Crustacean (mud crab)	Deposit feeder/ scavenger
Amphipoda indet_	Amphipod	Deposit feeder/grazer /scavenger	*Heteromastus filiformis*	Capitellid polychaete	Sub-surface deposit feeder
Anthopleura aureoradiata	Sea anemone	Filter feeder	*Macomona liliana*	Bivalve (wedge shell)	Suspension feeder
Aonides trifida	Spionid polychaete	Surface deposit feeder	*Macrophthalmus hirtipes*	Crustacean (crab)	Deposit feeder/ scavenger
Armandia maculata	Ophiliid polychaete	Deposit feeder	*Nemertea*	Nemertean worm	Carnivore
Austrovenus stutchburyi	Bivalve (cockle)	Suspension feeder	Nereididae (juvenile)	Juvenile polychaete (rag worm)	Predator/ scavenger
Ceratonereis sp.	Nereid Polychaete	Predator/ scavenger	*Nicon aestuariensis*	Nereid polychaete	Predator/ scavenger
Cominella glandiformis	Gastropod (mud whelk)	Predator/ scavenger	*Notoacmea helmsi*	Mollusk (limpet)	Grazer
Corophiidae	Amphipod	Deposit feeder/ scavenger	*Nucula hartvigiana*	Bivalve (nut shell)	Suspension feeder
Cumacea	Crustacean	Deposit/ suspension feeders	Oligochaeta	Oligochaete worm	Deposit feeder
Diptera	True fly (terrestrial)	Larvae- not feeding	*Perineries nuntia var. vallata*	Nereid polychaete	Predator/ Scavenger
(Diptera) Tipulidae	Crane fly	Larvae- not feeding	Polydorid	Spionid polychaete	Deposit/ suspension feeder
Diloma subrostrata	Gastropod (mud flat topshell)	Grazer	*Prionospio aucklandica*	Spionid polychaete	Deposit feeder
Edwardsia sp.	Sea anemone	Filter feeder	*Scolecolepides benhami*	Spionid polychaete	Deposit feeder
Eliminus modestus	Crustacean (barnacle)	Filter feeder	*Scolelepis* sp.	Spionid polychaete	Deposit feeder
Halicarcinus varius	Crustacean (pill box crab)	Deposit feeder/ scavenger	*Scoloplos cylindrifer*	Orbinid polychaete	Sub-surface deposit feeder
			Zeacumantus lutulentus	Gastropod	Grazer

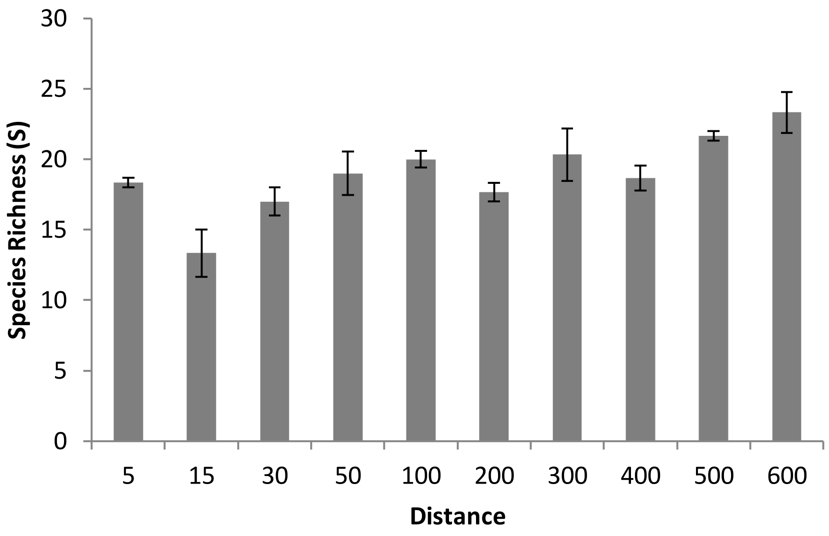

Figure 3.11 Species richness of macro faunal data summed to site level and averaged over three transects (A, B and C), at each distance from the shoreline (n=3), with error bars (±SE), Te Tāhuna o Rangataua. Full table can be found in Appendix 2.

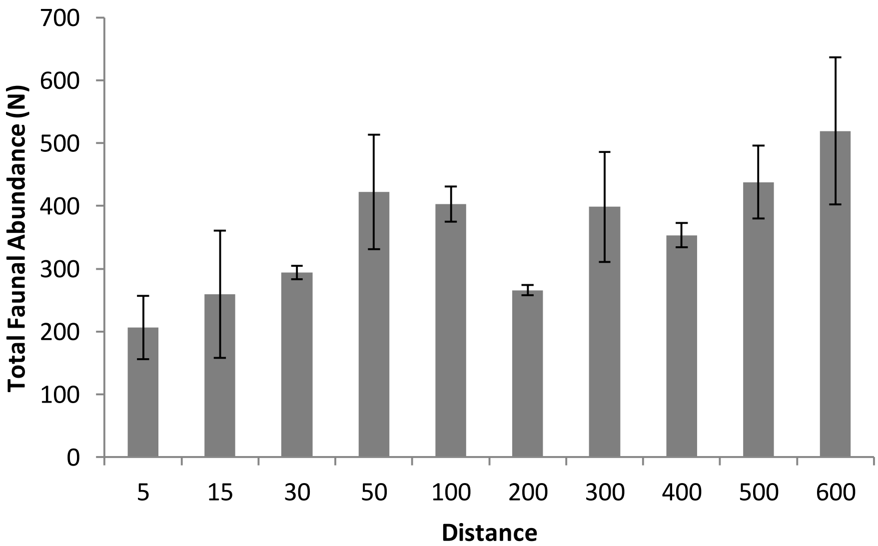

Figure 3.12 Total abundance of macro-faunal data summed to site level and averaged over three transects (A, B and C), at each distance from the shoreline, with error bars (±SE), Te Tāhuna o Rangataua. Full table can be found in the Appendix 2.

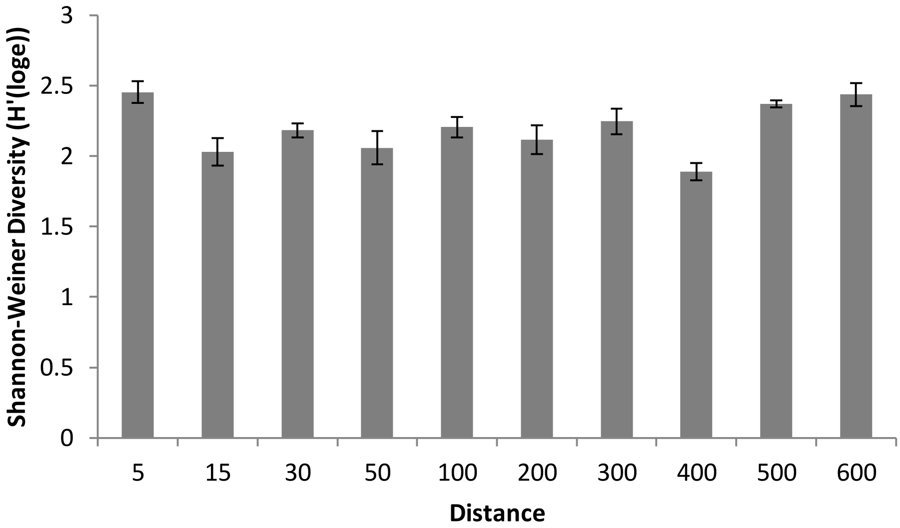

Figure 3.13 Shannon Wiener diversity indices (H') of macrofaunal data summed to site level and averaged over three transects (A, B and C), at each distance from the shoreline, with error bars (±SE), Te Tāhuna o Rangataua. Full table can be found in the Appendix 2

Data presented in Figure 3.11 indicates a slight increase in species richness with distance, with the exception of number of species found at the closest distance (5 m). The graph shows that the highest average of species richness is at the furthest site from the WWT ponds (600 m). Figure 3.12 shows an increase of total abundance of organisms along a gradient of distance (down shore/away from the WWT ponds), with the exception of total abundance, which declines at 200 m from the shoreline. At a distance of 600 m, total abundance is higher than other distances. This high abundance would be due to large numbers of organisms found at site C8. Shannon Wiener diversity indices do not change much along a gradient of distance, with each distance appearing to portray relatively similar values.

The Shannon Wiener index incorporates both species richness and species evenness to assess diversity of an area (Bennett, *et al.*, 2006). The values usually range between 1.5 and 3.5 and increase when biodiversity increases. From Figure 4.13, it appears that most sites have a medium level of biodiversity, with groups at distances 5 m and 600 m having slightly higher biodiversity levels than other groups. Overall, communities were numerically dominated by gastropods, in particular *Zeacumantus lutulentus,* grazers on the surface of the sediment and polychaete worms that live and feed within and on the surface of the sediment.

3.3 Multivariate Analyses

3.3.1 Principal Co-ordinates Analysis (PCO)

Results from the PCO indicate a pattern along the PC01 (horizontal) axis, with bubbles clustering together and showing a change in community composition based on a gradient of distance, potentially reflecting a gradient of decreasing nutrient and chl-α levels with distance from the WWT ponds (Figure 3.14). The first axis (PCO1) explained 42.2% of variation within the community data. Associated vector plots of taxa, found to have a correlation higher than 0.4, were overlaid on the ordination.

Fauna found to influence community composition at closer distances include polychaete worms *P. vallata, Ceratonereis* sp.; the amphipod Corophiidae*;* a barncacle *E. modestus; and* mud crabs *H. crassa and* Diptera (*see* Table 3.2 for full names and species descriptions).

Fauna found to characterize community composition at distances further away include polychaetes Polydorid, *Oligochaeta*, *Scolelepsis* sp., *A. trifida, P. auckandica:* Nereididae (juvenile), *S. cylindrifer*, *S. benhami* and *H. filiformis;* a crustacean, Cumacea*;* gastropods *C. glandiformis*, *Z. lutulentus*, *D. subrostrata;* a suspension feeding anemone *A. aureoradiata;* bivalves *M. liliana N. hartvigiana;* a limpet *N. helmsi;* and flat worms Nemertea.

3.3.2 Similarity Percentages Analysis (SIMPER)

The SIMPER analyses provide a breakdown of average similarity within groups and the average dissimilarity between groups based on individual animals contributions to a community. Two SIMPER analyses were performed. One which used all the site distances in the analysis and a second which grouped sites closest to the impact site (5 and 15 m) and sites furthest away (500 and 600 m). The first SIMPER analysis (grouped by site distance), indicated that macrofaunal assemblages across transects at each site are quite similar (61.1–82.5% similarity; Table 3.3).

Table 3.3 Average similarities within groups (grouped by distance from shoreline in metres) based on Bray-Curtis dissimilarities of fourth root transformed macro-faunal data.

Macro-faunal assemblages grouped by distance (m)	Average similarity (%)
5	68.5
15	61.1
30	76.9
50	71.5
100	81.0
200	74.3
300	74.3
400	82.5
500	77.9
600	76.43

Table 3.3 indicates that across transects A, B and C community composition of sites at each distance were very similar. Faunal communities at distances 5 m and 15 m were dominated by polychaete worms. At 5 m, the polychaete *Ceratonereis sp.* contributes the most to similarity within community composition (contrib%= 8.9), followed by Neriedidae juvenile polychaete worms (contrib%= 8.76, average abundance = 2.01) and the polychaete *Perineries nuntia var. vallata* (contrib%= 8.45).

Faunal composition within groups at 30 m and 50 m were dominated by the gastropod *Zeacumantus lutlentus*, followed by polychaete worms. Faunal composition within groups at 100 m, 200 m and 300 m were dominated by the polychaete *Scoloplos cylindrifer*, followed by the gastropod *Zeacumantus lutulentus* and other polychaete worm species. Faunal composition within groups 400 m, 500 m and 600 m were dominated by the gastropod *Zeacumantus lutulentus*, followed by the polychaete *Scoloplos cylindrifer* and *Nereididae* juvenile polychaetes. Community assemblages of neighbouring distances across sites (for example, between groups at 5 m and 15 m) seem to share dominant fauna.

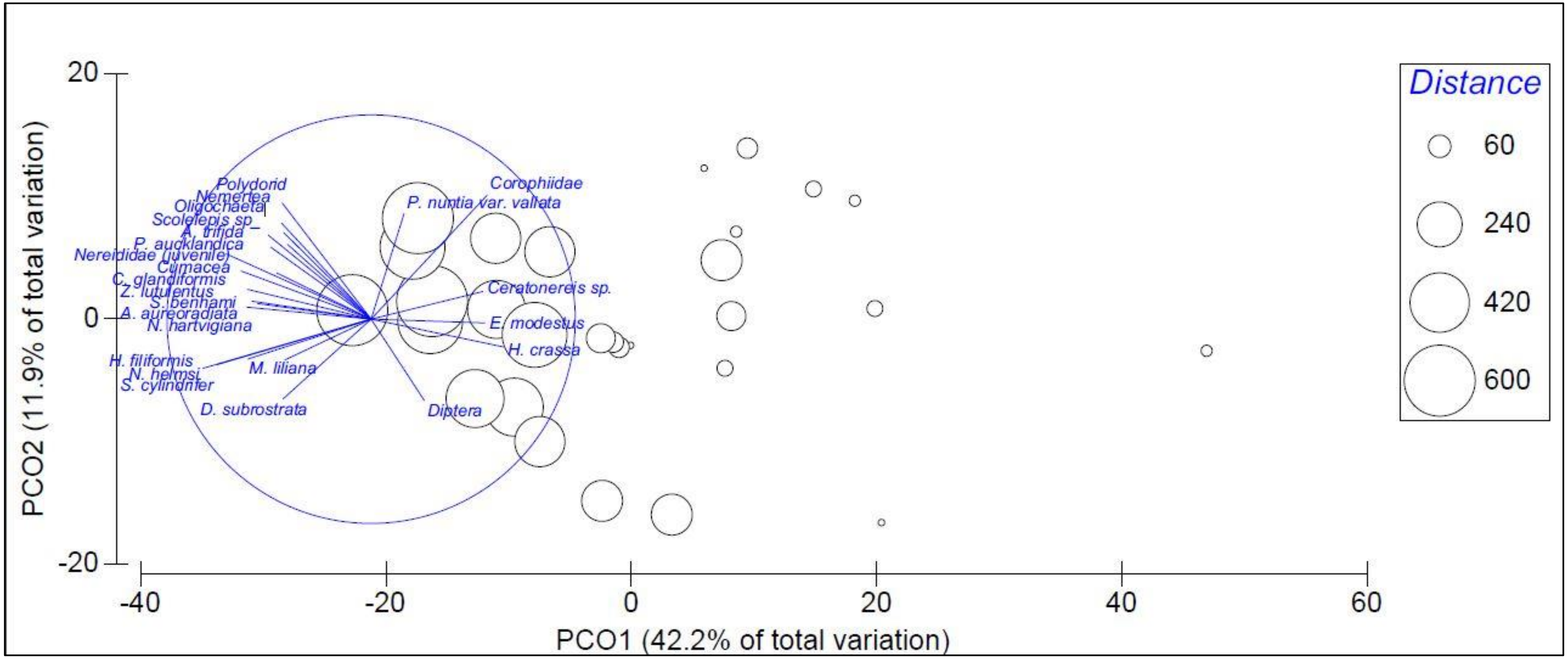

Figure 3.14 Two-dimensional PCO ordination (based on Bray-Curtis distances of fourth root transformed macrofaunal community data), with circle size indicating distance (m) from the WWT ponds. Plot is overlaid with associated vector plots of correlated taxa (Pearson $\rho > 0.4$). Percentage values indicate the percentage of variation in the resemblance matrix explained by each axis.

Table 3.4: Average dissimilarities between all groups (grouped by distance from shoreline in metres) based on Bray-Curtis dissimilarities of fourth root transformed macrofaunal data. All dissimilarity percentages over 40% are highlighted.

Groups (distances in m) compared	**Average dissimilarity (%)**	**Groups (distances in m) compared**	**Average dissimilarity (%)**	**Groups (distances in m) compared**	**Average dissimilarity (%)**
5 & 15	39.17	**5 & 300**	35.21	**30 & 500**	35.56
5 & 30	29.79	**15 & 300**	41.39	**50 & 500**	29.75
15 & 30	28.59	**30 & 300**	30.9	**100 & 500**	26.23
5 & 50	29.65	**50 & 300**	27.62	**200 & 500**	29.44
15 & 50	36.58	**100 & 300**	24.16	**300 & 500**	22.17
30 & 50	25.06	**200 & 300**	28.47	**400 & 500**	22.73
5 & 100	27.15	**5 & 400**	32.4	**5 & 600**	39.01
15 & 100	35.6	**15 & 400**	43.83	**15 & 600**	51.46
30 & 100	23.86	**30 & 400**	32.78	**30 & 600**	41.39
50 & 100	22.02	**50 & 400**	27.09	**50 & 600**	32.59
5 & 200	30.09	**100 & 400**	22.36	**100 & 600**	30.28
15 & 200	37.61	**200 & 400**	24.21	**200 & 600**	34.3
30 & 200	28.83	**300 & 400**	23.35	**300 & 600**	26.99
50 & 200	27.55	**5 & 500**	35.15	**400 & 600**	26.48
100 & 200	23.98	**15 & 500**	45.77	**500 & 600**	21.5

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Figure 3.4 is a summary of all dissimilarity percentages between each group;- dissimilarities greater than 40% are highlighted. Higher dissimilarity percentages between groups show that the two groups have distinctly different macro-faunal assemblages from each other. Macro-faunal assemblages at the 15 m sites showed the most dissimilarity to other sites, particularly those further from the WWT ponds (i.e. 300–600 m). Key species contributing to the observed differences between highly dissimilar groups (> 40% dissimilarity) can be found in Table 3.5.

Table 3.5 Breakdown of average dissimilarities between groups found to have an average dissimilarity higher than 40%. Species are ordered in decreasing contribution (top 40% only).

Groups 15 & 300 **Average dissimilarity = 41.39** Species	**Group 15 Av.Abund**	**Group 300 Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
Macomona liliana	0.00	1.70	2.96	3.33	7.16	7.16
Corophiidae	2.00	0.40	2.85	1.45	6.90	14.06
Scoloplos cylindrifer	1.60	3.10	2.82	1.04	6.82	20.88
Scolelepis sp.	0.69	1.82	2.38	1.52	5.75	26.63
Oligochaeta	0.76	1.78	2.38	1.92	5.74	32.37
Amphipoda indet_	0.00	1.35	2.27	5.12	5.49	37.86

Species	Group A Av.Abund	Group B Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Groups 15 & 400						
Average dissimilarity = 43.83						
Species	**Group 15 Av.Abund**	**Group 400 Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
Corophiidae	2.00	0.00	3.63	2.06	8.28	8.28
Macomona liliana	0.00	1.69	3.11	4.76	7.10	15.39
Scoloplos cylindrifer	1.60	2.95	2.93	1.06	6.69	22.07
Ceratonereis sp.	2.27	0.77	2.70	2.49	6.16	28.23
Scolelepis sp.	0.69	1.77	2.49	1.69	5.67	33.90
Diloma subrostrata	0.00	1.25	2.29	4.82	5.21	39.12
Groups 15 & 500						
Average dissimilarity = 45.77						
Species	**Group 15 Av.Abund**	**Group 500 Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
Macomona liliana	0.00	2.29	3.71	6.70	8.10	8.10
Amphipoda indet_	0.00	1.90	3.07	3.17	6.70	14.81
Nucula hartvigiana	0.00	1.90	3.05	3.45	6.65	21.46
Heteromastus filiformis	0.00	1.72	2.77	5.02	6.06	27.52
Corophiidae	2.00	0.44	2.57	1.51	5.62	33.14
Scoloplos cylindrifer	1.60	2.98	2.52	1.05	5.50	38.64
Oligochaeta	0.76	1.37	2.17	1.19	4.74	43.37
Groups 15 & 600						
Average dissimilarity = 51.46						
Species	**Group 15 Av.Abund**	**Group 600 Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
Macomona liliana	0.00	2.59	3.99	6.87	7.75	7.75
Nucula hartvigiana	0.00	2.38	3.52	2.28	6.84	14.59
Amphipoda indet_	0.00	2.01	3.03	3.53	5.88	20.47
Heteromastus filiformis	0.00	1.85	2.72	1.31	5.28	25.75
Scolelepis sp.	0.69	2.27	2.64	1.53	5.12	30.88
Corophiidae	2.00	0.33	2.59	1.46	5.03	35.90
Diloma subrostrata	0.00	1.55	2.42	3.65	4.70	40.60
Groups 30 & 600						
Average dissimilarity = 41.39						
Species	**Group 30 Av.Abund**	**Group 600 Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
Nucula hartvigiana	0.00	2.38	3.19	2.31	7.70	7.70
Macomona liliana	0.33	2.59	3.12	4.43	7.54	15.24
Corophiidae	2.10	0.33	2.74	1.40	6.62	21.86
Heteromastus filiformis	0.44	1.85	2.29	1.51	5.52	27.38
Amphipoda indet_	0.40	2.01	2.21	1.91	5.33	32.71
Cumacea	0.00	1.52	2.09	24.44	5.06	37.77
Ceratonereis sp.	2.49	1.03	1.95	1.89	4.71	42.48

From Table 3.4, the bivalves *Macamona liliana*, *Nucula hartvigiana* and the amphipod species Corophiidae appear to drive differences in community composition between groups. At a distance of 15 m, *Macamona liliana* or *Nucula hartivigiana* are not present (*see* Appendix 3). Between groups 5 m and 600 m, the dissimilarity percentage is also relatively high (39.01%), with *Macamona* *liliana* and *Nucula hartvigiana* also driving differences in community composition (contribution % of 9.22 and 8.18 respectively).

Interestingly, group 5 and 15 also appeared to have a relatively high average dissimilarity (39.17). The dominant taxa responsible for differences between the two distances was found to be Amphipoda (contribution % = 8.98), followed by Corophiidae, a species of amphipod (contribution % = 7.73). Amphipods were found to have particularly high counts within a few replicates in a site, which would account for the dissimilarities found between the two distances. For example, site 2 on transect B (at 15 m (*see* Figure 2.1 for site location)) was found to have 129 Corophiidae individuals, from only two of the 10 replicates within the site, while site 1 on transect B (at 5 m) had zero.

The second SIMPER analysis undertaken grouped sites at distances 5 and 15 m together to form a group labelled 'close' and sites at distances 500 and 600 m to form a group labelled 'far'. From table 4.5 the polychaete worms *Ceratonereis sp. and Perineries nuntia var. vallata* appear to be the dominant species contributing towards community composition closer to the WWT ponds and shoreline, followed by the gastropod *Zeacumantus lutulentus.* Within the group 'far' *Zeacumantus lutulentus* and the spionid polychaete *Scoloplos cylindrifer* are the most influential to community composition.

Table 3.6 Breakdown of average similarity of faunal assemblages within groups (the group "close" is comprised of sites at distances 5 and 15 m from the shoreline, the group "far" is comprised of sites at distances 500 m and 600 m from the shoreline). Analysis is based on Bray-Curtis dissimilarities of fourth root macro-faunal data. Species are listed in order of decreasing contributions (top 40% only).

Group Close (Sites at 5 & 15m) **Average similarity: 62.43** **Species**	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**	**Cum.%**
Ceratonereis sp.	2.11	7.44	5.42	11.91	11.91
Perineries nuntia var. vallata	2.10	7.01	6.19	11.23	23.14
Zeacumantus lutulentus	2.17	6.33	4.24	10.14	33.29
Scolecolepides benhami	1.93	6.08	5.97	9.73	43.02
Nereididae (juvenile)	1.90	5.94	8.60	9.51	52.53
Helice crassa	1.62	5.22	3.97	8.35	60.88
Scoloplos cylindrifer	1.74	3.72	1.28	5.97	66.85
Cominella glandiformis	1.33	3.28	1.34	5.25	72.10
Halicarcinus varius	1.13	3.07	1.36	4.91	77.01
Nicon aestuariensis	1.08	2.55	1.28	4.09	81.10

Group Far (Sites at 500 & 600m) **Average similarity: 77.96** **Species**	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**	**Cum.%**
Zeacumantus lutulentus	2.98	7.04	6.34	9.03	9.03
Scoloplos cylindrifer	2.83	6.64	7.77	8.51	17.54
Nereididae (juvenile)	2.67	6.09	10.57	7.81	25.35
Macomona liliana	2.44	5.71	10.48	7.33	32.68
Scolecolepides benhami	2.26	5.23	12.18	6.71	39.40
Perineries nuntia var. vallata	1.88	4.42	12.77	5.67	45.07
Scolelepis sp.	1.94	4.07	7.83	5.23	50.29
Amphipoda indet_	1.96	3.88	3.41	4.98	55.27
Polydorid	1.99	3.86	3.34	4.95	60.22
Nucula hartvigiana	2.14	3.74	3.10	4.79	65.01
Cominella glandiformis	1.66	3.68	5.33	4.72	69.74
Halicarcinus varius	1.27	2.80	6.91	3.59	73.33
Heteromastus filiformis	1.78	2.80	1.31	3.59	76.92
Anthopleura aureoradiata	1.39	2.77	6.83	3.55	80.47

As with the SIMPER by distance, it was found that the bivalves *Macomona liliana* and *Nucula hartvigiana* were the dominant fauna responsible for distinguishing the close sites from the far sites, with highest abundances found at a distance of 600 m (*M. liliana* $\bar{x}$ = 45.6, *N. hartvigiana* $\bar{x}$= 75.3, across three transects (n=3)). The capitellid worm *Heteromastus filiformis* was found to have higher abundances within sites far away, while the polychaete *Ceratonereis* sp. had higher average abundances within sites closer.

Table 3.7 Breakdown of average dissimilarity of faunal assemblages between groups close (sites at 5 and 15 m) and far (500 and 600 m). Analysis is based on Bray-Curtis dissimilarities of fourth root macrofaunal data (average abundances are fourth root transformed). Species are listed in order of decreasing contributions (top 40% only).

Groups Close & Far **Average dissimilarity = 42.85** **Species**	**Group Close Av.Abund**	**Group Far Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
Macomona liliana	0.00	2.44	3.66	7.21	8.53	8.53
Nucula hartvigiana	0.00	2.14	3.12	2.67	7.29	15.83
Heteromastus filiformis	0.28	1.78	2.39	1.63	5.57	21.39
Amphipoda indet_	0.92	1.96	1.92	1.38	4.48	25.88
Scoloplos cylindrifer	1.74	2.83	1.89	1.11	4.42	30.30
Oligochaeta	0.66	1.38	1.85	1.46	4.31	34.61
Corophiidae	1.32	0.39	1.82	1.13	4.26	38.86
Cumacea	0.20	1.33	1.78	1.78	4.15	43.02

3.3.3 Environmental data

The relationships among measured environmental variables were examined to identify those which were closely and consistently related and could, therefore, act as surrogates. The Draftsman Plot of Pearson correlations for normalised sediment data (Table 3.8) identified a number of relationships among environmental variables.

- Organic matter (AFDW) was strongly ($r > 0.8$) positively correlated with Pb and TP and to a slightly lesser degree, positively correlated ($r > 0.7$) with Cu, Zn, TN and silt and clay (< 63 µm).
- Pb was strongly positively correlated with TP, Zn and silt and clay ($r > 0.8$), as well as being positively correlated with TN ($r > 0.7$).
- TP was strongly positively correlated with TN ($r > 0.8$), as well as being positively correlated to Cu ($r > 0.7$).

It is important to note that any correlations with As, Cu and total TN may be partially confounded due to the assigned values to samples that fell below detection limits (As and Cu= 0.5, TN = 0.05 at some sites). Correlating continuous variables that have an arbitrarily assigned number at the lower end may reduce the intensity of the relationship (Robertson, *et al.*, 2002).

Table 3.8 Draftsman Plot containing Pearson correlations of normalised sediment data (n = 30) from ecological survey, Te Tāhuna o Rangataua. Sediment variables include OM (organic matter), As (arsenic), Cu (copper), Pb (lead), TP (total phosphorus), Zn (zinc), TN (total nitrogen), chl –α, gravel, very course sand, course sand, medium sand, fine sand, very fine sand and silt and clay (mud content).

	OM	**As**	**Cu**	**Pb**	**TP**	**Zn**	**TN**	**Chl-a**	**Gravel**	**V.C. sand**	**C. sand**	**M. sand**	**F. sand**	**V.F. sand**	**Silt & clay**
OM															
As	0.487														
Cu	0.782	0.285													
Pb	0.952	0.634	0.703												
TP	0.858	0.207	0.731	0.828											
Zn	0.771	0.767	0.454	0.817	0.514										
TN	0.774	0.148	0.575	0.726	0.832	0.539									
Chl-a	0.014	-0.367	0.053	0.011	0.349	-0.27	0.346								
Gravel	-0.101	-0.071	-0.076	-0.007	-0.036	-0.085	0.068	0.455							
V.C. sand	0.049	0.237	0.033	0.199	0.094	0.059	0.085	0.485	0.764						
C. sand	-0.067	0.487	-0.07	0.116	-0.137	0.048	-0.211	-0.021	0.221	0.653					
M. sand	-0.399	-0.255	-0.241	-0.414	-0.385	-0.512	-0.309	-0.032	0.185	0.187	0.408				
F. sand	-0.344	-0.832	-0.169	-0.5	-0.06	-0.594	-0.115	0.349	-0.107	-0.401	-0.707	-0.159			
V.F. sand	0.381	0.689	0.211	0.455	0.119	0.71	0.136	-0.413	-0.293	-0.217	-0.083	-0.696	-0.431		
Silt & clay	0.799	0.652	0.454	0.829	0.595	0.91	0.627	-0.288	-0.19	-0.072	-0.05	-0.52	-0.52	0.642	
Distance	-0.029	0.138	-0.167	-0.061	-0.367	0.186	-0.062	-0.569	-0.144	-0.412	-0.226	0.083	-0.234	0.363	0.183

3.3.4 Principal Co-ordinates Analysis (PCA) of environmental data

Environmental data was examined using PCA to explore the similarity of the sites based on physical and chemical properties (excluding macro-faunal data) and to determine which variables were most responsible for the differences among sites (Robertson, *et al.*, 2002).

The PCA plot generated from the normalised sediment data found that the PC1 axis (x-axis) explained 40.6% of the total variation and the PC2 axis (y-axis) explained 19.6% of the total variation (together explaining 60.2%), which suggests that the ordination plot described the data acceptably (Figure 3.15). The PCA showed a pattern of change along the PC2 axis corresponding with a gradient of distance (displayed with the bubble plot). The environmental variables largely responsible for variation among sites were chl-α and TP at distances closer to the impact sites and very fine sand influenced variation at distances further away.

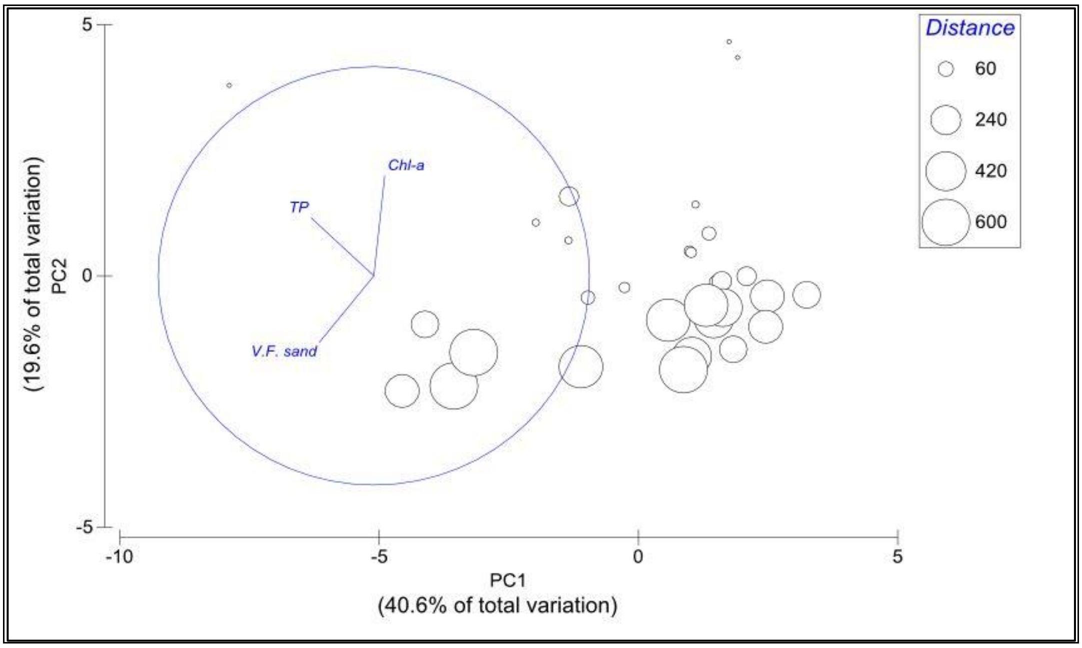

Figure 3.15 PCA of normalised sediment data against distance. Ordination is based on Euclidean distances and plot is overlaid with associated vector plots of correlated environmental variables (Pearson correlation > 0.4).

3.3.5 Distance based linear modelling

The relationship between the macrofaunal data and the biophysical environmental variables (including distance from the WWT ponds) was analysed using non-parametric multivariate regression (DISTLM). From the draftsman plot (Table 3.8). Pb was found to be highly correlated with other variables, including organic matter ($\rho = 0.95$) and was therefore excluded from the DISTLM analysis. As Pb correlates with many other variables, it may be omitted due to corresponding surrogates making it redundant. The variable that explained the greatest amount of variation in macrofaunal assemblages was distance (28.25%), the second variable was TP (16.9%), followed by TN (8.23%), fine sand (7.73%) (Table 3.9). The marginal test showed that many environmental variables were not statistically significant ($p > 0.05$) in terms of explaining variation in the community assemblages.

Table 3.9: Results of marginal tests of non-parametric multiple regression of multivariate species data on predictor variables for each variable taken individually (ignoring other variables). % Prop = percentage of variance in species data explained by that variable, P-value significant at <0.05. Significant values are highlighted.

Variable	SS(trace)	Pseudo-F	P	% Var.
OM	933.4	1.8559	0.094	6.22
As	924.85	1.8378	0.095	6.16
Cu	608.12	1.1818	0.251	4.05
Hg	2.2737E-13	4.2399E-16	1.000	1.51
TP	2538	5.6952	0.006	16.9
Zn	826.54	1.631	0.125	5.5
TN	1235.1	2.5095	0.033	8.23
Chl-a	2269.8	4.9864	0.003	1.5
Gravel	549.26	1.0631	0.263	3.66
V.C. sand	405.5	0.77713	0.573	2.7
C. sand	326.96	0.62326	0.703	2.18
M. sand	822.45	1.6225	0.128	5.48
F. sand	1160.1	2.3443	0.05	7.73
V.F. sand	1066.4	2.1406	0.084	7.1
Silt & clay	997.81	1.9931	0.081	6.65
Distance	4241.6	11.023	0.001	28.25

When placed in a model, variables may explain more variation in the species data than by themselves. This is due to the predictor variables themeselves being correlated and when building a model the overlapping of variables in their explanation of the species data must be taken into account (Anderson, *et al.*, 2004). When a model is built, the percentage of variation explained by a variable may be reduced. For example, total nitrogen appeared to explain a relatively high percentage of variance in the marginal results (8.23%, p = 0.033) but does not appear to be significant in the sequential model. From the draftsman plot (Table 4.7) it can be seen that total nitrogen is correlated with organic matter, total phosphorus and silt and clay.

Removing the effects of co-linearity between variables may reduce the percentage of variation explained by a variable but it may also increase it, adding significantly to the ability to explain variation within macrofaunal data (Anderson, *et al.*, 2004). Total phosporus appeared to explain a substantial amount of variability when observed alone (16.9%, p = 0.006), but when considered in a model, this was considerably reduced (8.95%, p = 0.015). Organic Matter appeared to be non-significant in the marginal tests, but added to the sequential model significantly (8.95%, p = 0.001). Although Chl α (1.5%, p = 0.003), and fine sand (7.73%, p = 0.05) appeared to explain some variation when taken individually (along with TN), they appeared to be insignificant in the combined model.

Table 3.10 Results of sequential tests of non-parametric multiple regression of species data on a set of predictor variables. The sequential model was built using step-wise selection of sets of variables (AICc selection), where percentage of variance explained by each variable added to the model is conditional on variables already in the model(Anderson, *et al.*, 2008). % Var = percentage of variation explained by a variable within the model, Cumul. Var (%) = cumulative percentage of variance explained.

Variable	**SS(trace)**	**Pseudo-F**	**P**	**% Var**	**Cumul.Var(%)**
+Distance	4241.6	11.023	0.001	28.25	28.25
+TP	1277.5	3.6322	0.015	8.51	36.76
+OM	1343.3	4.2838	0.001	8.95	45.7
+Gravel	781.58	2.6506	0.034	5.21	50.91

The results of the sequential analysis (Table 3.10) showed that a set of 4 variables had the greatest explanatory power, with 3 environmental variables (total phosphorus, organic matter and gravel (grain size >/= 2mm) and the predictor variable distance, together explaining 50.91% of the variance within the species data. The dbRDA plot (Figure 4.16) showed patterns similar to the PCO, with bubbles showing a pattern of change along a gradient of distance.

The plot was overlaid with predictor variables that contributed the most to variation, with distance and total phosphorus having the greatest contribution to variance. Distance influenced communities farther away while TP influenced communities closer. Gravel gravitates slightly toward the left on the ordination (along dbRDA1), appearing to influence variation found in communities further away. Organic matter appears to contribute less than the other variables but contributes greatly to the model as a whole. The DISTLM analysis indicates that these four predictor variables are all significant in terms of explaining variability among faunal communities within Te Tāhuna o Rangataua.

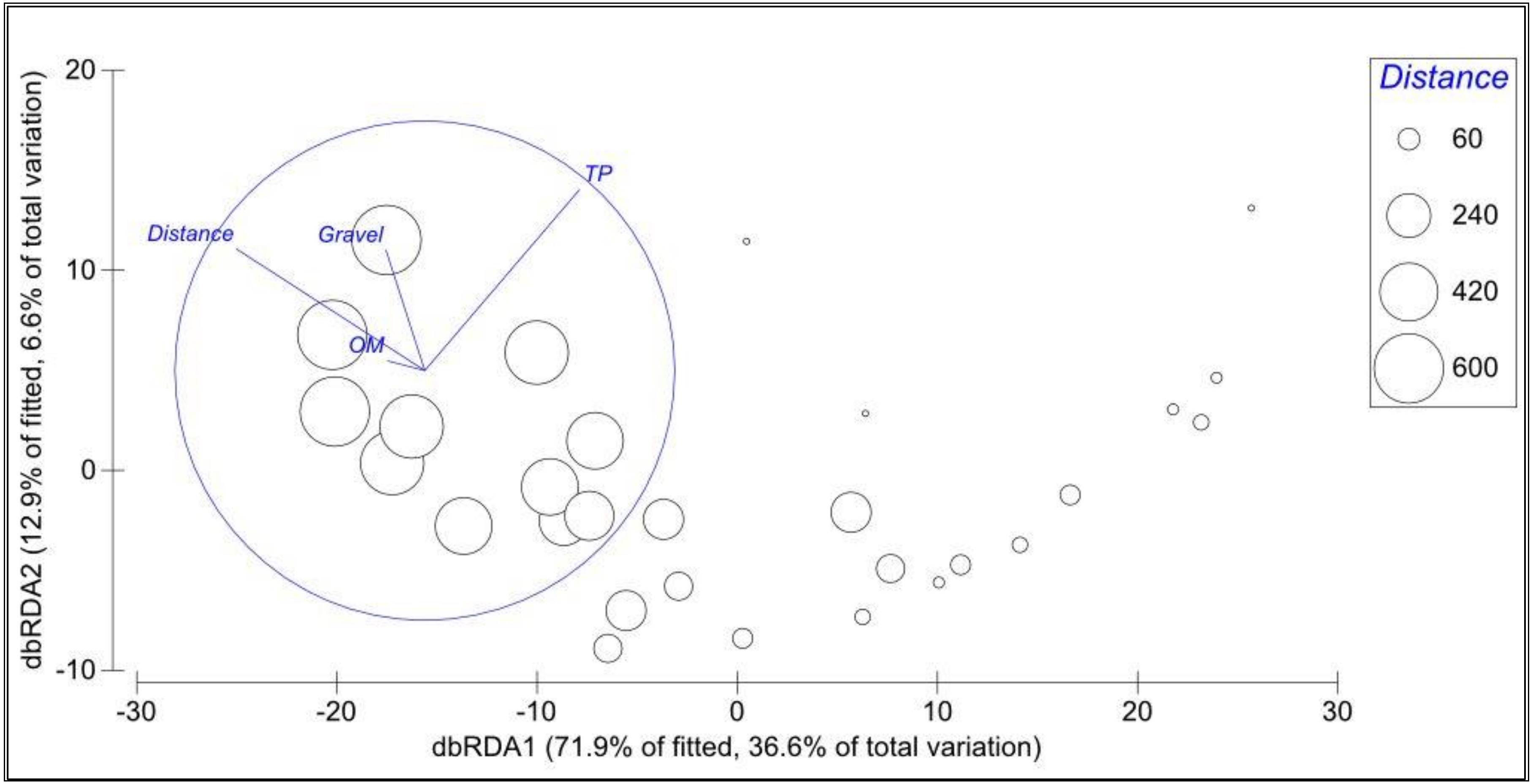

Figure 3.16 Distance-based redundancy analysis (dbRDA) constrained ordination relating predictor variables to species data. Analysis was performed on principal coordinate axes obtained from Bray-Curtis dissimilarities of fourth-root transformed species data against normalised environmental variables and the predictor variable distance. Plot is overlaid with key predictor variables identified from distance based linear modelling.

3.3.6 PCA of environmental variables for the Tauranga Harbour survey (Ellis, *et al.*, 2013) and Te Tāhuna o Rangataua.

Environmental data of 74 sites throughout the Tauranga Harbour and 30 sites within Rangataua Bay were examined using Principal Co-ordinates analysis, to determine sites that were similar or different based on environmental conditions. The PCA ordination generated from the environmental data for each site found that the PC1 axis explained 39.6% of the total variation among sites, while the PC2 axis explained 18.1% of the variation. From the associated vector plot (Figure 3.18), environmental variables, including heavy trace metals, nutrients and mud content showed a pattern consistent with the PC1 axis, indicating that these variables were responsible for explaining variation between sites, along this axis. Environmental variables, which showed a pattern along PC2 axis were the various sediment grain sizes and chlorophyll α, indicating these variables were largely responsible for variation between sites along this axis.

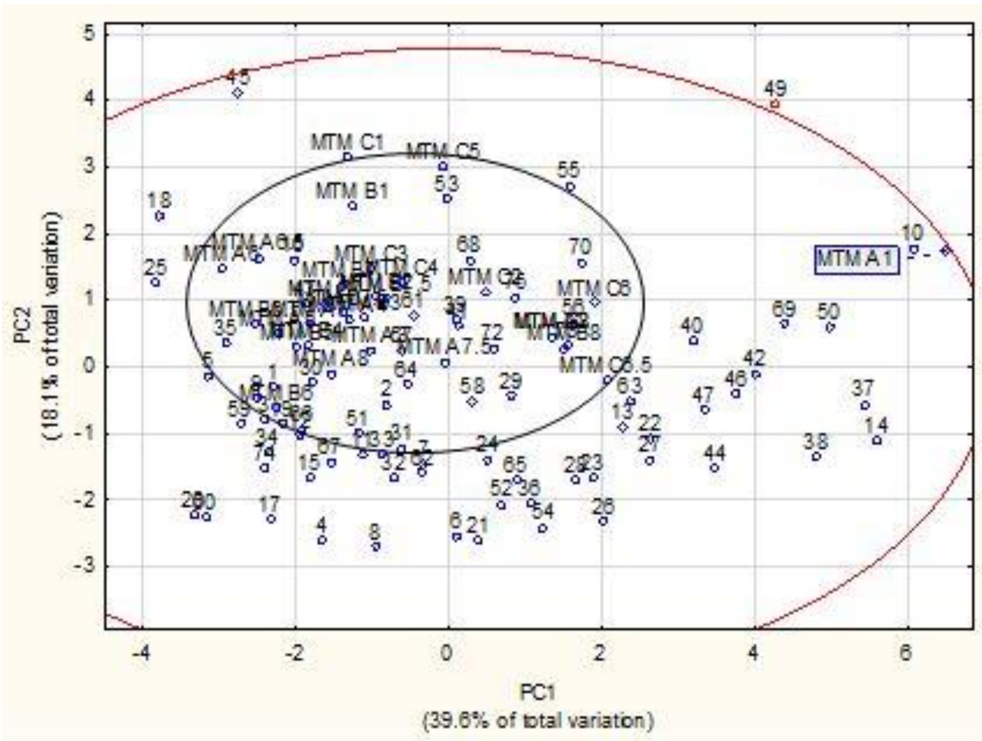

Figure 3.17: PCA ordination of environmental data from 74 sites within Tauranga Harbour (data from Ellis, *et al.* (2013)) and 30 sites from Te Tāhuna o Rangataua. Ordination is based on Euclidean distances. Te Tāhuna o Rangataua sites (labelled MTM) are circled, forming a cluster.

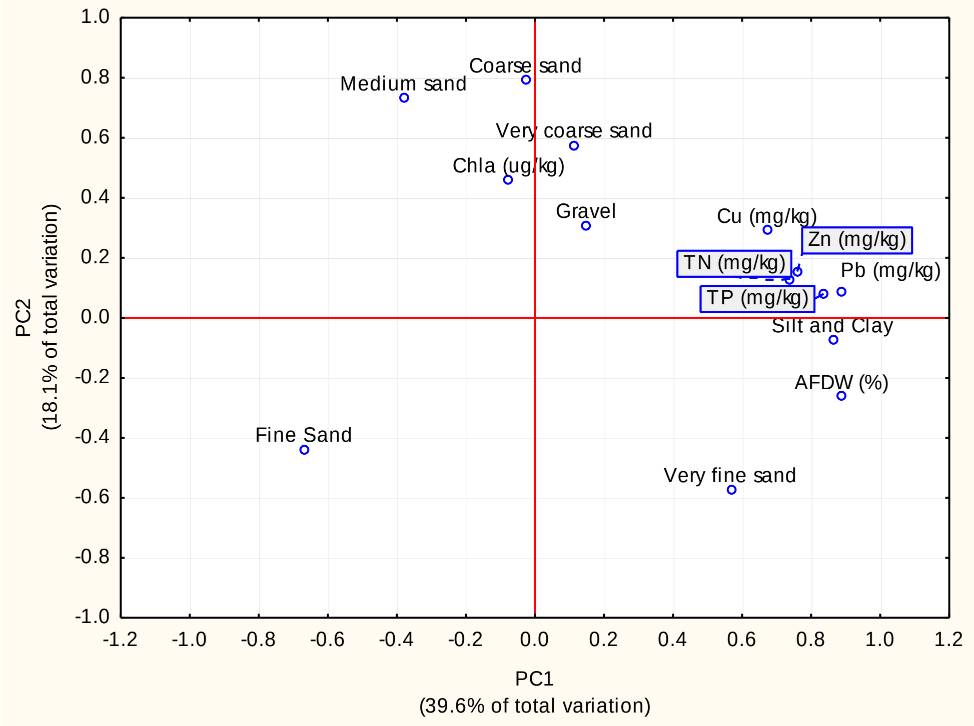

Figure 3.18: Associated vector plot (of Figure 4.17) of correlating environmental variables.

The Te Tāhuna o Rangataua sites within the PCA ordination form a subset which is separate from many other sites or estuaries within the Tauranga Harbour. Sites from the Rangataua area cluster together, apart from MTM A1, which was found to have higher values for nutrients, mud content and organic matter than other sites within the Rangataua area. Rangataua sites appear to follow a pattern consistent with the PC2 axis, indicating that chlorophyll α, medium sand, coarse sand, very coarse sand and gravel are the physico-chemical variables responsible for differences found between Te Tāhuna o Rangataua and the rest of the Harbour. This finding is consistent with previous analyses undertaken, in terms of highlighting chemico-physical variables of importance. The results are not surprising, considering the significantly high levels of chlorophyll α found within Rangataua Bay (12800–39100 µg/kg, $\bar{x}$= 23933 µg/kg) compared with the rest of the harbour (210–16000 µg/kg, $\bar{x}$=6078 µg/kg).

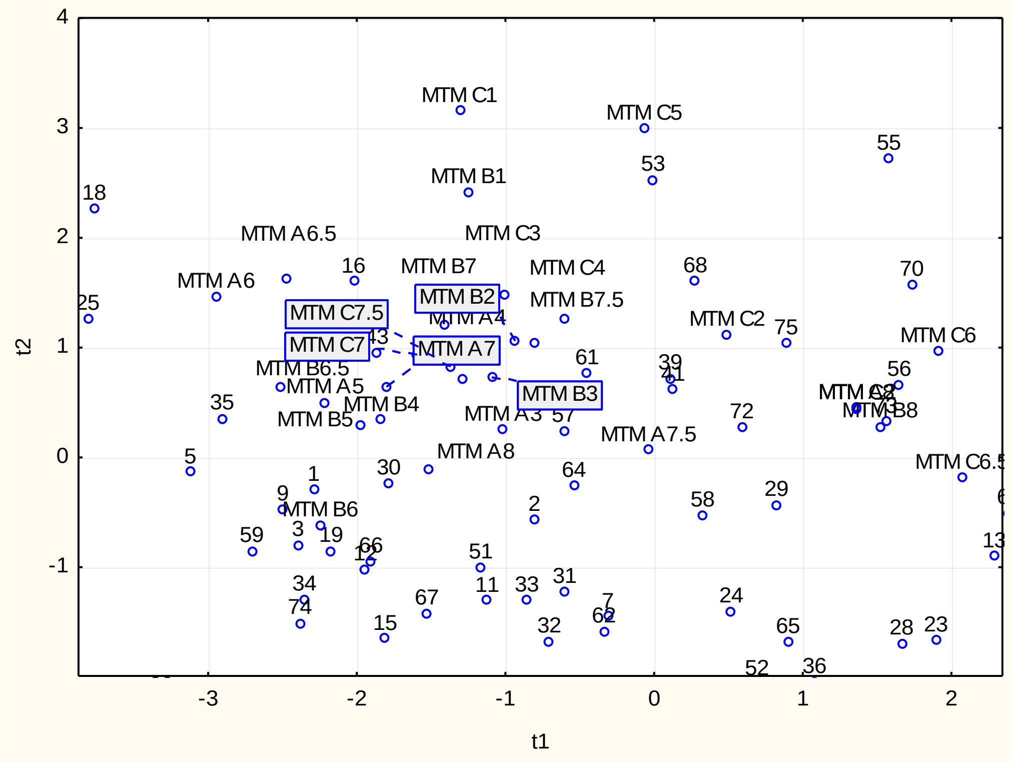

Figure 3.19: PCA ordination of environmental data from 74 sites within Tauranga Harbour (data from Ellis, *et al.* (2013)) and 30 sites from Rangataua Bay. Close up of Rangataua sites within ordination.

A few sites from the broad scale survey of Tauranga Harbour are found clustering within sites from Te Tāhuna o Rangataua, indicating that these areas have similar environmental conditions. These sites included; Site 16 (Bowentown flood delta), 39 (Ngakautuakina Pt), 41 (Omokoroa), 43 (Matakana Pt), 56 (Wairoa Est), 61 (Waikareao Entrance), 72 (Rangataua), and 75 (Welcome Bay) (*see* Figure 1.7 for map of Tauranga Harbour sites). These estuaries all vary in geomorphological features, all with differing degrees of protection, some being enclosed and others open sand spits. From Table 3.11, environmental variables across sites within Tauranga do not seem to vary significantly, with the exception of chl-α.

Table 3.11: Comparison of Environmental Variables between 74 sites within the MTM Broad scale survey of Tauranga Harbour (Ellis, *et al.*, 2013) and 30 sites within the survey of Te Tāhuna o Rangataua.

Environmental Variables	Sites similar to Rangataua (2011/2012)			All other sites within Tauranga Harbour (2011/2012)			Rangataua Ecological survey (2013)		
	Min	Max	Average	Min	Max	Average	Min	Max	Average
AFDW (%)	1.6	3.5	2.513	0.89	4.5	2.773	1.09	4.9	1.96
Gravel %	0.1	6.4	1.375	0.1	14.6	1.746	0.05	4.6	0.59
Silt and Clay %	3.3	15.1	9.663	0.6	48.9	12.61	3.9	28.2	9.2
TN (mg/kg)	280	580	412.5	140	1000	466.6	200	1500	600
Cu (mg/kg)	0.05	1.6	0.581	0.05	3	0.588	0.5	4	0.53
Pb (mg/kg)	1.5	4.3	2.55	0.05	5.4	2.413	1.2	5.4	2.1
TP (mg/kg)	120	190	156.3	53	340	164	90	470	159.6
Zn (mg/kg)	11	35	19.38	0.05	55	17.01	5	31	13
Chla (ug/kg)	4300	15000	7938	210	16000	5891	12800	49100	23900

Chapter 4

Discussion

The following discussion chapters will be in two parts. Part I will discuss benthic invertebrates, their responses to environmental condition and use as biological indicators. Part II will discuss in more detail the biophysical components of the study and how they link to overall benthic community structure.

Discussion Part I: Macro-Faunal Community Structure

Change in benthic community structure in response to environmental variables can be assessed in a number of ways. Macro-faunal data can be examined to assess changes in total biomass, the abundance of functional groups and abundance of individual species (Pearson & Rosenberg, 1978). Macro-faunal data is commonly analysed by using diversity indices, similarity indices and multivariate analysis. Biological information obtained throughout many studies over the years has led to the designation of indicator species or groups which are believed to respond in a predictable manner to specific environmental conditions (Pearson & Rosenberg, 1978).

In an environment which is relatively stable, benthic communities will undergo some level of natural fluctuation. Earlier studies suggested that for communities experiencing prolonged stability and minor or small-scale disturbances, that species diversity will be high but that abundances of each taxa will be moderate (Pearson & Rosenberg, 1978). More recent studies have found that communities which show highest species richness are areas which undergo an intermediate amount of disturbance through time, a school of thought known as the intermediate-disturbance model (Connell, 1978; Nybakken & Bertness, 2005). Understanding the processes which lead to diversity is complex and a challenge as estuarine benthic communities are systems exposed to high natural biophysical variability (de Juan & Hewitt, 2014).

Community composition is defined by a group of reoccurring species. ‘Limiting factors’ vary for each species and so the boundaries between different communities, which are representative of change in various environmental factors, are usually transitional and not well defined. Different species will gradually drop out (in response to factors exceeding their tolerance levels) while other species come in (Nybakken & Bertness, 2005).

Each species has a level of tolerance to any and all environmental factors and if the level of tolerance for any one environmental factor is exceeded, then that species will no longer be present. In addition, each species requires a minimum amount of key resources to survive. If any of these resources are depleted below the minimum requirement then that taxa will in turn disappear or its population parameters will change (Nybakken & Bertness, 2005).

4.1 Intertidal Zonation

Community specific patterns from the upper shore to the lower shore may be a result of responses to environmental variables that are influenced by biophysical intertidal gradients (Read, 1984). Delineating cause and effect within estuarine studies is made difficult due to a multitude of different elements, one of which is habitat specificity for differing organisms.

Sediments within the intertidal area of an estuary are dewatered and rehydrated periodically by a number of processes including evaporation, ebb-tide drainage, flood-tide inundation, terrestrial inputs, and precipitation during wet weather events. These events influence biochemical fluxes within the sediment pore-waters and suspension of particles and sediments (Tay *et al.*, 2012).

4.1.1 Biotic responses to tidal elevation

Invertebrates that occupy the intertidal area experience increased periods of exposure to air with higher elevation on shore which can lead to corresponding physiological stress. Stress from exposure includes overheating or overcooling dependant on the season, desiccation, damage from sun radiation and osmotic shock after heavy rainfall (Peterson, 1991). Physiological tolerance of these stressors can lead to faunal zonation which is distinctly visible within rocky shore assemblages, with horizontal banding of different organisms along the face of rock surfaces. Within sedimentary estuarine environments however, tidal zonation is not as visibly evident or studied as robustly. Soft sediment invertebrate communities are often dominated by infauna and therefore not easily observed (Peterson, 1991).

Peterson (1991) studied tidal zonation of marine invertebrates in soft shore environments in respect to biological and physical processes that are known to cause zonation on the rocky shore. The biological and physical processes underpinning zonation on the rocky shore are quite different within soft-sediment environments. More gradual patterns of zonation may occur on soft-sediment shores, with factors such as abiotic stress, predation, disturbance and competition varying with elevation and influencing community structure.

In soft shore environments, animals live on or within the sediment. Compared with rocky shore invertebrates, they are less vulnerable to desiccation, heat exposure and stress related to cessation of feeding and aerobic respiration. The sediment may provide a buffer against abiotic stressors, as well as remaining moist during low tide, with pore water inhibiting change in temperature and salinity (Peterson, 1991). However, many infaunal animals are most likely to cease feeding and aerobic respiration during the low tide, which may show a pattern of inhibited growth with increased elevation (Peterson, 1991).

Competition is arguably not as significant in soft sediment environments, at least for smaller animals within the sediment, compared with rocky shores. Often, there is not a limited surface space for animals to occupy and densities of animals may also be kept low by various other factors and disturbances. The influence of competition to species distribution may, however, be relative to each distinct environment.

4.1.2 Bioturbators

In coarser sediments, pore water may evaporate quickly, while in muddier sediments very little evaporation may occur, retaining water in the sediments at low tide. The presence of mud burrows and mobile infauna (bioturbators), which disturb the sediments, may however decrease water retention of the sediments. Mud burrows and sediment disturbance increases water movement through the upper sediment layers during tidal inundation, which in turn distributes heat between the mud and water more quickly than by molecular diffusion. In this way, the presence of burrowing animals can reduce temperature stressors to infuanal animals by reducing the vertical gradient of temperature within the sediment (Harrison & Phizacklea, 1987).

Biotrubators, including deposit feeders and burrow builders, can alter the physico-chemical nature of sediments and this may have an influence in faunal zonation. Disturbance of the sediment by active burrowers and deposit feeders, which ingest sediment and cause high volumes of turnover of the sediment, may inhibit or promote colonization by other animals (Peterson, 1991). Constant disturbance may prevent tube building benthic animals (which includes many polychaete species) from establishing. Bioturbation may also decrease biological oxygen demand by allowing oxygen to penetrate lower sediment layers and thus promoting increased biological activity. Bioturbators are suggested to inhabit areas higher up the shoreline, as all of their activity may make them more susceptible to predator fish species (Peterson, 1991).

Biotrubators observed in the Te Tāhuna o Rangataua included the deposit feeder *Macomona liliana,* and the mud crab *Helice crassa,* which reworks the sediment when building borrows. At sites within this study, *H. crassa* decreased in abundance with distance from the shoreline (Figure 4.1). This may be attributed across-shore differences in exposure time allowing for feeding. Displacement from the lower shore may occur due to predation from animals in the water column. Increased silt and clay may account for higher abundances of *H. crassa* at the 5 m sampling distance.

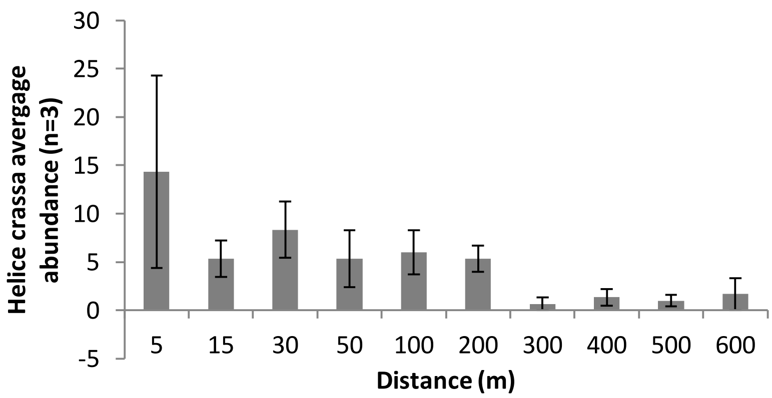

Figure 4.1: Average abundance (summed to site level) of *Helice crassa* across three transects (n=3), at each distance from the shoreline, with error bars (±SE), Te Tāhuna o Rangataua. Y axis = $\bar{x}$ of ten 300 cm^3 core samples.

4.1.3 Shorebird predation

During low tide, shorebirds are major predators in an estuarine environment. Many infuana lack protective shells and are easily preyed upon by differing species of shorebird. As well as this, many shorebirds have evolved different types of beaks and behaviours, allowing them to dig deeper into the sediment or penetrate the hard shells of molluscs. Benthic organisms which live higher on the shoreline are exposed for longer periods of time and are therefore more at risk of predation by shorebirds. Some shorebirds are migratory, exerting only a brief period of intense predation and therefore if zonation did occur, it would be seasonal (Peterson, 1991).

4.1.4 Recruitment

Larval settlement is an important biological process in colonisation of benthic assemblages. Settlement patterns are influenced by a myriad of factors, against the backdrop of an unpredictable, complex and fluid environment and it is therefore difficult to understand or predict. When considering hydrodynamic forces in propogule settlement, it is possible that larvae may passively settle where sediment deposition is also occurring. Sedentary soft-sediment invertebrates often have a period of mobility as juveniles, so settlement may not have as much influence on zonation (Peterson, 1991).

4.2 Species diversity and abundance

Environmental stress, in broad terms, is considered to negatively affect species abundance, diversity and richness (Stark, 1998), although when an area undergoes a disruption or disturbance, a rapid re-colonisation of opportunistic species may occur, leading to a spike in the abundance of these opportunistic taxa (Pearson & Rosenberg, 1978).

The trends found within the current survey show that abundance and richness have a slight increase with distance from the upper intertidal fringe, adjacent to the wastewater ponds, though species richness found at a distance of 5 m is greater than at 15 m and is characterised by polychaete worms and mud crabs (Figures 3.11 and 3.12). Total abundance appears to be lowest at a distance of 5 m. From this preliminary assessment it would appear that biological activity is somewhat limited closest to the treatment ponds. This could be due to environmental conditions exerting stress on local fauna, with certain species being unable to recruit to the upper intertidal area. Shannon-Weiner diversity did not show an observable trend over a gradient of distance (Figure 3.13).

When discussing results based on diversity indices, some scientists have expressed concerns about indices that summarize a large data set into a single number (Weisberg *et al.*, 1997). Compressing multi-dimensional community structure into a single number is considered an inadequate measure of a complex system (Thrush *et al.*, 2006). Pearson and Rosenberg (1978) highlight this when using diversity indices, such as the Shannon-Wiener function, the resulting co-efficients will not show structural changes when the numbers of species and individuals are low. This can often be the case when analysing benthic invertebrate data and in particular when benthic communities may be influenced by pollution. In saying this, measuring diversity indices are recommended to complement further analysis. Multivariate analysis of invertebrates was found in this study to be valuable in examining environmental effects.

4.3 Species as biological indicators

Macro-benthic communities impacted by watersheds with agricultural and urban land use are often associated with a reduction in species diversity, altered trophic interactions, reduced habitat availability and a decrease in overall productivity (Dauer *et al.*, 2000). Within estuarine environments, benthic communities which have undergone some form of degradation are dominated by small, opportunistic, rapidly growing species. These species typically have low functionality to ecosystem processes and therefore have little ecosystem value (Thrush, *et al.*, 2013). Even before a species is displaced from a habitat, functional changes to an ecosystem’s performance can take place. The loss of larger long-living infuana, can represent potential long term degradation in benthic condition (Thrush, *et al.*, 2006).

Estuarine macroinvertebrate assemblage character is a common tool utilized in environmental management, with predictive responses to different anthropogenic stressors being catalogued by a multitude of studies over the years. Tolerance values of different species have been developed using a variety of techniques including the use of ecotoxicology data, field based observations, knowledge of life history and best professional judgement. Species which make the most useful indicators will have narrow and specific environmental tolerances, which would give a clear benchmark of environmental condition (Pelletier *et al.*, 2010).

4.3.1 Characteristics of indicator species

Benthic invertebrates have many characteristics which make them suitable indicators of anthropogenic disturbance. Most important of these is direct exposure to contaminations and the diverse responses relative to the invertebrates' tolerances, feeding modes and trophic interactions (Pearson & Rosenberg, 1978; Weisberg, *et al.*, 1997). If an area is being polluted through anthropogenic activity, contaminants will often accumulate within the sediment in which benthic invertebrates live. Many animals living on or in the benthos are sedentary or move only short distances (millimetres to centimentres) and cannot evade adverse conditions, therefore receiving chronic exposure. Responses to adverse conditions are able to be measured over prolonged periods of time (Weisberg, *et al.*, 1997). Because of the animals' shorter life spans, they can show rapid responses to environmental changes that may otherwise be undetectable (Word, 1978).

Polychaete worm species are amongst the most diverse and abundant species within soft-sediment marine environments and are therefore an obvious choice to act as a representative species (group) for the assessment of the health of benthic communities (Dean, 2008). Their sedentary nature ensures they interact with the sediment and the water column of a particular area and experience exposure to toxic materials that may be present through time. The life cycle of most polychaetes can span from days to weeks, often producing high abundances of offspring, so changes in population structure as a response to environmental stress or pollution can be rapid and detectable (Dean, 2008).

The Principal Co-ordinate analysis (PCO, Figure 4.14) of macrofaunal data showed a change in community composition along a gradient of distance. Opportunistic taxa and scavengers such as neriedidae worms, amphipods and crabs were found to characterise composition at distances closest to the shoreline and a range of taxa including suspension/filter feeders, gastropods, bivalves and a variety of polychaete worm species, were dominant in lower tidal areas. This would suggest a change in environmental conditions influencing community structure, with tidal height and distances farther away from the WWT ponds having more favourable conditions which allow for animals of different feeding guilds and trophic levels to grow. An in depth examination of specific species and their abundances may give insight into which environmental variables are influencing distribution patterns.

4.3.2 Feeding guilds

To assess community structure, Word (1978) developed the Infaunal Trophic Index, believing that change in community composition in response to organic enrichment can be understood by examining the feeding mode of organisms present. Two feeding modes (suspension feeding and deposit feeding) were considered to be of most interest as the dominance of organisms using these feeding strategies would indicate the amount of particulate organic matter reaching the benthos.

(Word, 1978) suggested that when grouping organisms by feeding mode, uncertainties and mistakes may be reduced when identifying a broad range of taxa down to the species level. Indicator organisms were placed in one of four feeding groups (Group I, II, III and IV). Group I is dominated by suspension feeders, II by animals that feed on suspended particles or surficial detritus, III by surface detritus feeders and IV by animals that feed on detritus beneath the sediment surface. Group I are suggested to be present at reference sites with low Biological Oxygen Demand, group II are suggested to increase as BOD levels increase, Group III where BOD levels are increased further still and Group IV are suggested be indicator species of pollution and characterised by polychaete and oligochaete worm species. Group II and III species abundance is predicted to increase with proximity to wastewater outfalls but at a certain point, group II will begin to decrease. Group IV dominate impacted areas around outfalls, suggesting they are responding to high levels of organic matter or high concentrations of hydrogen sulphide, which is an intolerable condition for many other species (Word, 1978).

A Similarity Percentages analysis was undertaken which looked at macrofaunal assemblages within and between all sites (which were characterized by distance). The SIMPER analyses, along with the PCO analysis, showed a progressive change of faunal composition. Sites closest to the Wastewater ponds showed distinctly different dominance and composition of animals (Table 4.5), characterised by nereid polychaetes, *Ceratonereis* sp., *Perineries nuntia var. vallata* and Neriedidae juveniles; the polychaete *Scolecolepides benhami*; the opportunistic mud crab *Helice crassa;* and Corophiidae amphipods. The feeding modes of these animals include predator and deposit feeding scavengers (Table 4.1), feeding on detritus within or on the sediment surface, with infauna falling within Groups III and IV in the Infaunal Trophic Index of Word (1978), suggesting these animals may be present in response to organic pollution.

Species which appear to become more abundant at intermediate distances include the gastropod *Zeacumantus lutulentus*, the polychaete *Scoloplos cylindrifer* and Nereididae juvenile worms.

From the SIMPER analysis of close (characterised by distances at 5 and 15 m) vs far (distances 500 and 600 m) groups, ten species make up 80% of community contribution closest to the wastewater ponds, with feeding modes of the organisms including predators, scavengers, one grazer and deposit feeders(Table 4.5). Fourteen species contribute to 80% of the community composition at sites farthest away, with more diverse feeding modes amongst the taxa, including grazers, sub-surface deposit feeders, predator/scavengers, suspension feeders, deposit feeders, filter feeders or organisms that switch between these feeding strategies. The surface deposit feeding bivalves *M. liliana* and *N. hartvigiana* are responsible for the greatest difference in communities between sites close and far (Table 4.6), both bivalves only being found at distances further away. At a reference site, or sight which receives no anthropogenic inputs, there would be a more diverse range of feeding groups than at an impacted site (Word, 1978) and this trend appears to be reflected between sites close and far. When sediments become increasingly heterogeneous, this accommodates for different species resource requirements and communities may become more diverse, compared to environments which are homogenous (Pearson & Rosenberg, 1978)

Maurer *et al.* (1999) evaluated the Infaunal Trophic Index (ITI), believing the Index to have limitations, which would have implications when utilized for management purposes. Limitations included the Index’s sensitivity to abiotic factors (water depth, sediment organic matter and particle size), biotic factors and problems with allocating taxa to feeding groups. Maurer, *et al.* (1999) states that the ITI only allows for a single mode of feeding per taxa and that many species, in particular opportunistic species, are able to change their mode of feeding dependent on environmental conditions, which may lead to inaccurate allocation of organisms to feeding groups.

The common occurrence of particular species over a wide geographical range of polluted areas has led biologists to regard them as indicators. Pearson and Rosenberg (1978) highlight that within the wide range of marine environments, such species occurrences may be found in healthy areas just as much as they are found in impacted areas. Within organically enriched areas, capitellid polychaetes commonly occur, along with certain spionid polychaetes and gastropod molluscs, this leading them to be identified as indicators of organic enrichment. The use of groups of species to characterize a degree of pollution and relating these communities to environmental condition is considered much more useful than the use of a single species. Nereid polychaetes are often found as dominants in early successional phases of organically enriched areas, though species can vary between different geographical areas.

4.3.3 Suspension/filter feeder distribution

Few suspension/filter feeding taxa were found within Te Tāhuna o Rangataua. Those recorded included *Anthopleura aureoradiata, Austrovenus stutchburyi, Edwardsia* sp. and *Eliminus modestus* (Figure 4.1 of all taxa, Figure 5.2).

Of the filter and suspension feeders found within Te Tāhuna o Rangataua, slight increase in distribution lower down the shoreline may be attributed to longer exposure time and pressure from increased feeding cessation periods higher up.

Suspension feeders were rare across the sampled sites (Figure 5.2). Any number of environmental variables could contribute to species absences within the intertidal area. Consistent fine silt sedimentation over the years may have negatively affected filter feeding organisms, with populations no longer recruiting to the area, although abundances of *A. aureoradiata*, a suspension feeding anemone, are highest in areas with high mud content (Figure 4.2), which is inconsistent with this theory. The increase in *A. aureoradiata* at 600 m may be a response to a slight increase in organic matter at 600 m (Figure 3.1). As well as this, the estuarine anemones distribution is no doubt limited by the distribution of bivalves, as *A. aureoradiata* requires some sort of substrate with which to attach and grow and is often found growing on cockles.

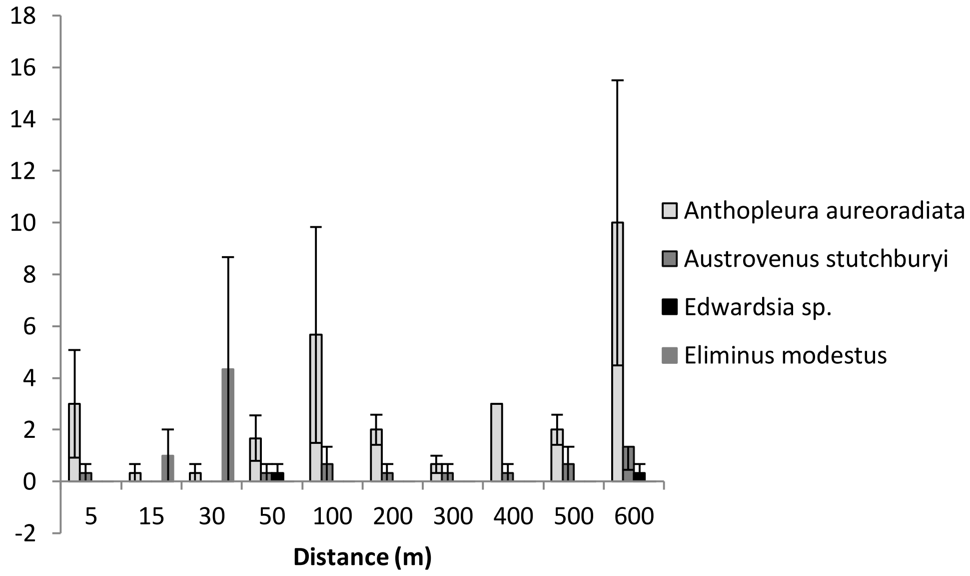

Figure 4.2: Average abundances (n=3, summed to site level), across three transects of suspension/filter feeders found in Te Tāhuna o Rangataua at distances from the shoreline, with error bars (±SE). Y axis = $\bar{x}$ of ten 300 cm^3 core samples.

Silt and clay suspended in the water column can also be detrimental to suspension feeders, with suspended particles directly effecting animals by clogging feeding structures, interfering with particle selection and requiring active movement of unwanted particles and reducing energy sources (Thrush *et al.*, 2004).

4.4 Species response to organic matter

Heteromastus filiformis is an opportunistic capitellid polychaete species which is known to grow in higher numbers within anaerobic sediments, a condition of areas that are high in organic content within the sediment. Reduced oxygen supply may be the most serious consequence to organisms as a result of organic enrichment (Pearson & Rosenberg, 1978). Within Te Tāhuna o Rangataua, *H. filiformis* was present occasionally, with a few organisms found sporadically between 5 m and 500 m and a high abundance of organisms found at 600 m.

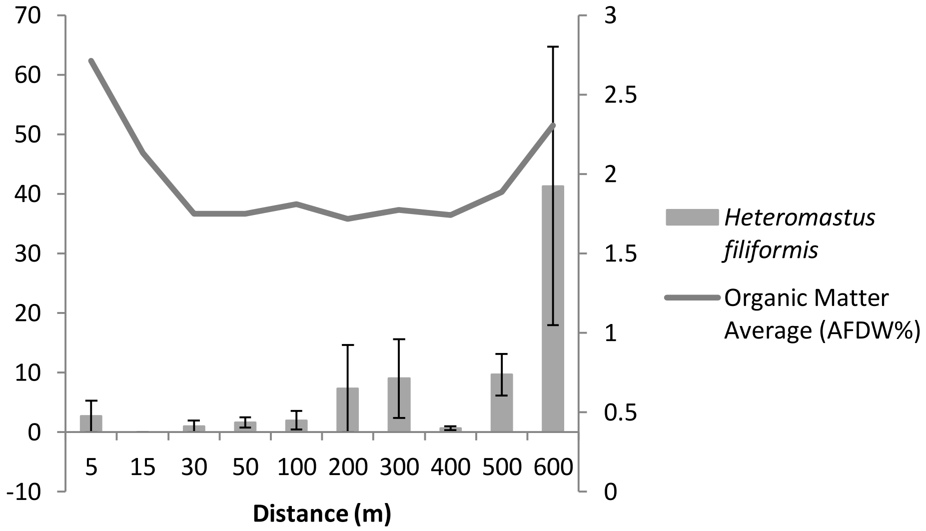

Figure 4.3: Average abundance (n=3, summed to site level) across three transects of *H. filiformis* (±SE), and average organic matter (n=3, see Fig. 4.1 for ±SE), at each distance from the shoreline, Te Tāhuna o Rangataua. Y axis = $\bar{x}$ of ten 300 cm^3 core samples.

A study of infaunal zonation along a tidal gradient by Read (1984) found that *H. filiformis* only occurred on the lower shore, suggesting they are subtidal species and higher abundances are a response to longer emersion periods and less environmental variability occurring than higher up the shore. This could account for *H. filiformis* increased abundance 600 m from the shoreline. Abundance increase may also reflect slightly elevated organic enrichment (Figure 3.1). Read (1984) also noted that *H. filifomis* was strongly positively correlated with organic detritus and the polychaete has been found to be associated with organic enrichment and anaerobic sediments.

Pearson and Rosenberg (1978) found that trophic groups within communities tend to change along a gradient of organic enrichment and there is a decrease in suspension feeders and increase in deposit feeders as organic content within the sediment increases. This is suggested to be, in part, due to the physical clogging of ciliary and siphonal mechanisms of suspension feeders, which can occur from inputs of suspended particulate organic matter and sediment instability, leading to reduced populations. As noted previously, only a few suspension/feeders were found in the area (Figure 4.2). In areas affected by eutrophic conditions community composition is expected to be made up of almost only detritus feeders.

Pearson and Rosenberg (1978) concluded that a small group of polychaetes which included *Capitella capitata* and *Scolelepsis fuliginosa* were associated with organically enriched areas, referring to them as enrichment opportunists. These species would readily recruit to areas in which environmental conditions had changed due to inputs of organic matter and populations would quickly grow due to lack of competition.

Fine scale estuarine monitoring undertaken in New Zealand (Robertson & Stevens, 2010, 2011) has assessed organic enrichment levels and corresponding invertebrates. Species that were found to be indifferent to organic enrichment included *Boccardia*, *Austrovenus stutchburyi* and *Macomona liliana*. Species tolerant to increased organic enrichment included Nemertea, *Perinereis vallata*, *Nicon aestuariensis*, *Scolecolepedis benhami* and *Paracorophium excavatum*. *Heteromastus filiformis*, oligochaetes and *polydora* sp. were considered opportunistic, thriving in enriched conditions.

Within Te Tāhuna o Rangataua, *Perinereis vallata, Scolecolepedis benhami and Ceratonereis* sp. (nereid worm), where found to characterize sites closest to the WWT plant. The presence of these organisms may be a reflection of slightly elevated organic content in the upper intertidal fringe.

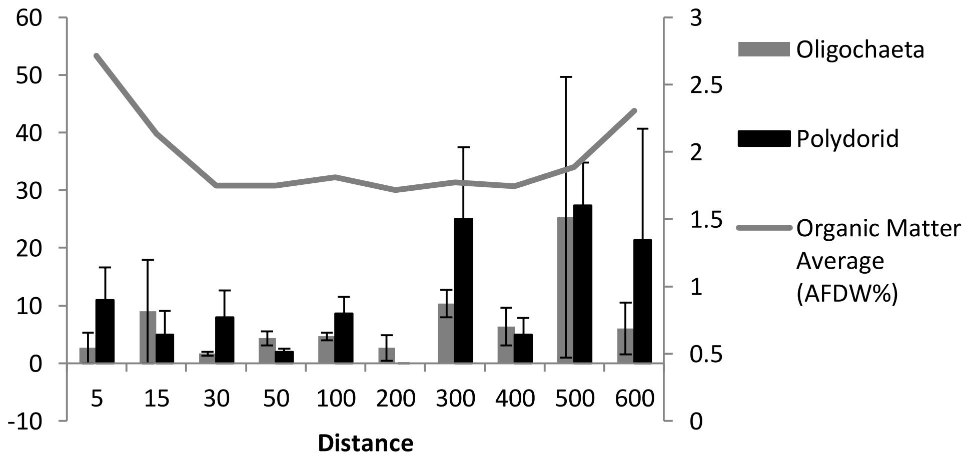

Figure 4.4: Average abundance n=3, summed to site level) across three transects of Oligochaeta and Polydorid sp. (±SE), and average organic matter (n=3, see Fig. 4.1 for ±SE), at each distance from the shoreline, Te Tāhuna o Rangataua. Y axis = $\bar{x}$ of ten 300 cm^3 core samples.

Within this study the composition of the invertebrate community reflected either low or elevated numbers of species that are sensitive to organic enrichment. From the figure (Figure 4.4) it can be seen that neither oligochaetes, nor polydorid abundance appears to directly reflect the trend found for organic matter content, though abundance could be loosely affiliated.

4.5 Wastewater seepages and salinity tolerance

Estuarine organisms experience a wide range of salinities. Fluctuation or reduced salinity is an added stress to estuarine fauna, with differing taxa tolerant to salinity concentrations (Pearson & Rosenberg, 1978). Stenohaline marine animals are organisms that are unable to tolerate changes in salinity and within estuarine environments are usually only found within the low tidal area closest to an inlets entrance, where salinity is higher. Euryhaline marine animals are able to tolerate large fluctuations and low salinity and can be found at varying distances along the tidal gradient of an estuary or close to areas of freshwater input. Brackish water animals, considered true estuarine animals, can be found within the mid-tidal area of an estuary and can neither tolerate fully marine waters or freshwater (Nybakken & Bertness, 2005).

A study by (Teske & Wooldridge, 2003) investigating the effect of salinity and grain size on macrobenthic distribution found that the nature of the sediment is more important than salinity in limiting distribution. Salinity tolerances for each species can change with season, temperature and other factors. Estuarine organisms are also able to evade temporarily unfavourable salinity concentrations, by burying within the sediment or by movement with tidal inundation, making it difficult to assign tolerance levels or estimate distribution patterns based on salinity. If salinity concentrations were to abruptly change however, species would not be able to acclimatize.

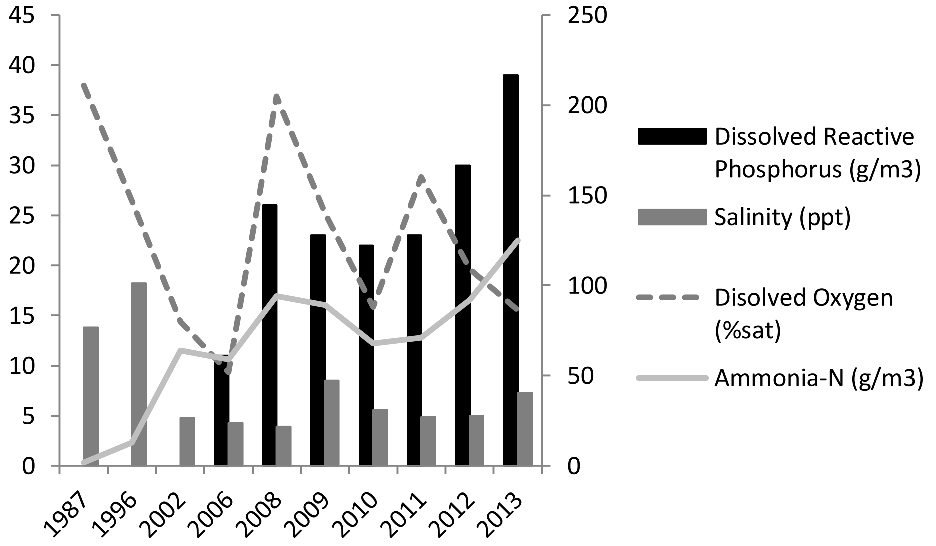

Figure 4.5: DRP, salinity (shown on the left axis) and DO and Ammonia-N values (shown on the right axis) sampled from seepage site W6, within Te Tāhuna o Rangataua, over 1987 to 2013. Data taken from TCC titiko and seepage monitoring report (Gibbons-Davies, 2013).

The TCC titiko and seepage monitoring report (Gibbons-Davies, 2013) provided time series data for dissolved reactive phosphorus, dissolved oxygen Ammoniacal nitrogen and salinity. DRP, and Ammonia-N both showed an increase from previous years within the seepage site W6 (Figure 4.5), indicating that the seepages are a source of nutrient pollution in the high intertidal area and that these inputs are increasing (*see* Figure 1.4 for seepage sites). Salinity was found to decrease or remain low within the area over the years due to the continued presence of the seepage sites. Dissolved oxygen is shown to fluctuate over the years, with DO percentage found to be lower when sampled in 2013. Salinity is very low from 2002 to 2013 (<10 ppt), which would inhibit many animals in the upper intertidal area. Biological activity within the area of the seepages is reported to be limited (Gibbons-Davies, 2013)

Pearson and Rosenberg (1978) notes that identification of species occurring in areas which are subject to changing salinity which is also receiving pollution (and in particular organic enrichment) is possible, as they frequently differ from species exposed to pollution in purely marine areas. Polychaete worms such as *Scolelepsis* sp., *Capitella* sp., and *Polydorid* sp. are suggested to live in areas with lower salinity. and oligochaete worms are suggested to be predominant in further reduced salinities (Pearson & Rosenberg, 1978; McLusky *et al.*, 1993).

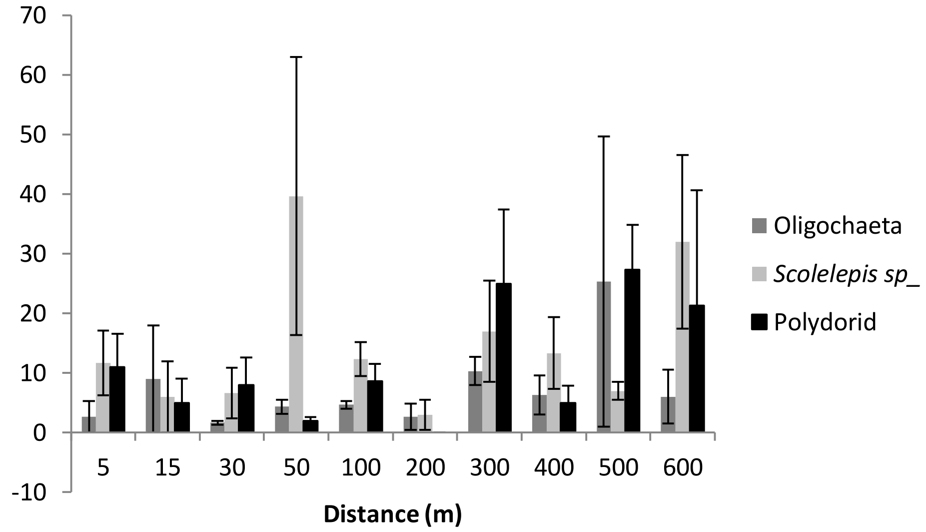

Figure 4.6: Average abundance (n=3, summed to site level) of Oligochaeta, *Scolelepsis* sp. and Polydorid sp. across three transects at each distance, with error bars (±SE), Te Tāhuna o Rangataua. Y axis = $\bar{x}$ of ten 300 cm^3 core samples.

From the current study, Figure 4.6 shows Oligochaeta and Polydorid species increase at a distance of 300 m, 500 m and 600 m from the shoreline. If lowered salinity was driving an increase in abundance, this may be from the Mangatawa Drain and Waitao inputs, although this relationship is speculative. When compared with other environmental variables, no strong patterns appear to emerge which could be influencing species distribution. Large error bars show faunal distribution may differ significantly at some distances along transects. Taking this into account, polychaetes were plotted along individual transects (A, B and C) to assess any patterns or trends. Apart from a high count of *Scolelepsis* sp. on transect B at 50 m (n=84), species abundance appeared to show the same pattern as seen in Figure 4.6. Polydorid distribution didn't follow any strong pattern, though a slight increase with distance could be discerned. The highest count for polydorid abundance was found at 600 m on transect C (n=60), though very low counts were found at 600 m on transect A (n=2) and B (n=2). The second highest count for oligochaete abundance was found at 30 m on transect C (n=27) and the highest count for oligochaete abundance was also found on transect C at 500 m from the shoreline (n=74), with no other sites having abundance counts near this magnitude. Transect C is the closest transect to all freshwater inputs in the area and counts of these species could be attributed to this.

4.6 Species response to fine silt/clay

Norkko *et al.* (2002a) studied macrofaunal sensitivity to fine silts and clay in a lab environment, comparable to sediments in the Whitford Embayment, Auckland. The study examined species distributions in relation to their tolerance of elevated suspended sediment concentration and deposits to the sediment surface. It was found that some taxa showed clear threshold responses to increasing silt/clay, others showed more gradual declines and a few taxa favoured high silt/clay sediments. A Mud Tolerance rating has been used for fine scale estuarine monitoring in New Zealand (Gibbs & Hewitt, 2004; Robertson & Stevens, 2010, 2011). Organisms within these studies appear to reflect mud/sand preferences' found by Norkko, *et al.* (2002a).

Short term deposited layers of clay (>2cm) was shown to have a negative effect to *Austrovenus stutchburyi* (cockle), while *Nucula hartvigiana* (nut clam) appeared to be more resilient (in terms of disturbance and reburial) (Norkko, *et al.*, 2002a). Within Te Tāhuna o Rangataua, cockles and pipi, which have strong sand preferences, were virtually absent.

The polychaete worm *Aonides* sp. and bivalve *Paphies australis* (pipi) were suggested to be highly sensitive to increase in silt/clay and *Aonides*, *Cominella glandiformis*, *Diloma subrosrata* and *A. aureoradiata* were suggested to have a strong sand preference (Norkko, *et al.*, 2002a; Robertson & Stevens, 2010, 2011). Within the Rangataua area sampled, many of these species abundances were negligible (*Aonides* sp., *Paphies australis*) or no strong patterns with silt/clay were detected (*Cominella glandiformis*, *Diloma subrosrata)*. *Austrovenus stutchburyi* (cockles), *Boccardia*, Orbinids (including *Scoloplos cylindrifer*) and *M. liliana* were suggested to be sensitive to fine silt/ clay (Norkko, *et al.*, 2002a), all having a preference for sandy environments (Robertson & Stevens, 2010, 2011).

Macomona liliana (wedge shell) is a deposit feeding species which prefers sandy environments and plays important roles in bioturbation. Deposit feeders such as *M. liliana* and interstitial species are maximally exposed to any pollutants present within the sediment and pore water (Dean, 2008). *Macomona liliana* lives at depths of 5-10cm, using a long siphon to deposit feed on surface deposits and particles in the water column and are negatively affected by increase in suspended sediment. The wedge shell's burrowing behaviour reworks the sediment, improving oxygenation and increasing nutrient fluxes (Robertson & Stevens, 2010). *Nucula hartvigiana* is an animal which actively contributes to the re-working of the sediment and high organic enrichment can cause a reduction in these types of organisms (Pearson & Rosenberg, 1978). Deposit feeders such as these maximize their exposure to pollutants within the water column whilst processing water during feeding (Dean, 2008). In a broad scale survey of the Tauranga harbour, Ellis*, et al.* (2013) found that as silt/clay increased, the abundance of *M. liliana* and *N. hartvigiana* populations decreased across estuaries.

The Similarity Percentages analysis undertaken among all sites within the Te Tāhuna o Rangataua survey, found that between many close and far sites *M. liliana* and *N. hartvigiana* were animals responsible for driving differences in infaunal composition (Table 3.4): with these animals, the distribution being restricted to sites lower on the shore. *M. liliana* and *N. hartvigiana* abundances within the Rangataua area were found to decrease with distance closer to the WWT ponds. Increased mud content in the area may limit the distribution of these bivalves.

Previous monitoring studies (Robertson & Stevens, 2010, 2011, 2013) have found that *H. filiformis* is suggested to have no response to increase in silt/clay, with *H. filiformis,* Nemertea*, Capitella* and the crab *Macrophthalmus hirtipes* preferring some mud but not in high percentages. Nereid polychaetes were suggested to have a slightly positive response to increased mud, with *Perinereis vallata*, *Nicon aestuariensis,* Nereididae (juveniles), *Scolelepsis* sp. (spionid worm) and *Amphibola crenata* all having a preference for muddy environments.

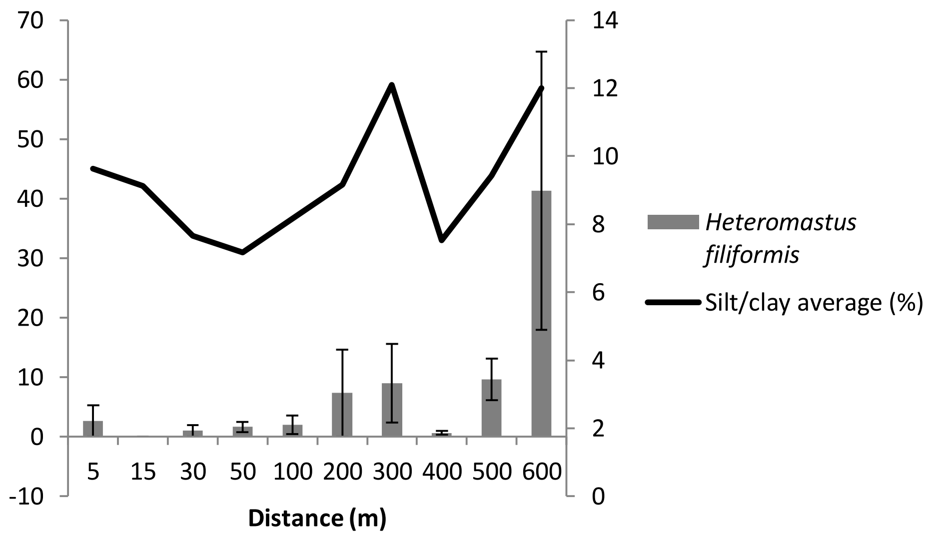

Figure 4.7: Average abundance (Y axis = $\bar{x}$ of ten 300 cm^3 core samples, summed to site level) across three transects of *H. filiformis* (±SE), and average silt/clay % (n=3, see Fig. 3.2 for ±SE), at each distance from the shoreline, Te Tāhuna o Rangataua.

H. filiformis is found to increase at distances 200 m, 300 m and 600 m which is reflective of the trend found for mud content, indicating that mud content may positively affect the distribution of the opportunistic polychaete worm (Figure 4.7).

Within Te Tāhuna o Rangataua, *A. crenata* was found higher along the shoreline, while *P. vallata*, *N. aestuariensis,* Nereidae (juveniles), were found high along the shoreline and at intermediate distances (Figure 4.8), none of which showed trends which directly reflect mud content, though loose associations could be inferred., due to the spikes of both at 200, 300 and 600 m. *Scolelepsis* sp. average abundance across transects (n=3) was found to have an increase at 300 and 600 m, as did silt/clay values (Figure 4.9) which could be the organisms' response to increase in mud content. This would not account for the large abundance found at a distance of 50 m from the shoreline.

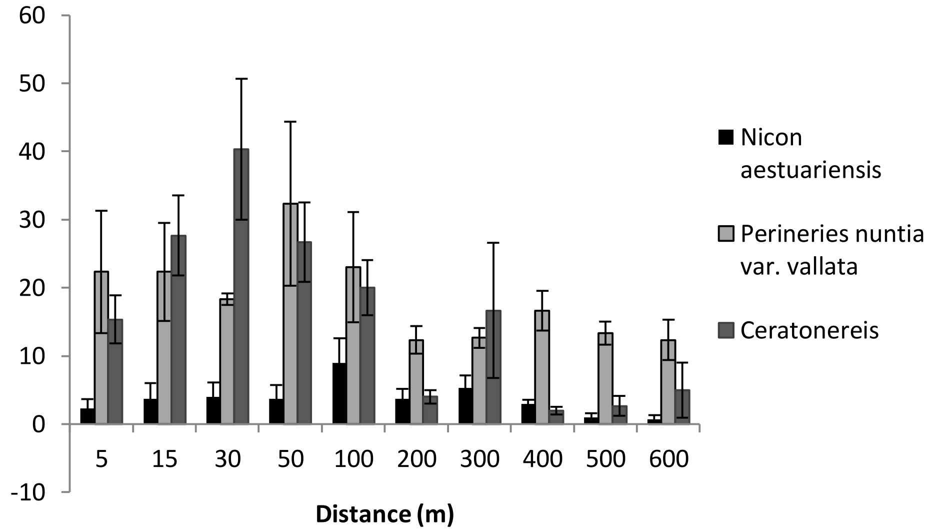

Figure 4.8: Average abundance across transects (n=3) for Nereid polychaete worms at each distance from the shoreline, with error bars (±SE), Te Tāhuna o Rangataua. Y axis = $\bar{x}$ of ten 300 cm^3 core samples.

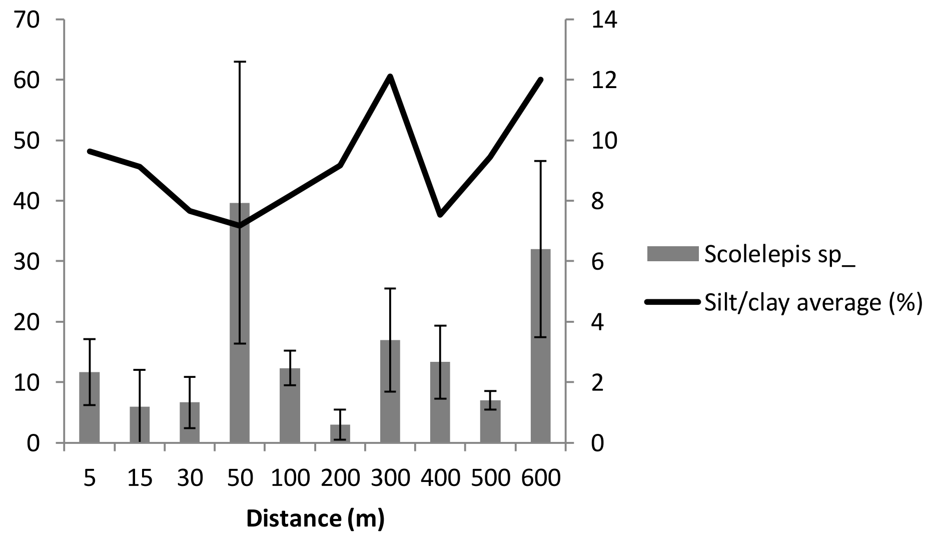

Figure 4.9: Average abundance across transects (Y axis = $\bar{x}$ of ten 300 cm^3 core samples, n=3) of *Scolelepsis* sp., with error bars (±SE) and silt and clay at each distance (n=3, see Fig. 3.2 for ±SE) from the shoreline, Te Tāhuna o Rangataua. First axis is organism abundance and second axis is silt/clay %.

Scolecolepides benhami, *Helice crassa,* Corophiidae (amphipod) and oligochaete worms are suggested to have a strong mud preference (Anderson, 2008; Robertson & Stevens, 2010, 2011), with highly positive responses to increase silt/clay (Norkko, *et al.*, 2002a). *S. benhami* has also been suggested to tolerate low salinities and its distribution is extends to the highest shore levels that are only briefly covered by the tide (Read, 1984).

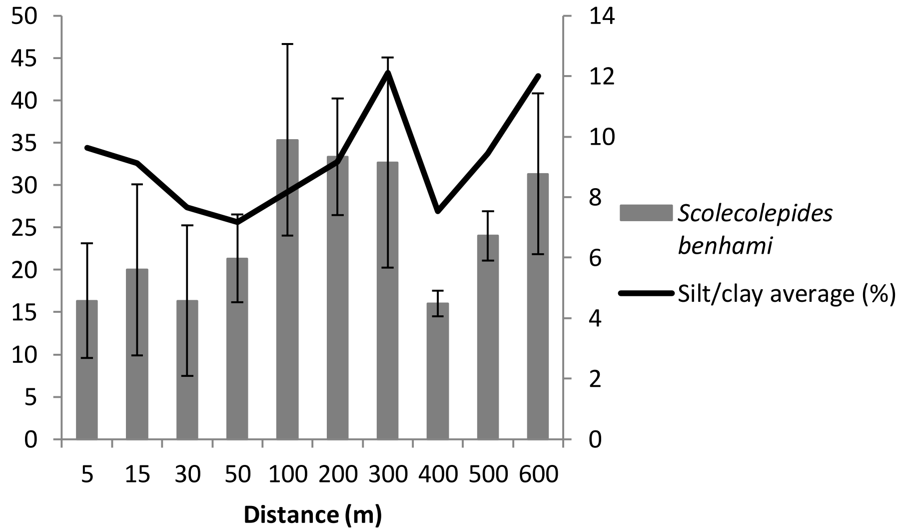

Figure 4.10: Average abundance across transects (Y axis = $\bar{x}$ of ten 300 cm^3 core samples, n=3) of *S. benhami*, with error bars (±SE) and silt and clay at each distance (n=3, see Fig. 3.2 for ±SE) from the shoreline, Te Tāhuna o Rangataua. First axis is organism abundance and second axis is silt/clay %.

The average distribution of *S. benhami* loosely followed the trend found for silt/clay along a gradient of distance (Figure 4.10). The sudden drop in both silt/clay and *S. benhami* and then steady increase to 600 m may indicate that the distribution of *S. benhami* is influenced by increased mud content within these areas. At a distance of 200 to 300 m, the area would be influenced by inputs from the Mangatawa Drain, slightly reduced salinity due to the stormwater inputs may also be influencing species distribution. Within Te Tāhuna o Rangataua, *S. benhami* (Figure 4.10) and *H. filiformis* (Figure 4.7) may be species indicative of increasing mud content.

The amphipod *Paracorophium* (Corophiidae) is a highly mobile, opportunistic species found within New Zealand estuaries, quickly colonising areas that have undergone disturbances. Such disturbances include algal matt growth and fine silt deposition (Robertson & Stevens, 2010). Within Te Tāhuna o Rangataua, high abundances of Coriiphidae (Figure 4.11) where found closer to the shoreline and were virtually absent from sites higher up the shore. Distribution did not follow a trend reflective of mud content, however higher abundances closer to the shoreline may be due to muddier sediments.

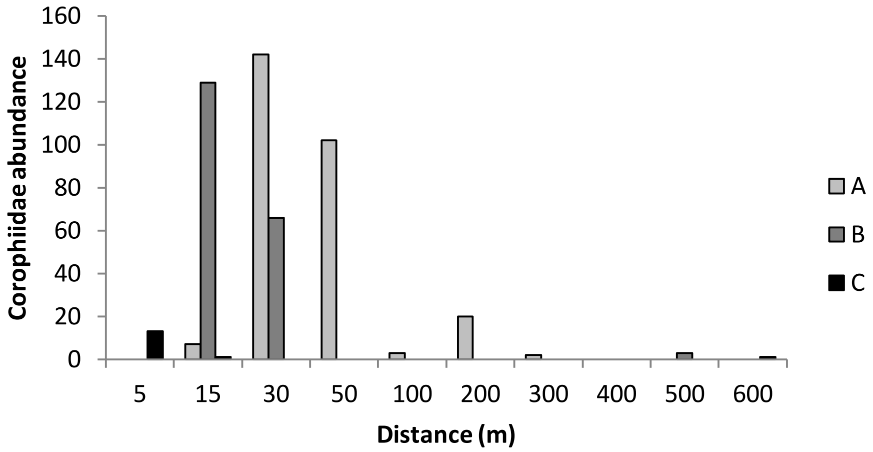

Figure 4.11: Abundance of Corophiidae (Y axis = $\bar{x}$ of ten 300 cm^3 core samples, n=30) along three transects (A, B and C) at each distance from the shoreline, Te Tāhuna o Rangataua.

A common characteristic of highly tolerant species to fine silt sedimentation is their mobility within or on the sediment surface. Suspension and deposit feeding bivalves are generally intolerant to fine sediments. Sensitive taxa generally occupy exposed, sandy intertidal flats, while tolerant taxa occupy upper reaches of estuarine environments or intertidal mudflats (Norkko, *et al.*, 2002a).

A dominant animal found within the sampling area of Te Tāhuna o Rangataua, was a grazing gastropod *Zeacumantus lutulentus.* The organism appears to be restricted to New Zealand, is common on mid-tidal mud flats throughout the North Island and prefers silty sand and muddy environments (Keeney *et al.*, 2013). The gastropod appears to thrive in the Rangataua area, which is not surprising considering the high levels of chlorophyll α that were found, providing an ample food source and the silt/mud sandflats in much of the area. The presence of grazer such as *Z. lutulentus* can play important roles in sediment productivity. By reducing microphytes, grazers may destabilize sediments and allow for oxygen penetration to the lower layers, thereby increasing oxygen flux to lower sediment layers (Thrush, *et al.*, 2006).

Z. lutulentus was not found in high numbers at the closest distances to the WWT ponds, regardless of the high concentrations of chl-α. This may be due to a number of factors, including decreased salinity, increased organic matter and the increased muddiness, especially within the mangrove dominated area.

Another dominant animal found across sites is the orbinid polychaete *Scoloplos cylindrifer*, a sub-surface deposit feeder (Figure 4.12). *S. cylindrifer* has been found to be a colonist after a disturbance (in particular sediment deposition) this being attributed in part, to the animals mobility during recruitment via sediment bedload transport (Norkko *et al.*, 2002b).

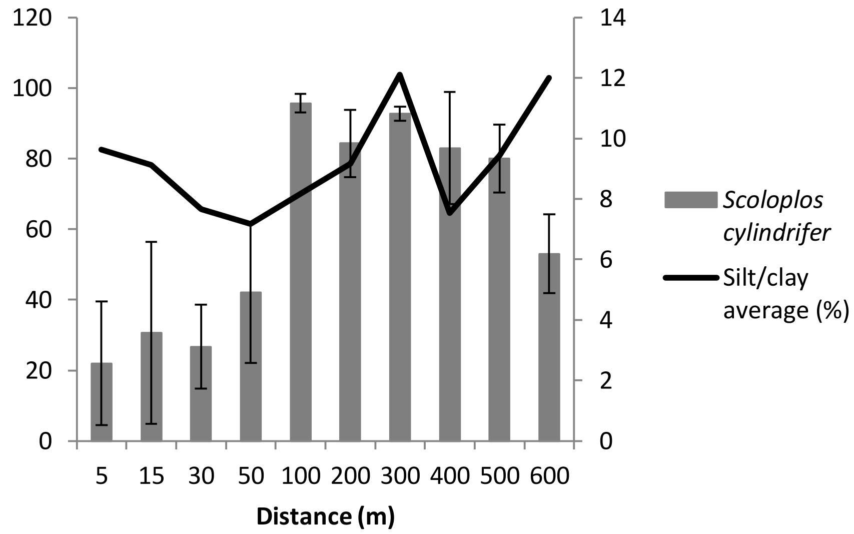

Figure 4.12: Average abundance across transects (Y axis = $\bar{x}$ of ten 300 cm^3 core samples, n=3) of *Scoloplos cylindrifer*, with error bars (±SE) and silt and clay at each distance (n=3, see Fig. 3.2 for ±SE) from the shoreline, Te Tāhuna o Rangataua. First axis is organism abundance and second axis is silt/clay.

4.7 Assessing species responses to other pollutants

Lead (Pb) was found to significantly correlate with many other variables (Table 3.7). One of these variables was phosphorus (r > 0.8), which was identified from the Distance Based Linear Modelling to be a predictor variable which was significant in explaining variation within macro-benthic assemblages, influencing community structure at sites higher on shore (Figure 3.16). As lead correlates with phosphorus, phosphorus may act as a surrogate for lead, indicating that lead may also be responsible for characterising assemblages at closer distances. As noted earlier, faunal composition at sites high along the shoreline were dominated by scavengers, nereid worms and amphipods (Figure 3.14), many of which have been identified in previous studies to be opportunistic and able to live in slightly degraded or polluted environments.

Although responses of specific species along a gradient of enrichment has been found in previous studies, such changes have not been proven for other pollutants, such as heavy trace metals (Pearson & Rosenberg, 1978). Pearson and Rosenberg (1978) suggested that such types of pollution are unlikely to favour any one species and that instead, a general and continuous decrease in biodiversity may be expected in the presence of other pollutants. Within Te Tāhuna o Rangataua, a shift in community composition was found with distance from the impact site (WWT ponds), with lowered biodiversity at the high tidal area. This could, in part, be attributable to contaminants entering the bay from the Mangatawa Drain.

Many toxicology tests have been undertaken for polychaete species over the years, with various studies showing that species may have varying LC_{50} responses to different toxicants and that closely related species may vary in their sensitivities to toxicants (Dean, 2008). Although these studies have led to a better understanding of toxicant effects on benthic species in a laboratory environment, applying these results to the field is made difficult with many confounding factors. Sensitivities of toxicants to organisms can be greatly affected by physico-chemical conditions within the sediment. The binding of toxicants to sediments may reduce their bioavailability to resident organisms, allowing them to live in polluted conditions.

Abiotic factors such as salinity, temperature, oxygen concentration and organic material can interfere with responses when assessing the effects of toxicants to benthic organisms. As well as this, assignment of particular species to an environmental condition is difficult as benthic communities in different geographic regions vary in their response to degraded environments (Dean, 2008). What has been made clear is that the effects of toxicants and pollution on benthic communities is very complex and the idea that any single species presence will always be indicative of a specific toxicant or environmental stress is incorrect. Dean (2008) suggests that polychaetes are, however, valuable as indicators of community diversity "within the confines of a specific sampling program" and that "species recognised as indicators should be viewed as specific for the area studied".

Although there is wide debate about labelling specific species as indicators of specific pollutants, studies have progressively moved from quantifying single species towards a community based approach in terms of indicating environmental stress in an area, with the effects of stress within assemblages being easier to measure (Giangrande *et al.*, 2005).

Accumulated concentrations of pollutants within sediments represent a high risk to the benthos and in particular to deposit-feeding infauna. Sediment ingested by organisms is one of the most significant uptake routes of sediment-bound contaminants such as trace metals (Casado-Martinez *et al.*, 2013).

4.8 Summary

Although tidal elevation is a significant characteristic of estuarine environments, within Rangataua Bay other environmental pressures are also exerting control over macrobenthic structure. From the results of this study, species which appear to be indicative of elevated mud content include the mud crab *H. crassa*, polychaetes *S. benhami* and *H. filiformis*, with *H. filiformis* also appearing to have a positive response to organic enrichment. Sensitive taxa which are not found within the upper intertidal areas are the bivalves *M. liliana* and *N. hartvigiana*. The environmental condition which influences their distribution will be explored further in the following chapters.

Hydrology and morphology of Te Tāhuna o Rangataua will be discussed in more detail within the next chapter, in which depositional endpoints and the processes which pre-empt them are examined. Of importance is the change in community structure that can be seen along a gradient of distance from the Wastewater treatment ponds and the biotic and abiotic interactions therein. The following chapter will discuss in detail the results of environmental variables and how they fit into the wider context of environmental condition and corresponding biodiversity in Te Tāhuna o Rangataua.

Chapter 5

Discussion: Part II; Environmental Condition

Wastewater seepages and discharge are generally associated with a suite of organic pollutants. This includes an increase in nutrients (phosphorus and nitrogen) and an increase in organic debris or particulate organic matter from the source of pollution. Increase in nutrient inputs will also stimulate primary production, leading to algal blooms (Tay, *et al.*, 2012). Nitrogen, phosphorus and organic matter are often linked with sediment grain size, with nutrients and organic content often found to be elevated with muddier sediments (Gillespie, *et al.*, 2012a). Figure 6.1 shows discolouration and increased wetness of sediment due to groundwater seepages from the Te Maunga wastewater ponds.

Figure 5.1: Wastewater seepage pollution in the upper intertidal fringe of Te Tāhuna o Rangataua. Photo taken February 2014.

5.1 Estuarine condition with reference to other New Zealand estuaries

From the Principal Co-ordinates analysis of Te Tāhuna o Rangataua and other estuaries within Tauranga Harbour (Ellis *et al.*, 2013), sites within Te Tāhuna o Rangataua form a subset, with the environmental variables of these sites causing them to cluster together within the larger set of estuaries from the broad scale survey (*see* Figure 3.17). This indicates that the general estuary condition is slightly to moderately enriched, as was found as a broad overview of all estuarine areas within the 2013 Tauranga Harbour survey. Within the Tauranga Harbour survey, nutrient and organic matter concentration in the harbour tended to decline with distance from the inner harbour region (which would include more sheltered and low-lying embayments that have less tidal influence) and associated rivers (Ellis, *et al.*, 2013).

Within Te Tāhuna o Rangataua, nitrogen levels appear to be slightly higher across sites (200–1500 mg/kg) than other estuaries within Tauranga Harbour, while phosphorus levels are around the same (see Table 3.13). Using the condition rating for nitrogen from Robertson and Robertson (2014) estuarine areas across Tauranga Harbour and within Te Tāhuna o Rangataua are low in nitrogen (250-1000 mg/kg), save for one site (Site A1, Fig 2.1), falling in the moderate nitrogen enrichment range (1000-2000 mg/kg) (Robertson & Robertson, 2014).

Across a total of 75 estuaries found in Tauranga Harbour, the highest organic content (AFDW) value was 4.5%, while the highest organic content value found within Te Tāhuna o Rangataua was 4.9% (Figure 3.13). Organic content is given a condition rating of 1-<2% = moderate, 2-<3.5% =high and >3.5% very high (Robertson & Robertson, 2014). Although the high percentage of organic content and elevated nitrogen level was confined to one area within this study, it is still indicative of high organic enrichment, in particular with comparison to other estuaries within the Tauranga Harbour and it is occurring closest to the Wastewater treatment ponds.

A link between mud content, organic matter and nutrients was evident across reference sites within the Estuarine Monitoring Protocol (Robertson, *et al.*, 2002). Sediments in most of the reference estuaries (excluding two, which were also found to be dominated by mud) were found to have an organic content of 1-2%, similar to sites found in the Rangataua area. The environmental variable of most significance in driving differences between Rangataua Bay and the rest of the Harbour was chlorophyll-α.

Comparisons of nutrient concentrations with the Estuarine Monitoring Protocol (Robertson, *et al.*, 2002) reference sites indicate that the Rangataua Bay study area was in a range typical for low to slightly enriched estuarine areas. Nutrient levels of reference estuaries within the EMP were found to vary between sites (200–1650 mg/kg), indicating reference estuaries were all undergoing differing degrees of enrichment. Two estuaries which were considered to be moderately enriched compared to other estuaries were Kaikorai and Kaipara with total nitrogen (TN) concentrations of 1630 mg/kg and 1650 mg/kg, respectively. One site within this study is comparative to these estuaries, Site 1 on Transect A (5 m from shoreline (Figure 2.1) was found to have the highest TN concentration (1500 mg/kg), along with slightly increased total phosphorus (TP) concentration (470 mg/kg)**,** the highest organic content (4.9 %) and the second highest percentage of mud content (20.2% silt and clay).

As well as being in close proximity to the Wastewater treatment ponds, mangroves dominate the area in which site A1 is located, which would influence build-up of fine silt sediments and organic detritus. Productivity within the sediments would be reduced in this area, which is reflected in the lower species count and abundance, with the few animals found at this site including scavengers such as mobile mud crabs, amphipods, neried polychaetes and a few deposit feeding polychaetes.

5.2 Depth, hydrodynamics and sediments

Estuaries can be categorised as either micro, meso or macro-tidal environments with tidal ranges; <2 m, 2-4 m and > 4 m, respectively. (Jackson, 2013). Estuaries that are shallow tend to be flood dominated, with sediment transport largely occurring from freshwater inputs. Sediments from river flow will be either coarse or fine, with coarser sediments settling to the seabed and finer sediments remaining suspended and being transported further afield (Jackson, 2013).

Within the intertidal zone bed irregularities occur due to different factors and this affects hydrodynamics in an area. These factors include drainage, drying events, biotrubation and protection by plant growth (Le Hir *et al.*, 2000).

Sediments may be continuously shifted by tidal inundation and wave action, as well as by animal disturbance such as bioturbation and predation. In soft shore environments with high wave-energy and strong currents, wave action is suggested to create an intertidal gradient. Sediments are sorted according to size, through re-suspension of finer sediments with wave action, while coarser sediments settle out. By this process sediments are suggested to be coarser higher on the shore while lower on the shore they are finer (Peterson, 1991). This would be more prevalent in open coastal areas or beaches, which experience high energy waves, with fine sediments remaining suspended leaving only coarser sand on the sea floor. Sediment transport induced purely by wave energy is much less important in estuaries than on the coastal shelf, however re-suspension by waves may still influence seaward sediment transport (Dronkers, 1986).

Grain size can influence benthic assemblages based on their mode of feeding, such that deposit feeders can be found in finer sediments, feeding on deposits on the bed floor and suspension feeders can be found in coarser sediments, with currents replenishing food supplies in the water column. This intertidal gradient is suggested to cause zonation in soft-sediment environments (Peterson, 1991). An along shore gradient in grain size of the sediments such as this however, is not observed from the data collected in Te Tāhuna o Rangataua. Gravel (grain size of >2mm) was found to have the highest percentage closest to the shoreline, though it did not show a trend along a gradient of distance. Fine silt and clay (< 63 µm) was also found to be high close to the shoreline as well as 400 m away, which suggests tidal inundation doesn't appear to create an observable gradient of coarse sand to fine sand.

Factors which determine whether an estuary is a source or a sink for sediments include tidal amplitude, hydraulic depth, sediment characteristics, fluvial processes and estuarine stratification (Jackson, 2013). Two transport modes exist for sediment types within an estuary; along the bottom, with coarse sediments pulled along the bed load and fine sediments moving with fluid mud, or by suspension in the water column (Dronkers, 1986).

The tide influences currents, which involve cross-shore and long-shore components. Tidal movement in a tidal basin consists of a complex geometrical system with a network of channels and tidal flats (Dronkers, 1986). Long-shore involves the filling and emptying of the intertidal area and cross-shore is the circulation of currents within the area. In shallow, semi-enclosed and flat intertidal areas such as Te Tāhuna o Rangataua, wave energy is not a predominant influence, though small waves may still be enough to re-suspend fine sediments (Le Hir, *et al.*, 2000). Sediment erosion, transport and deposition are influenced by tidal currents (cross-shore and long-shore) and wind circulation (Le Hir, *et al.*, 2000; Jackson, 2013) and incoming tides promote onshore transports of finer sediments (Jackson, 2013).

Cross-shore currents can depend on the width of the flat and they may exceed the long shore current if the area is particularly wide, as is the case in Te Tāhuna o Rangataua. Long-shore tidal currents (vertical to the shore) are influenced by large scale geomorphology of an area. Flood and ebb components of tidal inundation are important in influencing bed load transport, while slack durations are important for suspended matter transport. Slack duration is a short period where there is no tidal movement or tidal stress occurring (Le Hir, *et al.*, 2000).

The upper intertidal area only experiences slack at high tide and particles moving onshore during tidal inundation begin to settle in the upper region. Re-suspension of these deposited particles requires ebb-flow speed high enough to facilitate it, which is often not reached in the upper intertidal area (Le Hir, *et al.*, 2000). This is predominantly the case in shallow, enclosed areas such as Te Tāhuna o Rangataua. Tidal asymmetry can cause differences in the slack water period and is influenced by a number of factors. Cross-shore (parallel to the shore) asymmetry may be influenced by local topography and terrestrial inputs and cross-shore currents may be dominant when a flat has little slope (Le Hir, *et al.*, 2000), deflecting along-shore environmental gradients from occurring. Within intertidal flats which are wide, enclosed and have minimal wave energy, tidal chemico-physical gradients within the sediment are likely to be reduced and corresponding zonation of fauna is unlikely to be strongly developed (Read, 1984) .

5.2.1 Sediment inputs

Tidal asymmetry influences movement of coarse and fine sand in different ways and is suggested to affect suspension transport (and thus fine sediments) more than bed load transport. Fine sediments settle from the water column only at low current speeds, with fine sediment load settling in the period around slack water (Dronkers, 1986)

Sediments which have received organic enrichment or fine silt deposits are in areas where low current speeds are dominant at least some of the time. In these areas there is a greater proportion of fine sediments and often organic matter and nutrients will show a relative increase as well. Aside from low speeds allowing for more deposition, low speeds may also decrease possible oxygen renewal, increasing biological oxygen demand due to the aforementioned organic matter deposition (Pearson & Rosenberg, 1978).

The importance of drainage in hydrodynamic processes is specific to an intertidal area (Le Hir, *et al.*, 2000). Within the Rangataua area, runnels or channels created by drainage from the Mangatawa Drain, the Rocky Stream and the Waitao Stream would run at differing angles to the long shore current, into the area, causing cross-shore asymmetry (Figure 1.2 shows entrance points). Transport within these channels will greatly influence particle distribution and lead to several depositional endpoints. Increased wet weather events would greatly influence drainage and inputs into the area. Drainage from the Mangatawa Drain channel may also contribute to offshore transport of previously deposited and re-suspended particles to the middle reaches of the bay.

The highest percentages of gravel was found at sites B1 (4.6%) and C1 (2.3%), while site A1 was lower (0.5%) (*see* Appendix 1, Figure 7.1). Gravel percentage decreases 15 m from the shoreline and remains relatively constant across the rest of the sites along a gradient of distance (Figure 5.3). Sites B1 and C1 are closer to the drainage area of the Mangatawa Drain, receiving inputs of terrestrial sediment, with the results suggesting that gravel may be settling closer to the source and fine sediments being transported and depositing elsewhere in the area. Coarse sediment, such as gravel, is much heavier than sediment with smaller grain sizes and therefore loss of coarse sediment to the ocean is much smaller than fine sediments as it is less easily dispersed and re-suspended, tending to settle closer to its source (Hume, *et al.*, 2009).

From the Principal Co-ordinates analysis of environmental variables (Figure 3.15), it was found that very fine sand was a significant environmental feature, characterizing sites further away from the shoreline. Sources of very fine sand, along with silt and clay would be a culmination of inputs from all sources in the area (*see* Figure 1.2 for sources and Figure 5.2 for distributions). Silt and clay are highly charged particles which flocculate on contact with seawater, rapidly dispersing and depositing in an area (Thrush, *et al.*, 2004).

Silt and clay was found to be high at Site A1 (20.2%) and Site A2 (14.1%). As noted by Hume, *et al.* (2009), present day mud content is scaled as high (>20%), moderate (10–20%) and low (<10%). Mangrove cover in the upper intertidal area in which site A1 and 2 were located, would promote accumulation of silt and mud. Drainage of the Papamoa catchment area into the upper intertidal fringe of Te Tāhuna o Rangataua is probably the main source of both fine silt and coarse sand percentages found within the high tidal area. The Mangatawa discharge channel would have significant effects to the hydrodynamics of the area, in particular influencing the sea bed. To a lesser extent; the treated wastewater seepages may too have an effect with the continuous inputs of surficial freshwater may deflect processes on a local scale that would naturally occur with tidal inundation.

Due to margin alteration in Te Tāhuna o Rangataua, the area would have seen a shift from a productive and efficient filtering system of the marshland area, to an abrupt decrease in surface area for tidal flow in the upper intertidal fringe. This would have increased cross-shore currents and slack time at the upper intertidal limit, allowing for a long deposition time for particles to accumulate. The condensing of tidal transport mechanisms would have led to the area acting as a sediment trap and accumulation of muddy sediments would have supported a spread of unproductive mangrove growth, with the placement of the ponds forming an abrupt barrier between land and sea.

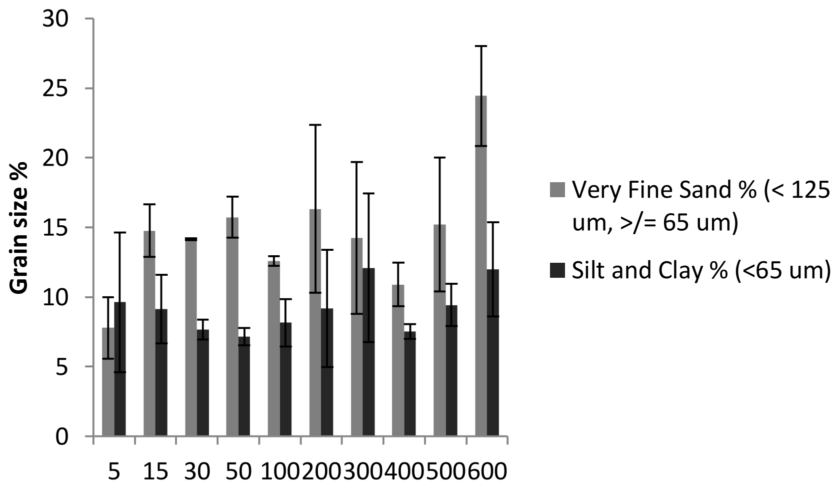

Figure 5.2: Very fine sand (<125 µm, >/=65 µm) and Silt/clay (<65 µm) average percentages (n=3) across transects with error bars (±SE), at each distance from the shoreline, Te Tāhuna o Rangataua.

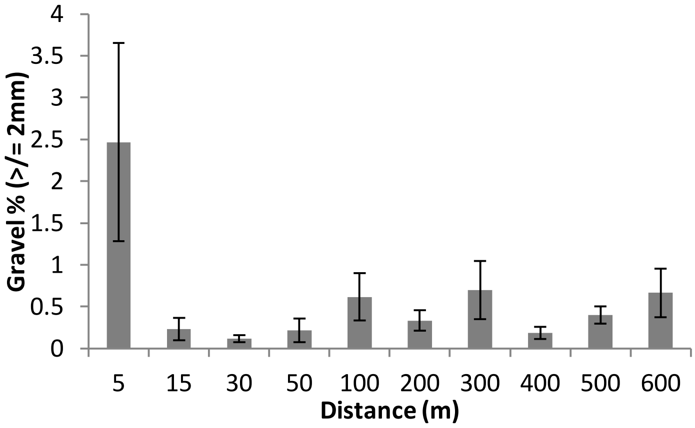

Figure 5.3: Average percentage (n=3) of gravel (>/= 2mm) across transects with error bars (±SE), at each distance from the shoreline, Te Tāhuna o Rangataua.

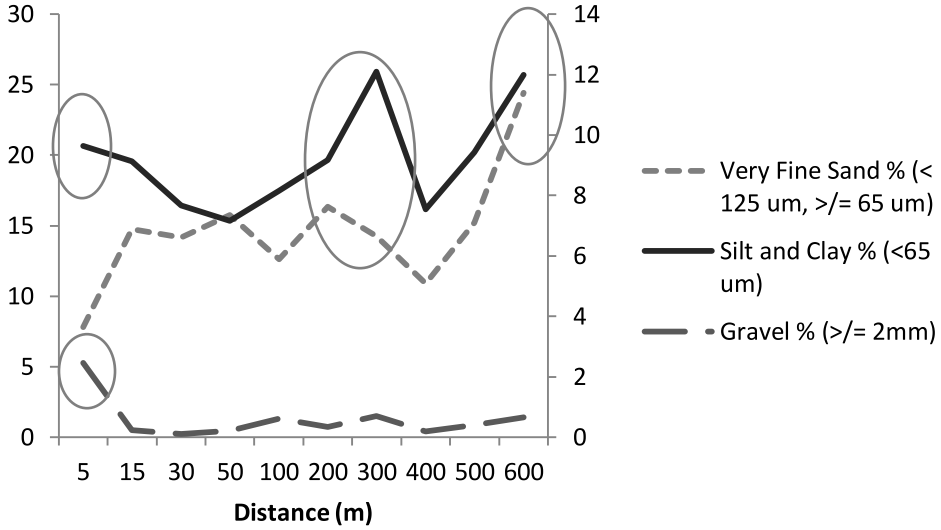

Figure 5.4: Average percentage (n=3) of differing sediment grain sizes along a gradient of distance from the shoreline (*see* Figures 6.1 and 6.2 for error bars). Axis 1 shows percentage of very fine sand % (<125 µm, >/=65 µm) and Axis 2 shows silt and clay % (<65 µm) and gravel % (>/= 2mm). Circles show depositional endpoints for differing grain sizes in Te Tāhuna o Rangataua.

Figure 6.4 shows depositional end points of silt/clay sediment classes that coincide with increases for both very fine sand and gravel, leading to the recognition of three areas prone to sediment accumulation. A spike in mud content was found at 200 and 300 m on Transect C (sites C6 and C6.5), with a mud percentage of 17.8% and 22.8%, respectively. The second highest percentages of very fine sand are found at distances 200 m and 300 m on transect C (26.5% and 25.1%, respectively). Accumulation of fine silt sediments would be occurring within this area, from the Mangatawa Drain and to a slightly lesser extent from the Waitao Stream. Very fine sand is found to be highest at 600 m on transect B (30.6%). Inputs from the Waitao Stream and surrounding sub-catchments (Figure 1.2) would influence sediment transport to this area.

Forestry covers a large proportion of the Waitao catchment (17.5%) and most sedimentation from forestry is released during the harvesting period, and it is predicted that sediment load from a mature forest is doubled during the harvesting phase (Hume, *et al.*, 2009). A number of control measures are suggested for decreasing sediment load during forestry harvesting. (Hume, *et al.*, 2009)

Hume et al (2009) estimated sedimentation rates occurring within each estuary and gave an indication of parts of the harbour where ecology may be at risk due to fine-silt sedimentation. The Welcome Bay and Rangataua area was "roughly indicated" to fall in the "ecological alert" category, suggesting an increase in sedimentation in both areas through time. Welcome Bay had a high present-day bed-sediment mud content (>20%), with a high sedimentation rate of >1.0 mm/year. Within the northeastern intertidal flats of Te Tāhuna o Rangataua (the upper intertidal fringe adjacent to the Te Maunga Wastewater Treatment ponds) mud content was moderate (10-20%) and sedimentation rate was high. This area is fringed with a dense growth of mangroves. The central reaches of Te Tāhuna o Rangataua was found to have a low mud content (<10%) and a moderate sedimentation rate (0.30-1.0 mm/year). The high sedimentation rates of both the north-eastern intertidal flats of Te Tāhuna o Rangataua and the adjacent Welcome Bay estuary indicate that muddy sediment will encroach into the central intertidal flats of the area. This is indicated to foster a corresponding spread of mangroves (Hume, *et al.*, 2009).

An integration of information collected and summarized by Hume, *et al.* (2009) provided a scale of ecological effects (low, medium and high) to which an estuary may be subjected to, based on the combined influence of catchment use and climate change. Te Tāhuna o Rangataua was identified as an estuary with a high potential for adverse ecological effects. Sedimentation resulting in poor water and sediment quality which negatively impacts estuarine ecology is difficult to reverse (Robertson & Robertson, 2014). Radical shifts in ecosystem structure may move to critical thresholds at which recovery may not be likely and value of the area is lost (Thrush, *et al.*, 2004).

5.3 Trace heavy metals

Generally, heavy metals were found to be elevated closest to the shoreline then fluctuate, before increasing towards the mouth of the bay. Lead and Zinc were found to increase towards the low tide, 600 m from shore, as was found with Arsenic and Copper. Three spikes in values can be seen on a gradient of distance for Cu, Zn and silt and clay (Figure 5.4, 5.5) at 5 m, 300 m and 600 m. The slightly elevated levels of these heavy metals and mud content are in the receiving environment of the discharges from the Mangatawa Drain and the Waitao Stream. As was found in the sediment study by (Park, 2003) Site 26 within the Rangataua area (see Figure 1.6 for site location), in which higher trace levels of these metals were found, is located in the receiving environment of storm water discharges from the Mangatawa drain, along with freshwater inputs from the Rocky Stream. Lead and zinc are typically associated with urban sources (Park, 2009).

At distances 5 to 15 m and 300 to 400 m from the shoreline there appears to be depositional endpoints for contaminants and fine particles, in which copper, lead, zinc and mud/silt are concentrated. This would be influenced by hydrodynamic processes in the area including the flow of the drainage channel consistently discharging into the area, tidal inundation and increased cross-shore currents and slack time at the upper intertidal limit. Land use within the Papamoa Hills catchment (Table 1.1) which contribute to the inputs of these pollutants include urban and road works. Storm water flow from State Highway 2 would contribute greatly to the inputs of heavy metal pollutants. Stormwater is produced when rainfall flows over surfaces, picking up pollutants before being channelled into a drainage network and eventually discharged to coastal environments. Due to the nature in which pollutants are collected, it is difficult to trace them from any single source (Botherway & Gardner, 2002).

Land use within the Waitao sub-catchment (found in Table 2.1), would influence the inputs of pollutants and sediment and include effects from agriculture, horticulture and to some extent developmental earthworks. An active quarry occurs within this catchment and together with the steep elevation of much of the surrounding land, would greatly influence inputs of fine silt and suspended particles.

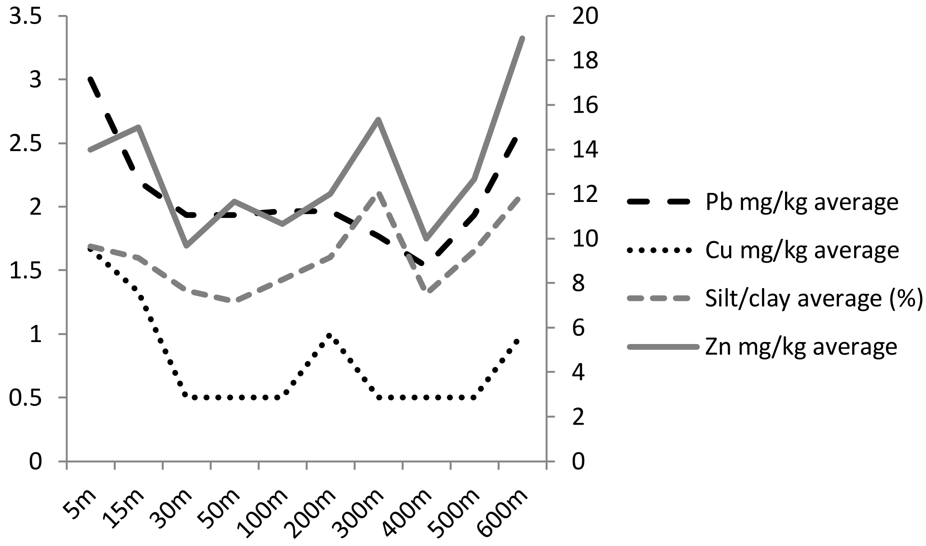

Figure 5.5: Average values across transects (n=3) for heavy metals and silt/clay along a gradient of distance from the shoreline, Ta Tāhuna o Rangataua. The first horizontal axis displays Cu and Pb concentrations, the second axis displays Silt and clay % and Zn concentrations.

With increased monitoring and awareness, toxic pollution within estuarine environments has begun to decline. Arsenic in small concentrations, typically less than 4 µg (Smedley & Kinniburgh, 2002), is a naturally occurring element in aquatic environments (Riedel *et al.*, 1987) and many environmental problems are the result of Arsenic mobilisation under natural conditions.

Within the Te Tāhuna o Rangataua, only small concentrations of arsenic were found, with the highest concentration of arsenic (As) found (2.1 µg) at 600 m from the shoreline. Transport and flux of arsenic involves several processes across the sediment/water interface, including redox chemistry and disturbance. Waves and currents will re-suspend sediments which may release concentrations of arsenic from interstitial water to the sediment surface. Activities of benthic fauna, such as bioturbators and tube dwellers, may release toxic materials from sediments (Riedel, *et al.*, 1987). The area in which higher concentrations of arsenic was found (the low tidal area) would have higher wave and current energy and be more prone to disturbances from animals with tidal inundation, thereby influencing observed concentrations. This may also indicate that increased sedimentation in the intertidal area (600 m from the shoreline) may be occurring, in which natural sources of arsenic may be confined to bottom sediments due to frequent smothering by sediment inputs.

Sediment constituents such as organic matter and metals are generally higher in muddy sediments and, in particular, metals are often closely associated with fine silts and clay through sorption (Robertson, *et al.*, 2002). With this in mind, higher concentrations of metals in sediments may be a reflection of higher silt/clay percentage within the sediment rather than increased contamination (Grant & Middleton, 1998).

Heavy metals are able to persist in the environment, capable of accumulating through the trophic web. In this instance, concentrations which are small within the sediment may persist for long periods of time and as a consequence, accumulations within organisms may eventually reach levels which have lethal or sub-lethal effects (Botherway & Gardner, 2002).

Long term persistence of heavy metal pollutants within Te Tāhuna o Rangataua may have negative effects to biota. These may be subtle and chronic and can occur at the trophic, community or ecosystem level. This would make it difficult to assess effects of such pollutants, against a backdrop of overwhelming variability within the environment.

5.4 Nutrients

Within the current study the Principal Co-ordinate analysis (Figure 3.15) found that total phosphorus and chlorophyll-α were environmental variables that were driving differences in environmental condition of sites closest to the WWT ponds comparative to sites further away. Total nitrogen was found to be positively correlated with TP ($r>0.8$) indicating that nitrogen is also a significant environmental variable at sites closer.

Distance based linear modelling undertaken, indicated TP was a significant environmental variable driving differences in community composition between sites closest to the WWT ponds and sites found lower on the shore (8.95% of variation explained by TP, $p = 0.015$ (Figure 3.10)).

TN and TP follow the same trend along a gradient of distance, with slightly elevated levels closest the shoreline. Elevations in nutrients closest to the WWT ponds were expected, due to the wastewater seepages. TN and TP then show a steady decrease along a gradient of distance and then a slight increase 600 m from the shoreline. Organic matter also shows a slight increase at 600 m (Figure 3.1). Nutrients entering the bay from the Waitao Stream may be influencing this increase, as well as catchment inputs from surrounding areas such as Kaitemako (Figure 2.1).

Phosphorus is a reactive element which follows a complex path in marine environments. Decline of phosphorus in aquatic waters has been attributed to improved agricultural practices and industrial and municipal water treatment (Van Damme *et al.*, 2005). Phosphorus from terrestrial inputs may be made more bioavailable with increased salinity and in the summer months, an increase in phosphate released from sediments may occur (Tay, *et al.*, 2012).

A study by Tay, *et al.* (2012) which assessed nutrient concentrations in two estuaries in the Tauranga Harbour, found a high correlation between nitrate and salinity and a lack of correlation between phosphate and salinity, suggesting that nitrate variations were controlled by change in freshwater inputs, while phosphate concentrations were more likely to be influenced by nutrient-cycling processes within the sediment. Nitrogen concentrations would generally show an increase with rainfall and runoff from surrounding catchments, though this would be influenced by the flushing of an estuary. An estuary which is not well flushed may show accumulation in nutrients, though with a flooding event, these concentrations may decrease.

The risk of eutrophication due to elevated nutrient concentrations is dependent on the time scale over which it persists in an environment. This, in turn is largely dependent on resident times and flushing of water bodies in an area (Tay, *et al.*, 2012). As discussed earlier, flushing times within Te Tāhuna o Rangataua may be reduced due to its morphology and distance from the Harbour mouth. Within the upper intertidal area high chlorophyll-α concentrations infer increased nutrient levels which has stimulated microalgal growth. The nutrient levels found are, however, considered only to be low or slightly elevated (Robertson & Robertson, 2014) and this may be due to depletion of nutrients in the summer months due to assimilation by the microphytobenthos. In shallow waters, much of the benthos is within the photic zone, which allows benthic microalgae to flourish, given nutrient availability is not limiting.

5.5 Chlorophyll-α

Chlorophyll-α is an abundant photopigment in living phytoplankton and is therefore a useful indication of microphytoplankton of the benthos. It is a useful measurement with which to track the fate of organic carbon derived from primary production (Ingalls *et al.*, 2000). Although chl-α is used to estimate primary production, primary production and biomass is not always related. The abundance of phytoplankton algae is largely dependent on flushing times, proximity to nutrient sources and light availability (ANZECC, 2000).

Chlorophyll-α may be useful to identify any major bloom occurrences. Reference estuaries surveyed within the EMP found chl-α concentrations to be widely variable and indicated low to moderate microalgal mat development across sites. Concentrations indicating conditions of extreme enrichment were not observed in any reference sites. Within the EMP, of the reference estuaries studied, the densest microalgal coverage was observed within the Avon-Heathcote estuary, and could be identified visually by a green-brown discolouration of the sediment. This was suggested to be caused by discharge from the Christchurch Wastewater Treatment Plant (Robertson, *et al.*, 2002).

Within the upper intertidal sites of Te Tāhuna o Rangataua, a discolouration of the sediment could be seen, which often coincided with sinking mud and a distinct sulphuric odour. This was most clearly observed near and within mangrove growth closest to the treatment ponds.

Microphytobenthic biomass can exhibit seasonal, tidal and diel variation which corresponds with change in temperature, light and nutrients availability. After a disturbance of the sediments, microphytobenthic algae are able to quickly recover if ample nutrients and light are present. Oscillations in productivity occur, with high productivity during low tide and a drop in productivity during high tide and at night. Variability can be either deterministic or random. Tidal, diel and seasonal variation may be predicted but fluctuations based on local environmental factors and sampling variability is more random (Blanchard *et al.*, 2002). Therefore, when using chl-α as an indicator of condition, sampling of time series data is essential in fully representing the nature of microphytobenthic biomass dynamics, taking into account the differing scales of variation.

Figure 5.6: Discolouration and seepage pollution of sediment in the upper intertidal area, adjacent to the Te Mauna Wastewater Ponds, in Te Tāhuna o Rangataua. Photos taken February 2013.

Algal growth is suggested to be limited by nutrients and in particular total nitrogen, based on the findings that dissolved nitrates usually become analytically undetected before dissolved phosphates (Risgaard-Petersen, 2003). Benthic microalgae are responsible for a large proportion of primary production within estuarine environments, with high concentrations at the sediment-water interface significantly affecting biochemical processes within the sediment, inducing fluxes in O2 concentrations and dissolved inorganic carbon concentrations (Risgaard-Petersen, 2003).

Temporal variability of microbenthic algae ranges from seasonal to intertidal. Growth is increased during summer with high temperatures, during daylight hours and during low tide, while concentrations may decrease during winter, during night time and at high tide due to reduced primary production, re-suspension, grazing and natural mortality (Blanchard *et al.*, 2001). Microphytobenthic communities may develop and act as barriers, preventing nutrient exchange with the water column (Conley *et al.*, 2007). It is suggested in intertidal mudflats, with the absence of the effects of re-suspension (which replenishes resources) and grazing on microbenthic algae (which provides space for growth), the benthic community would become unproductive. Concentrated blooms would occur, reaching biotic capacity within a system and the entire biological community would show low productivity (Blanchard, *et al.*, 2001).

In the upper intertidal area of Te Tāhuna o Rangataua, re-suspension would be limited due to the coupled effects of mangrove presence to the north-east and decreased flushing rates in the upper reaches. Animal grazing may be reduced in the immediate vicinity of the wastewater seepages, as a result of lowered salinity and increase in pollutants along the high tidal area. This was found for *Zeacumantus lutulentus*, a dominant grazer in the area. Figure 5.7 shows that closest to the wastewater treatment ponds and shoreline, where chlorophyll-α concentrations are at their highest, *Z. lutulentus* abundance is at its lowest. The SIMPER analysis of sites close to the WWT ponds and sites furthest away also found that *Z. lutulentus* was an important species in characterizing community assemblages at sites further away from the WWT ponds (Table 3.5) reiterating its increased absence closest to the shoreline. Without grazing pressure, the microphtyobenthos in the area has flourished to notably high concentrations.

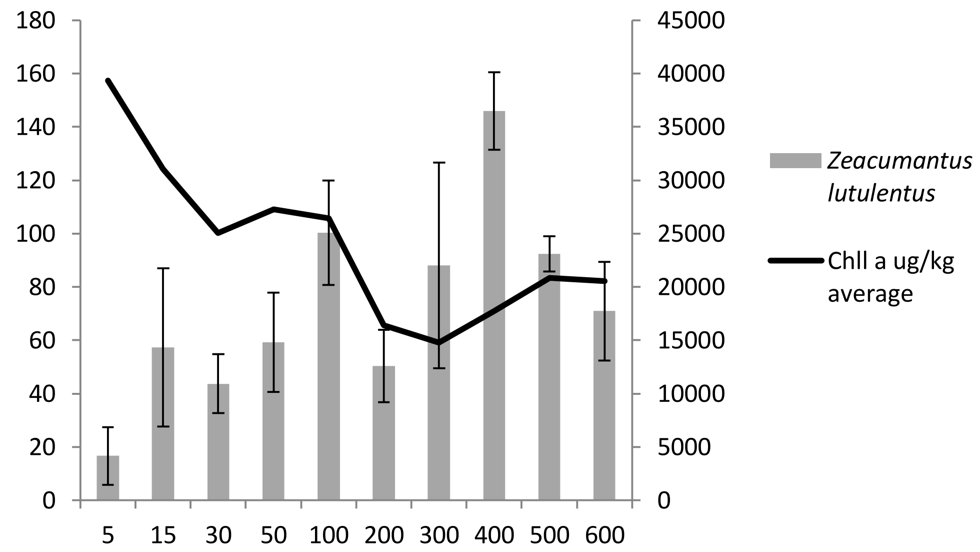

Figure 5.7: Average abundance across three transects (Y axis = $\bar{x}$ of ten 300 cm^3 core samples) of *Z. lutulentus,* with error bars (±SE) and chl-α at each distance (n=3, see Fig 3.6 for ±SE) from the shoreline in Te Tāhuna o Rangataua. First axis is organism abundance and second axis is chl-α.

Benthic microalgal communities are an important sink for nitrogen when they are highly productive, as a large amount is required for primary production. However, if microbenthic algal activity is reduced, their role as a sink is changed and they may become a source of nitrogen. As productivity becomes low and with a large biomass of already existing microbenthic algae, an increase in fresh organic matter eventuates once the algae dies (Conley*, et al.*, 2007). Microalgae enhances overall productivity within an area provided it is channelled to higher trophic levels (Van Damme, *et al.*, 2005).

5.5.1 Eutrophic conditions and anoxia

Of marine environments, estuaries are more susceptible to impacts of eutrophication, due to their often small volumes and enclosed nature (Hecky & Kilham, 1988). A study by Dauer, *et al.* (2000) which looked at benthic community composition and anthropogenic stressors in Chesapeake Bay, US, found that composition was most strongly related to low dissolved oxygen events. Low dissolved oxygen events were driven by high nutrient loads and resultant increases in eutrophication and were found to correlate with urban land use, from urban run-off, sewage and point source nutrient loading. There was however a disconnection between community condition and measures of eutrophic condition. Community composition was found to be only weakly correlated with measures used to indicate eutrophication (water column concentrations of nutrients and chl-α). The variability in chl-α concentration on a seasonal scale masked the link between benthic condition and chl-α production. Complex interactions between nutrient enrichment and phytoplankton biomass, increased organic deposition and lowered dissolved oxygen concentrations can be disconnected over space and time. The direct driving force in causing degradation in benthic communities within the study by Dauer, *et al.* (2000) was low dissolved oxygen levels.

Benthic algae may have positive or negative effects to organisms (Dyson *et al.*, 2007). Microalgal growth, in response to short term or long term nutrient inputs, may have differing effects to benthic condition. Short term nutrient enrichment and microalgal growth enhances benthic production by providing a source of food. Over longer time periods, biochemical oxygen demand is created by high nutrients and microalgae. An increase in organic matter and corresponding decomposition creates low dissolved oxygen concentrations which degrade benthic condition (Dauer, *et al.*, 2000).

During the summer months, a large proportion of decomposed benthic algae can accumulate in the sediment. This would lead to organic enrichment, which would increase oxygen uptake thus decreasing oxygen levels in the sediment. The general response of organisms under these conditions is decrease in suspension and surface deposit feeders and an increase in tolerant opportunistic organisms (Troell *et al.*, 2005). When sediments become anoxic for prolonged periods of time, most infaunal animals will disappear and breakdown of organic material within the sediment will shift from aerobic microbial degradation to the much slower process of anaerobic microbial degradation (Bianchi *et al.*, 2000). In these conditions, the sediments release previously-bound toxic hydrogen sulphide which many species cannot tolerate (Conley, *et al.*, 2007).

Dissolved oxygen concentrations are dependent on a number of factors including temperature, salinity, biological activity and other abiotic variables. Estuaries may become depleted in dissolved oxygen even under natural conditions. This can be intensified by organic matter from sources such as sewage effluent and dead plant material (ANZECC, 2000).

A study by Jordan *et al.* (1991) investigating nutrients in Rhode River Estuary, Maryland, Virginia found seasonal patterns of chl-α concentrations were generally opposite to those found for nitrogen, suggesting that seasonal depletion of nitrogen was occurring due to absorption by the phytoplankton. Jordan, *et al.* (1991) reported that phytoplankton production was found to consume nearly all of the dissolved nitrogen entering Rhode River estuary.

Nitrogen levels showed a range of values from low, moderate and slightly elevated within the Te Tāhuna o Rangataua study area, though TN values were not as high as one would expect given the high levels of chl-α (comparative to previous studies such as the broad scale survey of Tauranga Harbour (Ellis, *et al.*, 2013)). The high concentrations of chl-α closest to the shoreline would suggest that higher levels of nutrients may have also been in this area, prior to increased temperatures and light availability in the summer months. Assimilation of nitrogen by benthic microalgae to produce a large biomass, may have however depleted these levels, reflecting the trend found by (Jordan, *et al.*, 1991).

The slightly elevated levels of TN and TP found within the upper intertidal area suggests that the primary source of nutrients into the area is from the Wastewater treatment pond seepages. Microbenthic algae can assimilate nutrients from sediments and the water column, though nutrients released to the water column is capable of rapidly dispersing (Dyson, *et al.*, 2007).

Jordan, *et al.* (1991) also found that there was a seasonal flux in phosphate and that a large proportion of phosphate was bound to suspended particles within the water column. As well as this, deposited phosphate is suggested to be released from sediments most rapidly within the summer months due to the mineralization and dissolution of bound molecules on the seabed accelerating with increased temperature. Phosphate concentrations sampled from sediments during this time may not provide an accurate measurement of phosphate within the environment, due to the increase in phosphate concentrations released to the water column in the peak of summer (Jordan, *et al.*, 1991).

The SIMPER analysis of close (5 and 15 m) sites and far (500 and 600 m) sites found that nereid polychaetes *Ceratonereis* sp. and *Perineries nuntia var. vallata* were the dominant animals characterizing community composition within sites in the upper intertidal region. Neried polychaetes are a common fixture, found in high numbers in intertidal estuarine areas within New Zealand and this was reflected in the macrofaunal collection within Te Tāhuna o Rangataua. Nereids are known to be opportunistic organisms, colonizing areas with increased levels of organic content and having the ability to thrive in hypoxic conditions (Nkwoji, 2012), with the Nereididae family commonly described as anoxic and hypoxic tolerant (Ferreira-Cravo *et al.*, 2009).

From Figure 5.8 and 5.9, nereid distribution is generally higher in the upper intertidal, with small scale fluctuations. At a distance of 5 m, *Ceratonereis* sp. abundance is somewhat reduced and then shows an increase at 30 m where it then begins to decrease with distance from the shoreline. *Perineries nuntia var. vallata* also shows an increase at 50 m and subsequent decrease towards the low tidal area. As indicators of eutrophic or anoxic condition, TP, TN and chl-α are all at their highest values at a distance of 5 m. This could indicate that within the upper intertidal fringe, the area may be undergoing eutrophic conditions. Sediments may be oxygen depleted and as a response, community structure is devoid of certain organisms and opportunistic animals, such as the nereids found, occupy the area. It is interesting to note that even distribution of these anoxic tolerant animals appears to be slightly limited at a distance 5 m, in particular *Ceratonereis* sp. The production of toxic sulphides as a response of anoxic sediments, may be further limiting biotic growth in the area.

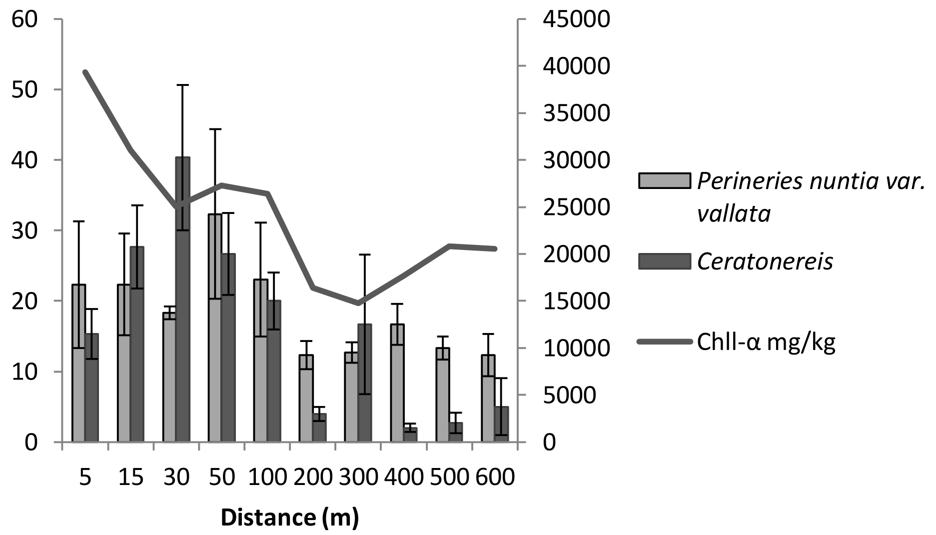

Figure 5.8: Average abundance (Y axis = $\bar{x}$ of ten 300 cm^3 core samples, n = 3) of *P. vallata* and *Ceratonereis* sp., with error bars (±SE) and Chl-α at each distance (n=3, see Fig 3.6 for ±SE) from the shoreline in Te Tāhuna o Rangataua. First Y axis is organism abundance and second Y axis is chl-α (mg/kg).

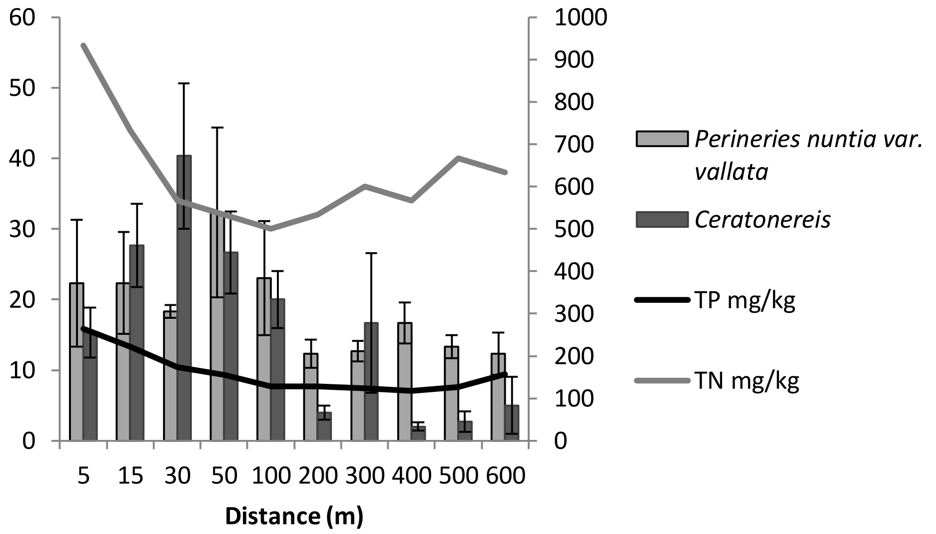

Figure 5.9: Average abundance (Y axis = $\bar{x}$ of ten 300 cm^3 core samples) across three transects, of *P. vallata* and *Ceratonereis* sp., with error bars (±SE) and Total Nitrogen and Total Phosphorus at each distance (n=3, *see* Fig 3.4 and 3.5 for ±SE) from the shoreline in Te Tāhuna o Rangataua. First Y axis is organism abundance and second axis is TN and TP (mg/kg).

5.5.2 The Redox Potential Discontinuity layer

The Redox Potential Discontinuity (RDP) depth is a useful measure for oxygen availability and microbial processes occurring within the sediment. A reduced RDP layer can give an indication of areas undergoing eutrophic conditions and anoxia. Increase in organic matter and algal growth can greatly reduce the depth of the RDP, decreasing productivity and sediment oxygenation and detrimentally effecting and displacing infuanal organisms. Sediments are more likely to become anoxic in muddy conditions, as diffusion through the fine particles is limited inhibiting oxygen penetration (Robertson & Stevens, 2011).

Measurements of Redox Potential profiles are considered reliable in reflecting sediment conditions, and in some cases more so than other variables (such as organic matter and chl-α) that are considered indicators of hypoxia (Pearson & Rosenberg, 1978), as redox potential layers show the long term and accumulative response to low oxygen and change in physico-chemical sediment structure. With low oxygen, redox potentials will show the sediment is under reducing conditions and unsuitable for animals. The threshold for dissolved oxygen saturation considered to represent the onset of hypoxia is 30% (Tay, *et al.*, 2012).

To further investigate whether anoxia is negatively affecting community structure and condition, measurements of the RDP layer would be very beneficial.

5.6 Summary

Apart from a small qualitative assessment of the Rangataua Area in 1974 (Larcombe, 1974), no quantitative baseline study of environmental characteristics was carried out prior to the reclamation and development of the Te Maunga Oxidation ponds and sewerage facility. Without this kind of information, it can only be postulated as to the extent of which reclamation and discharge of pollutants have changed the environment and its resident communities. From the data collected, a shift in community composition along a gradient of distance is evident and due to the various inputs and complex nature of hydrodynamics in the area, this change does not appear to be entirely reflective of intertidal gradients. From the statistical analysis and comparisons of specific species, composition appears to become less diverse and encompasses less productive animals closer to the shoreline, shifting to a composition of scavengers and opportunists reflective of a degraded environment. The following chapter will discuss the influence of sedimentation and input of pollutants in reducing productivity, with the upper intertidal area expected in the future to see increased deposition leading to a spread of mangrove growth in the area and associated loss of productivity.

Chapter 6

Estuarine Ecology, Environmental Management and Concluding Remarks

6.1 Biodiversity and ecosystem functioning

Compared with other marine ecosystems, estuaries are generally considered to be species poor (in terms of macro-fauna), yet, with the physico-chemical and biological interactions that occur within estuaries, they are functionally diverse. Species assemblages are intimately linked to the range of habitats that are characteristic of an estuary, with certain organisms altering the physical and chemical properties of the sediment and influencing ecosystem functioning (Thrush, *et al.*, 2013). Functional diversity in relation to species composition within an environment is suggested to be based on, not how many species are present but the behavioural traits of the species that are present and their functional roles in an ecosystem (Hewitt *et al.*, 2008).

There is a great importance in emphasising the intimate link between biodiversity and ecosystem services within marine environments, to create a universal understanding (for environmental management in particular) that any decrease in biodiversity will have negative consequences beyond the simple loss of species (Hewitt, *et al.*, 2008). There is a growing acknowledgment of positive relationships between biodiversity and processes within the environment that are vital to the services that ecosystems provide (Thrush, *et al.*, 2013). These relationships are however highly complicated and hard to quantify, where some services may be reliant on high biodiversity while others may be dependent on a few species or a specific functional group (Thrush, *et al.*, 2013).

Within soft sediments, large infaunal organisms are known to modify biogeochemical and particle gradients which, in turn, influence community composition. The effect macro-fauna have on nutrient and oxygen fluxes depends on their mode of feeding or activities within the sediment, for example if they subduct organic matter into the sediment or change concentrations of organic load at the surface (Thrush, *et al.*, 2006).

A study by Thrush, *et al.* (2006) looked at the effects of large organisms on sediment ecosystem functioning and community structure. Large infaunal organisms were removed from benthic communities, to assess changes in community structure and biochemical interactions. The experiment allowed assessment of chemical fluxes without the confounding environmental variables such as flow, grain size and tidal height. Oxygen production and nutrient uptake by microphytes was also investigated. Two functionally differing communities, a deposit-feeder-dominated community and suspension-feeder-dominated community were studied. Deep-burrowing deposit feeders (tellinid bivalve and polychaetes) and large suspension feeders (cockle) were removed from each of the communities, respectively.

It was hypothesised that, as a deposit feeder subducts organic matter into the sediments, this will stimulate microbial degradation and mineralizing processes within the sediment, aiding in the denitrification process. In contrast, suspension feeders produce biosolids that increase organic content at the sediment surface, adding to the detrital pool where oxygen is limited.

From the study by Thrush, *et al.* (2006) it was found that biodeposits from suspension feeders such as *Austrovenus stutchburyi* were much more important in biogeochemical fluxes in surficial sediments than first predicted. It was found that significant characteristics of dominant infauna included feeding mode, size, mobility and depth of feeding in relation to depth of defecation. One characteristic that was found to be most significant was size. Removal of larger animals showed distinct changes in community structure, such as increase in smaller opportunistic species such as polychaetes worms. Removal of both large deposit feeders and suspension feeders changed the structure of surface dwelling communities and biogeochemical fluxes (nutrient regeneration from organic content) important for productivity.

Within Te Tāhuna o Rangataua, the limited distribution of *M. liliana* and absence of *A. stutchburyi* will have important implications to the biophysical nature of the sediments and associated community structure. Thrush, *et al.* (2006) highlighted that larger infauna such as *M. liliana* and *A. stutchburyi* were important "ecosystem engineers", influencing fluxes of energy, matter and sediments within their habitats. Within Te Tāhuna o Rangataua, the loss of these animals may not only affect local productivity but ecosystems farther afield as well.

6.1.1 Bivalve distribution in Te Tāhuna o Rangataua

Distribution of bivalves in Te Tāhuna o Rangataua would have changed over the years due to both natural causes and anthropogenic stressors. A survey of the ecology of Tauranga Harbour was undertaken in the summer of 1990/91 by Environmental Bay of Plenty (Park & Donald, 1994), in which 10 sites were sampled for shellfish in Te Tāhuna o Rangataua. At each site, four core samples were taken which were 13 cm in diameter and extended 15cm into the sediment and animals retained on a 2 mm mesh were summed to site level.

Table 6.1: Bivalve shellfish data taken from the Tauranga Harbour Survey 1990/91 (Park & Donald, 1994) of 10 sites (four 13x15 core samples per site) within Te Tāhuna o Rangataua.

	Shellfish found in Te Tāhuna o Rangataua Bay over summer sampling period 1990/91		
Site	Cockle (*Austrovenus stutchburyi)*	Wedge shell (*Tellina liliana*)	Pipi (*Pahies australis*)
147	21	20	0
148	2	9	0
149	2	29	0
150	4	7	0
151	0	14	0
152	1	1	19
153	1	22	0
154	0	8	0
155	5	8	0
156	0	10	0
Total	36	128	19

Within the survey undertaken in 1990/91, 19 pipi were found at one site within the upper eastern area of Rangataua Bay, though nowhere else. This could be a chance recruitment, as they were absent from other sites and were not found in the current survey of the Rangataua area. Across the 30 sites within the current study, a total of 13 cockles were found. There did not appear to be any pattern along transects with only one or a few animals found, if present at a site. From the 1990/91 survey, cockle abundance appears to be slightly higher.

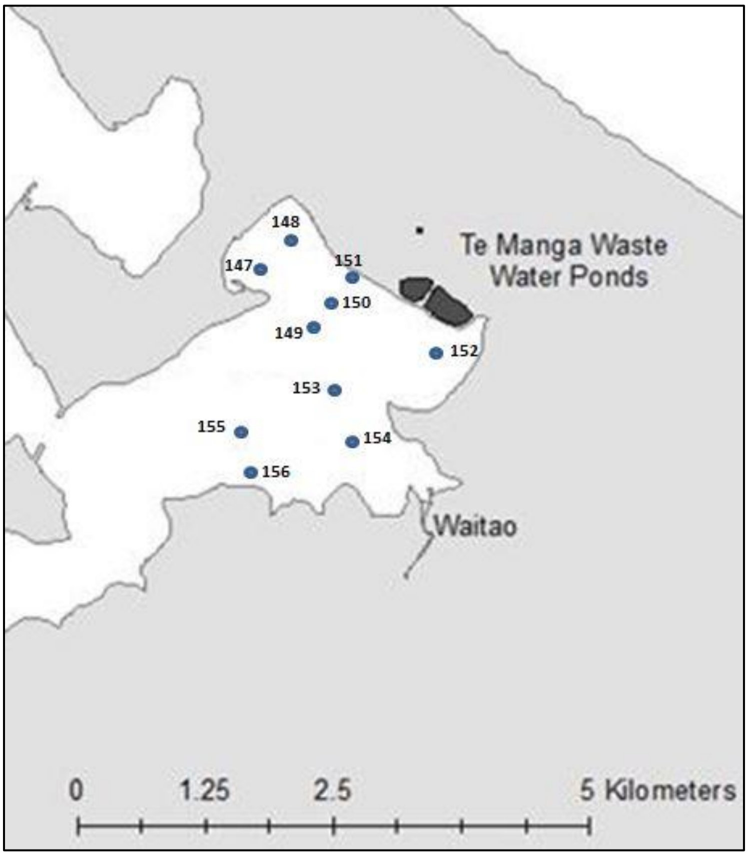

Figure 6.1: Sampling sites within Te Tāhuna ō Rangataua from the Tauranga Harbour Survey 1990/91 (Park & Donald, 1994).

From the current study, wedge shells no longer appear to live in the upper northern reaches of Rangataua, in which sites 147-149 from the 1990/91 survey were located. From this study, along Transect A (Figure 2.1) wedge shells only appeared at a distance of 300 m (n=3) from the shoreline, along transect B wedge shell was were found at a distance of 50 m away and along transect C wedge shell (n=1) was found at a distance of 30 m (n=1). These results vary considerably from the 1991 survey in which wedge shells were found within all sites in the upper reaches, with some sites appearing to have high counts of the shellfish. Recruitment of wedge shell to the upper intertidal fringe has diminished from 1991 to the present day.

6.1.1.1 Wedge shell ecology

As discussed earlier, *M. liliana* is a dominant bivalve considered to be an "ecosystem engineer" and *M liliana's* distribution is currently restricted to the lower intertidal region of Te Tāhuna o Rangataua. The ecology of this large benthic inverterbrate is important in influencing benthic condition within the area.

M. Liliana is a tellinid bivalve that has previously been named *Tellina liliana* but is recognized now as its own genus (Roper *et al.*, 1992). A study by Roper, *et al.* (1992) investigated the population biology of *M. liliana* within an estuarine area of Manukau Harbour, New Zealand. The results of the study by Roper, *et al.* (1992) indicated that juveniles colonising an area seemed to be related to the number of adults present, with highest recruitment occurring where adult densities were highest. Roper, *et al.* (1992), suggested that even though areas were similar in physical characteristics, some sites provided a better habitat for the species. Presence of adults may facilitate juvenile recruitment to an area. Tellinid bivalves have been previously found to be sensitive to pollution (Roper, 1990). Hydrogen sulphide has been found to negatively affect *Macoma balthica* individuals and population structure (Bonsdorff & Wenne, 1989). Aggressive polychaetes and amphipods have also been found to predate on juvenile *M. balthica*, though none such relationship was discerned with density of *M. liliana* within the study sites of Manukau Harbour (Roper, *et al.*, 1992).

As the survey from 1991 shows, *M. liliana* was once present in the upper intertidal fringe and the area adjacent to the WWT ponds. Distribution of *M. liliana* and *A. stuchburyi* was not restricted by tidal elevation or exposure, leading to the conclusion that change in environmental condition has caused displacement of these large bioturbators in the upper intertidal areas of the bay adjacent to the WWT ponds, in which they are no longer present. *Macamona liliana* are important surface-deposit feeders within communities. *A. stutchburyi* is a highly mobile bivalve which plays important roles in bioturbation, bulldozing surficial sediments, destabilising sediments and allowing for oxygenation of upper surface layers (Thrush, *et al.*, 2006)

Displacement of these species may be due to a number of environmental conditions. These include increase in eutrophic conditions (as indicated by TN, TP and chl-α (Figure 7.2)), increased pollutants and accumulation of mud leading to increased mangrove growth, with sediments in the upper intertidal reaches moving towards a state of un-inhabitability.

With the loss of these large and functionally important animals, sediment structure and species composition would have become further degraded. Understanding the changes which lead to the depletion of species that play positive roles in ecosystem functioning is key to understanding long-term environmental degradation (Thrush, *et al.*, 2004).

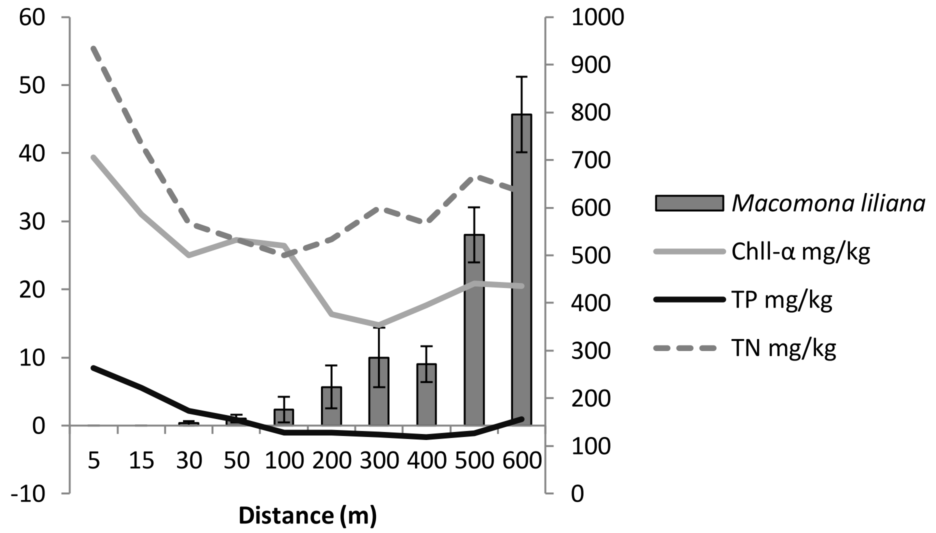

Figure 6.1: Average abundance across transects (n= 3, Y axis = $\bar{x}$ of ten 300 cm^3 core samples) of *Macomona liliana* with error bars (±SE), Total Nitrogen,Total Phosphorus and chl-α at each distance (n=3, see Fig 3.4,3.5 and 3.6 for ±SE) from the shoreline. First axis is organism abundance and second axis is TN, TP and chl-α mg/kg.

6.1.2 Mangrove ecology

Within Te Tāhuna o Rangataua, mangrove expansion has occurred within the upper intertidal area, adjacent to the WWT ponds in the years following reclamation of the area. This growth would have important implications to system dynamics within the whole of the area.

Land use change associated with agriculture, forestry and urbanization have increased sediment inputs into estuarine areas for many years and the has led to a corresponding spread in estuaries previously dominated by sandy sediments

(Lovelock *et al.*, 2007). Inputs of fine silt sediments are often closely linked to increased nutrient and contaminant pollution.

A study was undertaken by Lovelock, *et al.* (2007) to understand the coupling effects of nutrient enrichment and sedimentation at differing levels on the growth of mangrove forests. To investigate this, experimental nutrient enrichment was used in two mangrove areas with low and high degrees of sedimentation. It was found that within the area of high sedimentation, nutrient enrichment lead to a significant increase in leaf production comparative to the low sedimentation site, indicating that sedimentation enhances growth. To support this pattern, it was also found that during the winter months, the site with high sedimentation had sustained growth, while the area with low sedimentation showed negative growth. Other environmental factors may have also contributed to the results that were found. The study predicted that increased nutrients at the low sedimentation site would enhance growth more than at the high sedimentation site, though higher productivity was not found. This indicated that mangrove growth is slow in areas with low sedimentation compared with areas with high sedimentation and that, within areas of high sedimentation, nutrient enrichment is not the sole factor contributing to increased growth.

Lovelock, *et al.* (2007) suggested that mangrove growth within New Zealand may be due to the increase of suitable habitats being made available, either naturally or by human activities, coupled with increasing sedimentation. Sedimentation may alter mangrove receptiveness to increased nutrients as well as altering nutrient and carbon cycling. It was found from the study that internal nutrient conservation within the plants indicated that P availability was also higher at the high sedimentation site. Plant internal P recycling percentages were found to be lower at the high sedimentation site and higher at the low sedimentation site, which is expected if the low sedimentation is deficient in P, while the high site is replenished. This suggested that sedimentation enhances P availability thus allowing, along with nitrogen enrichment, enhanced growth.

6.1.2.1 Differing perceptions on mangrove growth

While the growth of mangroves are often perceived as important in supporting biodiversity, estuarine modification and increased inputs of sediment has over the years seen accelerated colonization and growth of mangroves in New Zealand, in areas which may not have been colonized naturally. This appears to be the case within the upper intertidal area of Te Tāhuna o Rangataua, with the accumulation of fine silts and nutrient inputs facilitating the spread of mangroves. In this respect, mangrove growth may displace other habitats such as sand-flats, mudflats and seagrass beds (Morrisey *et al.*, 2003).

The structure and function of mangrove stands have been found to differ with age. In a comparison of younger and older mangrove stands Morrisey, *et al.* (2003) found that there were more taxa in the younger, than the older stands. Different faunal composition coincided with differences in sediment characteristics between older and younger areas. Older mangrove stands had sediments which were more compacted and contained more organic matter and more pneumatophores, which was presumably due to highly reduced oxygen in the sediment. It was suggested that as mangroves become denser, sediments become more compacted and is inundated by the tide less frequently, leading to a decrease in faunal diversity and abundance, correlating with a decrease in sediment quality.

The invasion and extension of mangroves within New Zealand has affected many services valued by the public, as well as less obvious ecosystem services, due to displacement of other habitats and functionally important species (Thrush, *et al.*, 2013).

Due to increased sediment load within Te Tāhuna o Rangataua that has been predicted to occur in the future (Hume, *et al.*, 2009), coupled with the relatively long residence times and the further reduced tidal influence within mangrove stands (Morrisey, 1993), mangrove expansion within Te Tāhuna ō Rangataua seems highly likely in the future. As the mangroves become older and expand outward from the upper intertidal area, sediments may become increasingly muddy, compacted, eutrophic and degraded, exhibiting a corresponding loss of organisms. Due to further displacement of important fauna and habitats, this loss would not be limited to the mangrove stands but may extend to the entire intertidal area, with significant losses in biodiversity leading to an increasingly unproductive and degraded environment.

6.1.3 Environmental Management

Environmental implications of degraded estuarine areas have over the years been made clear. These include homogenisation of communities and ecosystems, decreased biodiversity and reductions in food-web complexities and decrease in organism size. Multiple stressors often interact in a way which can change environmental condition faster than animals and habitats can adapt. Ecological thresholds exist and may be reached due to subtle, chronic and cumulative impacts, in which if they are exceeded a fundamental shift in ecosystem processes will occur (Thrush, *et al.*, 2013).

In highlighting the magnitude or scope of a problem within an environment, it is important for ecologists to provide information to aid managers in minimizing risks and threats to ecological integrity. Ecological integrity is considered to incorporate the functionality and self-maintenance of an ecosystem (Thrush, *et al.*, 2013). Knowledge found through research may be most useful when applied to define thresholds to which detrimental effects may occur and predicting ecological responses. An important aspect of estuarine management is considering risks at the source of the problem, in particular land use and development (Thrush, *et al.*, 2004).

The need for an integrated, whole system approach to restoration of estuarine functioning is becoming widely recognized (Van Damme, *et al.*, 2005). An integral part of understanding ecosystem processes is recognizing knowledge gaps and establishing and developing monitoring programmes accordingly. To accomplish this, there is a need for an integrated monitoring programme which requires monitoring of not only the marine intertidal area, but inputs from all sources that influence an area. Spatial sampling should be sufficient to allow assessment of variability across an ecosystem (Van Damme, *et al.*, 2005).

Regional councils within New Zealand are focused on improving cost-effective assessments, by linking broad-scale surveys, which provides information in a broader context, and time series information, which provides information on natural variability versus variability as a response of anthropogenic stressors. Changes in ecological services that are provided by estuaries are often undocumented, which makes defining baselines against which to develop evidence based restorative management difficult (Thrush, *et al.,* 2013).

Careful land management would involve finding appropriate trade-offs between risks involved in land development and maintenance of ecological value, which is a difficult task. Sediment run off and hydrodynamic models, coupled with critical deposition thresholds, may predict ecological risk under differing levels of development and mangers can compare these risks and improve decision making. Change in abundance and distribution of sensitive species may also be modelled, with their responses to varying environmental condition highlighting species which are useful indicators of change. Management actions can occur before development stages begin, either at the site of development or through restoration and enhancement of buffering habitats within receiving environments (Thrush, *et al.*, 2004).

Habitat preference, recruitment patterns and species associations are complex and often studies may find altogether different responses and trends of species. What appears to be most important is that community structure is very dependent on the distinctive environment within which it is formed and elucidating information from results is dependent on collection methods employed. Confounding comparisons that may occur between studies are caused by methodological factors such as mesh size used to sieve macro-fauna, number of replicates collected within a site and of most importance, confusion amongst taxonomy (Pridmore, *et al.*, 1990). For taxonomic purposes, taking into account feeding habits may be of more use than specific species, although plasticity in feeding behaviour may also prove to create difficulties in assigning animals to functional groups.

The Estuarine Monitoring Protocol found that infaunal assemblages tended to characterize estuaries better than epifaunal assemblages, which is important in developing indicators and to facilitate the development of a "national biotic health index to assess biological condition of New Zealand estuaries". Limitations when using epifaunal assemblages as a measure of estuarine health are due largely to epifaunal species mobility. Animal assemblages living on the sediment surface can differ dependent on stage of tidal cycle, time of day and due to change in weather conditions. Bivalves which are not buried may also fall under this category (Robertson, *et al.*, 2002). Multivariate species responses may often give better indication of environmental condition as they are more sensitive to environmental change than a single species as an indicator (Clarke, 1993).

6.2 Concluding remarks

The management decisions which led to the reclamation of Te Tāhuna o Rangataua and the discharging of wastes into the area is suggested to have been based on the perception that marine ecosystems were highly resilient, open environments which could receive such pollution with minimal lasting effects. In terms of benthic community health, there has been a long lasting perception that marine systems had a high potential for dispersal and connectivity between habitats. There is growing evidence that the dispersal ability between habitats of many benthic species is limited (Thrush *et al.*, 2008). Limitations for dispersal within the Te Tāhuna o Rangataua may be exacerbated, due to the area being enclosed and shallow in nature. The loss of habitats may have, in turn, led to the loss of many local populations.

The wastewater ponds have been in place for many years now. When first established, the reclamation and corresponding removal and damaging of habitats in the Rangataua area would have had devastating and immediate effects to localized fauna. These effects would have been apparent and observable. What may not have been as observable is the cumulative impacts of the ponds over the years, which may have induced chronic effects to surrounding population. The loss of important habitats would have led to a reduction of overall biodiversity and indiscriminate pollutants may have had subtle chronic effects to biota.

The collection of information for the current study suggests that current input of pollution from the ponds is localized to the area directly adjacent to the ponds. What is much harder to determine, is any long lasting, subtle effects of the treatment ponds to the area.

Of great concern and not investigated within this survey, is the possibility of microbiological contamination to the area and to shellfish from the wastewater seepages. It would be expected that faecal coliforms would be found within macro-fauna, especially those located in the direct vicinity of the seepages. Currently, Manaaki taha Moana are investigating faecal coliform concentrations within an invertebrate species of importance to local iwi, titiko (*Amphibola crenata*) in the Te Tāhuna o Rangataua. The mud snail, which in early years (pre-reclamation) was collected in abundance, was an important source of food in Te Tāhuna o Rangataua. Since the placement of the ponds, titiko have not been collected for consumption from this area.

This study provides an in depth ecological base line study of the biological and physical characteristics of Te Tāhuna o Rangataua. Further efforts are required to fully understand ecological processes and trophic functioning that may be inhibited due to the presence of the WWT ponds. Many factors are likely to contribute to the observed differences in macro-faunal assemblages, not all pertaining to the presence of the ponds.

The results of the current fine scale ecological survey did not show evidence of heavy metal contamination build up within sediments to levels of concern of which immediate action is required to be taken. Although they are not at a toxic level now, the area may be prone to contaminant issues in the future, due to the area acting as a trap for terrestrial derived sediments and accumulation that may occur within these depositions.

A sediment survey of Tauranga Harbour conducted by Environment Bay of Plenty (now Bay of Plenty Regional Council) in 2008 highlighted the need to address management of storm water contaminants entering the harbour. Currently in the Bay of Plenty, as part of storm water discharge consent requirements, regular background monitoring programs are in place. Park (2008) suggests more intensive monitoring of the more sensitive receiving environments around marine storm water outfalls should also be undertaken.

6.2.1 Aims and objectives revisited

This study highlights that hydrodynamics and geomorphology of an estuary are key components to understanding the degree to which pollutants and sediments entering an area will affect benthic biodiversity and environmental condition.

The cumulative effects of sedimentation and nutrient enrichment within the upper intertidal fringe of Te Tāhuna o Rangataua may be causing degradation of sediment condition and resulting in loss of biodiversity. Key environmental parameters which may be driving this change include nutrients (nitrogen and phosphorus), fine silt sediments, corresponding benthic algal growth, increased organic matter and possible hypoxia within the sediments.

Significant spatial variation in abundances of individual taxa was found along a gradient of distance from the Wastewater treatment ponds (upper intertidal fringe) and differences in macro-faunal distribution was apparent between sites closest to the ponds and sites further away. Key benthic invertebrates which indicate change in community structure along a gradient of distance include the bivalve *Macomona liliana*, found within the low intertidal area and opportunistic polychaete worms and amphipods such as *Ceratonereis* sp., *Perinieries nuntia var. vallata* and Corophiidae, within the upper intertidal fringe.

To elucidate cause and effect of these changes is a complicated task. From this study, the observed differences between sites are not overly apparent, suggesting a complicated ecosystem structuring dynamic is occurring. For future study in the Rangataua area, incorporating measurements of oxygen saturation, the Redox Potential Discontinuity layer and salinity would greatly benefit understanding of microbial processes, sediment productivity and give a better indication of anoxic areas or areas exhibiting eutrophic conditions. Unexplained variability in species distribution is often a consequence of the fact that unmeasured parameters are also limiting abundances and interacting with measured variables in complex way (Anderson, 2008).

Although spatial variability is taken into account, there is no data that matches the fine scale survey undertaken within the study, to assess temporal variability of environmental factors and macrof-aunal assemblages. This was beyond the scope of the study time frame. Future monitoring will provide information on temporal variability, which may reflect responses to increase or decrease in certain stressors. Continuous monitoring data is required to discern anthropogenic influences from natural variability, to understand the effects of human activities on water resources (Van Damme, *et al.*, 2005). It is recommended that any future monitoring follow the same gradient style sampling adopted here, although less sites may prove to be enough to portray the area meaningfully and collect significant data; given that time, management requirements and budgetary constraints may not be able, or it may not be necessary, to replicate this study on the same scale.

This study set out to examine the environmental and ecological condition of Te Tāhuna o Rangataua against a backdrop of known anthropogenic stressors that have previously and currently impacted the area. The monitoring design was aimed specifically at assessing the impacts of the Te Maunga Wastewater Ponds and treated wastewater seepages. Biological activity was found to be reduced in the upper intertidal fringe, adjacent to the ponds and a change in macrobenthic community structure was found along a gradient of distance, indicative of changes one would find at distances from a degraded area. This study highlights the complexities of studying environments with such high natural variability and the need to move towards integrated ecosystem management approaches.

References

Anderson, M., Gorley, R. N., & Clarke, R. K. (2008). *Permanova+ for Primer: Guide to Software and Statistical Methods*. Plymouth, PRIMER-E Ltd.

Anderson, M. J. (2008). Animal-sediment relationships re-visited: Characterising species' distributions along an environmental gradient using canonical analysis and quantile regression splines. *Journal of Experimental Marine Biology and Ecology, 366*(1–2), 16-27.

Anderson, M. J., Ford, R. B., Feary, D. A., & Honeywill, C. (2004). Quantitative measures of sedimentation in an estuarine system and its relationship with intertidal soft-sediment infauna. *Marine Ecology Progress Series, 272*, 33-48.

Anderson, M. J., Hewitt, J. E., Ford, R. B., & Thrush, S. F. (2006). *Regional models of benthic ecosystem health: predicting pollution gradients from biological data.* Auckland Uniservices, University of Auckland prepared for Auckland Regional Council. 103pp.

ANZECC, A. (2000). *Australian and New Zealand guidelines for fresh and marine water quality.* Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand, Canberra, ACT, Australia, 1-103.

Atalah, J., & Crowe, T. P. (2012). Nutrient enrichment and variation in community structure on rocky shores: The potential of molluscan assemblages for biomonitoring. *Estuarine, Coastal and Shelf Science, 99*, 162-170.

Bennett, C., Gillespie, P., & Keeley, N. (2006). *Horoirangi Marine Reserve soft sediment baseline ecological survey.* Cawthron Report No. 1186. Prepared for Department of Conservation. 15 pp.

Bianchi, T. S., Johansson, B., & Elmgren, R. (2000). Breakdown of phytoplankton pigments in Baltic sediments: effects of anoxia and loss of deposit-feeding macrofauna. *Journal of Experimental Marine Biology and Ecology, 251*(2), 161-183.

Blanchard, G., Simon-Bouhet, B., & Guarini, J.-M. (2002). Properties of the dynamics of intertidal microphytobenthic biomass. *Journal of the Marine Biological Association of the UK, 82*(06), 1027-1028.

Blanchard, G. F., Guarini, J.-M., Orvain, F., & Sauriau, P.-G. (2001). Dynamic behaviour of benthic microalgal biomass in intertidal mudflats. *Journal of Experimental Marine Biology and Ecology, 264*(1), 85-100.

Bonsdorff, E., & Wenne, R. (1989). A comparison of condition indices of Macoma balthica (L.) from the northern and southern Baltic Sea. *Netherlands journal of sea research, 23*(1), 45-55.

Botherway, K. J., & Gardner, J. P. (2002). Effect of storm drain discharge on the soft shore ecology of Porirua Inlet, New Zealand. *New Zealand Journal of Marine and Freshwater Research, 36*(2), 241-255.

Bradley, J., Cameron, D., Evans, M., Holland, R., Hunter, P., & Mikaere, B. (2004). *Wastewater Consents Project: Assessment of Effects on the Environment. Project Number - 801/004480-10.* Tauranga City Council. 434pp.

Carpenter, S. R., Caraco, N. F., Correll, D. L., Howarth, R. W., Sharpley, A. N., & Smith, V. H. (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological applications, 8*(3), 559-568.

Casado-Martinez, M. C., Smith, B. D., & Rainbow, P. S. (2013). Assessing metal bioaccumulation from estuarine sediments: comparative experimental results for the polychaete Arenicola marina. *Journal of Soils and Sediments, 13*(2), 429-440.

Clarke, K. R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian journal of ecology, 18*(1), 117-143.

Clarke, K. R., & Warwick, R. M. (2001). Change in marine communities: an approach to statistical analysis and interpretation. Plymouth, PRIMER-E Ltd.

Conley, D. J., Carstensen, J., Ærtebjerg, G., Christensen, P. B., Dalsgaard, T., Hansen, J. L., & Josefson, A. B. (2007). Long-term changes and impacts of hypoxia in Danish coastal waters. *Ecological Applications, 17*(sp5), S165-S184.

Connell, J. H. (1978). Diversity in tropical rain forests and coral reefs. *Science, 199*(4335), 1302-1310.

Dauer, D. M., Ranasinghe, J. A., & Weisberg, S. B. (2000). Relationships between benthic community condition, water quality, sediment quality, nutrient loads, and land use patterns in Chesapeake Bay. *Estuaries, 23*(1), 80-96.

de Juan, S., & Hewitt, J. (2014). Spatial and temporal variability in species richness in a temperate intertidal community. *Ecography, 37*(2), 183-190.

Dean, H. K. (2008). The use of polychaetes (Annelida) as indicator species of marine pollution: a review. *Revista de Biolgia Tropical , 56*(4), 11-38.

Dronkers, J. (1986). Tidal asymmetry and estuarine morphology. *Netherlands Journal of Sea Research, 20*(2), 117-131.

Dyson, K. E., Bulling, M. T., Solan, M., Hernandez-Milian, G., Raffaelli, D. G., White, P. C., & Paterson, D. M. (2007). Influence of macrofaunal assemblages and environmental heterogeneity on microphytobenthic production in experimental systems. *Proceedings of the Royal Society B: Biological Sciences, 274*(1625), 2547-2554.

Ellis, J., Clark, D., Hewitt, J., Taiapa, C., Sinner, J., Patterson, M., Hardy, D., Park, S., Gardner, B., Morrison, A., Culliford, D., Battershill, C., Hancock, N., Hale, L., Asher, R., Gower, F., Brown, E., & McCallion, A. (2013). *Ecological Survey of Tauranga Harbour.*, Manaaki Taha Moana Research Report No. 13. Cawthron Report No. 2321. Prepared for Manaaki Taha Moana.56p. plus appendices.p.

Ellis, J. I., & Schneider, D. C. (1997). Evaluation of a gradient sampling design for environmental impact assessment. *Environmental Monitoring and Assessment, 48*(2), 157-172.

Ferreira-Cravo, M., Ventura-Lima, J., Sandrini, J. Z., Amado, L. L., Geracitano, L. A., Rebelo, M., Bianchini, A., & Monserrat, J. M. (2009). Antioxidant responses in different body regions of the polychaeta< i> Laeonereis acuta</i>(Nereididae) exposed to copper. *Ecotoxicology and environmental safety, 72*(2), 388-393.

Giangrande, A., Licciano, M., & Musco, L. (2005). Polychaetes as environmental indicators revisited. *Marine Pollution Bulletin, 50*(11), 1153-1162.

Gibbons-Davies, J. (2013). *Consent 62881 Annual report of titiko monitoring and seepages into Rangataua Bay 2013. TCC Ref: 6323110.* City Waters, Tauranga City Council. 23pp.

Gibbs, M. M., & Hewitt, J. E. (2004). *Effects of sedimentation on macrofaunal communities: a synthesis of research studies for ARC.* Technical Paper 294. NIWA Client Report: HAM2004-060, Hamilton

Gillespie, P., Clark, K., & Conwell, C (2007) *Waimea Estuary State of Envrinmental Monitoring: Fine-scale Benthic Assessment, April 2006.*

Gillespie, P., D, C., Clark, D., & Asher, R. (2012a). *Nelson Haven Fine-Scale Benthic Baseline 2012.* Cawthron Report N0. 2209.Prepared for Nelson City Council. 22pp.

Gillespie, P., Forrest, R., Clark, D., & Asher, R. (2012b). *Coastal effects of the Nelson (Bell Island) regional sewerage discharge: Benthic monitoring survey 2011* Prepared for Nelson Regional Swewrage Business Unit. Cawthron report No. 2068. 24 pp.

Grant, A., & Middleton, R. (1998). Contaminants in sediments: Using robust regression for grain-size normalization. *Estuaries, 21*(2), 197-203.

Harrison, S. J., & Phizacklea, A. P. (1987). Temperature fluctuation in muddy intertidal sediments, Forth Estuary, Scotland. *Estuarine, Coastal and Shelf Science, 24*(2), 279-288.

Hecky, R., & Kilham, P. (1988). Nutrient limitation of phytoplankton in freshwater and marine environments: a review of recent evidence on the effects of enrichment. *Limnology and Oceanography, 33*(4), 796-822.

Hewitt, J. E., Thrush, S. F., & Dayton, P. D. (2008). Habitat variation, species diversity and ecological functioning in a marine system. *Journal of Experimental Marine Biology and Ecology, 366*(1–2), 116-122.

Hume, T., & Swales, A. (2003). How estuaries grow old. *NIWA Water and Atmosphere, 11*(1), 14-15.

Hume, T. M., Green, M. O., & Elliot, S. (2009). *Tauranga Harbour Sediment Study: Assesment of predictions for management.* NIWA Client Report: HAM2009-139. Prepared for Environment Bay of Plenty. 116pp.

Ingalls, A. E., Aller, R. C., Lee, C., & Sun, M.-Y. (2000). The influence of deposit-feeding on chlorophyll-a degradation in coastal marine sediments. *Journal of marine research, 58*(4), 631-651.

Jackson, N. L. (2013). 10.12 Estuaries. In J. F. Shroder (Ed.), *Treatise on Geomorphology* (pp. 308-327). San Diego: Academic Press.

Jordan, T. E., Correll, D. L., Miklas, J., & Weller, D. E. (1991). Nutrients and chlorophyll at the interface of a watershed and an estuary. *Limnology and Oceanography, 36*(2), 251-267.

Keeney, D. B., Szymaniak, A. D., & Poulin, R. (2013). Complex genetic patterns and a phylogeographic disjunction among New Zealand mud snails Zeacumantus subcarinatus and Z-lutulentus. *Marine Biology, 160*(6), 1477-1488.

Larcombe, M. F. (1974). *Ecological Impact Report - siting of Oxidation ponds, Welcome Bay*. Auckland, Bioresearches Ltd; Steven and Fitzmaurice Consulting and Sanitary Engineers. .

Le Hir, P., Roberts, W., Cazaillet, O., Christie, M., Bassoullet, P., & Bacher, C. (2000). Characterization of intertidal flat hydrodynamics. *Continental Shelf Research, 20*(12), 1433-1459.

Lovelock, C. E., Feller, I. C., Ellis, J., Schwarz, A. M., Hancock, N., Nichols, P., & Sorrell, B. (2007). Mangrove growth in New Zealand estuaries: the role of nutrient enrichment at sites with contrasting rates of sedimentation. *Oecologia, 153*(3), 633-641.

Madarasz, A. (2006). *Ahuriri Estuary: Environmental Assessment and Monitoring*. EMI 06/16, HBRC Plan Number 3872.

Maurer, D., Nguyen, H., Robertson, G., & Gerlinger, T. (1999). The Infaunal Trophic Index (ITI): Its suitability for marine environmental monitoring. *Ecological Applications, 9*(2), 699-713.

McArdle, B. H., & Anderson, M. J. (2001). Fitting multivariate models to community data: a comment on distance-based redundancy analysis. *Ecology, 82*(1), 290-297.

McLusky, D., Hull, S., & Elliott, M. (1993). Variations in the intertidal and subtidal macrofauna and sediments along a salinity gradient in the upper Forth Estuary. *Netherland Journal of Aquatic Ecology, 27*(2-4), 101-109.

Morrisey, D., Skilleter, G., Ellis, J., Burns, B., Kemp, C., & Burt, K. (2003). Differences in benthic fauna and sediment among mangrove (Avicennia marina var. australasica) stands of different ages in New Zealand. *Estuarine, Coastal and Shelf Science, 56*(3), 581-592.

Morrisey, D. J. (1993). Environmental impact assessment—A review of its aims and recent developments. *Marine Pollution Bulletin, 26*(10), 540-545.

Nkwoji, J. A. (2012). Adaptions of three Nereid polychaetes to hypoxia in their environment. *Journal of Basic and Applied Scientific Research, 2*(3), 2828-2833.

Norkko, A., Talman, S., Ellis, J., Nicholls, P., & Thrush, S. (2002a). *Macrofaunal Sensitivity to Fine Sediments in the Whitford Embayment*. NIWA Client Report: ARC01266/2 prepared for Auckland Regional Council, Technical Publication No. 158. . 39pp.

Norkko, A., Thrush, S. F., Hewitt, J. E., Cummings, V. J., Norkko, J., Ellis, J. I., Funnell, G. A., Schultz, D., & MacDonald, I. (2002b). Smothering of estuarine sandflats by terrigenous clay: the role of wind-wave disturbance and bioturbation in site-dependent macrofaunal recovery. *Marine Ecology Progress Series, 234*, 23-42.

Nybakken, J. W., & Bertness, M. D. (2005). *Marine biology: an ecological approach - 6th edition*: Pearson Education, Inc., publishing as Benjamin Cummings.

Park, S. (2003). *Marine Sediment and Contaminants Survey (2001-03) of Tauranga Harbour.* ISSN 1775-9372, Environment Bay of Plenty.

Park, S. (2009). *Bay of Plenty Marine Sediment Contaminants Survey 2008.* Environmental Publication 2009/01.Environment Bay of Plenty. 24pp.

Park, S. (2011). *Sea lettuce and Nutrient Monitoring in Tauranga Harbour 1991-2010.* Bay of Plenty Regional Council. Environmental Publication 2011/06. 54p.p.

Park, S., & Donald, R. (1994). *Environment BOP Tauranga Harbour Regional Plan Environmental Investigations. Ecology of Tauranga Harbour.* Environment Bay of Plenty. Environment Report number 94/8. 177p.p.

Pearson, T., & Rosenberg, R. (1978). Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanogr. Mar. Biol. Ann. Rev, 16*, 229-311.

Pelletier, M. C., Gold, A. J., Heltshe, J. F., & Buffum, H. W. (2010). A method to identify estuarine macroinvertebrate pollution indicator species in the Virginian Biogeographic Province. *Ecological Indicators, 10*(5), 1037-1048.

Peterson, C. H. (1991). Intertidal Zonation of Marine-Invertebrates in Sand and Mud. *American Scientist, 79*(3), 236-249.

Pridmore, R. D., Thrush, S. F., Hewitt, J. E., & Roper, D. S. (1990). Macrobenthic community composition of six intertidal sandflats in Manukau Harbour, New Zealand. *New Zealand journal of marine and freshwater research, 24*(1), 81-96.

Read, G. B. (1984). Persistence of infaunal polychaete zonation patterns on a sheltered, intertidal sand flat. *New Zealand journal of marine and freshwater research, 18*(4), 399-416.

Riedel, G. F., Sanders, J. G., & Osman, R. W. (1987). The effect of biological and physical disturbances on the transport of arsenic from contaminated estuarine sediments. *Estuarine, Coastal and Shelf Science, 25*(6), 693-706.

Risgaard-Petersen, N. (2003). Coupled nitrification-denitrification in autotrophic and heterotrophic estuarine sediments: On the influence of benthic microalgae. *Limnology and Oceanography, 48*(1), 93-105.

Robertson, B., Gillespie, P., Asher, R., Frisk, S., Keeley, N., Hopkins, G., Thompson, S., & Tuckey, B. (2002). *Estuarine environmental assessment and monitoring: a national protocol. Part A. Development, Part B. Appendices, and Part C. Application.*, Prepared for supporting Councils and the Ministry for the Environment, Sustainable Management Fund Contract No. 5096. Part A. 93p. Part B. 159p. Part C. 40p plus field sheetsp.

Robertson, B., & Robertson, B. (2014). *Havelock Estuary Fine Scale Monitoring 2013/14.* Wriggle Coastal Management Ltd. Prepared for Marlborough District Council. 43pp.

Robertson, B., & Stevens, L. (2010). *Haldane Estuary. Fine Scale Monitoring 2009/10.* Nelson, Prepared for Environment Southland by Wriggle Coastal Managemnt.

Robertson, B., & Stevens, L. (2011). *Clifton Wastewater Treatment Plant Discharge. Consent Monitoring - New River Estuary Sediments* Prepared for Invercargill City Council. Wriggle Coastal Management. 27 pp.

Robertson, B., & Stevens, L. (2013). *Moutere Inlet. Fine Scale Monitoring 2013/14.* Nelson, Prepared for Tasman District Counsel by Wriggle Coastal Management 35p.

Roper, D. (1990). Benthos associated with an estuarine outfall, Tauranga Harbour, New Zealand. *New Zealand journal of marine and freshwater research, 24*(4), 487-498.

Roper, D., Pridmore, R., & Thrush, S. (1992). Recruitment to the macrobenthos of Macomona liliana (bivalvia: Tellinidae) in Manukau Harbour, New Zealand. *New Zealand journal of marine and freshwater research, 26*(3-4), 385-392.

Scholes, P., & McIntosh, J. (2009). *Water Quality of Bay of Plenty Rivers 1989-2008.* Environmental Publication 2009/11.Environment Bay of Plenty. 79pp.

Seabourne, M. (2009). *Mangatawa Drain Flood Bypass and TEM stormwater AEE.* New Zealand Transport Agency and Tauranga City Council. Project Number: 3932036. 354p.p.

Smedley, P., & Kinniburgh, D. (2002). A review of the source, behaviour and distribution of arsenic in natural waters. *Applied geochemistry, 17*(5), 517-568.

Stark, J. S. (1998). Heavy metal pollution and macrobenthic assemblages in soft sediments in two Sydney estuaries, Australia. *Marine and Freshwater Research, 49*(6), 533-540.

Stokes, E. (1980). *A History of Tauranga County*. Palmerston North, NZ: The Dunmore Press Limited.

Tay, H. W., Bryan, K., Lange, W. d., & Pilditch, C. (2013). The hydrodynamics of the southern basin of Tauranga Harbour. *New Zealand Journal of Marine and Freshwater Research, 47*(2), 249-274.

Tay, H. W., Bryan, K. R., Pilditch, C. A., Park, S., & Hamilton, D. P. (2012). Variations in nutrient concentrations at different time scales in two shallow tidally dominated estuaries. *Marine and Freshwater Research, 63*(2), 95-109.

Teske, P. R., & Wooldridge, T. H. (2003). What limits the distribution of subtidal macrobenthos in permanently open and temporarily open/closed South African estuaries? Salinity vs. sediment particle size. *Estuarine, Coastal and Shelf Science, 57*(1), 225-238.

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

Thrush, S., Townsend, M., Hewitt, J., Davies, K., Lohrer, A., Lundquist, C., Cartner, K., & Dymond, J. (2013). The many uses and values of estuarine ecosystems. 1.16 in: Dymond, J. (ed). *Ecosystem services in New Zealand: conditions and trends*: Manaaki Whenua Press, Lincoln. New Zealand.

Thrush, S. F., Halliday, J., Hewitt, J. E., & Lohrer, A. M. (2008). The effects of habitat loss, fragmentation, and community homogenization on resilience in estuaries. *Ecological Applications, 18*(1), 12-21.

Thrush, S. F., Hewitt, J. E., Gibbs, M., Lundquist, C., & Norkko, A. (2006). Functional role of large organisms in intertidal communities: community effects and ecosystem function. *Ecosystems, 9*(6), 1029-1040.

Troell, M., Pihl, L., Rönnbäck, P., Wennhage, H., Söderqvist, T., & Kautsky, N. (2005). *Regime shifts and ecosystem services in Swedish coastal soft bottom habitats: when resilience is undesirable*: Beijer International Institute of Ecological Economics.

Underwood, A. (1991). Beyond BACI: experimental designs for detecting human environmental impacts on temporal variations in natural populations. *Marine and Freshwater Research, 42*(5), 569-587.

Underwood, A. J. (1992). Beyond BACI: the detection of environmental impacts on populations in the real, but variable, world. *Journal of Experimental Marine Biology and Ecology, 161*(2), 145-178.

Updegraff, D., Stanton, D., & Spencer, M. (1977). Surface waters of Waimea Inlet and Nelson Ha Yen: A preliminary assessment of quality. *New Zealand Journal of Marine and Freshwater Research, 11*(3), 559-575.

Van Damme, S., Struyf, E., Maris, T., Ysebaert, T., Dehairs, F., Tackx, M., Heip, C., & Meire, P. (2005). Spatial and temporal patterns of water quality along the estuarine salinity gradient of the Scheldt estuary (Belgium and The Netherlands): results of an integrated monitoring approach. *Hydrobiologia, 540*(1-3), 29-45.

Weisberg, S. B., Ranasinghe, J. A., Dauer, D. M., Schaffner, L. C., Diaz, R. J., & Frithsen, J. B. (1997). An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. *Estuaries, 20*(1), 149-158.

Word, J. Q. (1978). The infaunal trophic index. *SCCWRP 1978 Annual Report, Southern California Coastal Research Project, El Segundo, CA, 1978*, 13-39.

Appendix 1

Table A1: Tables of Environmental data from sediment samples (n=30), from Te Tāhuna o Rangataua.

Site	Organic Matter (AFDW %)	Arsenic mg/kg dry wt	Cu mg/kg dry wt	Pb mg/kg dry wt	Hg mg/kg dry wt	TP mg/kg dry wt	Zn mg/kg dry wt
MTM A1	4.9	2	4	5.4	0.05	470	24
MTM A2	2.6	0.5	0.5	2.8	0.05	280	19
MTM A3	1.58	0.5	0.5	1.6	0.05	200	7
MTM A4	1.6	0.5	0.5	1.7	0.05	174	7
MTM A5	1.54	0.5	0.5	1.5	0.05	136	6
MTM A6	1.09	0.5	0.5	1.2	0.05	108	5
MTM A6.5	1.29	0.5	0.5	1.2	0.05	98	7
MTM A7	1.56	2	0.5	1.7	0.05	119	10
MTM A7.5	2.1	2	0.5	2.1	0.05	128	15
MTM A8	1.92	0.5	0.5	1.9	0.05	116	13
MTM B1	1.51	0.5	0.5	1.9	0.05	156	10
MTM B2	1.6	0.5	0.5	1.7	0.05	199	10
MTM B3	1.67	0.5	0.5	1.9	0.05	165	8
MTM B4	1.55	0.5	0.5	1.5	0.05	146	10
MTM B5	1.79	0.5	0.5	1.6	0.05	104	10
MTM B6	1.46	0.5	0.5	1.3	0.05	90	10

Site	Organic Matter (AFDW %)	Arsenic mg/kg dry wt	Cu mg/kg dry wt	Pb mg/kg dry wt	Hg mg/kg dry wt	TP mg/kg dry wt	Zn mg/kg dry wt
MTM B6.5	1.43	0.5	0.5	1.2	0.05	93	8
MTM B7	1.82	0.5	0.5	1.5	0.05	125	12
MTM B7.5	1.68	0.5	0.5	1.9	0.05	133	12
MTM B8	2.6	3	2	2.9	0.05	174	22
MTM C1	1.73	0.5	0.5	1.7	0.05	165	8
MTM C2	2.2	2	3	2.1	0.05	186	16
MTM C3	2	3	0.5	2.3	0.05	157	14
MTM C4	2.1	3	0.5	2.6	0.05	145	18
MTM C5	2.1	3	0.5	2.8	0.05	144	16
MTM C6	2.6	3	2	3.4	0.05	188	21
MTM C6.5	2.6	3	0.5	2.9	0.05	180	31
MTM C7	1.85	0.5	0.5	1.4	0.05	110	8
MTM C7.5	1.88	0.5	0.5	1.8	0.05	120	11
MTM C8	2.4	3	0.5	3.1	0.05	180	22

Site	Chl-α ug/kg	Gravel % (>/= 2 mm)	Very Course Sand % (< 2mm,>/= 1mm)	Course Sand % (< 1mm, >/= 500um)	Medium Sand % (<500 um,>/+250 um)	Fine Sand (<250µm, >/= 63 µm)	Very Fine Sand (<125 µm/=63 µm)
MTM A1	29900	0.5	0.9	3.2	20.5	42.8	12
MTM A2	31100	0.1	0.2	2.6	17.3	50.8	15
MTM A3	30400	0.1	0.1	2.3	18.6	56	14.1
MTM A4	36100	0.05	0.1	3.5	22.1	54.6	13.2
MTM A5	27500	0.05	0.3	4.5	23.8	53.3	11.9
MTM A6	18600	0.1	0.2	7.1	38.3	44.7	5.7
MTM A6.5	15500	1.3	0.5	11.4	34.2	37.5	8.4
MTM A7	14000	0.3	0.4	7	28	42.2	13.8
MTM A7.5	20600	0.2	0.3	4.3	19.3	40.6	24
MTM A8	21500	0.2	0.05	2.2	21.9	50.3	18.2
MTM B1	39100	4.6	4.2	6.9	25.7	47.4	6.9
MTM B2	35200	0.5	0.4	5.2	22.5	54	11.4
MTM B3	26300	0.05	0.5	6.7	21.9	50.4	14.1
MTM B4	26800	0.1	0.5	4.8	20.8	51.7	15.7
MTM B5	25000	0.9	0.3	3.9	21.5	53	12.9
MTM B6	15800	0.4	0.1	1.7	16.5	58.8	16.8

Site	Chl-α ug/kg	Gravel % (>/= 2 mm)	Very Course Sand % (< 2mm,>/= 1mm)	Course Sand % (< 1mm, >/= 500um)	Medium Sand % (<500 um,>/+250 um)	Fine Sand (<250µm, >/= 63 µm)	Very Fine Sand (<125 µm/=63 µm)
MTM B6.5	16000	0.7	0.05	3.4	32.1	47.5	9.2
MTM B7	16900	0.05	0.05	4.4	37.6	42.2	8.5
MTM B7.5	20000	0.5	0.2	6.6	28.7	41	14.1
MTM B8	18200	0.6	0.3	5	19.8	30.3	30.6
MTM C1	49100	2.3	3	9.4	35	41.3	4.5
MTM C2	26800	0.1	0.7	6.5	21	46.4	17.9
MTM C3	18400	0.2	1.2	12.6	29.6	34.2	14.3
MTM C4	18900	0.5	1.3	12.2	26.4	32.6	18.3
MTM C5	26700	0.9	3.6	20.9	29	21.6	13
MTM C6	14800	0.5	1.3	10.1	20.8	23	26.5
MTM C6.5	12800	0.1	0.4	3.9	14.5	33.3	25.1
MTM C7	22200	0.2	0.05	4.7	31.3	46.2	10.4
MTM C7.5	21900	0.5	0.01	3.7	28.9	51.3	7.5
MTM C8	21900	1.2	0.3	4.8	19.5	34.1	24.5

Appendix 2

Table A 2: Tables of species data, summed to site level (from 10 replicate core samples) at each site.

Transect	*Distance (m)*	*Amphibola crenata*	**Amphipoda indet_**	*Anthopleura aureoradiata*	*Aonides trifida*	*Armandia maculata*	*Austrovenus stutchburyi*
MTM A1	5	3	24	0	0	0	1
MTM A2	15	0	0	0	0	0	0
MTM A3	30	10	2	0	0	0	0
MTM A4	50	0	283	2	1	0	1
MTM A5	100	4	0	2	0	0	0
MTM A6	200	0	14	1	0	0	0
MTM A6.5	300	0	9	1	1	0	1
MTM A7	400	0	1	3	0	0	0
MTM A7.5	500	0	2	3	0	0	0
MTM A8	600	0	2	1	0	0	3
MTM B1	5	26	21	7	0	0	0
MTM B2	15	0	0	1	0	0	0
MTM B3	30	0	0	1	0	0	0
MTM B4	50	0	0	0	0	0	0
MTM B5	100	3	0	14	0	0	0
MTM B6	200	0	1	2	0	0	1
MTM B6.5	300	0	3	0	1	0	0
MTM B7	400	0	4	3	0	0	1
MTM B7.5	500	0	33	1	5	0	2
MTM B8	600	0	23	9	2	0	0
MTM C1	5	0	2	2	0	0	0
MTM C2	15	0	0	0	0	0	0
MTM C3	30	0	0	0	0	0	0
MTM C4	50	0	1	3	0	0	0
MTM C5	100	0	5	1	0	0	2
MTM C6	200	0	1	3	0	0	0
MTM C6.5	300	0	1	1	0	0	0
MTM C7	400	0	2	3	0	0	0
MTM C7.5	500	0	20	2	1	0	0
MTM C8	600	0	50	20	3	2	1

Transect	**Distance (m)**	***Ceratonereis* sp.**	***Cominella glandiformis***	**Corophiidae**	**Cumacea**	**Diptera**	**(Diptera) Tipulidae**
MTM A1	5	14	3	0	0	2	1
MTM A2	15	16	0	7	0	0	0
MTM A3	30	25	6	142	0	0	0
MTM A4	50	16	16	102	0	0	0
MTM A5	100	20	8	3	0	0	0
MTM A6	200	3	12	20	0	0	3
MTM A6.5	300	35	8	2	29	0	0
MTM A7	400	3	8	0	2	0	0

Transect	Distance (m)	*Ceratonereis* sp.	*Cominella glandiformis*	Corophiidae	Cumacea	Diptera	(Diptera) Tipulidae
MTM A7.5	500	3	17	0	9	0	0
MTM B1	5	10	10	0	0	0	0
MTM B2	15	32	9	129	0	0	0
MTM B3	30	60	18	66	0	0	0
MTM B5	100	13	9	0	0	0	0
MTM B6	200	6	6	0	0	0	0
MTM B6.5	300	14	4	0	7	0	0
MTM B7	400	2	5	0	7	0	0
MTM B7.5	500	5	3	3	0	0	0
MTM B8	600	2	10	0	4	0	0
MTM C1	5	22	12	13	0	0	0
MTM C2	15	35	3	1	2	0	0
MTM C3	30	36	13	0	0	0	0
MTM C4	50	28	10	0	0	0	0
MTM C5	100	27	8	0	0	0	0
MTM C6	200	3	5	0	0	0	0
MTM C6.5	300	1	5	0	0	0	0
MTM C7	400	1	9	0	5	0	0
MTM C7.5	500	0	8	0	8	0	0
MTM C8	600	0	3	1	9	0	0

Transect	Distance (m)	*Diloma subrostrata*	*Edwardsia* sp.	*Eliminus modestus*	*Halicarcinus varius*	*Helice crassa*	*Heteromastus filiformis*
MTM A1	5	2	0	0	3	34	0
MTM A2	15	0	0	3	1	9	0
MTM A3	30	7	0	13	4	13	0
MTM A4	50	1	1	0	0	11	0
MTM A5	100	5	0	0	1	10	0
MTM A6	200	0	0	0	2	8	0
MTM A6.5	300	4	0	0	3	0	5
MTM A7	400	1	0	0	2	0	1
MTM A7.5	500	4	0	0	3	0	4
MTM A8	600	9	0	0	1	0	0
MTM B1	5	2	0	0	7	7	8
MTM B2	15	0	0	0	4	3	0
MTM B3	30	0	0	0	1	3	0
MTM B4	50	1	0	0	3	4	3
MTM B5	100	6	0	0	6	2	1
MTM B6	200	3	0	0	5	4	0
MTM B6.5	300	0	0	0	1	0	0
MTM B7	400	2	0	0	7	3	0
MTM B7.5	500	1	0	0	3	1	16
MTM B8	600	9	1	0	1	0	43
MTM C1	5	1	0	0	4	2	0
MTM C2	15	0	0	0	0	4	0

Transect	Distance (m)	*Diloma subrostrata*	*Edwardsia* sp.	*Eliminus modestus*	*Halicarcinus varius*	*Helice crassa*	*Heteromastus filiformis*
MTM C3	30	0	0	0	1	9	3
MTM C5	100	14	0	0	3	6	5
MTM C6	200	6	0	0	4	4	22
MTM C6.5	300	4	0	0	0	2	22
MTM C7.5	500	0	0	0	5	2	9
MTM C8	600	2	0	0	5	5	81

Transect	Distance (m)	*Macomona liliana*	*Macrophthalmus hirtipes*	Nemertea	Nereididae (juvenile)	*Nicon aestuariensis*
MTM A1	5	0	2	0	10	5
MTM A2	15	0	2	0	2	0
MTM A3	30	0	4	0	16	1
MTM A4	50	0	1	1	30	7
MTM A5	100	0	1	0	20	2
MTM A6	200	0	0	1	27	1
MTM A6.5	300	3	1	1	52	8
MTM A7	400	14	0	0	25	4
MTM A7.5	500	33	2	1	42	0
MTM A8	600	37	0	5	49	0
MTM B1	5	0	0	1	12	1
MTM B2	15	0	1	1	8	3
MTM B3	30	0	2	0	19	3
MTM B4	50	1	0	0	18	0
MTM B5	100	6	0	1	54	14
MTM B6	200	6	1	0	22	4
MTM B6.5	300	9	1	0	52	2
MTM B7	400	8	0	1	29	3
MTM B7.5	500	31	0	0	40	2
MTM B8	600	56	0	4	24	0
MTM C1	5	0	0	1	32	1
MTM C2	15	0	3	0	40	8
MTM C3	30	1	3	0	28	8
MTM C4	50	2	3	1	45	4
MTM C5	100	1	3	1	75	11
MTM C6	200	11	5	0	14	6
MTM C6.5	300	18	9	0	12	6
MTM C7	400	5	0	0	47	2
MTM C7.5	500	20	2	2	89	1
MTM C8	600	44	1	2	83	2

Transect	**Distance (m)**	***Notoacmea helmsi***	***Nucula hartvigiana***	**Oligochaeta**	***Perineries nuntia var. vallata***	**Polydorid**
MTM A1	5	0	0	0	5	0
MTM A2	15	0	0	0	10	0
MTM A3	30	0	0	1	18	2
MTM A4	50	0	0	2	56	1
MTM A5	100	0	1	4	39	3
MTM A6	200	0	0	7	12	0
MTM A6.5	300	0	8	15	15	46
MTM A7	400	0	0	0	16	5
MTM A7.5	500	1	3	2	15	40
MTM A8	600	6	1	2	8	2
MTM B1	5	0	0	0	27	18
MTM B2	15	0	0	0	35	13
MTM B3	30	0	0	2	17	17
MTM B4	50	0	1	6	17	2
MTM B5	100	1	0	6	17	11
MTM B6	200	1	0	0	9	0
MTM B6.5	300	0	0	7	13	26
MTM B7	400	0	0	11	12	10
MTM B7.5	500	0	10	0	15	14
MTM B8	600	2	34	1	11	2
MTM C1	5	0	0	8	35	15
MTM C2	15	0	0	27	22	2
MTM C3	30	0	0	2	20	5
MTM C4	50	0	0	5	24	3
MTM C5	100	0	0	4	13	12
MTM C6	200	0	1	1	16	0
MTM C6.5	300	0	13	9	10	3
MTM C7	400	1	3	8	22	0
MTM C7.5	500	0	46	74	10	28
MTM C8	600	0	191	15	18	60

Transect	Distance (m)	*Prionospio aucklandica*	*Scolecolepides benhami*	*Scolelepis* sp.	*Scoloplos cylindrifer*	*Zeacumantus lutulentus*
MTM A1	5	0	3	1	4	3
MTM A2	15	0	3	0	0	5
MTM A3	30	0	6	4	11	22
MTM A4	50	0	11	30	3	25
MTM A5	100	0	17	9	101	99
MTM A6	200	0	37	8	68	50
MTM A6.5	300	0	51	30	89	148
MTM A7	400	0	17	5	107	123
MTM A7.5	500	2	24	6	92	79
MTM A8	600	2	31	59	75	102

Transect	Distance (m)	*Prionospio aucklandica*	*Scolecolepides benhami*	*Scolelepis* sp.	*Scoloplos cylindrifer*	*Zeacumantus lutulentus*
MTM B2	15	0	19	0	10	108
MTM B3	30	0	9	15	19	51
MTM B4	50	0	26	84	55	89
MTM B5	100	0	33	18	93	135
MTM B6	200	0	20	0	84	74
MTM B6.5	300	1	9	20	93	100
MTM B7	400	0	18	10	53	173
MTM B7.5	500	0	19	10	61	99
MTM B8	600	2	15	9	45	73
MTM C1	5	3	21	15	5	9
MTM C2	15	0	38	18	82	59
MTM C3	30	0	34	1	50	58
MTM C4	50	0	27	5	68	64
MTM C5	100	0	56	10	93	67
MTM C6	200	0	43	1	101	27
MTM C6.5	300	0	38	1	96	16
MTM C7	400	0	13	25	89	142
MTM C7.5	500	6	29	5	87	99
MTM C8	600	2	48	28	39	38

Appendix 3

Table A 3: Table of species diversity indices (S, N and H'), based on marco-faunal data summed to site level (10 replicates per site).

Site	Distance	Species Richness (S)	Total abundance of organisms (N)	Shannon-wiener diversity (H'(loge))
A1	5	18	120	2.300659849
A2	15	10	58	2.022596205
A3	30	19	307	2.081204846
A4	50	21	601	1.838630768
A5	100	19	349	2.073252235
A6	200	17	274	2.268278311
A6.5	300	24	565	2.411516916
A7	400	17	337	1.81570472
A7.5	500	22	387	2.324016138
A8	600	21	423	2.273835567
B1	5	18	296	2.550107014
B2	15	15	376	1.863661905
B3	30	16	303	2.239456709
B4	50	16	365	2.10206961
B5	100	20	443	2.220433813
B6	200	17	249	1.925270668
B6.5	300	18	363	2.09426554
B7	400	19	331	1.836397428
B7.5	500	21	374	2.374113557
B8	600	23	382	2.52314074
C1	5	19	203	2.509555712
C2	15	15	344	2.199047137
C3	30	16	272	2.227102768
C4	50	20	301	2.235465398
C5	100	21	417	2.320900625
C6	200	19	274	2.153960595
C6.5	300	19	267	2.233087926
C7	400	20	392	2.011482986
C7.5	500	22	553	2.415172692
C8	600	26	753	2.512716312

Appendix 4

Table A4: Water quality of surface flows and seepages, between 1987 and 2013), collected for Consent 62881 Annual report of Titiko monitoring and seepages into Rangataua Bay 2013 (Gibbons-Davies, 2013).

Site	Year	Flow (l/s)	Disolved Oxygen (%sat)	Salinity (ppt)	Dissolved Reactive Phosphorus (g/m3)	Nitrate-N (g/m3)	Ammonia-N (g/m3)
W3	1987	-	80	0	-	-	0.17
	1996	>100	80	0.2	-	-	0.46
	2002	100	76	0.4	-	-	0.25
	2006	120	82	4.6	0.012	0.44	0.21
	2008	80	182	15.5	<0.004	0.08	<0.01
W4	1987	<0.01	124	9.5	-	-	1.9
	1996	<0.01	73	17	-	-	0.64
	2002	<0.01	22	16.2	-	-	8.17
	2006	0.002	26	22	0.061	<0.002	2.08
W5	1987	<0.01	53	5.2	-	-	
	1996	<0.01	89	9.8	-	-	11
	2002	<0.01	81	8.8	-	-	8.9
	2006	0.008	95	11.3	4.8	0.004	6.8
W5a	1996	<0.01	83	4.4	-	-	46
	2008	0.006	57	20.8	5.9	<0.01	23
	2009	0.008	16	19.5	5	<0.010	16
W6a	2008	0.003	50	17.1	17	<0.01	71
	2009	0.018	83	7.4	23	0.064	130
	2010	0.01	107	10.5	25	0.069	110
	2011	0.02	150	6.4	33	0.036	133
	2012	0.095	51	6	31	0.119	154
W6	1987	<0.01	211	13.8	-	-	1.7
	1996	<0.01	147	18.2	-	-	13
	2002	<0.01	80	4.8	-	-	64
	2006	0.02	52	4.3	11	<0.002	59
	2008	0.06	205	3.9	26	0.003	94
	2009	0.009	140	8.5	23	0.027	89
	2010	0.04	88	5.6	22	0.021	68
	2011	0.01	160	4.9	23	0.022	71
	2012	0.045	109	5	30	0.015	92
	2013	0.029	86.2	7.3	39	0.005	125
W7	1987	0.02	103	2.8	-	-	14
	1996	0.2	120	4.2	-	-	41
	2002	<0.01	16	11.6	-	-	100
W8	1987	<0.01	132	15	-	-	0.9
	1996	0.1	205	19.6	-	-	0.2
	2002	0.1	164	21	-	-	0.1
	2006	0.004	64	17.9	3.1	0.019	14