The physical and ecological impacts of mangrove expansion and mangrove removal: Tauranga Harbour, New Zealand

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ABSTRACT

The mangrove *Avicennia marina* var. *australasica* is rapidly colonising intertidal sandflats within a number of estuaries of the North Island of New Zealand. Many local residents perceive this change to be detrimental to the ecology and aesthetics of their estuaries, yet little empirical data is available to support these perceptions. Coastal managers are presently developing management strategies associated with either the maintenance or the removal of mangrove habitat with limited information available to predict the impacts of either course of action. This study was developed to investigate the physiognomic characteristics of the mangrove stands, and the physical and ecological impacts of their expansion within three embayments of Tauranga Habour: Welcome Bay, Waikareao Estuary and Waikaraka Estuary. Removal of mangrove vegetation within Waikaraka Estuary provided an ideal site to assess the physical changes that occur in response to this activity.

Detailed field measurements of plant physiognomy of the mangroves within Welcome Bay, Waikareao Estuary and Waikaraka Estuary identified a limited vertical growth of < 10 cm per year, resulting in mean plant heights < 1.5 m. The climatic conditions limiting plant growth appeared to also limit the development of below-ground biomass (root mass). The 2 to 4 kg per m^{-2} of mangrove biomass under mangroves within Waikaraka Estuary is one of the lowest reported to date. Some mangrove sites within Tauranga Harbour produced pneumatophores at densities of ~ 700 m^{-2}. This high density of pneumatophores increases the structural complexity of the substrate which was found to dampen the strength of tidal currents, in turn promoting sedimentation and limiting sediment re-suspension. The morphological reflection of this process was measured using Rod Surface Elevation Tables (RSETs), buried base plates, erosion pins and sediment traps. Typically surface sediments within mangrove colonies were mud-dominated, and sedimentation provided substrate accretion up to 21 mm yr^{-1} in the upper reaches of the study sites. Substrate accretion was also observed on un-vegetated sandflats in some upper-estuary and mid-estuary locations which may promote continued mangrove colonisation by elevating topography relative to the lower elevation limits for seedling survival of between 0.0 and 0.3 m MSL.

A perception that mangrove colonisation has displaced bivalve populations was disproved in this study, at least within the upper estuary environments. A similar suite of benthic macro-invertebrates were encountered within both the mangrove and the un-vegetated tidal flat habitats. These benthic communities were dominated by deposit-feeding organisms such as polychaetes, and an absence of bivalves was common across both habitats.

Approximately one hectare of above-ground mangrove vegetation (10% of the total coverage) was removed from Waikaraka Estuary between 2005 and 2007, which resulted in a lowering of the surface topography at average rates of 15 to 17 mm yr^{-1}. Some textural change of the surface sediments also occurred, with much of the silt fraction being redistributed. It was predicted that a maximum of 9 kg of sediment, including organics, could be released for every square metre of mangroves that is removed. Any coastal management decisions pertaining to mangrove removal must consider the capacity of an estuary's sediment transport system to flush these increased sediment (and organic) loads.

Mangroves are a highly visible indicator of coastal change. It appears that increased sediment loads within the past 100 + years have provided a suitable environment to allow mangroves to flourish. Once established, mangroves further modify the estuary by trapping muddy sediments at sites where their density is high, and tide and wave activity is weak. Benthic ecology in upper-estuaries has, at some point, shifted from a filter-feeding community to a deposit-feeding community more suited to higher sedimentation and increased fine sediment. Altered sediment regimes as a result of land-clearing, agriculture and urbanisation, are likely to drive much of the morphological and ecological change that has been observed in this study.

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Special thanks to the local residents of the Waikareao, Welcome Bay and Waikaraka catchment care groups who supported my research. Dave and Rosemary Lind provided me with not only access to my field site, but also with valuable local knowledge, enormous encouragement and some wonderful shortbread.

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And of course, to my family; I was gone for quite some time. Thank you for your patience and constant support.

In memory of Sue: a great sister, and wonderful teacher.

PREFACE

The body of this thesis comprises five topic chapters (Chapters 3-7). At the time of submission Chapters 3 and 4 had been published in peer-reviewed journals, and Chapter 7 had been published in peer-reviewed conference proceedings.

I was responsible for all aspects of the field work and laboratory work. Two exceptions are noted below. All data analysis and interpretation of results are mine.

Conny Faust, a visiting intern from Germany, contributed to laboratory work associated with determining mangrove below-ground biomass (Chapter 5). Conny undertook lab analysis on 8 sediment cores collected in an area that had been cleared of mangroves which represents < 50% of the lab work associated with the chapter.

Dr Norm Mason, co-author of the conference paper “The benthic ecology of expanding mangrove habitat, Tauranga Harbour, New Zealand“ (Chapter 7), provided statistical advice and created a modified PERMANOVA test to analyse macroinvertebrate data.

A CD can be found attached to the back cover of this thesis which includes the following:

- a copy of this thesis;
- raw data file of macroinvertebrate taxonomic counts;
- field measurements of mangrove plant physiognomy.

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Introduction and Thesis Outline

1.1 INTRODUCTION

There has been a significant increase in the coverage of mangrove vegetation within Tauranga Harbour over the past 40-50 years (Park, 2004), a pattern of coastal change that is also occurring in many other estuaries and embayments within the natural range of mangrove habitat of the North Island of New Zealand.

For many local residents the mangroves and associated muddy substrate are seen as a negative development, for the following reasons:

- they hinder water access for recreation;
- their development can lead to habitat loss for certain intertidal (sandy) benthic organisms and avifauna;
- the muddy substrate associated with mangroves is regarded as aesthetically unpleasant; and
- concerns that mangroves may increase flood hazard in upper estuaries.

1.2 MOTIVATION FOR THE STUDY

Mangroves (*Avicennia marina (*subsp. *Australasica* (Walp.) J. Everett) are currently considered a protected native species under the New Zealand Resource Management Act. Increasing demands for the removal of mangroves is placing pressure on governing bodies to make management decisions with little or no scientific knowledge of the effects of mangrove removal on the estuarine ecosystem. Debate continues among conservation organisations and territorial authorities as to the impact the removal of mangroves will have on coastal erosion, sedimentation and estuarine ecology. In the meantime ad-hoc intervention by local residents continues, sometimes illegally, ranging from simply clearing seedlings to removing sections of established forest. It is essential that decision-makers can accurately assess the local effects of removal activity in order to determine if these effects will be 'less than minor', in accordance with the Resource Management Act.

Figure 1.1 Mangrove seedlings positioned in front of a mangrove stand on the north shores of Welcome Bay, Tauranga Harbour. (Photo: D. Stokes, 2005).

The obvious change occurring in response to mangrove colonisation is the reduction of bare intertidal habitat. To what extent this physical change is modifying the sedimentology, surface topography and benthic ecology is unknown, mostly because mangrove expansion in New Zealand has only recently been recognised as a coastal management issue. To provide effective management frameworks for changing estuaries it is important to enhance our knowledge of the mangrove ecosystem that is central to some of that change.

Figure 1.2 Mangrove vegetation is cut and placed into piles, allowed to dry then incinerated. Waikaraka Estuary, 2005. (Photo: D. Stokes, 2005)

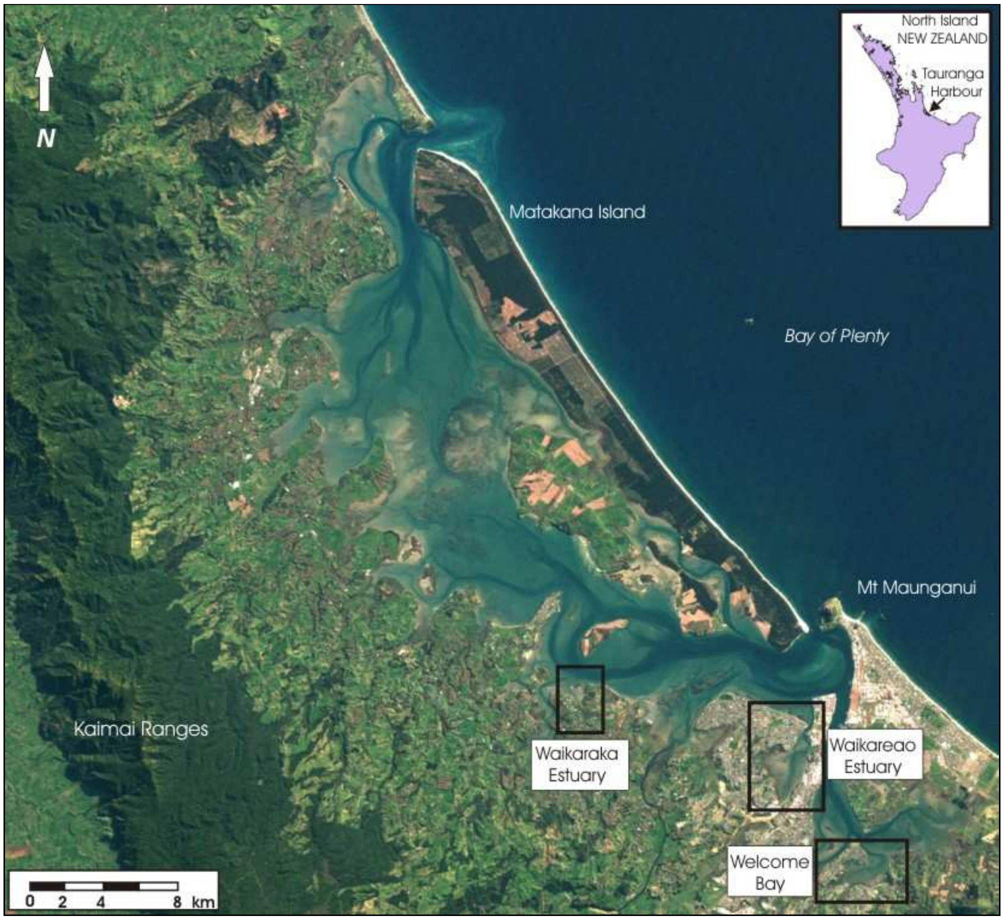

Figure 1.3 **Map showing location of study sites within Tauranga Harbour. Tauranga City is situated along the peninsula between Waikareao Estuary and Welcome Bay. Aerial photograph sourced from Environment Bay of Plenty (2003).**

1.3 AIMS AND OBJECTIVES

Initially, the general aim of this research was to investigate the nature of sedimentation associated with mangroves and determine the physical and ecological response to their removal in Welcome Bay and Waikareao Estuary. Because of the time delays experienced in the resource consent process, the proposed extensive clearing did not take place within the timeframe of this study.

The focus of the study steered more toward an investigation of the mangrove forest characteristics and the influence mangrove vegetation has on the physical environment and benthic community structure. Waikaraka Estuary Manager's Group (composed mostly of residents living close to the estuary margins) however, was permitted to commence clearing in 2005 and so this site was added to the study to provide some insight into the physical changes that occur after mangrove removal.

The research encompasses the following objectives:

- investigate the physical changes that have occurred due to mangrove expansion within sites where mangrove removal was prohibited;
- determine the physical changes that have occurred in response to the removal of mangrove vegetation;
- identify the morphometric characteristics of temperate mangrove forests and determine the contribution of mangrove below-ground biomass to the sedimentary environment;
- examine the influence of the above-ground structures of *Avicennia marina* on tidal flows and sedimentation;
- evaluate past sedimentation rates and explore the effects of on-going mangrove expansion on sedimentation and geomorphology;
- determine benthic community composition of mangrove habitat and draw comparisons with adjacent bare tidal flats.

To meet these objectives a significant field program was initiated to provide data on:

- above-ground and below-ground mangrove plant physiognomy;
- temporal topographical and sedimentological changes;
- hydrodynamic conditions of mangrove habitat; and
- benthic ecology of mangrove and adjacent bare intertidal habitat.

1.4 BENEFITS OF THE RESEARCH

The management of urbanised estuaries is hindered by a lack of information relating to the consequences of both the expansion of mangroves and the impact of their removal. This study endeavours to address the paucity of information relating to the physical changes that can occur due to mangrove expansion. Results of this study contribute to the understanding of mangrove removal on intertidal morphology, should this become an accepted management practice. The field program was also designed to provide baseline data on benthic ecology for future monitoring, and to investigate the influence of mangroves on benthic community structure. Plant morphometrics were measured to contribute to the small botanical knowledge-base of temperate *Avicenna marina*.

1.5 THESIS STRUCTURE

Each chapter of the thesis introduces a stand-alone topic relevant to the understanding of temperate mangrove dynamics and their influence on geomorphology and ecology. Some overlap in field methods occurs in Chapters 3 and 4, although different sites are discussed in each chapter.

Additional data was collected in Waikaraka and Waikareao estuaries to draw comparisons between a large sub-estuary with an associated large catchment, exposure to stronger tidal velocity and wave activity (Waikareao) and a site associated with a smaller catchment and estuarine area, exposed to lower tidal and wave velocities. Waikaraka Estuary provided easier access, so this site was chosen over Welcome Bay to represent the smaller system. Sediment cores were collected to between 2 and 3 m depths at both sites, sediment traps were installed and additional plant characteristics such as canopy size and stem diameter were measured.

Initially it was planned to analyse the results of elevation change and sediment characteristics for Waikareao and Waikaraka and in a subsequent chapter investigate the differences between the small and large system using the additional data. Because mangrove removal was postponed at Waikareao Estuary and Welcome Bay, it seemed more appropriate to present results based on ‘cleared’ (Waikaraka Estuary) and ‘non-cleared’ (Waikareao Estuary and Welcome Bay) systems and as such the additional information was sometimes excluded. Stratigraphic descriptions of the sediment cores collected in Waikaraka Estuary are included in Chapter 4 and additional plant morphometric data is presented in Chapter 5. Core log descriptions of Waikareao Estuary can be found in Appendix I, and additional plant morphometrics measured in Waikareao Estuary are included in the raw data file of all plant physiognomy data that can found on the CD attached to the back sleeve of this thesis.

1.5.1 CHAPTER 2 Mangroves and the research study area

Chapter 2 provides a general background to the topics investigated within this thesis. The current state of knowledge of New Zealand mangrove habitat is addressed. International literature pertaining to mangroves and geomorphology is reviewed to highlight the potential impacts of mangrove expansion on the physical estuarine environment. Site-specific research is summarised.

1.5.2 CHAPTER 3 Physical changes driven by mangrove expansion

This chapter documents the spatial changes of mangrove habitat in Waikareao Estuary and Welcome Bay since the 1940s and discusses surface elevation change in both mangrove habitat and adjacent bare intertidal areas. The mangrove habitat is a key driver to environmental change, and as such knowledge of their stand characteristics is included in this chapter to enhance our understanding of the interaction between mangrove and geomorphology. The key findings of this chapter were published in the *Journal of Coastal Research* in January 2010.

1.5.3 CHAPTER 4 Changing sedimentary environments: influence of mangrove expansion and mangrove removal on estuarine morphology

The impacts of mangrove removal on the physical environment are largely unknown. This chapter focuses on one of the three study sites, Waikaraka Estuary, which was the only site where legal clear-felling of mangrove vegetation was undertaken. Changes to surface elevation and sedimentology after mangrove removal are discussed. Physiognomy of the mangrove stands is also addressed. This chapter was published in the *International Journal of Ecology and Development*, in June 2009.

1.5.4 CHAPTER 5 Characteristics of below-ground structures of temperate mangroves

Review of mangrove literature revealed a knowledge gap pertaining to the development of below-ground biomass of New Zealand mangroves. The mangroves in Tauranga Harbour are growing toward the natural southern limit and it would be expected that the climatic influence on growth (particularly compared to their tropical counterparts) would also influence the below-ground structures. Biomass of mangrove roots will influence the extent of retention or release of sediments and the rate of topographical change associated with mangrove removal. An investigation of the spatial variability in below-ground biomass, its general composition, and temporal changes due to mangrove removal, is outlined in this chapter.

1.5.5 CHAPTER 6 Implications for the future: Waikaraka Estuary

The potential morphological change as a consequence of continued mangrove expansion are explored in this chapter. The influence of mangrove vegetation on tidal currents and suspended sediment was investigated to better understand the site-specific sediment transport processes. Mangrove removal is presently a major influence on intertidal morphology, and estimates of sediment loads associated with this activity were calculated using empirical data of topographical change.

1.5.6 CHAPTER 7 Benthic ecology of temperate mangroves

Intertidal benthic organisms can be sensitive to physical and hydrodynamic changes. As mangroves colonise an intertidal zone they also alter the physical environment via increased deposition of fine sediments and, over time, reduce tidal inundation heights and the duration of inundation. This chapter examines the differences in the community structure of macroinvertebrates of both mangrove habitat and adjacent bare intertidal habitat, and attempts to evaluate the significance of the physical characteristics of each habitat in determining species diversity, abundance and composition. Chapter Seven was peer-reviewed and published in the Proceedings of the Coasts and Ports Conference 2009, Wellington.

1.5.7 CHAPTER 8 Summary and implications

Chapter 8 summarises the findings of the thesis. Drawing on the main conclusions from the previous chapters, implications of both mangrove expansion and mangrove removal are discussed, along with key considerations for the provision of effective management decisions pertaining to mangrove habitat. Aspects of this study that would benefit from further research are addressed and knowledge gaps are highlighted.

1.6 LITERATURE CITED

Park, S., 2004. Aspects of Mangrove Distribution and Abundance in Tauranga Harbour. Environment BOP Environmental Publication 2004/16. Whakatane, New Zealand, 49 p.

Chapter 2
Mangroves and the Research Study Area

2.1 WHAT IS A MANGROVE?

Mangroves are woody plants that grow at the interface between land and sea (Kathiresan and Bingham, 2000). The term 'mangrove' encompasses a number of plant species that have adapted to exist under conditions of high salinity, tidal inundation and anaerobic soils (see Kathiresan and Bingham, 2000 for details on mangrove taxonony). Evolutionary convergence has resulted in a considerable number of plant species that exhibit a variety of adaptations to cope with similar environmental stressors experienced within the mangrove habitat (mangal). Mangrove growth form reflects climatic and edaphic conditions and range from stunted shrubs of less than 1 m in height to majestic trees extending to 40 m (Ellison and Farnsworth, 2000).

Ninety percent of the world's mangroves are found in warm humid areas such as South Mexico to Colombia, the Caribbean, North Brazil and SE Asia to North Queensland. In these environments mangroves are generally tall, dense and comprised of a diversity of plant species. There is a decrease in the number of species found toward the southern and northern limits of mangal geographical distribution, a response to temperature sensitivity (Kathiresan and Bingham, 2000).

Australia is a good example of the species gradients. Along the wet north east coast of Queensland 20 species can be found, while only four grow across on the dry west coast (Duke, 1990). Species diversity also declines with increasing latitude south along both the east and west coasts of Australia (Duke et al., 1998). In southern Victoria, single-species stands of *Avicenna marina* var. *australasica,* the same species found in New Zealand, thrive in a few pockets of the coast. Interestingly, trees growing at similar latitude to those southern stands here in New Zealand are generally taller. For example, Western Port Bay stands (lat 38° 20'S) were measured at around 4 m (Van der Valk and Attiwill, 1984) and further west at Barwon Heads (lat 38° 28'), trees stand at 2-3 m (Stokes, 2002). The southernmost mangroves in the world grow in Corner Inlet, Wilsons Promontory

(Victoria, Australia) at a latitude of 38°54 South (Crisp et al., 1990), and generally have a mean stand height of 1-2 m (Hindell and Jenkins, 2004).

All mangroves face a number of challenges, one being respiration in anaerobic muds. Mangroves display a range of root structures that protrude above the sediment surface (see Fig 2.1). *Avicennia* plants grow vertical breathing roots called pneumatophores which generally extend up to 30 cm in height. The density of pneumatophores tends to be greater where sediments are anaerobic or polluted (Kathiresan and Bingham, 2000), where a single tree can have more than 10,000 pneumatophores (Hogarth, 1999).

The primary role of pneumatophores is one of gas exchange, channelling oxygen into the below-ground root system surrounded by anoxic sediments. The network of below-ground fine rootlets (less than 1 mm diameter) and larger lateral roots enables transfer of oxygen in water-logged conditions, but also assist in stabilising the plant in unconsolidated sediments and against the force of tidal and wave action (Kathiresan and Bingham, 2000). Mangroves typically produce very dense root networks, and studies have reported higher relative root mass for mangrove plants than for upland forest types (for example Saintilan, 1997; review by Komiyama et al., 2008).

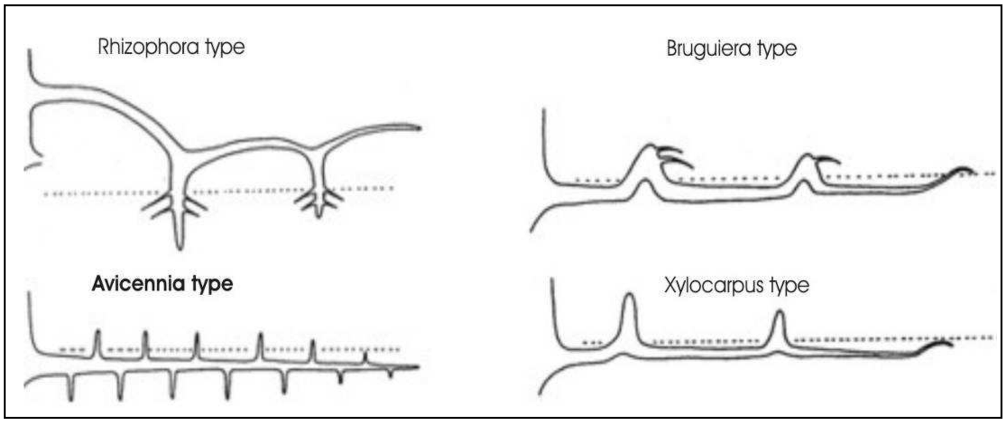

Figure 2.1 Examples of mangrove root structures (adapted from Hogarth, 1999*).*

Vivipary is an efficient method of colonisation. *Avicennia*, along with many other mangrove genera, produce seeds which germinate while still attached to the parent tree. These ‘live young‘, called propagules, drop from the parent tree generally over the summer months and their buoyancy allows distribution by tidal movements. The length of time they can remain buoyant varies with species and with local conditions (Hutchings and Saenger, 1987) and the numbers released can vary from year to year (Clarke and Myerscough, 1993). Propagules tend to establish close to the parent tree, and many others will be found toward high tide level where they settle after tidal-driven dispersal movements. Seedling survivorship generally seems highest on the seaward edge of mangrove stands where there is less competition for light. A combination of sediment type (e.g. mud content) and tidal regime (i.e. flow velocity, inundation period) will influence initial establishment (Clarke and Myerscough, 1993).

Figure 2.2 Seedlings from propagules dropped in summer 2005 under an established mangrove shrub in Waikareao Estuary (left), and seaward of the mangrove fringe at Welcome Bay (right). Photos: D Stokes, August 2005.

2.2 THE ECOLOGICAL SERVICES OF MANGROVES

Mangroves provide a number of ecological services. In many coastal systems a host of fauna have evolved to thrive within the mangal environment. Numerous studies of tropical mangrove habitats have established the abundant biodiversity and rich productivity of these ecosystems (Alfaro, 2005). The mangrove system is still poorly understood, however, due to the complexity and variety of the systems classified under 'mangrove habitat' (Ellison and Farnsworth, 2000). The following general concepts tend to hold true for most systems, however:

1. The nature of the dense and extensive root networks (aerial and subaerial) of mangroves can accelerate sedimentation and thus plays a

significant role in landscape evolution (Woodroffe, 1992; Cahoon et al., 2003);

2. Mangrove forests often act as sediment traps, providing a secondary service of trapping material that could otherwise pollute or smother benthic organisms. In trapping the sediment, mangrove forests also provide shelter and food sources (bacteria, fungi and macroalgae) for grazing animals (Kathiresan and Bingham, 2000);
3. Fish and invertebrate species may use mangrove habitat as a nursery ground (Robertson and Duke, 1987; Laegdsgaard and Johnson, 1995; Mumby et al., 2004; review by Manson et al., 2005);
4. Mangrove systems host a variety of organisms that utilise the tree canopy, including insects and birds (Dingwall, 1984; review by Morrisey et al., 2010); and
5. Mangrove forests store large quantities of organic carbon in their living and dead biomass (Lovelock, 2008) and the decomposition of plant material adds significant organic matter to the detrital food web (Woodroffe, 1985b; Hogarth, 1999).

Humans have utilised mangrove ecosystems for a range of purposes, such as medicine, animal fodder, and their wood has been exploited for fuel and construction. The mangrove environment itself provides home and harbour to a variety of pelagic and benthic organisms collected for food.

Recent studies have demonstrated the potential buffering capability of mangrove forests against storm and tsunami (Dahdouh-Guebas et al., 2005; Kathiresan and Rajendram, 2005), however urban development, mining and over-exploitation of mangrove resources have contributed to dramatic rates of mangrove destruction world-wide. Presently, it is the rapid rise in aquaculture that poses the largest threat to mangrove forests (Alongi, 2002).

2.3 THE GLOBAL STATE OF MANGROVES

Globally, some 35% - 86% of the world's mangrove forests have been lost during the last quarter century (Duke et al., 2007), causing a decline in functional diversity, reducing their carbon contribution to coastal ecosystems, and limiting their capacity to act as an atosmpheric carbon sink (Duke et al., 2007). In tropical areas destruction of mangrove habitat often occurs during construction of large-scale shrimp farming, a practice which has been linked to accelerated coastal erosion, increased pollutants, and also to the collapse of natural shrimp larvae stocks (Hogarth, 1999).

Destruction of mangrove habitat will alter the remaining physical and ecological systems, and while mangrove forests are known to enhance marine biodiversity (e.g. Mumby et al., 2004), few studies have closely examined the changes that occur if they are removed (Granek and Ruttenberg, 2008). The few studies that examine the effects of mangrove removal suggest that the clearing of mangrove vegetation will drive changes in abiotic factors such as temperature, light and nutrient supply which can in turn lead to increases in algal biomass (Granek and Ruttenberg, 2008), altered zooplankton community composition (Granek and Frasier, 2007) and a decline in macroinvertebrate diversity and abundance (Fondo and Martens, 1998).

2.4 MANGROVE EXPANSION

Increases in spatial distribution of mangroves have been observed in southern (temperate) Australia (Coleman, 1998; Stokes, 2002; Rogers et al., 2005), as well as some sub-tropical and tropical sites that are not experiencing net losses through natural disturbance or deliberate removal, for example, some areas of Moreton Bay in southern Queensland (Manson et al., 2003), some protected embayments in Brazil (Souza Filho et al., 2006), river mouths in some locations in Thailand (Thampanya et al., 2006) and wetlands in Taiwan (Lee and Shih, 2004). In some areas where mangroves have been removed for coastal development, rapid recolonisation has been reported (Lee and Shih, 2004; Benfield et al., 2005).

In temperate regions of Australia, New Zealand and southern United States, mangrove and salt marsh communities extensively co-exist (Saintilan et al., 2009). In temperate South-east Australia mangrove and saltmarsh communities are located predominantly within drowned river valleys and barrier estuaries (Roy et al., 2001), similar to New Zealand (Healy et al., 1996). Historical changes to mangrove and salt marsh community distribution has been governed by geomorphic processes, particularly sedimentation (Saintilan et al., 2009). Saintilan (1997) suggests that as infilling progresses, mangroves are replaced with salt marsh as the river progrades seaward. More recently, surveys have demonstrated a loss of salt marsh to landward encroachment of mangrove within numerous South-east Australian estuaries (summarised by Saintilan and Williams, 2000). Increased sediment delivery into the system which promotes vertical accretion and soft substrate for propagule establishment within the salt marsh was a suggested cause. The additional effect of increased nutrients may also lead to increased fecundity of the mangroves (Saintilan et al., 2009). The relationship between the subsidence or autocompaction of marsh surfaces and increased relative sea-level appears to be initiating the landward encroachment of mangroves (Rogers et al., 2005; Rogers et al., 2006). These studies suggest that groundwater recharge is a significant driver of variability in surface elevation after autocompaction was found to strongly correlate with El Nino drought conditions.

In the abovementioned cases mangroves are native species. On the Hawaiian islands, however, mangroves were introduced toward the beginning of the 1900s (Allen, 1998) to reduce soil erosion (Enoki, 2004). Their on-going colonisation along the banks of canals and in harbours has some negative effects such as impeded drainage, offensive odours and increased mosquito populations. Some positive effects have been acknowledged however, due to their capacity to improve water quality via sediment retention (Allen, 1998). Manual clearing of above-ground mangrove structures has been a form of mangrove control in some parts of Hawaii for over 10 years (Allen, 1998), though no studies have been published on the effects of these activities. It has been noted, however, that the control effort has been hampered by rapid re-colonisation of seedlings (Cox and Allen, 1999).

In the absence of mangrove clearance by human intervention, what would we expect to see? During periods of more rapid sea level rise (such as between 4100 and 3700 years BP) mangroves were found to retreat landward, whereas during times where sea-level rise has been more gradual (such as the last 2000 years) mangrove habitat has tended to keep pace (Ellison, 2008). Because mangroves are sensitive to relative sea-level, sediment supply as well as changes in local sea-level will influence the persistence and spatial evolution of a mangrove forest (Krauss et al., 2008). Over time estuaries continue to infill, and in the absence of any geological subsidence, and with sufficient sediment supply, intertidal sites may slowly rise out of the tidal frame. As mangroves generally grow between mean sea level and mean high water (Ellison, 2008), high sediment loads into an estuary can shift and/or extend the position of suitable substrate for mangrove growth (Swales et al., 2007).

Once established, mangrove aerial and subaerial root networks can accelerate sedimentation (Furukawa et al., 1997; Quartel et al., 2007) and impede sediment re-suspension (Phuoc and Massel, 2006; Van Santen et al., 2006), thus speeding up the rate of landscape evolution (Woodroffe, 1992). The structural interference from mangrove roots and trunks creates a significant drag against water movement, increasing the friction force by a factor of at least 25 from that of non-vegetated surfaces (Furukawa and Wolanski, 1996). The result is preferential deposition of fine sediment among mangrove root structures (Furukawa and Wolanski, 1996; Massel et al., 1999).

Within the diversity of mangrove taxa, *Avicennia marina* has been identified as a colonising species (Saenger, 1982). They are prolific producers of seedlings, and in order to flourish they require only some time exposed on each tidal cycle (e.g. the intertidal zone), a suitable substrate to anchor, and a sufficiently quiet hydrodynamic environment to prevent breaking or uprooting (Hogarth, 1999). The preferred substrate for *Avicennia* is a partially muddy one (de Lange and de Lange, 1994), and so any increase in (silty) terrestrial sediments assists colonisation.

The colonisation of mangroves typically does not continue unchecked. Seedlings establish on intertidal areas or creek banks as they require some time exposed, but they can easily be broken in the event of strong tidal currents or wind wave action (Taylor, 1983). Occasional catastrophic events have caused dieback of both seedlings and mature trees. In New Zealand, severe frosts can cause widespread damage (Beard, 2006), while at tropical locations the fungus *phytopthora* has been responsible for the large-scale loss of mangroves. *Phytophora* was also identified at a smaller scale along the Piako River, New Zealand (Maxwell, 1993 in Morrisey et al., 2007).

2.5 STATE OF KNOWLEDGE: MANGROVES IN NEW ZEALAND

2.5.1 Mangrove biogeography and physiognomy

Avicennia marina subsp. *australasica* (called 'Manawa' by Maori) is an indigenous plant that has grown along New Zealand coastlines for 19 million years (Sutherland, 2003). Historically the range of mangroves was greater, with radiocarbon dating and pollen dating placing mangroves around Hawkes Bay (Mildenhall, 2001) and Poverty Bay (Mildenhall, 1994) some 6000-7000 years ago, 140 km further south than the present natural limit which is latitude 38° on both the west coast (Kawhia Harbour) and east coast (Kutarere, Ohiwa Harbour) (Beard, 2006). Manawa are found in sheltered harbours, estuaries, embayments and lagoons (Figure 2.3).

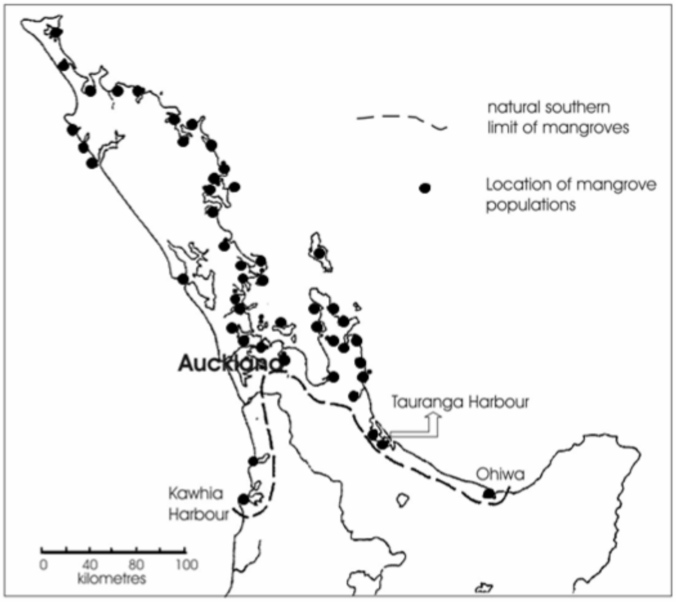

Figure 2.3 Locations of mangrove populations in New Zealand (adapted from Osunkaya and Creese, 1997).

A review of research findings related to New Zealand mangroves (Morrisey et al., 2007) outlined the main theories for the latitudinal boundaries of mangroves in the North Island. Firstly, it had been considered that distribution was constrained by cold temperatures (frosts) (Sakai et al., 1981), or a combination of unsuitable ocean currents and coastal geomorphology acting to limit colonisation (de Lange and de Lange, 1994). More recently, it has been proposed that distribution may be constrained by the physiological limits of mangroves under chilling temperatures which are well above freezing (Beard, 2006).

Structural complexity exists even where mono-species mangrove populations are found, as a result of spatially variable edaphic and hydrodynamic conditions. Plant height and morphology will differ considerably both within sites and among sites (Kuchler, 1972; Taylor, 1983; Woodroffe, 1985a; Crisp et al., 1990). Variation in tree size has been documented in most New Zealand studies (Kuchler, 1972; Taylor, 1983; Burns and Ogden, 1985; Crisp et al., 1990; Osunkaya and Creese 1997; May, 1999; Morrisey et al., 2003; Ellis et al., 2004). Tall (> 6 m) trees can be found in the far north of the North Island (Figure 2.4), and mature trees can stand as low as 1 m.

Table 2.1 Plant morphology characteristics of New Zealand mangroves.

Author	Site	Avg Tree Height	Avg Tree Density	Avg Pneumatophore Density
May, 1999	Rangaunu Harbour, Northland	Site 1. 6.23 m Site 4. 1.68 m	Site 1. 0.125 m^{-2} Site 4. 0.3667 m^{-2}	Not given
Young and Harvey, 1996	Piako River mouth, Firth of Thames	Not given	Not given	< 250 m^{-2}
Ellis et al., 2004	Whitford Embayment, Auckland	Site 2. 0.9 m Site 5. 2.32 m	Not given	Site 2. 109 m^{-2} Site 5. 192 m^{-2}
Alfaro, 2005	Matapouri Estuary Northland	Site 7. 3.6 m Site 1. 4.7 m	Site 7. 2.54 m^{-2} Site 1. 1.98 m^{-2}	Site 7. 88 m^{-2} Site 1. 181 m^{-2}
Morrisey et al., 2003	Manukau Harbour Auckland	Plants established 1987: Site 1/1 1.6m Site 3/2 0.6m	Site 1/1 12.8 m^{-2} Site 3/2 48.8 m^{-2}	Site 1/1 4 m^{-2} Site 3/2 29.2 m^{-2}
Morrisey et al., 2003	Manukau Harbour Auckland	Plants pre 1939: Site A/1 4.6 m Site C/3 3.2 m	Site A/1 0.5 m^{-2} Site C/3 0.45 m^{-2}	Site A/1 414 m^{-2} Site C/3 411 m^{-2}

Often the smaller, stunted tree form is found toward the southern limit of *Avicennia's* natural range, however this is not always the case. Earlier studies suggested a relationship between latitude and mean tree size (Kuchler, 1972; Crisp et al., 1990), however no systematic relationship between latitude and growth form (e.g. stunted 'shrub' or taller 'tree') was found by de Lange and de Lange (1994), suggesting other factors contribute to growth form and growth rate. Plant morphology characteristics of some New Zealand locations are included in Table 2.1.

Figure 2.4 An *Avicennia* tree at Mangawhai, exceeding 5 m in height. (Photo: C. Beard, 2003).

Figure 2.5 Dense canopy cover of mangroves shrubs (< 1.5 m tall) in Waikaraka Estuary.

Mangroves are known to grow in many substrates, however the preferred substrate appears to be a muddy one. Taller trees and faster growing saplings have been found in muddy parts of an upper estuary (Ellis et al., 2004) and a stunted growth form was observed on substrate with <50 % mud (de Lange and de Lange, 1994). Other possible restrictions to plant growth are poor drainage (Crisp et al., 1990), high salinity (Crisp et al., 1990) and nutrient availability (Schwarz, 2002). *Avicennia* successfully colonise sediments in shallow, low energy intertidal environments where seedlings are exposed for part of the tidal cycle (Clarke and Myerscough, 1993). A combination of a rising topography from increased sediment loads, and increasing mud content, provides suitable conditions for seedlings to anchor in many New Zealand estuaries (Swales et al., 2007).

The productivity of mangroves in New Zealand has been demonstrated by measuring litter production (Woodroffe, 1985a; May, 1999). Litter volumes of up to 8 t ha^{-1} yr^{-1} were reported for 3 to 4 m trees, with expected lower volumes for stunted plants (Woodroffe, 1985a). Rates of litterfall documented to date are comparable to values reported for sub-tropical and temperate Australia (Morrisey et al., 2007). Decomposition of mangrove detritus (litterfall, below-ground roots) provides organic matter and nutrients to the estuarine system. Of the few New Zealand studies undertaken, some investigate the breakdown of twigs and leaves (e.g. Woodroffe, 1985b), while others have included below-ground root material (Albright, 1976). There is still, however, little information on both the amounts of below-ground biomass created by New Zealand mangroves (and any spatial or temporal variation thereof), and rates of detrital decomposition of the structural and feeding roots of *Avicennia*.

2.5.2 Ecology of New Zealand mangroves

Presently there is only a small body of work quantifying benthic, terrestrial or pelagic ecology associated with New Zealand mangrove systems. These studies have addressed key questions such as the impact of high rates of mud accumulation on benthic community composition (Ellis et al., 2004), or compared benthic diversity between young and mature mangrove zones (Morrisey et al., 2003), or across different estuarine habitats such as seagrass, sandflats, mudflats and mangroves (Alfaro, 2005). Lower than expected abundance and diversity of benthic invertebrates was highlighted in these studies, with benthic communities dominated by gastropods, polychaetes, amphipods and decapods (Ellis et al., 2004; Alfaro, 2005; Alfaro, 2006).

Spatial variability in benthic community composition also is also evident. For example, Alfaro (2006) recorded populations of grazing snails *Turbo smaragdus* and *Diloma subrostrata*, within a small estuary in the more sub-tropical part of the

North Island, whereas these species were rarely encountered at other locations (Morrisey et al., 2003; Ellis et al., 2004). Grain size could be an influencing factor, as the mangrove substrate reported by Alfaro was sand-dominated, rather than the mud and fine-sand dominated surface sediments sampled by Morrisey et al. (2003) and Ellis et al. (2004).

In addition, benthic community composition differs between young and mature mangrove stands, where relatively higher macroinvertebrate diversity was recorded in stands that were < 12 years old compared to trees of 60 years. This is counter to the popular concept of greater biological diversity where trees have reached a state of maturity. The authors speculated that a greater terrestrial diversity could be present due to the increased structural complexity of the older plants and a less habitable benthic environment brought about by sediment compaction (Morrissey et al., 2003).

The decapod *Helice Crassa* (mud crab) is commonly found in New Zealand mangrove sites (e.g. May, 1999; Morrisey et al., 2003; Ellis et al., 2004; Alfaro, 2005). In tropical systems decapods play a key carbon cycling function by consuming mangrove leaf litter. It is unknown if the smaller New Zealand crabs perform a similar function, with the possibility that amphipods and deposit-feeding snails are a greater influence on the decomposition of organic matter (Morrisey et al., 2007).

Tropical mangrove systems are often cited as important nursery grounds for many fish species (see review by Manson et al., 2005), yet these habitats are often morphologically very different to temperate systems. Tropical mangrove forests are characterised by a diversity of plant species and root morphologies that can provide fish and larvae with a variety of food sources and protective structures (Hogarth, 1999). Fringing forest zones can be permanently inundated with tidal water, allowing a continuous connection to the marine environment. Mangrove populations in New Zealand are generally less diverse in their morphology and typically experience episodic and shallow tidal inundation, which could influence the diversity and abundance of fish that utilise this environment.

Sampling of eight estuaries in northern New Zealand identified 17 fish species that were caught in mangrove habitat, although the assemblages were dominated by common species of yellow-eyed mullet, grey mullet, pilchards and anchovies. A higher abundance of short-finned eel (*Anguilla australis)* was found to correlate with structural complexity of mangrove forests (Morrisey et al., 2007).

In similar latitudes in Australia, fish assemblages were sampled on the edge of mangrove forests and on the adjacent mudflats (Westernport Bay and Corner Inlet, 38 ° latitude). Overall, a greater abundance of juvenile fish was found in the mangroves however there appeared to be no habitat preference for larger subadult and adult fish (Hindell and Jenkins, 2004). A similar study undertaken in the Barwon River, Victoria, concluded that the system was relatively low in species richness and overall abundance. Furthermore, species richness was lower in the mangrove habitat compared to pneumatophore zones and subtidal channels (Smith and Hindell, 2005).

These and other studies of fish diversity identified a similar suite of fish species due to their locational bias towards south and south-eastern Australia and northern New Zealand (Morrisey et al., 2010). Collectively, results suggest that although mangroves provide habitat for fish, many of those species identified are small, have little or no commercial value, and are often also abundant in surrounding habitats (Morrisey et al., 2010).

2.5.3 Mangrove expansion in New Zealand

Captain James Cook documented mangroves in the Firth of Thames, and at Whitianga, in the late 1700s (Crisp et al., 1990) while other reports placed Tauranga Harbour as the southern limit of mangrove biogeography in the 1880s (Burns and Ogden, 1985). The unanswered question is whether mangroves were as prevalent before European settlement as they are presently. Land reclamation for farming, industrial and residential developments intensified in the 1920s and saw the removal of thousands of hectares of mangal (Crisp et al., 1990). A push for mangrove conservation in the 1970s (e.g. Chapman, 1976a, 1976b) contributed to new legislation aimed at the protection of mangroves in New Zealand, with the Harbours Amendment Act 1977 prohibiting infilling for agricultural use (Morrisey et al., 2007). The Resource Management Act (1991) later identified mangroves as a native plant, and therefore a protected species. Today a growing number of coastal residents are calling for legislation change, this time to remove the protection status from mangroves in light of the expansion of mangrove habitat in many harbours and embayments.

An increase in mangrove coverage over the last 50 – 60 years is now well documented (Schwarz, 2002; Brownell, 2004; Park, 2004; Mom, 2005; Swales et al., 2007). Studies have quantified temporal changes to mangrove coverage through the analysis of aerial photos dating back to the 1940s (Park, 2004; Mom, 2005; Swales et al., 2007). Without earlier pictorial evidence, it is not possible to determine longer-term net loss or gain of mangrove habitat (Morrisey et al., 2007). In light of the anectodal evidence of mangrove removal in the 1920s, it is reasonable to caution that the baseline taken from the 1940 aerial photographs is not necessarily representative of an undisturbed site.

Increasingly, studies of New Zealand estuaries are highlighting a link between estuarine sedimentation and mangrove expansion (Young and Harvey, 1996; Ellis et al., 2004; Swales et al., 2007). Infilling of harbours and estuaries is a natural, on-going physical process (Healy et al., 1996), however sedimentation rates (Table 2.2) appear to have increased significantly since European settlement in response to extensive clearing of native vegetation and subsequent soil erosion (Hume and McGlone, 1986; Hume and Dahm, 1992; Sheffield et al., 1995; Swales et al., 2002b; Swales et al., 2007). A correlation between high sedimentation and increased mangrove growth suggests nutrient (nitrogen) enrichment of the coastal environment is likely to play a secondary role, after sedimentation, in facilitating the expansion of mangroves (Lovelock et al., 2007). Climate change is another potential contributor to the recent changes in mangrove habitat (Morrisey et al., 2007).

Regardless of the catalysts of mangrove expansion in New Zealand, the impacts of this environmental change must be investigated fully so that any debate concerning their management can be an informed one.

2.6 THE STUDY AREA: TAURANGA HARBOUR

2.6.1 The physical setting

The study sites (Welcome Bay, Waikaraka and Waikareao estuaries) are small embayments situated within Tauranga Harbour. The harbour is a large (over 200 km^2) barrier enclosed estuarine lagoon (Healy and Kirk, 1992) that extends roughly 40 km along the Bay of Plenty coast (Davies-Colley and Healy, 1978). The lagoon is impounded by Matakana Island, a sandy barrier spit that has developed between two tombolos, Bowentown to the north and Mount Maunganui to the south. Similar to other barrier enclosed estuaries of New Zealand, extensive tidal flats are exposed at high tide (Healy et al., 1996).

The Geology

The Kaimai Ranges separate the Waikato and the Tauranga basins. The ranges are made up of Miocene – Pliocene basalt to rhyolitic rocks that were uplifted during a period of activity along the Hauraki Fault around 1-2 Ma (Briggs et al., 1996). The Tauranga Basin formed over the last 2 to 4 million years through the process of subsidence (Whitbread-Edwards, 1994) associated with activity of the Taupo Volcanic Zone (Davis and Healy, 1993). Tectonic controls of uplift and subsidence vary from site to site in New Zealand (Berryman and Hull, 2003) and to date opposing views have been presented as to whether the Tauranga Basin is still subsiding or currently stable (Shepherd et al., 1997).

Thick ignimbrite deposits are the prominent geological features of the Tauranga Basin. Toward the harbour margins, the ignimbrites are overlain with Holocene and Late Pleistocene alluvium and tephras (Harmsworth, 1983). A number of Miocene rhyolite domes protrude through the plateaus (Briggs et al., 1996), one of which (Minden Peak) creates a watershed for the Waikaraka Estuary catchment. Some ignimbrites, particularly in the central basin area, are non-welded and as such, are prone to erosion. The terraces along the north-west margin of Waikareao Estuary, the largest field area of this study, are made up of Te Ranga Ignimbrites, a non-welded deposit that is structurally weak and prone to gully erosion (Briggs et al., 1996), as shown in Figure 2.6.

The Harbour system

Tauranga Harbour possesses two tidal entrances, Katikati Inlet to the north, and the Tauranga Entrance at the south-eastern end of the harbour, which is the access route to the Port of Tauranga (Davies-Colley and Healy, 1978). A channel has been dredged across the ebb delta to improve ship navigation (Davies-Colley and Healy, 1978). Tidal velocities can peak at 2 m s^{-1} on spring-ebb tides at the Katikati inlet throat (Hume and Herdendorf, 1992), and peak spring-ebb tides of 1.2 -1.3 m s^{-1} have been recorded either side of the Tauranga inlet gorge (Davies-Colley and Healy, 1978).

Seventy percent of Tauranga Harbour is exposed at low tide, therefore a combination of climate (inducing wind-waves) and tidal stage (providing inundation or exposure) will influence the entrainment of sediment over the greater part of the harbour floor (de Lange and Healy, 1990). The dominant orientation of sediment transport in small embayments within Tauranga Harbour have been reported as flood-dominated (White, 1979) and interchangeable depending on the season (Hope, 2002), with reduced current speeds across the tidal flats compared to adjacent tidal channels (Perano, 2000).

Bottom sediments of both harbour entrances and ebb deltas consist mostly of medium and coarse sand with some shelly gravel (Davies-Colley and Healy, 1978; Kruger, 1999). Fine sands and muds accumulate near the head of the many sub-estuaries of the harbour (White, 1979; Hope, 2002; Park, 2003).

2.6.2 The Climatic setting

The mean summer monthly maximums experienced in Tauranga township range from 22 to 24 °C over the months December to March. Mean winter maximums range from 14 to 15 °C and minimums from 5 to 6 °C. Mean rainfall is around 1,200 mm per year [1]. The dominant wind directions measured at Tauranga Aerodrome tend to be north to north-east and west to south-west with the strongest (10.5 – 22.5 m s^{-1}) mostly coming from the west and south-west (Hope, 2002).

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[1] Climate data for the observation period 1969-1998, accessed via MetService.com. Data on MetService.com supplied by National Institutue of Water and Atmospheric Research (NIWA)

The Bay of Plenty region experiences occasional tropical cyclone systems that tend to travel south-east, bringing strong winds and heavy rains (Quayle, 1984), such as the event that caused numerous slope failures along the terraces of Waikareao Estuary after 309 mm of rain fell in a 24 hour period[1] (see Figure 2.6). The La Nina phase of the El Nino Southern Oscillation (ENSO) tends to provide more opportunity for cyclonic weather (de Lange, 2000). An analysis of storm surges and associated wind events suggests a period of higher frequency and greater magnitude of storm surge occurred between 1960 and 1976, a cycle which may be prevailing presently (de Lange, 2000).

Figure 2.6 Slope failures on the margins of Waikareao Estuary after heavy rain in May 2005.

2.6.3 Land Use Changes and Sedimentation

Significant changes in land use have occurred within and around Tauranga Harbour since European settlement (150 – 200 years). Forested areas of the Tauranga basin have been cleared for agricultural and horticultural purposes, and a growing human population is creating growth in the building sector and an increase in the amount of earthworks being undertaken. Construction of causeways, bridges and the port facility has altered the physical and hydrodynamic environment of the harbour. One example is Waikareao Estuary (one of the field sites of this study), where the tidal entrance has been narrowed from 400 m to 200 m as a result of land reclamation and the construction of road and rail causeways (White, 1979).

Sedimentation within Tauranga Harbour has been cited as a leading public concern (Lawrie, 2005), however contemporary and historic rates of infilling have yet to be fully investigated. An attempt was made to utilise radio carbon dating

(^{14}C) and radio isotope techniques (in this instance, ^{137}Cs) to infer sedimentation rates in Waikaraka Estuary (Hope, 2002). Cesium (^{137}Cs) is a by-product of nuclear weapons testing and peaks found in sediment cores correspond to nuclear test dates undertaken in 1953, 1955-1956 and 1963-64 (Swales et al., 2002a). The results of ^{14}C dating suggested sedimentation rates within Waikaraka estuary of 0.05 mm yr^{-1}, a lower result than commonly reported for other New Zealand estuaries (listed in Table 2.2). No peak of ^{137}Cs was detected, however. In the absence of a cesium peak, the sedimentation rate calculated from the single carbon dating sample can provide only an average rate of deposition, and any temporal variation of sedimentation rates which are commonly recorded in the stratigraphy of other New Zealand estuaries (see Table 2.2), are not identified in this instance.

Estuarine sediments are often sourced regionally, at relatively short distances, and so land-use practices in the surrounding catchment will influence the volumes of sediment entering an estuary. Although rates of infilling in Tauranga Harbour are yet to be investigated (beyond this study at least), it has been suggested that fine-grained, catchment-derived sediments are accreting in the upper reaches of many of the quieter embayments of Tauranga Harbour, particularly along the western harbour from Katikati to Te Puna (Hope, 2002; Park, 2003). However, temporal changes in the quantity of terrestrial sediment entering the tidal system are largely unknown. A one-off, extensive study estimated sediment yields entering Tauranga Harbour over the monitoring period of July 1990 to June 1991 (Surman, 1999). The study addressed the erosional state of freshwater streams entering the harbour, and reported monthly measurements of suspended sediment concentration in freshwater inflow from the larger streams and rivers. Interestingly, the highest suspended sediment concentrations and the highest sediment yields were not arriving from the largest inflow at Wairoa River (7 g m^3), but from the Kopurereroa catchment (49 g m^3) which drains into the Waikareao Estuary (Surman, 1999). The lower yields from the Wairoa River may be due to the damming upstream for hydro-electricity (Perano, 2000). Also, the Waikareao catchment is a large one, and one that has experienced considerable earthworks over the last 30 years. Park (2003) suggests that water quality may have improved since Surman's study in 1990-1991, following monitoring of Kopurererua Stream (Waikato estuary) in 2001 that yielded a mean suspended sediment volume of 14 g m^3.

2.6.4 Mangroves in Tauranga Harbour

Mangroves are found in most of the low energy embayments of Tauranga Harbour (Figure 2.7). An almost exponential increase in mangrove coverage was reported for seven sub-estuaries of the harbour between 1943 and 2003 (Park, 2004; Figure 2.8). A small decline in mangrove habitat was recorded after 1999, presumably due to unauthorised vegetation clearance by local residents.

Canopy cover identified on aerial photos of the 1940s and 1950s was usually less than one hectare, increasing to between 5 ha at Waimapu Estuary and 35 ha at Te Puna Estuary by 2000 (Park, 2004).

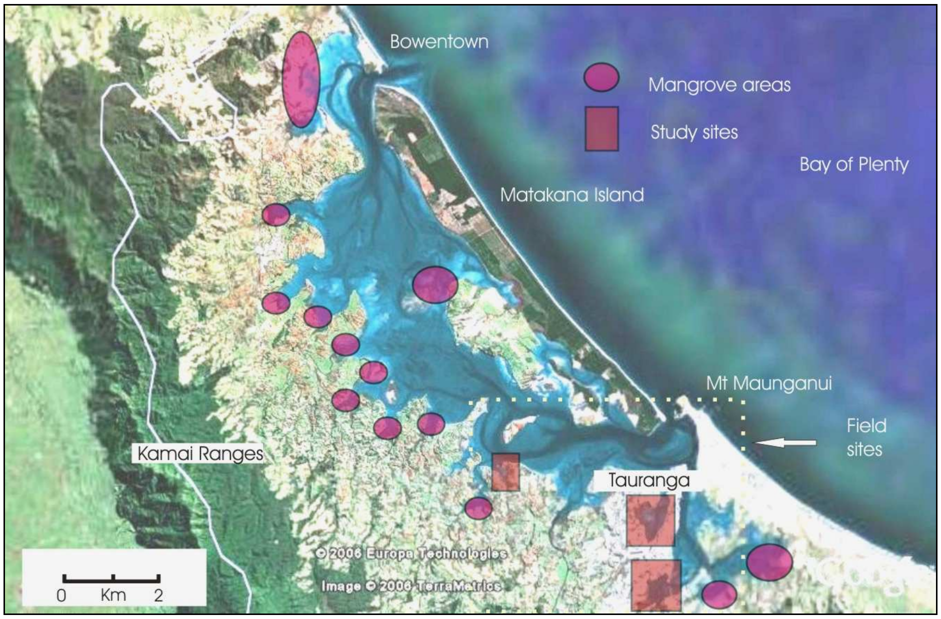


Figure 2.7 Location of mangrove habitat within Tauranga Harbour (Park, 2004). Study sites are outlined. Image from Google Earth, 2006.

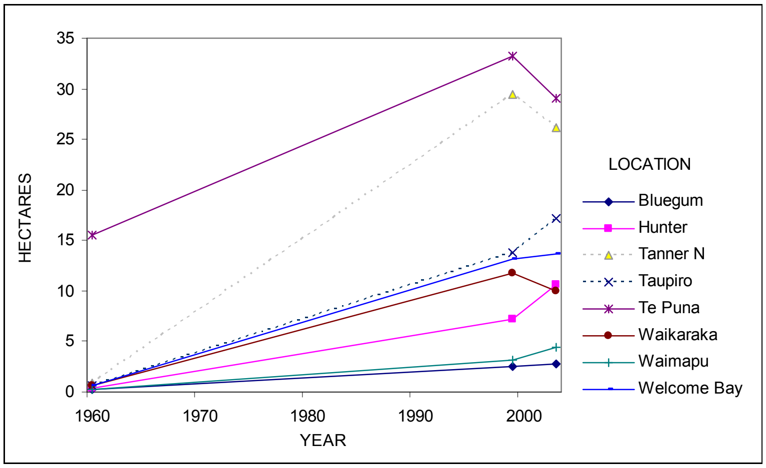


Figure 2.8 Increase in mangrove coverage within sub-estuaries of Tauranga Harbour, between 1960 and 2003. Adapted from Park, 2004.

Table 2.2 Sedimentation rates measured in New Zealand estuaries, including dating techniques used.

AUTHORS	LOCATION	METHOD	SEDIMENTATION RATES (mm yr^{-1})		LAND USE	GRAIN SIZE
Hume & McGlone, 1986	Waitemata Harbour	^{14}C Pollen	2 mm 3 mm <1.5mm	Present day 1840 - 1985 Pre-Polynesian settlement	Rural and urban	Upstream mostly mud; decreasing but always more than 50%
Sheffield, 1995	Whangamata Harbour	^{14}C ^{210}Pb Pollen	20 mm 18 mm < 1 mm	from 1940s; from 1920s-1940s pre European settlement	Mining; Forestry; Steep land	Predominantly fine sand over intertidal flats
Swales et al., 1997	Mahurangi Estuary	Cores Probes modelling	2-21 mm Double the sediment loads of many other catchments in Auckland	since 1850	Pasture and bush steepland	muddy sands and alternative mud/sand beds.
Swales et al., 2002a	Pakuranga Estuary (Auckland)	Pollen ^{137}Cs	3-33 mm 1-1.6 mm 0.2-0.6 mm 3 fold increase in soil erosion over pasture because of urbanization	Urbanisation European settlement Polynesian settlement	Urban development	Intertidal mud and fine sand
Swales et al., 2002b	Auckland estuaries	Pollen ^{210}Pb ^{137}Cs	1.5 - 34.5 mm Mangrove sites to 30 mm	Post 1950	Urban development	mostly muddy fine-sands.
Ellis et al., 2004	Whitford Embayment, Auckland	Traps – mthly for 7 months	0 to 23 mm		Urban development	55 – 99% mud
Swales et al., 2007	Firth of Thames	^{210}Pb	100 mm 20 mm	mangroves since 1950 1850-1920 deforestation	Pasture Forestry, mining	Muds and fine sands

Figure 2.9 **Mangrove seedlings prograding across bare intertidal flats seaward of established dense mangrove habitat (roughly 50 m width) bordered by saltmarsh habitat (in the foreground), Waikareao Estuary.**

2.6.5 Ecology of Tauranga Harbour

The benthic ecology of mangrove habitat in Tauranga Harbour is mostly unidentified. Ecological studies undertaken within the Harbour have to date focused on the population structure and function of mollusc species of subtidal and (unvegetated) intertidal habitat (e.g. Cole et al., 2000; Gouk, 2001). The flood tidal delta of the Tauranga entrance appears to support a wide diversity of bivalves, with 31 taxa identified by Cole et al. (2000). The most common species identified in the area were *Paphies australis, Tawera spissa* and *Ruditapes largillerti* (Hull, 1996). Bivalve diversity and abundance has been found to decline in the upper reaches of other North Island locations, where the substrate contains mud and/or experiences increased turbidity (Thrush et al., 2004; Norkko et al., 2006). *Paphies australis* (pipi) appear to be particularly susceptible to increased turbidity (Teaioro, 1999).

2.6.6 Summary

The mangrove systems in New Zealand may not be as diverse or productive as their tropical counterparts, however ecosystem services are still provided via the contribution of organic matter to the detrital food web and habitat for some benthic, pelagic and terrestrial species. Through the use of aerial photographs, increase in mangrove coverage has been documented at a number of locations within their natural range, and in Tauranga Harbour canopy cover has increased significantly since the 1960s. Increased sediment loads are a suggested leading causal factor in the changes to the vegetation mosaic, however other potential influences such as climate (e.g. less chilling temperatures) and greater nutrient loads have not been widely investigated.

The complex above-ground root structures of mangrove plants generate a substantial influence on the substrate morphology, however the extent to which mangrove expansion in New Zealand has amplified sediment retention is largely unknown. Because of the variability in forest structure and catchment characteristics, continued site-specific studies are required to address this question. Reduced biodiversity and increased coastal erosion are often highlighted as deleterious consequences of the large-scale removal of mangroves in tropical regions, however detailed studies of the physical or ecological changes that occur after mangrove removal are few. Temperate mangrove habitats are vastly different ecosystems which, to date, have received little attention in relation to their evolution, ecology and the impacts of their removal.

2.7 LITERATURE CITED

Albright, L.J., 1976. In situ degradation of mangrove tissues (note). *New Zealand Journal of Marine and Freshwater Research* 10, 385-389.

Alfaro, A.C., 2005. Benthic macro-invertebrate community composition within a mangrove/seagrass estuary in northern New Zealand. *Estuarine, Coastal and Shelf Science* 66, 97-110.

Alfaro, A.C., 2006. Tidal migration influences the zonation of grazing snails (*Turbo smaragdus*) in a mangrove-seagrass estuary, northen New Zealand. *Estuaries and Coasts* 29(5), 731-736.

Allen, J.A., 1998. Mangroves as alien species: the case of Hawaii. *Global Ecology and Biogeography Letters* 7, 61-71.

Alongi, D.M., 2002. Present state and future of the world's mangrove forests. *Environmental Conservation* 29, 331-349.

Beard, C., 2006. Physiological constraints on the latitudinal distribution of the mangrove *Avicennia marina* (Forsk.) Vierh. subsp. *australasica* (Walp.) J. Everett in New Zealand. Unpublished PhD. thesis, University of Waikato, 203 p.

Benfield, S.L., Guzman, H.M., and Mair, J.M., 2005. Temporal mangrove dynamics in relation to coastal development in Pacific Panama. *Journal of Environmental Management* 76, 263-276.

Berryman, K. and Hull, A., 2003. Tectonic controls on Late Quaternary shorelines: a review and prospects for future research. In: Goff, J. R., Nichol, S. L. and Rouse, H. L. (Eds.), *The New Zealand Coast*. Dunmore Press and Whitireia Publishing, Palmerston North, 305 p.

Brownell, B., 2004. *Muddy Feet - Firth of Thames Ramsar Site Update. Report prepared for Environment Waikato, Hauraki District Council and Thames Coromandel District Council*. EcoQuest Education Foundation, Pokeno, New Zealand, 217 p.

Briggs, R. M., Hall, G. J., Harmsworth, G. R., Hollis, A. G., Houghton, B. F., Hughes, G. R., Morgan, M. D. and Whitbread-Edwards, A. R., 1996. *Geology of the Tauranga Area: Occasional Report No. 22*. Department of Earth Sciences, University of Waikato, Hamilton, New Zealand, 56 p.

Burns, B.R., and Ogden, J., 1985. The demography of the temperate mangrove [*Avicennia marina* (Forsk.) Vierh.] at its southern limit in New Zealand. *Australian Journal of Botany* 10, 125-133.

Cahoon, D.R., Hensel, P., Rybczyk, J., McKee, K.L., Proffitt, C.E., and Perez, B.C., 2003. Mass tree mortality leads to mangrove peat collapse at Bay Islands, Honduras after Hurricane Mitch. *Journal of Ecology* 91, 1093-1105.

Chapman, V.J., 1976a. *Mangroves and salt marshes of the Kaipara Harbour: a study with proposals for preservation of areas supporting the harbour ecosystem*. Department of Lands and Survey, Auckland, 28 p.

Chapman, V.J., 1976b. *Mangroves and salt marshes of the Herekino, Whangape and Hokianga Harbours: a study with proposals for preservation of areas supporting the harbour ecosystem.* Department of Lands and Survey, Auckland, 28 p.

Clarke, P.J., and Myerscough, P.J., 1993. The intertidal distribution of the grey mangrove (*Avicennia marina*) in southeastern Australia: the effects of physical conditions, interspecific competition, and predation on propagule establishment and survival. *Australian Journal of Ecology* 18, 307-315.

Cole, R. G., Hull, P. J. and Healy, T. R., 2000. Assemblage structure, spatial patterns, recruitment, and post-settlement mortality of subtidal bivalve molluscs in a large harbour in north-eastern New Zealand. *New Zealand Journal of Marine and Freshwater Research* 34, 317-329.

Coleman, P.S.J., 1998. Changes in a mangrove/samphire community, North Arm Creek, South Australia. *Transactions of the Royal Society of South Australia* 122, 173-178.

Cox, E.F., and Allen, J.A., 1999. Stand structure and productivity of the introduced Rhizophora mangle in Hawaii. *Estuaries and Coasts* 22 (2a), 276-284.

Crisp, P., Daniel, L., and Tortell, P., 1990. *Mangroves in New Zealand: Trees in the Tide*. GP Books, Wellington, New Zealand, 69 p.

Dahdouh-Guebas, F., Jayatissa, L.P., Di Nitto, D., Bosire, J.O., Lo Seen, D., and Koedam, N., 2005. How effective were mangroves as a defence against the recent tsunami? *Current Biology* 15, 1337-1338.

Davies-Colley, R. J. and Healy, T. R., 1978. Sediment and hydrodynamics of the Tauranga Entrance to Tauranga Harbour. *New Zealand Journal of Marine and Freshwater Research* 12(3), 225-236.

Davis, R.A., and Healy, T.R., 1993. Holocene coastal depositional sequences on a tectonically active setting: southeastern Tauranga Harbour, New Zealand. *Sedimentary Geology* 84, 57-69.

de Lange, W. P., 2000. Seasonal, interannual, and decadal variability of storm surges at Tauranga, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 34, 419-434.

de Lange, W., and de Lange, P.J., 1994. An appraisal of factors controlling the latitudinal distribution of Mangrove (*Avicennia marina* var. *resinifera*) in New Zealand. *Journal of Coastal Research* 10, 539-548.

de Lange, W. and Healy, T. R., 1990. Renourishment of a flood-tidal delta adjacent beach, Tauranga Harbour, New Zealand. *Journal of Coastal Research* 6, 627-640.

Dingwall, P.R., 1984. Overcoming problems in the management of New Zealand mangrove forests, in: Teas, H.J., (Ed.) *Physiology and management of mangroves*. The Hague, Dr W Junk Publishers, pp. 97-106.

Duke, N.C., 1990. Phenological trends with latitude in the mangrove tree *Avicennia Marina*. *Journal of Ecology* 78, 113-133.

Duke, N.C., Ball, M.C., and Ellison, J.C., 1998. Factors influencing biodiversity and distributional gradients in mangroves. *Global Ecology and Biogeography Letters* 7, 27-47.

Duke, N.C., Meynecke, J.O., Dittman, S., Ellison, A.M., Anger, K., Berger, U., Cannicci, S., Lee, S.Y., Marchand, C., Nordhaus, I., and Dahdouh-Guebas, F., 2007. A world without mangroves? *Science* 317, 41-42.

Ellis, J., Nicholls, P., Craggs, R., Hofstra, D., and Hewitt, J., 2004. Effects of terrigenous sedimentation on mangrove physiology and associated macrobenthic communities. *Marine Ecology Progress Series 270*, 71-82.

Ellison, A.M., and Farnsworth, E.J., 2000. Mangrove Communities, in: Bertness, M.D., Gaines, S.D., and Hay., M.E., (Eds.), *Marine Community Ecology*. Sinauer Associates Inc, Sunderland, MA, 550 p.

Ellison, J.C., 2008. Long-term retrospection on mangrove development using sediment cores and pollen analysis: a review. *Aquatic Botany* 89, 93-104.

Enoki, T., 2004. Management of forest ecosystems on islands: the mangrove in Hawaii as an alien species. *The Okinawan Journal of American Studies* 1, 62-66.

Fondo, E.N., and Martens, E.E., 1998. Effects of mangrove deforestation on macrofaunal densities, Gazi Bay, Kenya. *Mangroves and Salt Marshes* 2, 75-83.

Furukawa, K., and Wolanski, E., 1996. Sedimentation in mangrove forests. *Mangroves and Salt Marshes* 1, 3-10.

Furukawa, K., Wolanski, E., and Mueller, H., 1997. Currents and sediment transport in mangrove forests. *Estuarine Coastal and Shelf Science* 44, 301-310.

Gouk, S.G., 2001. The Population Dynamics and Production of *Paphies australis* (Pipi) in the Southern Basin, Tauranga Harbour, New Zealand. MSc. thesis, University of Waikato, 133 p.

Granek, E.F., and Frasier, K., 2007. The impacts of red mangrove (*Rhizophora mangle)* deforestation on zooplankton communities in Bocas del Toro, Panama. *Bulletin of Marine Science* 80, 905-914.

Granek, E., and Ruttenberg, B.I., 2008. Changes in biotic and abiotic processes following mangrove clearing. *Estuarine, Coastal and Shelf Science* 80, 555-562.

Harmsworth, G. R. 1983. Quaternary Stratigraphy of the Tauranga Basin. Unpublished MSc. thesis, University of Waikato, 342 p.

Healy, T. R., Cole, R. G. and De Lange, W., 1996. Geomorphology and ecology of New Zealand shallow estuaries and shorelines. In: Nordstrom, K. F. and Roman, C. T. (Eds.), *Estuarine Shores: Evolution, Environments and Human Alteration*. John Wiley & Sons, Chichester, pp. 115-154.

Healy, T. R. and Kirk, R. M., 1992. Coasts. In: Soons, J. M. & Selby, M.J. (Eds.), Landforms of New Zealand. Longman Paul, Auckland, pp. 81-104.

Hindell, J.S., and Jenkins, G.P., 2004. Spatial and temporal variability in the assemblage structure of fishes associated with mangroves (*Avicennia marina*) and intertidal mudflats in temperate Australian embayments. *Marine Biology* 144, 85-395.

Hogarth, P.J., 1999. *The Boilogy of Mangroves*. Oxford, Oxford University Press. 228 p.

Hope, H.M., 2002. Sediment dynamics of Waikaraka Estuary, a small, semi-enclosed estuarine system in the upper reaches of a developed harbour. Unpublished MSc. thesis, University of Waikato. 126 p.

Hull, P. J. 1996. The ecology of bivalves on Centre Bank, Tauranga Harbour. Unpublished MSc. thesis, University of Waikato, 101 p.

Hume, T. and Dahm, J., 1992. *An investigation of the effects of Polynesian and European land use on sedimentation in Coromandel estuaries*. Consultancy Report, 6104, Water Quality Centre, DSIR, Waikato Regional Council, Hamilton, 55 p.

Hume, T. M. and Herdendorf, C. E., 1992. Factors Controlling Tidal Inlet Characteristics on Low Drift Coasts. *Journal of Coastal Research* 8(2), 355-375.

Hume, T. and McGlone, M. S., 1986. Sedimentation patterns and catchment use change recorded in the sediments of a shallow tidal creek, Lucas Creek, Upper Waitemata Harbour, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 20, 677-687.

Hutchings, P., and Saenger, P., 1987. *Ecology of Mangroves*. St Lucia, University of Queensland Press, 369 p.

Kathiresan, K., and Bingham, B., 2000. Biology of Mangroves and Mangrove Ecosystems. *Advances in Marine Biology* 40, 84-259.

Kathiresan, K., and Rajendran, N., 2005. Coastal mangrove forests mitigated tsunami. *Estuarine Coastal and Shelf Science* 65, 601-606.

Komiyama, A., Ong, J. E. and Poungparn, S., 2008. Allometry, biomass, and productivity of mangrove forests: A review. *Aquatic Botany* 89(2), 128.

Krauss, K.W., Lovelock, C.E., McKee, K.L., Lopez-Hoffman, L., Ewe, S.M.L., and Sousa, W.P., 2008. Environmental drivers in mangrove establishment and early development: A review. *Aquatic Botany* 89, 105-127.

Kruger, J. C. 1999. Sedimentation at the entrance channel of Tauranga Harbour, New Zealand. Unpublished MSc. thesis, University of Waikato, 244 p.

Kuchler, A.W., 1972. The Mangrove in New Zealand. *New Zealand Geographer* 28, 113-129

Lawrie, A., 2005. *Tauranga Harbour Integrated Management Review: 2005/22*. Environment Bay of Plenty, Whakatane, New Zealand, 52 p.

Laegdsgaard, P. and Johnson, C. R. 1995. Mangrove habitats as nurseries: unique assemblages of juvenile fish in subtropical mangroves in eastern Australia. *Marine Ecology Progress Series* 126, 67-81.

Lee, H.Y., and Shih, S.S., 2004. Impacts of vegetation changes on the hydraulic and sediment transport characteristics in Guandu mangrove wetland. *Ecological Engineering* 23, 85-94.

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

Lovelock, C., 2008. Soil respiration and belowground carbon allocation in mangrove forests. *Ecosystems* 11, 342-354.

Manson, F.J., Loneragan, N.R., and Phinn, S.R., 2003. Spatial and temporal variation in distribution of mangroves in Moreton Bay, subtropical Australia. A comparison of pattern metrics and change detection analyses based on aerial photographs. *Estuarine, Coastal and Shelf Science* 57, 653-666.

Manson, F. J., Loneragan, N. R., Harch, B. D., Skilleter, G. A. and Williams, L. 2005. A broad-scale analysis of links between coastal fisheries production and mangrove extent: A case-study for northeastern Australia. Fisheries Research 74(1-3), 69.

Massel, S.R., Furukawa, K., and Brinkman, R.M., 1999. Surface wave propagation in mangrove forests. *Fluid Dynamics Research* 24, 219-249.

May, J., 1999. Spatial variation in litter production by the mangrove *Avicennia marina* var. *australasica* in Rangaunu Harbour, Northland, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 33, 163-172.

Mildenhall, 1994. Early to mid Holocene pollen samples containing mangrove pollen from Sponge Bay, east coast, North Island, New Zealand. *Journal of the Royal Society of New Zealand* 24(2), 219-230.

Mildenhall, 2001. Middle Holocene mangroves in Hawkes Bay. *New Zealand Journal of Botany* 39(3), 517-521.

Mom, G.A.T., 2005. Effects of management of the ecology of mangrove communities in New Zealand. Unpublished PhD. thesis, Auckland University, 366 p.

Morrisey, D., Beard, C., Morrison, M., Craggs, R., and Lowe, M., 2007. *The New Zealand Mangrove: review of the current state of knowledge*. Auckland Regional Council Technical Publication Number 325, Auckland, 155 p.

Morrisey, D.S., Skilleter, GA; Ellis, JI; Burns, BR; Kemp, CE; 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, 581-592.

Morrisey, D. J., Swales, A., Dittmann, S., Morrison, M. A., Lovelock, C. E. and Beard, C. M., 2010. The ecology and management of temperate mangroves. Oceanography and Marine Biology: An Annual Review, Vol 48. Boca Raton, Crc Press-Taylor & Francis Group. 48, pp. 43-160.

Mumby, P.J., Edwards, A.J., Arias-Gonzalez, J.E., Lindeman, K.C., Blackwell, P.G., Gall, A., Gorczynska, M.I., Harborne, A.R., Pescod, C.L., Renken, H., Wabnitz, C.C.C., and Llewellyn, G., 2004. Mangroves enhance the biomass of coral reef fish communities in the Caribbean. *Nature* 427, 533-536.

Norkko, J., Hewitt, J. E. and Thrush, S. F., 2006. Effects of increased sedimentation on the physiology of two estuarine soft-sediment bivalves, *Austrovenus stutchburyi* and *Paphies australis*. *Journal of Experimental Marine Biology and Ecology* 333(1), 12-26.

Osunkoya, O.O., and Creese, R.G., 1997. Population structure, spatial pattern and seedling establishment of the grey mangrove, *Avicennia marina* var. *australasica*, in New Zealand. *Australian Journal of Botany* 45, 707-725.

Park, S., 2003. *Marine Sediment and Contaminants Survey (2001-2003) of Tauranga Harbour*. Environment Bay of Plenty Environmental Publication No. 2004/16, Environment Bay of Plenty, Whakatane, New Zealand, 23 p.

Park, S., 2004. *Aspects of mangrove distribution and abundance in Tauranga Harbour*. Environment Bay of Plenty Environmental Publication 2004/16, Whakatane, New Zealand, 41 p.

Perano, K. M. 2000. Sediment transport on intertidal flats, in a tide-dominated environment, Wairoa Estuary, Tauranga, New Zealand. Unpublished MSc. thesis, University of Waikato, 158 p.

Phuoc, V.L.H., and Massel, S.R., 2006. Experiments on wave motion and suspended sediment concentration at Nang Hai, Can Gio mangrove forest, Southern Vietnam. *Oceanologia* 48, 23-40.

Quartel, S., Kroon, A., Augustinus, P.G.E.F., Van Santen, P., and Tri, N.H., 2007. Wave attenuation in coastal mangroves in the Red River Delta, Vietnam. *Journal of Asian Earth Sciences* 29, 566-575.

Quayle, A.M. 1984. *The Climate and Weather of the Bay of Plenty Region*. New Zealand Meteorological Service Miscellaneous Publication 115 (1) 2nd Edition, Ministry of Transport, Wellington, 56 p.

Robertson, A. I. and Duke, N. C., 1987. Mangroves as nursery sites: comparisons of the abundance and species composition of fish and crustaceans in mangroves and other nearshore habitats in tropical Australia. *Marine Biology* 96(2), 193-205.

Rogers, K., Saintilan, N., and Heijnis, H., 2005. Mangrove encroachment of salt marsh in Western Port Bay, Victoria: the role of sedimentation, subsidence and sea level rise. *Estuaries* 28, 551-559.

Rogers, K., Wilton, K. M. and Saintilan, N. 2006. Vegetation change and surface elevation dynamics in estuarine wetlands of southeast Australia. *Estuarine, Coastal and Shelf Science* 66(3-4), 559-569.

Roy, P. S., Williams, R. J., Jones, A. R., Yassini, I., Gibbs, P. J., Coates, B., West, R. J., Scanes, P. R., Hudson, J. P. and Nichol, S. 2001. Structure and function of south-east Australian estuaries. *Estuarine Coastal and Shelf Science* 53, 351-384.

Saenger, P., 1982. Morphological, anatomical and reproductive adaptations of Australian mangroves. In: Clough, B.F. (Ed.), *Mangrove Ecosystems in Australia: Structure, Function and Management*. Australian Institute of Marine Science, Townsville, in association with Australian National University Press, Canberra, pp. 153–191.

Saintilan, N., 1997. Above- and below-ground biomass of mangroves in a sub-tropical estuary. *Marine and Freshwater Research* 48, 601-604.

Saintilan, N., Rogers, K. and McKee, K., 2009. Salt marsh-mangrove interactions in Australia and the Americas. In: Perillo and E. Wolanski, (Eds). Coastal Wetlands: an integrated ecosystem approach. Elsevier B.V, pp. 855-883.

Saintilan, N. and Williams, R. 2000. Short Note: the decline of saltmarsh in southeast Australia: results of recent surveys. *Wetlands (Australia)* 18(2), 49-54.

Sakai, A., Paton, D.M., and Wardle, P., 1981. Freezing resistance of trees of the south temperate zone, especially subalpine species of Australasia. *Ecology* 62, 563-570.

Schwarz, A., 2002. *The role of nutrients in contributing to mangrove expansion*. National Institute of Water & Atmospheric Research Ltd, Hamilton, New Zealand 24 p.

Sheffield, A.T., Healy, T.R., and McGlone, M.S., 1995. Infilling rates of a steepland catchment estuary, Whangamata, New Zealand. *Journal of Coastal Research* 11, 1294-1308.

Shepherd, M. J., McFadgen, B. G., Betts, H. D. and Sutton, D. G., 1997. Formation, landforms and palaeoenvironment of Matakana Island and implications for archaeology. Dept. of Conservation, Wellington, New Zealand, 99 p.

Smith, T. M. and Hindell, J. S. 2005. Assessing effects of diel period, gear selectivity and predation on patterns of microhabitat use by fish in a mangrove dominated system in S.E. Australia. Marine Ecology Progress Series 294, 257-270.

Souza Filho, P.W.M., Farias Martins, E.d.S., and da Costa, F.R., 2006. Using mangroves as a geological indicator of coastal changes in the Braganca macrotidal flat, Brazilian Amazon: A remote sensing data approach. *Ocean & Coastal Management* 49, 462-475.

Stokes, D., 2002. Tidal dynamics and geomorphology of the lower Barwon tidal inlet. Unpublished Honours research thesis, La Trobe University, 63 p.

Surman, M., 1999. *Tauranga Harbour Sediment Source Survey*, Environmental Publication No. 98/13. Environment Bay of Plenty, Whakatane, New Zealand, 141 p.

Sutherland, J.I., 2003. Miocene petrified wood and associated borings and termite faecal pellets from Hukatere Peninsula, Kaipara Harbour, North Auckland, New Zealand. *Journal of the Royal Society of New Zealand* 33, 395-414.

Swales, A., Bentley, S.J., Lovelock, C.E., and Bell, R., 2007. Sediment processes and mangrove-habitat expansion on a rapidly-prograding muddy coast, New Zealand, in: Kraus, N.C., and Rosati, J.D., (Eds.), *Coastal Sediments '07*, Volume 2. New Orleans, Louisiana, American Society of Civil Engineers, pp. 1445-1454.

Swales, A., Hume, T. and Green, M., 1997. Holocene sedimentation and recent human impacts on infilling in a drowned valley estuary. 13th Australasian Coastal and Ocean Engineering Conference & 6th Australasian Ports and Harbours Conference, Christchurch, pp. 895-900.

Swales, A., Hume, T., McGlone, M., Pilvio, R., Ovenden, R., Zviguina, N., Hatton, S., Nicholls, P., Budd, R., Hewitt, J., Pickmere, S., and Costley, K., 2002b. Evidence for the physical effects of catchment sediment runoff preserved in estuarine sediments: Phase II (field study). Prepared for Auckland Regional Council, NIWA Client Report. HAM 2002-067. Hamilton, National Institute of Water & Atmospheric Research Ltd, 174 p.

Swales, A., Williamson, R.B., Van Dam, L.F., Stroud, M.J., and McGlone, M.S., 2002a. Reconstruction of urban stormwater contamination of an estuary using catchment history and sediment profile dating. *Estuaries* 25, 43-56.

Taylor, F.J., 1983, *The New Zealand Mangrove Association*, Tasks for Vegetation Science, Volume 8. The Hague, Dr W Junk Publishers, pp. 77 - 79.

Teaioro, I. 1999. The effects of turbidity on suspension feeding bivalves. Unpublished MSc. thesis, University of Waikato, 109 p.

Thampanya, U., Vermaat, J.E., Sinsakul, S., and Panapitukkul, N., 2006. Coastal erosion and mangrove progradation of Southern Thailand. *Estuarine, Coastal and Shelf Science* 68, 75-85.

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

Van Der Valk, A.G., and Attiwill, P.M., 1984. Decomposition of leaf and root litter of *Avicennia marina* at Westernport Bay, Victoria, Australia. *Aquatic Botany* 18, 205-221.

Van Santen, P., Augustinus, P.G.E.F., Janssen-Stelder, B.M., Quartel, S., and Tri, N.H., 2006. Sedimentation in an estuarine mangrove system. *Journal of Asian Earth Sciences* 29, 566-575.

Whitbread-Edwards, A.R., 1994. The volcanic geology of the Western Tauranga Basin. Unpublished MSc. thesis, University of Waikato, 107 p.

White, J. 1979. Recent sediments of Waikareao Estuary, Tauranga, North Island, New Zealand. Unpublished MSc. thesis, University of Waikato, 197 p.

Woodroffe, C., 1985a. Litter production and decomposition in the New Zealand mangrove, *Avicennia marina* var. *resinifera*. *New Zealand Journal of Marine and Freshwater Research* 16, 179-188.

Woodroffe, C., 1985b. Studies of a Mangrove Basin, Tuff Crater, New Zealand: I. Mangrove biomass and production of detritus. *Estuarine Coastal and Shelf Science* 20, 265-280.

Woodroffe, C., 1992. Mangrove sediments and geomorphology, in: Robertson, A.I., and Alongi, D.M., (Eds.), *Tropical Mangrove Ecosystems*, Volume 41. Washington, D.C., American Geophysical Union, pp. 7-41.

Young, B.M., and Harvey, E.L., 1996. A spatial analysis of the relationship between mangrove (*Avicennia marina* var. *australasica*) physiognomy and sediment accretion in the Hauraki Plains, New Zealand. *Estuarine, Coastal and Shelf Science* 42, 231-246.

Chapter 3
Physical changes driven by mangrove expansion

3.1 PREFACE

With an increasing public perception that mangrove expansion is negatively impacting on the estuarine environment, there is a clear need to identify and quantify any changes triggered by their establishment. Site-specific studies identifying the geomorphic changes attributable to mangrove expansion are required before suitable management strategies can be developed.

A resource consent was submitted by Tauranga City Council in 2003, requesting approval to clear any mangrove vegetation that had established since 2001 within four sub-estuaries of Tauranga Harbour (Waimapu, Matua, Waikareao and Welcome Bay). A legal condition associated with the submitted consent was the requirement of ecological and sedimentological monitoring at both Welcome Bay and Waikareao Estuary. Tauranga City Council therefore funded this Ph.D. research as a means of meeting the consent conditions. The resource consent application, although approved, was subsequently appealed which prevented any mangrove clearance to take place during the time-frame of this study. Tauranga City Council required the monitoring of the physical and ecological parameters to continue, however. As such, the study continued but with an increased focus on understanding the relationship between mangroves and the rate of expansion across tidal flats, and subsequent physical changes to the harbour floor. The delay in mangrove clear-felling provided an opportunity to study the contemporary estuarine environment before intervention.

The findings presented in this chapter have been published in the Journal of Coastal Research, with the following citation :

> Stokes, D.J., Healy, T.R. and Cooke, P.J. *2010.* Expansion dynamics of mono-specific, temperate mangroves and sedimentation in two embayments of a barrier-enclosed lagoon, Tauranga Harbour, New Zealand. *Journal of Coastal Research*, 26 (1), 113-122.

The journal article reports on the surface sediments of the upper estuaries of Welcome Bay and Waikareao Estuary, Tauranga Harbour. Temporal changes to the lateral coverage of mangrove habitat are assessed, and the influence of mangroves on sedimentology and surface topography are investigated. The plant physiognomy is also reported, which identifies the general characteristics of estuarine *Avicennia* stands growing under the climatic constraints of cooler temperatures. The estuaries have evolved from bare flats with a 1-2 m wide mangrove fringe in 1940, to dense mangrove stands reaching 75 m wide. It is important to broaden our understanding of mangrove dynamics in these temperate settings, as little specific botanical investigation has been undertaken.

3.2 INTRODUCTION

A noticeable environmental change that has occurred in numerous estuaries of the North Island of New Zealand is the rapid aerial expansion of mangroves (*Avicennia marina* subsp. *australasica)* over previously uncolonised tidal flats. Because of a growing public concern about this phenomenon, particularly the mud accumulation, mangrove removal has occurred in some locations, both legally and illegally. The public perception is that mangrove removal will result in flushing of the unwanted mud. Before mangrove removal becomes an accepted norm it is important to study the physical environment in its present state. The dynamics of mangrove colonisation in the temperate and single-species context, and the resultant changes to the physical estuarine environment are rarely documented. In this study we explore the spatial evolution of mangrove forests in two estuaries within a larger harbour, and investigate the relationships between mangrove structures, sedimentology and surface accretion.

Tidal flats evolve over time in response to tidal fluxes which move sediment, nutrients and organic material. The rate of infilling, or evolution, will depend on estuary type, the size of the estuary basin and the sediment supply. Sediment supply and estuarine sedimentation are influenced by catchment topography, land-use and climate (Woodroffe, 1983). Harbours and estuaries situated in climatic regions which experience high and or heavy rainfall may receive huge volumes of sediment (e.g. Saad et al., 1999), particularly if the region is dominated by agricultural soils (Alongi et al., 2005). For example, the Firth of Thames (New Zealand) is located at the terminal end of a steep catchment that has been largely deforested. Rainfall of over 1 m per year delivers up to 185,000 t yr^{-1} of sediment to the Firth each year and this has resulted in sedimentation of up to 20 mm yr^{-1}

on the intertidal mudflats, and 100 mm yr^{-1} within the fringing mangrove forest (Swales et al, 2007).

The translation of sediment supply to surface accretion of tidal flats and wetland systems is a complex process that is primarily influenced by tides and waves. Any prediction of surface accretion of wetlands is further complicated by sediment compaction, root development and groundwater supply (Cahoon and Lynch, 1997; Rogers et al., 2005). Even in environments that receive limited supplies of allochtonous sediment, surface accretion is still possible due to the development of peat as belowground mangrove root systems decompose (Cahoon et al., 2003; McKee et al., 2007). Altered groundwater supplies may also influence surface elevation (Cahoon and Lynch 1997), as observed by Rogers et al. (2005) in Homebush Bay, Australia, where a reduction in groundwater recharge during drought correlated with lower rates of surface elevation gain.

Sedimentation is vital in maintaining the elevation of coastal and estuarine wetlands, particularly under rising sea levels (Nichols et al., 2007). It has been suggested, however, that the rate of physical change occurring within estuaries in New Zealand has accelerated due to increased sediment yields as a result of erosion from human-induced land-use changes (Sheffield et al., 1995; Swales, Hume and Green, 1997; Swales, et al*.,* 2002; Hayward, et al*.,* 2006). It has been suggested that this increase in terrigenous sediment has provided a suitable substrate for mangrove colonisation (Young and Harvey, 1996; Ellis, et al*.,* 2004; Swales, et al*.,* 2007). Once established, mangrove aerial and subaerial root networks can further accelerate sedimentation (Furukawa, Wolanski and Mueller*,* 1997; Quartel et al., 2007) and therefore speed up the rate of landscape evolution (Woodroffe, 1992). Fine sediment is preferentially deposited among mangrove root structures (Furukawa and Wolanski, 1996; Massel, Furukawa and Brinkman*,* 1999), and in areas of seaward mangrove migration, the change from sandy to muddy substrate may result in a change or loss of biodiversity and abundance of benthic species (Thrush et al., 2003; Ellis et al*.,* 2004; Thrush et al*.,* 2004).

3.2.1 Techniques to measure surface elevation and sediment accretion

Changing surface elevation of wetlands and tidal flats is influenced by the processes of sedimentation, erosion and compaction. A variety of methods have been developed to measure both sediment accumulation and elevation changes in intertidal environments (Thomas and Ridd, 2004).

Firstly, it is beneficial to define the geomorphic processes which are to be measured. Sediment deposition is defined by many as the emplacement of particles on the seabed (McKee et al., 1983). This may include primary and secondary fluxes of particles (Lund-Hansen et al., 1997). Vertical sediment accumulation (also termed sedimentation or vertical accretion) refers to the thickness of these sediment deposits (Larcombe and Woolfe, 1999). Surface elevation change, or surface accretion, on the other hand, identifies the movement of the intertidal seabed surface relative to a subsurface datum (Cahoon et al., 1995) and inherently includes, although does not separate, processes of sediment deposition, sediment compaction, and/or subsurface movement driven by groundwater, biological processes or subsurface tectonic activity (Cahoon et al., 1995). The term shallow subsidence has been used to describe these processes (Cahoon et al., 1995). The rates of sediment accumulation and surface elevation change can be investigated by using a combination of field methods which are described below.

3.2.2 Methods to measure surface elevation change/surface accretion

Erosion pins

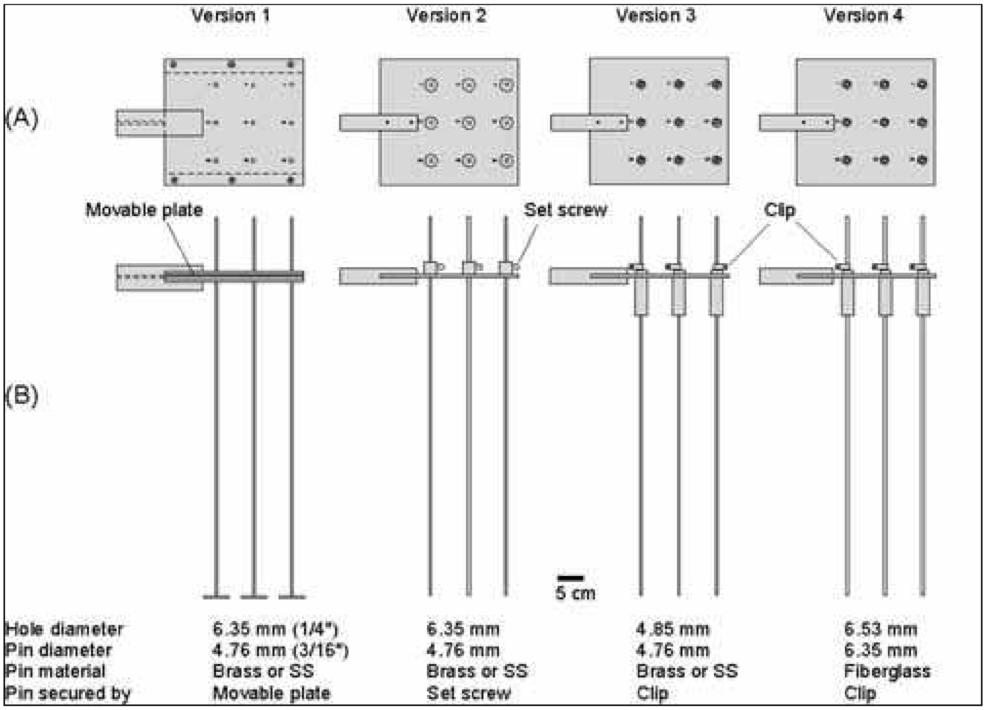
Erosion pins are robust, inexpensive and relatively quick to deploy (Thomas and Ridd 2004). Pins are planted in the ground with a designated height protruding above the sediment surface. Any change to the proportion of erosion pin above the sediment surface is read by an observer at time intervals from months to years. The reference datum is the bottom of the pin therefore this method can measure elevation change, which includes subsurface processes to the depth of that pin (Cahoon and Lynch, 1997). The length used in previous studies varies from around 30 cm to over 2 m depending on the expected sediment loads of the study site (Thomas and Ridd, 2004). The accuracy of this method has not been specified in published surveys, although Thomas and Ridd (2004) suggest readings can be estimated to the nearest millimetre.

A number of studies have measured surface elevation in mangroves using stakes or pins (sometimes also called graduated pegs), deployed in various lay-outs and densities. Bird (1971) and Spenceley (1977) investigated the land-building capacity of the pneumatophores of *Avicennia marina* in southern (temperate) Australia using a series of stakes. In New Zealand, Young and Harvey (1996) installed plots of stakes at increasing densities to examine the relationship between pneumatophore density and rates of surface accretion. Results suggest a positive correlation between pneumatophore density and surface accretion at densities > 250 m^{-2}. In addition to their experimental design, Young and Harvey installed transects of stakes through mangrove habitat to investigate spatial gradients of increasing surface accretion with distance from the mangrove fringe. Krauss et al. (2003) considered the degree of influence that the three main mangrove root structures (pegs, pneumatophores and knee roots) play on sediment retention. In this instance, one metre long steel pins were buried to a depth of 0.7 m, making sure each pin was no closer than 0.1 m to any other to reduce any effect of the pin on sediment retention.

Sediment Erosion Tables (SETs)/Rod surface elevation tables (RSETs)

The Sediment Erosion Table (SET) was designed specifically for intertidal surveys (Boumans and Day, 1993; Cahoon et al., 1995) and can measure both positive and negative elevation changes. The device consists of two main components. A permanent rod is driven into the substrate, preferably to refusal, and then cemented in place. A detachable arm is attached to the permanent rod on each field visit. At the end of the arm is a small square disc (table) with a series of holes that allow pins to be passed through it and then positioned on the ground surface (Figure 3.1). The length of each pin above the SET, or table, relates directly to the distance between the table and the ground and provides a measure of elevation change. The resolution of the device has been measured at between ± 1.3 mm (Cahoon et al., 2002b) and ± 1.5 mm (Cahoon and Lynch, 1997).

The sediment erosion table (SET) has been renamed ‘surface elevation table’ because this is a more apt description of the processes it measures (Cahoon et al., 2002a). The method has also been improved with the development of the rod surface elevation table (RSET) which has a lighter measuring arm than the original SET (Cahoon et al., 2002b). When using the RSET, pins are passed through a series of holes that run along a narrow steel arm which has replaced the square table of the SET (Figure 3.2).



	Version 1	Version 2	Version 3	Version 4
Hole diameter	6.35 mm (1/4")	6.35 mm	4.85 mm	6.53 mm
Pin diameter	4.76 mm (3/16")	4.76 mm	4.76 mm	6.35 mm
Pin material	Brass or SS	Brass or SS	Brass or SS	Fiberglass
Pin secured by	Movable plate	Set screw	Clip	Clip

Figure 3.1 The table design changes of the SET. Image from Cahoon et al., 2002a.

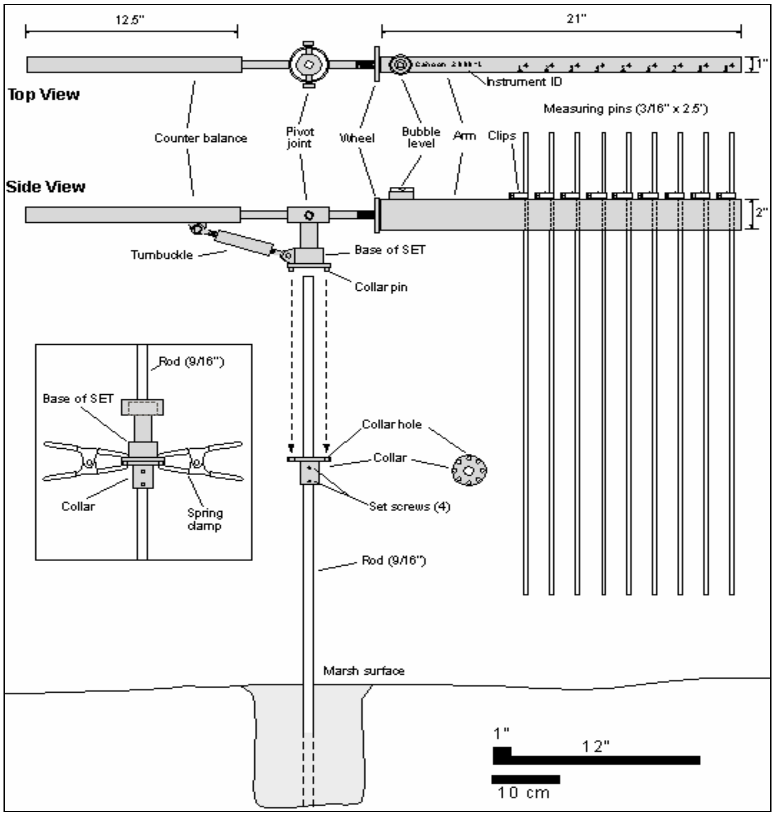


Figure 3.2 The newer RSET design, from Cahoon et al., 2002b.

There are now RSETs installed in two New Zealand sites; Tauranga Harbour (this study), and the Firth of Thames (Swales et al., 2007). A number of studies have utilised the SETs in the context of Australian mangrove systems (Rogers et al., 2005; Rogers et al., 2006) and tropical systems in the Carribean and the United States (McKee et al., 2007; Cahoon and Lynch, 1997; Whelan et al., 2005; and Cahoon et al., 2003).

3. *Short term radionuclides*

The radionuclides ^{234}Th and ^{7}Be have a short half life (24 and 53 days respectively) and so are often used to measure accumulation of sediments over a short time scale (Thomas and Ridd, 2004). This is an expensive technique due to the cost of processing samples which often prohibits intensive sampling. Studies instead tend to use either large spatial sampling intervals, and/or is used as a supplementary technique (Thomas and Ridd, 2004).

3.2.3 Methods to measure sediment accumulation

1. *Marker horizons*

Marker horizons have been used extensively in intertidal environments, with varying success. The method involves spreading a layer of material, such as sand or feldspar over the sediment surface (French and Spencer, 1993; Cahoon and Lynch, 1997). Small cores are collected months to years after the horizons have been laid, and net vertical accretion is then measured as the rate of accumulation above the marker horizon (French and Spencer, 1993).

The use of marker horizons is inexpensive, which benefits studies of large survey areas or for comparisons of different environments. This method will not provide robust results in all environments, particularly if a site receives limited fresh sediment, strong tidal currents which can potentially flush the marker medium, or where bioturbation from crabs and worms mixes the surface sediment with the marker (Thomas and Ridd 2004).

2. *Buried Plates/Tiles*

A similar method is the burial of solid plates. The size and material of the plate or tile may vary (Saad et al., 1999; Fan et al., 2004), however the concept is the same. A plate is placed at a recorded depth below the surface. The plates are left for one month or more after installation to allow stabilization of sediments above the marker levels. A series of readings of the depth of sediment above the buried plate are averaged to provide a value of vertical accretion, with an accuracy of + 2mm (Saad et al., 2004)

A variation to the marker methods above is to use an anchored tile (Pasternack and Brush, 1998). A ceramic tile is positioned flush with the intertidal surface and anchored by a buried steel rod. On repeated visits, all the material that had settled onto the tile is collected, dried and weighed. A variation of this collection method is the filter pad technique (Reed, 1989; French and Spencer, 1993) which uses pre-weighted filter papers to collect sediment. The filter paper is removed at each visit then dried and weighed. The results are given in a dry weight per unit area, typically in mg cm^{-2} as opposed to a vertical depth of sediment such as that provided by the marker horizons and buried plates.

3. Sediment Traps

Sediment traps are widely used in riverine and estuarine environments to measure sediment accumulation. Although the trap design specifications differ, they are typically cylindrical tubes with a closed bottom and open top and are buried with the opening sitting slightly above the substrate. In deeper environments they are positioned at a designated height in the water column and held in position by frames or rope (Thomas and Ridd, 2004).

There is still some debate as to the most appropriate design, particularly the ratio of length to opening, mostly because it is difficult to determine their accuracy (Gust and Kozerski, 2000). The likelihood of resuspension of material is reduced by increasing the aspect ratio of a trap. Butman (1986) suggested that a length-to-opening ratio of 6:1 improved trap efficiency, whereas Gust observed a close to zero bottom shear stress at the base of a trap (with a turbulent eddie at the surface of the trap) with a ratio of 10:1. Sediments collected in traps positioned in intertidal areas are less likely to be resuspended than those deployed in oceanic and lake environments, primarily because exposure during low tide allows time for sediments to settle before the next tidal inundation.

A study on the limitations of sediment traps suggest they over-estimate settling flux because they interrupt the natural processes whereby particles will be repeatedly resuspended before final burial in bottom sediments (Kozerski, 1994). Even so, they are an appropriate device to identify the type of sediment that is settling onto the intertidal surfaces and to provide an indication of gross sediment accumulation in the absence of readily available superior technology. Traps have been used in numerous studies to investigate sedimentation rates associated with mangroves, although with a range of trap ratios from 1:4 (Wolanski et al., 2005) to 1:10 (Ellis et al., 2004), or in some instances, not specified (Furukawa et al., 1997; Victor et al., 2004).

3.3 STUDY OBJECTIVES

The aims of this chapter are: (i) to document the spatial evolution of mangrove forests using remote sensing and GIS tools; (ii) to record plant architecture characteristics of these temperate mangrove stands; and (iii) to determine the physical changes occurring within the study sites due to the spread of mangrove habitat. In so doing, this chapter addresses the following thesis objectives:

a) determining morphometric characteristics of mangrove forests within Welcome Bay and Waikareao Estuary,

and;

b) identifying the physical changes that have occurred due to mangrove expansion, including surface sedimentology and changes to surface elevation due to sedimentation and/or erosion.

3.4 STUDY SITE

The two embayments selected for this study were Waikareao Estuary and Welcome Bay, within Tauranga Harbour (Figure 3.3). They are two of many re-entrant bays found along the landward (west) flank of the harbour, a barrier enclosed estuarine lagoon (Healy et al*.,* 1996), located on the east coast of the North Island of New Zealand (Lat. 37° 40'S, Long. 176° 03'E). Urbanisation is pronounced around the fringes of both embayments and large-scale land clearing for further development is on-going within the Waikareao catchment. The harbour experiences a semi-diurnal tidal regime with a mean spring tidal range around 2.2 m at the southern entrance (Davis and Healy, 1993). Peak tidal currents flowing through this entrance have been measured at 1.2 – 1.3 m s^{-1} (Davies-Colley and Healy, 1978). Welcome Bay is situated roughly 10 km from the harbour entrance and therefore likely to experience lower tidal velocities than the 0.7 m s^{-1} maximum measured at the narrow entrance of Waikareao Estuary (White, 1979), which is 4.5 km south of the harbour mouth. Tauranga Harbour is dominated by intertidal flats (White, 1979; Davis and Healy, 1993; Lelieveld, Pilditch and Green, 2004), with 70 % of the interior exposed at low tide (de Lange and Healy, 1990). The harbour interior is dominated by sandy sediments with an increased proportion of mud in the upper reaches (White, 1979; Hope, 2002; Mom, 2005). The intertidal flats of the study sites are exposed for approximately five hours between low and high tide, with an increase in exposure of 30 to 40 minutes toward the head of both estuaries. Each site possesses a unique set of geomorphic and hydrological features that will influence the amount and type of sediment entering the embayment, and the hydrodynamic forces (tidal and meteorological) that will interact with the sediment transport processes (Table 3.1).

Table 3.1 Morphometric characteristics, freshwater input, and sediment yields of Welcome Bay and Waikareao Estuary, Tauranga Harbour, New Zealand. *taken from Surman (1999).

Estuary	Estuary size (km^2)	Catchment size (km^2)	North & north-east maximum fetch	West & south-west max. fetch	Distance from ocean entrance (km)	Mean freshwater flow l/sec^{-1}	Suspended Sediment Yield t/yr^{-1}
Welcome Bay	1.07	20	5 km	1.7	10	179 *	280 *
Waikareao Estuary	3.25	75	6 km	1.6	4.5	2,450 *	37,940 *

The region experiences a temperate climate with mean summer monthly maximums ranging from 22 to 24 °C over the months of December to March. Winter maxima range from 14 to 15 °C and minimums from 5 to 6 °C. Mean rainfall for the Tauranga area is around 1,200 mm per year. Westerly winds dominate in strength and frequency with occasional tropical cyclonic systems moving South and South-east (Quayle, 1984).

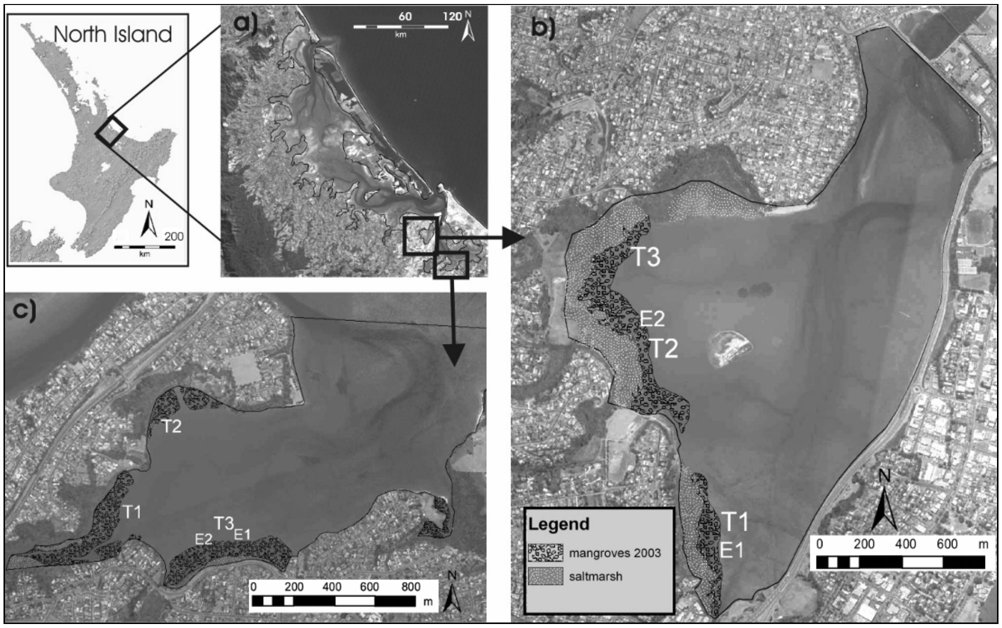


Figure 3.3 **Welcome Bay (c) and Waikareao Estuary (b), embayments of Tauranga Harbour (a), New Zealand. T1, T2, T3 = transect locations. E1 and E2 = exclusion plots used to determine seedling establishment and survival.**

3.5 METHODS

3.5.1 Mangrove Expansion

Spatial assessment of changes in the lateral coverage of mangrove vegetation has been mapped using aerial photographs dated 1943, 1982, 1996 and 2003. All analog data were transformed into geo-referenced digital format using ArcMap software, according to the New Zealand Map Grid co-ordinate system.

3.5.2 Monitoring Lay-out

Transects for the purpose of monitoring surface elevation change and sediment texture, were established at three sites within Welcome Bay and Waikareao Estuary, and five stations were marked along each of these transects. Stations 1 and 2 were positioned within mangrove habitat, 20 m and 10 m landward of the seaward mangrove fringe, respectively. Station 3 marked the seaward edge of mangrove habitat and was not sampled due to the potential for unpredicted mangrove removal. In analysis, Station 3 was termed ‘0 m’. Station 4 was located on the bare tidal flats, 10 m from the mangrove fringe, and Station 5 positioned 40 m seaward of the mangrove fringe.

3.5.3 Tree Physiognomy

Forest structural variables of tree height and pneumatophore density were measured at three sites within each of the estuaries. At each site, three 16 m^2 plots were randomly selected and marked, and all tree heights recorded. The plots were positioned between 5 m and 25 m from the tidal-flat/mangrove boundary, minimising edge effect. As such, all data recorded is of trees located in the zone where most interaction between tides and sediment occurs. It is important to note that the oldest trees positioned proximal to the landward margins are not included in this study. One plot per site was revisited in 2007 and all trees re-measured to determine general vertical growth rates.

First year seedlings that establish beyond the seaward edge of mature mangrove vegetation are removed by local residents, to prevent on-going colonisation. In order to gather preliminary data on the rate of seedling establishment in the absence of a natural cohort, two 12 m x 12 m 'exclusion plots' were marked out, with posts hammered in to each corner and tape used to mark the plot boundary. All seedlings seaward of the mature tree margin were counted within each plot during each field visit (2005 to 2007).

3.5.4 Surface Elevation Changes

Two methods have been used in this study to measure surface elevation change. An unobtrusive option was required for the bare intertidal flats due to human (and dog) traffic and so stainless steel base-plates (1 m^2) were buried approximately 15 cm below the stratum, positioned 10 m and 40 m seaward of the mangrove edge along 3 transects (see Figure 3.3 for transect locations). All plates were left to settle for over one month before initial readings were taken. Twenty readings of the depth of sediment above the plate were recorded on each visit and averaged to give a single value of surface elevation change. Smaller, 10 cm x 10 cm plates were trialed within the mangrove zones, with limited success as the high density of pneumatophores prohibited even placement and burial. These were replaced with stainless steel erosion pins (length 0.7 m, diameter 5 mm), positioned in clusters of seven pins (with a minimum of 10 cm between each pin), and driven into the substrate with 0.2 m remaining above the surface. The seven readings on each visit were averaged to give a single value of elevation change for each cluster of pins. Results are expressed as 'surface elevation change' rather than 'sedimentation rate'. Sedimentation rates are a measure of the volume of sediment that has accumulated above a given surface, whereas 'surface elevation change' incorporates any influence of sediment compaction, shallow subsidence or changes in root-mass (Cahoon et al*.,* 2000).

Surface accretion of mangrove habitat is at least partially influenced by the production and decomposition of mangrove roots (Cahoon et al., 1995). The placement of pins to a depth of 0.5 m penetrated beyond the mangrove root zone to ensure any biological processes were included. This also increased the stability of the pins, and although any vertical movement was not directly measured against any known datum, the first reading after installation mostly showed no change in surface elevation, which would suggest the pins were held firmly into their vertical position.

A shell bed up to 30 cm deep was widespread across the sandflats of both study sites, starting at depths of around 15 cm. Rather than dislodge the shell beds and disrupt the overall structure of the sediments, it was preferred to place the baseplates above the shell layer. The shallower burial of the plates also meant they could be located using a metal detector should any of the markers be destroyed or removed.

Generally, using a single method provides data for statistically valid analysis. For the purpose of this study, a single method to monitor surface elevation was not suitable. The combination of buried base plates used on the bare tidal flats and erosion pins among the mangroves was necessary to address safety issues and comply with the concerns of local residents. As such, the analysis identified general trends rather than statistically significant differences between sites and habitats.

3.5.5 Sediment Texture

Triplicate surface sediment samples were collected at each station during the southern-hemisphere summer (February) and winter (June or July) from winter 2005 to summer 2007. Samples were treated with 10% hydrogen peroxide (H_2O_2) to remove organic material. Calgon was then added for deflocculation, and samples analysed for grain size distribution using the Malvern Mastersizer S Version 2.19.

3.5.6 Sediment Traps

Sediment accumulation rates for Waikareao Estuary were measured using cylindrical PVC sediment traps, with a closed bottom, an internal diameter of 70 mm and an aspect ratio of 1:8, similar to Butman (1986). Traps were deployed in proximity to the three permanent transects with the trap mouth positioned 10 cm above the bed to eliminate the addition of any bed surface transport from saltation (Ellis et al., 2004). At each site, the sediment traps were positioned at the following positions (a) 10 m into mangrove habitat from the bare flat boundary, (b) 5 m into mangrove habitat from the bare flat boundary, and (c) 10 m from mangrove edge onto the bare tidal flat. Traps were collected after one month, returned to the lab and the contents were oven-dried to a constant weight. Sediment accumulation rates of dried sediment are expressed in grams *per* square metre *per* month (g m^2 mo^{-1}).

3.6 RESULTS

3.6.1 Mangrove Expansion

The temporal increase in aerial extent of mangroves between 1943 and 2003 is documented in Table 3.2. Because of the poorer quality of the 1943 aerial photographs it is possible that some fringing terrestrial vegetation was included in the outline of mangrove habitat. Even so, analysis suggests mangrove habitat covered < 1 % of the intertidal area of Welcome Bay and Waikareao Estuary in 1943 and was only a narrow strip of trees found toward the head of the estuary. Colonisation occurred mostly in the upper reaches prior to 1982, after which new cohorts appeared toward the middle of both estuaries and then continued expanding seaward (Figure 3.4). Between 1996 and 2003 Waikareao experienced a threefold increase in mangrove habitat, whereas Welcome Bay experienced a slight net reduction in mangrove area due to some manual clearing of mature shrubs by local residents, as well as the removal of seedlings. Any further seaward expansion has been halted as a result of human intervention via removal of seedlings on bare flats after each annual drop.

Table 3.2 Areal variation in mangrove coverage of Welcome Bay and Waikareao Estuary, Tauranga Harbour.

ESTUARY	Estuary area (ha)	Mangrove coverage (ha) in 1943	% cover in 1943	Mangrove coverage (ha) in 1982	% cover in 1982	Mangrove coverage (ha) in 1996	% cover in 1996	Mangrove coverage (ha) in 2003	% cover in 2003
Welcome Bay	160	0.4	<1	7.2	4.5	15.2	9.5	14.6	9
Waikareao Estuary	260	??	< 1	2.1	<1	4.5	1.7	15.7	6

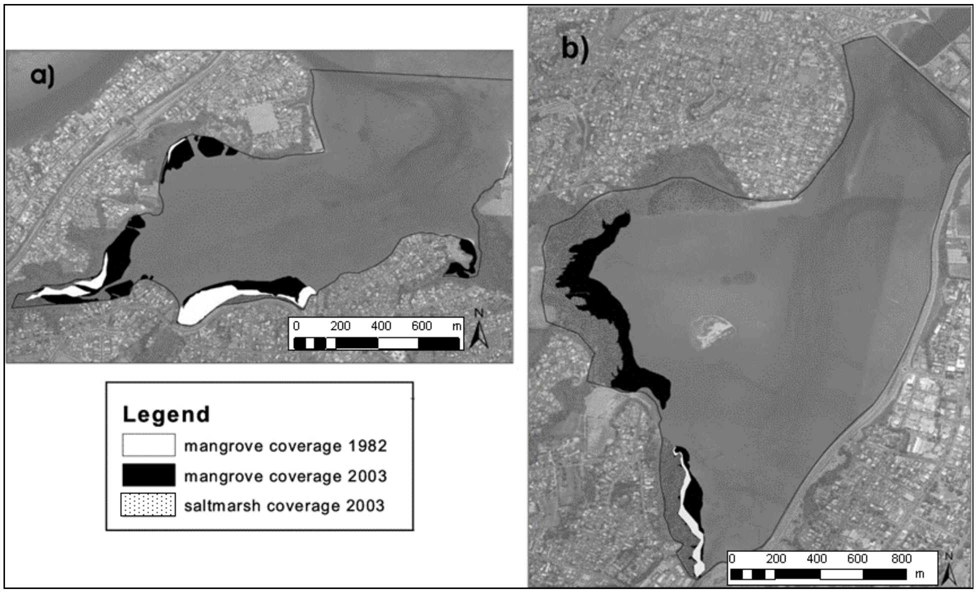


Figure 3.4 2003 aerial photographs (courtesy of Environment Bay of Plenty) of Welcome Bay (a) and Waikareao Estuary (b) with overlays of mangrove coverage in 1982 and 2003.

3.6.2 Tree Physiognomy

Mangrove stands within Welcome Bay are generally taller than those at Waikareao, with mean shrub heights ranging from 1.17 m to 1.43 m (Figure 3.5A), and also generally show a lower tree density (Figure 3.5B). A mean mangrove height increase of 7.5 cm yr^{-1} (± 1 cm) calculated for the two sites of Waikareao is close to double that of 3 cm yr^{-1} (± 1 cm) computed for Welcome Bay (Figure 3.5A). Pneumatophore density (Figure 3.5C) ranged from 204 m-^{2} (± 100) to just under 540 (± 48) m-^{2}. The highest numbers were recorded near the head of the estuary, at Site 1, in Waikareao whereas the highest values in Welcome Bay were found at Site 3, closer to the middle of the estuary.

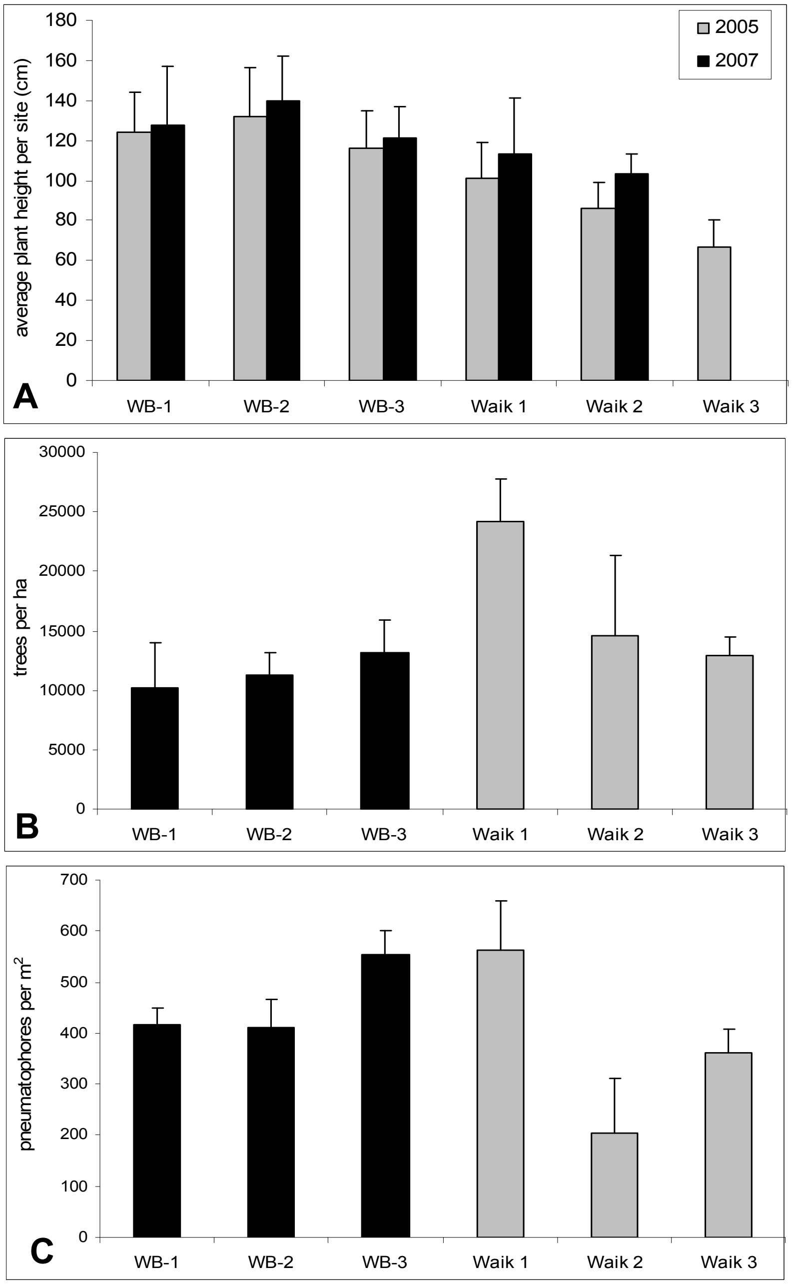


Figure 3.5 **Mean plant heights (SE) at each monitoring site within Welcome Bay (WB) and Waikareao Estuary (Waik) measured in 2005 (grey) and 2007 (black) (A). Tree density (SE) in Welcome Bay (black) and Waikareao (grey) (B); and pneumatophore density (SE) (C).**

3.6.3 Seedling Establishment

A stepped increase in seedling numbers was recorded over a period of two years at Waikareao Estuary, with mortality of < 40% occurring between summer 2006 and summer 2007. Highest seedling densities of 14 per m^2 were counted in Plot 2, toward the middle of Waikareao Estuary, as well as the highest survival rate of approximately 80% from summer to winter 2006. A subsequent increase in seedling density of 63% was calculated between winter 2006 and summer 2007. Welcome Bay exclusion plots maintained lower seedling numbers of < 1 per m^2 in Plot 1 and < 2 per m^2 in Plot 2 (Figure 3.6).

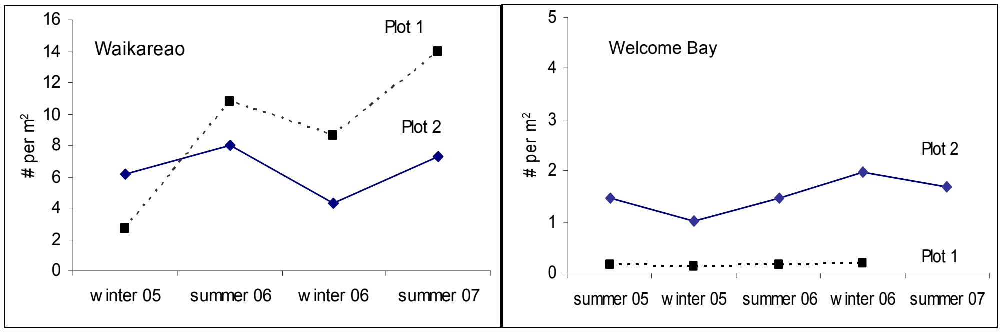


Figure 3.6 **Seedling densities calculated for each 12 m^2 exclusion plot at Waikareao Estuary and Welcome Bay.**

3.6.4 Surface Sediment Texture

A general trend of coarsening texture in surface sediments along a transect was recorded, with silty sediments (3-9 – 62.5 µm, as defined by Folk, 1974) dominant within mangrove stands, and fine sands found on the bare tidal flats (Figure 3.7). An exception, however, was seen in Transect 2 at Waikareao which displayed a considerably smaller range in mean grain size, with mangrove and bare tidal flats sediments consisting of very fine or fine sands. Total Organic Content (TOC) of sediment is higher in mangrove habitat (~ 6 to 13 %) than adjacent bare flats (~ 2 to 6 %), as shown in Figure 3.8, with Welcome Bay mangrove sediments possessing slightly higher TOCs than Waikareao Estuary. A strong relationship between sediment mud content and TOC is typical of estuarine sediments, and was evident in analysis of 2007 data, as shown in Figure 3.9 ($R^2 = 0.75$)

An assessment of temporal variation in surface sediment texture highlighted a marked increase in mud content of samples collected on bare flats of Transects 1 and 2 in Welcome Bay on 8 February 2006 (southern hemisphere summer), possibly as a result of increased terrestrial sediment input after heavy rainfall (82 mm in 48 hours) the week prior to monitoring. Samples were also collected in Waikareao Estuary one day after heavy rainfall (129 mm) on 19 May 2005 (southern hemisphere winter), however there appeared to be no discernable variation in overall textural composition on that occasion.

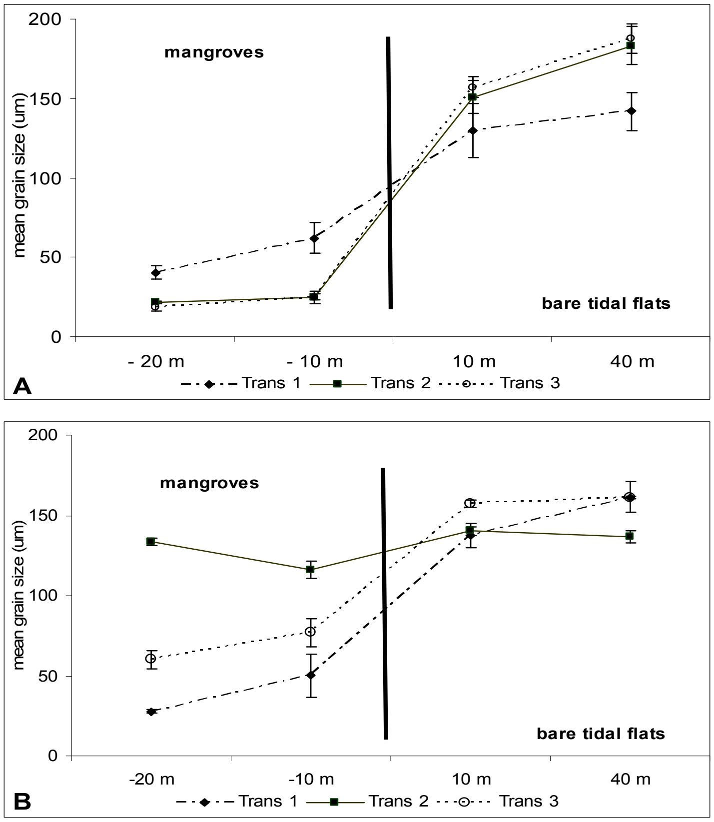


Figure 3.7 Mean (± SE) grain size at each station of Transects 1, 2 and 3 at Welcome Bay (A) and Waikareao Estuary (B).

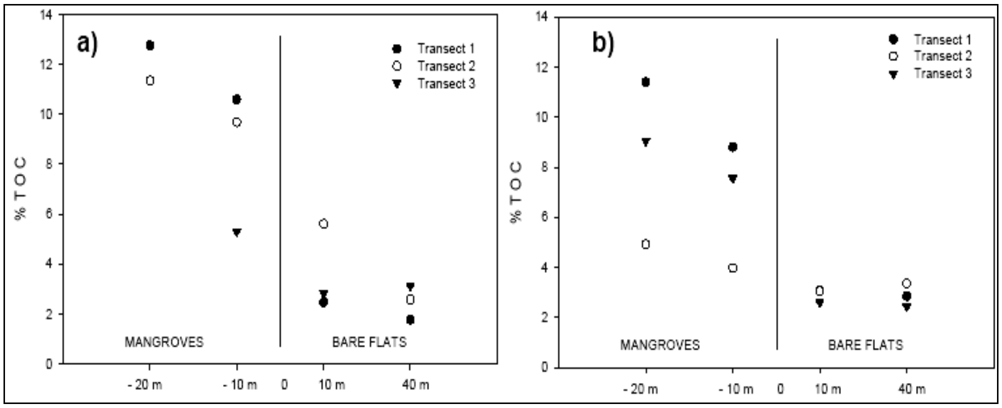


Figure 3.8 Total Organic Content of sediment samples collected along each transect of Welcome Bay (a) and Waikareao Estuary (b), in February (summer) 2007.

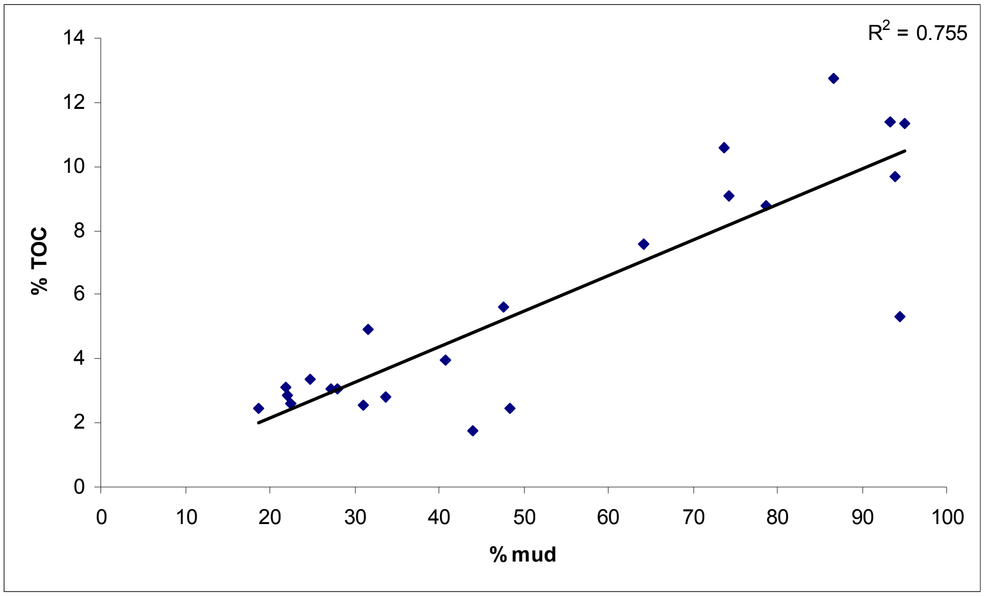


Figure 3. 9 **A strong relationship between sediment mud content and % Total Organic Carbon is evident from analysis of 2007 data for both Waikareao and Welcome Bay (R^2 = 0.755).**

3.6.5 Sediment Traps

Results from sediment traps installed over the summer months of mid-January to mid-Febuary 2007 and mid-Feburary to mid-March 2007 are displayed in Figure 3.10. The most noticeable trends were that (1) higher accumulation rates were measured toward the head of the estuary (Transect 1), particularly over the second deployment; and (2) sediment traps positioned on the bare intertidal area 5 m beyond the mangrove fringe received more sediment than those positioned inside the adjacent mangrove stands. The greatest sediment accumulation was recorded during the deployment from mid-Feburary to mid-March (~ 32,000 g m^2 mo^{-1}) which is more than double that measured at the same bare-flat location for the previous deployment period. Accumulation rates within the mangroves range from 1,600 to almost 29,000 g m^2 mo^{-1}, and intertidal sediment accumulation ranged from 3,370 to 32,000 g m^2 mo^{-1}. Mean grain size of the trapped sediment was 51 µm (± 18) and TOC was 9% (± 3.7).

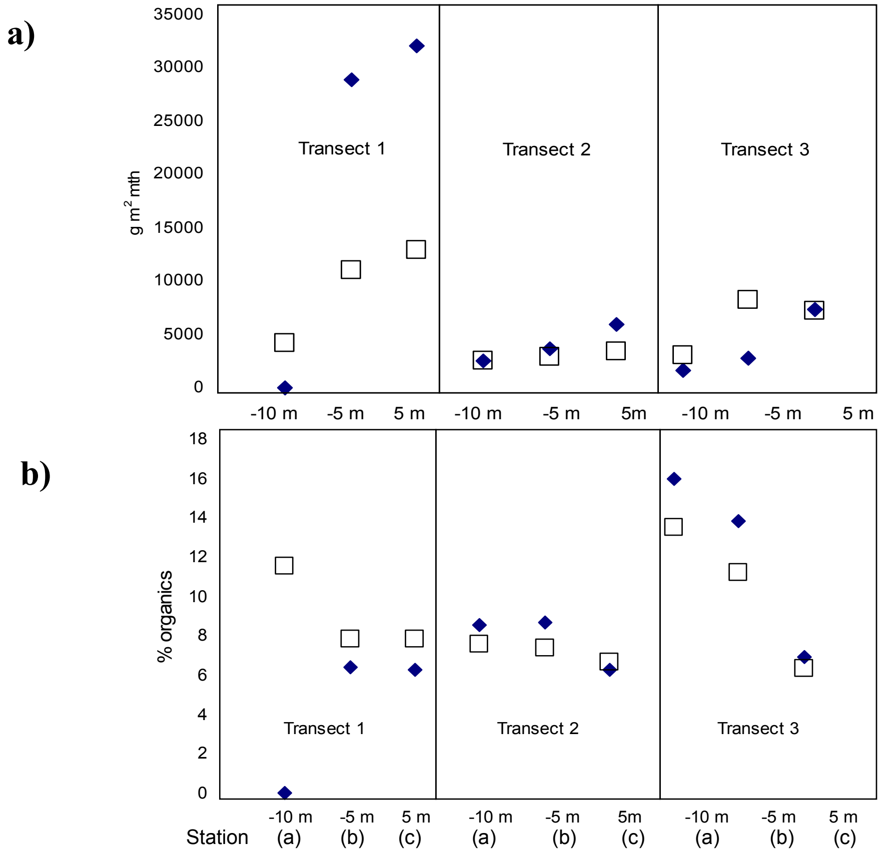


Figure 3.10 **Volume (a) and % organic content (b) of sediment collected in traps deployed for January 2007 (diamonds) and February 2007 (open squares) along Transects 1, 2 and 3 at Waikareao Estuary. a = mangrove habitat 10 m from bare flat boundary; b = mangrove habitat 5 m from bare flat boundary; c = 10 m onto bare flat from mangrove edge. Transect 1, station (a) for January was vandalised so zero values equate to null results.**

3.6.6 Surface Elevation Change

Incremental change in surface elevation measured in mangrove habitat within Welcome Bay and Waikareao Estuary, between mid-2005 and early 2007, is displayed in Figures 3.11 and 3.13. Similarly, temporal changes of surface elevation across bare flat habitat are shown in Figures 3.12 and 3.14. Erosion pins and base plates were installed in Waikareao Estuary in March 2005 and the first readings were taken in May 2005. No change in elevation within the mangrove habitat between the time of installation and May 2005 was recorded, therefore the May 2005 measurements are not included in the graphs. Erosion pins at Transect 3 were reinstalled in May 2005 due to vandalism.

Base plates and erosion pins were installed in Welcome Bay in April 2005 and the first measurements were taken in July 2005. Similar to Waikareao, the July measurements also showed no change in surface elevation and therefore not included in the analysis.

The erosion pins were a functional and cost-effective method of measuring surface elevation. Variation in surface accretion within each cluster of pins reflects the small depressions and domes associated with the mangrove root system. Introducing a greater density of erosion pins to each cluster may have reduced the standard deviation associated with the calculation of mean elevation.

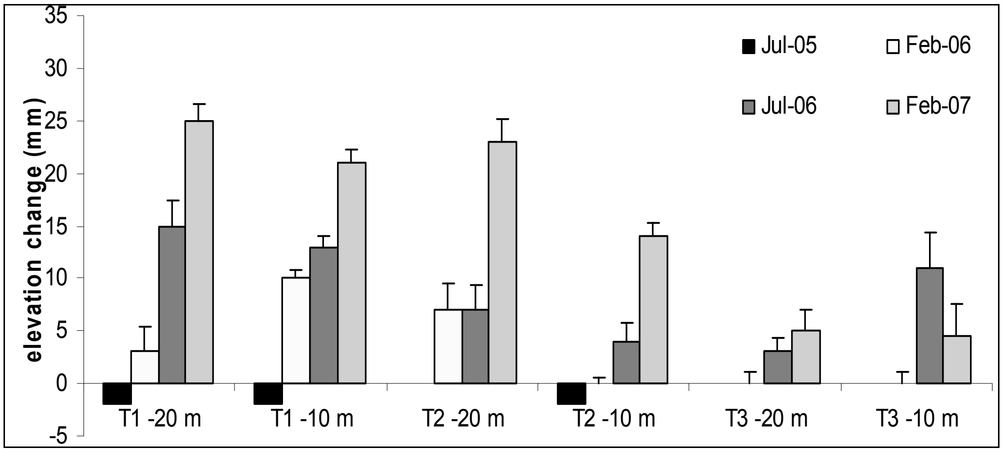


Figure 3.11 **Surface elevation change (+ SE) measured on each field visit between July 2005 and February 2007 in mangrove habitat at Waikareao Estuary, 20 m and 10 m landward of the mangrove fringe at Transects 1, 2 and 3 (T1, T2 and T3).**

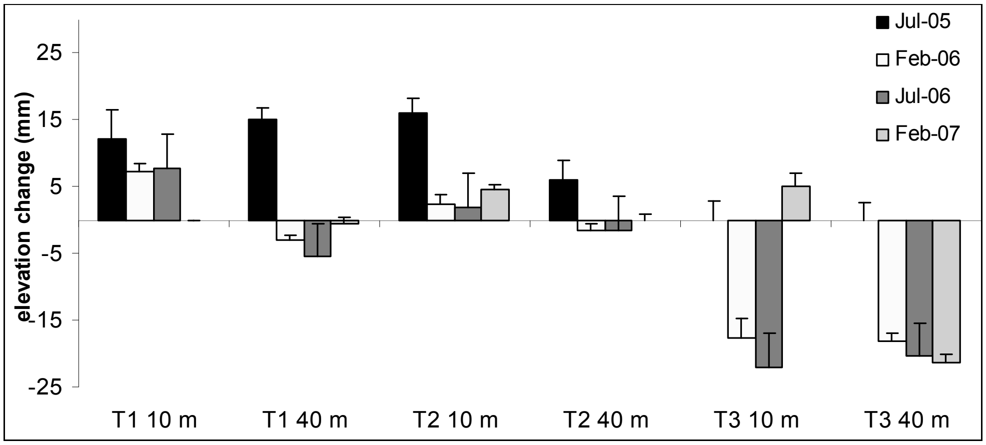


Figure 3.12 **Surface elevation change (+ SE) measured on each field visit between July 2005 and February 2007 on bare intertdal flats at Waikareao Estuary, 10 m and 40 m beyond the mangrove fringe.**

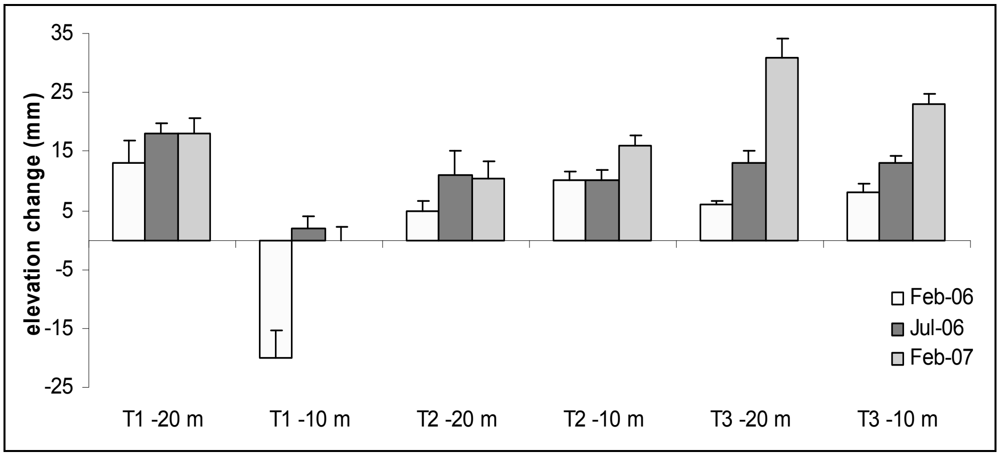


Figure 3.13 Surface elevation change (+ SE) measured on each field visit between February 2006 and February 2007 in mangrove habitat at Welcome Bay, 20 m and 10 m landward of the mangrove fringe.

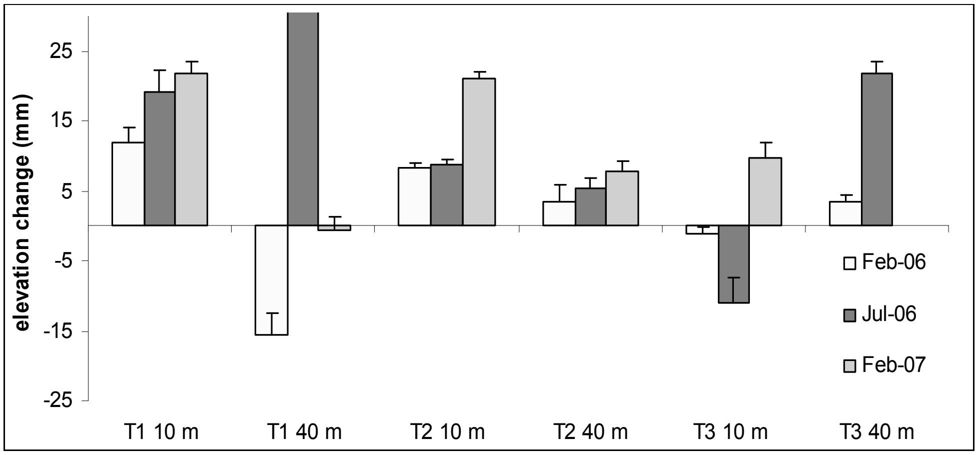


Figure 3.14 Surface elevation (+ SE) change measured on each field visit between February 2006 and February 2007 on bare intertdal flats at Welcome Bay, 10 m and 40 m beyond the mangrove fringe.

Annual rates of surface elevation are shown in Figure 3.15. The greatest increases of 16.5 mm yr^{-1} and 21 mm yr^{-1} were recorded toward the head of the estuary at Waikareao (Transect 1, Station 1), and mid-estuary at Welcome Bay (Transect 3, Station 1), respectively (Figure 3.15). Along all transects, higher values of topographical change were recorded within mangrove vegetation 20 m from the mangrove/intertidal flat boundary relative to measurements taken 10 m from the boundary. Topographical changes documented for bare flats adjacent to mangrove vegetation mostly reflect a fall in surface elevation at Waikareao.

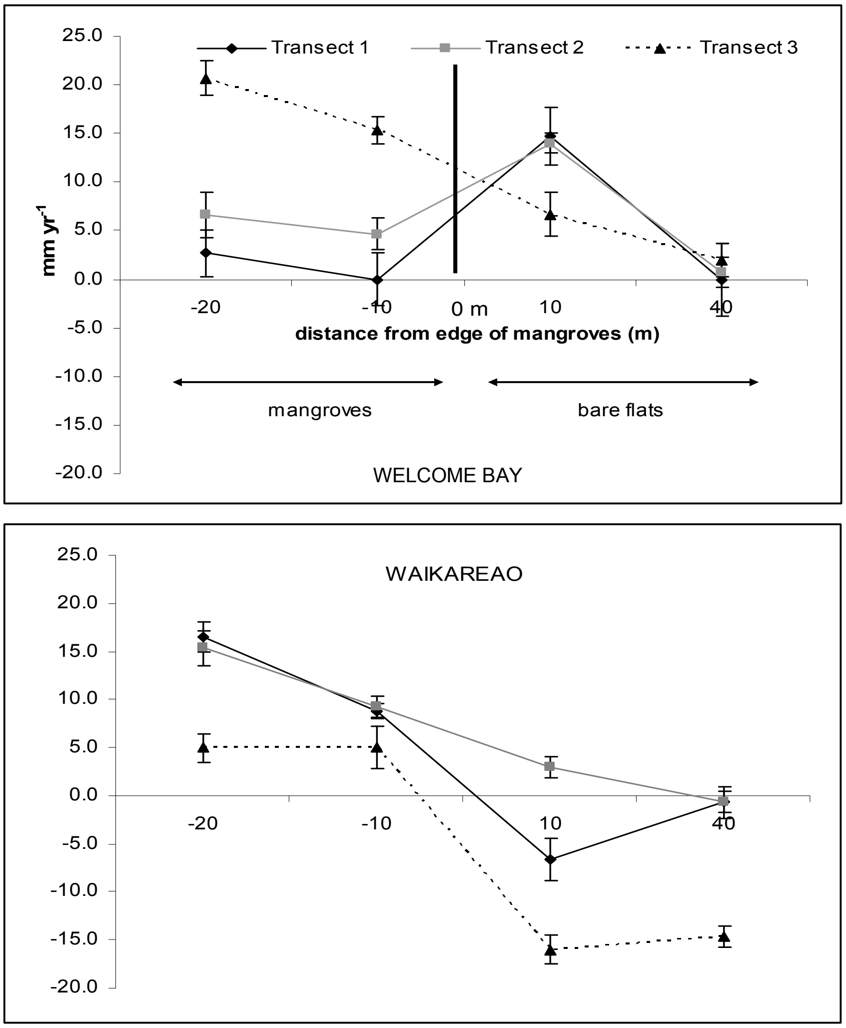


Figure 3.15 Annual rate of surface elevation (mm yr^{-1}) along each transect at Welcome Bay and Waikareao Estuary. Error bars represent the mean of standard errors from each averaged result (*e.g.* each field visit).

Interestingly, at Welcome Bay lower rates of accretion in mangroves at Transects 1 and 2 (0.0 to 6.7 mm yr^{-1}) corresponded to much higher rates of elevation on the bare tidal flats 10 m beyond the adjacent mangrove fringe (14.0 to 14.7 mm yr^{-1}). This high rate of accretion did not extend to the tidal flat Station 5 positioned 40 m from the mangrove fringe however, where values range from -0.7 to 0.7 mm yr^{-1}. Estuary-averaged rates of surface elevation were calculated to be 10 mm yr^{-1} (± 4.9) for Waikareao Estuary, and 7.4 mm yr^{-1} (± 8) for Welcome Bay. These averaged results and relatively high standard deviations illustrate variability not only between estuaries, but also within each estuary.

3.7 DISCUSSION

Data from this study show considerable mangrove expansion at both sites within Tauranga Harbour since the 1960s. Other studies have documented mangrove expansion in New Zealand (Burns and Ogden, 1985; Young and Harvey, 1996; Morrisey et al*.*, 2003; Ellis et al*.*, 2004), however this finding is not ubiquitous (Mom, 2005). Mangrove expansion has often been attributed to increased sedimentation resulting from land-use changes (Burns and Ogden, 1985; Young and Harvey, 1996; do Amaral et al*.*, 2006). Increased nutrient loads are also often associated with human impacts on estuaries, and a recent study has found that once mangroves are established, nutrients can play a secondary role in mangrove expansion by enhancing plant growth (Lovelock et al., 2007). Continued seaward migration of mangrove habitat would likely occur within the study sites in the absence of human intervention, as demonstrated by the seedling survival of > 50% within the exclusion plots at Waikareao Estuary. Low seedling numbers counted in the plots of Welcome Bay were surprising and may be as a result of either eager local residents or exposure to cross-shore waves during strong north or north-east wind events.

Mangrove expansion has also been documented in numerous studies of south-eastern Australian (reviewed in Saintilan and Williams, 2000). Spatial analysis has idenfied a trend of landward incursion of mangroves in to salt marsh rather than in the seaward direction demonstrated by many New Zealand studies (Schwarz, 2002; Morrisey et al., 2003; Ellis et al., 2004; Mom, 2005; Swales et al., 2007). The effects of sediment autocompaction and subsidence have been identified as the geomorphic processes responsible for the invasion of salt marsh in some sites in southeastern Australia, particularly within locations that experience drought conditions sufficient to lower the water table (Rogers et al., 2005; Whelan et al., 2005). The stark contrast in spatial evolution of mangrove and saltmarsh in southeastern Australia compared to New Zealand could be due to the high rate of active infilling of New Zealand embayments compared to their counterparts on stable continental margins (Hume et al., 1989; Sheffield et al., 1995; Swales et al., 2002) which could reduce the effect of autocompaction. Alternatively, climatic conditions (i.e. fewer or milder drought conditions) may contribute by maintaining groundwater levels.

The mangrove habitat of Welcome Bay and Waikareao Estuay is comparatively young, with most plants < 25 years old. Plants stand < 1.5 m in height, and appear to be growing vertically at less than 10 cm yr^{-1}, likely due to physiological limitations of low temperatures experienced at this latitude (Beard, 2006). All but one monitoring site has dense vegetation with > 90 % canopy cover. Transect 2 at Waikareao Estuary was the exception, and although this site has a similar number of plants per hectare to Transect 3, canopy cover is $\leq$ 50 %. This mangrove site also possesses a fine-sand substrate rather than silt, and fewer pneumatophores. It is likely a combination of age (or youngness) of the vegetation (Young and Harvey, 1996) and the dominance of relatively coarser grained sediment that determines the lower pneumatophore density at this location. Pneumatophore densities measured at all other transects in this study were found to be mostly higher than other New Zealand locations (Young and Harvey, 1996; Morrisey et al., 2003; Ellis et al., 2004; Alfaro, 2005), possibly because of the high mud content, considerable sediment accretion observed in this study, and the associated anoxia.

The sedimentary environment of mangrove habitat is, for the most part, distinctly different to adjacent bare tidal flats. Mangrove habitat within both estuaries is characterized by silt-dominated sediments with total organic content of 5-13%. Bare tidal flats, conversely, have a bed surface consisting of fine-sand with lower levels of organics. The one exception to this observation, however, was toward the middle of Waikareao Estuary (Transect 2) which maintained fine sand at all stations. Pneumatophore densities and plant cover are lower here, but interestingly the bed-level was rising at 10 mm yr^{-1} (Station 2) and 15 mm yr^{-1} (Station 1). Sediment trap results suggest that less sediment is settling at the Transect 2 mangrove sites compared with Transect 1, yet rates of surface elevation are within 1 mm of the Transect 1 stations. The cause of this can only be speculated, but could be because (1) the location is a receiving environment for incoming marine sediments during storm events (keeping in mind that silt, probably terrestrial, was collected in sediment traps); or (2) its location behind a small island made it an area of dampened tidal flows, allowing the settling out of fine sand and silt, with the low pneumatophore densities allowing re-suspension and transport of the clay and silt particles during wind-wave events.

The greatest increases in surface elevation measured in this study were not always found to occur in mangrove habitat. Along four of the six transects a higher rate of surface elevation was recorded in mangrove habitat compared with the adjacent bare flats, despite the lower sediment availability (as demonstrated by sediment trap results). This trend was documented in Stokes, Healy and Cooke (2009) and Van Santen et al. (2006), and is reportedly due to less re-entrainment within the vegetation areas compared with the bare flats (Furukawa and Wolanski, 1996; Van Santen et al., 2006). The two transects positioned on the north side of Welcome Bay (Transects 1 and 2), however, recorded surface elevation rates of around 16 mm yr^{-1} on the bare flats (Station 4) which is more than double that recorded in the adjacent mangrove habitat. Welcome Bay is the narrower of the two embayments and is twice the distance from the ocean entrance (therefore experiencing reduced tidal velocities); fetch distances are smaller, as is the freshwater input. A combination of these parameters would influence sedimentation on the tidal flat and provide less opportunity for re-suspension. This study would have benefited from a deployment of sediment traps within Welcome Bay to determine if the rising bed level of bare tidal flats is due to high sediment loads or simply due to lower tidal velocities.

Sediment accretion in mangroves is a well-documented phenomenon (e.g. Cahoon and Lynch, 1997; Saad et al., 1999; Victor et al., 2004; Alongi et al., 2005), however the rate of both sedimentation and surface elevation change varies greatly, dependent upon a range of factors such as sediment supply, tidal regime, vegetation type and root density (Cahoon and Lynch, 1997; Krauss, Allen and Cahoon, 2003). Rates of surface elevation change in mangrove habitat presented in this study range from < 1 mm to 21 mm yr^{-1}, similar to sedimentation rates reported in other New Zealand estuaries (Young and Harvey, 1996; Ellis et al., 2004). Sedimentation rates measured in mangrove habitat in Florida and temperate Australia ranged from 2 to 8 mm yr^{-1} (e.g. Lynch et al., 1989; Cahoon and Lynch, 1997; Rogers, Wilton and Saintilan, 2006). Higher sedimentation rates (up to 100 mm yr^{-1}) have been recorded in some mangrove fringed estuaries at sites that receive very high sediment loads, both in New Zealand (Swales et al., 2007) and elsewhere (Alongi et al., 2005; Van Santen et al., 2006).

3.8 CONCLUSIONS

The aims of this study were to describe two sites of expanding temperate estuarine mangrove forest and to determine whether sedimentation within these localities was driven solely by the presence of these mangroves. A combination of erosion pins and buried base plates were used to determine changes in surface elevation, and grain size analysis provided an overview of surface sediment texture. The mangrove sites were characterized by dense stands of shrubs, mostly measuring <1.5 m in height, and increasing in vertical height at < 10 cm yr^{-1}. Since the 1940s mangroves have colonised approximately 15 hectares within both Welcome Bay and Waikareao Estuary, which represents ~ 9 and 6% respectively of the estuarine intertidal zone. Seedlings are currently being cleared by local residents, however, survival rates of up to 80% in exclusion plots suggests that, in the absence of human intervention, continued expansion would be likely. The public perception, and intuitive assumption, that mangroves are actively trapping mud is confirmed by grain size analysis that shows mangrove sediments to be dominated by silt and clay, whereas adjacent tidal flats are made up mostly of fine sands. The main physical change occurring as a result of mangrove habitat development is an increase in surface elevation at rates of up to 21 mm yr^{-1}. Sedimentation, however, is not solely occurring within mangrove habitat. Surface elevation rates of 5 mm yr^{-1} to 15 mm yr^{-1} were recorded on the more protected bare tidal flats fronting the mangrove fringe. It would appear, therefore, that mangroves are not the sole driver of topographical change at the study sites.

3.9 PHOTO GALLERY: Waikareao and Welcome Bay

Figure 3.16 Mangrove plant and seedlings at Transect 1, Waikareao Estuary (left) and Transect 3, Welcome Bay (right).

Figure 3.17 Measuring plant heights of mangroves at Transect 1, Waikareao Estuary, 2005.

Figure 3.18 Exclusion Plot 1, Waikareao Estuary, February 2007.

Figure 3.19 Exclusion Plot 2, Waikareao Estuary, February 2007.

Figure 3.20 Mangrove vegetation and seedling fringe at Transect 1, Waikareao Estuary, February 2007.

Figure 3.21 **Seedlings (12 – 24 months old), positioned seaward of Transect 2, Waikareao Estuary, February 2007.**

Figure 3.22 **Measuring plant heights at Transect 2, Welcome Bay.**

Figure 3.23 Sea lettuce (*Ulva*) covering mudflats that front the mangroves at Transect 3, Welcome Bay, July 2005.

Figure 3.24 Exclusion Plot 1, Welcome Bay, May 2005.

3.10 LITERATURE CITED

Alfaro, A. C., 2005. Benthic macro-invertebrate community composition within a mangrove/seagrass estuary in northern New Zealand. *Estuarine, Coastal and Shelf Science* 66(1-2), 97-110.

Alongi, D. M., Pfitzner, J., Trott, L. A., Tirendi, F., Dixon, P. and Klumpp, D. W., 2005. Rapid sediment accumulation and microbial mineralization in forests of the mangrove Kandelia candel in the Jiulongjiang Estuary, China. *Estuarine, Coastal and Shelf Science* 63(4), 605-618.

Beard, C. 2006. Physiological constraints on the latitudinal distribution of the mangrove *Avicennia marina* (Forsk.) Vierh. subsp. *australasica* (Walp.) J. Everett in New Zealand. PhD thesis, University of Waikato, 203 p.

Bird, E.C.F. 1971. Mangroves as land builders. *Victorian Naturalist* 88, 189–197.

Boumans, R., Day Jr., J.W., 1993. High precision measurements of sediment elevation in shallow coastal areas using a sedimentation– erosion table. *Estuaries* 16, 375–380.

Burns, B. R. and Ogden, J., 1985. The demography of the temperate mangrove [*Avicennia marina* (Forsk.) Vierh.] at its southern limit in New Zealand. *Australian Journal of Botany* 10, 125-133.

Butman, C.A., 1986. Sediment trap biases in turbulent flows: results from a laboratory flume study. *Journal of Marine Research* 44, 645–693

Cahoon, D. R., Reed, D. J. and Day, J. W., 1995. Estimating shallow subsidence in microtidal salt marshes of the southeastern United States: Kaye and Barghoorn revisited. *Marine Geology* 128(1-2), 1-9.

Cahoon, D. R. and Lynch, J. C., 1997. Vertical accretion and shallow subsidence in a mangrove forest of southwestern Florida, U.S.A.. *Mangroves and Salt Marshes* 1, 173-186.

Cahoon, D.R., Marin, P.E., Black, B.K., and Lynch, J.C., 2000. A method for measuring vertical accretion, elevation, and compaction of soft, shallow-water sediments. *Journal of Sedimentary Research* 70, 1250-1253.

Cahoon, D. R., Lynch, J. C. , Hensel, P., Boumans, R., Perez, B. C., Segura, B. and Day, J. W. Jr., 2002a. A device for high precision measurement of wetland sediment elevation: I. Recent improvements to the sedimentation-erosion table. *Journal of Sedimentary Research.*72, No. 5., 730-733.

Cahoon, D. R., Lynch, J.C., Perez, B.C.; Segura, B., Holland, R.D., Stelly, C., Stephenson, G., Hensel, P., 2002b. High-Precision Measurements of Wetland Sediment Elevation: II. The Rod Surface Elevation Table. *Journal of Sedimentary Research* 72(5), 734-739.

Cahoon, D. R., P. Hensel, J. Rybczyk, K. L. McKee, C. E. Proffitt and B. C. Perez., 2003. Mass tree mortality leads to mangrove peat collapse at Bay Islands, Honduras after Hurricane Mitch. *Journal of Ecology* 91(6), 1093-1105.

Davies-Colley, R.J., and Healy, T.R., 1978. Sediment and hydrodynamics of the Tauranga Entrance to Tauranga Harbour. *New Zealand Journal of Marine and Freshwater Research* 12, 225-236.

Davis, R.A., and Healy, T.R., 1993. Holocene coastal depositional sequences on a tectonically active setting: southeastern Tauranga Harbour, New Zealand. *Sedimentary Geology* 84, 57-69.

de Lange, W., and Healy, T.R., 1990. Renourishment of a flood-tidal delta adjacent beach, Tauranga Harbour, New Zealand. *Journal of Coastal Research* 6, 627-640.

do Amaral, P.G.C., Ledru, M.-P., Branco, F.R., and Giannini, P.C.F., 2006. Late Holocene development of a mangrove ecosystem in southeastern Brazil (Itanhaem, State of Sao Paulo). *Palaeogeography, Palaeoclimatology, Palaeoecology* 241, 608-620.

Ellis, J., Nicholls, P., Craggs, R., Hofstra, D. and Hewitt, J., 2004. Effects of terrigenous sedimentation on mangrove physiology and associated macrobenthic communities. *Marine Ecology Progress Series* 270, 71-82.

Fan, D. D., Li, C. X., Wang, D. J., Wang, P., Archer, A. W., Greb, S. F., 2004. Morphology and sedimentation on open-coast intertidal flats of the Changjiang Delta, China. *Journal of Coastal Research* 43, 23-35.

French, J. R. and Spencer, T., 1993. Dynamics of sedimentation in a tide-dominated backbarrier salt marsh, Norfolk, UK. *Marine Geology* 110, 315-331.

Furukawa, K. and Wolanski, E., 1996. Sedimentation in mangrove forests. *Mangroves and Salt Marshes* 1(1), 3-10.

Furukawa, K., Wolanski, E. and Mueller, H., 1997. Currents and sediment transport in mangrove forests. *Estuarine Coastal and Shelf Science* 44(3), 301-310.

Gust, G. and Kozerski, H. P., 2000. In situ sinking-particle flux from collection rates of cylindrical traps. *Marine Ecology Progress Series* 208, 93-106.

Hayward, B. W., Grenfell, H. R., Sabaa, A. T., Morley, M. S. and Horrocks, M., 2006. Effect and timing of increased freshwater runoff into sheltered harbor environments around Auckland City, New Zealand. *Estuaries and Coasts* 29(2), 165-182.

Healy, T. R., Cole, R. G. and De Lange, W., 1996. Geomorphology and ecology of New Zealand shallow estuaries and shorelines. In: Nordstrom, K. F. and Roman, C. T. (Eds.), Estuarine Shores: Evolution, Environments and Human Alteration. John Wiley & Sons, Chichester, pp. 115-154.

Hope, H. M., 2002. Sediment dynamics of Waikaraka Estuary, a small, semi-enclosed estuarine system in the upper reaches of a developed harbour. MSc. thesis, University of Waikato, 126 p.

Hume, T., Fox, M. E. and Wilcock, R. J. 1989. Sedimentation rates as indicated by organochlorine contaminants in the Manukau Harbour, New Zealand. *Journal of the Royal Society of New Zealand* 19, 305-317.

Kozerski, H. P., 1994. Possibilities and limitations of sediment traps to measure sedimentation and resuspension. *Hydrobiologia* 284(1), 93-100.

Krauss, K. W., Allen, J. A. and Cahoon, D. R., 2003. Differential rates of vertical accretion and elevation change among aerial root types in Micronesian mangrove forests. *Estuarine, Coastal and Shelf Science* 56(2), 251-259.

Larcombe, P., Woolfe, K.J., 1999. Increased sediment supply to the Great Barrier Reef will not increase accumulation at most coral reefs. *Coral Reefs* 18, 163–169.

Lelieveld, S.D., Pilditch, C.A., and Green, M.O., 2004. Effects of deposit-feeding bivalve (*Macoma liliana)* density on intertidal sediment stability. *New Zealand Journal of Marine and Freshwater Research* 38, 115-128.

Lovelock, C. E., Feller, I.C., Ellis, J., Schwarz, A.N.., 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.

Lund-Hansen, L.C., Valeur, J., Pejrup, M., Jensen, A., 1997. Sediment fluxes, re-suspension and accumulation rates at two windexposed coastal sites and in a sheltered bay. *Estuarine, Coastal and Shelf Science*. 44, 521– 531.

Lynch, J.C., Meriwether, J.R., McKee, B.A., Vera-Herrera, F., and Twilley, R.R., 1989. Recent accretion in mangrove ecosystems based on ^{137}Cs and ^{210}Pb. *Estuaries* 12 284 -299.

Massel, S.R., Furukawa, K., and Brinkman, R.M., 1999. Surface wave propagation in mangrove forests. *Fluid Dynamics Research* 24, 219-249.

McKee, B.A., Nittrouer, C.A., DeMaster, D.J., 1983. Concepts of sediment deposition and accumulation applied to the continental shelf near the mouth of the Yangtze River. *Geology* 11, 631– 633.

McKee, K. L., Cahoon, D. R. and Feller, I. C. 2007. Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. *Global Ecology and Biogeography* 16(5), 545-556.

Mom, G. A. T. 2005. Effects of Management of the Ecology of Mangrove Communities in New Zealand. PhD thesis, Auckland University, 366 p.

Morrisey, D. S., GA; Ellis, JI; Burns, BR; Kemp, CE; 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, 581-592.

Nicholls, R.J., Wong, P.P., Burkett, V.R., Codignotto, J.O., Hay, J.E., McLean, R.F., Ragoonaden, S. and Woodroffe, C.D., 2007. Coastal systems and low-lying areas. Climate Change 2007, In: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Parry, M.L. Canziani, O.F., Palutikof, J.P., van der Linden P.J. and Hanson, C.E. (Eds) Cambridge University Press, Cambridge, UK, 315-356.

Pasternack, G. B. and Brush, G.S., 2001. Seasonal variations in sedimentation and organic content in five plant associations on a Chesapeake Bay tidal freshwater delta. *Estuarine Coastal and Shelf Science* 53, 93-106.

Quartel, S., Kroon, A., Augustinus, P. G. E. F., Van Santen, P. and Tri, N. H., 2007. Wave attenuation in coastal mangroves in the Red River Delta, Vietnam. *Journal of Asian Earth Sciences* 29(4), 576.

Quayle, A.M., 1984. *The climate and weather of the Bay of Plenty region*. Wellington: New Zealand, Ministry of Transport, New Zealand Meteorological Service, 56 p.

Reed, D., 1989. Patterns of sediment deposition in subsiding coastal salt marshes, Terrebonne Bay, Louisiana: the role of winter storms. *Estuaries* 12(4), 222-227.

Rogers, K., Saintilan, N. and Heijnis, H. 2005. Mangrove Encroachment of Salt Marsh in Western Port Bay, Victoria: the Role of Sedimentation, Subsidence and Sea Level Rise. *Estuaries* 28(4), 551-559.

Rogers, K., Wilton, K. M. and Saintilan, N., 2006. Vegetation change and surface elevation dynamics in estuarine wetlands of southeast Australia. *Estuarine, Coastal and Shelf Science,* 66(3-4), 559-569.

Saad, S., Husain, M. L., Yaacob, R. and Asano, T., 1999. Sediment accretion and variability of sedimentological characteristics of a tropical estuarine mangrove: Kenaman, Terengganu, Malaysia. *Mangroves and Salt Marshes* 3, 51-58.

Saintilan, N. and Williams, R., 2000. Short Note: the decline of saltmarsh in southeast Australia: results of recent surveys. *Wetlands (Australia)* 18(2), 49-54.

Schwarz, A., 2002. *The role of nutrients in contributing to mangrove expansion*. National Institute of Water & Atmospheric Research Ltd, Hamilton, New Zealand 24 p.

Sheffield, A.T., Healy, T.R., and McGlone, M.S., 1995. Infilling rates of a steepland catchment estuary, Whangamata, New Zealand. *Journal of Coastal Research* 11,1294-1308.

Spenceley, A.P., 1977. The role of pneumatophores in sedimentation processes. *Marine Geologoy.* 24, M31–M37.

Stokes, D.S., Healy, T.R. and Cooke, P. J. 2009. Surface elevation changes and sediment characteristics of intertidal surfaces undergoing mangrove expansion and mangrove removal, Waikaraka Estuary, Tauranga Harbour, New Zealand. *International Journal of Ecology and Development* 12, W09, 75-87.

Surman, M., 1999. *Tauranga Harbour Sediment Source Survey. Environment Bay of Plenty Report 98/13*, Whakatane: New Zealand, 141 p.

Swales, A., Bentley, S. J., Lovelock, C. E. and Bell, R., 2007. Sediment processes and mangrove-habitat expansion on a rapidly-prograding muddy coast, New Zealand. *Coastal Sediments '07,* New Orleans, Louisiana, American Society of Civil Engineers, pp. 1445-1454.

Swales, A., Hume, T. and Green, M., 1997. Holocene sedimentation and recent human impacts on infilling in a drowned valley estuary. *13th Australasian Coastal and Ocean Engineering Conference & 6th Australasian Port and Harbour Conference Proceedings* (Christchurch, New Zealand), pp. 895-900.

Swales, A., Williamson, R. B., Van Dam, L. F., Stroud, M. J. and McGlone, M. S., 2002. Reconstruction of urban stormwater contamination of an estuary using catchment history and sediment profile dating. *Estuaries* 25(1), 43-56.

Thomas, S. and Ridd, P. V. 2004. Review of methods to measure short time scale sediment accumulation. *Marine Geology* 207, 95-114.

Thrush, S., Hewitt, J., Cummings, V., Ellis, J., Hatton, C., Lohrer, A. M. and 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. F., Hewitt, J. E., Norkko, A., Nicholls, P. E., Funnell, G. A. and Ellis, J. I., 2003. Habitat change in estuaries: predicting broad-scale responses of intertidal macrofauna to sediment mud content. *Marine Ecology Progress Series* 263, 101-112.

Van Santen, P., Augustinus, P. G. E. F., Janssen-Stelder, B. M., Quartel, S. and Tri, N. H., 2006. Sedimentation in an estuarine mangrove system. *Journal of Asian Earth Sciences* 29(4), 566-575.

Victor, S., Golbuu, Y., Wolanski, E. and Richmond, R. H., 2004. Fine sediment trapping in two mangrove-fringed estuaries exposed to contrasting land-use intensity, Pulau, Micronesia. *Wetlands Ecology and Management* 12, 277-283.

Whelan, K.R.T., Smith, T. J., Cahoon, D. R., Lynch, J. C. and Anderson, G.H., 2005. Groundwater control of mangrove surface elevation: Shrink and swell varies with soil depth. *Estuaries* 28(6), 833-843.

White, J., 1979. Recent sediments of Waikareo Estuary. MSc. thesis, University of Waikato, 197 p.

Wolanski, E., Fabricius, K., Spagnol, S. and Brinkman, R., 2005. Fine sediment budget on an inner-shelf coral-fringed island, Great Barrier Reef of Australia. *Estuarine, Coastal and Shelf Science* 65(1-2), 153-158.

Woodroffe, C. ,1983. Development of mangrove forests from a geological perspective. In: Teas, H.J. (Ed), Tasks for vegetation science. The Hague, Dr W Junk Publishers. 8: pp. 1-17.

Woodroffe, C., 1992. Mangrove sediments and geomorphology, *in:* Robertson, A. I. and Alongi D. M. (Eds.), *Tropical Mangrove Ecosystems*. Washington, D.C., American Geophysical Union, pp. 7-41.

Young, B. M. and Harvey, E. L., 1996. A spatial analysis of the relationship between mangrove (*Avicennia marina* var. *australasica*) physiognomy and sediment accretion in the Hauraki Plains, New Zealand. *Estuarine, Coastal and Shelf Science* 42(2), 231-246.

Chapter 4
Changing sedimentary environments: the influence of mangrove expansion and mangrove removal on estuarine geomorphology

4.1 PREFACE

Waikaraka Estuary is a small and narrow sub-estuary of Tauranga Harbour. In recent years a number of residents local to the area expressed a growing concern at the increasing dominance of mangroves in the upper estuary where previously sandy intertidal flats were accessed by walkers, kayakers and swimmers. The 'Waikaraka Catchment Managers' group was formed, and in 2004 the group was granted resource consent to remove any mangrove vegetation that had established post-1984. Representatives of the territorial authority used GPS co-ordinates acquired from photogrammetry to demarcate the boundary between areas to be cleared and areas to be maintained. The physical removal of above-ground vegetation and propagules has since been undertaken in stages by community volunteers. The process of removal involves cutting the plants at the sediment surface and placing them into piles on the sandflats. Once the detritus has dried it is incinerated on-site.

This chapter describes the physical changes to the harbour bed that have occurred in response to the removal of above-ground mangrove vegetation. The present-day estuarine environment of Waikaraka Estuary is described using a combination of plant morphometrics, surface topography and sedimentology. Sedimentary environments underlying the contemporary surface sediments are also described, and historical sedimentation rates are discussed.

The key findings within this chapter were published in the International Journal of Ecology and Development:

> Stokes, D.J., Healy, T.R. and Cooke, P.J., 2009. Surface elevation changes and sediment characteristics of intertidal surfaces undergoing mangrove expansion and mangrove removal, Waikaraka Estuary, Tauranga Harbour, New Zealand. *International Journal of Ecology and Development*, 12 (W09) 88-106.

D.J. Stokes was responsible for all field work, lab work, and the formation of the journal article cited above. Co-authors were added to the journal article in acknowledgement of their contribution to fieldwork (Cooke, P.J.) and editorial advice during preparation of the manuscript (Healy, T.R.).

4.2 INTRODUCTION

Sedimentation on intertidal flats has been studied extensively (Fan et al., 2004 and references therein) with a number of studies investigating the sedimentation and trapping mechanisms of mangrove vegetation (see for example Spencely, 1977; Cahoon and Lynch, 1997; Furakawa and Wolanski, 1996; Krauss et al., 2003; Rogers et al., 2006). In contrast, the remobilisation and transport of sediment across intertidal areas as a result of mangrove removal has rarely been reported.

Although active sedimentation is a typical condition of most estuarine environments (Nichols & Biggs, 1985; Healy et al., 1996), studies of sedimentation rates in New Zealand estuaries report increased rates of infilling since European settlement (Hume and McGlone, 1986; Hume and Herdendorf, 1992; Sheffield et al., 1995; Swales et al., 1997; Ellis et al., 2004). This has been attributed to land use changes, particularly where native forest has been removed for agriculture, forestry or urbanization (Hume and McGlone, 1986; Healy et al., 1996; Hayward et al., 2006). Rapid sedimentation will not only influence the geomorphology of an estuary, but can negatively impact on estuarine ecology through smothering benthic fauna and muddying water which can result in lower productivity of benthic and pelagic organisms (Thrush et al., 2003; Thrush et al., 2004).

A number of studies have reported sediment accumulation within mangrove vegetation, both overseas (Cahoon and Lynch, 1997; Alongi et al., 2005; Van Santen et al., 2006; Victor et al., 2006; Wolanski et al., 2006) and in New Zealand (Young and Harvey, 1996; Ellis et al., 2004). The vegetation density increases friction, resulting in a reduction of water flow velocities (Massel et al., 1999); and the above-ground root structures act to create micro-turbulence capable of maintaining sediment in suspension during flood tides, which then settles during periods of slack water (Furukawa and Wolanski, 1996). The mangrove root zone also acts to bind sediment once it has settled, as noted in Woodroffe (1992).

Spatial gradients are often highlighted in studies of mangrove sedimentation, with higher rates of accretion recorded in the mangrove fringe (Furukawa and Wolanski, 1996; Cahoon and Lynch, 1997; Saad et al., 1999; Alongi et al., 2005; Rogers et al., 2006). Other factors such as sediment supply (Woodroffe, 1992), tidal range (Rogers et al., 2006) and forest root structures (Young and Harvey, 1996; Cahoon and Lynch, 1997; Krauss et al., 2003) have also been found to influence sedimentation rates in mangrove vegetation.

An increase in mangrove coverage over recent decades has been documented in many harbours and embayments in the upper North Island of New Zealand (Burns and Ogden, 1985; de Lange and de Lange, 1994; Young and Harvey, 1996; Ellis et al., 2004; Swales et al., 2007). It has been suggested that the increase in mangrove coverage is a response to estuarine infilling, and may also be linked to periods of calm weather and increased nutrient inputs associated with human land-use (Swales et al., 2007). Waikaraka Estuary is one of a number of embayments within Tauranga Harbour where the monospecific stands of *Avicenna marina* subsp. *australasica* are expanding their range. The catchments surrounding Tauranga Harbour have been converted to horticultural and agricultural land, with an urban fringe closer to the harbour margins.

Rates of contemporary sedimentation within Tauranga Harbour are largely unknown, however it is likely to follow the trend of increasing sedimentation reported for other New Zealand estuaries (e.g. Sheffield et al., 1995). It is imperitive that we develop an understanding of historical and contemporary sedimentary environments of these estuaries and embayments. A reconstruction of the sedimentary history will provide us with an understanding of the scale of recent geomorphological change, and allow for more reliable predictions of future change in response to climate change (i.e. increased storm events and rainfall bringing greater sediment loads) and sea level rise.

4.2.1 Dating methods

Radionuclide analysis, radiocarbon dating and pollen analysis are all tools used to reconstruct estuarine sedimentation history in New Zealand (e.g. Hume and McGlone, 1986; Swales et al., 1997; Swales et al., 2007). The most powerful analysis comes from the use of a combination of these techniques, although cost is often prohibitive of such a comprehensive methodology. Radiocarbon and pollen techniques allow inquiry into long term sedimentation rates, whereas radionuclides are most often utilised in studies that wish to interpret impacts of human occupation and associated land-use changes up to 30 years (^{137}Cs) and 150 years ago (^{210}Pb).

Radionuclide analysis

The choice of radionuclide analysis is often determined by the timeframe over which a researcher expects to be measuring. For example, ^{234}Th has a half life of 24 days, and ^{7}Be of 53.3 days and so are useful for measuring sediment accumulation for short time scales (Thomas and Ridd, 2004). Radioisotopes such as Lead-210 (^{210}Pb) and Cesium-137 (^{137}Cs - $t_{1/2}$ 30) are more commonly used to reconstruct the sedimentation history of an estuary associated with European settlement (Sheffield, et al., 1995; Swales et al., 2002b).

Cesium-137 was introduced into the atmosphere as a by-product of nuclear testing that commenced in the 1950s and early 1960s (Lynch et al., 1989). Peaks in annual Cesium deposition in New Zealand correspond to nuclear tests undertaken in 1953, 1955-1956 and 1963-1964 and the maximum depth of cesium detected in estuarine sediment cores in New Zealand, therefore, is taken to correspond with the year 1953 (Swales et al., 2002b).

Sediment accumulation rates (SAR) can also be estimated using analysis of Lead-210 (^{210}Pb) in the sediment profile. Lead-210 is a naturally occurring radioisotope with a half life of 22 years (Sheffield et al., 1995). There are two key components to the analysis of Lead-210. Firstly, there is the “supported” ^{210}Pb activity which is derived from the decay of the parent gas ^{226}Ra (Radon). Additional to the ‘supported’ activity is the contribution of atmospheric lead which is the by-product of ^{222}Rn decay once it has escaped the earth’s crust (summarised in Lynch et al., 1989). Based on the assumption that the supported ^{210}Pb and ^{226}Ra are in equilibrium, the excess or unsupported ^{210}Pb is calculated after which a rate of sediment accumulation can be modeled (Swales et al., 2002a).

Radiocarbon dating

Radiocarbon dating can be used to determine estimates of long-term sedimentation. The technique is used to date fossil carbon from organisms or plant material which has been buried and preserved. Accelerated Mass Spectrometry (AMS) dating is used where only small samples or fragments can be collected, and where the error margin needs to be tighter than that available from conventional techniques (Ellison, 2008).

Pollen analysis

Pollen analysis has been used alongside radionuclide techniques to strengthen the interpretation of sedimentation rates in New Zealand estuaries (Hume and McGlone 1986; Sheffield et al., 1995; Swales et al., 2002a). Pollen and spores preserved in estuarine sediments hint at the vegetation that was prevalent in the surrounding catchment. Major changes to the vegetation types and dominance, such as those initiated by deforestation for human habitation and the onset of plantation forestry or pastoral agriculture, can be used to interpret sedimentation rates in vertical cores (e.g. Hume and McGlone, 1986; Sheffield et al., 1995; Swales et al., 2002a).

4.3 STUDY OBJECTIVES

There is little information available pertaining to sedimentation rates in mangrove, and an even larger knowledge gap concerning the effects of mangrove removal (Granek and Frasier, 2007). Accordingly, the aim of this study was to document the dynamics of mangrove expansion at Waikaraka Estuary and investigate sedimentation rates in the presence of mangroves, and topographical and sedimentological changes measured after mangrove removal. Description of plant height, plant density and pneumatophore density is incorporated to increase our understanding of the mangrove stand dynamics where plants are growing near their southern climatic limits.

This chapter addresses the following thesis objectives:

1) Describe the morphometric characteristics of mangrove forests within Waikaraka Estuary;

(2) Identify the physical changes that have occurred due to mangrove expansion; and

(3) Investigate the physical impacts of mangrove removal.

4.4 SETTING

Tauranga Harbour is situated within the Bay of Plenty region, on the east coast of the North Island of New Zealand (Lat. 37° 40’S, Long. 176° 03’E, Figure 4.1). It is a large barrier-enclosed estuarine lagoon (over 200 km^2) with extensive sandy tidal flats exposed at low tide (Healy et al., 1996). On the landward side of the estuarine lagoon a number of re-entrant bays drain local catchments. The Waikaraka Estuary is bound by a small catchment of just under 10 km^2, and the estuary area itself, including mangroves, is 0.5 km^2. The surrounding catchment incises ignimbrite geology underlying some Holocene and Late Pleistocene alluvium and tephras closer to the harbour margins (Harmsworth, 1983; Briggs et al., 1996). All native forest has been removed from the Waikaraka catchment, which is now dominated by kiwifruit and citrus orchards.

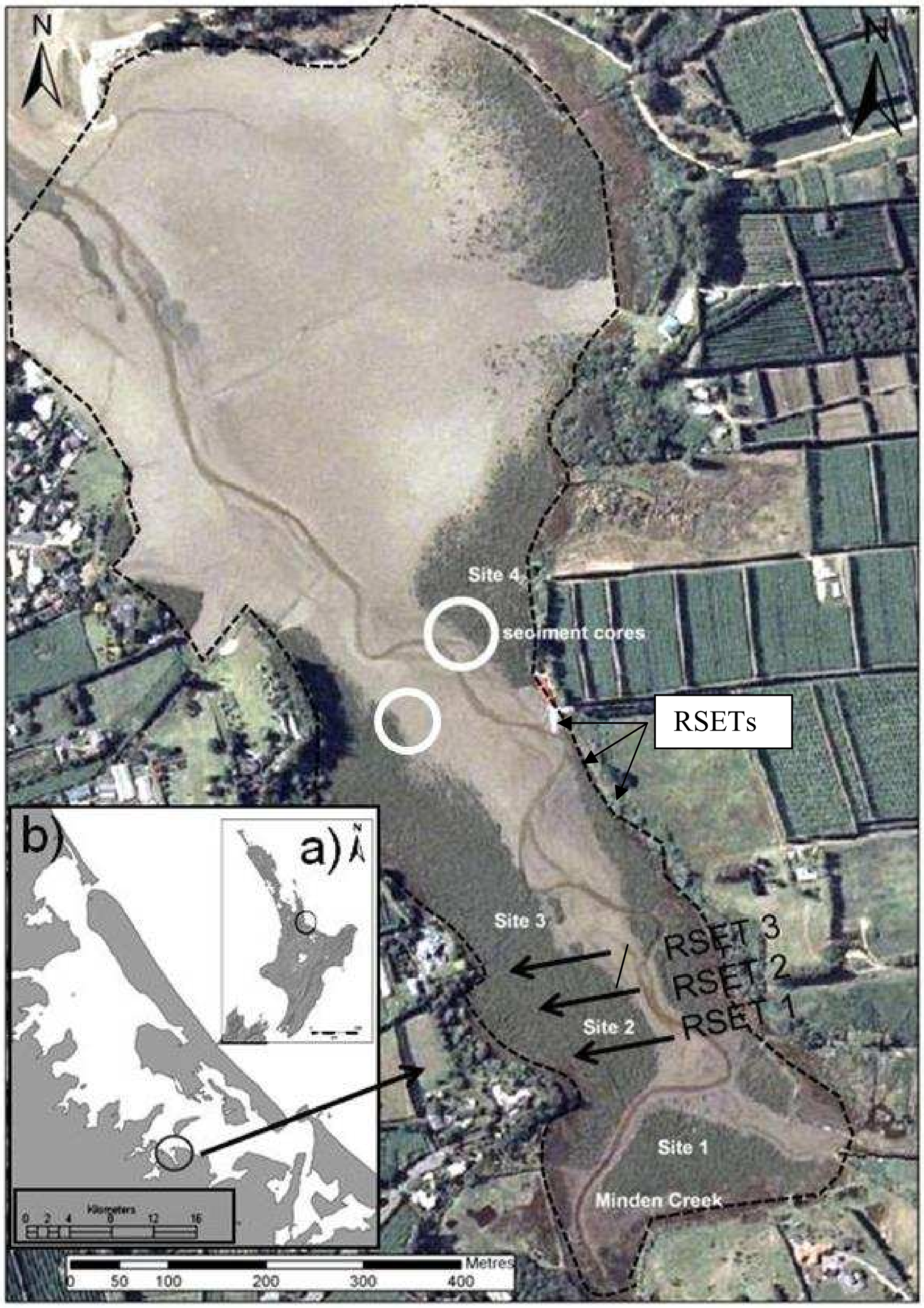


Figure 4.1 **Waikaraka Estuary, a narrow estuary positioned along the western margins of Tauranga Harbour (b). Sample collection sites and RSET locations are labeled. 'Estuary area' represented by hatched line, outlined for determination of mangrove coverage as % of estuary area. Aerial photograph (2003) courtesy of Environment Bay of Plenty.**

Freshwater discharge into Waikaraka Estuary is considerably smaller than neighbouring embayments. The main tributary, Minden Creek, contributes a mean annual flow of 92 l^{s-1}, compared to the neighbouring Te Puna estuary which receives 792 l^{s-1} (Hope, 2002). Tides at the entrance of Waikaraka estuary have been measured as meso-tidal, ranging from 2.1 m at spring tides, to 1.4 m during neap tides, with the tidal range decreasing to 0.6 - 0.7 m in the upper estuary (Hope, 2002). Mangrove stands in the middle and upper estuary are inundated only during the final stage of high tide and the mangrove and cleared plots closer to the estuary mouth (Site 4, see Figure 4.1) are covered 30 to 45 minutes earlier.

4.5 METHODS

4.5.1 Mangrove Physiognomy

Plant height, stem girth at 0.05 m above stratum, and pneumatophore density were measured at four sites along the estuary (Figure 4.1). At each of these sites, three 4 x 4 m plots were randomly selected, marked out, and all trees measured for the above-mentioned parameters. Pneumatophores were counted in three separate, randomly selected 1 m^2 quadrats within each plot. Mean values reported in Table 4.2 represent results of the three plots combined for each site.

4.5.2 Sediment characteristics

In July 2006 (southern-hemisphere winter) triplicate sediment samples were collected along transects at Sites 2, 3 and 4. In February 2007 (summer), sites were resampled, with the inclusion of Site 1, to provide baseline grain size data in light of potential mangrove removal in the future. Two sampling stations were located inside mangrove habitat, and three stations on the bare flats (Figure 4.3f). Samples were also collected at three locations within cleared plots 1, 2 and 3, in May 2005, and again in summer 2006 and 2007.

Sediment samples were treated with 10 % hydrogen peroxide (H_2O_2) to remove organic material. Calgon was then added for deflocculation, and samples analysed for grain size distribution using the Malvern Mastersizer S. Version 2.19.

Three sediment cores, 1.5 m – 3 m in length, were collected in 70 mm diameter aluminium tubes using a vibracorer (Figure 4.3). Cores were returned to the lab for stratigraphic logging, and sub-samples were removed for grain size analysis and color notations, using Maunsell color charts.

It was only possible to collect cores in proximity to the main access point, which is mid-way along the estuary, roughly 25 m south of Site 4 (see Figure 4.1 for site location). Core (a) represents the sediment profile beneath a recently cleared mangrove zone; Core (b) adjacent mudflats within 25 m of the cleared mangrove zone, and Core (c) was collected toward the middle of the intertidal flats, approximately 15 m east of the main tidal channel. A short core (35 cm deep) was collected within the mangroves in the vicinity of Site 2, in the middle (longitudinally) of the mangrove zone.

An additional series of cores were collected in 2007, 100 m west of Site 4 on intertidal flats and within mangroves on the adjacent side of the tidal channel (see Figure 4.1). Of three long cores, one yielded a contiguous 160 cm sediment profile of the unvegetated tidal flats. From this core, a cockle (*Austrovenus stuchburyi*) shell was collected at 155 cm depth and submitted to the Waikato Radiocarbon Dating Laboratory where Accelerated Mass Spectrometry (AMS) was used to calculate a ^{14}C date. Wood samples collected at various depths were sent to SCION Laboratory (New Zealand) for identification, however due to their advanced state of decomposition, only very broad taxonomic classifications could be provided.

Sediment texture and colour, and broad descriptions of mineralogy (under petrological microscope) were also described.

A smaller core (7 cm diameter x 70 cm) collected in the mangroves opposite Site 4 was analysed for ^{210}Pb to provide a sedimentation history. On return to the laboratory, the core was split and sub-sampled at 1 cm intervals then oven-dried to a constant weight. Approximately 10 grams of each sub-sample were retained for grain size analysis using a Malvern Mastersizer S Version 2.19, after 48 hours in 10 % hydrogen peroxide (H_2O_2) and 24 hours in Calgon. Selected samples were then analysed at the State Key Laboratory of Nanjing University for ^{210}Pb. Sediment Accumulation Rates (SAR) were determined from regression analysis of log-transformed data (Swales et al., 2002).

4.5.3 Surface elevation changes from erosion pins and RSET

Surface elevation changes on the mudflat surface were measured with a series of Rod Surface Elevation Tables (RSET), as described in Cahoon et al. (2002) (Figure 4.3). Benchmark poles were driven 3 m into the substrate with around 50 cm protruding from the estuary floor, then further stabilised with cement. A detachable arm with nine measuring pins attaches to the benchmark pole via a rod-collar coupling device, and for this study was rotated 180°, giving a total of 18 readings per RSET, which were then averaged after each visit to give a single value of surface elevation. Confidence intervals for the measured height of an individual pin were measured at {+/-} 1.3 mm in a mangrove forest (Cahoon et al., 2002). Each RSET benchmark was manually surveyed one month after installation, and again 14 months later to ensure the poles had maintained their original position.

Three transects of four RSETs were positioned in the upper estuary in the vicinity of Sites 2 and 3 (see Figure 4.1 for site location and Figure 4.3f for transect lay-out). RSETs are a permanent fixture in the environment and because of the potential for injury or interference, only three transects were installed. The intertidal RSETs along Transect 1 were positioned in Cleared Plot 3 (10 m from mangrove fringe) and Cleared Plot 1 (20 m from mangrove fringe) while RSET Transects 2 and 3 were positioned within mangroves and on bare tidal flat to assess variation in surface elevation changes in the absence of mangrove removal.

Stainless steel erosion pins were installed at 15 locations within the cleared areas as well as the mangrove zones at Sites 1 and 4, the locations of which are displayed in Figure 4.7. Erosion pins (0.7 m long, 5 mm diameter) were deployed in clusters of seven pins (Figure 4.3) and driven into the substrate with 0.2 m remaining above the sediment surface. The height above substrate of the seven pins was averaged to provide a single measurement of elevation change. Erosion pins have been used in other mangrove environments (e.g. Spenceley, 1977), and although the accuracy has not been specified in published surveys, it can be estimated to the nearest millimetre (Thomas and Ridd, 2004).

Site 4 was partially cleared of mangroves in mid-March 2006, roughly one year after sections in the vicinity of Site 2 and 3. Cleared Plot 1 (CP1) was cleared on 21 May 2005; CP2 on 13 August 2005 and CP3 on 30 August 2005 (Figure 4.2).

In this study, recorded measurements from RSETs and erosion pins are referred to as 'surface elevation change'. These devices measure the rise or fall in the substrate, therefore any sediment compaction, shallow subsidence, root decomposition, or root growth are incorporated in the result of elevation change (Cahoon et al., 2000). It is important to note that the RSETs have a base datum 3 m below the surface, which is much deeper than the pins at 0.5 m, although the datum of both instruments is positioned at depths below the mangrove root zone. Both techniques will therefore measure the processes of root development and root decomposition that are likely to influence surface accretion at these sites. Any sediment compaction between 0.5 m and 3 m will be picked up by the RSET but excluded by the pins.

Most studies that use RSETs also put down marker horizons as a method of differentiating between rates of surface elevation change and depths of sediment accretion. The use of marker horizons was discussed more fully in Chapter 3. To this end, marker horizons of glass beads were emplaced on the surface at locations in proximity to each RSET. Unfortunately the success of the marker horizons in this study was limited. This was due to a) removal of some of the posts that marked each location (therefore making it difficult to find); b) difficulty in extracting small cores; and c) some trampling of the surface. Because of the limited time-frame of the study, rather than re-establish a second series of marker horizons, sediment traps were deployed as an alternative. The difference between the two methods, however, is that traps measure the weight of sediments collected over an area rather than a vertical depth of sediment accumulation.

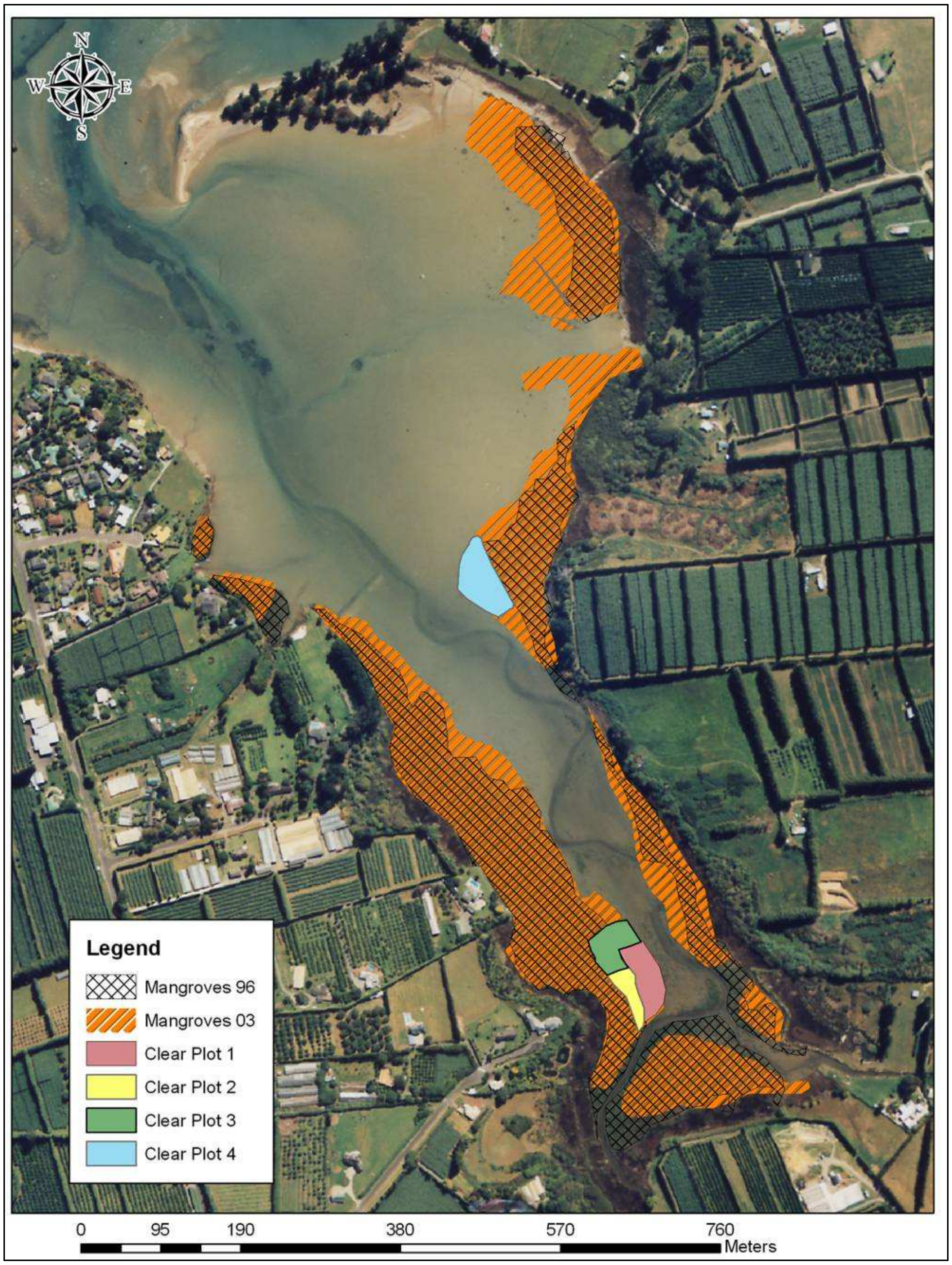


Figure 4.2 **Aerial vertical image of Waikaraka Estuary, 2003. Mangroves have expanded to cover approximately 115,000 m^2. Four plots have been cleared of mangrove vegetation since April 2005, totaling 9,600 m^2.**

4.5.4 Sediment traps

Sediment accumulation, or gross sediment deposition, was measured using cylindrical PVC sediment traps with an internal diameter of 70 mm and an aspect ratio of 1:8 (Figure 4.3). Sediment traps have been used in mangrove environments to investigate temporal and spatial variability in sediment loads that move across the intertidal surface (Ellis et al., 2004; Wolanski et al., 2005; Victor et al., 2006). They provide an inexpensive and robust method to gain some understanding of not only the load of sediment but also the characteristics of the sediment that is settling out of the water column.

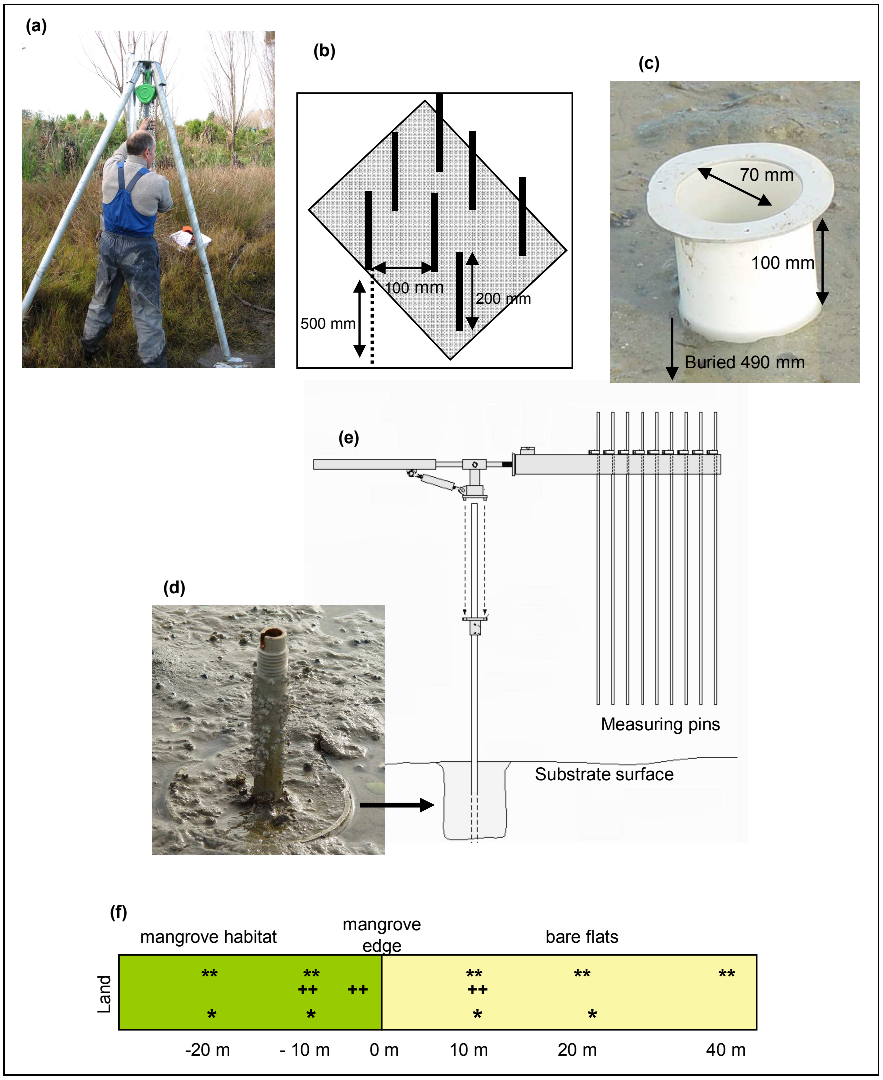


Figure 4.3 **Images of instruments used in this field study. (a) Tripod component of the motorized vibracorer used to collect sediment cores; (b) schematic diagram of erosion pin cluster, (c) sediment traps installed on bare intertidal flats and within mangrove zones; (d) the permanent benchmark of the RSET device; (e) conceptual diagram of the portable RSET arm with adjustable measuring pins (from Cahoon et al., 2002); (f) spatial lay-out of transects for * RSET positions, ** collection of surface sediments, ++ sediment traps.**

The limitations of this technique however lie in their inability to reflect the on-going natural process of periodic settlement and resuspension that is likely to occur before longer term deposition occurs, and therefore are likely to over-estimate sediment flux (Kozerski, 1994).

Traps were deployed for approximately one month in May and June 2006 (winter) and January and February 2007 (summer). Transects of sediment traps were installed at the four monitoring sites, with two traps inside the mangroves and one on the bare flats (Figure 4.3f). Sediment accumulation rates of dry sediment are expressed in g m^2 mo^{-1}. A combination of tampering, mishandling and growth of filamentous algae over traps, has reduced the final analyses however.

4.6 RESULTS

4.6.1 Mangrove Expansion

Temporal change of the planimetric distribution of mangrove vegetation in the estuary has been mapped using aerial photographs dated 1943, 1982, 1996 and 2003. Mangrove coverage in 1943 was approximately 16,000 m^2. In 1982 mangroves had colonised seaward, increasing the area of mangrove vegetation to 29,000 m^2 and by 1996 mangroves had expanded to cover approximately 100,000 m^2, including the previously bare sandier areas south-east of the estuary mouth, in the vicinity of Site 4 (Figure 4.4). Between 1996 and 2003 further colonisation increased mangrove coverage to 115,000 m^2 (Table 4.1).

Table 4.1 Area (m^2) and percent coverage of total estuarine area of mangrove coverage measured from aerial photographs dated 1943, 1982, 1996 and 2003.

Year	Area of estuary covered by mangroves (m^2)	% of estuary covered by mangroves
1943	16,000	3
1982	29,000	6
1996	100,000	20
2003	115,000	23

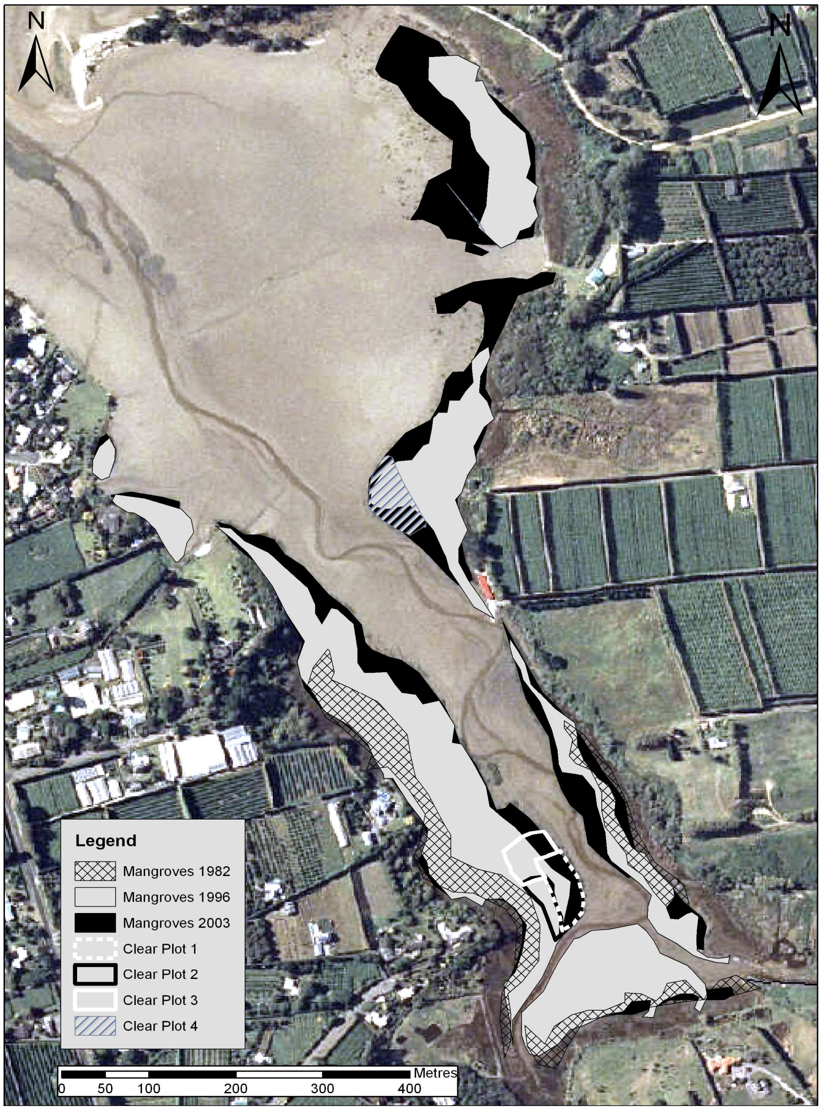


Figure 4.4 **2003 aerial photo showing mangrove coverage in Waikaraka Estuary in 1982, 1996 and 2003. Areas cleared of mangroves during the study period are also outlined.**

4.6.2 Mangrove Physiognomy

Average plant heights, measured within each 16 m^2 plot, range from 0.68 m (± 0.11 m) to 1.21 m (± 0.18 m). Standard error around mean plant height was sufficiently low that the three plots at each site were grouped together for further analysis. Mean plant height appears to have no correlation with the age of the mangrove stands studied, with the youngest (Site 4) and oldest (Site 1) stands displaying similar mean plant heights of 1.03 m and 1.04 m respectively (Table 4.2). Stem density is highest and stem diameter lowest at Site 3, where mean plant height is lower than all other stands (0.76 m). Average pneumatophore density at Site 4, where shrubs have been growing for less than 20 years, is 282 per m^2, which is less than 50 % of the 694 per m^2 measured at Site 1.

Table 4.2 Plant height, density, stem diameter and pneumatophore density values displayed by site (mean ± SD)

SITE	Mean plant density (m^2)	Mean plant height (m)	Mean stem diameter (m)	Mean pneumatophore density (m^2)
Site 1	1.5	1.04 (0.22)	.049 (.032)	694 (99)
Site 2	0.9	1.10 (0.16)	.048 (.030)	470 (86)
Site 3	2.5	0.76 (0.15)	.029 (.020)	535 (202)
Site 4	1.3	1.03 (0.22)	.035 (.026)	282 (33)

4.6.3 Surface Sediment Characteristics

The greatest mud (particle size < 63 μm, as defined by Folk (1974), incorporating % clay and % silt) content of surface sediments is found within Site 1 (93%), toward the head of the estuary. Mud content exceeds 50% for all mangrove and cleared sites, however some spatial variability between and within mangrove sites is evident. The undisturbed bare flats at Sites 1, 2 and 3, however, possess mud content < 40 % and therefore contain > 60 % sand (Table 4.3).

An increase in grain size across bare flats of Transect 2 in summer 2007 compared with winter 2006 is apparent, with the opposite trend occurring at bare flat locations of Transects 3 and 4 (Figure 4.4). No clear seasonal fluctuation is discernible in mangrove habitat due to the range of grain sizes recorded.

Table 4.3 Surficial sediment textural analyses for sites under mangroves, cleared of mangroves and on undisturbed bare flats in the Waikaraka Estuary. Samples collected February 2007.

	Mangroves					cleared of mangroves					undisturbed intertidal flats				
	site name	% clay	% silt	% sand	mean grain (μm)	site name	% clay	% silt	% sand	mean grain (μm)	site name	% clay	% silt	% sand	mean grain (μm)
Site 1	TP1-1	17	76	7	22	TP1-3	6	49	45	93	TP1-5	5	31	64	138
	TP1-2	15	78	7	21	TP1-4	6	46	48	88					
Site 2	TP2-1	14	62	24	55	TP2-3	7	49	43	94	TP2-5	4	22	73	213
	TP2-2	10	46	44	114	TP2-4	5	46	49	98					
Site 3	TP3-1	9	41	50	173	TP3-3	7	52	41	85	TP3-5	5	33	62	136
	TP3-2	14	69	17	39	TP3-4	7	50	43	91					
Site 4	TP4-1	0	70	13	32	TP4-3	13	72	15	33	TP4-5	5	47	48	84
	TP4-2	15	65	20	47	TP4-4	11	66	23	44					
Mean		**12**	**64**	**23**	**63**		**8**	**54**	**39**	**78**		**5**	**33**	**62**	**143**
Std Dev		5	13	16	53		3	10	13	25		1	10	11	53

Temporal variation in sediment texture of the cleared plots is displayed in Figure 4.5. Results exhibit an apparent increase in grain size from winter 2005 to summer 2006, however this is obscured by the considerable variation in grain sizes recorded for summer 2006.

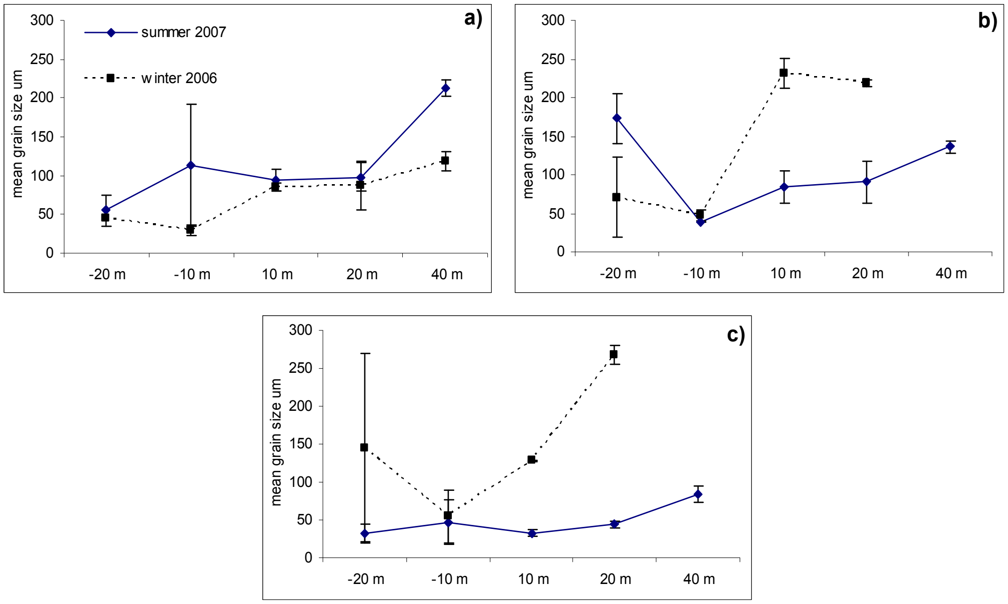


Figure 4. 5 **Seasonal grain size variation measured in Winter 2006 and Summer 2007 along transects at Site 2 (a), Site 3 (b) and Site 4 (c). Samples collected 20 m (-20 m) and 10 m (-10) landward of mangrove fringe and 10, 20 and 40 m seaward of the mangrove fringe.**

In August 2005, mean grain size within cleared plots ranged from 22 µm (± 3) at Clear Plot 2, to 53 µm (± 24) at Clear Plot 3, being medium silt to very-fine sand. In February 2007, mean grain size ranged from 82 µm (± 43) at Clear Plot to 94 µm (± 27) at Clear Plot 3 (very fine sand) (Table 4.3).

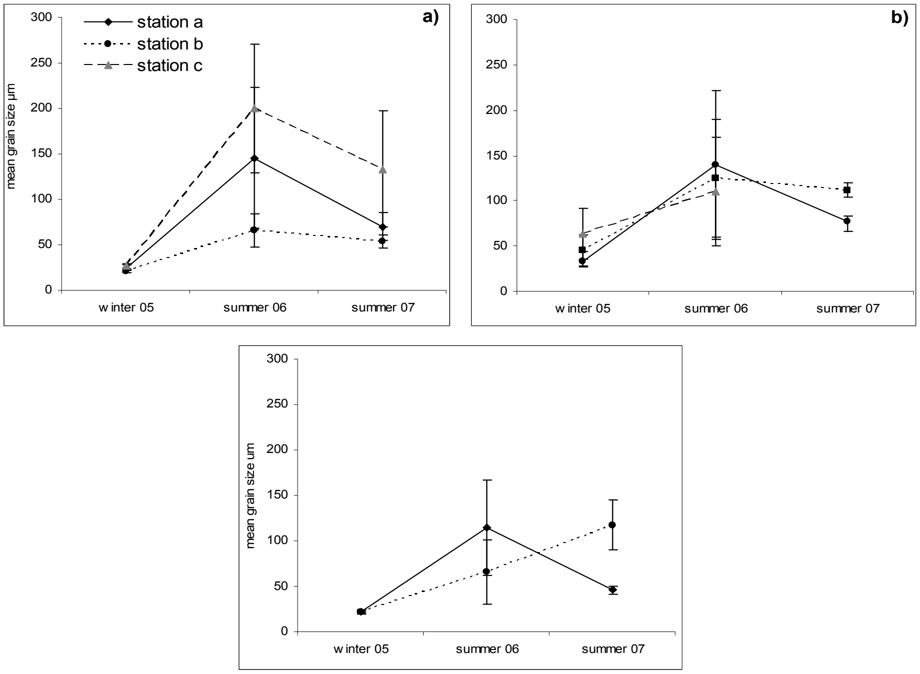


Figure 4. 6 **Surface grain size (µm) and standard deviation of the average of 3 samples per collection station within Cleared Plot 1 (a), Cleared Plot 2 (b) and Cleared Plot 3 (c) from samples collected within 3 months of mangrove clearance (winter 2005), and the following summer 2006 and summer 2007.**

4.6.4 Sediment Cores: Site 4

Sedimentary features of three cores collected in the vicinity of Site 4 are displayed in Figure 4.7. A comparison of the surface facies indicates a deeper, finer-grained surface layer in the mangrove zone (Core (a)). Mangrove rootlets are most dense in the upper 15 cm of black silty sand. Shell material is absent in Core (a), whereas coarse and fine sands are coupled with shell hash and shell material in Cores (b) and (c), indicative of intertidal deposits.

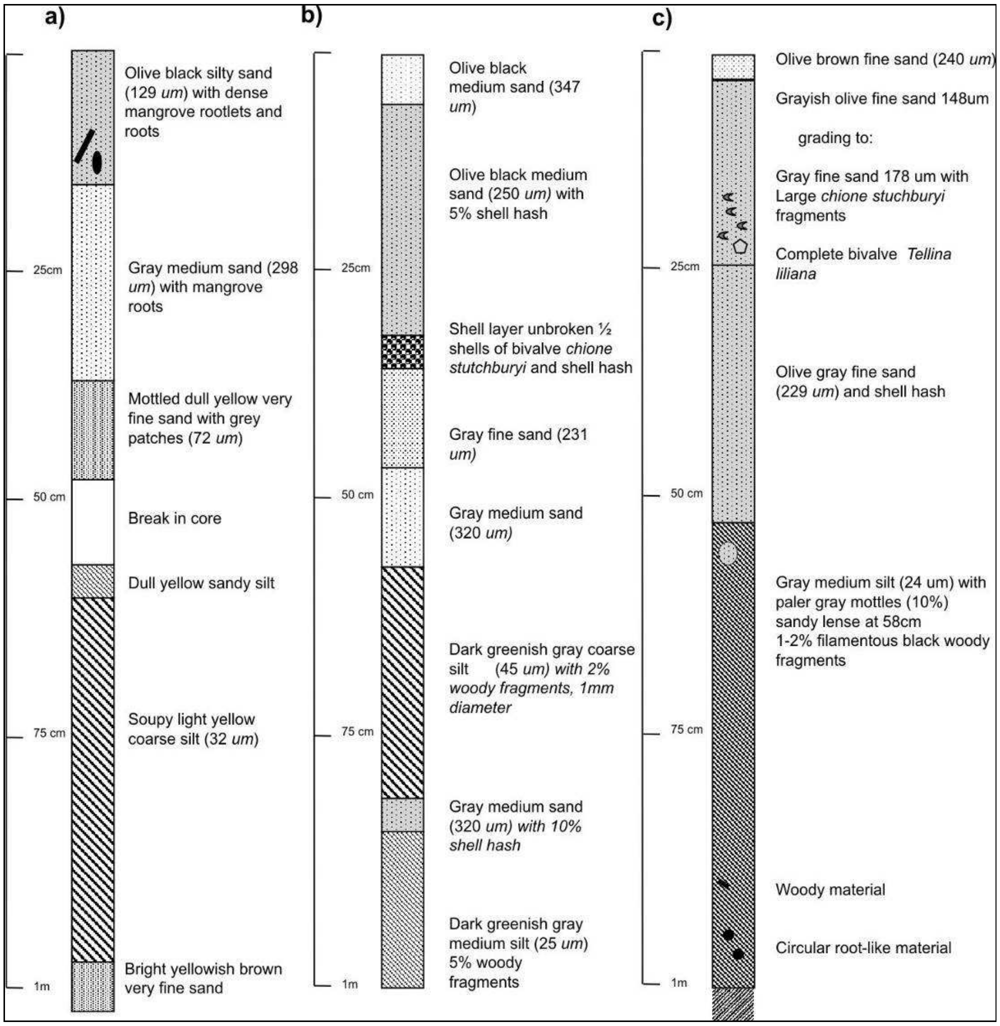


Figure 4. 7 **Core stratigraphy for a site cleared of mangroves 3 months before core collection, Core (a); bare flats within 25 m of the cleared mangrove zone, Core (b); and 15 m east of the main tidal channel, Core (c).**

The most noticeable change to the sedimentary units occurs at depths of around 50-55 cm in all cores, where overlying sandy beds are replaced with unconsolidated silts which penetrate to depths beyond 1 m in Core (c), 85 cm in Core (b) and just under 1 m in Core (a). Shell material is not present in these lower facies, except for a 4 cm sandy layer with shell hash found at 80 cm in Core (b), representative of a tidal channel or intertidal sand-flat environment. The coarse silt found between 55 cm and 95 cm in Core (a) is “soupy” which could indicate groundwater penetration, a zone of poor water filtration, or a bed of degrading volcanic sediment containing smectite (Harmsworth, 1983).

The short core collected at Site 2 was found to have a surface layer to 8 cm consisting of olive-black, medium silt (16-22 µm). Mean grain size then changed to coarse silt and very-fine sand to a depth of 25 cm, below which was medium and fine sands to 35 cm. Comparison between the short core and Core (a) suggests the finer silt fraction has been removed from the surface of Core (a), which was cleared of mangroves three months prior to collection.

4.6.5 Sediment Cores: opposite Site 4 (western margins)

The sediment profile under the intertidal sandflats adjacent to Site 4 can be broadly separated into an upper and a lower unit (Figure 4.8C and 4.8D). The upper 95 cm of the core contained alternating beds of poorly-sorted and angular fine, medium and coarse sands. Medium sand dominated the upper 25 cm of the core, underlain by sands grading from fine sand at a depth of 26 cm, to coarse sand at 42 cm. Coarse sands positioned between 42 and 50 cm are underlain by very-fine sand and coarse silt deposits between 50 and 95 cm which are intermittently interrupted by narrow beds of medium sand at 55 – 60 cm and again at 75 to 85 cm and a fine sand deposit between 92-95 cm. Mineralogy was similar at all depths, and consisted of pumice granules, quartz, mica, glass shards, hornblende and shell hash.

The lower 95 – 160 cm of the core was dominated by well-sorted silts which were similar in their mineralogy to the overlying deposits, with a slight textural fining with depth to the base of the core. Shell-hash contained within these silts was < 5 %. Woody material collected below 1 m was broadly identified as podocarp (possibly Matai) and podocarp bark. A cockle shell at the base of the core (155 cm) was ^{14}C dated at 7181 years ($\underline{+}$ 38 BP).

The top 8 cm of the mangrove core analysed for ^{210}Pb analysis and dating contained brown and black silts (see Table 4.4 for mean grain size). Mangrove roots found in this upper layer continued to a depth of 20 cm. Very-fine sand between the surface and 8 cm, graded to medium sand at 22cm. A deposit of very fine sand extends from 24 to 42 cm, which overlies medium and fine sands that continue to the base of the core at 62 cm (Figure 4.8A and 4.8B). Bulk density increases with depth, attributable to a combination of sediment compaction and coarsening grain size (Table 4.4). Surface sediments of mangrove silts were found to have a bulk density of < 0.6 g cm^3, compared to 0.9 to 1.19 g cm^3 calculated for the underlying sandy deposits.

Results of the ^{210}Pb analysis identified variable sedimentation rates over time. The top 6 cm of the core were not included in radionuclide analysis due to the likelihood of sediment mixing which is known to skew results (Lynch et al., 1989; Smoak and Patchineelam, 1999). To construct a timeline associated with sedimentation rates it is important to include these surface sediments. To accommodate for this in the creation of an approximate timeline, an average rate of surface elevation of 3.5 mm yr^{-1} was calculated from results of RSET measurements reported in Stokes et al. (2009), equating to an approximate period of deposition of 17 years (1990 – 2007).

Interpretation of the ^{210}Pb results suggest that sediment accumulation rates were rapid between circa 1920 – 1925 at 36 mm yr^{-1}, followed by a relatively slower, but still considerable sedimentation of 10 mm yr^{-1} between approximately 1925 and 1950. Sedimentation slowed again between 1950 and 1990 to 2.3 mm yr^{-1}.

These results suggest that 60 cm of estuarine sediments have accumulated on the western, mid-estuary intertidal flats since *c.* 1920, equating to 90 years of sedimentation. The date of 7181 years BP ($\pm$ 38) provided by ^{14}C dating suggests that the estuarine sediments at depths of 60 – 155 cm accreted at an average rate of 0.1 mm yr^{-1}.

Table 4.4 Depth profile of mean grain size (*µm*) and bulk density of sediments analysed for ^{210}Pb analysis.

CORE DEPTH (cm)	MEAN GRAIN SIZE (µm)	BULK DENSITY g cm^3	CORE DEPTH (cm)	MEAN GRAIN SIZE (µm)	BULK DENSITY g cm^3
1-2	-	0.526	29-30	129	1.194
2-3	46.7	0.597	33-34	128	1.128
3-4	30.6		37-38	137	-
6-7	36.8	0.512	41-42	183	1.138
9-10	94.3	0.650	45-56	249	-
13-14	111.3	0.945	49-50	253	-
17-18	262	1.171	53-54	259	-
21-22	254	-	57-58	187	1.002
25-26	154	-	62-63	138	0.914

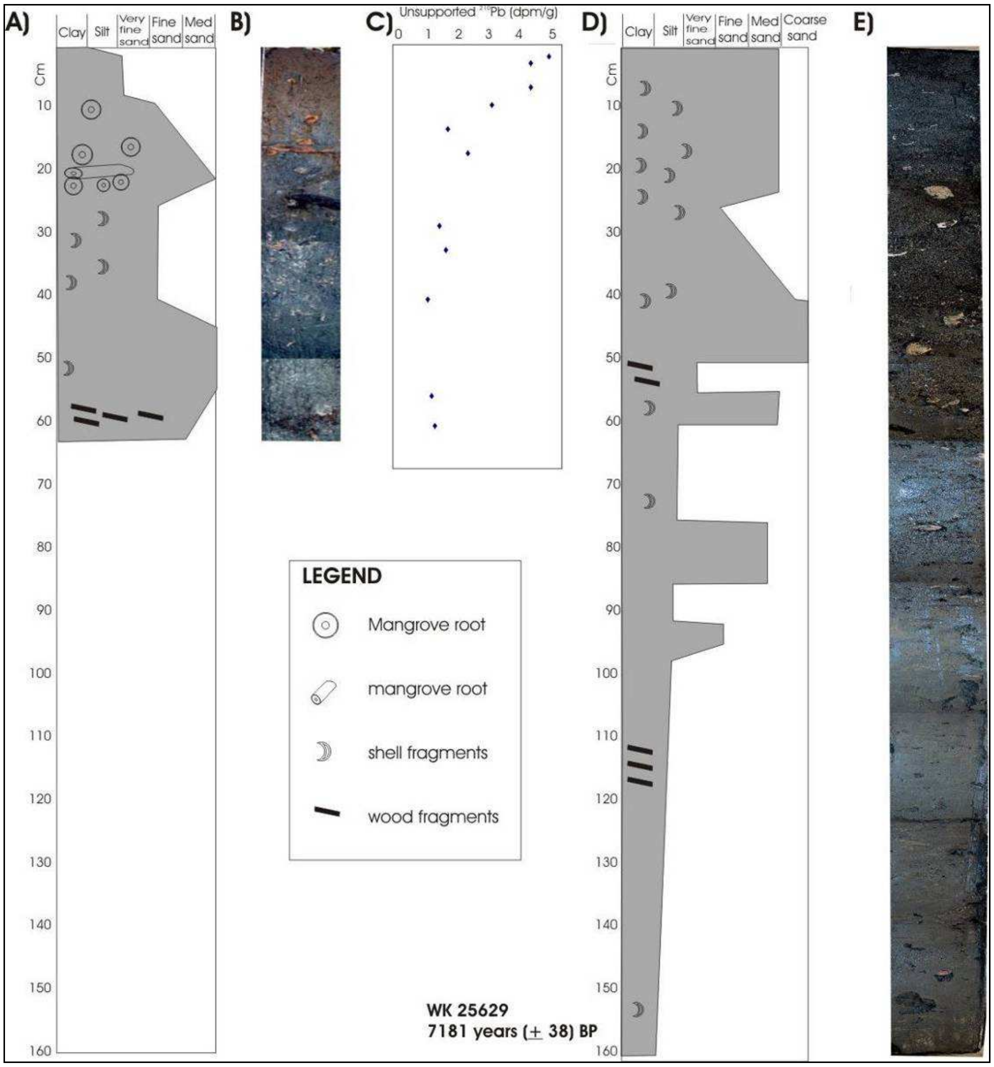


Figure 4.8 **Sediment characteristics of a 63 cm core collected in mangrove habitat (A) and (B) and Unsupported ^{210}Pb (C) used to determine Sediment Accumulation Rates. Sediment characteristics of a 160 cm core collected from adjacent unvegetated intertidal flats west of the main tidal channel of Waikaraka Estuary are also displayed (D) and (E).**

4.6.6 Surface Elevation Change

Annual rates of surface elevation change displayed in Figure 4.9 show a reduction in surface elevation ranging from 9 mm yr^{-1} to 38 mm yr^{-1} within the zones cleared of mangrove vegetation, and mostly an increase in surface elevation within mangrove vegetation, ranging from – 5 mm yr^{-1} to 14 mm yr^{-1}. Highest rates of surface elevation increase were recorded inside mangrove habitat along RSET Transect 2 (6 mm yr^{-1} and 14 mm yr^{-1}).

Figure 4.10 demonstrates the cumulative decline in surface elevation recorded within the areas cleared of mangroves. Cumulative surface elevation change along the RSET transects shows an apparent stability of the bare flats in the vicinity of RSET Transect 2, whereas the bare flats of the cleared plots (RSET Transect 1) experienced a continual fall in surface elevation. Migration of a small channel was observed in the vicinity of RSET Transect 3, which is reflected in the fall in surface elevation at the 20 m RSET in March 2007. Figure 4.10 also illustrates an overall increase in surface elevation measured over time at most mangrove RSET stations, although some temporal variation is evident.

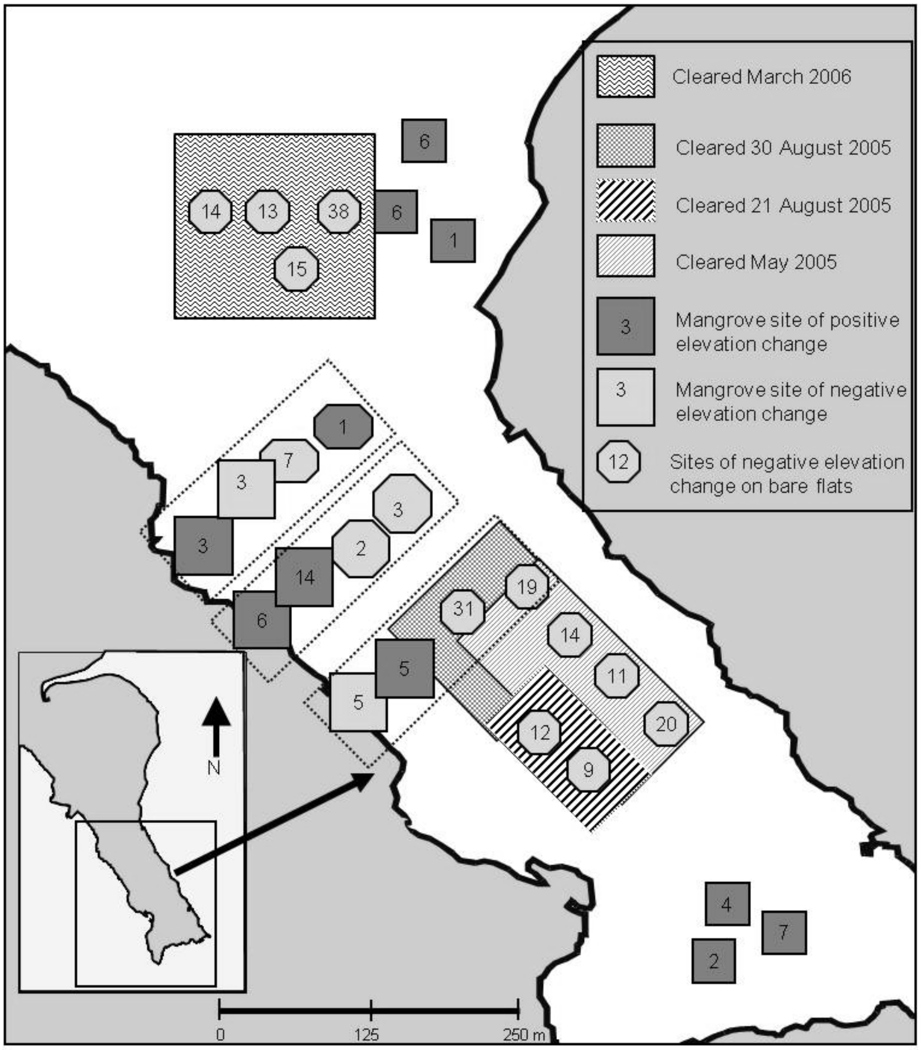


Figure 4.9 **Graphic illustration of annual rates of surface elevation change, calculated from erosion pin and RSET measurements for the monitoring period March/April 2006 to March 2007. RSET transects are outlined with hatched line.**

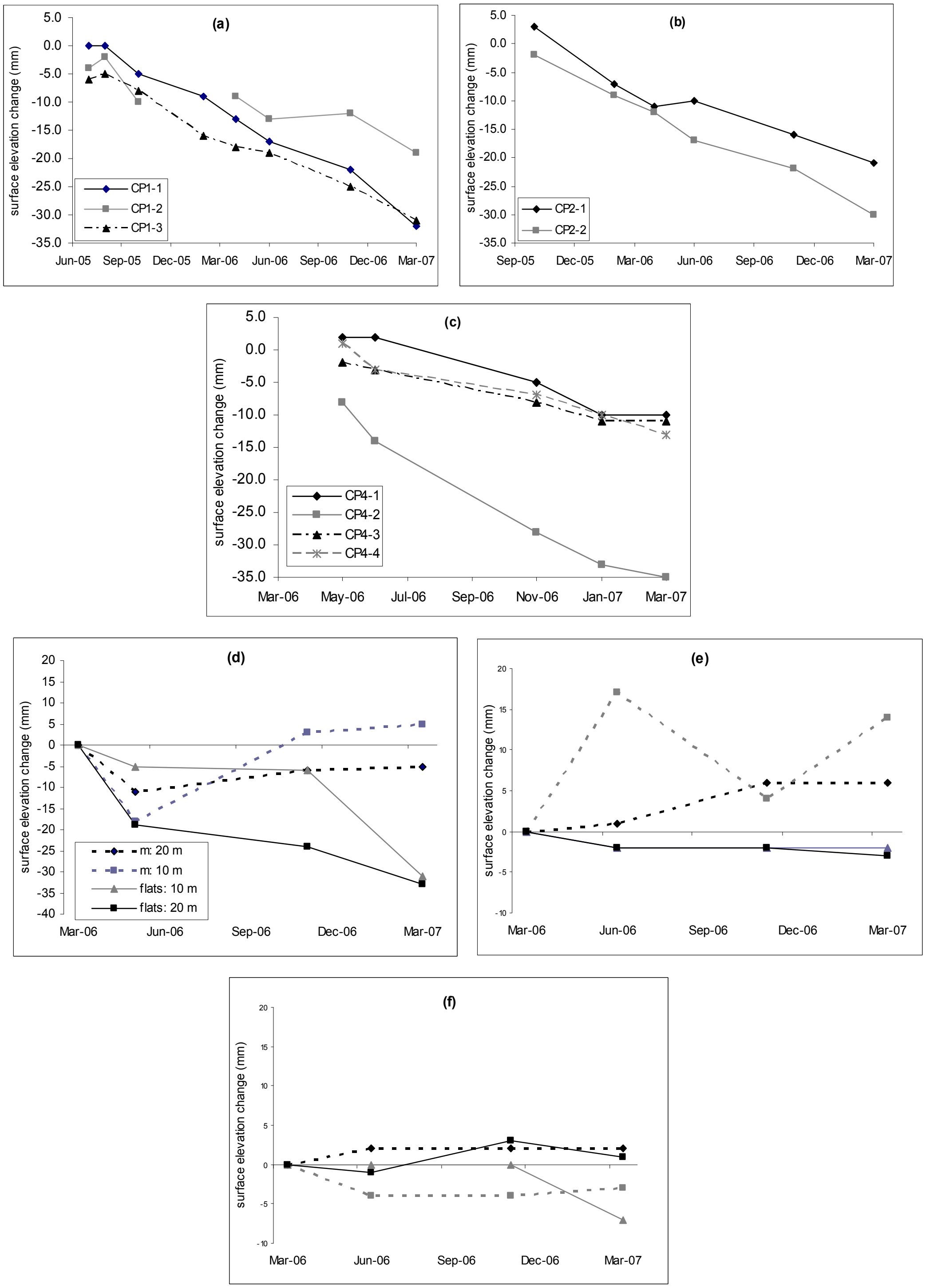


Figure 4.10 **Cumulative surface elevation change within Cleared Plot 1 (a), Cleared Plot 2 (b) and Cleared Plot 4 (c); and along RSET Transects 1 (d), RSET Transect 2 (e) and RSET Transect 3 (f). Mangrove locations of RSET transects are represented by hatched lines.**

4.6.7 Sediment Accumulation Rates from Sediment Traps

Sediment trap results exhibit variation in sediment accumulation rates, with the greatest accumulation occurring on the bare flats where values ranged from 1,200 to 6,000 g m^2 mo^{-1} (Figure 4.11). Site 4, closer to the estuary mouth, generally shows the highest values of sediment accumulation.

Figure 4.12 suggests there is no linear relationship between rates of sediment accumulation and total rainfall or highest rainfall intensity for the trap deployment periods (rainfall data from NIWA Climate Data Centre, Tauranga Aerodrome recording station).

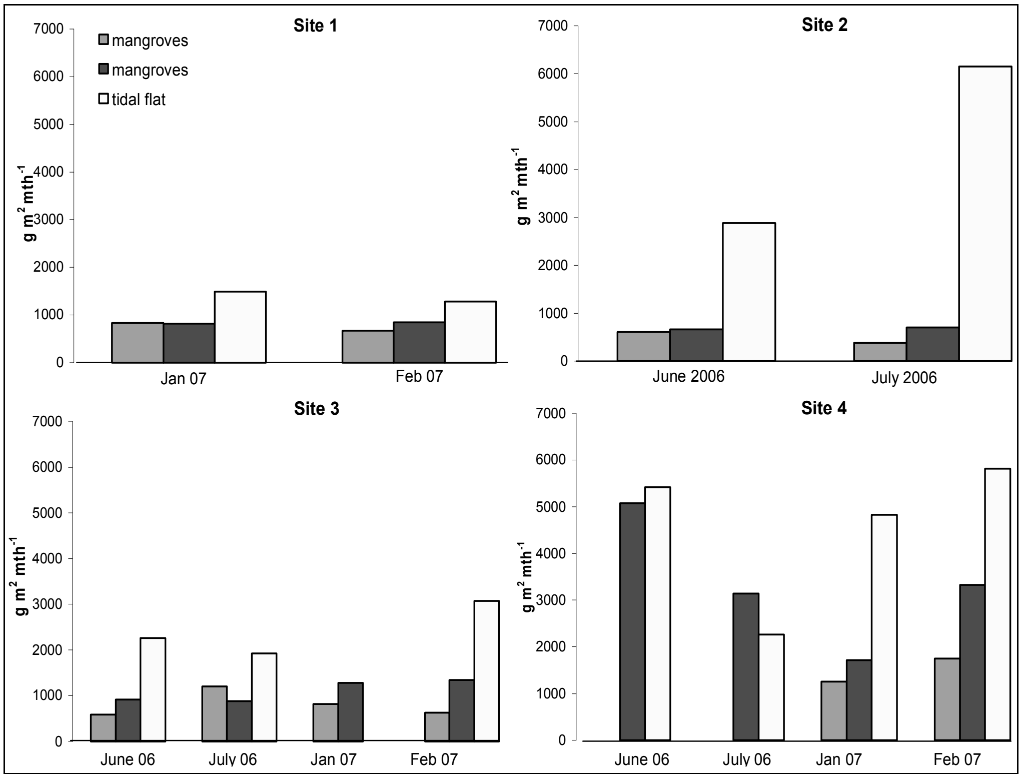


Figure 4.11 **Sediment Trap accumulation rates (g m^2 mo^{-1}) for June 2006, July 2006, January 2007 and February 2007. Pale grey columns represent trap locations within mangrove habitat 10 m from mangrove fringe; dark grey columns represent sites 5 m from mangrove fringe, and white columns represent traps positioned on bare flats 10 m from the mangrove fringe.**

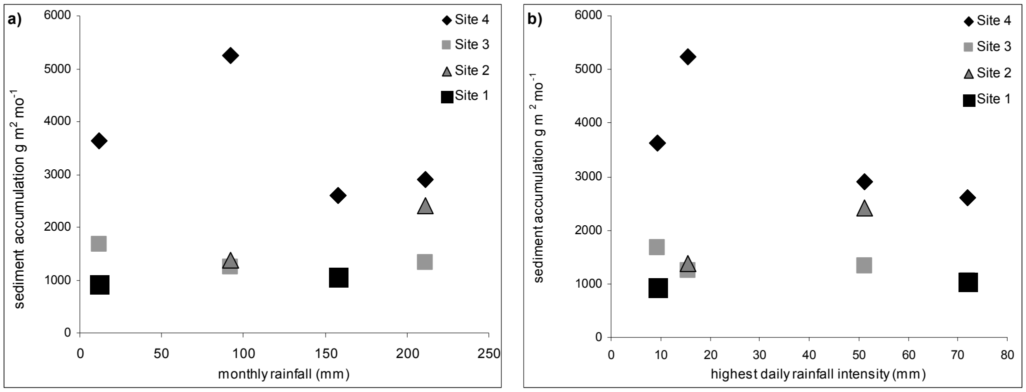


Figure 4.12 **Sediment trap results of sediment accumulation plotted against rainfall intensity and total rainfall for the trap deployment periods June 2006, July 2006, January 2007 and February 2007.**

4.7 DISCUSSION

4.7.1 Historical sedimentation

The geomorphic evolution of Waikaraka Estuary over the last century can be linked to increased sedimentation triggered by deforestation of the catchment. Quarrying of the local rhyolite in this area has also increased sediment deposits in the estuary. Analysis of ^{210}Pb dated sediments collected toward the seaward end of the neck of the estuary identified a sharp transition from slow deposition of silts to more rapid sedimentation associated with poorly-sorted sands. This change was found 1 m below the surface on the west side of the channel and approximately 0.5 m below the surface east of the channel (Stokes et al., 2009), and may correlate with the onset of land-clearance for agriculture. A bed of angular coarse sands positioned 42 cm below the surface was analysed using ^{210}Pb and dated as being deposited within the 1920s. This deposit may be representative of the impacts from rhyolite quarrying, which commenced production at that time and continued through to the 1960's. Sediment accumulation rates of 35 mm yr^{-1} occurred in the early years of quarry operation and fell to 10 mm yr^{-1} between 1925 and 1950, with a further reduction of sedimentation (2.3 mm yr^{-1}) recorded over the last half century.

The average SAR of < 0.1 mm yr^{-1} from ~ 7000 BP to approximately 1920, inferred from ^{14}C results, is likely to be an underestimate. The use of carbon dates to interpret historic sedimentation rates provides only a net rate of sediment accumulation and does not account for any sediment compaction which is likely to occur under load. Pre-European sedimentation rates determined for other North Island estuaries range from < 1 mm yr^{-1} (Sheffield et al., 1995; Swales et al., 2002a) to < 1.5 mm yr^{-1} (Hume and McGlone, 1986).

The lag between the increased sedimentation associated with European land-use and the onset of rapid mangrove colonisation in the 1950s/1960s can be explained, at least partially, by the critical tidal limits required for the survival of *Avicennia* seedlings (Swales et al., 2007). The lower elevation limit (LEL) for *Avicennia* is typically between Mean Sea Level (MSL) (Clarke and Myerscough, 1993) and 0.3 m above MSL (Swales et al., 2007). The sediment accumulation rate of 10 mm yr^{-1} between approximately 1925 and 1950, as discussed above, would afford sufficient vertical growth of the tidal flats to bring the surface topography close to mean sea level and therefore provide suitable conditions for seedling survival. Historical SAR's toward the head of the estuary would be relatively higher due to the proximity of the sediment supply and minimal tidal influence, and therefore likely to have reached the critical elevation for mangrove survival earlier. This gradient of SAR is common (Swales et al., 2002a; Ellis et al., 2004;), and further evidenced by aerial photo analysis of sites in Tauranga Harbour which identifies narrow mangrove stands isolated to the upper estuary (Site 1 and 2 as identified in Figure 4.1) in the early 1940s. Presently, the mangrove stands in Waikaraka Estuary are positioned mostly at or above 0.3 m above MSL (Moturiki datum) and the unvegetated sandflats are mostly at MSL (Park, 2004).

4.7.2 Mangrove expansion and contemporary sedimentation

The purpose of this study was to report on the mangrove expansion at Waikaraka Estuary and investigate the physical changes that have occurred as a result of mangrove removal. Photogrammetry documented a 23 % increase in mangrove coverage over the total estuary area between the years 1943 to 2003, with the greatest rate of expansion occurring between 1982 and 1996. The expansion rate has subsequently slowed, possibly as a result of human intervention via physical removal of propagules from the estuary. The main driver for mangrove expansion at this site may be sedimentation. The Waikaraka Estuary catchment area has experienced considerable land clearance since European settlement (approximately 150 - 200 years), and during this time sediment loads entering the estuary appear to have been greater than the present-day. Prior to the 1980s, stock grazing, land reclamation and rubbish dumping were all permissible activities in

New Zealand estuaries, which may have truncated any estuarine vegetation establishment during that time. Recent prohibition of these activities may play some role in the success of mangrove expansion. Other possible factors include increases in nutrient run-off as a result of agricultural and horticultural activities, or a reduction in the occurrence of chilling temperatures during the establishment phase of mangrove propagules (Beard, 2006).

Mangrove shrubs in Waikaraka Estuary display a mean plant height of <1.5 m, in contrast to other New Zealand sites where tree heights range between 2 and 6 m in similar physical conditions (Young and Harvey, 1996; Osunkoya and Creese, 1997; May, 1999; Morrisey et al., 2003; Ellis et al., 2004; Alfaro, 2005). The study site is located toward the southern limit of mangrove distribution in New Zealand, and the limited plant growth can be attributed to climatic stress (Beard, 2006). Spatial variation in plant height is commonly found in mangrove habitat (e.g. Burns and Ogden, 1985; Ellis et al., 2004) and in this study could not be attributed to age. Other possible causes such as salinity (Crisp et al., 1990) and nutrient availability (Fry et al., 2000; Naidoo, 2006) were not measured.

Pneumatophore densities measured in this study are higher than those reported in other New Zealand estuaries (Young and Harvey, 1996; Morrisey et al., 2003; Ellis et al., 2004; Alfaro, 2005) which may be due to the high mud content of surface sediments (Ellis et al., 2004). The low pneumatophore density measured within the youngest stand of mangroves in Waikaraka Estuary is consistent with a reported correlation between increasing plant age and higher pneumatophore densities (Morrisey et al., 2003). Pneumatophore density has also been found to correlate with increased sediment trapping capability (Young and Harvey, 1996). Sediment trapping occurs within the mangrove vegetation at the study site, evidenced by the recorded increase in surface elevation. Surface elevation change averaged 3 mm yr^{-1}, which is less than that recorded in other New Zealand estuaries (Swales et al., 1997; Ellis et al., 2004; Swales et al., 2007), although this is similar to values recorded in Florida (Cahoon and Lynch, 1997), Vietnam (Van Santen et al., 2006) and temperate Australia (Rogers et al., 2005; Rogers et al., 2006).

Sedimentation rates are influenced by sediment supply into the estuary, and hydrodynamic processes (Furukawa et al., 1997). As Waikaraka Estuary receives a relatively low volume of freshwater inflow, it is likely that suspended sediment input will also be relatively low, particularly in light of the small catchment area (10 km^2).

The establishment of mangrove vegetation on previously bare tidal flats initiates a substantial change in surface sediment characteristics. Interpretation of core stratigraphy and surface sediment analysis suggests that bed material of fine and medium sand representative of the bare intertidal flats, is replaced by silt-dominated sediment once mangroves become established. The depth of mud is likely to vary spatially within the estuary, and was found to extend to a depth of 8 cm in the vicinity of a well-established mangrove stand located roughly equi-distant between the mouth and head of the estuary. Interestingly, medium and coarse silts were also found at depths of around 55 cm below the surface, suggesting that the study site has experienced accumulation of finer-grained material in the past.

Rates of surface elevation change associated with mangrove vegetation at Waikaraka Estuary ranged from -5 mm to 14 mm yr^{-1}. The rate of surface elevation change is spatially and temporally variable with no clear seasonal fluctuations discernible over the monitoring period. A relationship between sedimentation with distance from the head of the estuary has been reported in other studies (Young and Harvey, 1996), but was not evident at this site. Higher values of surface elevation change recorded mid-estuary coincide with lower values along the RSET transects either side, suggesting the existence of a narrow depositional zone within this section of the estuary. This could be the result of tidal currents pushing released sediment from neighbouring cleared zones into this mangrove zone (approximately 200 m downstream), or may simply be due to a topographical/hydrodynamic anomaly favouring deposition at this location.

Sediment availability (determined from sediment traps) is lower within mangrove habitat than on the adjacent bare flats, further demonstrating the trapping capabilities of mangroves at the study site, particularly as the higher sediment accumulation rates of the bare flats do not result in a net gain in surface elevation. This trend of decreasing sediment load between the bare tidal flats and vegetation zone, coupled with increasing sedimentation into fringing mangrove habitat, has been discussed by other authors and is considered to be a function of both the trapping capability of high vegetation density (Furukawa and Wolanski, 1996), and erosional episodes of the less stable sediments on the bare tidal flats (Van Santen et al., 2006). The monitoring undertaken in this study coincided with mangrove clearing activities, therefore the sediment accumulation rates quoted may not reflect typical, or ambient, sediment availability but is likely to reflect the injection of released sediment from cleared zones. A positive correlation between rainfall and sediment accumulation has been reported in other studies (Saad et al., 1999; Van Santen et al., 2006), however this trend was not evident during the periods of trap deployment at Waikaraka Estuary, possibly due to this remobilisation of sediment.

Since May 2005 approximately 9,600 m^2 of mangrove vegetation has been removed from Waikaraka Estuary, resulting in significant changes to surface topography. Surface elevation within cleared areas declined at rates of 9 to 38 mm yr^{-1} (average 14 mm yr^{-1}). The decomposition of mangrove root material has been found to contribute significantly to surface subsidence, following a study of mass tree mortality (Cahoon et al., 2003). Unfortunately, marker horizons were unsuccessful in this study and as such it is not possible to separate the processes of sediment erosion and root-mass decomposition. An apparent increase in grain size between winter 2005 and summer 2007, mostly of no more than 30 µm, is skewed by a systematic and substantial increase in grain size documented for summer 2006, coupled with a considerable range of mean values. Possible explanations for this anomaly are a) a function of spatially variable root-mass decomposition resulting in zones of released sediments along with trapped, coarser sediments within areas where root mass is still significant, b) the temporary exposure of underlying coarser material, c) the response to a period of increased flow velocities; d) an artifact of sample collection. A fining of surface texture between winter 2006 and summer 2007 occurred on bare flats adjacent to cleared zones at two of three sampling locations, which could possibly be due to deposition of silt released from nearby cleared areas.

4.8 CONCLUSION

The distribution and expansion of mangrove habitat in Waikaraka Estuary over the last 60 years is reported and the changes in surface topography and surface sediment as a result of mangrove removal are documented. Mangrove coverage has increased from 16,000 m^2 in 1943, to 115,000 m^2 in 2003. This expansion may be linked to markedly increased sedimentation after the 1920s which resulted in a rapid infilling of the site whereby an increased area of intertidal habitat was elevated to the critical elevation limits for mangrove colonisation. The measured mean tree heights of less than 1.5 m are significantly shorter than mangroves growing in warmer regions of New Zealand, inferring climatic limitations to growth. Annual rates of surface elevation change within mangrove habitat (using erosion pins and RSETs) averaged 3 mm, which demonstrates sediment trapping by mangrove vegetation. In contrast, after mangrove clearance a reduction in surface elevation occurred, ranging from 9 mm to 38 mm yr^{-1} (mean 14 mm yr^{-1}). Concurrent to this decline in surface elevation is an increase in mean grain size (<53 µm to ~ 78 µm), indicating remobilisation of some of the silt fraction as a result of a) the loss of above-ground plant architecture which dampens water flow;

and b) decomposition of root material which previously held sediment. This study demonstrates that the removal of mangroves results in some remobilisation of sediment, mostly in the silt fraction. It is important to note, however, that 18 months after mangrove clearing the remaining sediment is still finer than that of the surrounding bare flats.

4.9 ACKNOWLEDGEMENTS

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4.10 PHOTO GALLERY – WAIKARAKA ESTUARY

Figure 4.13 Aerial view of Waikaraka Estuary (in foreground), and catchment, looking from the south-west. Photo courtesy of Lawrie Donald, 2006.

Figure 4.14 Aerial view of Site 1 mangroves (top of photo) and Site 2 (right) of Waikaraka Estuary, looking north. Clear Plots 1, 2 and 3 are outlined. Dark circles seen on the cleared substrate show locations where debris has been incinerated on-site. Photo courtesy of Lawrie Donald, 2006.

Figure 4.15 Aerial view of Waikaraka Estuary showing Site 4, lower-right; and southern end of Clear Plot 3 (left). View looking south. Photo courtesy of Lawrie Donald, 2006.

Figure 4.16 Mangrove plant, pneumatophores and seedlings, Site 2.

Figure 4.17 Mangrove vegetation at Site 4. Plants approximately 1 m in height.

Figure 4.18 The detachable arm, in place on RSET of Transect 1. Each pin is numbered to match the same numbered hole along the arm. Measurements were taken on the east and west sides of the RSET base; and foot traffic restricted to north and south of the RSET base.

Figure 4.19 Cleared Plot and debris piles (1-2 m high) fronting Site 4, April 2006.

Figure 4.20 Debris piles after clearing at Site 4. Photo taken on an incoming tide, April 2006.

Figure 4.21 Footprint in a cleared plot showing anoxic muds, surface debris, and mat of fine roots below the surface.

4.11 LITERATURE CITED

Alfaro, A.C., 2005. Benthic macro-invertebrate community composition within a mangrove/seagrass estuary in Northern New Zealand. *Estuarine, Coastal and Shelf Science* 66, 1-2, 97-110.

Alongi, D.M., Pfitzner, J., Trott, L.A., Tirendi, F., Dixon, P., and Klumpp, D.W., 2005. Rapid sediment accumulation and microbial mineralization in forests of the mangrove *Kandelia Candel* in the Jiulongjiang Estuary, China. *Estuarine, Coastal and Shelf Science* 63(4), 605-618.

Beard, C., 2006. Physiological constraints on the latitudinal distribution of the mangrove *Avicennia marina* (Forsk.) vierh. subsp. *australasica* (Walp.) J. Everett in New Zealand, PhD thesis, University of Waikato, 203 p.

Briggs, R.M., Hall, G.J., Harmsworth, G.R., Hollis, A.G., Houghton, B.F., Hughes, G.R., Morgan, M.D., and Whitbread-Edwards, A.R., 1996. *Geology of the Tauranga Area*. Occasional Report No. 22, Department of Earth Sciences, University of Waikato, New Zealand, 56 p.

Burns, B.R., and Ogden, J., 1985. The demography of the temperate mangrove *[Avicennia Marina* (Forsk.) Vierh.] at its southern limit in New Zealand. *Australian Journal of Botany* 10, 125-133.

Cahoon, D.R., and Lynch, J.C., 1997. Vertical accretion and shallow subsidence in a mangrove forest of southwestern Florida, U.S.A. *Mangroves and Salt Marshes* 1, 173-186.

Cahoon, D.R., Marin, P.E., Black, B.K., Lynch, J.C., 2000. A method for measuring vertical accretion, elevation, and compaction of soft, shallow-water sediments. *Journal of Sedimentary Research* 70(5), 1250-1253.

Cahoon, D.R., Lynch, J.C., Perez, B.C., Segura, B., Holland, R.D., Stelly, C., Stephenson, G., and Hensel, P., 2002. high-precision measurements of wetland sediment elevation, II. The Rod Surface Elevation Table. *Journal of Sedimentary Research* 72, 734-739.

Cahoon, D.R., Hensel, P., Rybczyk, J., McKee, K.L., Proffitt, E. and Perez, B.C. 2003. Mass tree mortality leads to mangrove peat collapse at Bay Islands, Honduras after Hurricane Mitch. *Journal of Ecology* 91, 1093-1105.

Clarke, P. J. and Myerscough, P. J., 1993. The intertidal distribution of the grey mangrove (*Avicennia marina*) in southeastern Australia: the effects of physical conditions, interspecific competition, and predation on propagule establishment and survival. *Australian Journal of Ecology* 18, 307-315.

Crisp, P., Daniel, L., and Tortell, P., 1990. *Mangroves in New Zealand: trees in the tide*, GP Books, New Zealand, 68 p.

de Lange, W., and de Lange, P.J., 1994. An appraisal of factors controlling the latitudinal distribution of mangrove *(Avicennia marina* var. *resinifera)* in New Zealand. *Journal of Coastal Research* 10, 539-548.

Ellis, J., Nicholls, P., Craggs, R., Hofstra, D., and Hewitt, J., 2004. Effects of terrigenous sedimentation on mangrove physiology and associated macrobenthic communities. *Marine Ecology Progress Series* 270, 71-82.

Ellison, J. C., 2008. Long-term retrospection on mangrove development using sediment cores and pollen analysis: A review. *Aquatic Botany* 89, 93-104.

Fan, D. D., Li, C. X., Wang, D. J., Wang, P., Archer, A. W. and Greb, S. F., 2004. Morphology and sedimentation on open-coast intertidal flats of the Changjiang Delta, China. *Journal of Coastal Research*, 43, 23-35.

Folk, R.L., 1968. *Petrology of Sedimentary Rocks,* Hemphill, Texas, 170 p.

Furukawa, K., and Wolanski, E., 1996. Sedimentation in mangrove forests. *Mangroves and Salt Marshes* 1, 3-10.

Furukawa, K., Wolanski, E. and Mueller, H., 1997. Currents and sediment transport in mangrove forests. *Estuarine Coastal and Shelf Science* 44(3), 301-310.

Fry, B., Bern, A. L., Ross, M. S. and Meeder, J. F., 2000. Isotope15 N studies of nitrogen use by the red mangrove, *Rhizophora mangle* L. in South Florida. *Estuarine Coastal and Shelf Science* 50, 291-296.

Granek, E. F. and Frasier, K., 2007. The impacts of red mangrove (*Rhizophora* mangle) deforestation on zooplankton communities in Bocas del Toro, Panama. *Bulletin of Marine Science* 80(3), 905-914.

Harmsworth, G.R., 1983, *Quaternary stratigraphy of the Tauranga Basin*, MSc. thesis, University of Waikato, 342 p.

Hayward, B. W., Grenfell, H. R., Sabaa, A. T., Morley, M. S. and Horrocks, M., 2006. Effect and timing of increased freshwater runoff into sheltered harbor environments around Auckland City, New Zealand. *Estuaries and Coasts* 29(2), 165-182.

Healy, T.R., Cole, R.G., and De Lange, W., 1996. Geomorphology and ecology of New Zealand shallow estuaries and shorelines, in: Nordstrom, K.F., and Roman, C.T., (Eds.), *Estuarine shores, evolution, environments and human alteration*. John Wiley & Sons, Chichester, pp. 115-154.

Hope, H.M., 2002. Sediment dynamics of Waikaraka Estuary, a small, semi-enclosed estuarine system in the upper reaches of a developed harbour, MSc. thesis, University of Waikato, 126 p.

Hume, T., and McGlone, M.S., 1986. Sedimentation patterns and catchment use change recorded in the sediments of a shallow tidal creek, Lucas Creek, Upper Waitemata Harbour, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 20, 677-687.

Hume, T.M., and Herdendorf, C.E., 1992. Factors controlling tidal inlet characteristics on low drift coasts. *Journal of Coastal Research* 8, 355-375.

Kozerski, H. P., 1994. Possibilities and limitations of sediment traps to measure sedimentation and resuspension. *Hydrobiologia* 284(1), 93-100.

Krauss, K.W., Allen, J.A., and Cahoon, D.R., 2003. Differential rates of vertical accretion and elevation change among aerial root types in Micronesian mangrove forests. *Estuarine, Coastal and Shelf Science* 56, 251-259.

Lynch, J. C., Meriwether, J. R., McKee, B. A., Vera-Herrera, F. and Twilley, R. R., 1989. Recent accretion in mangrove ecosystems based on ^{137}Cs and ^{210}Pb. *Estuaries* 12(4), 284 -299.

Massel, S.R., Furukawa, K., Brinkman, R.M., 1999. Surface wave propagation in mangrove forests. *Fluid Dynamics Research* 24, 219-249.

May, J., 1999. Spatial variation in litter production by the mangrove *Avicennia marina* var. *australasica* in Rangaunu Harbour, Northland, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 33, 163-172.

Morrisey, D.S., GA; Ellis, J.I.; Burns, B.R.; Kemp, C.E.; 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, 581-592.

Naidoo, G., 2006. Factors contributing to dwarfing in the mangrove *Avicennia marina*, *Annals of Botany* 97, 1095-1101.

Nichols, M.M. and Biggs, R.B., 1985. Estuaries, in: Davis, R.A., Jr. (Ed.), *Coastal Sedimentary Environments*, Springer-Verlag, New York, pp. 77-186

Osunkoya, O.O. and Creese, R.G., 1997. Population structure, spatial pattern and seedling establishment of the grey mangrove*, Avicennia marina* var. *australasica,* in New Zealand. *Australian Journal of Botany* 45(4), 707-725.

Park, S., 2004. Aspects of Mangrove Distribution and Abundance in Tauranga Harbour. Environment Bay of Plenty (EBoP) Environmental Publication 2004/16. Whakatane, New Zealand, 49 p.

Rogers, K., Saintilan, N. and Cahoon, D., 2005. Surface elevation dynamics in a regenerating mangrove forest at Homebush Bay, Australia. *Wetlands Ecology and Management* 13, 587-598.

Rogers, K., Wilton, K.M., and Saintilan, N., 2006. Vegetation change and surface elevation dynamics in estuarine wetlands of southeast Australia. *Estuarine, Coastal and Shelf Science* 66, 559-569.

Saad, S., Husain, M. L., Yaacob, R. and Asano, T., 1999. Sediment accretion and variability of sedimentological characteristics of a tropical estuarine mangrove: Kenaman, Terengganu, Malaysia. *Mangroves and Salt Marshes* 3, 51-58.

Sheffield, A.T., Healy, T.R., and McGlone, M.S., 1995. Infilling rates of a steepland catchment estuary, Whangamata, New Zealand. *Journal of Coastal Research* 11, 1294-1308.

Smoak, J. M. and Patchineelam, S. R., 1999. Sediment mixing and accumulation in a mangrove ecosystem: evidence from ^{210}Pb, ^{234}Th and ^{7}Be. *Mangroves and Salt Marshes* 3, 17-27.

Spenceley, A.P., 1977. The role of pneumatophores in sedimentary processes. *Marine Geology* 24, M31-M37.

Stokes, D.J., Healy, T.R. and Cooke, P.J., 2009. Surface elevation changes and sediment characteristics of intertidal surfaces undergoing mangrove expansion and mangrove removal, Waikaraka Estuary, Tauranga Harbour, New Zealand. *International Journal of Ecology and Development* 12, 209, 75-87.

Swales, A., Hume, T., and Green, M., 1997. Holocene sedimentation and recent human impacts on infilling in a drowned valley estuary. 13th Australasian Coastal and Ocean Engineering Conference & 6th Australasian Port and Harbour Conference, Volume 1, Christchurch, Centre for Advanced Engineering, pp. 895-900.

Swales, A., Williamson, R. B., Van Dam, L. F., Stroud, M. J. and McGlone, M. S., 2002a. Reconstruction of urban stormwater contamination of an estuary using catchment history and sediment profile dating. *Estuaries* 25(1), 43-56.

Swales, A., Hume, T., McGlone, M., Pilvio, R., Ovenden, R., Zviguina, N., Hatton, S., Nicholls, P., Budd, R., Hewitt, J., Pickmere, S. and Costley, K., 2002b. Evidence for the physical effects of catchment sediment runoff preserved in estuarine sediments: Phase II (field study). Prepared for Auckland Regional Council. NIWA Client Report: HAM 2002-067. National Institute of Water & Atmospheric Research Ltd, Hamilton, New Zealand, 174 p.

Swales, A., Bentley, S. J., Lovelock, C. E. and Bell, R., 2007. Sediment processes and mangrove-habitat expansion on a rapidly-prograding muddy coast, New Zealand. *Coastal Sediments '07*, New Orleans, Louisiana, American Society of Civil Engineers, pp. 1445-1454.

Thomas, S., and Ridd, P.V., 2004. Review of methods to measure short time scale sediment accumulation. *Marine Geology* 207, 95-114.

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

Thrush, S.F., Hewitt, J.E., Norkko, A., Nicholls, P.E., Funnell, G.A., and Ellis, J.I., 2003. Habitat change in estuaries, predicting broad-scale responses of intertidal macrofauna to sediment mud content. *Marine Ecology Progress Se*ries 263, 101-112.

Van Santen, P., Augustinus, P.G.E.F., Janssen-Stelder, B.M., Quartel, S., and Tri, N.H., 2006. Sedimentation in an estuarine mangrove system. *Journal of Asian Earth Sciences* 29, 4, 566-575.

Victor, S., Neth, L., Golbuu, Y., Wolanski, E., and Richmond, R.H., 2006. Sedimentation in mangroves and coral reefs in a wet tropical island, Pohnpei, Micronesia. *Estuarine, Coastal and Shelf Science* 66(3-4), 409-416.

White, J., 1979. Recent sediments of Waikareao Estuary, Tauranga, North Island, New Zealand. MSc. Thesis, University of Waikato, 197 p.

Wolanski, E., Fabricius, K., Spagnol, S., Brinkman, R. 2005. Fine sediment budget on an inner-shelf coral-fringed island, Great Barrier Reef of Australia. *Estuarine, Coastal and Shelf Science* 65(1-2), 153-158.

Wolanski, E., Williams, D., and Hanert, E., 2006. the sediment trapping efficiency of the macro-tidal Daly Estuary, tropical Australia. *Estuarine, Coastal and Shelf Science* 69, 291-298.

Woodroffe, C., 1992. Mangrove sediments and geomorphology, in: Robertson, A. I. and Alongi, D. M. (Eds.), *Tropical Mangrove Ecosystems.* American Geophysical Union, Washington D.C., pp. 7-41.

Young, B.M., and Harvey, E.L., 1996. A spatial analysis of the relationship between mangrove *(Avicennia marina* var. *australasica)* physiognomy and sediment accretion in the Hauraki Plains, New Zealand. *Estuarine, Coastal and Shelf Science* 42, 231-246.

Chapter 5
Characteristics of below-ground biomass of temperate mangroves

5.1 PREFACE

The ultimate goal for many people participating in mangrove removal activities in Tauranga Harbour is to encourage a return to sandy intertidal flats. This study has documented a coarsening of sediment texture after the removal of above-ground mangrove, however any return to a sandy substrate will require not only the erosion of surface sediments, but also of the root mass that remains. The density of below-ground biomass in temperate mangrove and its temporal persistence after above-ground structures are removed is largely unknown. This chapter attempts to address this knowledge gap.

5.2 INTRODUCTION

Mangroves play a significant role in the global carbon cycle, and are estimated to account for approximately 15% of the total carbon that accumulates in marine sediments (Jennerjahn and Ittekkott, 2002). That carbon contribution is often measured by quantifying mangrove biomass (see Komiyama et al., 2008 for a review of techniques and key trends) of which the production of root networks is a significant component (Lovelock, 2008).

The amount of mangrove biomass is a function of productivity, age, organic matter allocation and exportation strategies (Cintron and Novelli, 1984). In turn, productivity will be affected by climate, zonation, succession, soil properties and nutrient status (McKee, 1993; Saenger and Snedaker, 1993; Lovelock et al., 2004; Lovelock et al., 2005; Komiyama et al., 2008). Primary tropical forests have been reported to produce above-ground biomass up to 460 t ha^{-1} (Putz and Chan, 1986). Mangroves in higher latitudes are generally less productive and the smaller biomass may be related to different climatic conditions such as lower temperatures, reduced solar radiation and precipitation (Saenger and Snedaker, 1993). Studies of Australian temperate mangrove report total biomass of 270 – 290 t ha^{-1} (Briggs, 1977) while Woodroffe (1985) measured above-ground biomass of low (dwarf) mangroves in New Zealand at ~ 70 t ha^{-1}, and medium height trees at ~ 240 t ha^{-1}.

Mangroves tend to partition biomass differently to their terrestrial counterparts, by producing a higher relative root mass (Briggs, 1977; Saintilan, 1997a; Tamooh et al., 2008). This has been attributed to an unstable sedimentary environment (Hutchings and Saenger, 1987). The ratio of above-ground to below-ground biomass production can be be influenced by age (Fromard et al., 1998; Tamooh et al., 2008) and salinity (Saintilan, 1997a; Sherman et al., 2003), although few studies have addressed both above and below-ground biomass in their analysis (Tamooh et al., 2008). Where an attempt has been made to measure below-ground biomass of mangrove, studies report variable percentages or ratios of above- to below-ground partitioning with root structures providing up to 60 % of the total biomass (Briggs, 1977; Mackey, 1993; Saintilan, 1997a; Comley and McGuiness, 2005).

Mangrove roots resist decay because they grow in saturated, low oxygen soils (Middleton and McKee, 2001). This is beneficial to the contribution of peat production and therefore to the maintenance of surface elevation in environments with low allochtonous sediment supplies (Briggs, 1977; Cahoon and Lynch, 1997; McKee et al., 2007). However, this characteristic of mangroves will extend the timeframes of ‘rehabilitation’ of mudflats (i.e. a return to a sandy surface) where mangroves have been removed, particularly in locations protected from strong tidal and wave velocities.

The rate at which mangrove root biomass decomposes will be influenced by species (Middleton and McKee, 2001), location and tidal elevation (Huxham et al., 2010), the density of fine and structural roots (Van der Valk and Attiwill, 1984) and nitrogen availability (Huxham et al., 2010). Organic material is known to decay faster at the water sediment interface (Albright 1976), and seasonal differences in decay rates have also been observed (Van der Valk and Attiwill, 1984; Mackey and Smail, 1996). Of the few studies that have investigated the decomposition of below-ground biomass, fewer still have examined decomposition at locations where all root material is unproductive because of dieback or clear-cutting (Albright, 1976). Often, weight loss of root material buried in litterbags is used to estimate decomposition rates (e.g Van der Valk and Attiwill, 1984; Mackey and Smail, 1996; Middleton and McKee, 2001) in preference to destroying above-ground structures to produce complete root mortality.

It would be useful to predict the timeframes required for a site to release mangrove detritus after above-ground structures have been removed. This release could be from partial decomposition and subsequent flushing, or from total decomposition. Before any such predictions can be attempted, some preliminary investigation into the composition of *Avicennia* below-ground biomass in Tauranga Harbour, and the rates at which mangrove roots decompose is vital.

5.3 STUDY OBJECTIVES

The expectation of some New Zealand care groups and coastal managers is that the removal of above-ground mangrove structures from their estuaries will result in rehabilitation (i.e. a return to clear, sandy intertidal flats). If the remaining below-ground biomass is both abundant and temporally persistent, then this expectation may be unrealistic. The production of below-ground biomass by these temperate mangroves must be quantified as a first step in predicting the likely impacts of mangrove removal.

In this study, spatial variability of biomass and morphometric characteristics of both the above-ground and below-ground structures of *Avicennia marina* subsp. *australasica* is investigated and the decomposition of below-ground biomass after mangrove removal is evaluated.

5.4 STUDY SITE

Waikaraka Estuary is a small (0.5 km^2) and narrow, funnel-shaped sub-estuary located on the western boundary of Tauranga Harbour (Lat. 37°40’S, Long. 176°03’E; Figure 5.1). The site experiences semi-diurnal tides with a tidal range of ~ 2 m. All fringing mangrove habitat has a substrate surface containing > 50 % mud, which is inundated on each tide, to depths of up to approximately 0.5 to 0.75 m.

In 2003, mangrove habitat covered approximately 23 % of the Waikaraka Estuary. Mangrove removal, undertaken by local residents, has occurred within the estuary since then, mostly since 2005. The removal has local government approval, and has been undertaken in stages. Only the above-ground vegetation, including pneumatophores, is removed by bruschutters, which ensures water infiltrates into the remaining root material below the surface.

The locations of monitoring sites for this study were largely determined by the mangrove removal activity. Sites 1 and 2 incorporate a zone of mangrove habitat and two adjacent areas that were cleared in May and August 2005. Partial mangrove clearance occurred at Site 3 in March 2006 (Figure 5.1).

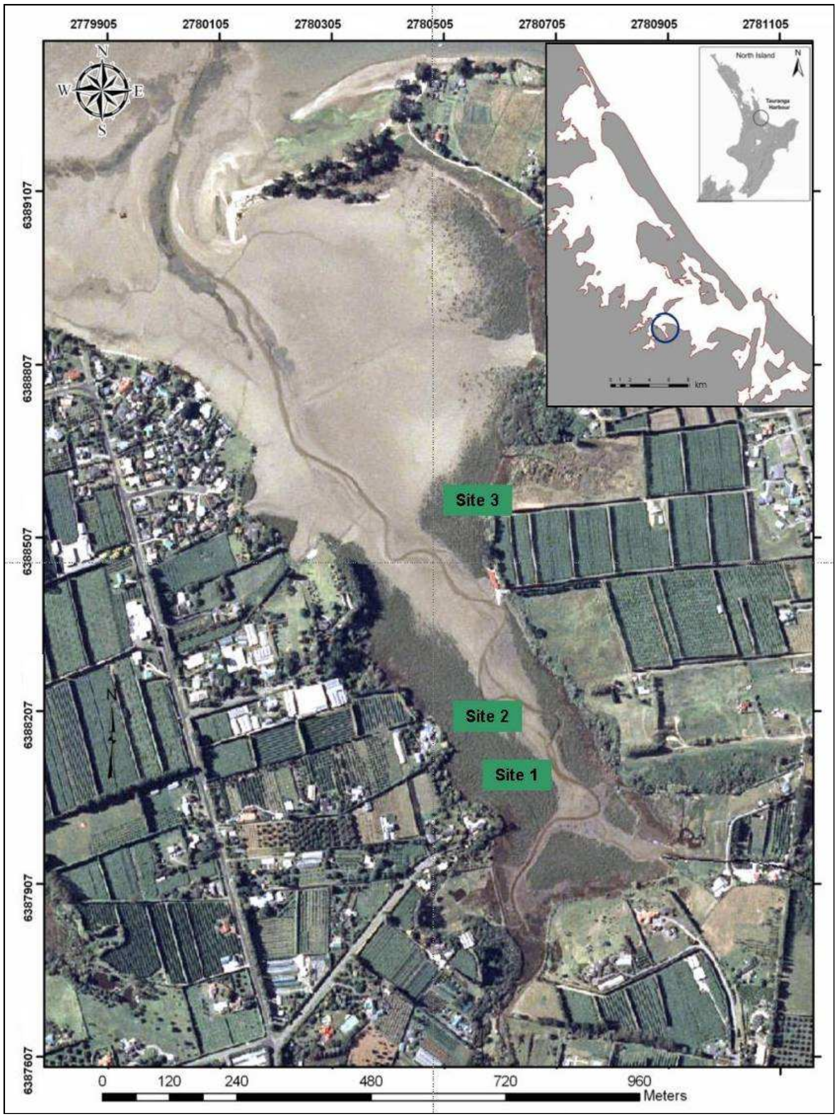

Figure 5.1 **Location of Waikaraka Estuary, positioned along the westerm margins of Tauranga Harbour (inset). Sample collection sites are labeled. Intertidal vegetation consists of monospecific stands of *Avicennia marina*. The aerial photo (courtesy of Environment Bay of Plenty) was taken in 2003 and shows the mangrove stands at Sites 1, 2 and 3 before removal activity took place. NZ Map Grid co-ordinates are included (left and top of image).**

5.5 METHODS

5.5.1 Plant physiognomy

To describe the above-ground forest and plant structure, the following parameters were measured for every tree within three 4 x 4 m plots, at each monitoring site:

1. Plant height;
2. Stem girth;
3. Canopy width of short and long axis.

The plants in Tauranga Harbour are best described as shrubs, and rarely exceed heights of 1.5 m (Stokes et al., 2010), therefore it was not suitable to measure stem diameters at breast height, which is the usual practice (Cintron and Novelli, 1984). Instead, stem girths were measured 5 cm above the substrate, following Woodroffe (1985). Canopy diameter is important as a measure to indicate the extent of crowding within a stand of trees (Cintron and Novelli, 1984). Canopy dimensions were measured directly, across the longest and shortest axis of each tree, and the average of the two represents 'canopy diameter'. Basal area of each plot was calculated using stem diameter, and represents the space covered by tree stems, expressed as m^2 *per* hectare.

5.5.2 Below-ground biomass

An initial pilot study was undertaken in October 2005 to investigate the spatial variability of below-ground biomass. A visiting graduate student undertook much of the field and lab work associated with the pilot study with the aim of dertermining the feasibility of on-going field sampling to quantify biomass in areas where above-ground structures had been removed thus preventing any spatial correlation between plant characteristics (i.e. height, density), distance from trunk etc. Results of the study allowed comparisons between an area that was cleared in May 2005 and a neighbouring area that was cleared in August 2005 in the vicinity of Site 1. The methodology is outlined under 5.5.2a.

A smaller core was used in a subsequent field collection in 2007 to reduce handling time and limit destruction to sampling sites that were also sampled for surface sediments and faunal composition (Chapter 7). This sampling regime included the collection of material from mangrove habitat and cleared areas nearby for a direct comparison of biomass, as outlined in 5.5.2b. Litterbags were also trialled to measure decomposition rates of mangrove roots (5.5.2c).

5.5.2a. Density of dead biomass (cleared zone)

In October 2005, 10 cores were collected across an area cleared of mangroves in May 2005 and August 2005, providing biomass estimates at 2 months post-clearance and 5 months post-clearance. The cores (13 cm diameter) were collected to a depth of 25 cm. In the lab, cores were cut into 5 cm vertical sections and the root material separated from the sediments. Oven-dried structural roots (> 2 mm diameter) and fine roots were weighed separately. Sediments from each vertical section were collected and analysed for total carbon content (% loss after ignition) and grain size (using Malvern LaserSizer). A student T-test was performed to determine any significant difference in mean biomass and grain size between the two post-clearance periods.

5.5.2b. Biomass comparisons between mangrove and adjacent zones cleared of vegetation.

In the southern hemisphere summer (February) 2007, six cores (4.5 cm diameter) were collected to a depth of 20 cm within mangrove habitat, and 6 cores collected from the adjoining intertidal flats where mangroves had been removed at Sites 1, 2 and 3. The sediment from each core was removed using a 1 mm sieve. The remaining plant material was then separated into 'fine roots' (< 2 mm diameter) and structural roots (> 2 mm diameter). The sorted material was dried at 60 °C to a constant weight, and the dry weight was recorded. Differences between sites were analysed using the one-way ANOVA procedure in Statistica, and significant differences were determined from a post-hoc Tukey's tests. Data was log transformed prior to analyses where required.

Due to difficulty in removing the biomass cores, the shorter depth of 20 cm was decided upon whilst in the field. It is likely that results therefore underestimate true values of total biomass, however field observations suggest that the density of root material very quickly declines beyond depths of 20 to 25 cm.

5.5.2c. Litter-bag study of biomass decomposition

Mangrove root material was collected in April 2006, three weeks after above-ground structures had been felled. Fine roots were separated from structural roots (i.e. roots with a diameter of > 2 mm), rinsed with tap water and then air dried to a constant weight. Root litter bags with a 1 mm^2 mesh were then filled with 5 g of fine roots and then placed within a plot located on the tidal flats 20 m south of Site 3, at a depth of 10 cm. Four bags were collected from the plot at intervals extending over 20 months.

5.6 RESULTS

5.6.1 Mangrove physiognomy

Location influenced stand characteristics, with significant differences found between sites for all measured parameters ($p < 0.01$). Overall, plant heights did not exceed 1.2 m, with the shortest stand (mean height of 0.76 m) found at Site 2.

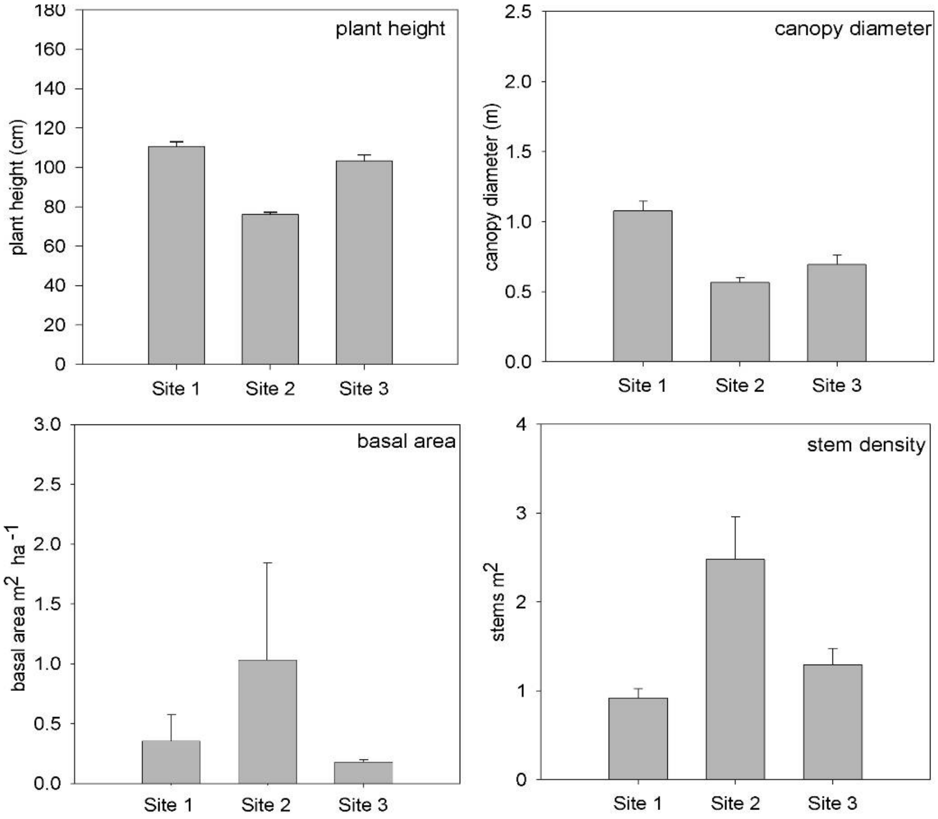

Figure 5.2 Plant characteristics (+ SE) of mangrove habitat within Waikaraka Estuary: a) vertical plant height cm; b) canopy diameter m; c) basal area m^2 ha^{-1}; d) stem density $^{-}m^2$.

This mid-estuary mangrove area also possessed the smallest canopy diameter of < 0.6 m, but the highest stem density (>2 m^{-2}), and corresponding basal area (Figure 5.2). Tallest mean tree height of 1.1 m was measured at Site 1, where the stand was found to possess a larger mean canopy diameter of 1.1 m, and a lower stem density of < 1 m^{-2}.

5.6.2 Mangrove below-ground biomass

Below-ground biomass of live mangroves ranged from 2.1 kg m^{-2} at Site 3, to 3.5 kg m^{-2} (Figure 5.3). Sites 1 and 2 measured similar mean biomass of 3.5 kg m^{-2}, however Site 1 appeared to contain comparatively lower structural root mass (0.85 kg m^{-2} ± 0.3). Root material collected within the three mangrove sites was dominated by 60 to 70% fine root material.

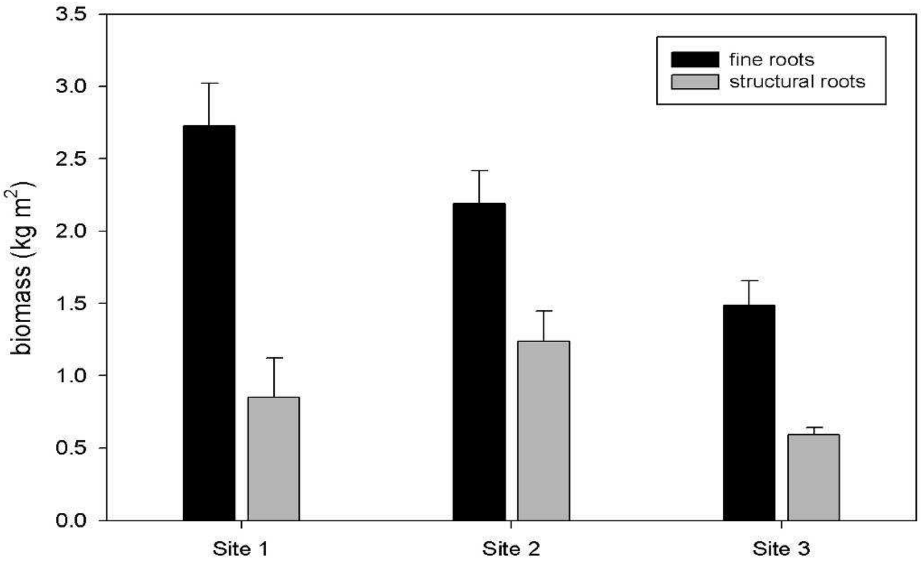

Figure 5.3 **Average biomass (+ SE) separated into fine roots (< 2 mm diameter) and structural roots (> 2 mm diameter) from sediment cores collected in mangrove habitat, February (summer) 2007. n = 6.**

5.6.3 Comparison of mangrove and neighbouring cleared habitat

Spatial variability in biomass collected from Sites 1, 2 and 3, in February 2007, is evident in Figure 5.4, which displays the contribution of fine and structural roots to sediments of both the mangrove and neighbouring cleared habitats at these locations. All sites show less biomass in the cleared area, when compared with the adjacent mangrove habitat, with Site 2 and Site 3 showing significant differences in a T-test ($p < 0.05$ and < 0.01 respectively). The only significant difference in total below-ground biomass found in an ANOVA post-hoc Tukey test, however, was between the cleared zones of Site 1 and Site 3 ($p < 0.01$).

Apparent decomposition can be loosely interpreted from biomass differences between vegetated and cleared habitat. Results suggest the greatest decomposition occurred at Site 3 where approximately 50% less biomass was evident in cores in the cleared site compared to the neighbouring mangrove habitat. Conversely, apparent loss between mangrove and cleared habitat at Site 1 was < 10% (Figure 5.4).

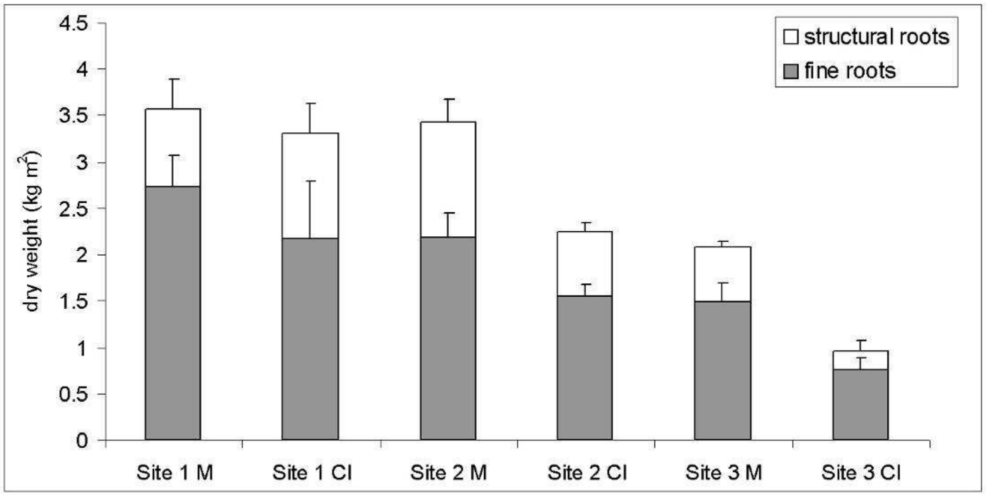

Figure 5.4 Below-ground biomass (+SE) of mangrove habitat (M) and neighbouring intertidal flats cleared of above-ground vegetation (Cl). Cores collected in February 2007 (n=6).

5.6.4 Morphometrics of dead (cleared) below-ground biomass

An investigative sampling regime was undertaken in 2005 to identify spatial variability of below-ground biomass across one site that had been clear-felled. The location had been cleared in two stages, 3 months apart. Results could therefore be interpreted as decomposition 2 months and 5 months after tree mortality. Below-ground biomass differences were significant ($p < 0.01$) between locations. An average of 4.7 kg m^{-2} ($\pm$ 0.4) was found across cores nominated '2 months post clearance', compared to 2.5 ($\pm$ 0.24) kg m^{-2} for the '5 months after post-clearance' location (Figure 5.6a). This observation was coupled with a significant increase in mean grain size, from silt to fine-sand at depths of 0-5 cm ($p < 0.001$) and 5–10 cm ($p < 0.05$) (Figure 5.6b).

The proportion of root biomass with depth (Figure 5.6c) shows some variability, although there is no consistent decline in biomass with depth. The greater mass of root material occurred within the top 5 cm of the core, and also at 15-20 cm below the surface. It was found that fine root material contributed < 50 % of the total biomass (Figure 5.6d). The highest total organic content (TOC) from Loss on Ignition was found in sediments at a depth of 15-20 cm (13% ± 4), whereas TOC measured in the overlying sediments ranged from ~ 6 to 8% (Figure 5.6e).

5.6.5 Litterbag results

Linear regression identified patchy results across the 20 month sampling period and no significant trend of litter weight loss associated with decomposition time could be identified (Figure 5.5). An apparent linear decrease in biomass for the first 3 months (up to 13%) is thereafter skewed by variable results for the remaining 17 months.

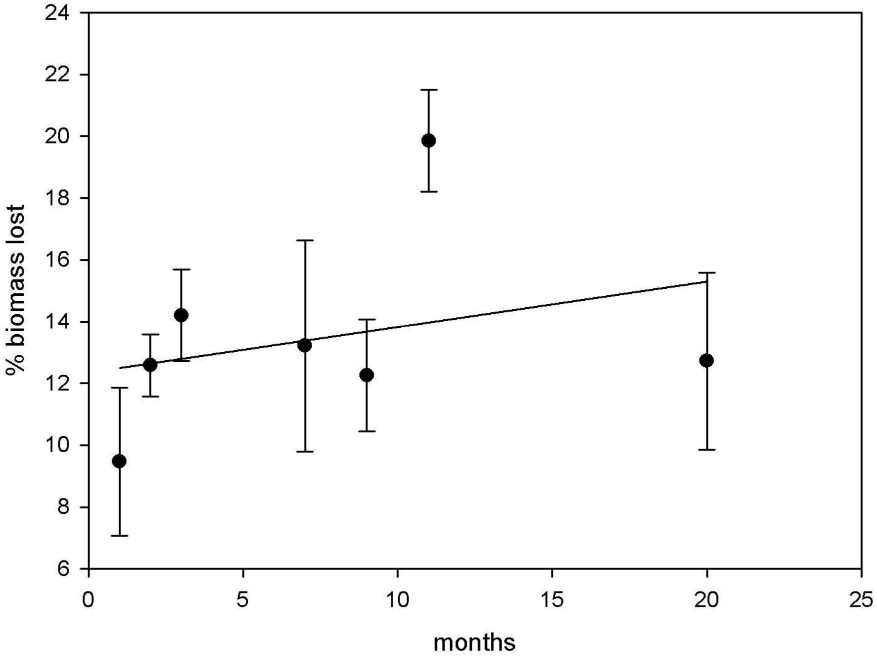

Figure 5.5 **Relationship between percentage of dry weight (± SE) of mangrove roots lost and time in months after burial in litterbags 10 cm below the surface (n = 4). R^2 = 0.09**

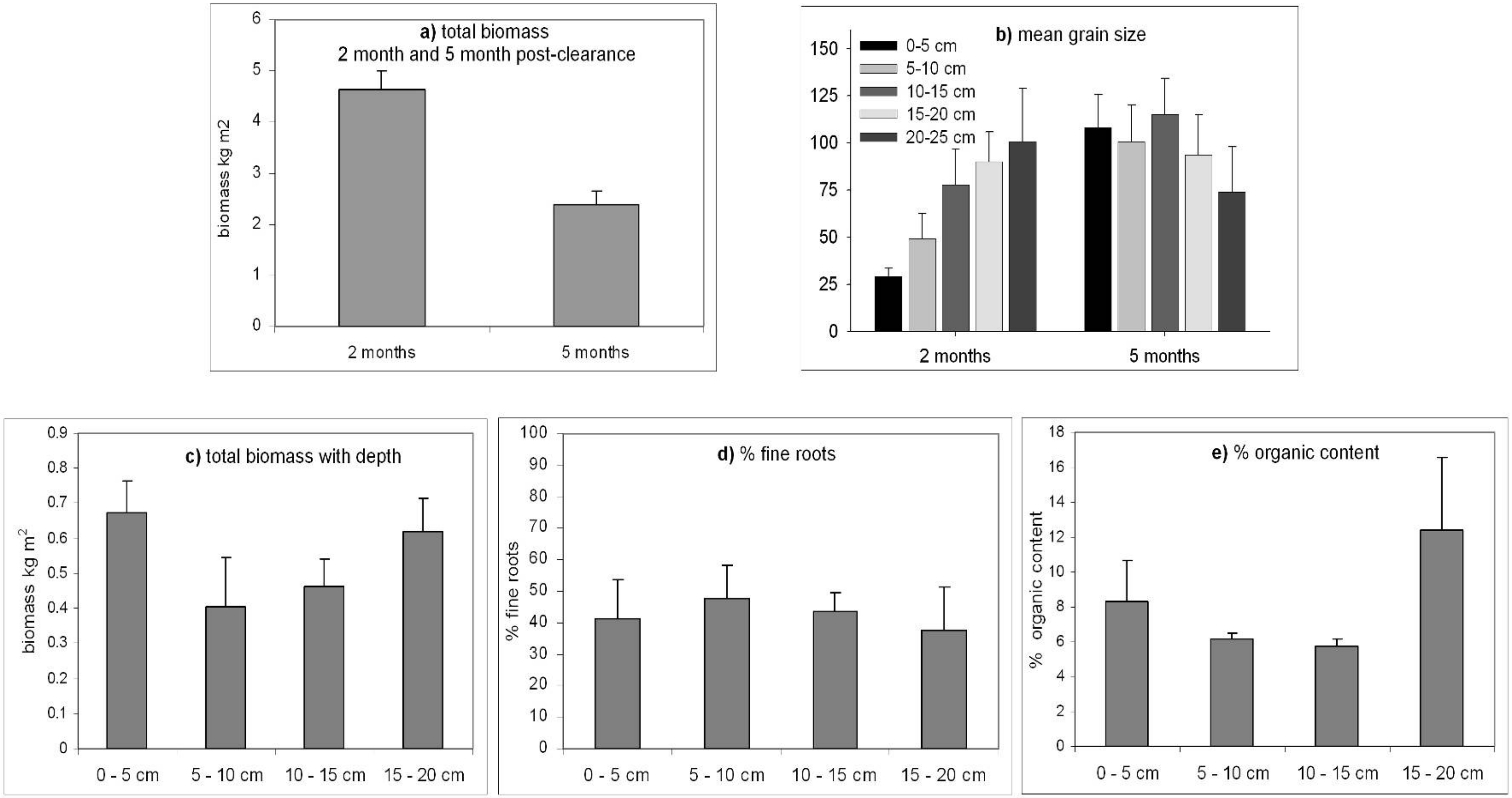

Figure 5.6 (a) Total below-ground biomass (dry weight kg m^{-2}) from cores collected to a depth of 20 cm in an area cleared of mangroves 2 months previous, and 5 months previous to collection; (b) mean grain size (microns) of sediments collected in biomass cores; (c) total biomass (dry weight) found in 5 cm vertical sections of 5 month post-clearance cores (n = 5); (d) % fine roots of biomass with depth; - 5 month post-clearance (n = 5) (e) % total organic content of sediments with depth for 5 month post-clearance cores (n = 5).

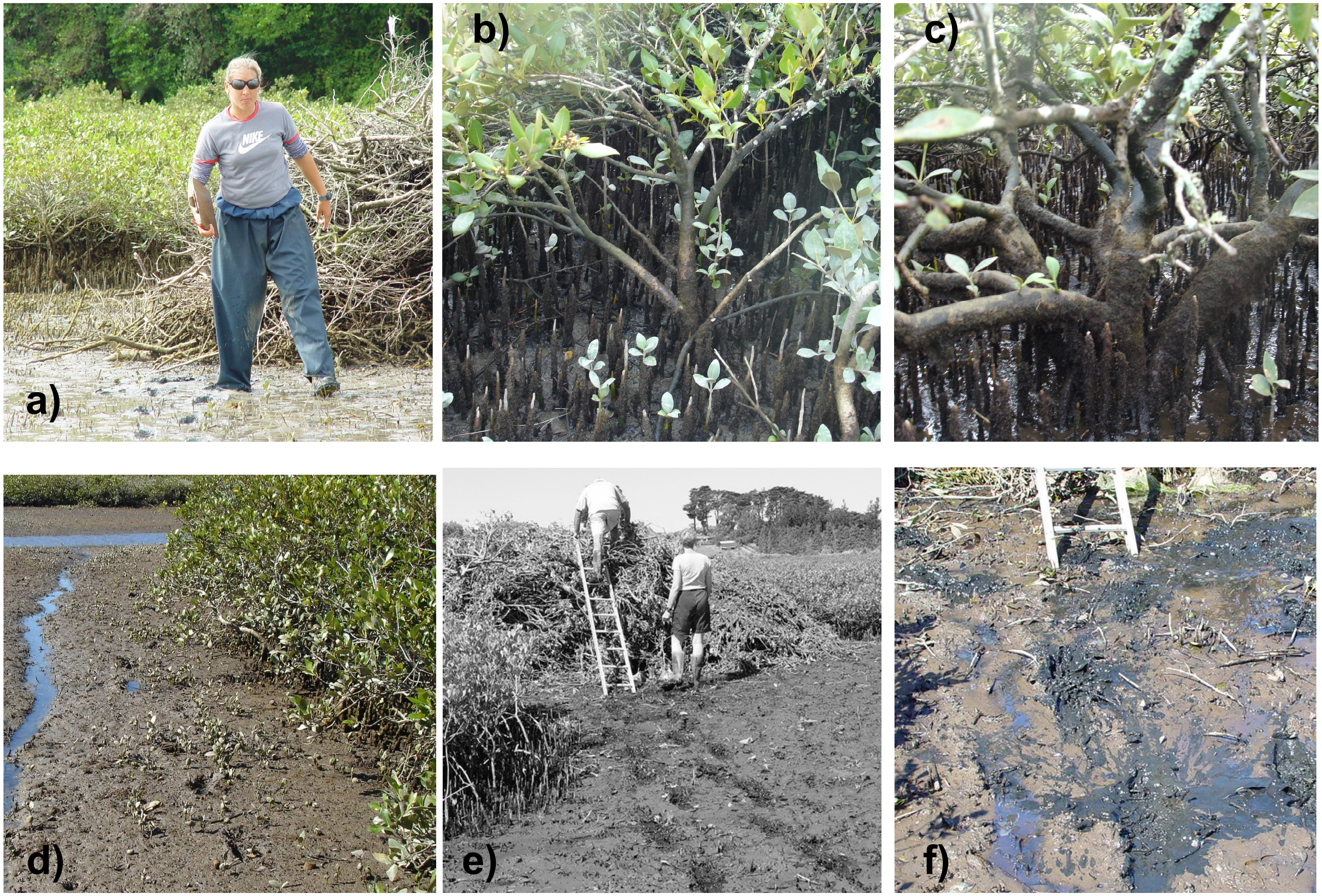

Figure 5.7 **Images: (a) Mangrove stand behind a pile of debris from mangrove clearing activity (April 2006); (b) Typical plant structure of 10-20 year old plants at Site 3 – mean heights approx. 0.7 m; (c) Typical plant structure of older plants closer to landward margins; (d) cleared habitat at Site 1, approximately 18 months after clearance; (e) cleared debris piles, 2005; (f) anoxic black muds and root material of cleared habitat evident in footprints (2005).**

5.7 DISCUSSION

The mangrove populations in Waikaraka Estuary consist of densely populated ‘low’ trees of 1 to 1.25 m, and ‘stunted’ plants of < 1 m, following physiognomic descriptions from mangroves elsewhere in New Zealand (Kuchler, 1972; Woodroffe, 1985).

Below-ground biomass of mangrove populations in Waikaraka Estuary ranged from approximately 20 t ha^{-1} to 40 t ha^{-1} (2 – 4 kg m^{-2}). This is far from the higher end values of >200 t ha^{-1} reported for tropical primary mangrove forests (e.g. Komiyama et al., 1987). *Avicennia* populations near Sydney with tree heights of 6–7 m were estimated to accommodate 147 and 160 t ha^{-1} of below-ground biomass including pneumatophores (Briggs, 1977), similar to 109 – 126 t ha^{-1} reported by Mackey (1993) for an Australian site at a lower latitude. A further study of sub-tropical *Avicennia* reported below-ground biomass estimates of 30 to 80 t ha^{-1} (Saintilan, 1997a). At first glance this latter finding appears similar to this study, however direct comparison is complicated by the different approach used to determine below-ground biomass, whereby Saintilan (1997a) separated live and dead root material to produce an estimate of living biomass, while elsewhere (and in this study) all root material was included in the measurements (Briggs, 1977; Komiyama et al., 1987; Mackey, 1993).

The low values of below-ground biomass reported in this study are not surprising, considering the lower growth form of the populations, the higher latitude which would influence photosynthetic productivity (Beard, 2006), as well as the young age of most of the trees (Cintron and Novelli, 1984; Mackey, 1993; Komiyama et al., 1987). Allometric studies of mangrove biomass allude to a relationship between above-ground and below-ground biomass, although the reported ratio’s of such have been found to vary because of differences in conditions such as salinity (Saintilan, 1997a; Saintilan, 1997b; Sherman et al., 2003), nutrient supply (Saintilan 1997b) and tree age (Komiyama et al., 1987; Mackey, 1993; Tamooh et al., 2008). With this in mind, it seems reasonable to expect that short trees with narrow canopy diameters would produce relatively lower biomass, and indeed a strong correlation between canopy diameter and biomass for low trees was identified by Woodroffe (1985).

5.7.1 Decomposition of below-ground biomass

Three methods were used in this study to investigate the decomposition of mangrove below-ground biomass. Interestingly, results from each of the three methods provided different estimates.

Results from the litterbag study suggest an initial loss in biomass of 14% within 3 months, however this apparent trend in the data is subsequently blurred by variable and inconsistent values of biomass loss across the remaining 17 months of the study. Similarly, Van der Valk and Attiwill (1984) recorded intial weight loss of fine roots over the first 40 days after which no more weight loss was detectable, a trend which was also observed by Albright (1976). A similar process was reported by Woodroffe (1985) when measuring decay of mangrove leaves which were found to lose half their weight rapidly and then degrade at a much slower rate.

Van der Valk and Attiwill (1984) suggests there is some error inherent in litterbag studies, however the 15% loss in fine roots over a 270 day period reported therein contrasts greatly to the apparent 14% loss in 90 days recorded in Waikaraka Estuary. Fine roots appear to decay more quickly than main roots (Van der Valk and Attiwill, 1984), and the exclusion of the main structural roots from the litterbags in this study may partially explain the initial high rate of decay.

The second approach used in this study was to collect sediment cores within two adjoining cleared zones. One zone was cleared 2 months prior to core sampling, while the neighbouring seaward zone was cleared 5 months prior to sampling. As such, results provide an indication of biomass loss over a 3 month period. The results suggest that 52% of below-ground biomass was lost over that 3 month period, which appears to be unusually high when compared to other studies (Albright, 1976; Van der Valk and Attiwill, 1984). This could be explained by the fact that the cores were collected on the same day, in two different plots, rather than one plot revisited two months, then again five months after mangroves were removed. Therefore, these core results provide a comparative result only and age differences in the mangroves that were cleared could be partially responsible for this result.

The third field method identified comparative differences of below-ground biomass between core samples collected in existing mangrove habitat, and those collected within adjoining cleared zones. The results provide a snapshot of spatial trends in both mangrove below-ground biomass, and biomass degradation. The percentage difference (total dry weight) between cleared and vegetated habitat ranged from 8% at Site 1 (18 months post-clearance) to 54 % at Site 3 (11 months post-clearance). In comparison, Albright (1976) found that after 7 years, a patch of dead mangroves had lost 69% of its roots and 55% of its pneumatophores, extrapolated out to a degradation rate of 12% per year. Middleton and McKee (2001) reported mangrove tissue degradation rates of 0.098 % loss day^{-1}, which is roughly in the middle of Albright's estimate and the apparent degradation of >54% per year reported in this study (Site 3). The mangrove stands in Waikaraka Estuary experience semi-diurnal tidal inundation, and it is possible that daily inundation provides sufficient surface flushing of decomposed organic matter to promote faster decomposition of the remaining material at the water/sediment interface (Albright, 1976). It is also possible that the cleared areas had less biomass to start with, when compared to the adjacent remaining mangroves, as a consequence of their location seaward of the nominated 'mangrove habitat' which would therefore deem them relatively younger.

Within areas cleared of above-ground vegetation in Waikaraka Estuary, surface elevation fell at an average rate of 14 mm yr $^{-1}$ (Stokes et al., 2009 – Chapter 4). Substrate collapse has been observed after mass mangrove mortality from hurricane activity, however at a slightly lower rate of 11 mm yr^{-1} (Cahoon et al., 2003). Cahoon et al. (2003) suggest the topographical change was driven by subsidence of the mangrove peat, whereas in Waikaraka some sediment erosion is occurring, as evidenced by an increase in grain size over time. Elsewhere, deficits in surface accretion have been linked to shifts in groundwater during drought conditions (Rogers et al., 2005). It is likely that some root compaction is also occurring in Waikaraka, however the separation of erosional and subsidence processes were not attempted in this study.

5.8 CONCLUSIONS

This study has investigated the below-ground biomass of a developing, temperate mangrove system. Low mean plant height and small canopy diameter is reflected in low values of below-ground biomass. This can be explained partly by the growth-limiting climate, while site differences of biomass can be attributed to stand age. Spatial variability of apparent decomposition rates was evident from results of core analyses. Decomposition rate estimations in this study were mostly higher than has been reported elsewhere, and this may be attributed to a combination of daily tidal inundation, and low initial biomass. However, it could be expected that it will be a number of years before all below-ground biomass of felled mangrove habitat will degrade in Waikaraka Estuary, and this will influence rates of fine sediment release, the contribution of dissolved organic carbon, and the resultant intertidal topography.

5.9 ACKNOWLEDGEMENTS

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5.10 LITERATURE CITED

Albright L., 1976. In situ degradation of mangrove tissue (note). *New Zealand Journal of Marine and Freshwater Research* 10(2), 385-389.

Beard C., 2006. Physiological constraints on the latitudinal distribution of the mangrove *Avicennia marina* (Forsk.) *Vierh*. subsp. *australasica* (Walp.) J. Everett in New Zealand. PhD Thesis, University of Waikato, 203 p.

Briggs S.V., 1977. Estimates of biomass in a temperate mangrove community. *Australian Journal of Ecology* 2, 369-373.

Cahoon, D. R. and Lynch, J. C., 1997. Vertical accretion and shallow subsidence in a mangrove forest of southwestern Florida, U.S.A. Mangroves and Salt Marshes 1, 173-186.

Cahoon D.R., Hensel P., Rybczyk J., McKee K.L., Proffitt C.E. and Perez B.C. 2003. Mass tree mortality leads to mangrove peat collapse at Bay Islands, Honduras after Hurricane Mitch. *Journal of Ecology* 91, 1093-1105.

Cintron G., and Novelli Y.S., 1984. Methods for Studying Mangrove Structure. In: Snedaker, S.C. and Snedaker J.G. (Eds.), *The mangrove ecosystem: research methods*. Paris, UNESCO**,** pp. 91-114.

Comley, B. W. T. and McGuiness, K. A., 2005. Above- and below-ground biomass and allometry of four common northern australian mangroves. *Australian Journal of Botany* 53, 431-436.

Fromard, F., Puig, H., Mougin, E., Marty, G., Betoulle, J. L. and Cadamuro, L., 1998. Structure, above-ground biomass and dynamics of mangrove ecosystems: new data from French Guiana. *Oecologia* 115(1-2), 39-53.

Hutchings P., and Saenger P., 1987. *Ecology of Mangroves*. St Lucia, University of Queensland Press, 369 p.

Huxham, M., Langat, J., Tamooh, F., Kennedy, H., Mencuccini, M., Skov, M. W. and Kairo, J., 2010. Decomposition of mangrove roots: effects of location, nutrients, species identity and mix in a Kenyan forest. *Estuarine, Coastal and Shelf Science* 88(1), 135.

Jennerjahn, T. C. and Ittekkot, V. 2002. Relevance of mangroves for the production and deposition of organic matter along tropical continental margins. *Naturwissenschaften* 89(1), 23-30.

Komiyama, A., Ogino, K., Aksornkoae, S. and Sabhasri, S. 1987. Root biomass of a mangrove forest in southern Thailand. 1. Estimation by the trench method and the zonal structure of root biomass. *Journal of Tropical Ecology* 3, 97-108.

Komiyama A., Ong J.E. and Poungparn S., 2008. Allometry, biomass, and productivity of mangrove forests: a review. *Aquatic Botany* 89, 128-137.

Kuchler, A. W. 1972. The Mangrove in New Zealand. *New Zealand Geographer* 28, 113-129.

Lovelock, C., Feller, I., McKee, K., Engelbrecht, B. and Ball, M., 2004. The effect of nutrient enrichment on growth, photosynthesis and hydraulic conductance of dwarf mangroves in Panamá. *Functional Ecology* 18(1), 25.

Lovelock C.E., Feller I.C., McKee K.L. and Thompson R., 2005. Variation in mangrove forest structure and sediment characteristics in Bocas del Toro, Panama. *Caribbean Journal of Scienc*e 41, 456-464.

Lovelock, C., 2008. Soil respiration and belowground carbon allocation in mangrove forests. *Ecosystems* 11, 342-354.

Mackey A.P., 1993. Biomass of the mangrove *Avicennia marina (*Forsk.) Vierh. Near Brisbane, south-eastern Queensland*.* *Marine and Freshwater Research* 44(5), 721-725.

Mackey, A. P. and Smail, G. 1996. The decomposition of mangrove litter in a subtropical mangrove forest. *Hydrobiologia* 332(2), 93-98.

McKee, K. L., 1993. Soil physicochemical patterns and mangrove species distribution - reciprocal effects? *Journal of Ecology* 81, 477-487.

McKee K.L., Cahoon D.R. and Feller I.C., 2007. Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. *Global Ecology and Biogeography* 16, 545-556.

Middleton B.A. and McKee K.L., 2001. Degradation of mangrove tissues and implications for peat formation in Belizean island forests. *Journal of Ecology* 89, 818-823.

Putz, F. E. and Chan, H. T., 1986. Tree growth, dynamics, and productivity in a mature mangrove forest in Malaysia. *Forest Ecology and Management* 17(2-3), 211-230.

Rogers, K., Saintilan, N. and Cahoon, D. 2005. Surface elevation dynamics in a regenerating mangrove forest at Homebush Bay, Australia. *Wetlands Ecology and Management* 13(5), 587-598.

Saenger, P. and Snedaker, S., 1993. Pantropical trends in mangrove above-ground biomass and annual litterfall. *Oecologia* 96(3), 293-299.

Saintilan N., 1997a. Above- and below-ground biomass of mangroves in a sub-tropical estuary. *Journal of Marine and Freshwater Research* 48(7), 601-604.

Saintilan, N., 1997b. Above- and below-ground biomasses of two species of mangrove on the Hawkesbury River estuary, New South Wales. *Marine and Freshwater Research* 48, 147-152.

Sherman, R. E., Fahey, T. J. and Martinez, P., 2003. Spatial patterns of Biomass and aboveground net primary productivity in a mangrove ecosystem in the Dominican Republic. *Ecosystems* 6(4), 384-398.

Stokes D.J., Healy T.R., and Cooke P.J., 2009. Surface elevation changes and sediment characteristics of intertidal surfaces undergoing mangrove expansion and mangrove removal, Waikaraka Estuary, Tauranga Harbour, New Zealand. *International Journal of Ecology and Development* 12(9), 88-106.

Stokes D.J., Healy T.R., and Cooke P.J., 2010. Expansion dynamics of monospecific, temperate mangroves in two embayments of a barrier-enclosed lagoon, Tauranga Harbour, New Zealand. *Journal of Coastal Research 26(1), 113-122.*

Tamooh, F., Huxham, M., Karachi, M., Mencuccini, M., Kairo, J. G. and Kirui, B., 2008. Below-ground root yield and distribution in natural and replanted mangrove forests at Gazi bay, Kenya. *Forest Ecology and Management* 256(6), 1290.

Woodroffe C., 1985. Studies of a Mangrove Basin, Tuff Crater, New Zealand: I. Mangrove Biomass and Production of Detritus. *Estuarine Coastal and Shelf Science* 20, 265-280.

Van Der Valk A.G. and Attiwill P.M., 1984. Decomposition of leaf and root litter of *Avicennia marina* at Westernport Bay, Victoria, Australia. *Aquatic Botany* 18 205-221.

Chapter 6
Implications for the future: Waikaraka Estuary

6.1 PREFACE

Chapters 3 and 4 have discussed mangrove expansion and surface accretion. In this chapter the influence of tidal currents and sediment loads on the scale of this geomorphic process is investigated. The data presented in previous chapters is used here to underpin a conceptual model of estuarine geomorphology resulting from continued mangrove expansion. An alternative evolution for Waikaraka Estuary is on-going mangrove removal, and estimates of sediment loads that could be introduced into the sediment transport system are provided. The implication of sea-level rise on the sustainability of the wetlands is also explored.

This chapter contributes to achieving the following thesis objectives:

- *To determine the physical changes that have occurred in response to the removal of mangrove vegetation; and*
- *To explore the effects of on-going mangrove expansion on sedimentation and geomorphology.*

6.2 INTRODUCTION

Estuaries are receiving environments for terrestrial sediments, and therefore are vulnerable to the effects of increased sediment loads generated by catchment practices such as forestry and urbanisation. Estuaries are natural sediment sinks (Healy et al., 1996), and for many sites in New Zealand physical changes have increased in pace and extent due to human induced land-use changes (Hume and McGlone, 1986). The increase in sediment to these estuaries can initiate changes in intertidal topography and sedimentology, with potentially more intertidal area experiencing fewer and shallower tidal inundations (e.g. Swales et al., 2007). Reduced tidal inundation frequency and/or inundation height can initiate vegetation colonisation which then accelerates morphological progression (Thom et al., 1975). As such, changes in the distribution and mosaic of estuarine vegetation may reflect large-scale modification to the sediment delivery regime and/or hydrodynamics.

Over the last 50 - 60 years, declining seagrass coverage and increasing mangrove coverage has occurred in many temperate and sub-tropical estuaries of New Zealand. These changes are often identified using aerial photographs therefore limiting the reference timeframes to the first photographs, which are generally from the 1940s. At a number of localities, the loss of seagrass habitat has been attributed to increased sedimentation and turbidity, and increased nutrient loading (Turner and Schwarz, 2006). In Tauranga Harbour, 34% of the seagrass beds (predominantly subtidal) disappeared between 1959 and 1996 (Park, 1999) while an exponential increase in mangrove habitat was observed over a similar period (Park, 2004).

Pethick (1981) suggested a negative feedback mechanism exists in wetland systems whereby increased sediment inputs result in an increase to the marsh surface elevation. This in turn decreases tidal inundation which then causes a decrease in sediment accretion. Where sediment supply is abundant, an estuary that has evolved to 'maturity' will exhibit expansive mangrove that are dissected by tidal creeks. Landward margins become dryer and plant succession moves to shrubbier mangroves and saltmarsh species (Thom, 1975). Sedimentation processes will alter at this stage, with sub-tidal channels funneling sediments out to the coast (Thom, 1975; Woodroffe, 1992). Continued supply of fresh sediments to estuary wetlands can then only occur when tidal waters overtop the creek banks (Thom, 1975).

The scale and speed of coastal evolution where mangrove are found is dependent on water depth, tidal current velocity, initial suspended sediment concentrations and the morphology of the mangrove forest (Massel et al., 1999). Field investigations, laboratory experiments and modeling exercises have identified the following key relationships between mangrove forests and sedimentation:

- Current velocities of tidal water moving through mangrove forests rarely exceed 0.1 m s^{-1} (Furukawa and Wolanski, 1996), well below the 0.3 m s^{-1} required to re-suspend fine sediments (Wolanski et al., 1995), meaning once sediment has settled in mangrove vegetation, it is rarely re-suspended.

- Suspended sediment loads in the tidal water decreases in mangrove forests as it progressively settles out. Furukawa and Wolanski (1996) reported that 50 % of the suspended sediment settled out within 25-30 m of the seaward/tidal-channel edge of the mangrove stand. Findings reported by Victor et al. (2006), are similar to an exponential decrease in sedimentation rates with distance from the tidal creek, identified by Furukawa et al. (1997).
- Fine sediment remains in suspension until high tide (Van Santen et al., 2006), due to turbulence created by dense, protruding mangrove root structures (Furukawa and Wolanski, 96); with sediment deposition then taking place during slack water.
- Density of forest structures will influence dissipation of energy – e.g. model results of Massel et al. (1999) identified an increase in wave attenuation with greater density of above-ground forest structures.
- Tidal inundation height will influence the distance wave energy is transmitted (Massel et al., 1999). For example, Phuoc and Massel (2006) found that twice the distance through a mangrove forest had to be covered to dissipate 50% of the wave energy, if maximum inundation height exceeded 2.1 m, compared to wave energy transmission when water heights peaked at < 2 m.

Many studies that quantify the relationship between coastal wetland vegetation and sedimentation have explored the potential impact of future sea-level rise on the sustainability of these systems. Eustatic sea-level is projected to rise between 20 and 60 cm during the 21st century (IPCC, 2007). The coast of New Zealand has seen an average sea-level rise of between 1.6 and 1.77 mm yr^{-1} over the past 100 years (Bell et al., 2000; Hannah, 2004), which is close to the global trend (Hannah, 2004). More recently sea-level rise (SLR) around New Zealand has slowed, with a 3.3 mm ($\pm$ 0.4) SLR calculated over the past 15 years (Hannah, 2004). Regardless, some locations within New Zealand that experience strong subsidence will be at greater risk from future SLR (Hannah, 2004). It is largely unknown if Tauranga Harbour is subsiding at any significant rate, indeed there are opposing views as to whether the harbor is experiencing subsidence or uplift (Shepherd et al., 1997). The Rangitiki Plains coastline, south of the Tauranga

Harbour basin, appears to be subsiding at 0.4 – 2 m/1000 years, or an average of 2 mm yr^{-1} (Gibbs et al., 1992). The Plains are situated closer to the Wellington Fault so it is probable that the Tauranga basin, which is further north, would be subsiding at a slower rate, if at all.

The stability of mangrove and saltmarsh populations in light of such projections depends on their ability to maintain surface elevations relative to sea-level (Cahoon et al., 1999; McKee et al., 2007). Vertical accretion must therefore keep pace with relative sea-level rise (eustatic sea-level plus any local subsidence) (Cahoon et al., 1995). Vulnerability to sea-level rise will be variable depending on the extremity of changes to freshwater, sediment and nutrient delivery into a wetland system (Day et al., 2008). It has been suggested that coastal wetlands could adjust to predicted climate change, but in combination with human impacts wetlands may be significantly affected (Day et al., 2008).

6.3 STUDY OBJECTIVES

Waikaraka estuary, similar to other embayments within Tauranga Harbour, is evolving in response to increased sedimentation and the progradation of mangrove. This chapter explores the interactions of mangrove colonization and sediment transport as the estuary evolves. Observations of sedimentation and mangrove dynamics within Waikaraka Estuary are used to forecast the potential impact of 50 years of continued mangrove encroachment. Conversely, the potential consequences of mangrove removal on stability of the remaining wetland are discussed with a focus on the additional stressor of sea-level rise.

6.4 STUDY AREA

Tides at the entrance of Waikaraka Estuary have been measured as meso-tidal, ranging from 2.1 m at spring tides, to 1.4 m during neap tides, with the tidal range decreasing to 0.6 - 0.7 m in the upper estuary tidal channel (Hope, 2002). Mangrove stands in the middle and upper estuary are inundated only during the final stage of high tide and the mangrove and cleared plots closer to the estuary mouth (Site 4, see Figure 6.1) are covered 30 to 45 minutes earlier. Generally, inundation within the mangrove habitat is < 1 m mid-estuary and < 0.5 m in the upper estuary. The mangrove forest floor is positioned at or above 0.3 m Mean

Sea Level (Moturiki datum) and the unvegetated sandflats are mostly at MSL (Park, 2004).

Data presented in Chapter 4 uderpin a conceptual model which is outlined in Section 6.7. Below is a brief summary of the key findings detailed in Chapter 4, and the main characterstics of Waikaraka Estuary:

- Waikaraka Estuary has an intertidal area of 0.5 km^2;
- Mangrove coverage increased from < 1.5 ha in 1943 to 11.5 ha in 2003;
- Between 2003 and 2009 10% of that mangrove vegetation was cleared, leaving below-ground biomass in-situ;
- Average surface accretion measured within mangrove habitat was 3 mm yr^{-1};
- Surface elevation change following the removal of mangroves averaged -4 mm yr^{-1};
- Sediment accumulation (from trap results) shows variable sediment loads. Site 4 receives between 2,200 – 6,000 g m^2 mo^{-1} seaward of the mangrove fringe, and 1,100 to 5,000 g m^2 mo^{-1} within mangrove vegetation.

6.5 TIDAL DYNAMICS AND SUSPENDED SEDIMENT TRANSPORT: *Waikaraka Estuary*

6.5.1 Methods

Tidal currents, water depth, and suspended sediment concentrations (SSC) were recorded at Site 4. Station A was positioned 5 m seaward of the mangrove fringe, and Station B located 5 m into the mangrove vegetation to provide an 'inside' and 'outside' mangrove comparison. The depth of tidal flooding was recorded using DOBIE pressure sensors, and Optical Backscatter Sensors (OBS) measured SSC. The OBS sensors were positioned 10 cm above the bed and any mangrove pneumatophores within 0.2 m of the sensor were removed to reduce interference. OBS were calibrated in a settling tank, using sediment collected at the site. The OBS's and DOBIE's were programmed to burst every 8 minutes, for a two minute duration, at 5 Hz. Tidal currents were measured using a Triton Acoustic Doppler Velocimeter (ADV) with sensors also positioned 10 cm above the bed. The

ADVs were programmed to burst every minute. The deployment covered a spring tide phase from 21 – 24 December, 2007. The standard error of the ADV's is recorded with a 0.1 cm sec^{-1} resolution of the mean velocity (SonTek 2003).

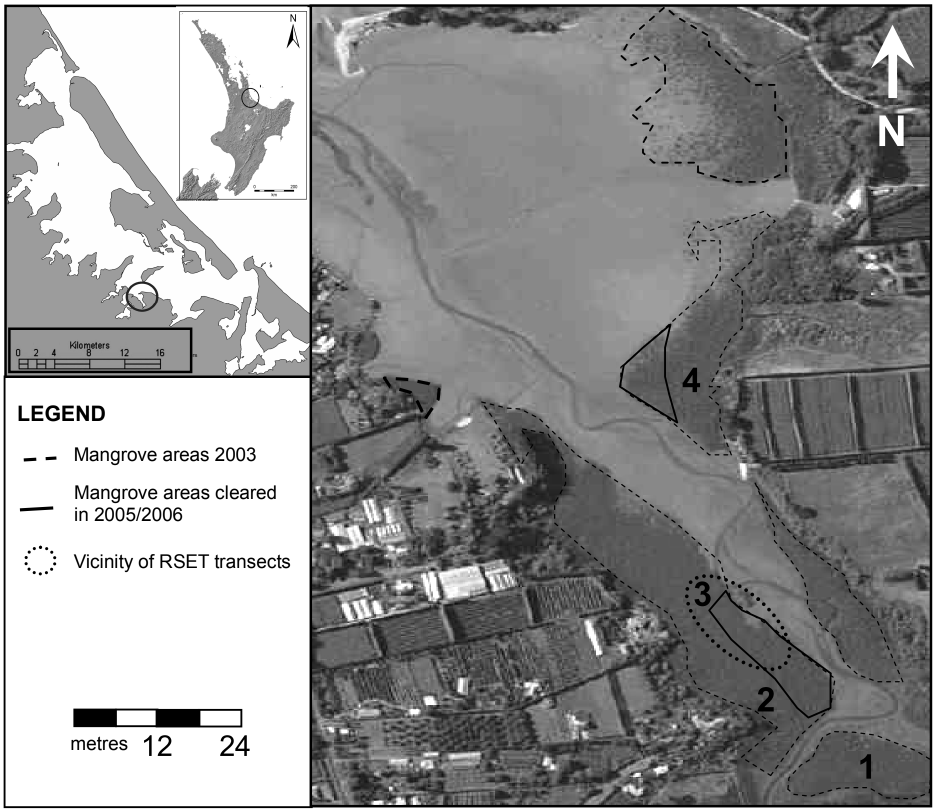

Figure 6.1 **The study area of Waikaraka Estuary (right), positioned within Tauranga Harbour, North Island, New Zealand (upper left). Monitoring site locations are numbered, and areas cleared of mangroves are outlined.**

6.5.2 Results

The semi-diurnal tidal oscillations were measured during a spring tidal phase, with maximum water depths ranging from 0.7 – 0.8 m, at both Station A (outside mangroves) and Station B (inside mangroves). Inundation periods measured by DOBIE pressure sensors ranged from between 230 and 280 minutes per tide (Figure 6.2).

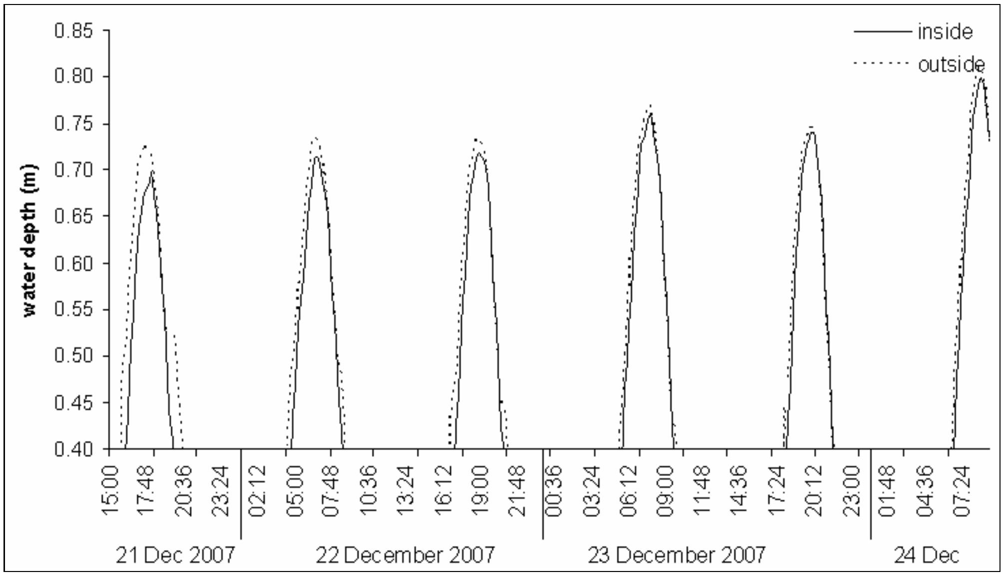

Figure 6.2 **Water depths measured by DOBIE's during tidal inundations for the spring tide cycle from the 21/12/2007 to 24/12/2004, inside and outside mangroves.**

The dampening effect of mangrove vegetation is evident, with reduced tidal currents recorded inside the mangroves. Results of the ADVs suggest weak, calm-weather current speeds on both the flood and ebb tide at both locations. These currents were mostly well below the 0.3 m sec^{-1} required for fine sediment re-suspension (Wolanski et al., 1995). Current speeds within the mangroves were typically less than 0.01 m sec^{-1}, compared with currents entering the mangroves, which were generally between 0.01 and 0.02 m sec^{-1} (Figure 6.3). It is important to note that the velocities measured by the ADV's are close to the 0.001 m sec^{-1} resolution of the standard error of the instruments, although no device was available to measure tidal velocities in the field at a higher resolution.

Suspended sediment concentrations, were typically < 75 mg l^{-1}. An exception to this observation occurred on the first day of monitoring, when the OBS positioned on the unvegetated mudflat reached saturation at ~ 600 mg l^{-1} which was maintained for the duration of the flood and ebb tide. SSC inside the mangrove habitat ranged mostly between 20 – 25 mg l^{-1}, and a modest decrease (< 20 %) in turbidity occurred as each tide progressed (Figure 6.4).

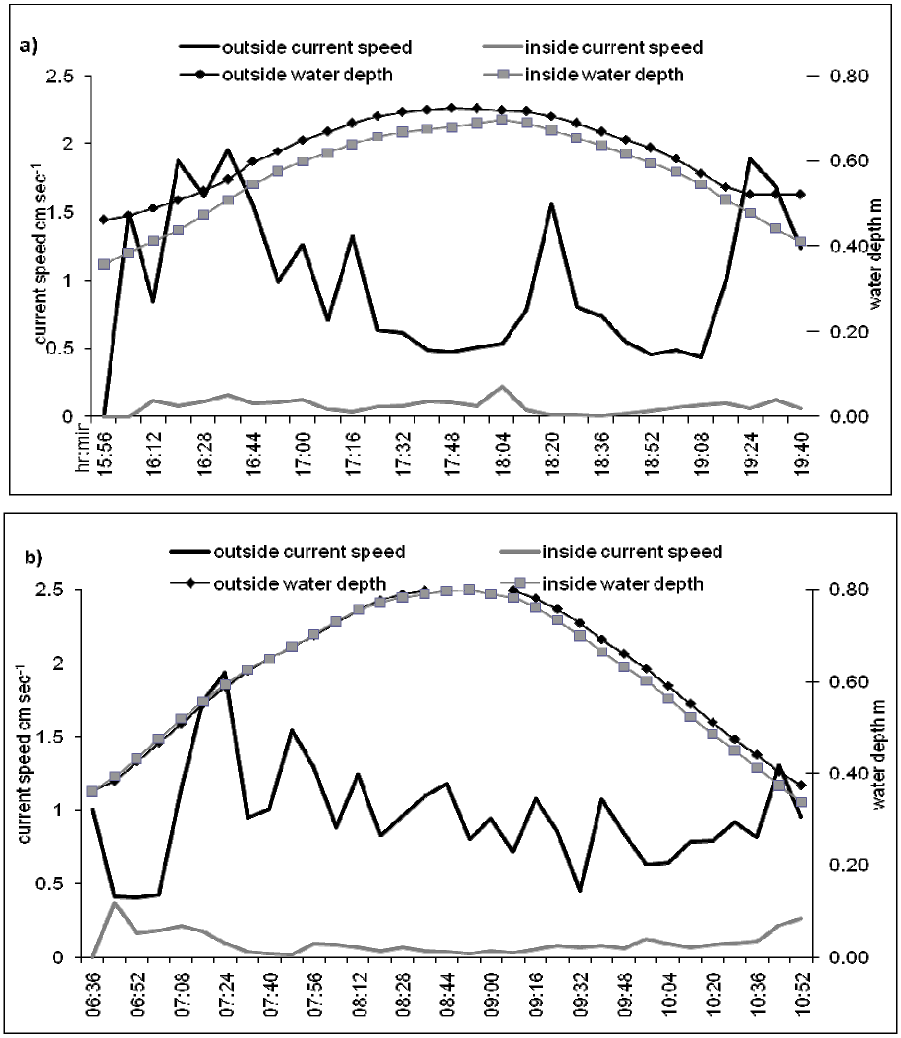

Figure 6.3 **Inundation height (m) and current speed (cm sec $^{-1}$) inside mangroves ('inside') and outside mangroves ('outside') measured at Site 4, over a spring tidal cycle on 21 December (a) and the 24 December (b), 2007. The X axis annotations represent hour:minute.**

Instrumentation results suggest weak flood-tide currents that reach the boundary of the mangrove vegetation, are further weakened by the dense network of mangrove trunks and pneumatophores. Despite the weak tidal currents, flood and ebb waters contained an average of approximately 25 mg l^{-1} of SSC. Suspended sediment concentrations fluctuated mostly within the range of 25 and 75 mg l^{-1} outside the mangroves, with slight peaks at the onset of both the flood and ebb flows, while a gradual decrease in SSC was evident within the mangroves with no discernible increases after commencement of the ebb tide. Apparent peaks in SSC of >500 mg l^{-1} were recorded at the mudflats (Station A) during the onset of flood tide inundation for three of the six tides monitored. These peaks lasted < 20 minutes, and may represent either the presence of a turbid fringe or could be an instrument failure due to the shallow water.

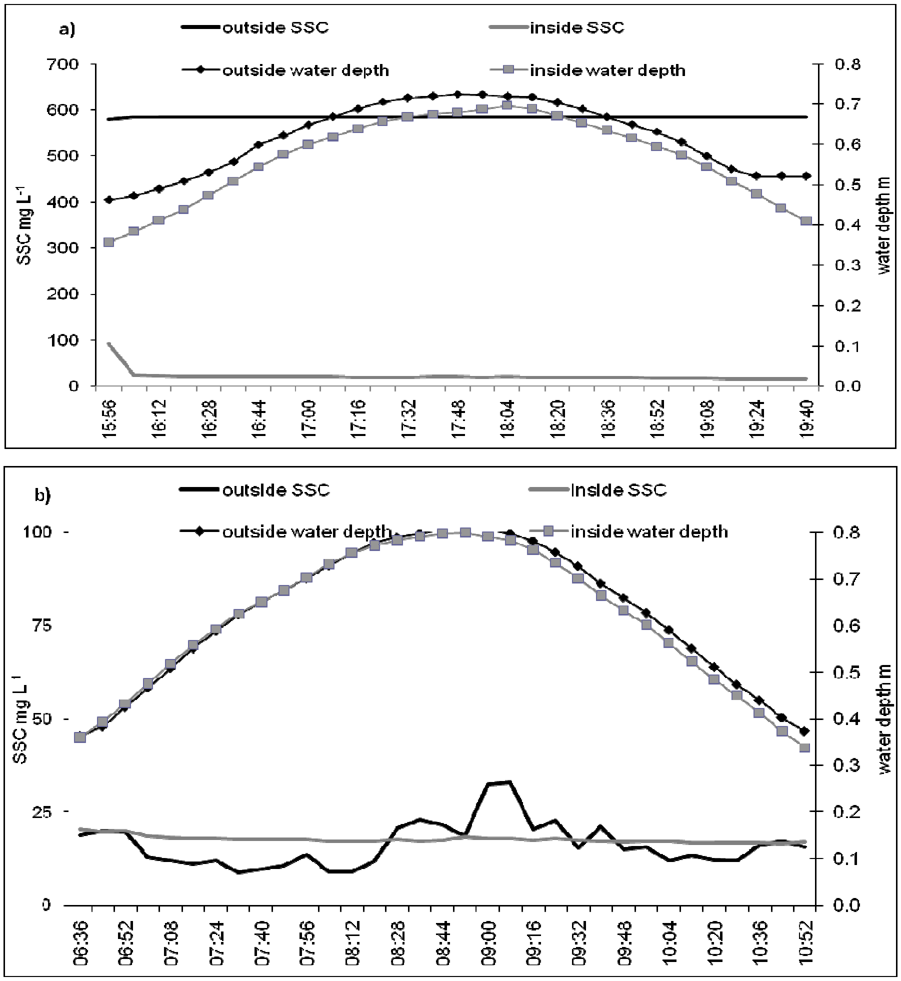

Figure 6.4 Inundation height (m) and Suspended Sediment Concentration (SSC) (mg L $^{-1}$) inside mangroves ('inside') and outside mangroves ('outside') measured at Site 4, over a spring tidal cycle on 21 December (a) and the 24 December (b), 2007. The X axis annotations represent hour:minute.

6.6 SEDIMENT LOADS ASSOCIATED WITH MANGROVE REMOVAL

6.6.1 Methods

Results from the erosion pin data presented in Chapter 4 are used here to estimate annual sediment loads released from intertidal areas that have been cleared of mangrove vegetation, using the following equation:

$$SEC\ yr^{-1}\ x\ p\ x\ BD = mass\ released\ m^{2}\ yr^{-1}$$

where: *SEC = surface elevation change*
p = perimeter of cleared area in m
BD = bulk density of surface sediments

6.6.2 Results

Three areas of mangrove vegetation in the vicinity of Site 2 at Waikaraka Estuary were cleared between March and October 2005 (Clear Plots 1, 2 and 3), totaling approximately 6,400 m^2. Further clearing in March 2006 removed an additional 3,600 m^2 of mangroves at Site 4 (Clear Plot 4). Monitoring of erosion pins identified some spatial variation in the rate of substrate subsidence/erosion that occurred after mangrove clearing, with plot averages of between 14 and 17 mm yr^{-1} (see Chapter 4.6 for details). Estimates of the sediment loads associated with these topographical changes are outlined in Table 6.1. The results suggest that approximately 94 m^3 of surface sediments were released as a result of mangrove clearance at Cleared Plots 1, 2 and 3 between March 2005 and March 2006. Clearing of vegetation in Plot 4 in March 2006, contributed a further 61 m^3 of sediments over a 12 month period. Erosion of the sea floor is on-going, and surface elevation rates slowed slightly between the 12 and 18 months post-clearance period (Table 6.1). Calculations of maximum sediment volumes for the 12 to 18 month period suggests a further 45 m^3 was removed from Cleared Plots 1 2 and 3 over that time (which coincides with the first 6 months of erosion at Clear Plot 4).

An estimate of mass per unit area is based on the bulk density of mangrove surface muds calculated for ^{210}Pb analysis (presented in Chapter 4). Results suggest that the removal of above-ground mangrove structures leads to the release of between 7.4 and 8.9 kg m^{-2} of surface material over the first 12 months, based on average surface elevation rates within each cleared plot (Table 6.1).

The material re-suspended after mangrove removal will be mostly mineral sediments, however it is important to consider the proportionality of micro and macro organic material, including mangrove root fragments. It is probable that between 8 and 17 % of the sediments consists of organic carbon, following analysis of sediments found in sediment traps. The contribution of mangrove biomass (> 1 mm diameter root fragments which are excluded from sediment organic tests) to the weight or volume of post-clearance spoil, is more difficult to predict. A study of mangrove below-ground biomass within Waikaraka Estuary (Chapter 5) highlighted spatial variability in the mangrove biomass, ranging from 2.5 kg m^{-2} (dry weight) to 4.5 kg m^{-2}. These values represent biomass collected to a depth of 20 cm, and if a percentage of that is calculated to correlate with the < 2 cm decrease in surface elevation recorded in the cleared plots, biomass values are approximately 0.45 kg m^{-2} (using the higher biomass). This represents 5 to 6% of the estimated total weight of re-suspended matter.

Within a 12 month period of Cleared Plots 1, 2 and 3 undergoing mangrove removal, a potential maximum of ~ 50,000 kg of sediments, including micro and macro-organics, was released. Toward the end of that 12 month period Plot 4 was cleared, contributing a further 32,000 kg in addition to the on-going release of sediments from Cleared Plots 1, 2 and 3. It is important to note, however, that an unknown parameter in this study is that of mangrove root collapse which can occur after mangrove mortality, as discussed by Cahoon et al. (2003). If this was significant, the volume of re-suspended sediment would be considerably lower.

Table 6.1 Dimensions of intertidal flats of Waikaraka Estuary that were cleared of above-ground mangrove structures in 2005 and 2006. Plots 1, 2 and 3 were cleared of mangroves in 2005, and Plot 4 in March 2006.

			EROSION PERIOD 1 to 12 MONTHS				**EROSION PERIOD 12 to 18 MONTHS**			
Clear Plot	Area Cleared (m^2)	Bulk density kg m^3	Surf. El. Change 1-12 months (m)	12 month max. vol. m^3	Max. weight of released material kg	Weight kg $^-m^2$	Surf. El.change 12-18 months	12-18 month max. vol. m^3	Weight 12-18 month kg	Weight kg $^-m^2$
1	2 600	526	0.014	36	19 146	7.36	0.0057	14.8	7 795	3.0
2	1 200	526	0.015	18	9 468	7.89	0.0105	12.6	6 628	5.52
3	2 600	526	0.0155	40	21 198	8.15	0.0065	16.9	8 889	3.42
4	3 600	526	0.017	61	32 191	8.94	0.0065	23.4	12 308	3.42

6.7 IMPLICATIONS FOR FUTURE MORPHODYNAMIC CHANGE

6.7.1 Background to a conceptual model

Based upon the following data and observations, a conceptual model was developed to consider the potential geomorphological changes in Waikaraka Estuary should mangrove expansion continue:

1. Mangrove expansion at Site 4, between 1982 and 1996, ranged from a seaward migration of between 15 m and 110 m over the 14 year period. An average seaward migration rate is approximately 40 m, equating to 2.8 m year^{-1}, or 28 m decade^{-1}. The mangroves fringing the western margins (Site 3) have colonised at a more spatially consistent rate equating to approximately 35 m decade^{-1}. A conservative approximate of 10 m decade^{-1} was used to develop the conceptual model of geomorphic development. It is noted, however, that establishment phases can be episodic rather than consistent (Swales et al., 2007). Soon after 1996, local residents began removing seedlings and therefore modifying the forest dynamics. It is not possible, therefore to evaluate any change to the natural rate of mangrove colonisation over the past decade.

2. Annual rates of surface elevation reported by Stokes et al. (2009), indicate spatially variable substrate accretion. However, at site 4 surface elevation appeared to decline in magnitude with distance from the mangrove fringe (in the landward, or flood tide direction), from 6 mm yr^{-1} at 10 m, to 1 mm yr^{-1} at 30 m. From this, an assumption was made that substrate accretion of 3 mm yr^{-1} is likely at 20 m. Surface elevation measurements across the three RSET transects in the vicinity of Site 3, provided an average substrate accretion of 7 mm yr^{-1} at 10 m and 4 mm yr^{-1} at 20 m landward of the mangrove fringe.

3. Spring tide water level elevations measured at Site 4 show a maximum inundation height of 0.8 m at the mangrove fringe. Neap inundation heights are likely to be < 0.5 m, considering a neap tidal range of 1 m measured in the adjacent channel (Hope, 2002) and reduced water depth with distance from the channel (Kathiresan, 2003). As such, it is assumed that little suspended sediment is reaching landward margins of mangrove habitat beyond 30 m from the seaward fringe.

6.7.2 Results

Figure 6.5 illustrates the possible topographical changes resulting from mangrove expansion of 10 m per decade, for a period of 50 years, at Site 4. The surface profile shown in Figure 6.5 is annotated with Stations A through to E to assist with the discussion of temporal and spatial change. The conceptualized surface profile after 50 years, demonstrates the development of a convex sea floor topography in response to increased sediment deposition associated with mangroves expansion. Station B became the location of a topographical high reaching 0.26 m due to the added surface height attributed to its location in established mangroves at Year 10, and subsequent accretion of sediments at this position over the subsequent 30 years. Over time, as the mangrove fringe moved closer to the channel and as the surface elevation of Stations C and D increased, the difference in topography became less pronounced, or the gradient between the two locations decreased.

Mangrove habitat toward the landward edge of the mangrove zone (i.e. Station A) quickly becomes subject to lower suspended sediment loads, shallower inundation, and also fewer inundation events, as a result of the 10 m per decade colonisation rate. Sedimentation rates therefore become negligible within 20 – 30 years, which results in the development of a relative depression. It is probable that this topographical depression would become more pronounced due to shrinkage associated with dessication of surface sediments. A similar profile shape, showing a lower surface at the landward end of the transect, relative to the mangrove fringe, was recorded during a topographical survey of the RSET transects undertaken in 2005. It is also likely that a similar geomorphology would develop across the mangrove forest occupying the neck of the estuary (Site 2 and Site 3).

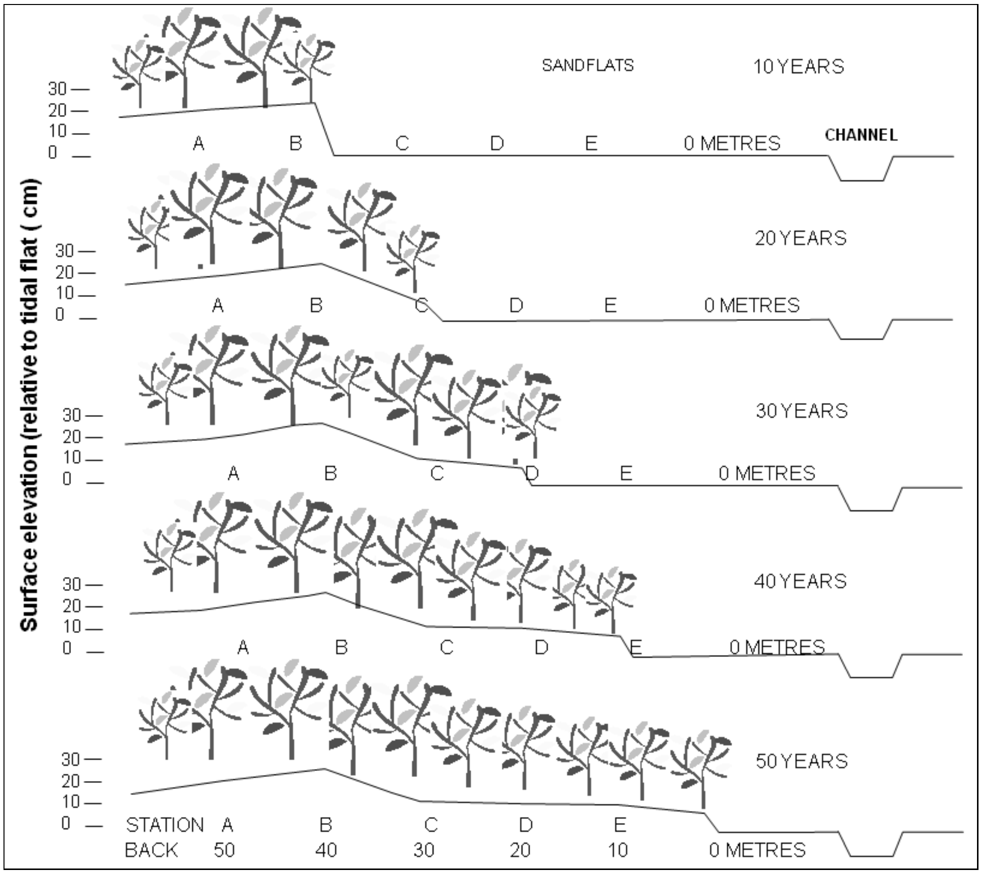

Figure 6.5 **Conceptual model: mangrove expansion and surface topography, in ten year increments, at Site 4.**

6.8 DISCUSSION

6.8.1 Tidal influences on contemporary sedimentation

Mangrove habitat is presently (2009) covering approximately 20% of the intertidal area of Waikaraka Estuary, which represents a significant driver of geomorphic change to the study site. Sedimentation rates in mangroves appear to have slowed in comparison to rates prior to mangrove colonization, however mangroves act to enhance background sedimentation levels (Young and Harvey, 1996), promote mud deposition, and contribute organic material to the carbon cycle of the estuary.

This process of textural change is driven by the interaction between tidal flows and the complex and dense mangrove structures (Furukawa et al., 1997; Massel et al., 1999; Phuoc and Massel, 2006). The high density of pneumatophores and trunks create an increased friction to flow (Furukawa and Wolanski, 1996), which in this study has resulted in reduced tidal current speeds of < 0.01 m sec^{-1} within 5 m of the mangrove fringe. These sluggish tidal currents promote sediment deposition, with re-suspension unlikely. Any erosion events within the mangrove habitat would therefore only occur during high energy events (Woodroffe, 1983) and/or high rainfall events (Tolhurst et al., 2005).

Interestingly, no strong tidal asymmetry in suspended sediment or tidal current velocity was identified at Waikaraka Estuary over the 4 day deployment of OBS and current meters. Indeed, the strength of the tidal flows, both inside and outside the mangroves, was well below the 0.3 m sec^{-1} required for re-suspension of sediment fines (Wolanski et al., 1995). SSC in mangroves at Waikaraka is not influenced by tidal stage (i.e., no flood peaks), instead showing a gradual decline over the inundation period.

Tidal current velocities of <0.02 m s^{-1} measured on the mudflats seaward of mangroves at Site 4 are considerably lower than that reported in studies of tidal dynamics within mangrove systems that fringe tidal creeks. For example, Wolanski (1992) found current velocities leaving tidal channels regularly exceeded 1 m s^{-1}, whereas current speeds remained >0.07 m s^{-1} in mangrove forest 50 m from the creek edge. Van Santen et al. (2006) reported a dampening of flood tide velocities across mudflats fronting a mangrove forest in Vietnam which ranged from 0.15 to 0.5 m s^{-1}. Tidal currents through fringing mangroves, measured by Van Santen et al. (2006), generally did not exceed 0.03 m s^{-1} which is still higher than the 0.01 m s^{-1} measured in mangroves at Waikaraka Estuary. One factor that will act to attenuate tidal velocities in the fringing mangroves at Waikaraka is the fronting 'cleared zone' which extends approximately 30 m toward the channel. The mudflat surface of this cleared area consists of watery muds with numerous protruding tree and pneumatophore stumps which will be producing some frictional force against the tidal currents. Figure 6.6 illustrates an area recently cleared of mangroves and highlights the roughness of the bed surface.

Figure 6.6 Cleared area (Plot 1) two days after mangroves were removed in May 2005. Distance from left to right (middle of photo) approximately 40 m.

Sediment loads, as measured by sediment traps, were highest on the bare mudflats in Waikaraka. Despite the relatively higher sediment loads and SSC observed over the mudflats the mudflat surface is subsiding at >10 mm yr^{-1}, suggesting either a scouring surface or a collapsing substrate induced by root collapse after mangrove removal, or more likely a combination of the two. Couple this with sluggish tidal flows and it can be expected that much of the re-suspended sediment is delivered into the neighbouring mangrove habitat on an incoming tide.

Suspended sediment entering mangroves at Site 4 appear to settle over relatively short distances, with between 30% and 60% of the initial SSC deposited within 10 m of the seaward mangrove fringe. This is typical of sedimentation processes in mangroves, however the volume and gradient of deposition will be site-specific and reliant on inundation height and sediment supply (Furukawa and Wolanski, 1996). For example, Furukawa and Wolanski (1996) measured a 50% decrease in SSC within 35 m of the tidal creek/mangrove edge, whereas Victor et al. (2006) suggested much of the incoming suspended sediment in a microtidal site may be deposited within the seaward 25 m of mangrove forest. Similarly, Van Santen measured high sediment trap accumulation rates in sparse, pioneering mangrove vegetation of 20 – 40 g cm^2 yr^{-1}, compared with 0.5 – 2.5 g cm^{-1} yr^{-1} in dense mangrove habitat approximately 30 m landward. To draw a site comparison, accumulation rates on the mudflats fronting mangroves in Waikaraka were between 1 and 7 g cm^2 yr^{-1} which reflects the smaller sediment yields delivered into Waikaraka Estuary.

6.8.2 Mangrove expansion

The predicted future geomorphology in the presence of mangrove expansion at Waikaraka Estuary was based on empirical data of substrate accretion, combined with an assumption of a constant expansion of mangroves of 10 m per decade. The conceptual model highlights the potential for seaward colonisation of mangroves to limit vertical growth of the seafloor on the landward side of the colony. Sediment trap results and surface elevation changes measured at Site 4 suggest that the majority of the incoming sediment is deposited within 20 – 30 m of the mangrove fringe. The progression of the mangrove fringe results in the older landward mangroves being subject to less tidal inundation, and therefore less sediment supply, as they move back relative to the tidal frame. The resultant cross-shore profile becomes convex, with development of a relative depression toward the landward side of the mangrove forest which may become hyper-saline and compacted. The landward margins could therefore see a progression in plant communities dominated by either stunted mangrove plants or a shift to saltmarsh species such as *Juncus.*

Progression of mangrove habitat can occur only where intertidal areas are positioned above mean sea level. At Waikaraka, over 90% of the estuary is above mean sea level (Park, 2004), indicating the potential for continued mangrove expansion. Following the evolutionary progression subscribed by Thom (1975) and Woodroffe (1992), for example, mangrove could continue to prograde across the tidal flats with the eventual morphology being a contiguous mangrove/saltmarsh wetland dissected by tidal channels.

Surface elevation of mangrove habitat in the Waikaraka Estuary averaged 3 mm yr^{-1}, similar to the average rate of sea-level rise (Hannah, 2004). Numerous studies have identified a feedback mechanism at play in wetlands whereby surface elevation keeps pace with rising sea-level (Lynch et al., 1989; Cahoon and Lynch, 1997). Although a positive correlation between increased tidal inundation and suspended sediment has been identified (e.g. Temmerman et al., 2003), substrate accretion is a function of a complex set of physical and biological processes (Cahoon and Lynch, 1997). It has been suggested elsewhere that below-ground biomass can contribute to surface accretion (McKee and Faulkner, 2000; Cahoon et al., 2006; McKee et al., 2007), and this becomes particularly important in regions of low terrestrial sediment input such as coral islands (Krauss et al., 2003; Day et al., 2008).

The impact of sea-level rise in Tauranga Harbour becomes a more serious issue upon consideration of the effects of mangrove removal.

6.8.3 Mangrove removal

Between 2003 and 2009 approximately 10% of the mangrove habitat in Waikaraka Estuary was cleared. The associated decline in surface elevation of between 14 mm yr^{-1} and 17 mm yr^{-1} would amplify the effects of sea-level rise by effectively increasing the depth of tidal inundation. The double effect could effectively lower relative sea-level more than 20 mm yr^{-1}. This may have wider implications if all mangroves are cleared because no buffer would exist to protect the remaining saltmarsh from potentially higher tidal inundation. In Waikaraka, similar to the other sites of this study, there is little to no room for any landward progression of saltmarsh to accommodate significant increases in RSLR.

Mangrove removal in Waikaraka Estuary is altering both the forest dynamics and the estuarine geomorphology. In the 12 months after mangroves were felled approximately 7.4 – 8.9 kg of sediment was released for each square metre that was cleared. These ‘released’ loads will include mangrove roots, micro-organics (i.e. algae) and mineral sediments.

By converting vertical measurements of substrate change to sediment mass, as mentioned above, an assumption is made that the seafloor changes are due to surface erosion. What is not taken into account is the effect of root collapse, which has been found to occur as fine roots die and compact, effectively lowering the relative surface level. This process was identified after Hurricane Mitch caused tree mortality in Honduras, which resulted in a collapse of the underlying mangrove peats of 11 mm yr^{-1} (Cahoon et al., 2003). Mangrove sediments in Waikaraka Estuary are less biogenic than the peat soils identified by Cahoon et al. (2003), making any attempt to estimate the effect of root mortality on seafloor collapse difficult in this instance.

The impacts of any sediment redistribution as a response to vegetation removal in Waikaraka Estuary can only be speculated. Considering the quiet nature of the tidal regime in the regions where mangroves are being cleared, it is probable that surface sediments will be entrained and re-deposited within the estuary, particularly within the remaining mangroves where tidal velocities are extremely low. A remnant mangrove fringe may therefore be an important consideration for coastal management purposes, particularly in its potential to buffer the detrimental effects that high turbidity and sedimentation can impose on estuarine ecology, such as bivalve productivity (Thrush et al., 2003; Norkko et al., 2006).

6.9 CONCLUSION

Mangroves have colonised over 25 % of the intertidal surface of the Waikaraka Estuary and have in turn enhanced sedimentation and surface elevation which appears to be keeping pace with sea-level rise. This study has described the potential geomorphic evolution of the intertidal flats should mangrove encroachment continue. Ninety per cent of the intertidal area is presently positioned at, or close to, the range of elevation suitable for mangrove establishment, which suggests on-going mangrove expansion is probable if wave and climate conditions are suitable. Tidal currents measured during calm weather spring-tide events identified very low tidal velocities which are sufficient to transport suspended sediment into the mangrove community, but insufficient to resuspend material once it has integrated with the substrate surface. Seaward sections of the mangrove population have been cleared and this has resulted in a lowering of the intertidal surface at a mean rate of 17 mm yr^{-1}, which in effect increases the height of tidal inundations at these locations. These finding have implications for the sustainability of the remaining mangrove and salt marsh should sea-level rise continue. The impact of clear-felling, in terms of sediment dynamics, is the release and redistribution of up to 9 kg of sediments (including organics and mangrove roots) for each square metre that is cleared of vegetation.

Any future investigation into the effects of mangrove removal would benefit from a study of the implications of root collapse (as opposed to sediment erosion) on the subsidence of the harbor floor, as the scale of this process will influence the accuracy of any estimation of sediment loads. In light of the potential release of significant volumes of fine sediments, however, any consideration of mangrove removal should closely investigate the nature of the tidal transport system to accommodate the associated increased sediment budget.

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6.11 LITERATURE CITED

Bell, R., Goring, D. and de Lange, W. 2000. Sea-level change and storm surges in the context of climate change. *IPENZ (Institute of Professional Engineers New Zealand) Transactions* 27(1), 1-10.

Cahoon, D.R., Day Jr., J.W., Reed, D.J., 1999. The influence of surface and shallow subsurface soil processes in welan elevation: a synthesis. *Current Tropical Wetland Biogeography* 3, 72-88.

Cahoon, D. R., Hensel, P. F., Spencer, T., Reed, D. J., McKee, K. L. and Saintilan, N., 2006. Coastal wetland vulnerability to relative sea-level rise: wetland elevation trends and process controls. In: J. T. A. Verhoeven, B. Beltman, R. Bobbink and D. E. Whigham (Eds) Wetlands and natural resource management. Ecological Studies. Berlin, Springer-Verlag. 190, pp. 271-292.

Cahoon, D. R. Hensel, P., Rybczyk, J., McKee, K.L., Proffitt, C.E. and Perez, B.C. 2003. Mass tree mortality leads to mangrove peat collapse at Bay Islands, Honduras after Hurricane Mitch. *Journal of Ecology* 91(6), 1093-1105.

Cahoon, D. R. and Lynch, J. C., 1997. Vertical accretion and shallow subsidence in a mangrove forest of southwestern Florida, U.S.A. *Mangroves and Salt Marshes* 1, 173-186.

Cahoon, D. R., Reed, D. J. and Day, J. W. 1995. Estimating shallow subsidence in microtidal salt marshes of the southeastern United States: Kaye and Barghoorn revisited. *Marine Geology* 128(1-2), 1-9.

Day, J. W., Christian, R. R., Boesch, D. M., Yanez-Arancibia, A., Morris, J., Twilley, R. R., Naylor, L., Schaffner, L. and Stevenson, C. 2008. Consequences of climate change on the ecogeomorphology of coastal wetlands. *Estuaries and Coasts* 31, 477-491.

Furukawa, K., and Wolanski, E., 1996. Sedimentation in mangrove forests. *Mangroves and Salt Marshes* 1, 3-10.

Furukawa, K., Wolanski, E., and Mueller, H., 1997. Currents and sediment transport in mangrove forests. *Estuarine Coastal and Shelf Science* 44, 301-310.

Gibbs, J., Sheffield, A., Foster, G., 1992. A standardised coastal sensitivity index based on an initial framework for physical coastal hazards information. Science and Research Series No. 55, Department of Conservation, Wellington, New Zealand, 103 p.

Hannah, J., 2004. An updated analysis of long-term sea level change in New Zealand. *Geophysical Research Letters* 31, L03307

Healy, T. R., Cole, R. G. and De Lange, W., 1996. Geomorphology and ecology of New Zealand shallow estuaries and shorelines. In: Nordstrom, K. F. and

Roman, C. T. (Eds.), Estuarine Shores: Evolution, Environments and Human Alteration. John Wiley & Sons, Chichester, pp. 115-154.

Hope, H. M., 2002. Sediment dynamics of Waikaraka Estuary, a small, semi-enclosed estuarine system in the upper reaches of a developed harbour. Unpublished M.Sc thesis, University of Waikato, 126 p.

Hume, T. and McGlone M. S., 1986. Sedimentation patterns and catchment use change recorded in the sediments of a shallow tidal creek, Lucas Creek, Upper Waitemata Harbour, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 20, 677-687.

IPCC (Intergovernmental Panel on Climate Change), 2007. Climate Change 2007: The science basis. Contribution of Working Group 1 to the Fourth Assessment Report, Cambridge University Press, Cambridge, UK.

Kathiresan, K., 2003. How do mangrove forests induce sedimentation? *Revista De Biologia Tropical* 51(2), 355-359.

Krauss, K. W., Allen, J. A. and Cahoon, D. R., 2003. Differential rates of vertical accretion and elevation change among aerial root types in Micronesian mangrove forests. *Estuarine, Coastal and Shelf Science* 56(2), 251-259.

Lynch, J. C., Meriwether, J. R., McKee, B. A., Vera-Herrera, F. and Twilley, R. R. 1989. Recent accretion in mangrove ecosystems based on 137Cs and 210Pb. *Estuaries* 12(4), 284 -299.

Massel, S.R., Furukawa, K., and Brinkman, R.M., 1999. Surface wave propagation in mangrove forests. *Fluid Dynamics Research* 24, 219-249.

McKee, K. L., Cahoon, D. R. and Feller, I. C. 2007. Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. *Global Ecology and Biogeography* 16(5), 545.

McKee, K.L., and P.L. Faulkner. 2000. Mangrove peat analysis and reconstruction of vegetation history at the Pelican Cays, Belize. *Atoll Research Bulletin* 468: 46-58

Norkko, J., Hewitt, J. E. and Thrush, S. F., 2006. Effects of increased sedimentation on the physiology of two estuarine soft-sediment bivalves, *Austrovenus stutchburyi* and *Paphies australis*. *Journal of Experimental Marine Biology and Ecology* 333(1), 12-26.

Park, S., 1999. Changes in Abundance of Seagrass (*Zostera* spp.) in Tauranga Harbour from 1959-96. Environment BOP Environmental Report 99/30. Environment Bay of Plenty, Whakatane, New Zealand: 19 p.

Park, S., 2004. Aspects of mangrove distribution and abundance in Tauranga Harbour. Environment Bay of Plenty Environmental Publication 2004/16, Whakatane, New Zealand, 41 p.

Phuoc, V.L.H., and Massel, S.R., 2006. Experiments on wave motion and suspended sediment concentration at Nang Hai, Can Gio mangrove forest, Southern Vietnam. *Oceanologia* 48, 23-40.

Rogers, K., Saintilan, N. and Cahoon, D., 2005. Surface elevation dynamics in a regenerating mangrove forest at Homebush Bay, Australia. *Wetlands Ecology and Management* 13(5), 587-598.

Sheffield, A. T., Healy, T. R. and McGlone, M. S. 1995. Infilling rates of a steepland catchment estuary, Whangamata, New Zealand. *Journal of Coastal Research* 11(4), 1294-1308.

Shepherd, M. J., McFadgen, B. G., Betts, H. D. and Sutton, D. G., 1997. Formation, landforms and palaeoenvironment of Matakana Island and implications for archaeology. Wellington, N.Z., Dept. of Conservation. 77 p

SonTek/YSI, Inc., 2003. Triton System Manual: Firmware Version f.f. SonTek YSI Inc., San Diego, CA, 213 p.

Smoak, J. M. and Patchineelam, S. R., 1999. Sediment mixing and accumulation in a mangrove ecosystem: evidence from ^{210}Pb, ^{234}Th and ^{7}Be. *Mangroves and Salt Marshes* 3, 17-27.

Stokes, D.J., Healy, T.R., Cooke, P.J., 2009. Surface elevation changes and sediment characteristics of intertidal surfaces undergoing mangrove expansion and mangrove removal, Waikaraka Estuary, Tauranga Harbour, New Zealand. *International Journal of Ecology and Development* 12 (W09), 88-106.

Swales, A., S. J. Bentley, Lovelock, C.E. and Bell, R., 2007. Sediment processes and mangrove-habitat expansion on a rapidly-prograding muddy coast, New Zealand. *Coastal Sediments '07*, New Orleans, Louisiana, American Society of Civil Engineers, pp. 1445-1454

Swales, A., Williamson, R. B. Van Dam, L. F., Stroud, M. J., McGlone, M. S., 2002. Reconstruction of urban stormwater contamination of an estuary using catchment history and sediment profile dating. *Estuaries* 25(1), 43-56.

Temmerman, S., Govers, G., Meire, P. and Wartel, S. 2003. Modelling long-term tidal marsh growth under changing tidal conditions and suspended sediment concentrations, Scheldt estuary, Belgium. *Marine Geology* 193, 151-169.

Thom, B. G., Wright, L. D., Coleman, J.M., 1975. Mangrove ecology and deltaic-estuarine geomorphology: Cambridge Gulf-Ord River, Western Australia. *The Journal of Ecology* 63(1), 203.

Thrush, S. F., Hewitt, J. E., Norkko, A., Nicholls, P. E., Funnell, G. A. and Ellis, J. I., 2003. Habitat change in estuaries: predicting broad-scale responses of intertidal macrofauna to sediment mud content. *Marine Ecology Progress Series* 263, 101-112.

Tolhurst, T. J., Friend, P. L., Watts, C., Wakefield, R., Black, K. S. and Paterson, D. M., 2005. The effects of rain on the erosion threshold of intertidal cohesive sediments, *Aquatic Ecology* 40(4), 533-521.

Turner, S. E. and Schwarz, A.M. 2006. Biomass development and photosynthetic potential of intertidal *Zostera capricorni* in New Zealand estuaries. *Aquatic Botany* 85, 53-64.

Van Santen, P., Augustinus, P.G.E.F., Janssen-Stelder, B.M., Quartel, S., and Tri, N.H., 2006. Sedimentation in an estuarine mangrove system. *Journal of Asian Earth Sciences* 29, 566-575.

Victor, S. L., Leinson, N., Yimnang, G., Wolanski, E., Richmond, R.H., 2006. Sedimentation in mangroves and coral reefs in a wet tropical island, Pohnpei, Micronesia. *Estuarine, Coastal and Shelf Science* 66(3-4), 409-416.

Wolanski, E., 1992. Hydrodynamics of mangrove swamps and their coastal waters. *Hydrobiologia* 247(1-3), 141-161.

Wolanski, E., King, B., and Galloway, D., 1995. Dynamics of the turbidity maximum in the Fly River estuary, Papua New Guinea. *Estuarine, Coastal and Shelf Science*, 40: 321-337.

Woodroffe, C., 1983. Development of mangrove forests from a geological perspective. In: Teas H.J. (Ed), Tasks for vegetation science. The Hague, Dr W Junk Publishers, Vol. 8, pp 1-17.

Woodroffe, C., 1992. Mangrove sediments and geomorphology. In: A. I. Robertson and D. M. Alongi (Eds) Tropical Mangrove Ecosystems. Washington, D.C., American Geophysical Union. 41, pp. 7-41.

Young, B. M. and Harvey, E.L., 1996. A spatial analysis of the relationship between mangrove (*Avicennia marina* var. *australasica*) physiognomy and sediment accretion in the Hauraki Plains, New Zealand. *Estuarine, Coastal and Shelf Science* 42(2), 231-246.

Chapter 7
Benthic ecology of temperate mangroves

7.1 PREFACE

Estuarine benthic populations provide a number of ecological services. Intertidal macroinvertebrate communities are an important food source for wading birds and some bottom-feeding species such as flounder and stingrays (Cole et al., 2000). Bivalves are also an important food source for humans and are highly valued 'kai moana' for coastal Maori. Infauna can generally be separated into functional groups according to their feeding modes. For example, bivalve species such as the cockle (*Austrovenus stutchburyi*) are filter-feeders, and polychaetes such as the *Nicon* species, are surface deposit feeders. In the action of feeding and processing food, both functional groups rework and oxygenate sediments and recycle suspended sediments (Thrush et al., 2003).

An increased understanding of the susceptibility to turbidity and sedimentation by various benthic organisms has provided the opportunity to measure these parameters for environmental monitoring purposes. Significant shifts in the community composition of estuarine benthic ecosystems can alarm coastal managers to potentially detrimental shifts in the physico-chemical environment. Regional councils in New Zealand are responsible for monitoring the health of coasts and estuaries, and as such, undertake either seasonal or six monthly benthic sampling. Because of the cost and time involved, macroinvertebrate sampling is typically restricted to key sites which rarely includes mangrove habitat.

Supporters of mangrove removal often cite a reduction in benthic diversity as a negative impact of mangrove expansion. To date, there is little information available to confirm these assumptions. Baseline data of macroinvertebrate communities in Tauranga Harbour are provided in this chapter, and the influence of mangroves on benthic diversity and benthic community composition is discussed. The infaunal data presented in this chapter was peer-reviewed and published in the Proceedings of the biennial Australasian Coasts and Ports Conference, 2009 (Wellington, New Zealand) which was an amalgamation of the New Zealand Coastal Society, Australasian Coastal and Ocean Engineering and Australasian Ports and Harbour conferences. Representatives of regional councils and crown research institutes involved in coastal monitoring typically attend this event, and therefore it seemed an appropriate arena to publish and present this work.

The citation is:

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The lead author was responsible for all field work, macroinvertebrate identification and preparation of the manuscript, under the supervision of co-author T.R. Healy. Norm Mason was included as a co-author in acknowledgement of his contribution to the software manipulation and subsequent running of the PERMANOVA multivariate statistical analysis.

This chapter addresses the thesis objective of determining the influence of mangroves on benthic community composition.

7.2 INTRODUCTION

A growing public perception is that the expansion of mangroves in New Zealand estuaryies has initiated a decline in benthic biodiversity. Anecdotal evidence of decreased bivalve abundance is often used to reinforce this argument. As such, mangrove removal is being increasingly considered by coastal managers as a process to improve the benthic ecosystem health of estuaries in New Zealand. However there is little direct evidence to link mangrove colonization with reduced biodiversity.

Two general trends in estuarine geomorphology and ecology are well documented:

1. Once mangroves have established over intertidal flats sedimentation rates may be enhanced and mud content of the surface sediment is likely to increase (Furukawa et al., 1997; Quartel et al., 2006).
2. Resident benthic communities that inhabit the intertidal zone are known to alter in response to increased terrestrial sediment loads (Cummings et al., 2003), with bivalves being particularly vulnerable (Norkko et al., 2006).

As such, it could be expected that as the intertidal habitat becomes colonised by mangroves, the benthic community composition will alter. Results of the few studies undertaken to date indicate low benthic macroinvertebrate diversity and abundance exists in mangrove habitat in New Zealand, however these findings are often coupled with a similarly low diversity of adjacent intertidal mudflats (Ellis et al., 2004). This would suggest that benthic communities are responding to increased silt/clay as a result of higher sedimentation rather than simply to the presence of mangroves. Surface macroinvertebrate communities are dominated by grazing snails which can show distinct distribution patterns that are controlled by dessication, wave action, temperature and salinity (Kathiresan and Bingham, 2000).

Studies of mangrove and mudflat habitat have mostly described the same functional groups of benthic macroinvertebrates. The species identified often vary, however, as do the estuarine sediment regimes of the sites investigated (Morrisey et al., 2003; Ellis et al., 2004; Alfaro, 2005). Further investigation is therefore required to identify the response of benthic community to the spread of mangrove ecosystems. This research aims to quantify and compare benthic invertebrate populations within both mangrove habitat and adjacent unvegetated intertidal flats. The results identify the potential impacts of mangrove expansion in upper estuarine environments where increased inputs of terrestrial sediments are influencing the present-day geomorphology.

7.3 STUDY AREA DESCRIPTION

Sampling for this study was undertaken in 3 sub-estuaries, namely, Welcome Bay, Waikareao Estuary and Waikaraka Estuary. Transect 1 was located closest to the head of the estuary, and Transect 3 toward the middle of the system, with Transect 2 roughly halfway between (Figure 7.1). The two study sites of Welcome Bay and Waikareao Estuary were chosen for their hydrodynamic differences (Table 7.1). Welcome Bay represents a lower energy system, and Waikareao estuary a more open, higher energy environment with considerably greater terrestrial sediment inputs. Waikaraka Estuary, which is also a narrow, low-energy embayment, was included in the study to monitor sedimentological and topographical changes after council-approved removal of above-ground mangrove vegetation.

Mangroves and nearby unvegetated flats of Transects 2 and 3 in both Welcome Bay and Waikareao are exposed for around 5 hours over most low tides, while Transect 1 at both sites is inundated roughly 30 minutes later. Spring tide inundation height, based on observation of markings on the fringe mangroves, does not exceed ~0.75 m. In comparison, Waikaraka estuary experiences reduced inundation periods, with the monitoring locations of Transects 1 and 2 generally exposed for up to 8 hours per semi-diurnal tide (Hope, 2002) and Transect 3 is exposed for around 7 hours. Spring tidal inundation ranges from < 0.5 m at Transects 1 and 2 to ~ 0.75 m at Transect 3 (see Figure 7.1 for transect locations).

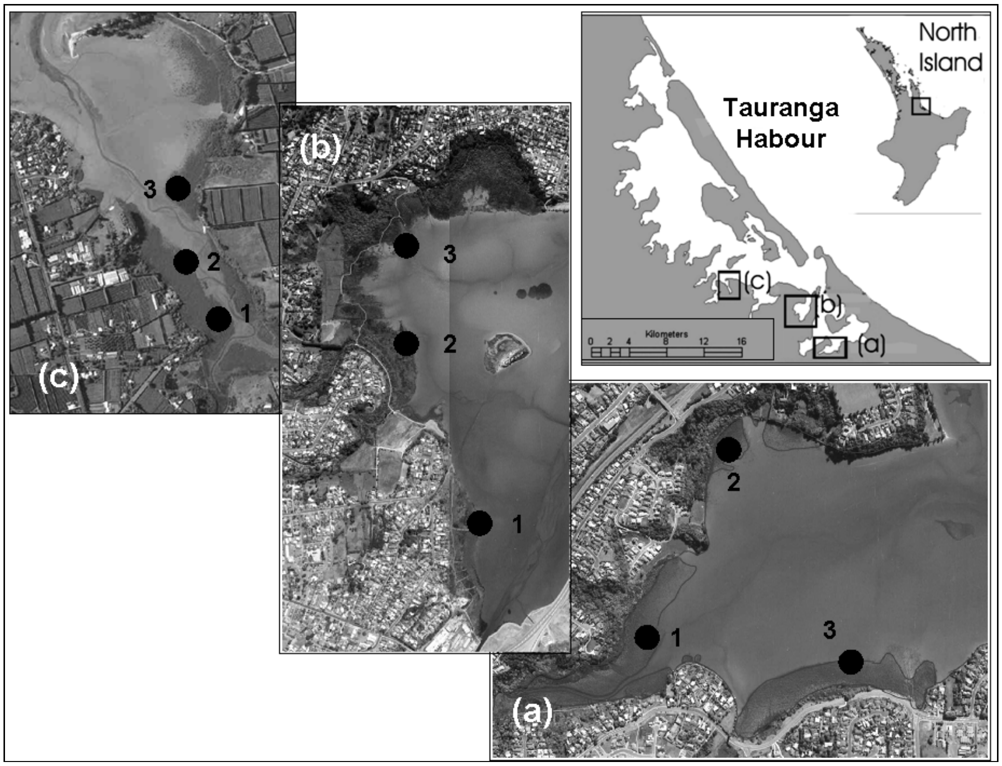


Figure 7.1 **Transect locations marked on 2003 aerial photos of Welcome Bay (a), Waikareao Estuary (b), and Waikaraka Estuary (c), North Island, New Zealand. Photo's courtesy of Environment Bay of Plenty.**

Table 7. 1 Estuary and catchment characteristics of Waikareao Estuary (Waik), Welcome Bay (WB) and Waikaraka Estuary (TP), taken from Park, 2003 *; Surman, 1999 ** and Hope, 2002^.

	Waik	WB	TP
Estuary area (km^2)	3.25	1.6	0.5
Mangrove coverage 1980 (ha)	2.2	8.3	3
Mangrove coverage 2003 (ha)	16	11.6	11.5
Catchment size (km^2) *	74	20	10
% catchment urban *	8	16	1
% catchment horticulture or pasture *	54	84	86
% scrub/forest *	40	1	13
Distance from harbour entrance km	4.5	10	12
Freshwater inflow yields - mean of recorded flows in L/s	2450**	179**	92^

7.4 METHODS

7.4.1 Sampling

Three transects were marked out at each of the three study sites. The seaward fringe of mangrove vegetation was marked as ‘0 m’. Sampling of mangrove habitat was undertaken 20 and 10 m landward of the mangrove/tidal flat boundary (labelled as M20 and M10). Stations on the unvegetated intertidal flats were positioned 10 m (labeled F10) and approximately 40 m (labeled F40) from the mangrove fringe. Sampling was undertaken in February 2006 (southern hemisphere summer), July 2006 (winter), and February 2007 at Waikareao Estuary and Welcome Bay, and July 2006 and February 2007 at Waikaraka Estuary.

Benthic macrofauna were collected in 3 replicate 13 cm diameter x 20 cm cores. The cores were taken roughly 1 m apart, at each location. All samples were sieved on-site during high tide through a 1000 µm mesh. The 1 mm sieve was used in preference to the more commonly used 0.5 mm sieve in this study because of the difficulty in separating infauna from the mangrove roots while on-site. Absolute abundance of some populations are underestimated using the 1 mm mesh, however a study on the effect of mesh size on taxonomic resolution suggested the larger mesh will make little difference to the spatial patterns for macrofauna assemblages (James, et al., 1995). Collected organisms were preserved in isopropyl alcohol and later stained with Rose Bengal and all organisms were enumerated and identified to the lowest possible taxonomic level.

Surface macrofauna (epifauna) were counted within three randomly placed 1 m^2 quadrats at each sampling location prior to the collection of benthic cores.

7.4.2 Statistics

Similarities in community structure were established using non-metric multi-dimensional scaling (MDS) based on Brays Curtis similarity matrices, square-root transformed where required, using PRIMER software. Community composition and species diversity was further investigated using PERMANOVA techniques, written in C++ script (available on request) and performed using R software. A nested approach was used, with factors ‘mangrove/non-mangrove’; ‘station’ ‘transect’ and ‘estuary’; nested within ‘season’ and ‘year’. Homogeneity of variance was checked using Levene’s test, and although the result indicated variance across groups of the species composition data, highly significant results from the PERMANOVA test (< 0.001) were supported by MDS, and as such were included in the data interpretation. Univariate indices and benthic community composition are described using data associated with the 2007 (summer) sampling.

Nested ANOVA was undertaken to assess the influence of transect, habitat (mangrove or sand/mudflat), and sampling time/season on square-root transformed abundance data for the dominant epifaunal species within each estuary.

7.4.3 Environmental variables

Environmental variables examined in this study were sediment grain size and organic matter content. Surface samples were collected in triplicate from the top 1-2 cm of the substrate surface. Grain size was measured with a Malvern Mastersizer-S Longbed after 48 hours in hydrogen peroxide and 24 hours in Calgon. Organic content was determined by weight lost after ignition (500° C for 5 hours). Surface elevation changes at the study sites are reported in detail in Chapters 3 and 4.

7.5 RESULTS

7.5.1 Environmental Parameters

The mangrove and bare habitats were distinguishable by their sediment characteristics. Fine sand was dominant on the unvegetated flats, while mud (silt and clay) was abundant in mangrove at Welcome Bay, and all except Transect 2 at Waikareao, where fine sand was measured at all stations. Mud still constituted up to 50 % of the unvegetated intertidal locations (Figure 7.2).

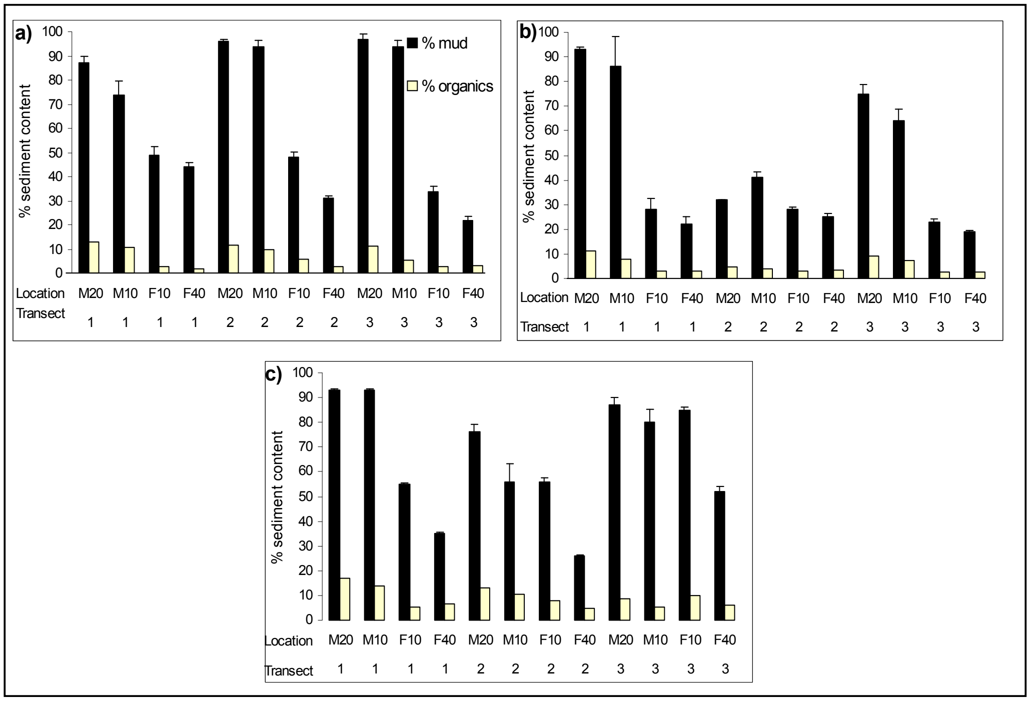


Figure 7.2 Average (± SE) percent mud content and total organic content of sediments collected in mangrove (M20 and M10) and unvegetated sampling locations (F10 and F40) at Welcome Bay (a), Waikareao Estuary (b) and Waikaraka Estuary (c), February 2007.

Organic content was generally higher in mangrove sediments, with the highest values found in Waikaraka, which may reflect the presence of decaying root material after mangrove removal (Figure 7.2).

Increased surface topography was found to occur at most locations in the upper estuary locations of Waikareao and Welcome Bay with maximum increases in surface elevation of 18 mm yr^{-1}. More sediment appears to be accumulating on the bare flats of Transects 1 and 2 at Welcome Bay, compared to the adjacent mangroves.

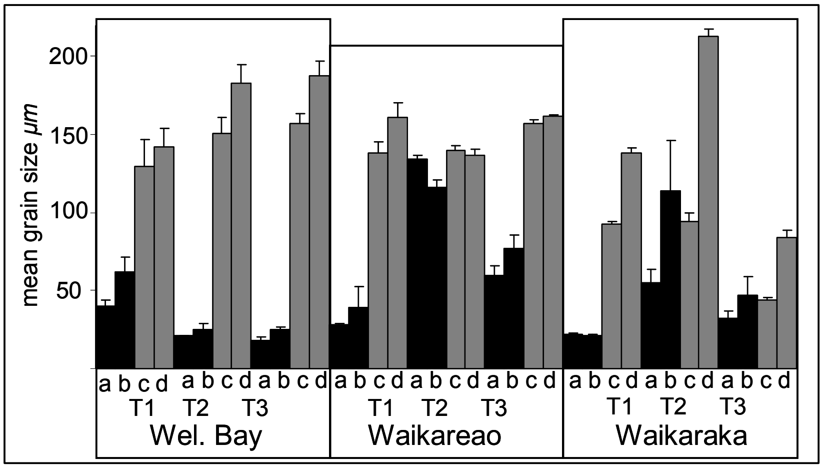


Figure 7.3 **Mean grain size (+ SE) of sediments collected in February 2007 in mangrove (black) and unvegetated sampling locations (grey) at Welcome Bay, Waikareao Estuary and Waikaraka Estuary.**

Erosion, or a fall in surface elevation, is occurring on the bare flats at Waikareao in the vicinity of Transect 3. Erosion rates of over 20 mm yr^{-1} were recorded at Waikaraka due to release of some sediment fines and the decomposition and collapse of mangrove roots after above-ground mangrove vegetation was removed (Table 7.2).

Table 7.2 **Surface elevation changes measured in mangrove (M20 and M10) and unvegetated sampling locations (F10, F40) at Welcome Bay, Waikareao Estuary and Waikaraka Estuary.**

Transect	habitat & distance (m)	W. Bay annual accretion (mm)	Waikareao annual accretion (mm)	Waikaraka annual accretion (mm)
1	M20	2	14	4
1	M10	0	7.5	4
1	F10	13	6	**
1	F40	1	0.6	**
2	M20	6	13	-5
2	M10	5	8	5
2	F10	12	-2.5	-31
2	F40	1	0.4	**
3	M20	18	5	4
3	M10	13	5	4
3	F10	6	-13	-14
3	F40	5	-13	**

7.5.2 Macrobenthic Community Composition

All locations sampled were found to have low macrofaunal abundance. Unvegetated sandflats often had slightly higher numbers of individuals (between 2 and 6 individuals per core) than the adjacent mangrove habitat (1 to 2.7). Similarly, sandflats mostly had slightly higher total taxa compared to mangrove habitat, though the average total taxa at all locations was always < 4 per 0.01 m^2 core (Table 7.5). Comparative differences in univariate indices between habitats were less evident within Waikaraka Estuary however.

The benthic macrofaunal community consisted of gastropods, polychaetes and decapods, with a similar suite of species found across both habitats, though with patchy abundance. Numerically dominant species (listed in Table 7.3) were mostly surface deposit feeders, with the exception of the predators *Perinereis nuntia* and *Cominella glandiformis*. Very few bivalves were found. One cockle (*Austrovenus stutchburyi*) was recorded for the tidal flats of Waikareao, and two individuals in Waikaraka, where one *Macomona liliana* was found at the 40 m station of each of the three transects. Interestingly, the crab *Helice crassa* was more abundant in mangroves compared to unvegetated sites, with the reverse trend found for *Macropthalmus* sp.

Table 7.3 Dominant taxa found in mangrove and unvegetated intertidal flat habitat pooling 2007 sampling data from Waikareao, Welcome Bay and Waikaraka.

Habitat and dominant taxa	Faunal group	feeding mode	% explained for no. of occurrences	% explained for total no. of individuals
Mangroves				
Helice crassa	Decapod	surface deposit feeder	15.9	13.6
Macropthalmus spp	Decapod	surface deposit feeder	11.5	7.7
Eatoniella spp	Gastropod		10.6	24.9
Zeacumantus lutulentus	Gastropod	surface deposit feeder	9.3	11.4
Ceratoneresis spp	Polychaete		7.5	10.5
Tidal flats (10 m from mangroves)				
Macropthalmus spp	Decapod	surface deposit feeder	16.5	15.4
Zeacumantus lutulentus	Gastropod	surface deposit feeder	11.0	13.0
Helice crassa	Decapod	surface deposit feeder	10.4	9.1
Nicon aestuariensis	Polychaete	surface deposit feeder	9.1	9.4
Ceratoneresis spp	Polychaete		7.9	9.8
Tidal flats (40 m from mangroves)				
Zeacumantus lutulentus	Gastropod	surface deposit feeder	14.5	19.1
Macropthalmus spp	Decapod	surface deposit feeder	9.8	7.0
Ceratoneresis spp	Polychaete		9.2	12.7
Scolecolepides spp	Polychaete	surface deposit feeder	7.5	9.4
Nicon aestuariensis	Polychaete	surface deposit feeder	6.4	4.7

Multivariate analyses, displayed in Table 7.4, indicate that the presence of mangroves, location and transect influenced macrofaunal community composition ($p < 0.001$). The significant seasonal effect on mean taxa was due to the presence of crab (*Halicarcinus cookii, Halicarcinus whiteii* and *Hemigrapsus edwardsii*) and polychaete species (*Hetermoastus filiformis* and *Aglophamus macroura*) in summer 2006 that were mostly not found during proceeding sampling events.

Table 7.4 p values from nested PERMANOVA undertaken to assess differences in macrobenthic species composition and species richness amongst sampling locations. Significant p values (< 0.01) are indicated in bold.

Source	Observed	Expected	*p*
Species composition			
Year	1.38	0.95	0.13
Season	2.76	1.71	0.04
Estuary	11.09	11.09	1.0
Transect	**7.20**	**4.13**	**<0.001**
station/position	**2.59**	**0.77**	**<0.001**
Mangrove or flats	**5.19**	**0.77**	**<0.001**
Mean Taxa			
Year	0.101	0.84	0.78
Season	**9.40**	**0.92**	**<0.001**
Estuary	1.04	1.61	0.52
Transect	0.00	0.85	1.0
Station/position	**11.50**	**0.95**	**<0.001**
Mangrove or flats	**29.09**	**0.94**	**<0.001**

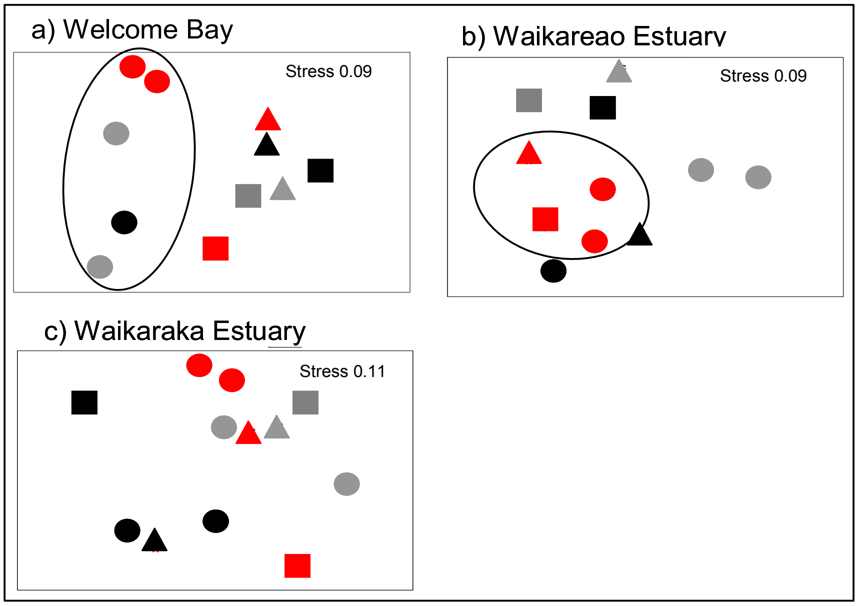


Figure 7.4 Multidimensional scaling (MDS) ordination of macroinvertebrate data from all locations sampled in February 2007. Circles = mangroves; squares = F10; and triangles = F40. Black = Transect 1; red = Transect 2 and grey = Transect 3.

The MDS ordination for macroinvertebrate community data revealed no clear separation of habitats when all sites (estuaries) and locations were combined, consistent with the similar suite of organisms found across mangrove and unvegetated locations.

When sites were analysed separately, however, mangrove and unvegetated habitats separated clearly at Welcome Bay. The four locations along Transect 2 at Waikareao cluster closely, consistent with similarities in sediment characteristics of these locations (Figure 7.4).

7.5.3 Epifauna community composition

Regular occurrences of gastropods were restricted to two species, the mud snail *Amphibola crenata* and the horn shell, *Zeacumantus lutulentus.*

Spatial distribution of *Amphibola crenata* (mudsnail) populations was variable, both within estuaries and between estuaries. Habitat was not found to be a significant factor influencing mudsnail abundance within Welcome Bay when all transects and seasons were combined in a nested ANOVA (Table 7.6). Transect and sampling period appeared to influence abundance however, because of the higher numbers encountered in mangrove habitat in summer 2006, and a relatively low average abundance across all stations of Transect 2 for all seasons (Figure 7.5).

Table 7.5 Results of nested ANOVA on square root transformed data of *Amphibola crenata* counts, for Welcome Bay, summer 2006, winter 2006 and summer 2007.

	df	MS	F	P
Transect	1	7.7	5.6	0.020*
season	1	11.46	8.34	0.005*
transect*habitat	1	2.25	1.64	0.20
transect*season	3	4.33	3.15	0.28*
habitat*season	1	1.06	.77	0.38
transect*habitat*season	3	3.48	2.53	0.62

An almost reverse trend was observed in Waikareao Estuary, where *Amphibola* were found in higher numbers of up to 60 m^{-2} ($\pm$ 30) in mangrove habitat of Transects 2 and 3, compared to average abundances of < 10 m^{-2} at all sandflat/mudflat locations (Figure 7.6). Sampling period was also a significant factor ($p < 0.01$) (Table 7.7).

Table 7. 6 Macroinvertebrate taxa identified from cores collected within each sampling location at Welcome Bay, Waikareao Estuary and Waikaraka Estuary in February 2007. Values are average abundancer per 0.01 m^2 core (n=3). Sand/mudflat stations F10 and F40 are written as 10 m and 40 m (distance from mangrove edge).

Species/taxa		Site	T1 M	T1 10 m	T1 40 m	T2 M	T2 10 m	T2 40 m	T3 M	T3 10 m	T3 40 m
Zeacumantus lutulentu	G	W Bay	0.5	1.0	2.7	0.3	1.3	0.7	1.0	1.3	2.0
		Waikareao									
		Waikaraka						2.0			
Eatoniella sp.	G	W Bay									
		Waikareao	1.7	1.0		1.2	1.0	2.0			0.3
		Waikaraka	1.7	1.2		0.5		1.0			
Amphibola crenata	G	W Bay									
		Waikareao							0.3		
		Waikaraka						1.0	1.0		
Cominella glandiformis	G	W Bay					0.7				0.0
		Waikareao					1.7				0.7
		Waikaraka							1.0	1.0	1.0
Helice crassa	D	W Bay	0.5			0.3			1.0		
		Waikareao		0.5			0.3		0.7		
		Waikaraka	0.3	0.3	1.3		1.0		1.3	1.0	
Macropthalmus sp.	D	W Bay		0.7	0.7	0.3	0.7			1.0	0.3
		Waikareao		0.5	0.3	0.3			0.3	0.7	
		Waikaraka				1.0	1.5		2.0	1.0	1.0
Nicon aestuariensis	P	W Bay		0.3	1.3			0.3		2.0	0.3
		Waikareao		0.5		0.3			0.3	0.3	
		Waikaraka	1.0				1.0	1.0	1.0	2.0	2.0
Family Nereidae	P	W Bay			0.7	0.3	0.3	0.7			
		Waikareao			0.3		2.3	0.3		1.0	2.0
		Waikaraka	0.7	0.3				1.0			
Ceratonereis sp.	P	W Bay			0.3						
		Waikareao			0.3	0.3	0.3	0.3	0.3	2.7	0.3
		Waikaraka	0.5	0.3							
Scolecolepides sp.	P	W Bay	0.5	0.3			0.3	1.7	0.7	0.7	0.7
		Waikareao			0.3	0.3				0.3	0.7
		Waikaraka			2.0						
Perinereis nuntia	P	W Bay							0.7		
		Waikareao			1.3	0.3	0.3			0.3	0.3
		Waikaraka	0.3	0.2		1.0					
Species Richness		W Bay	2.5	1.7	3.3	1.5	2.3	3.3	2.0	3.3	3.0
		Waikareao	1.3	2	2	2.2	3.3	1.3	2	3.7	3
		Waikaraka	1.6	2	2.7	1	2.3	2.3	1.5	1.3	1.7
Abundance		W Bay	1.7	2.7	6	1.5	3.3	5	2.7	5.7	4.7
		Waikareao	1	2.5	2.7	2.3	6	2.7	2	6	4.3
		Waikaraka	2.7	3.7	4.3	1	2.7	2.7	2	1.7	2

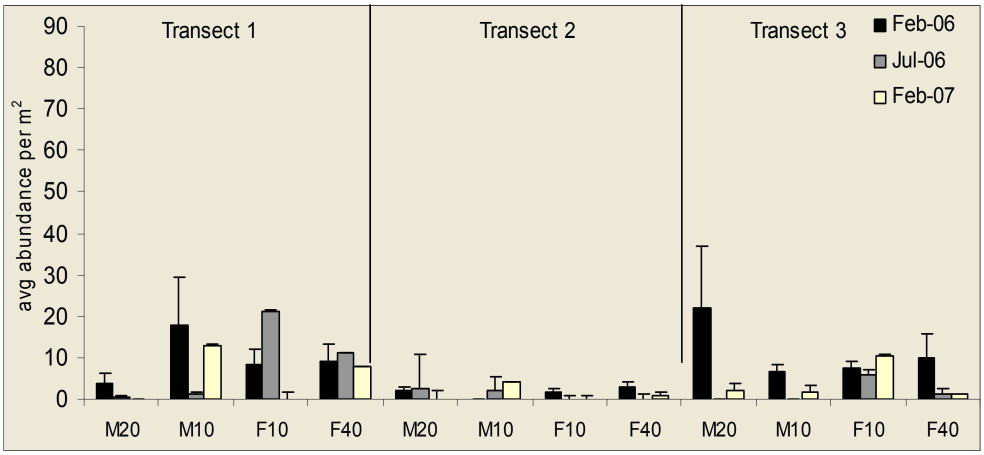


Figure 7.5 Average number (+SE) of ***Amphibola crenata*** (mudsnails) counted in 1 m^2 quadrats (n=3) at all locations in Welcome Bay in February 2006, July 2006 and February 2007.

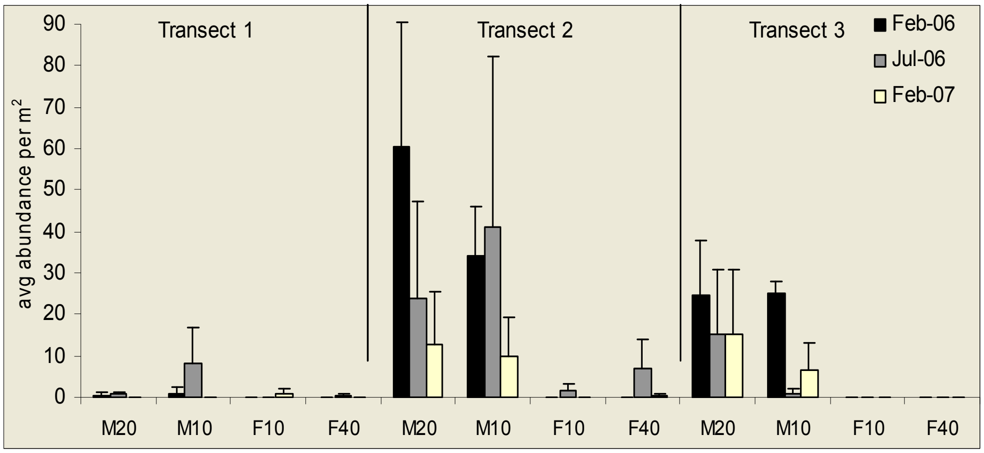


Figure 7.6 Average number (+SE) of ***Amphibola crenata*** (mudsnails) counted in 1 m^2 quadrats (n=3) at all locations in Waikareao Estuary in February 2006, July 2006 and February 2007.

Table 7.7 Results of nested ANOVA on square root transformed data of ***Amphibola crenata*** counts, for Waikareao Estuary, summer 2006, winter 2006 and summer 2007

	df	MS	F	P
Transect	2	56.2	40.7	.000*
habitat	1	221.8	160.5	0.000*
season	2	9.1	6.6	.002*
transect*habitat	2	37.1	26.8	.000*
transect*season	4	6.5	4.7	.002*
habitat*season	2	12.1	8.8	.000*
transect*habitat*season	4	5.2	3.8	.007*

The abundance and distribution of *Amphibola crenata* within Waikaraka Estuary are shown in Figure 7.7. The highest average abundance of 53 m^{-2} (± 5) was observed on the mudflats of the upper estuary (Transect 1), whereas this gastropod was rarely encountered in mangrove habitat.

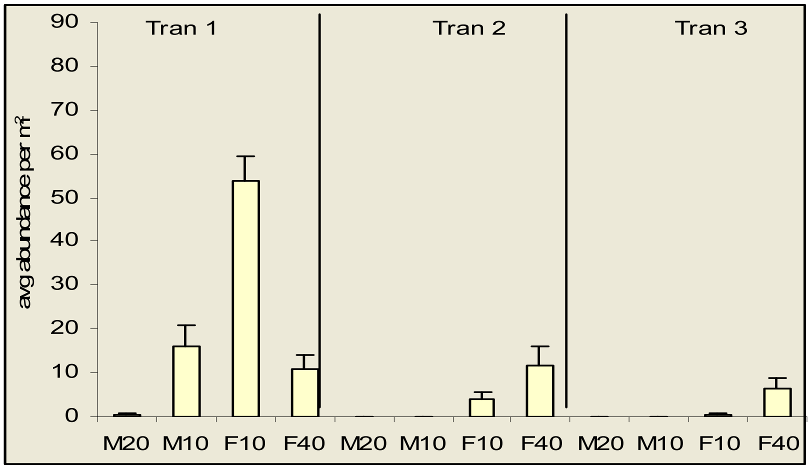


Figure 7.7 **Abundance (+ SE) of *Amphibola crenata* counted in 1 m^2 quadrats (n=3) at all sampling locations in Waikaraka Estuary in February 2007.**

The patchy distribution of *Zeacumantus lutulentus* in Welcome Bay shows some habitat preference, with variable abundance across sampling periods (Table 7.8). Significantly lower numbers of *Zeacumantus* were observed on the sandflats in winter 2006 ($p < 0.05$), although season did not significantly influence already low numbers within the mangrove sites (Figure 7.8).

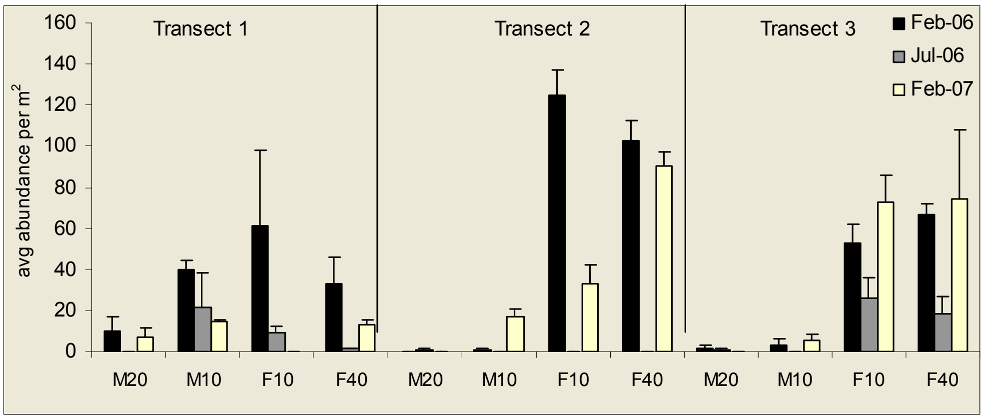


Figure 7.8 **Average number (+ SE) of *Zeacumantus* sp. counted in 1 m^2 quadrats (n=3) at all locations in Welcome Bay in February 2006, July 2006 and February 2007.**

Table 7.8 Results of nested ANOVA on square root transformed data of *Zeacumantus* abundance for Welcome Bay, summer 2006, witner 2006 and summer 2007.

	df	MS	F	P
Transect	1	.7	.23	.630
Season	**1**	**12.8**	**4.03**	**.048***
transect*season	6.0	1.88	.138	
transect*habitat	**1**	**117.9**	**37.06**	**.000***
season*habitat	**1**	**16.0**	**5.02**	**.027***
transect*season*habitat	**3**	**10.2**	**3.21**	**.027***

Zeacumantus lutulentus were rarely encountered in Waikareao Estuary, where individuals were mostly restricted to the mudflat habitats of Transects 1 and 2 in winter 2006, and summer 2007 (Figure 7.9). Abundance varied significantly with transect, habitat ($p < 0.001$) and season ($p < 0.05$), with an almost complete absence within mangrove habitat (except for Transect 3, summer 2006).

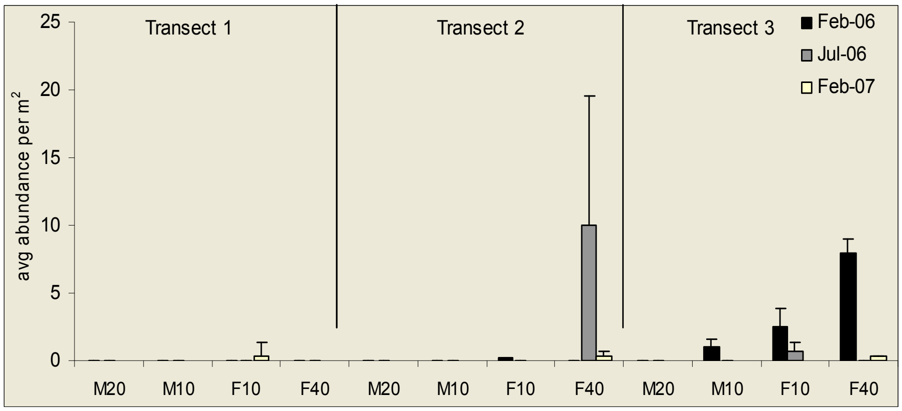


Figure 7.9 Average number (+ SE) of *Zeacumantus* sp. counted in 1 m^2 quadrats (n=3) at all locations in Waikareao Estuary in February 2006, July 2006 and February 2007. Note different scale on Y axis.

Zeacumantus were the dominant taxa on the mudflats of Transects 2 and 3 at Waikaraka Estuary (> 100 m^{-2}). This compared to < 2 m^{-2} across all mangrove locations (Figure 7.10).

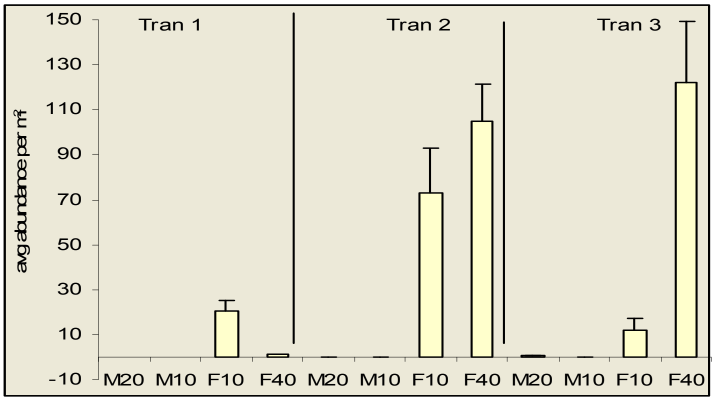


Figure 7.10 Abundance (+ SE) of *Zeacumantus sp.* Counted in 1 m^2 quadrats (n=3) at all sampling locations in Waikaraka Estuary in February 2007.

7.6 DISCUSSION

This study was designed to determine the differences in macrobenthic communities between mangrove habitat and adjacent unvegetated sandflats. A description of surface sediment characteristics was included to investigate the influence of mud accumulation on species diversity and abundance.

The mud (clay and silt) content of mangrove sediments of Welcome Bay, Waikareao and Waikaraka estuaries ranged from around 60% to 98%, while the adjacent flats were dominated by fine sand. Mud content of between 20 and 50% was found on the bare flats however, implying these locations also retain terrestrial sediments. Sediment surfaces are mostly experiencing accretion, as evidenced by positive surface elevation changes measured in both mangrove and bare flat habitats.

7.6.1 Benthic infauna

Benthic communities across both mangrove and bare flat habitats of the study were populated by mud-tolerant organisms which were predominantly surface deposit-feeders such as polychaetes, decapods and gastropods. In contrast to high crab abundance and diversity documented in many tropical estuarine systems (Frusher et al., 1994), two decapods were commonly found (*Helice crassa* and *Macropthalmus hirtipes*), however only in low numbers. *Helice crassa* tolerate increases in mud content (Thrush et al., 2003), although their preference for well-drained compact sediments (Jones and Marsden, 2005) may go some way to explaining the low numbers counted in this study.

The numerical dominance of polychaetes is consistent with findings of other field studies that have identified a correlation between increased terrestrial sedimentation with greater abundance of this group of surface deposit-feeders (Pridmore et al., 1990; MacFarlane and Booth, 2001; Morrisey et al., 2003; Thrush et al., 2003; Ellis et al., 2004). Furthermore, temperate soft-sediments are generally dominated by a small number of species (Pridmore et al., 1990). Annelids have differing habitat preferences, however, and the presence of *Nicon* and *Scolecolepides* species supports modeling predictions of a positive effect of increased mud content with these species (Thrush et al., 2003). The use of 1 mm sieves in this study is likely to have excluded smaller macroinvertebrates and therefore underestimate the absolute abundance of some populations (James et al., 1995). The results of this study provide a good comparison between vegetated and bare intertidal habitats however.

Bivalves were mostly absent across all locations in this study. *Paphies australis* and *Austrovenus stutchburyii* have been found in mangrove habitat of other New Zealand estuaries (Alfaro, 2005), however these were sites dominated by fine sand (with < 15% mud), as opposed to the mud-dominated sediments reported here. It is likely that the relatively high mud content of the bare flats and > 90% mud of the mangrove habitat in this study is a causal factor in the absence of bivalves as sensitivity to increased turbidity and sediment mud content of filter-feeding invertebrates is well documented (Ellison and Farnsworth, 2000; Thrush et al., 2003; Norkko et al., 2006).

Differences in benthic communities were detected between the two key habitats, mostly as a result of slightly higher abundances of the same suite of organisms found in the unvegetated flats more so than the occurrence of habitat-specific taxa. The low benthic diversity and abundance documented for both the mangrove and bare flat habitats in this study is consistent with observations of declining diversity and abundance associated with increasing sediment mud content (Thrush et al., 2003; Ellis et al., 2004; Lohrer et al., 2004; Thrush et al., 2004; Rodrigues et al., 2006). Sedimentation of as little as 3 mm has been considered to have a deleterious effect on macrobenthic communities (Lohrer et al., 2004), so it is reasonable to assume that the sedimentation rates of up to 18 mm yr^{-1} reported in this study are sufficient to influence the composition of benthic fauna. Field observations of extensive bivalve beds (predominantly *Austrovenus stutchburyi*) buried 10 – 15 cm below the estuary surface indicate a large environmental change has occurred to shift the benthic community from one dominated by filter-feeding organisms to one now composed of macroinvertebrates with feeding strategies adapted to a mud-dominated environment.

7.6.2 Surface macrofauna

Patterns of epifaunal distribution were unique to each estuary, however the similarly low diversity of species was limited to two key species; *Amphibola crenata* and *Zeacumantus lutulentus.* In Waikareao Estuary, *Amphibola crenata* were found in higher abundance in mangrove habitat, whereas they were almost completely excluded from mangroves in Waikaraka Estuary. *Zeacumantus lutulentus* were rarely found in numbers > 2 m^{-2} in Waikareao, compared to 50 to 120 m^{-2} counted on the mudflats/sandflats of Welcome Bay and Waikaraka Estuary. These variable results demonstrate the patchy distribution of gastropod populations in estuarine environments, which commonly occurs (Thrush et al., 1994; Chapman and Tolhurst, 2004) in response to variable organic content, mud content, pH, salinity and nutrient availability (Macfarlane and Booth, 2001). Epifaunal communities may be further controlled by tree cover (and therefore shading), leaf litter, and tidal characteristics (Lundquist et al., 2006).

The same gastropod species identified in this study have also been reported elsewhere in New Zealand where mud content in sediments is usually >20% (May, 1999; Morrisey et al., 2003; Ellis et al., 2004). The presence of muddy surface sediments may also explain the absence of *Diloma subrostrata* and *Turbo smaragdus* in the Tauranga Harbour sites, as these gastropods have been found in pneumatophore zones of a sandier New Zealand estuary (Alfaro, 2005).

The highest numbers of *Zeacumantus lutulentus* were observed on the mudflats of Transect 2 and 3, in Waikaraka Estuary. This could be an opportunistic response to increased sediment organics made available by the mangrove clearing at these sampling locations.

Mangrove habitat appeared to limit the abundance of gastropods in some instances, although occational relatively high densities were observed under mangroves. For example, mudsnails appeared to prefer the mangrove habitat of

Transects 2 and 3 in Waikareao Estuary. There are numerous potential controls that could explain this spatial distribution. Additional to influences of pH, temperature and food sources (which were not measured in this study) are some potential morphological effects. Firstly, the canopy cover of the mangrove habitat along Transects 2 and 3 at Waikareao Estuary, is typically <50%, therefore there is only patchy shade at these sites (Figure 7.11). Pneumatophore density and mud content are also lower than observed in mangrove habitat at Welcome Bay and Waikaraka Estuary.

Figure 7.11 ***Amphibola crenata*** **grazing amongst mangrove seedlings and pneumatophores at Waikareao Estuary (Transect 3).**

7.7 CONCLUSION

This study has provided further evidence of the impacts of terrestrial sedimentation on benthic communities. The sediment characteristics of the mangrove and unvegetated intertidal habitat were markedly different. It appears, however, that the silt and clay content of the unvegetated habitat was still sufficient to limit macrofaunal species diversity and abundance, exclude bivalves and encourage a benthic community dominated by deposit-feeding polychaetes, gastropods and decapods. Any coastal management strategies employed to improve the ecosystem health of an estuary must therefore consider the negative effects of catchment-based sediment loads on the diversity of benthic communities that exist both in mangrove habitat and on adjacent bare intertidal surfaces.

Variabile gastropod community composition was observed across the three sub-estuaries of this study. Low gastropod diversity was common across both the mangrove and the sandflat/mudflat habitats, which suggests that a range of factors are likely to be controlling distribution patterns, rather than simply the presence or absence of mangroves. *Amphibola crenata* were encountered in highest densities under mangroves within Waikareao Estuary, which is contrary to a growing public perception that mangroves exclude this common, mud-tolerant gastropod.

7.8 ACKNOWLEDGEMENTS

Thanks to Penny Cooke for her untiring field support and to Sarah Hailes for the lead author’s introduction to invertebrate taxonomy. Assistance in the field by a number of postgraduate students was greatly appreciated, and enjoyed.

7.9 PHOTO GALLERY

Figure 7.12 Shell layer buried by 15 cm of fine sand, Welcome Bay, Transect 3.

Figure 7.13 *Austrovenus stutchburyi* shells found in the buried shell layer at Transect 3, Welcome Bay. April 2005.

7.10 LITERATURE CITED

Alfaro, A.C., 2005. Benthic macro-invertebrate community composition within a mangrove/seagrass estuary in northern New Zealand. *Estuarine, Coastal and Shelf Science* 66, 97-110.

Chapman, M. G. and Tolhurst, T. J., 2004. The relationship between invertebrate assemblages and bio-dependant properties of sediment in urbanized temperate mangrove forests. *Journal of Experimental Marine Biology and Ecology* 304(1), 51-73.

Cole, R. G., Hull, P. J. and Healy, T. R., 2000. Assemblage structure, spatial patterns, recruitment, and post-settlement mortality of subtidal bivalve molluscs in a large harbour in north-eastern New Zealand. New Zealand *Journal of Marine and Freshwater Research* 34, 317-329.

Cummings, V., Thrush, S., Hewitt, J., Norkko, A., and Pickmere, S., 2003. Terrestrial deposits on intertidal sandflats: sediment characteristics as indicators of habitat suitability for recolonising macrofauna. *Marine Ecology Progress Series* 253, 39-54.

Ellis, J., Nicholls, P., Craggs, R., Hofstra, D., and Hewitt, J., 2004. Effects of terrigenous sedimentation on mangrove physiology and associated macrobenthic communities. *Marine Ecology Progress Series* 270, 71-82.

Ellison, A.M., and Farnsworth, E.J., 2000. Mangrove Communities, in: Bertness, M.D., Gaines, S.D., and Hay., M.E., (Eds.) Marine Community Ecology: Sinauer Associates, Sunderland, MA, USA, 550 pp.

Frusher, S.D., Giddins, R.L., and Smith, T.J., 1994. Distribution and abundance of grapsid crabs (Grapsidae) in a mangrove estuary: effects of sediment characteristics, salinity tolerances, and osmoregularity ability. *Estuaries* 17, 647-654.

Furukawa, K., Wolanski, E., and Mueller, H., 1997. Currents and sediment transport in mangrove forests. *Estuarine Coastal and Shelf Science* 44, 301-310.

Hope H.M. (2002) Sediment dynamics of Waikaraka Estuary, a small, semi-enclosed estuarine system in the upper reaches of a developed harbour. MSc thesis, University of Waikato, New Zealand, 126 pp.

James, R.J., Lincoln Smith, M.P., and Fairweather, P.G., 1995. Sieve mesh-size and taxonomic resolution needed to describe natural spatial variation of marine macrofauna. *Marine Ecology Progress Series* 118, 187-198.

Jones, M.B., and Marsden, I.D., 2005. Life in the Estuary, illustrated guide and ecology. Canterbury University Press, Christchurch, New Zealand, 179 pp.

Kathiresan, K. and Bingham, B., 2000. Biology of Mangroves and Mangrove Ecosystems. *Advances in Marine Biology*, 40, 84-259.

Lohrer, A.M., Thrush, S.F., Hewitt, J.E., Berkenbusch, K., Ahrens, M., and Cummings, V.J., 2004. Terrestrially derived sediment: response of marine macrobenthic communities to thin terrigenous deposits. *Marine Ecology Progress Series* 273, 121–138.

Lundquist, C., S. Thrush, S., Hewitt, J., Halliday, J., MacDonald, I., Cummings, V., 2006. Spatial variability in recolonisation potential: influence of organism behaviour and hydrodynamics on the distribution of macrofaunal colonists. *Marine Ecology Progress Series* 324, 67-81.

MacFarlane, G.R., and Booth, D.J., 2001. Estuarine macrobenthic community structure in the Hawkesbury River, Australia: relationships with sediment physicochemicalal and anthropogenic parameters. *Environmental Monitoring and Assessment* 72, 51-78.

May, J., 1999. Spatial variation in litter production by the mangrove *Avicennia marina* var. *australasica* in Rangaunu Harbour, Northland, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 33, 163-172.

Morrisey, D.S., Skilleter, G.A; Ellis, J.I.; Burns, B.R.; Kemp, C.E.; 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, 581-592.

Norkko, J., Hewitt, J.E., and Thrush, S.F., 2006. Effects of increased sedimentation on the physiology of two estuarine soft-sediment bivalves, Austrovenus stutchburyi and Paphies australis. *Journal of Experimental Marine Biology and Ecology* 333, 12-26.

Park, S., 2003. Marine Sediment and Contaminants Survey (2001-2003): Environment Bay of Plenty Operations Report. Environment Bay of Plenty, Whakatane, 23 pp.

Pridmore, R.D., Thrush, S.F., Hewitt, J.E., and 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, 81-96.

Quartel, S., Kroon, A., Augustinus, P.G.E.F., Van Santen, P., and Tri, N.H., 2007. Wave attenuation in coastal mangroves in the Red River Delta, Vietnam. *Journal of Asian Earth Sciences* 29, 566-575.

Rodrigues, A.M., Meireles, S., Pereira, T., Gama, A., and Quintino, V., 2006. Spatial patterns of benthic macroinvertebrates in intertidal areas of a Southern European estuary: the Tagus, Portugal. *Hydrobiologia* 555, 99-113.

Surman, M., 1999, Tauranga Harbour. Sediment Source Survey, Environment B.O.P Operations Report: Whakatane, Environment Bay of Plenty, 141 pp.

Thrush, S., Pridmore, R. D., Hewitt, J.E., 1994. Impacts on soft-sediment macrofauna: the effects of spatial variation on temporal trends. *Ecological Applications* 4(1), 31-41.

Thrush, S.F., Hewitt, J.E., Norkko, A., Nicholls, P.E., Funnell, G.A., and Ellis, J.I., 2003. Habitat change in estuaries: predicting broad-scale responses of intertidal macrofauna to sediment mud content. *Marine Ecology-Progress Series* 263, 101-112.

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

Chapter 8
Summary and Conclusions
Temperate mangrove dynamics: consequences of expansion and removal

8.1 INTRODUCTION

The expansion of mangroves has led to these native plants being considered a maligned pest in certain localities around New Zealand. This expansion has resulted in ecological modifications to the effected estuaries in New Zealand's north island. Their presence is often blamed for other changes that have occurred, such as reduced bivalve abundance and the development of softer and muddier intertidal surfaces, even though these perceptions are based largely on anecdotal evidence. The development of management strategies for these estuaries, in response to the public opposition to mangrove conservation, is made more difficult by the lack of empirical data that identifies the physical and ecological implications of both the presence of mangroves and their continued lateral expansion. This lack of data makes it difficult for decision-makers to predict the types and scale of impacts on the surrounding estuarine system resulting from any removal of established mangroves.

From a botanical viewpoint, the New Zealand *Avicennia* forests present an interesting system for detailed physiognomic study. Mangrove stands located in estuaries such at Tauranga Harbour are growing close to the southern limit of *Avicennia's* natural range, and at this latitude the climate will influence mangrove growth forms. The other interesting factor is that as there is only one species, inter-species competition can be excluded as a variable in the study of plant morphology.

With these conditions in mind, an extensive field study was developed to investigate the morphological characteristics of temperate mangrove forests in Tauranga Harbour. The effect and interactions of above-ground mangrove vegetation on the movement and deposition of suspended sediment was assessed, and the resultant impacts on estuarine geomorphology were characterised. The study also evaluated the influence of mangrove expansion on benthic community composition.

8.2 KEY FINDINGS

8.2.1 Mangrove physiognomy

Mono-specific stands of *Avicennia marina* in Tauranga Harbour were found to be spatially variable in their morphological characteristics. They can, however, be broadly described as densely populated, temperate *Avicennia* forests with close to 100 % canopy cover in most locations. The latitudinal location of the harbour appears to be limiting vertical growth to < 10 cm yr^{-1} and plant heights rarely exceeded 1.2 m. Despite a slow growth rate, vivipary proved to be productive, with seedling survival of up to 80 % on un-vegetated sandflats, suggesting continued sandflat colonisation is probable. Counts of pneumatophores within mangroves of Tauranga Harbour identified some of the highest densities reported to date, for both New Zealand mangroves and temperate mangroves elsewhere, with pneumatophores protruding at densities of up to ~ 700 m^{-2}. This may be a response to the > 90 % mud content of surface sediments.

Mangroves are known to display a greater below-ground to above-ground ratio of biomass than their terrestrial counterparts. The climate affecting mangrove growth in New Zealand also influences the production of structural and feeding roots. Not surprisingly, the limited plant growth observed in Tauranga Harbour was reflected in relatively lower below-ground biomass, where the root networks provided biomass of between 2 and 4 kg per m^{-2} (20 to 40 t ha^{-1}). Accurate estimations of decomposition rates after tree mortality were not achieved here, partly because of the spatial variability relative to the age of the plants, and tidal dynamics (i.e. flushing rates). This study has, however, provided some indication of the density of below-ground material that can be found in New Zealand mangroves, as a first step to further investigate the processes and timeframes involved in their decomposition after mangrove removal.

8.2.2 Mangrove expansion

Periods of greatest lateral migration of the mangrove fringe occurred at different times within the three study sites, although prior to 1982 mangroves were mostly limited in their coverage to narrow fringes in the mid-estuary regions, and wider stands in the upper reaches. There are a number of reasons why this may be the case, and it is not possible to determine if mangrove populations had been extensive prior to urban and agricultural development of the estuarine margins. However, one necessary requirement for successful mangrove colonisation is related to surface topography.

Using Waikaraka Estuary as a case study, the link between periods of increased sedimentation stemming from modified land-use practices and the onset of mangrove colonisation was recognised. The use of ^{210}Pb dating identified a period of high sedimentation in Waikaraka Estuary between *c* 1920 and 1950. Substrate accretion continued at a slower rate after that time which, cumulatively, provided sufficient uplift of the harbour floor relative to mean sea level to promote the survival of *Avicennia* seedlings across most of the upper estuary.

The upper reaches of all three study sites (Welcome Bay, Waikareao Estuary and Waikaraka Estuary) are experiencing substrate accretion, and are therefore susceptible to on-going mangrove colonisation. The area that is most likely to experience extensive mangrove expansion is the intertidal sand-flat positioned west of Matupae Island, in Waikareao Estuary. The net substrate accretion of 4 mm yr^{-1} in this region would suggest that surface topography will remain within the lower elevation limit required for seedling survival. Indeed, field observations of a scattering of one to two year old seedlings across a 40 m width of sandflats supports this prediction. Frost-kill or uprooting from wave action may provide some limiting influence, though the difficulty for planners lies in the inability to predict any return-periods of these events.

8.2.3 Mangroves and estuarine geomorphology

The interaction of mangrove structures and tidal currents promotes a depositional environment, and once sediment has settled, it is less likely to be re-suspended.

This geomorphic process was identified through the use of sediment traps which demonstrated a discrepancy between sediment accumulation rates (i.e. sediment availability) and substrate accretion, whereby mangroves received less sediment than the adjacent unvegetated sand/mudflat but experienced a higher rate of substrate accretion.

Mangroves are driving geomorphological change on two main fronts: (1) the selective trapping of silt and clay particles, resulting in a muddy substrate; and (2) enhancing substrate accretion. Interestingly, substrate accretion is also occurring in mangrove habitat positioned on fine-sand deposits (Waikareao estuary), which suggests that the determination of elevation levels is a more important indicator than sediment type when predicting mangrove expansion. Any reduction in sediment loads into an estuary, regardless of grain size, is therefore an important consideration for the spatial management of mangrove populations.

The scale of topographical change associated with mangrove expansion will be influenced by the migration rate of the mangrove fringe. Sedimentation and substrate accretion occurs mostly within a 20 to 30 m width of fringing mangroves, therefore, this accretion zone will move relative to the progressing mangrove fringe. Continued expansion may also result in the development of hyper-saline depressions toward the landward margins of the mangrove stands as a result of this spatial gradient in sedimentation.

8.2.4 What does this mean for intertidal benthic ecology?

The assumption that mangroves are the cause of bivalve displacement was disproved in this study, at least for the upper reaches of Tauranga Harbour's sub-estuaries. Benthic macroinvertebrate surveys of Welcome Bay, Waikareao Estuary and Waikaraka Estuary highlighted an absence of bivalves across both mangrove and sandflat/mudflat habitats. The macroinvertebrate communities were dominated by deposit-feeding organisms, in slightly lower abundances in mangrove habitat when compared with unvegetated sites.

The muddy nature of the surface sediments and high rates of sedimentation, will be imparting some influence on the benthic community composition.

The presence of relict shellfish beds under 10 – 15 cm of sediment indicate a sharp transition in benthic ecology which has most likely occurred as a result of a modified sediment regime. Increased contaminant loads (nutrients, heavy metals) and reduced tidal inundation may also be important, however these characteristics were not quantified in this study.

8.2.5 Impacts of mangrove removal

Mangroves provide a valuable carbon sequestration role in the coastal environment. By removing mangroves, the store of carbon is released. A further potential negative impact of mangrove removal is the likelihood of increased relative sea-level. The removal of above-ground mangrove vegetation initiates a decline in surface elevation up to 17 mm yr^{-1}. If we superimpose projected sea-level rise onto this 17 mm yr^{-1}, it could be expected that cleared estuaries may in effect experience an increase in relative sea-level of approximately 20 mm yr^{-1}. The relatively deeper tidal inundations that would result may stress remaining mangrove and saltmarsh habitat. This is an important consideration, particularly as much of the present-day landward boundaries of the Tauranga sub-estuaries cannot accommodate any landward migration of wetland vegetation.

The release of silts from surface sediments occurs as a result of mangrove removal, however the length of time before a return to a sandy substrate will be dependent on the flushing ability of the estuary in question and the depth of the silt beds. In general this appears to be a process that must be looked at in terms of at least 3 to 5 years (considering silt was still part of the sediment matrix of cleared sites monitored in this study after 2 years), and potentially much longer in locations that receive limited tidal flushing (i.e. near hard structures such as groins and causeways).

Estimates of sediment loads associated with mangrove removal suggest up to 9 kg of sediments may be released over a 12 month period for each square metre that is cleared, including biogenic material of micro and macro-organics. This estimate is a maximum value only, in that the effect of root compaction (as opposed to sediment erosion) could not be quantified.

Mitigation measures in relation to mangrove removal could include the retention of a mangrove border which would trap some of the additional fine sediments released from the cleared areas. A "catch-22" situation emerges when contemplating the most appropriate form of mangrove management. If a mangrove fringe is maintained, then some form of on-going management will be required to restrict seedling progression. Conversely, if mangroves are cleared to the landward margins of the site (i.e. total clearance), increased silt deposits could potentially blanket intertidal areas and impact on existing benthic communities.

The downstream (seaward) effects of mangrove clearance will be dependent upon the tidal regime and the incidence of surface waves required to flush sediment fines out of the estuary. Other impacts of mangrove removal that were not quantified in thus study are:

(1) the effect of increased organic loads on the production of algae (i.e. potential for algal blooms);
(2) the chemical environment of the remaining substrate, which is high in sulfates in the early months;
(3) the interaction of mangrove root decomposition, root compaction, and sediment release; and
(4) changes to benthic communities in response to removal and subsequent sedimentological changes.

8.3 IMPLICATIONS FOR COASTAL MANAGERS

The findings discussed above are unlikely to result in simple solutions to the 'mangrove issue'. However, some key considerations pertinent to the development of estuarine management strategies for mangrove-fringed estuaries are:

- The continued substrate accretion that is occurring on bare sandflat locations in the upper estuaries is an indicator that mangrove expansion will continue, at least to the lower-elevation-limit of *c* 0.3 m above Mean

Sea Level. Some mitigation will be required if the maintenance of unvegetated sandflats is an ecological objective. Local catchment care groups around Tauranga Harbour have been clearing mangrove seedlings for a number of years, and are well aware that clearing seedlings is a far easier task than removing established shrubs.

- The removal of mangroves in the upper estuaries is not going to necessarily provide a suitable environment for the rehabilitation of bivalve populations. This study suggests other factors are modifying the composition of benthic communities.

- Rates of physical change, or ‘recovery’ of the harbour floor are likely in the event of mangrove removal, however, this is a slow process which could be further slowed or reversed, if too much vegetation is removed, or if removed too quickly. It is vital that any consideration of mangrove removal includes an assessment of the tidal regime and its ability to flush out the increased sediment loads.

- The maintenance of a mangrove fringe is likely to assist in trapping some of this released sediment, however it will then present another management dilemma in that the removal of seedlings will be required *ad infinitum* to prevent any future colonisation.

8.4 PROSPECTS FOR FUTURE RESEARCH

This study has investigated the geomorphological impacts of mangrove expansion and mangrove removal. A valuable next step would be to quantify the processes of root compaction and root decomposition after mangroves are removed. An increased understanding of how these processes influence surface topography would assist with any future assessments of sediment budgets and improve the accuracy of any sediment transport modeling associated with mangrove clearance. The placement of marker horizons would evaluate the scale of which sub-surface processes such as compaction contribute to changing surface elevation.

Benthic macro-invertebrate populations of mangrove and intertidal sandflats were described in this study. Due to the complexities of animal migration and recruitment, a multiple year study would be required to determine the impact of mangrove removal on benthic ecology, and to identify any 'recovery' resembling a filter-feeding dominated benthic population.

The removal of mangroves from intertidal environments is a fairly new concept to the coastal managers and researchers in New Zealand. Experimental flume studies could facilitate a greater understanding of complex processes, such as:

- any increase or decrease in biofilm production after mangrove removal, and the influence on erosion potential of surface sediments;
- decomposition pathways in cleared areas to identify the temporal changes to the sediment chemistry; and
- the influence of intermediate bed roughness of cleared mudflat surfaces, as well as influence of the high pneumatophore densities, on tidal currents and sedimentation.

8.5 CONCLUSION

Coastal managers are striving to find a balance between social values pinned to coastal environments, and sustaining the ecological diversity of these ecosystems. This study has identified the potential for continued mangrove colonisation, and the potential for a significant elevation to the sediment budget should mangroves be removed. It will be a balancing act on the part of the coastal manager to satisfy the social and ecological objectives associated with estuarine management, however with a detailed knowledge of the site-specific processes involved, some balance can be achieved.

Intertidal sediment cores: Waikareao Estuary, Tauranga Harbour

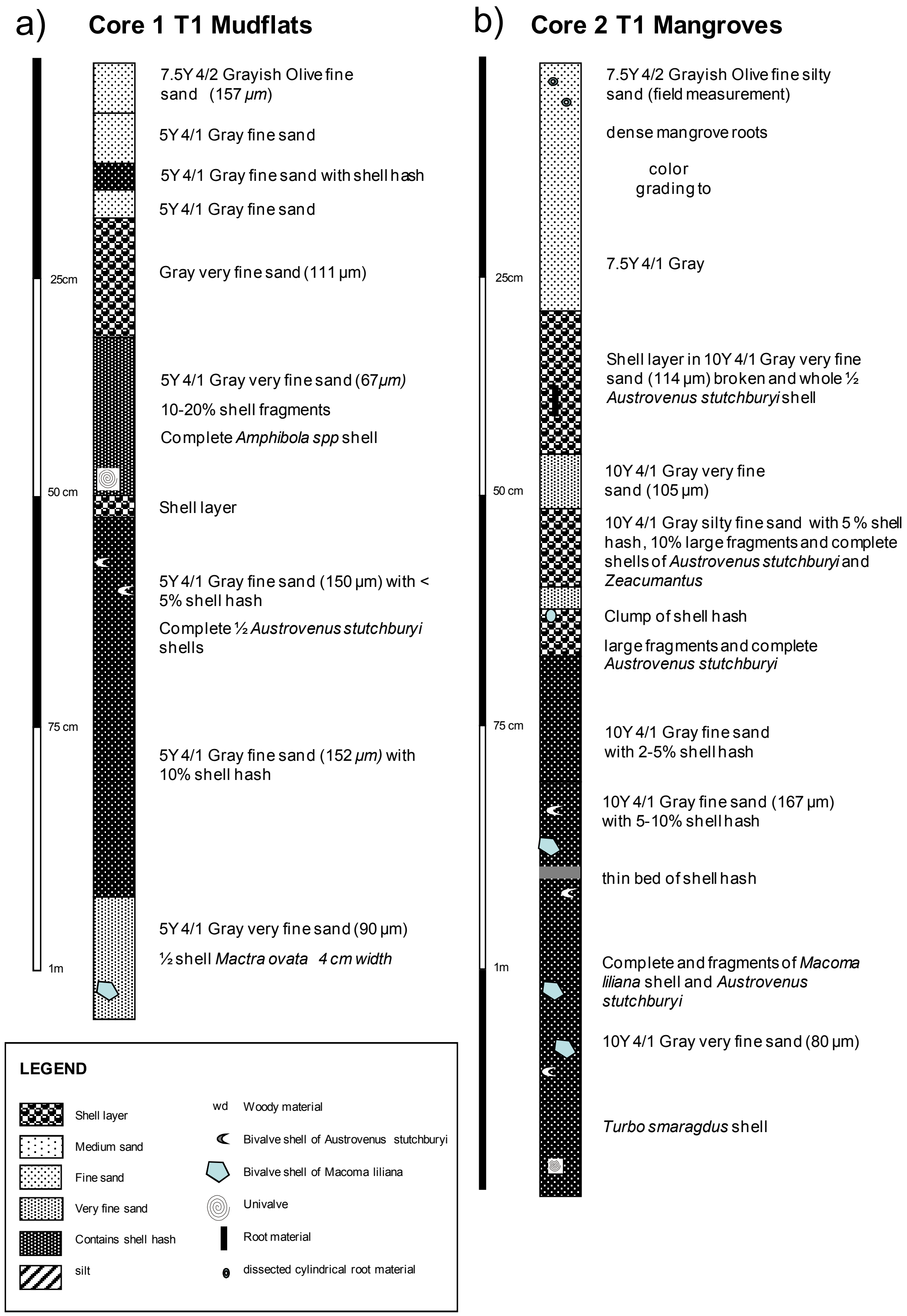
Collected by:

Chris McKinnon, Penelope Cooke, Debra Stokes

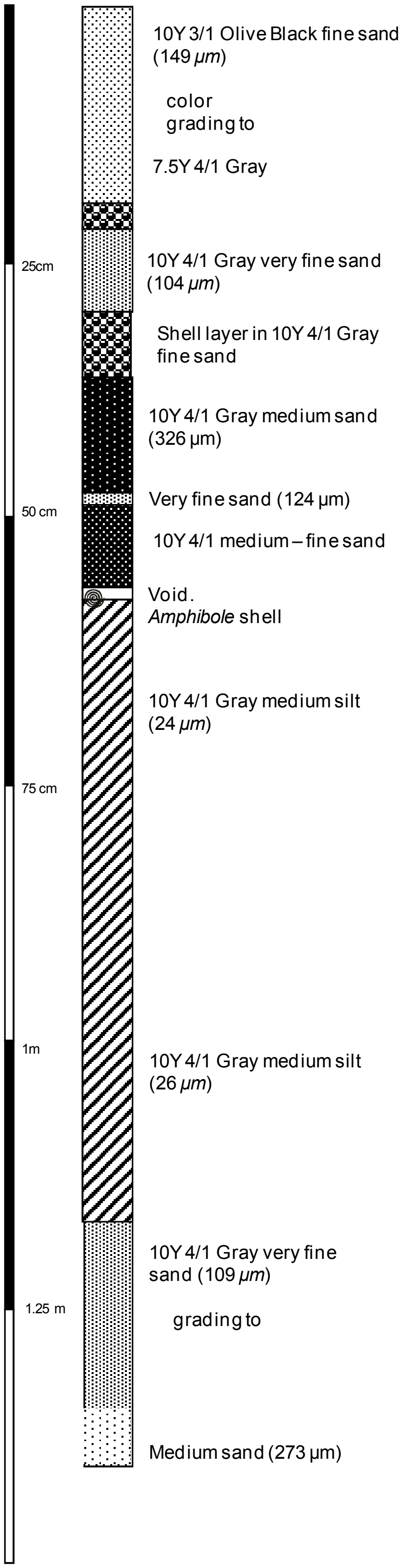
15 November, 2005, using custom-built vibracorer (courtesy of Chris McKinnon)

CORE LOCATIONS

Core a)	Transect 1 mudflats. Core depth 1.0 m	E 2788140 N 6385175
Core b)	Transect 1 mangrove habitat. Core depth 1.5 m mangroves, 1.5m	E2788102 N 6385156
Core c)	Transect 2 sandflats. Core depth 1.8 m	E2787876 N 6385841
Core d)	Transect 2 mangrove habitat. Core depth 1.4 m	E2787833 N 6385827


a)
Core 1 T1 Mudflats
25cm
50 cm
75 cm
1m
7.5Y 4/2 Grayish Olive fine sand (157 *µm*)
5Y 4/1 Gray fine sand
5Y 4/1 Gray fine sand with shell hash
5Y 4/1 Gray fine sand
Gray very fine sand (111 µm)
5Y 4/1 Gray very fine sand (67*µm)*
10-20% shell fragments
Complete *Amphibola spp* shell
Shell layer
5Y 4/1 Gray fine sand (150 µm) with < 5% shell hash
Complete ½ *Austrovenus stutchburyi* shells
5Y 4/1 Gray fine sand (152 *µm)* with 10% shell hash
5Y 4/1 Gray very fine sand (90 µm)
½ shell *Mactra ovata 4 cm width*
b)
Core 2 T1 Mangroves
25cm
50 cm
75 cm
1m
7.5Y 4/2 Grayish Olive fine silty sand (field measurement)
dense mangrove roots
color grading to
7.5Y 4/1 Gray
Shell layer in 10Y 4/1 Gray very fine sand (114 µm) broken and whole ½ *Austrovenus stutchburyi* shell
10Y 4/1 Gray very fine sand (105 µm)
10Y 4/1 Gray silty fine sand with 5 % shell hash, 10% large fragments and complete shells of *Austrovenus stutchburyi* and *Zeacumantus*
Clump of shell hash
large fragments and complete *Austrovenus stutchburyi*
10Y 4/1 Gray fine sand with 2-5% shell hash
10Y 4/1 Gray fine sand (167 µm) with 5-10% shell hash
thin bed of shell hash
Complete and fragments of *Macoma liliana* shell and *Austrovenus stutchburyi*
10Y 4/1 Gray very fine sand (80 µm)
Turbo smaragdus shell
LEGEND
Shell layer
Medium sand
Fine sand
Very fine sand
Contains shell hash
silt
wd Woody material
Bivalve shell of Austrovenus stutchburyi
Bivalve shell of Macoma liliana
Univalve
Root material
dissected cylindrical root material

c) **Core 3 T2 Sandflats**



d) **Core 4 T2 Mangroves**