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Heavy Metal Contamination from Landfills in Coastal Marine Sediments: Kiribati and New Zealand

A thesis
submitted in partial fulfilment
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Abstract

Landfill leachates are a concern in the Pacific Region where they may contribute contaminants to the coastal marine environment. Poor waste management and pollution of coastal waters are amongst the major environmental problems in Kiribati, particularly in South Tarawa. An investigation of areas adjacent to coastal landfill sites; Betio, Kiribati and Auckland, New Zealand was undertaken. The Kiribati case study investigated metal contamination in marine sediments at an operational landfill while the New Zealand study was adjacent to a coastal landfill decommissioned in the 1970s. Surficial sediments (top 15 cm) were collected along transects. At both the New Zealand and Kiribati sites, 3 transects adjacent to the landfill and 1 control transect were sampled. The sediments were analyzed for particle size distribution, organic matter content, and Cd, As, Cr, Cu, Pb, Hg, Ni and Zn concentrations. The pH was also measured.

The Kiribati study site had a groundwater pH of 7.14 - 8.85, and sediment materials were dominated by sand with a low organic matter content (1.60 - 2.21 %). At the Kiribati sites Cd, As, and Ni were below the detection limits. The Cr, Cu, Pb and Zn concentrations were lower at the Kiribati control transect than the landfill transects. Cr level decreased away from the landfill indicating the landfill as a possible source. However, Cu and Zn did not show any distribution pattern suggesting other potential sources (port and shipwrecks) may have contributed to the elevated levels. At the Kiribati landfill and control transects the Cr, As, Cd, Cu, Pb, Ni, and Zn concentrations were below the Effects Range-Low (ERL) and the Threshold Effects Levels (TEL) of the Sediment Quality Guidelines (SQGs) of the National Oceanographic and Atmospheric Administration (NOAA) and the Florida Department of Environmental Protection, indicating no potential adverse ecological effects on the biota. At the Kiribati control site the Cr, As, Cd, Cu, Pb, Ni, and Zn concentrations were within the background ranges published for clean reef sediment but the concentration at the landfill transects exceeded the background ranges. The

concentration of mercury at both the control and landfill transects in Kiribati exceeded the Effects Range-Median (ERM) and the Probable Effects Level (PEL) of the SQGs indicating potential adverse ecological effects on the local benthic communities.

The New Zealand study site sediments had a pH of 6.22 - 7.24, and comprised up to 90 % clay/silt, with an organic matter content of 5 - 22 %. At the New Zealand landfill transects Arsenic concentrations decreased away from the landfill indicating the landfill as a possible source. Other metals such as Cr did not show a pattern of distribution along the transects, or with depth, suggesting that the landfill was not the only source of these metals. There was a weak correlation between organic matter content, particle size distribution, and metal enrichment.

At the New Zealand site, there were no marked differences in metal concentrations between the landfill and control transects suggesting the landfill was not the only source of metals and that the wider urban or industrial run-off may have contributed. All the metal concentrations, except Hg and Zn, exceeded the ERL and the TEL values indicating the potential for adverse ecological effects of metals on the benthic communities. At the New Zealand site the Hg and Zn concentrations exceeded both the ERM and PEL of the SQGs and are considered highly contaminated.

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Acronyms

ADB	Asian Development Bank
ARC	Auckland Regional Council
BTC	Betio Town Council
ECD	Environment and Conservation Division
EDXRF	Energy Dispersive X-Ray Fluorescence
EEZ	Exclusive Economic Zone
EIA	Environmental Impact Assessment
ERL	Effects Range-Low
ERM	Effects Range-Median
LOI	Loss on Ignition
MAF	Ministry of Agriculture and Forestry
MELAD	Ministry of Environment, Lands and Agricultural Development
NOAA	National Oceanographic and Atmospheric Administration
PEL	Probable Effects Level
PICs	Pacific Island Countries
RMA	Resource Management Act
SAPHE	Sanitation, Public Health and Environmental Improvement Project
SOPAC	South Pacific Applied Geoscience Commission
SPREP	Secretariat of the Pacific Regional Environment Programme
SQGs	Sediment Quality Guidelines
TEL	Threshold Effects Level
UNEP	United Nations Environmental Programme
WCC	Waitakere City Council

Chapter One: Introduction

1.1 Background

The study of marine sediments provides useful information in marine, environmental and geochemical research about pollution of the marine environment (Calace *et al.* 2005). Urban developments and industrial activities contribute to the introduction of significant amounts of contaminants (among them trace metals) into the marine environment and affect directly the coastal systems where they are often deposited (Angelidis 1995; Dassenakis *et al.* 2003). The growing apprehension about the potential effects of sediment toxicity poses to marine fauna and flora and the risk posed to the environment by the contaminants accumulated by the sediments have aroused an increase in research interests in marine sediments (Vernet 1991; Calace *et al.* 2005).

Sediment is a matrix of materials, made up of detritus, inorganic and organic particles, and is relatively heterogeneous in terms of its physical, chemical and biological characteristics (Sarkar *et al.* 2004) and consist of a wide range of particle sizes, including gravels, sand, silt and clay (Fergusson 1990). Sediments are well known to act as a major sink for many of the more persistent organic and inorganic chemicals introduced into the aquatic environment by atmospheric deposition, erosion of the geological matrix, or from anthropogenic sources (such as industrial effluents, mining wastes, etc.) (Pempkowiak *et al.* 1999; Sarkar *et al.* 2004; Calace *et al.* 2005). Sediments also act as potential sources of pollution for the surrounding water and benthic flora and fauna by releasing sorbed contaminants back to the overlying water column should remobilization occur through any disturbance (Denton *et al.* 1997; Sarkar *et al.* 2004; Adamo *et al.* 2005). Contaminants of major concern found in the marine coastal environment include heavy metals, persistent organic pollutants (POPs), oils (hydrocarbons), nutrients and radioactive substances (UNEP 1999).

Contamination of the environment, and in particular the marine environment by heavy metals has received considerable attention as evidenced by an increase of publications appearing in the literature (Chandra 2002). Environmental contamination from heavy metals is of major concern because they show behavior similar to those of persistent toxic chemicals. Metals are partitioned amongst soluble phases, suspended and bottom sediments and biota in aquatic systems (Gangaiya *et al.* 2001).

Sediments play a useful role in the assessment of heavy metal contamination (Gangaiya *et al.* 2001). The partitioning behavior of heavy metals is such that they tend to accumulate in sediments to levels that are several orders of magnitude higher than in the surrounding water (Denton *et al.* 1997). Further, their deposition rates are generally related to their rates of input in the surrounding water (Forstner 1990). The analysis of heavy metals in the sediments permits detection of contaminants that may be either absent or in low concentrations in the water column, and their distribution in coastal sediments provides a record of the spatial and temporal history of pollution in a particular area or ecosystem (Binning & Baird 2001). Therefore, the chemical analysis of sediments, is very important from the environmental pollution point of view because sediment concentrates metals from aquatic systems, and represents an appropriate medium in monitoring of environmental pollution (Denton *et al.* 1997; Sarkar *et al.* 2004).

The main heavy metals forming the focus of this study are Arsenic (As), Cadmium (Cd), Chromium (Cr), Copper (Cu), Lead (Pb), Mercury (Hg), Nickel (Ni), and Zinc (Zn). Hereafter these heavy metals will be referred to as the contaminants of concern. The reason for focusing on these particular metals is that they are known to be present in landfill leachate and pose significant threat to human health and the environment (Leong *et al.* n.d.; Pattle Delamore Partners Ltd 1999; Sedgwisk 2005).

1.2 Environmental Legislation

In New Zealand, there is an increasing amount of government policy and legislation tailored to deal with managing the risks associated with contaminants, including metals, in the environment. The development of this policy has been a consequence of past agricultural and industrial practices that have resulted in the contamination of the environment, the recognition of the need to manage the risks to human health and the environment arising from this contamination, and to prevent new contamination (Cavanagh & Coakley 2005).

In an effort to improve the regulation and management of metals in the environment, policy makers are developing policy tools (e.g., legislation and guideline) to help manage the risks these contaminants present. The main legislation governing the regulation of metal contaminants and their effects are the Resource Management Act (RMA) 1991 and the Hazardous Substance and New Organisms Act (HSNO) 1996 (Cavanagh & Coakley 2005).

The Resource Management Act (RMA) 1991 is the primary piece of legislation for the sustainable management of natural and physical resources, including controlling the adverse effects of contaminants in the environment. Under the RMA, metals and metal compounds generally meet the definition of a contaminant (New Zealand Government 1991; Cavanagh & Coakley 2005). As such, district and regional councils must issue resource consents, according to the provisions set out in the relevant council plans to control the use of metal contaminated land and to prevent or mitigate adverse effects associated with the use of the land including the discharge of metals into the environment.

There are other pieces of legislation that control the levels of metals in the environment. For instance, the Hazardous Substances and New Organism (HSNO) Act 1996 and the Food Act 1981 have provisions to prevent the adverse effects of metals on human health and the environment (Cavanagh & Coakley 2005). The

purpose of the Hazardous Substance and New Organism (HSNO) Act 1996 in relation to metal contamination is:

"to protect the environment, and the health and safety of people and communities, by preventing or managing the adverse effects of hazardous substances".

This purpose is achieved by a central process of approving new and existing hazardous substances administered by the Environment Risk Management Authority (ERMA) New Zealand, regulations that specify performance-based controls, and a compliance and enforcement system (Cavanagh & Coakley 2005).

The Food Act 1981 governs the sale of food in New Zealand, and includes protection of public health. The Act sets out the standards and it is an offence not to comply with the applicable food standards. One such standard is the Australia New Zealand Food Standard Code, which outlines maximum levels (MLs) for specified food items. MLs have been set for metals including arsenic, cadmium, lead and mercury (Cavanagh & Coakley 2005).

In Kiribati, the Kiribati Environment Act 1999 is the principal piece of legislation that promotes the sustainable management of natural and physical resources through developing national standards to promote sustainable development and monitors those standards through environmental auditing. This includes controlling developments through the Environmental Impact Assessment (EIA) process and controlling potentially polluting activities through the pollution control licensing system (Government of Kiribati 2000).

The Kiribati Environment Regulations 2001, which was enacted to support the Kiribati Environment Act 1999, has a list of prescribed activities. Under the Regulations, these activities, which include landfilling operation, require a pollution control licence from the Ministry of Environment, Lands and Agricultural Development (MELAD) to control the discharge of contaminants to land, water and air and to prevent or mitigate adverse effects of the discharge into the environment

(Government of Kiribati 2001). Specifically, the Kiribati Environment Regulations 2001 has a guideline on maximum acceptable limits of certain soil pollutants that includes all the contaminants of concern in this study. The guideline sets an action level for each contaminant, a level at which the consent authority will request further investigations to determine the likely environmental and health effects of the contaminants (Government of Kiribati 2001).

1.3 General information on Kiribati

The Republic of Kiribati is made up of 33 coral islands and atolls located in the middle of the Pacific. The islands straddles the equator between 4º 43' N and 11º 25' S and lies west of the International Dateline between 169º 32' and 150º 14' W (Gangaiya 1994; Kaly 1996). It consists of three main island groups – the Gilbert Group in the west, the Phoenix Group in the centre, and the Northern and Southern Line Islands to the east. The islands are extremely small with a total land area of 822.8 km² and an Exclusive Economic Zone (EEZ) area of 3.5 million km² between 5º N and 11ºS and 170ºE and 150ºW, giving a land area to ocean area ratio of 1:4000 (Gangaiya 1994; SPREPa 1994). The islands are isolated and fragmented, spreading over some 3,870 km from Banaba (Ocean Island) on the west to Kiritimati (Christmas Island) in the east; and 2,050 km from Teraina (Washington Island) in the Northern Line islands to Flint in the Southern Line Islands (Figure 1) (SPREPa 1994).

Kiribati is located in the dry belt of the equatorial oceanic climate zone, with mean daily temperatures ranging from 26ºC to 32ºC, with the recorded highs and lows being 22ºC and 37ºC respectively. Annual rainfall is extremely variable, both annually and between islands, with the annual average in the Gilbert Group ranging from about 1,000mm for the drier islands near the equator to 3,000mm for the wetter islands in the far north (Gangaiya 1994; SPREPa 1994).

The atoll soils of Kiribati are reported as being possibly the most infertile in the world and are young (3,000 to 4,000 years old). Their properties are consistent with those expected for young atoll soils – shallow, alkaline (high pH 8.2 to 8.9), coarse

textured and mineralogy dominated by the carbonates. This is essentially very similar to the coral-limestone parent material from which they are derived (Gangaiya 1994; SPREPa 1994).

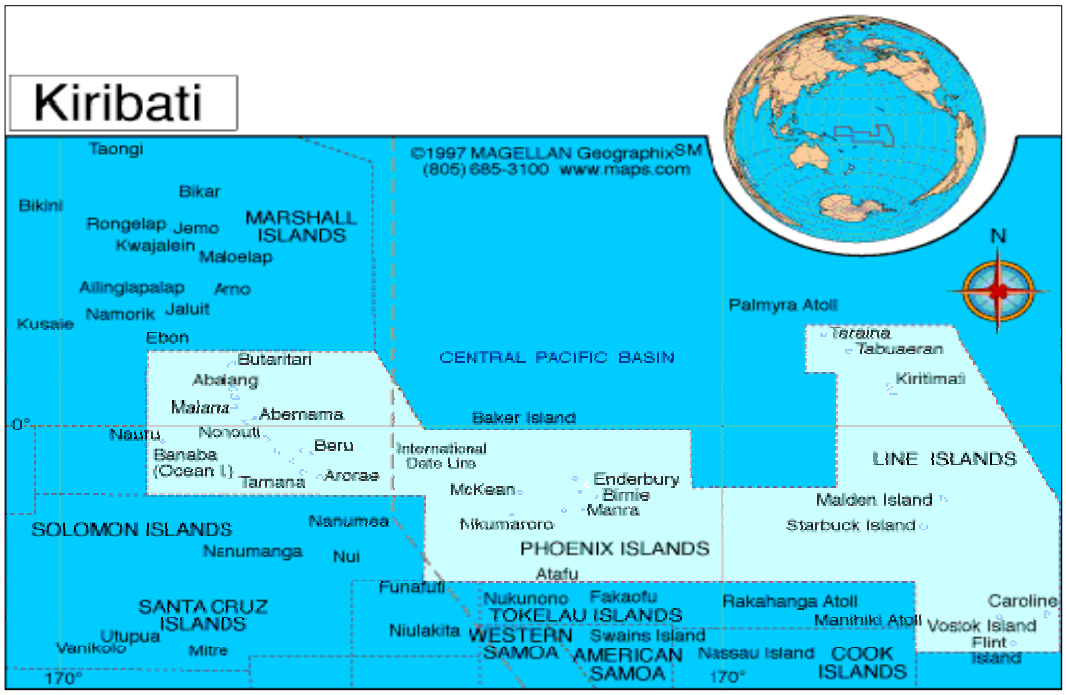


Figure 1.1: Map of Kiribati

The administrative centre of Kiribati is located in Tarawa, an island in the middle of the Gilbert Group. The total population for Kiribati is 84,494 according to the 2000 census with a population growth rate of 1.69. From this total population, around 43% of the national population resides in the main urban centres on South Tarawa (Tiroa 2001).

Kiribati solid waste generation was estimated to be approximately 30,840 tonnes/yr based on the 2000 population (UNEP 2000), with South Tarawa generating 13,402 tonnes/yr. Study reports on waste characterization of domestic waste since1994 found that the bulk of the waste stream is organic waste. The results of these surveys are summarized in Table 1.1 (Kiribati SAPHE Project n.d.; Teiwaki 2004).

Table 1.1: Summary of domestic waste composition surveys (Unit: Wt %)

Classification	Gangaiya (1994)	PPTA report (ADB/Royds, 1996	ISEU report (ADB, 1997	SWCSMP report (EU/Sinclair, 2000)	Teiwaki 2004
All Organics	80	62	76	51.3	48
Papers	2	14	5	7	3
Plastics	2	12	5	7.2	8
Glass/Ceramics	3	-	3	13.6	15
All metals	7	7	10	9.4	19
Textile/Rubber	<1	3	<1	3	-
Miscellaneous	6	2	-	8.5	7
Total	100	100	100	100	100

Notes: 1. Gangaiya (1994), SWCSMP/Sinclair (2000) and Teiwaki (2004) were based on household surveys.

2. PPTA (1996) was based on a waste dumpsite survey at the Betio landfill.

3. The general trend indicates an increased in use of plastics, paper and metal and decrease in organic waste.

1.4 Objectives of the Study

The overall aim of this study was to carry out a comparative investigation on heavy metal contamination in coastal marine sediments from two separate landfills: one in Tarawa, Kiribati and the other in Auckland, New Zealand. More specifically, the objectives were to:

1) Acquire baseline heavy metal data necessary to provide an evaluation of the ecological health of the surrounding aquatic environment and for future Environment Impact Assessment (EIA) for landfills,

- Metals considered are As, Cd, Cr, Cu, Pb, Hg, Ni and Zn

2) Assess the extent of contamination by investigating the distribution of heavy metals in marine surficial sediments,
3) Contribute to understanding of the environmental performance of landfills on coastal margins.

1.5 Structure of Thesis

Chapter 2 is a literature review that introduces the main contaminants of concern and examines the key parameters that can affect the mobility and fate of metals in marine sediments. Chapter 3 covers the description of the two study sites used as case studies in this study. Chapter 4 presents the field methodology and the experimental procedure for the physical and chemical analysis of the samples. Chapter 5 presents the results and a discussion of the entire study and an overview of the effectiveness of the work undertaken. The findings of the study are summarized in Chapter 6 with the main conclusions drawn and future directions for continued research are recommended.

Chapter Two: Heavy Metal Contamination in Marine Sediments

2.1 Introduction

Environmental pollution with toxic metals is becoming a global phenomenon. As a result of the increasing concern with the potential effects of the metallic contaminants on human health and the environment, the research on fundamental, applied and health aspects of trace metals in the environment is increasing (Vernet 1991).

Advances in knowledge of the concentrations and distributions of trace metals in the marine environment have occurred since the mid 1970s (Burton & Statham 2000). This is mainly due to developments in procedures for contamination-free sampling, the adoption of clean methodologies for handling and analysis of samples, and increased application of improved analytical methods such as inductively coupled plasma-mass spectrometry (HG-ICP-MS) (Burton & Statham 2000; Plant *et al.* 2003).

Heavy metals occur naturally as they are components of the lithosphere and are released into the environment through volcanism and weathering of rocks (Fergusson 1990). However, large-scale release of heavy metals to the aquatic environment is often a result of human intervention (Mance 1987; Denton *et al.* 1997). Coastal regions are some of the most sensitive environments and yet they are subject to growing human pressures (David 2003) because of increasing urbanization, industrial development, and recreational activities. Therefore, pollution levels are often elevated in the coast because of nearby land based pollution sources (Fergusson 1990; Angelidis 1995).

Industrial processes that release a variety of metals into waterways include mining, smelting and refining. Almost all industrial processes that produce waste discharges are potential sources of heavy metals to the aquatic environment (Denton *et al.* 2001). Domestic wastewater, sewage sludge, urban runoff, and leachate from solid waste disposal sites are also obvious sources of heavy metals into rivers, estuaries and coastal waters (Mance 1987). A proportion of the total anthropogenic metal input in the sediments in nearshore waters, adjacent to urban and industrial growth centers, comes from the combustion of fossil fuels. Other potential sources include ports, harbours, marinas and mooring sites, also subjected to heavy metal inputs associated with recreational, commercial, and occasionally, military, boating, and shipping activities (Denton *et al.* 1997).

The study of contamination of the coastal marine environment from anthropogenic sources is receiving considerable attention, in particular for small island countries that are surrounded by vast stretches of ocean. In small islands, coastal environments play a key role in the life of the local population serving as fishing grounds and recreational areas (Chandra 2002). A large and increasing proportion of the world's population is concentrated in coastal areas. Coastal environment contamination in small island countries is of concern because livelihoods in these countries are intimately linked to the resources of the coastal zone (Gangaiya *et al.* 2001). In order to maintain a productive marine environment, which in turn can allow humans to derive maximum benefits from it, it is essential that sustainable systems are adopted in coastal areas. Human activities on land, and immediately adjacent to coasts, pose a significant threat to the health, productivity and biodiversity of the marine environment (Chandra 2002).

In the Pacific Region, domestic sewage-discharges, solid waste from domestic, industrial, and construction activities, fertilizer use, sediments, and increasing toxic wastes from industrial, agricultural, and domestic sources are identified as significant sources of land-based pollutants (UNEP 2000). Domestic wastes dominate the waste stream from land-based sources. However, given the relatively small quantities of hazardous and toxic materials (in particular persistent organic pollutants (POPs) and

heavy metals) they are posing a real concern based on the known and potential effects on the marine environment (UNEP 2000).

A land-based pollution sources case study done in Kiribati in 1994 found that solid wastes, domestic wastewater, industrially related wastes, possibly agricultural wastes such as pesticides and household hazardous substances, contribute to the pollution of marine environments, particularly Tarawa, the capital island (Gangaiya 1994). SPREP (1994b) also identified poor waste management and pollution of coastal waters amongst the major environmental problems found in Kiribati particularly on South Tarawa. One of the important Kiribati environmental documents, the Kiribati National Environmental Management Strategy (NEMS) recommended five key strategies for attaining sustainable development in Kiribati. Improving waste management and pollution control was one of these recommendations. Some of the strategies identified to help to achieve this objective were improving management and disposal of solid waste and sewage, and controlling of marine pollution (SPREP 1994a).

A number of studies in the Pacific Region found that poor waste management was a major contributor of oils, pesticides and heavy metals to soil and coastal marine environments, particularly the lagoons (Kaly 1996; Morrison & Munro 1999). However, UNEP (2002) identified the environmental assessments of the discharges from existing dumpsites as one of the most significant areas where new or additional data is required to provide a comprehensive coverage of chemical contaminant sources in the Region.

2.2 Pacific Islands Region

The Pacific Region was defined as the area between latitudes 23º N and 23º S of the Equator and between longitudes 130º W and 120º E of the International Dateline (Figure 2.1). The Pacific Islands Region is incredibly diverse, and encompasses a wide variety of geographical features – high islands and low-lying atolls, populations,

'cultural areas' of Melanesia, Polynesia, and Micronesia, economies and politics within its 22 island countries (UNEP 2000; UNEP 2002). The Pacific Region is vast spreading over an area of 30 million square kilometers, more than 98 per cent of which consists of ocean.

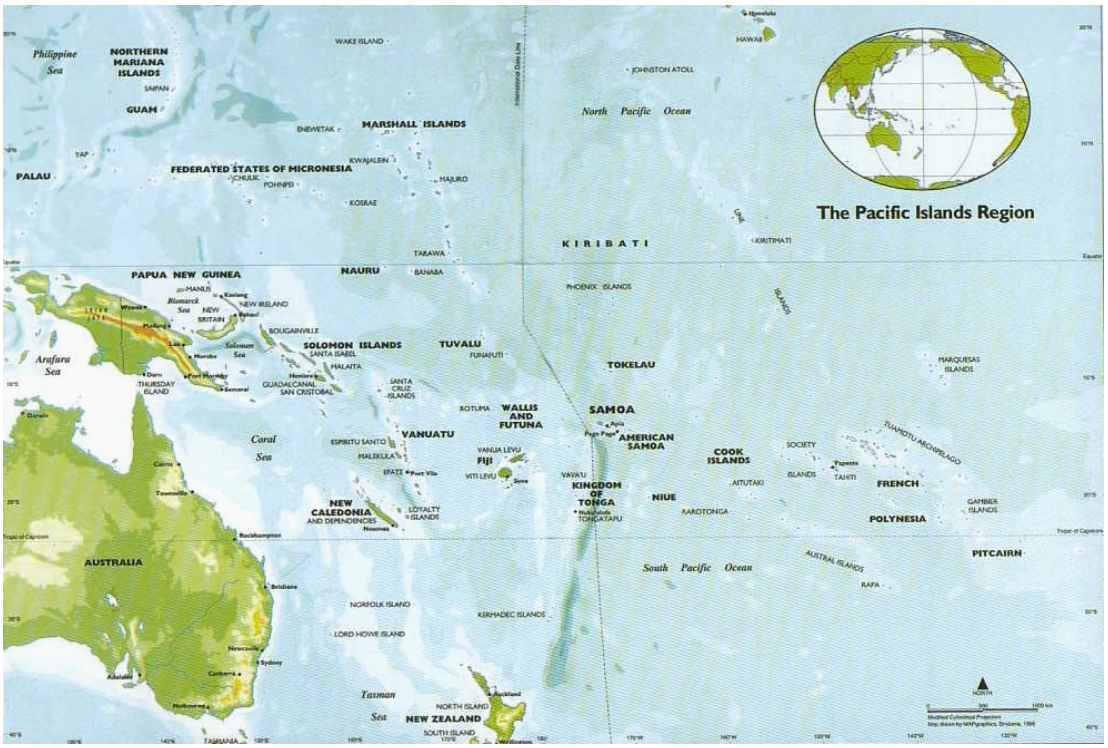


Figure 2.1: A map of the Pacific Islands Region (Source: SPREP (2000)).

The Pacific area is three times larger than either the USA or China. As the land area is only about 550,000 square kilometers, the Pacific Region consists mainly of a large number of small countries or territories separated by substantial stretches of ocean (Morrison & Munro 1999). Of its 7500 islands, only about 500 are inhabited (UNEP 2002). The total population of the Pacific Region was estimated at 2.8 million in 2000, and the current regional growth rate is 2.2 per cent with more than half of the Pacific Region's population being teenagers (UNEP 2002).

This 'isolation' of islands/countries has both advantages and disadvantages. This isolation complicates transport and communications but the lack of major industrial centres and the associated infrastructure of ports, shipping traffic, factory complexes

and waste disposal sites is seen by those concerned with the environment as a significant advantage (Morrison & Munro 1999).

The marine environment is the foundation of society and economies in the Pacific. The importance of the aquatic environment to the people of the Pacific cannot be overstated. It is the coastal area that most people make their living and work. It is also in the coastal region and the marine environment where most planning for expansion of the economic base is concentrated (UNEP 2000). The marine environment is a primary source of food and revenue for most nations in the region, and it is essential that this is not be adversely affected by the land-based pollutants in particular the chemical contaminants (UNEP 2002).

*2.*3 Waste Management in the Pacific

Waste management in the South Pacific has become a serious environmental problem. It is a much bigger issue than many people appreciate, both within the region and outside (Morrison & Munro 1999). Even though the management of waste is an issue of global concern, the problem is particularly serious for land- and resource-poor and ecologically fragile Pacific island nations. "As populations increase and exceed the capacity of small island resources to sustain them, as the islands become more dependent on the global cash and consumer economy, and as island societies loose their close traditional links with their island environment, waste management and sustainable use of limited island resources become much more serious health, cultural, economic and environmental problems" (Thaman *et al.* 2003).

The South Pacific Regional Environment Programme (SPREP) now known as the Secretariat of the Pacific Regional Environment Programme, established in 1982, was tasked by the governments of the Pacific island nations with coordinating environment activities in the region to address the issue (Morrison & Munro 1999). SPREP has, for almost a decade spearheaded an active South Pacific Regional Pollution Prevention, Waste Minimization and Management Programme which has

been used as the umbrella and justification for research and enabling activities related to waste management undertakings in the region (Thaman *et al.* 2003). In addition, waste management issues at the regional level are addressed in the SPREP Convention, the Waigani and Basel Conventions, which they have provisions requiring good management of wastes for protection of the natural resources of the Region. However, a number of countries are still to ratify these conventions (Morrison & Munro 1999).

Solid waste generation in the South Pacific Region is increasing at a rapid rate. As urbanization increases and the local economies are transformed from traditional subsistence economies to cash economies there is an increased use of non-biodegradable materials and products (UNEP 2000). Waste management in the Pacific Islands reflects the generation and disposal of waste in the region which is quite low, (0.2-1.0 kg/person/day) compared to developed nations such as Australia (2-2.5kg/person/day) (Thaman *et al.* 2003). Though, some of the more developed Pacific Islands especially Fiji and Samoa show the generation of waste generation is higher and increasing (Table: 2.1) .

A comparison of waste generation data generated from studies undertaken in the 1990s indicates that average solid waste generation rates in Pacific island nations have increased from about 0.42kg/capita/day to about 0.66kg/capita/day. This represents a 57 per cent increase in waste generation over 9 years and this is attributed to an increase in paper products (mainly disposable nappies) and plastics (Thaman *et al.* 2003).

In terms of Pacific Islands waste composition, there is a degree of uniformity in the composition of urban domestic solid waste with organic/biodegradable material tending to dominate (Table 2.2) (Morrison & Munro 1999). Other major components are plastics (bottles, bags and other packaging), glass, metals, paper and cardboard. Minor components, but less difficult to manage; include batteries (dry and wet), white metal goods, and occasionally chemicals and containers, including pesticides containers.

Table 2.1: Waste disposal figures for selected Pacific Island Countries

Country	Population	Average Waste Generation rate (kg/capital/day)	Biodegradable content (%)
Cook Is (Rarotonga)	19,600	0.19	35
FSM (Pohnpei)	134,000	0.38	19
Fiji (Lautoka)	825,000	0.92	68
Kiribati (South Tarawa)	76,000	0.33	20
Marshall Islands (Majuro)	51,000	0.38	46
Papua New Guinea	4,400,000	0.41	62
Samoa (Apia)	170,000	1.05	61
Solomon Islands (Honiara)	385,000	0.22	65
Tonga (Nukualofa)	97,000	0.82	47
Tuvalu (Funafuti)	10,000	0.43	52
Vanuatu((Port Villa)	175,000	0.65	71
Average		**0.55**	**49**

Notes: Collated from studies carried out between 1996-1999 by various organizations at bracketed cities (adopted from Thaman *et al*. (2003))

A number of waste disposal methods can be found in the Pacific Islands; apart from reuse and recycling, these include landfill, incineration, composting, open burning, burying in small packages (Morrison & Munro 1999). Throughout the Pacific Region, there is an indiscriminate dumping of the increasing stream of solid domestic waste in mangroves, creeks, lagoon, and estuary locations. These areas are ecologically important as the coastal habitats support a high natural productivity, yet have also historically been the preferred choice for dump sites (Kaly 1996; Thaman *et al.* 2003).

Landfills have been identified as a major contributor to the contamination of nearshore lagoon waters and edible shellfish (Kaly 1996).

Table 2.2: Typical South Pacific domestic solid waste data

Component	Percentage by Weight (range)
Garden waste	14-44
Food waste	2-45
Plastic	4-16
Paper	2-13
Glass	2-17
Metals	8-17
Bone	<1
Rubber/leather	0-2
Bulk Density	**100-350 kg/m^3**
Generation rate	**0.19-0.52 kg/capita/day**

(adopted from Morrison & Munro (1999))

Landfill operations in the Pacific Islands are generally unsatisfactory. Improper waste disposal has been identified as contributing to increase the magnitude of the negative impacts of waste generation and disposal (Thaman *et al.* 2003). Landfill sites can vary from uncontrolled tipping/dumping through controlled dumping to basic sanitary landfill and full sanitary landfills (Morrison & Munro 1999). Leachate from landfills poses a serious concern in the region. UNEP (2000) suggested that even though leachate production has not been quantified, it contributions to the contamination of the marine environment are likely to be significant.

Based on previous studies and observations over more than 35 years in the Pacific Islands, some of the more important adverse impacts of inappropriate waste management and disposal include (Thaman *et al.* 2003);

- Deterioration of human health and safety
- Loss of aesthetic values
- Pollution of rivers, lakes, groundwater and potable waster resources
- Spread of invasive plants, pests and diseases
- Eutrophication of slow-moving rivers, bays and lagoons
- High coral mortality
- Loss of tourism value
- Habitat destruction and loss
- Air pollution
- Increased mortality of fish and other organisms

Given the incredible diversity of the Pacific Islands Region, the nature and volume of waste streams and waste management problems and solutions vary considerably both between countries and within countries (Thaman *et al.* 2003). Most of the Pacific island nations face critical problems with regard to waste management. Some of the common problems include: insufficient government priority and political support; lack of finance; no long term planning or business planning; poor landfill siting, design, planning and management; lack of skilled personnel; lack of awareness of the problems caused by poor waste management and physical limitations to the establishment of landfill sites (Morrison & Munro 1999). Other problems include poor handling of clinical waste; insufficient recycling and reuse, including limited reuse of organic wastes, septic sludge, sewage sludge and effluent (Morrison & Munro 1999).

Recent studies stressed that waste management problems are particularly serious on very small islands, in particular low-lying atolls like Kiribati where there is little available land area for disposal sites, limited soils for capping and site reclamation purposes, and where waste disposal heavily impacts on limited potable water and the

coastal and nearshore marine environment (Thaman *et al.* 2003). In atoll islands, due to the limited land, landfills tend to be built on the coastal margins, particularly the lagoons due to its relatively low engineering cost. Previous studies also have identified the serious health and environmental issues caused by water pollution on limited freshwater resources and nearshore and coastal marine environments affecting Pacific communities. The predicted rising sea-levels associated with global warming and the inevitable loss of land and freshwater resources will further intensifying these problems (Thaman *et al.* 2003).

Studies found that the generation and disposal of wastes has direct and indirect connections to economic development and therefore it needs serious attention. If not appropriately dealt with, it can lead to serious negative impacts on national development undertakings, including tourism and trade, food supplies, public health and the environment (Morrison & Munro 1999; Thaman *et al.* 2003).

2.4 Heavy metal contamination in the Pacific Region and Kiribati

People from outside often view the Pacific as an unspoiled tropical paradise (Chandra 2002). The Pacific has an attractive image internationally, a picture of white sandy beaches, clear lagoon waters, thriving rich coral reef communities and lush green vegetation is widely used to attract travelers from all over the world (Morrison *et al.* 1996 ; Thaman *et al.* 2003). The possibility that contamination of the environment exists is often not considered. While clear lagoon waters and white beaches do exist in places, the situation is changing with rapidly increasing populations and industrial development contributing to problems that now need serious attention (Morrison *et al.* 1996).

Pacific island nations are going through a phase of rapid urbanization, especially around the coastal regions where urban developments are mostly centred, and in particular the development of ports as points of entry of goods and services (Chandra 2002). The increasing tendency of rural population migrating to urban centres causes

population imbalance shifting towards the urban areas. Ultimately, the shift in population densities poses considerable pressure on the existing services including solid waste disposal.

Heavy metal compounds are found in many of the materials and processes of regional industrial activities and to a lesser extent agricultural activities (UNEP 2000). Lead, Cadmium, and Chromium, which were used for anti-algae and fungi-fouling paints for marine craft and structures. More toxic, and of documented environmental degradation in the Pacific, is the use of organotins, such as TBT in anti-fouling paints. Sandblasting of ships in preparation for painting has also contributed to the release of heavy metals into the marine environment. Mining activities are contributors to the region's environmental load of heavy metals. Other industrial operations such as foundries, metal plating and processes involving combustion and use of petroleum products are also contributors (UNEP 2000). In the Pacific island nations, the release of heavy metals to the marine environment is increasing as industrial development continues because either there are no regulatory mechanisms, or monitoring and enforcement of environmental regulations is difficult due to lack of laboratory facilities and enforcement mechanisms.

Information on heavy metals in the South Pacific environment is limited, except for Suva, Fiji which has attracted a number of environmental pollution studies recently, including heavy metal contamination investigations, for example in the studies of Naidu and Morrison (1994), Gangaiya *et al*. (2001) and Chandra (2002). Generally, heavy metal pollution in the Pacific Region varies with site. Levels found in one country usually indicate the degree of development and industrialization of that particular site (Chandra 2002). Urbanization plus the changing lifestyles of Pacific Islanders means moving away from traditionally subsistence lifestyles to cash-based, consumer goods societies, where the volume and complexity of waste products have increased considerably (UNEP 2000). The composition of waste has changed with the biodegradable proportion of the waste decreasing, accompanied by an increase in non-biodegradable waste such as nicad and lithium batteries, waste oil, and food and

drink cans (Chandra 2002). These contribute to heavy metal problems in the marine environment.

The data available on metals for the Tarawa lagoon, Kiribati, is limited to metals (Hg, Cu, Cr, Cd, and Pb) in shellfish reported in Naidu *et al*. (1991), based on sampling in 1987 (Table 2.3). As expected, the results were reasonably low. However, the level of cadmium was higher than usually found in shellfish in unpolluted waters and there was no obvious explanation for the elevated levels of cadmium found at the time (Naidu *et al.* 1991).

Table 2.3: Trace metal concentrations in *Anadara maculosa* ("te bun") from Tarawa Lagoon, Kiribati

Location	Hg (µg/kg)	Cu (µg/g)	Cr (µg/g)	Cd (µg/g)	Pb (µg/g)
Teaoraereke	5.6	2.1	1.1	0.7	0.2
Bikenibeu	5.5	18.0	2.7	1.2	0.2
Eita	<0.1	1.0	0.4	1.0	0.5

(adopted from Naidu, *et al.* (1991))

2.5 Landfill leachate generation

One of the most significant impacts of landfills on the surrounding environment comes from the generation of leachate (North *et al.* 2004) and therefore, environmental monitoring programmes for landfills tend to focus on the investigation of leachate impacts, particularly any contamination of ground and surface water. Leachate is the contaminated liquid resulting in the waste decomposition and percolation of rain water, through wastes stored in the landfill or dump site that have entered the landfill through external sources (Saleem 1999; Taulis 2005). Surface

water flow and ground water may also contribute (Johnsson 1995). As water passes through the landfill, it may 'leach' pollutants from the disposed waste to areas deeper in the soil (Saleem 1999).

Solid waste decomposition begins immediately after burial and involves a complex series of physical, chemical, and biological processes, occurring simultaneously (Scrudato & Pagano 1991; Johnsson 1995; Taulis 2005). The physical processes involve the compression, dissolution, absorption, and adsorption, while the chemical processes include the oxidation of metals and the reactions involving organic acid and dissolved carbon dioxide. The biological processes, which involve aerobic and anaerobic organisms, are the main processes directly responsible for the generation of landfill gas (mainly methane and carbon dioxide) and leachate (Taulis 2005).

Leachate may contain elevated levels of metals that pose a risk to human health and the environment and therefore it is important to understand the processes involved in leachate formation and composition (Taulis 2005). Biological decomposition of organic matter depends on microorganism activity which, in turn, is dependent on the age, depth, compaction, density, and compaction of wastes, and the concentration of toxic constituents, oxygen levels, moisture, temperature and pH (Johnsson 1995). Moisture has been determined to be the most significant factor to refuse decomposition (Scrudato & Pagano 1991) and water is consumed by in the microbiological processes inside the landfill (Johnsson 1995).

The main biological processes operating in the landfill can be summarized into four main phases;

Phase 1: Immediately after initial placement of refuse, aerobic organisms rapidly break down degradable refuse in the presence of oxygen resulting in the production of carbon dioxide, organic compounds, heat, and water. The pH in this phase is generally neutral (Scrudato & Pagano 1991; Johnsson 1995). Usually, organisms thrive for only a few days under these conditions because oxygen is no longer available

once the refuse is covered. Levels for Biochemical Oxygen Demand (BOD) and Chemical Oxygen Demand (COD) also become elevated (Taulis 2005).

Phase 2: Aerobic organisms die off, and hydrolytic, fermentative, and acetogenic bacteria, which can thrive in the absence of oxygen, take over the decomposition process (Taulis 2005). Carbon dioxide (50-90% of the gas generated) and volatile fatty acids are produced causing a decrease in pH (Scrudato & Pagano 1991). Leachate becomes chemically aggressive resulting in metal species becoming mobilized and may form metal complexes (Johnsson 1995; Taulis 2005).

Phase 3: In this initial methanogenic phase, the acids produced in Phase 2 are utilized by the increasing number of methanogenic bacteria and converted to methane and carbon dioxide. The leachate pH increases and the organic content of leachate decreases (Scrudato & Pagano 1991; Johnsson 1995). The BOD and COD concentrations decrease as volatile fatty acids are consumed (Taulis 2005).

Phase 4: In this stable phase of the methanogenic phase, methane production reaches its maximum but then decreases after the carboxylic acids are consumed (Taulis 2005). As the methanogenic phase removes soluble organic compounds, the concentrations of metals, inorganics, fatty acids, COD and BOD all drop while the pH continue to increase (Johnsson 1995). Leachate becomes stable.

In the methanogenic phase, metal concentrations tend to be low because the high pH enhances both sorption and precipitation of metals (Taulis 2005). Therefore, metals become more readily adsorbed onto particulates and thereby concentrate in the sludge and refuse residue resulting in the immobilization of metals (Scrudato & Pagano 1991).

However, metals are not always attenuated in landfills, as there are processes that can potentially mobilize them to solutions. Organic matter plays an important role in these processes (Taulis 2005). The formation of metal complexes to organic ligands and sorption to colloids by humic material can effectively mobilize metals by increasing their concentration in the leachate (Taulis 2005).

In municipal waste landfills, all the above four phases take place at one time or another and each phase of the stabilization process has a characteristic leachate composition (Johnsson 1995). Monitoring data from the Horotiu (Hamilton) and Rosedale Rd. (Auckland) landfills illustrate the differences between young and old leachates (Table 2.4) (Taulis 2005). Also important to consider is that the amount of leachate generated at a particular site is strongly dependent upon whether the landfill being investigated is in operation or closed (Taulis 2005).

In most situations, landfills are gradually filled and therefore these phases tend to overlap over time. Therefore, it is fair to say that in reality no landfill has a single 'age' but rather a family of different ages, associated with different cells within the landfill complex progressing towards ultimate stabilization (Johnsson 1995).

2.6 Metal contaminants, uses and sources

Heavy metals, that will be referred to in this study as contaminants of concern that will form the focus of this study are As, Cd, Cr, Cu, Hg, Ni, Pb, and Zn. Table 2.5 summarizes the diversity of uses in industry of the metals of interest in this study. These metals are known to be present in landfill leachates and their presence poses a serious human health and environmental concern (e.g. mercury and cadmium). Such focus is because of their known toxicity (with regard to human health and ecology) or their ability to bioaccumulate and move through the food chain. Bioaccumulation processes may adversely affect food resources by making them unfit for human consumption (Denton *et al.* 1997). Table 2.6 summarizes the present day sources of metals in New Zealand to the coastal environment.

Table 2.4: Metal contaminants in leachate from Horotiu (1999) and Rosedale Rd. (1998) landfills

	Horotiu Landfill		Rosedale Rd. Landfill	
Metal contaminant (mg/l)	**Cell closed for more than 10 years**	**Currently operating cell**	**Composite from the whole site**	**Currently operating cell**
Aluminium	0.046	4.1	*	*
Cadmium	<0.001	<0.001	<0.005	<0.005
Calcium	85	380	69	86
Chromium	0.0012	0.12	0.015	0.218
Iron	3.7	32	8.2	7.57
Lead	0.001	0.014	<0.02	<0.02
magnesium	20	200	36.5	82
Manganese	0.67	4	*	*
Nickel	0.012	0.14	0.034	0.093
Potassium	80	740	131.2	540
Sodium	57	970	222	812
Zinc	0.015	0.8	0.145	0.158

- Not reported (adopted from Taulis (2005))

Table 2.5: Past and Present Industrial Uses of the Heavy Metals Examined

Metal	Uses of Metals and Compounds
Arsenic	Component of pesticides; wood preservative; alloys; semi-conductors; medicines; glass and enamels
Cadmium	Electroplating (anticorrosion coatings); thermoplastic stabilizers, e.g. in PVC; Ni-Cd batteries; alloys; solders; catalysts; engraving; semi-conductors; TV tube phosphorus; pigments in paints and plastics; glass ceramics; biocides
Chromium	Metallurgy-ferrochromium alloys, refractory bricks; electroplating; industrial dyes; ink; tanning; paint; wood preservative; glass making; cement production
Copper	Electrical industry; alloys, e.g. brass; chemical catalyst; anti-fouling paint; algaecide; wood preservative
Lead	Storage batteries; leaded gasoline; pigments; read lead paint; ammunition; solder; cable covering; anti-fouling paint; glazing; PVC stabilizers
Mercury	Chlorine production; electrical apparatus; anti-mildew paint; instruments; catalysts e.g. for PVC and acetaldehyde production; pesticides; preservatives; pharmaceuticals; dentistry; anti-fouling paint
Nickel	Metallurgy – steel and other alloys; electroplating; catalyst; rechargeable Ni-Cd batteries
Zinc	Zinc based alloys; brass and bronze; galvanizing; rolled zinc; paints; batteries; rubber; sacrificial anodes on marine water craft

(adopted from Denton *et al,*. (1997))

The term "heavy metal" is used to describe metals having density greater than 5-6 g cm^{-3} (Sedgwick 2005). Heavy metals are often referred to as trace metals, occurring naturally in low concentrations in organisms, although the term trace metal might imply the presence of an essential requirement by organisms for that particular metal (Furness & Rainbow 2000). Heavy metals can be divided into two groups namely; i) metals essential to at least some organisms (micronutrients) such as As, Cr, Cu, Ni, and Zn and ii) nonessential heavy metals with no known biological function which include Hg, Pb and Cd (Furness & Rainbow 2000; Denton *et al.* 2001). Even though micronutrients are essential to some forms of life, they become toxic at high concentrations (Sedgwick 2005).

Table 2.6: Major sources of heavy metals to the coastal environment in New Zealand

Source	Metals	Comment
Urban storm-water	Cu, Pb, Zn	Build-up near urban areas
Contaminated sediment from past industrial discharges	As, Cr, Cu, Pb, Zn	Gradual burial and dispersal
Industrial spills, poor yard practices	Numerous	On-going problem
Anti-fouling paints and boating activities	Cu, Zn, Hg	Local contamination (e.g. marina)
Geothermal discharges	Hg	Widespread Hg in fisheries food chain. Local geothermal discharges

(adopted from Williamson *et al.* (2003))

2.6.1 Arsenic (As)

Arsenic has become increasingly important in geochemistry due to its significance to human health. The concentration varies considerably, partly in relation to geology and partly due to the activity of human (Plant *et al.* 2003). The most important source of arsenic to the aquatic environment is domestic wastewater, reflecting the common use of arsenic in household preparations and in small industrial discharges to municipal waste systems (Denton *et al.* 2001). Other contributing sources to the environment include mining and smelting, combustion of arsenical coals, petroleum recovery, refining, chemical production and use, use of biocides including wood preservatives, use of fertilizers, the manufacture and use of animal feed activities, and the development of high-arsenic groundwater for drinking water and irrigation (Plant *et al.* 2003).

Arsenic levels in sediments are often elevated when iron levels are high and the association with organic matter for arsenic appears to be of less significance compared with other metals. Arsenic tends to be strongly sorbed onto sediments which depends on the anoxic/oxic conditions of the sediment environment (Fergusson 1990).

The average crustal abundance of arsenic is 1.5 μg/g with concentration levels of <10 μg/g reported to be of typical concentration for relatively clean environments. Levels in grossly contaminated sites found to reach high levels of 3732 μg/g ((Denton *et al.* 2001; Plant *et al.* 2003)

Arsenic is highly toxic and can lead to many health issues in humans (Plant *et al.* 2003). High levels of inorganic arsenic (ranging from 60,000 μg/kg in food or water) can be fatal while lower levels (300 to 30,000 μg/kg in food or water) may cause stomach and intestine irritation, with symptoms like stomach ache, nausea, vomiting, and diarrhea (Finkelman 2005). The most characteristic effect of long-term oral exposure to inorganic arsenic is associated with skin changes.

2.6.2 Cadmium (Cd)

Cadmium is usually present as complex oxides, sulfides, and carbonates in zinc, lead and copper ores (Finkelman 2005). Cadmium in several aspects is similar to zinc and therefore, it is always associated with zinc in mineral deposits (Callender 2003). Cadmium is extremely toxic to most plants and animal species particularly in the form of free cadmium ions (Denton *et al.* 1997). The major sources of cadmium include metallurgical industries, municipal effluents, sewage sludge and mine wastes, fossil fuels and some phosphorus containing fertilizers (Denton *et al.* 2001).

In sediments, cadmium does not appear to be absorbed to colloidal material, but organic matter, such as humic substances and organic debris, appear to be the main sorption material for the metal. Cadmium levels tend to increase with decrease in size and increase in density in terms of partition of sediment samples by size and density (Fergusson 1990). The sorption of cadmium to sediments, and to the clay content, increases with pH. The release of cadmium from the sediment is influenced by a number of factors including acidity, redox conditions and complexing agents in the water. Under alkaline conditions, cadmium is less mobile (Fergusson 1990). Also under saline conditions, Cl^- ion affects the mobility of cadmium (Morrison, *pers. comm*.).

The average concentration of cadmium in the lithosphere is ~0.1µg/g and it is strongly chalcophilic (Callender 2003). Concentrations in pristine areas are <0.2 µg/g with levels exceeding 100 µg/g at severely contaminated sites (Naidu & Morrison 1994).

Cadmium can potentially harm human health depending upon the form of cadmium present, the amount taken in, and whether the cadmium is eaten or breathed. If the levels are high enough, the cadmium in the kidney will cause kidney damage and also may cause bones to become fragile and easily broken (Finkelman 2005)

2.6.3 Chromium (Cr)

Chromium like zinc, is one of the most abundant heavy metals in the lithosphere with an average concentration of about 69 and mercury content in carbonate sediments is reported to be 0.03 μg/g (Callender 2003). Chromium is moderately toxic to aquatic organisms. Major coastal marine contributors of chromium are dominated by input from rivers, urban runoff, domestic and industrial wastewaters and sewage sludge (Denton *et al.* 1997). Also other major sources in the aquatic environment include the waste stream from electroplating and metal finishing industry (Table 2.4) (Callender 2003; Finkelman 2005).

Levels of chromium in marine sediments range from 2.4 at unpolluted sites to 749 μg/g at grossly contaminated sites. Under the US EPA sediment quality classification, sediments with chromium levels of less than 25 μg/g are considered as non-polluted whereas sediments with 25-75 μg/g and >75 μg/g are considered to be moderately polluted and heavily polluted with chromium respectively (Denton *et al.* 1997). Calcareous sediments of biogenic origin, such as those found in coral reefs, are often lower and normally contain 3-5 μg/g (Denton *et al.* 2001).

Chromium is carcinogenic to humans and long term exposure has been associated with lung cancer in workers exposed to levels in air that in the order of 100 to 1,000 times higher than usually found in the environment (Finkelman 2005).

2.6.4 Copper (Cu)

Copper is a moderately abundant heavy metal with mean concentration in the lithosphere of about 39 μg/g. It is an essential trace element for the growth of most aquatic organisms however it becomes toxic to aquatic organisms at levels as low as 10 μg/g (Callender 2003). Heavily polluted sediments have been reported to exceed 200 μg/g. Inputs of copper into the natural waters come from various sources

including mining, smelting, domestic and industrial wastewaters, steam electrical production, incinerator emissions, and the dumping of sewage sludge (Denton *et al.* 1997). Algaecides and antifouling paints are identified as major contributors of copper to harbor areas whereas coastal waters are generally receiving inputs from rivers and atmospheric sources (Denton *et al.* 1997). In New Zealand, major coastal discharges of copper come from urban storm-water, contaminated sediments from past and present industrial discharges, antifouling and boat activities (Table 2.6) (Williamson *et al.* 2003).

Copper has a high affinity for clay mineral fractions, especially those rich in coatings containing organic carbon and manganese oxides (Callender 2003) and as a result, residues are often elevated in sediments near localized sources of inputs (Denton *et al.* 1997). Callender (2003) reported that the adsorption behavior of copper in natural systems is strongly dependent on the type and concentration of inorganic and organic ligands.

Copper is essential for good health. However, exposure to higher doses can be fatal. Long term exposure to copper results in nose irritation, mouth, and eyes, and cause headaches, dizziness, nausea, and diarrhea (Finkelman 2005).

2.6.5 Lead (Pb)

Inorganic lead is moderately toxic to aquatic flora and ranks behind mercury, cadmium, copper and zinc in the order of toxicity to invertebrates. Organolead compounds, on the other hand, particularly the alkyl-lead compounds used as antiknock agents in gasoline, are considered toxic to any forms of life (Denton *et al.* 1997).

The main sources of Pb in natural waters include manufacturing processes (especially metals), atmospheric deposition (e.g. from pyrometallurgical nonferrous metal production; the combustion of leaded fuels; the burning of wood and coal; and the

incineration of municipal refuse). Other sources include domestic wastewaters, sewage and sewage sludge (Denton *et al.* 1997).

Lead species are strongly sorbed to Fe/Mn oxides, which are reported to be more important than association with clays and organic materials (Fergusson 1990). The sorption of lead onto Fe/Mn oxides is not affected by aging, but does decrease with an increase in Cl^- ion concentration, due to the formation of lead chloride complexes. The sorption process is pH dependent. As a result, lead is immobile in the aquatic environment and tends to accumulate in sediments close to its point of entry (Morrison, *pers. comm.*). Lead is reported to be in the 15-50 µg/g range for coastal and estuarine sediments around the world (Denton *et al.* 1997) with < 25 µg/g in clean coastal sediments.

Lead enters the body through breathing or swallowing and the main target for lead toxicity is the nervous system. Lead exposure may also cause weakness in fingers, wrists, or ankles and miscarriage for pregnant women (Finkelman 2005).

2.6.6 Mercury (Hg)

Mercury is significantly toxic to aquatic organisms especially when it is present in the organic form (Denton *et al.* 1997). The most common organic mercury compound that microorganisms and natural processes generate from other forms of mercury is methylmercury. This compound is particularly a concern due to its ability to build up in certain edible freshwater and marine fish that are many times higher than that of the surrounding water levels (Finkelman 2005).

Today, the primary anthropogenic source of mercury to the environment is discharge from coal-burning power plants followed by atmospheric fallout from other sources like the high temperature combustion processes (e.g. municipal refuse incineration), chemical manufacturing processes, and discharge of domestic wastes (Denton *et al.* 1997; Fitzgerald *et al.* 2003). Levels of mercury in urban runoff are often very low though relatively high concentrations have been reported in oil and petroleum

products (Denton *et al.* 1997). In New Zealand, geothermal discharges are contributing to mercury discharge to the coastal environment (Williamson *et al.* 2003). However, inputs of mercury into the environment through the atmosphere are of particular concern. Such significant inputs coupled with long-distance transport of mercury have resulted in elevated concentrations of mercury in fish far away from anthropogenic sources (Fitzgerald *et al.* 2003).

Baseline levels of total mercury are reportedly in the order of 30 ng/g (parts per billion) in clean sediments, which is also the suggested mercury content in carbonate sediments (Denton *et al.* 1997; Fitzgerald *et al.* 2003). However, levels found in pristine sediments from the tropical north Queensland coast of Australia were below 30 ng/g of levels in pristine environment and ranged from 4-6 ng/g (Denton *et al.* 1997).

Mercury has high affinities for organic carbon as well as sulfides. In marine and estuarine environments, where seawater provides sufficient sulfate, rates of sulfate reduction are affected mostly by availability of organic matter and temperature (Fitzgerald *et al.* 2003). In seawater, mercury is associated with surface active organic materials and mercury ions and the metal are rapidly sorbed by sediments. Factors influencing the sorption of mercury, follows the order (Fergusson 1990):

surface area > organic > cation exchange capacity (CEC) > grain size

Mercury may enter the body by consuming contaminated fish, shellfish or marine mammals (whales, seals) that come from mercury-contaminated waters. Once mercury is in the blood stream, it moves rapidly to most tissues and the brain. The nervous system is very sensitive to mercury. The kidneys are also sensitive because mercury accumulates in the kidneys and causes higher exposures to such tissues, hence more damage (Fitzgerald *et al.* 2003).

2.6.7 Nickel (Ni)

Nickel is moderately toxic to most species of aquatic plants, though it is one of the least toxic inorganic agents to invertebrates and fish. The major source of discharge to natural waters is municipal wastewater followed by smelting and the refining of nonferrous metals (Denton *et al.* 2001). Also mine drainage effluents are known to be major contributors due to high concentrations of nickel found in the discharges (Finkelman 2005). Typically, nickel residues in sediments can be up to 100 µg/g or higher but may fall below 1 µg/g in some clean coastal waters (Denton *et al.* 1997) with the average concentration of nickel in the lithosphere of 55 µg/g (Callender 2003).

Major uses of nickel include its metallurgical use as an alloy, plating and electroplating, a major component of Ni-Cd batteries, and as a catalyst for hydrogenating vegetable oils (Callender 2003). Therefore, nickel enters the landfills when objects including batteries containing or coated with such metal discarded and ended up in the landfill (Leong *et al.* n.d.).

In the bottom sediments of estuaries in which anaerobic conditions often occur, sulfide tends to control the mobility of nickel. However, under aerobic conditions, the solubility of nickel is mainly controlled by either the co-precipitate $NiFe_20_4$ or $Ni(OH)_{2(s)}$ (Callender 2003).

Some of the most serious health effects due to exposure to nickel include reduced lung function some nickel compounds are reported to be carcinogenic to humans and metallic nickel my also be carcinogenic (Finkelman 2005).

2.6.8 Zinc (Zn)

Zinc is a very common environmental contaminant and usually outranks all other metals considered in this study in terms of abundance and it is commonly found in association with lead and cadmium (Denton *et al.* 1997; Finkelman 2005). Although it is not regarded as particularly toxic, it is sometimes released into the sea in substantial quantities (Denton *et al.* 2001). Major sources of Zinc to the aquatic environment include the discharge of domestic wastewaters; coal-burning power plants; manufacturing processes involving metals; and atmospheric fallout (Denton *et al.* 2001). Approximately one third of all atmospheric zinc emissions are from natural sources, the rest come from nonferrous metals, burning of fossil fuels and municipal wastes, and from fertilizer and cement production (Denton *et al.* 2001; Callender 2003).

Sediments are known as major sinks for zinc in the aquatic environment, and residues in excess of 3000 μg/g have been reported close to mines and smelters (Denton *et al.* 2001). The highest sedimentary zinc levels are found to be from enclosed harbors reaching as high as 5700 μg/g. This is mainly due to restricted water circulation and also particularly prone to zinc contamination from a variety of localized sources including brass and galvanized fittings on boats, wharves and piers; zinc-based anti-corrosion and anti-fouling paints (Denton *et al.* 1997).

The average zinc content of the lithosphere is approximately 80 μg/g (Callender 2003) and depending on the local geology, sediments from uncontaminated waters typically contain zinc concentration in the order of 5-50 μg/g. However, levels normally encountered in carbonate sediments from pristine reef waters, away from coastal developments are usually less than 1 μg/g (Denton *et al.* 2001).

Taking excess zinc into the body through food, water and dietary supplements can have adverse effects on health. Ingesting high levels of zinc for several months may cause anemia, damage to pancreas, and decrease levels of high-density lipoprotein (HDL) cholesterol (Finkelman 2005).

2.7 Factors affecting metal concentrations and fate in marine sediments

Enrichment of trace metals from anthropogenic sources in the estuaries and marine environments has become a serious human health and environmental concern. Once trace metals are deposited in sediments, they undergo a series of physical, biological and chemical processes termed "diagenesis", which determines new chemical phases in the sediment (Williamson & Wilcock 1994). Diagenetic processes have a strong influence on the form of trace elements. Trace metals occur in a number of different forms, mainly in the dissolved (e.g. free element ion, complexed by inorganic ions such as chloride and sulphide, and complexed by dissolved organic matter) and in the solid state (adsorbed onto surfaces of clays, element oxides, organic material, co-precipitated with sediment phases and incorporated into organic matter) (Williamson & Wilcock 1994). The highest proportion is usually in the solid phase. How an element is bound in sediment determines the biologically active fraction and its fate and cycling (Williamson & Wilcock 1994).

Understanding the processes affecting trace metals concentration and their fate in sediments is important in gauging appropriate designs for sampling and monitoring programmes (Williamson & Wilcock 1994) and planning for appropriate remediation options.

2.7.1 Inputs (proximity to source)

Increased inputs of metals in such forms available for association with sediments ultimately result in increases in metal concentrations in sediments (Luoma 2000). Metal enrichment in the sediments is usually located close to past and present sources of pollution (Williamson & Wilcock 1994) and in particular the sheltered environments where discharge is restricted (Williamson *et al.* 1992). This is supported by numerous studies which have demonstrated that metal inputs from human activities are reflected by metal concentrations in sediments (Luoma 2000).

Simply, concentrations drop with distance away from the source of input as determined by the physical processes that dilute and redistribute the particles with which metals associate such like the dilution with less contaminated sediments (Fergusson 1990; Luoma 2000). However, the rate of decline in concentration with distance from sources of pollution is extremely variable (Williamson & Wilcock 1994). Metal concentration in sediments are not only determined by metal inputs but also effected by other complex factors such like sediment characteristics and reactions at particle surfaces that influence the quantity of metal adsorbed, and reduction/oxidation reactions (Luoma 2000).

Many marine and estuarine systems are subject to multiple metal inputs from human activities that make dilution trends hard to determine. For example, Luoma (2000) reported that in San Francisco Bay, metal distributions in sediments are complex, reflecting local metal inputs, zones of dilution around each, and localities outside the influence of point source inputs. These complexities make it difficult to characterize regional trends in contamination of sediments of the Bay.

2.7.2 Sediment texture

Metal concentrations in marine and estuarine sediments are determined not only by metal inputs but also by sediment characteristics. Grain size distribution has been recognized to affect trace metal concentrations in estuarine and marine sediments (Luoma 2000). Generally, trace metals are mainly concentrated in the silt/clay sediment fraction, consisting of particles with a grain size <0.063 mm (Krumgalz *et al.* 1992). The enrichment of the silt/clay fraction by anthropogenic trace metals is due to the large specific area of this fraction and to the strong adsorptive properties of clay minerals (Krumgalz *et al.* 1992).

Fine grained sediment of this particular fraction has a high absorption potential because the sediment has a large surface area to ratio, and contain large amounts of interstitial water (Loomb 2001). The feature of clay that is of significance regarding the trace metals is their ability to absorb metal ions by their outer sheath of hydroxyl groups (Fergusson 1990). The surface of clays may be also negatively charged which

is crucial in providing potential absorption sites for metal ions (Fergusson 1990; Williamson *et al.* 1992). Clay minerals also have a higher surface area to volume ratio and can absorb material into their lattice framework (Loomb 2001).

Luoma (2000), however, found that the relationship of metal concentration and grain size distribution is sometimes complicated. For example, silt/clay sizes particles in estuarine and marine sediments are usually aggregates and particle aggregates can complicate the relationship of metal concentration and surface area. Another potential factor that can affect this relationship is the physical and chemical processes that may affect the degree of accumulation of metal-reactive sediment components on the sediment surface (e.g., movement of water across a redox boundary may facilitate accumulation of iron and manganese oxides) (Luoma 2000).

2.7.3 Sediment composition

One of the most important sediment characteristics is the concentration of sediment components. The concentrations of the important metal-reactive components of sediments (Fe and Mn hydrous oxides, sulphides and organic materials) can vary among estuarine and marine environments (Luoma 2000) and are found to increase as sediment textures become finer (Williamson & Wilcock 1994). The sediment components (particulate matter) contain many phases that strongly adsorb metals such as amorphous iron oxides, manganese oxides, and polar organic matter (humic acid, carbohydrates, proteinous material) (Williamson *et al.* 2003).

The nature of Fe oxides, organic materials, or Mn oxides will determine site densities for sediment components; and the relative concentrations of Fe, Mn and organic materials will determine aggregate differences in binding site density or binding intensity (Luoma 2000). Amorphous iron hydrous oxide (FeOOH) has a large surface area per unit weight. It is abundant and it may be an important medium for trace metal adsorption. Concentrations do not change greatly with depth or redox conditions, though are strongly dependent on sediment texture (Williamson & Wilcock 1994). Amorphous manganese oxide (MnOOH) concentration is lower than iron oxides, though its rapid cycling between reduced and oxidized forms, and

between sediments and water mean that it is important in trace metal mobilization from sediments (Williamson & Wilcock 1994).

Generally organic matter content of sediments increase as the sediment texture becomes finer (Williamson & Wilcock 1994; Denton *et al.* 2001). The presence of organic matter can potentially increase metal concentrations in sediment by adsorption of metals from surrounding environment onto organic material (Loomb 2001). Also dead organisms in sediments may carry the heavy metals with them, either taken in by the organism while alive or sorbed on to the animal before or after death (Fergusson 1990) and this contribute directly to the metal levels in the sediments. Organic compounds containing metal ions may also be sorbed onto Fe-Mn oxides.

Organic materials can affect metal species solubilization by complexing the metal ions, but they can also take metal ions out from the solution and contribute to the sediments. Decomposition of organic material produces organic ligands that may extract metals from the sediments which can effectively mobilize metals by increasing their concentration in the water (Fergusson 1990; Taulis 2005). Interestingly, the Irving-Williams series of increasing stability of metal complexes, is the same order as increasing mobilization (Fergusson 1990);

$$Mn < Fe < Co < Ni < Cu > Zn$$

Therefore, the subsequent fate of metal-reactive components of sediment (Fe and Mn hydrous oxides, and organic materials) will determine the fate of the associated heavy metals (Williamson *et al.* 2003).

2.7.4 Reduction/oxidation reactions

Biological oxidation of metabolisable forms of organic carbon has been recognized as the most important factor in early diagenesis (Williamson & Wilcock 1994). In the surface sediments, organic matter is decomposed by organisms in the presence of

oxygen. In finer sediments such as clay, with the exception of coarse sediments and other high-energy areas where there is rapid advection of oxygenated water, this decomposition uses up oxygen in the sediment quickly. The decomposition occurs faster than the rate of diffusion of oxygen into the sediments and, as a result, most sediments are anaerobic just below the surface (Williamson *et al.* 2003). The decomposition of organic matter proceeds under anoxic conditions using alternative electron acceptors to oxygen, such as nitrate, manganese and iron oxides, and sulphate to oxidize (metabolise) organic carbon (Williamson & Wilcock 1994; Williamson *et al.* 2003). This oxidation, together with the resulting anoxic conditions, produces large changes to the form of iron, manganese and sulphur, which are important in 'binding' trace metals in sediment and releasing them to the overlaying water (Williamson & Wilcock 1994).

For instance, when sulphate is used, sulphide is produced and heavy metals react with the sulphides and produce soluble precipitates (Williamson *et al.* 2003). Sulphide also reacts with Fe to form black ferrous sulphide and pyrites (FeS_2) to give the subsurface sediments their characteristic black and grey color. Mercury is different. While it undergoes similar reactions to other metals, microbiota changes a proportion to organo-mercury (methylmercury) a toxic compound that has the ability to bio-accumulate in the food chain (Williamson *et al.* 2003).

In sediments with high inputs of organic matter, reactions are concentrated close to the sediment-water interface. In addition, iron, manganese and sulphur in many oxidized and reduced forms can react with each other abiotically. All these reactions contribute to the complex sediment chemistry (Williamson & Wilcock 1994).

2.7.5 Adsorption/desorption

The release of metals from oxic sediments can happen as the salinities increase or reduction/oxidation condition changes due to the high ionic strength of seawater and potential competition for binding sites with Ca and Mg (Luoma 2000). The flux of oxygen from the water to the sediments controlled the reaction at the interface, hence the depletion of oxygen consequently limits the release of the metals into the water.

The extent of losing the metals from oxic sediments can be reduced by readsorption to precipitating Fe and Mn oxides (Luoma 2000). The diagram in figure 2.2 illustrates how the pollutants move within the hydrosphere.

Metal removal from estuarine sediments can occur due to desorption of metals from particulates to solution when freshwater and seawater mix and this has been demonstrated in the laboratory experiments (Luoma 2000). However, readsorption also takes place at the freshwater/seawater interface in estuaries due to the formation of new particles, increased turbidity, increased pH and the occurrence of complex recirculation patterns (Williamson & Wilcock 1994; Luoma 2000).

The above processes are not only fueled by the changing redox environment and pH but also by the metabolism of carbon organic matter, changing environmental conditions brought about by bioturbation (muds and sands) or waves (sand) and irrigating the sediment (Williamson & Wilcock 1994).

2.7.6 Physical transport

Sediment type and dynamics are known to affect contaminant concentration and fate (Williamson & Wilcock 1994). In coastal environments with low hydrodynamic energy, fine particulates tend to be trapped, while in areas where hydrodynamic energy is high, fine particulates are 'moved on' (Williamson *et al.* 2003). There are 3 main zones that can be distinguished based on three types of processes (Williamson & Wilcock 1994; Williamson *et al.* 2003).

i) areas of accumulation (deposition). This is where fine materials are continuously being deposited. Wave and current energies are very low in this area;
ii) areas of transportation. This is where fine materials are deposited discontinuously, i.e. periods of accumulation are interrupted by periods of periods of remobilization (generally of short duration and associated with storms); and

iii) areas of erosion. This is where there is little deposition of fine materials. Wave and/or current energies are high in this area.

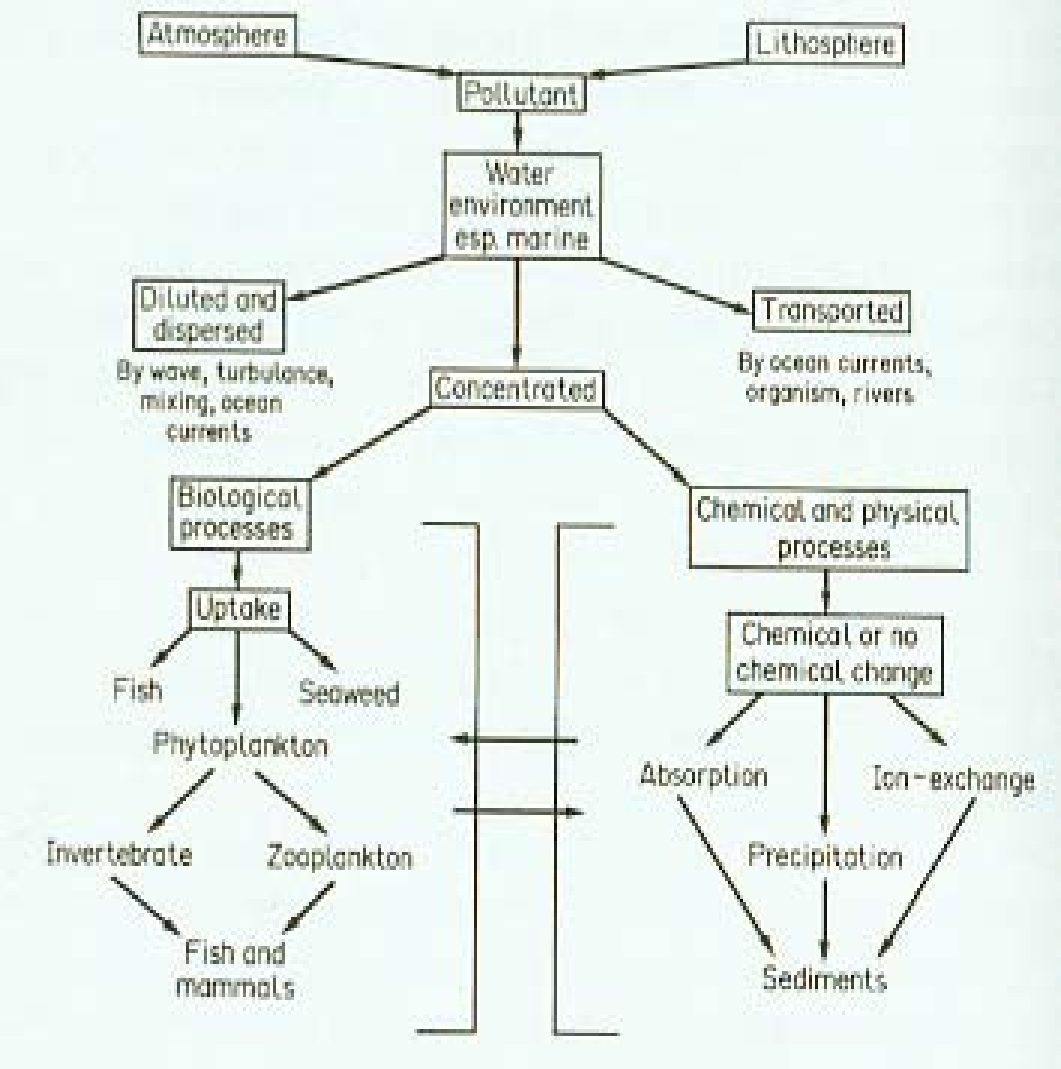


Figure 2.2: Movement of pollutants in the hydrosphere (source Fergusson (1990))

Bare rock, gravels and sand characterise areas of erosion while sediments in areas of transport are diverse; ranging from muds to sands. The deposits within areas of accumulation are typically muddy, with a high water and organic matter content (Williamson & Wilcock 1994).

For most metals, the intermediate fate is the deposition area where the finest sediments generally accumulate (Luoma 2000; Williamson *et al.* 2003). These areas

tend to be sheltered estuaries and embayments, and deeper water offshore. The ultimate fate of heavy metals is burial given that metals do not break down. When buried, heavy metals become immobilized as insoluble sulphide precipitates due to the decomposition of organic material (dead animals, plant material) (Williamson *et al.* 2003).

2.7.7 Bioturbation

Numerous studies have suggested that bioturbation strongly affects the physical, chemical and biological properties of sediments (Williamson & Wilcock 1994). Biological mixing is more important in muddy sediments than sandy sediments, where strong physical processes occur. There are two groups of animals based on the magnitude of sediment disturbance; i) large individual predators, such as rays and crabs, may shift large amount of sediment when searching for food, and ii) smaller animals such as worms and gastropods. The latter burrow and ingest sediment, shift only small amounts of sediments, but their high population densities make them important in sediment turnover (Williamson & Wilcock 1994; Hayward *et al.* 2004). This process of bioturbation is responsible for the homogeneous nature of the top 60 cm of the sediments of Chesapeake Bay (Fergusson 1990).

Bioturbation has a major impact on the chemistry of muddy sediments. Oxygen penetration in unbioturbated sediments is highly restricted, and the redox boundary occurs a few millimetre below the surface (Williamson & Wilcock 1994). Marine organisms, such as polychaetes and crabs, create extensive 'honeycombs' in mud flats up to 60 cm in depths allowing oxygen to be transported deep within the sediments via burrows (Leong *et al.* n.d.; Williamson & Wilcock 1994). These 'honeycombs', in turn, provide low tide irrigation of burrow water, which maintains oxic conditions when tidal flats are submerged (thus facilitating the exchange of solutes between sediments and water). In this way, the sediment becomes a honeycomb of oxic and anoxic environments (Leong *et al.* n.d.).

Burrowing organisms also can alter the levels and the speciation of heavy metals in sediments by affecting the sediment stratigraphy to become blurred, particle size

altered, pore spaces and pH changed (Fergusson 1990). Consequently, metal concentrations do not appear to change rapidly with depth (Williamson & Wilcock 1994). The organisms may also bio-accumulate heavy metals and remove them from the sediment profile (Fergusson 1990) hence decreasing the metal concentrations in the sediments.

2.8 Effects of heavy metal contamination in sediments

2.8.1 Sediments (metal-polluted)

Heavy metals are preferentially transferred from the dissolved to the particulate phase (Gangaiya *et al.* 2001) and this results in the elevation of metal concentrations in estuaries and marine sediments. Therefore, concentrations often exceed those in overlying water by several orders of magnitude (Langston 2000). Because sediments can accumulate heavy metals, concentrations can be high and ultimately become potentially toxic (Williamson *et al.* 2003). Exposure and uptake of even a small fraction of sediment-bound metal by organisms could have significant toxicological significance, in particular where conditions favor bioavailability. In addition, increased metal concentrations in pore water may contribute significantly to sediment toxicity (Langston 2000).

Evidence of fatal effects of metal-polluted sediments can be determined by the absence of sensitive species or by the development of resistance mechanisms and adaptation in tolerant forms such like efficient excretory features in organisms (Langston 2000). Binning and Baird (2001) reported that the density and diversity of nematode communities in the Swaetkops River estuary, South Africa, are influenced by the degree of heavy metal contamination in the sediments. Many of the metals have no known biological function in the marine environment, but can act synergistically with other chemical species to increase toxicity (Binning & Baird 2001). The potential impacts of accumulating levels of heavy metals can be estimated

by comparing the concentrations of contaminants of interest present in sediments with sediment quality guidelines (SQGs) (Williamson *et al.* 2003).

The Sediment Quality Guidelines (SQGs) have been derived from large databases, which correlate the incidence of adverse biological effects with the concentrations of contaminants in sediments (Long *et al.* 1995) and are used in the evaluation of sediment contamination and potential ecotoxicological effects (Casado-Martinez *et al.* 2005). One of the most widely used SQGs for estuarine and marine environments, which will be applied in this study as well, is a set of effects-range guidelines developed by the National Oceanic and Atmospheric Administration (NOAA) (McCready *et al.* 2005) in the U.S. (Long *et al.* 1995). There are two levels of risks considered under this SQG – effects low range (ERL) and effects range-moderate (ERM) (Guerra-Garcia & Garcia-Gomez 2005). Concentrations below the ERL value are rarely associated with biological effects while concentrations above the ERL but below the ERM indicate a possible range in which effects would occur occasionally. Concentrations above ERM are associated with frequent adverse ecological effects on the benthic communities (Guerra-Garcia & Garcia-Gomez 2005).

2.8.2 Ecology

Given that metal concentrations in sediment are normally higher than those in the overlying water, macrophytes, at the primary production level, rooted in these enriched sediments tend to have higher concentrations than the sediment (Mance 1987). This is due to the uptake of metals not just only from the sediment via the roots but also from the surrounding water.

The effect of heavy metal contaminants in the sediment on benthic organisms can be either acute or chronic (cumulative) (Binning & Baird 2001). No matter whether metals are essential or not, all heavy metals form an important group of enzyme inhibitors when natural concentrations are exceeded. Therefore, organisms living in or adjacent to metal contaminated sediments may suffer toxic effects that can be fatal in highly contaminated situations to relative abnormal metabolic adjustments at the

sub-lethal level (Denton *et al.* 2001). In addition, metal enrichment in estuaries and coastal environments is a major concern as heavy metals have the ability to bio-accumulate in the tissues of various biota, and can ultimately affect the distribution and density of benthic organisms, as well as the composition and diversity of infaunal communities (Binning & Baird 2001).

A wide range of criteria to assess the impact of metals on marine organisms have been developed during the last few years and consequently, effects are now recognized at a much lower levels than in earlier LC_{50} studies (Langston 2000). Growth, reproduction, and recruitment are usually the processes most susceptible to metal stress.

The toxicity of a trace element to an organism depends on the metal chemical species, its concentration and the organism being affected (Sedgwick 2005). As for the organism, toxic effects occur when excretory, metabolic, storage, and detoxicification mechanisms are no longer have the capacity to match uptake rates (Langston 2000). This capacity may vary between phyla, species, populations, even individuals and can also depend on the stage in the life history of the organism (Langston 2000).

2.8.3 Human health

The consumption of marine food is the principal path to human exposure to heavy metals. Effects on humans can be observed after either a one-off exposure to a large non-lethal dose (acute) or after repeated exposure to lower doses (chronic) (Sedgwick 2005).

Heavy metal contamination has become a subject of public interest because humans have been harmed by metal contamination (Chandra 2002). One of the well-documented incidents of metal contamination that attracted greater public interest was an industrial mercury contamination on Minimata Bay in Japan caused by severe health effects in consumers of fish and shellfish from the area and led to the identification of methyl mercury as an extremely toxic compound (Chandra 2002).

Methyl mercury can bio-accumulate in the food chain and its accumulation in some fish represents a risk to those people who consume a large proportion of fish in their diet (Williamson *et al.* 2003).

Chapter Three: Study Sites Description

3.1 Introduction

This chapter presents a detailed description of the two study sites; the Betio (Tarawa), Kiribati study site and Te Atatu Peninsula (Auckland), New Zealand study site.

The objective of this chapter is to look at each site independently, particularly on those features that are likely to have a bearing on the investigation of heavy metal contamination in the coastal marine environment. The chapter will begin with the description of the existing environment followed by the background description of the landfills for each case study.

3.2 Kiribati – case study

3.2.1 Physical setting

The study site was undertaken on the islet of Betio, Tarawa atoll. The capital island is part of the Gilbert Islands group, which is located in the central Pacific Ocean just north of the equator (01º 20'N; 173º, 00'E) (Weber & Woodhead 1972; Kaly 1996). The atoll is triangular in shape, with approximate dimensions of 25 km E-W and 30 km N-S. Betio islet lies at the extreme western point of Tarawa, and was isolated from the rest of South Tarawa until June 1987 when the causeway was built (Figure 3.1) (Howorth & Radke 1991). The V-shaped atoll is made up of chain of islets along the southern and north-eastern perimeters of the atoll, enclosing a wide, shallow lagoon, open to the west (Consulting Environmental Engineers 1985). The islets are narrow, rarely more than 100 to 400 metres wide (Byrne 1991). The

southern part of the atoll rim is now completely linked by causeways (Forbes & Biribo 1996). It is 93.4 km in circumference with a lagoon area of 400 km², but has a land area of only 21 km² with maximum elevations of about 2-3 metres (Zann 1982; Forbes & Biribo 1996).

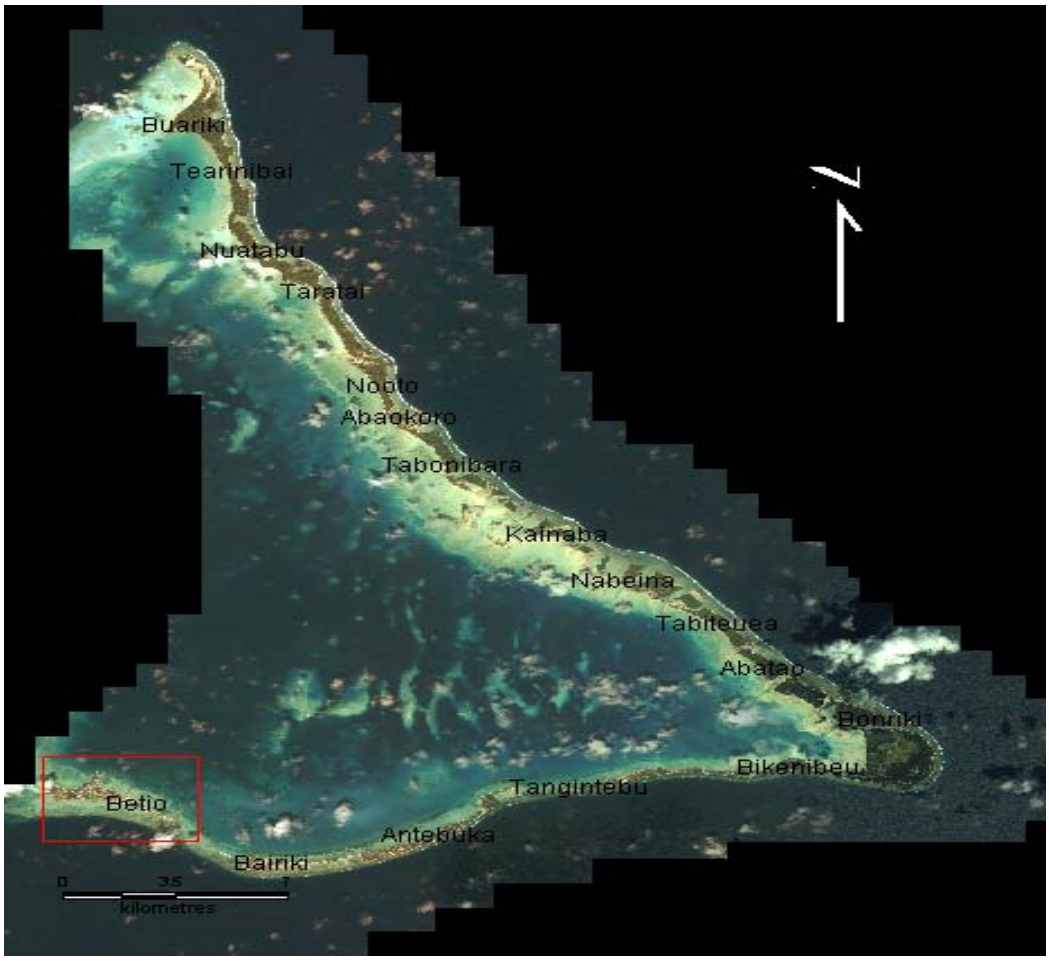


Figure 3.1: Tarawa atoll showing the location of Betio Islet (2004 IKONOS Imagery – SOPAC Archive).

3.2.2 Tarawa atoll lagoon

Despite its size (400 km²), the lagoon is relatively shallow, with an average depth between 12 and 15 m, and a maximum depth of 25 m. Irregular topography characterizes the lagoon floor, with numerous patch reefs, and coral pinnacles are fairly common (Weber & Woodhead 1972; Richmond 1990). Along the western margin of the atoll, the reef is submerged to depths of about 8 to 10 metres. The main

entrance to the lagoon is a breach in the leeward, submerged reef, located just north of Betio (Weber & Woodhead 1972).

Tarawa lagoon is the most important water body on Tarawa. It plays a major part in the life of the local population residing on Tarawa (41,194 at the 2000 census). Many people are dependent on the lagoon to meet their minimal nutritional requirements and it is widely used for fishing, gleaning shellfish and recreational activities. With the 2000 population exceeding 40,000 for Tarawa, it would mean that 20 times as many people may rely on the lagoon for their food source as did only forty years ago (Abbott & Garcia 1995b).

Over the last quarter century, Tarawa lagoon has become heavily impacted by the human activities. Primary sources of anthropogenic disturbance in these waters include the Betio Port and Shipyard (boat repair shop), wharves, navigation channels and the various land-based pollution sources, mainly the Betio municipal landfill (plus 2 new landfills – Anderson causeway and Bikenibeu), domestic wastewater, industrial wastes, agricultural activities, household hazardous substances and non-point urban run off (Gangaiya 1994).

The construction of causeways, which now connect all islets on the southern arm of the atoll, significantly disrupted sediment transport regimes on the coast and affected the circulation and flushing of the lagoon water. Land reclamation and coastal dredging are also common. An intensive study on circulation in the lagoon water suggested that opening the causeways will enhance flushing of the lagoon only about 10 percent faster than at present (Abbott & Garcia 1995a).

A stream of studies on the lagoon indicated a deterioration of the lagoon's natural resources since the 1970's (Abbott & Garcia 1995a). Some of these previous studies include the investigation on the ecology of the lagoon (Zann 1982; Paulay 1995) and water quality and contamination (Naidu *et al.* 1991).

3.2.3 Formation and sediment geology

Tarawa atoll is a limestone cap covering a peak of a submarine mountain range several thousand metres high. Little is known of its geology, but it is probably similar to the well-studied Bikini and Eniwetok atolls in the Marshall Islands to the north (Zann 1982). These submarine mountains, of volcanic origin, probably once emerged above sea level in pre-Tertiary times but were later eroded to sea level by natural wave action. During the Eocene period corals grew around the edges over the plateaus during periods of subsidence or higher sea levels. During periods of emergence, they were exposed as high, flat topped islands (Zann 1982).

Coral growth was extensive around the islands during the Pleistocene when the sea level lowered to over 100m below the present. The gradual rise in level over the past 20,000 years covered the islands, but coral growth continued upwards at a similar rate to create a perimeter of coral (Zann 1982). Vertical growth rates at Tarawa vary between 5 and 8m per 1000 years, and this has resulted in a predominantly coral head/mixed branching-detrital facies beneath the windward margin (Marshall 1991).

The present islands were formed in recent times as the reef top was eroded and sediments were deposited by waves and currents. The islets of Tarawa were developed on a raised coral conglomerate. This is typically 0.5-1.3 metres above mean sea level and forms natural groynes in many places along the ocean shore and locally on the lagoon shore as well (Forbes & Biribo 1996). The islets are composed primarily of coral rubble, coral and foraminfera sands and beach rock overlying a limestone platform (Kaly 1996).

Weber and Woodhead (1972) examined textural properties, mineralogical composition of and biological diversity among the Tarawa lagoon sediments. Their study found: a) a westward coarsening of lagoon sediment; b) aragonite is the most abundant mineral component; and c) weathering of grains increases to the east making identification more difficult. At the extreme western part of the lagoon, the constituent carbonate grains appear to be fresh and not greatly abraded hence making

it relatively easily identified. However, the general trends for four different biological components are percentage coral increasing to the west and percentage Halimeda, mollusks, and echinoderm increasing towards the east. Halimeda fragments are the overall dominant component of the lagoon sediments. This pattern is consistent with the observation that live corals increase towards the west (Environment Consultants Kiribati 2002).

Little is known about the vertical distribution of sediments in the Tarawa lagoon. Studies using lagoonal boreholes and high-resolution geophysical surveys in the area around the Betio-Bairiki causeway indicate the unconsolidated sediment that varies in thickness between about 1 m and at least 7.5 m (Figure 3.1). The sediment is primarily made up of a mixture of mud (primarily silt), sand, and gravel (mainly coral heads), however, the components that made up the sediments vary significantly over short distances and appear to be related to the proximity of patch reefs and pinnacles. The coarser materials appeared to be produced at the reefs while the finer material is accumulating away from the reefs (Richmond 1990).

3.2.4 Climate

The climate is an oceanic-equatorial one. Tarawa is located in the tropical convergence zone where annual rainfall is relatively high at 1.5 m (Kaly 1996). The climate is governed by the ocean: for most of the year (March to October) prevailing light East to South-East trade winds bring settled weather; while the stormy season is associated with westerly winds. The general trend for Betio long-term rainfall (1947-2004) is consistent with this pattern with high rainfall occurs during the period of westerly winds (November to March) (Figure 3.2).

When the Southern Oscillation Index (SOI) is negative these conditions occur more frequently (Zann 1982; Kaly 1996). Cyclones are rare as tropical cyclones rarely form within 5º either side of the equator.

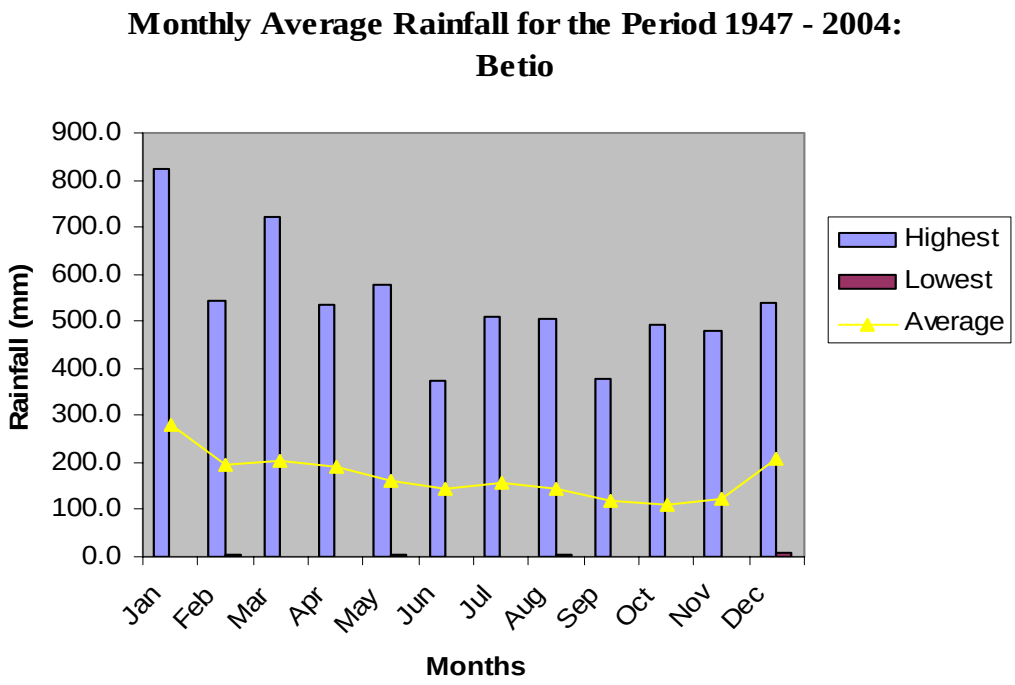


Figure 3.2: Average monthly rainfall for the period 1947 – 2004: Betio (Source: Kiribati MET).

Trade winds blow from the east about 81% of the time (Kaly 1996; Environment Consultants Kiribati 2002). A study in 1990 established the probability analysis of wind speed indicating that winds in the sector North-East to South-East are prevailing. The results indicated that the following wind speeds occur with a return period of 50 years:

330º :23.3m/sec

0º :15.1m/sec

30º :15.1m/sec

The most appropriate value for Betio is a wind speed of 23.3m/sec (50 year return period) (Environment Consultants Kiribati 2002). Air temperature varies little from the mean of 25ºC (at night) to 32ºC (at day) while water temperature remains close to a mean of 32ºC (Zann 1982).

3.2.5 Ecology

The marine ecology of Tarawa lagoon is influenced by the unusual combination of environmental and human-related factors that characterize the lagoon (Paulay 1995). Tarawa lies in the equatorial upwelling region, and therefore the concentration of inorganic nutrients, phytoplankton biomass, and productivity are considerably higher than that of atolls in the subtropical gyres (Kaly 1996; Environment Consultants Kiribati 2002). Because of Tarawa's unusual triangular shape, with islets closing off much of the eastern and southern sides, most of the exchange between the lagoon waters and the surrounding ocean occurs across the submerged western barrier reef (Figure 3.1). This limited exchange creates marked west-east and north-to-south gradients in nutrients, phytoplankton, sediment composition and bioerosion across the atoll lagoon (Paulay 1995).

Coral cover and diversity also follow this general pattern as a consequence of the one-sided opening of the atoll. Algae are abundant and the reefs are dominated by fragmenting corals in much of the central lagoon, probably due to high productivity. Many of the reefs in the northern lagoon and western part of the lagoon appear healthy but little live coral remains in the southeastern lagoon (Paulay & Kerr 1995). This is reflected in the sedimentological gradients discussed earlier.

Suspension and deposit feeders dominate the macrobenthos of the deep lagoon; mainly echinoids, gastropods, bivalves and polychaetes. The shallower lagoon slope is dominated by an assemblage of echinoids, polychaetes and mollusks. Seagrasses are locally dominant producers in the benthos and support a greater abundance of shellfish. Both the seagrass beds and associated shellfish resources appear to have increased substantially since World War II, highly likely as a result of increased fertilization by sewage-derived nutrients (Paulay 1995). Zann (1982) noted severe damage to lagoon corals of southwest Tarawa reefs corals resulting in large area of in-situ dead colonies (as high as 95% of the lagoon bed surface). Some of this damage is man induced via dredging and pollution.



Figure 3.3: Betio's extensive mud flat during low tide.

Betio's wide (average 250 metres) lagoon sand flats were biologically impoverished (Figure 3.3) (Zann 1982). Organisms present in lagoon sediments include the gastropod *Oliva* and crab *Calappa*. Filamentous algae, holothurians *H. atra* and *S. chlorotus* and snails *Strombus luhuanus* and *S. gibberulus* were also present (Zann 1982). A blue green algae was dominant close to the Betio landfill with few invertebrates (Kaly 1996).

3.2.6 Coastal sedimentation

Several studies have been done on sediment type and composition, sediment transport and coastline changes (accretion/erosion) in South Tarawa (Webb 2005). These studies established that the soft coastal environments of South Tarawa are dynamic and sensitive environments, which are subject to natural change over time. Also disturbance and additional changes have occurred due to engineering on a domestic (hand-built seawall) and national (causeways, wharves, channels) level (Webb 2005).

Tides are semidiurnal with a maximum tidal range of 2.3 m (average close to1.5 m) (Howorth & Radke 1991). Tidal currents tend to be weak, except in the deep channel through the western reef where current speeds are high. The nearshore current is estimated to be 0.3 to 0.5 knots (0.25 m/s) (Environment Consultants Kiribati 2002). Currents through the inter-islet channels vary with tidal cycle and wind direction, but are predominantly lagoon-ward due to the wave set up on the ocean side reef (Kaly 1996).

A design wave height of 1.7 m has been calculated for the Betio Port area. Clearly, the design wave height for any seawall/breakwater at Betio should not be lower than 1.7 m. Lagoon waves generated by the northeast-east winds and the westerlies play a key role to the sedimentation, erosion and accretion of the lagoon coast on the southern rim and south west of Tarawa atoll (Environment Consultants Kiribati 2002).

Transport patterns of the lagoon sediments inferred from the historical aerial photos, current satellite images, and field observation indicate a predominantly eastward transport pattern. This possibly relates to tidally driven flows rather than wind-dominated (easterly trade winds) transport. However, the lagoon coastline is now subject to heavy level of disturbance (seawalls, groynes, navigation channels, reclamation and beach mining) that net transport patterns are frequently disrupted and therefore, hard to conclusively establish (Webb 2005).

3.2.7 Urbanization

The population of Tarawa atoll was 41,194 at the 2000 census of which 36,717 (89%), reside on the southern arm of the atoll, known as South Tarawa, with a land base of only 1571 ha mainly where the urban development is restricted (Forbes & Biribo 1996; Tiroa 2001). The 2000 population for Betio islet was 12268 and was projected to have a population density of 8700 per km² in 2010 (Forbes & Biribo 1996).

The huge urban population has been well supported by the many developmental changes that have taken place over past fifteen years (Environment Consultants Kiribati 2002). There has been a significant economic development, with an increasing number of buildings, both commercial and residential, for government, churches and private ventures. Building has doubled since independence from Britain in 1979.

Urban sprawl and a growing population place increased environmental pressure on infrastructure (water resources and waste disposal services), social services (school and health care) and environmental services (environmental degradation). Due to changing values and lifestyles, increasing population densities and urbanization on South Tarawa, including Betio, there are now evident problems with overcrowding, pollution, overexploitation of resources, sanitation and public heath. Problems with the supply of safe drinking water, water for washing, sanitation, the proper disposal of solid and hazardous wastes, and coastal degradation have been ongoing for at least the past 20 years on South Tarawa (Gangaiya 1994; Kaly 1996).



Figure 3.4: Betio Islet showing the landfill and the Betio Port on the right (2004 IKONOS Imagery – SOPAC Archive).

3.2.8 Background on Betio Landfill

Betio landfill, also known as Red Beach landfill, is located at the historical site named "Red Beach" on the western side of the Betio Port, Betio islet (Figure 3.4). The Betio Town Council (BTC) first used this site as a dumping area in 1987 without any seawall (Iangaibo, *pers. comm*.) (Figure 3.5). A seawall made up of sand bags and reef mud was constructed by the Public Works Division (PWD) in 1997 to contain the rubbish at the site and since then the BTC used the site to dump solid waste from Betio (Figure 3 .6) (Table 1.1).

The site covers an area of approximately 1.8 ha, and has a capacity of about 54,000 m^3 with a seawall measuring 310 m in length and 3.05 m in height (Figure 3.7) (Kiribati SAPHE Project n.d.).

The Betio landfill currently in operation has secure galvanized fencing around the perimeter and receives solid waste from the BTC, commercial and private entities. The site is currently approximately one-third full, with remaining capacity of around 30,000 m^3. However, compacting and leveling of the dumped waste by compactor or heavy machinery has not been undertaken (Kiribati SAPHE Project n.d.).

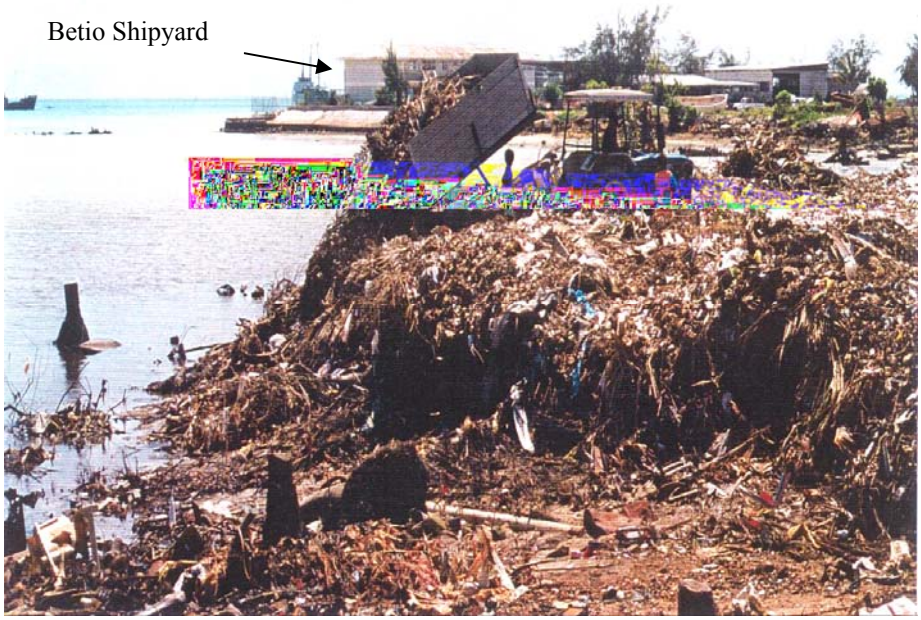


Figure 3.5: The BTC tractor emptying its trailer at the Red Beach dumping site before a seawall was built to form the Betio Landfill (after Royds (1996)).



Figure 3.6: Landfill seawall built in 1997 looking into the landfill.

NB: The two photos figure 3.5 (1996) and figure 3.6 (2005) were taken from the same angle. Note the roof of the Betio shipyard building in the background

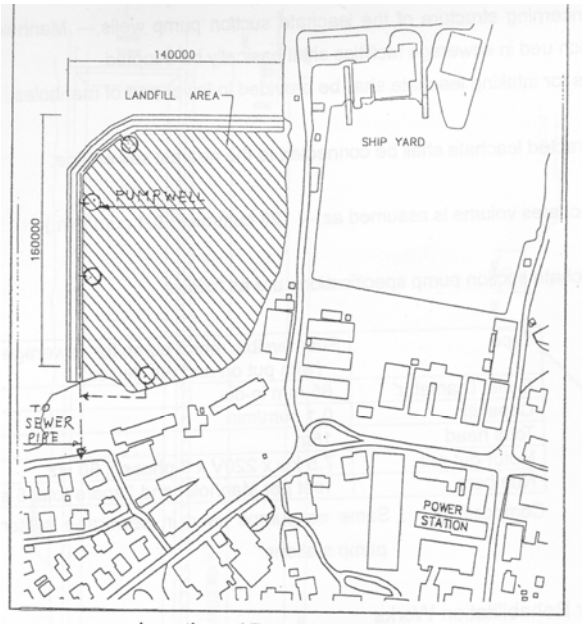
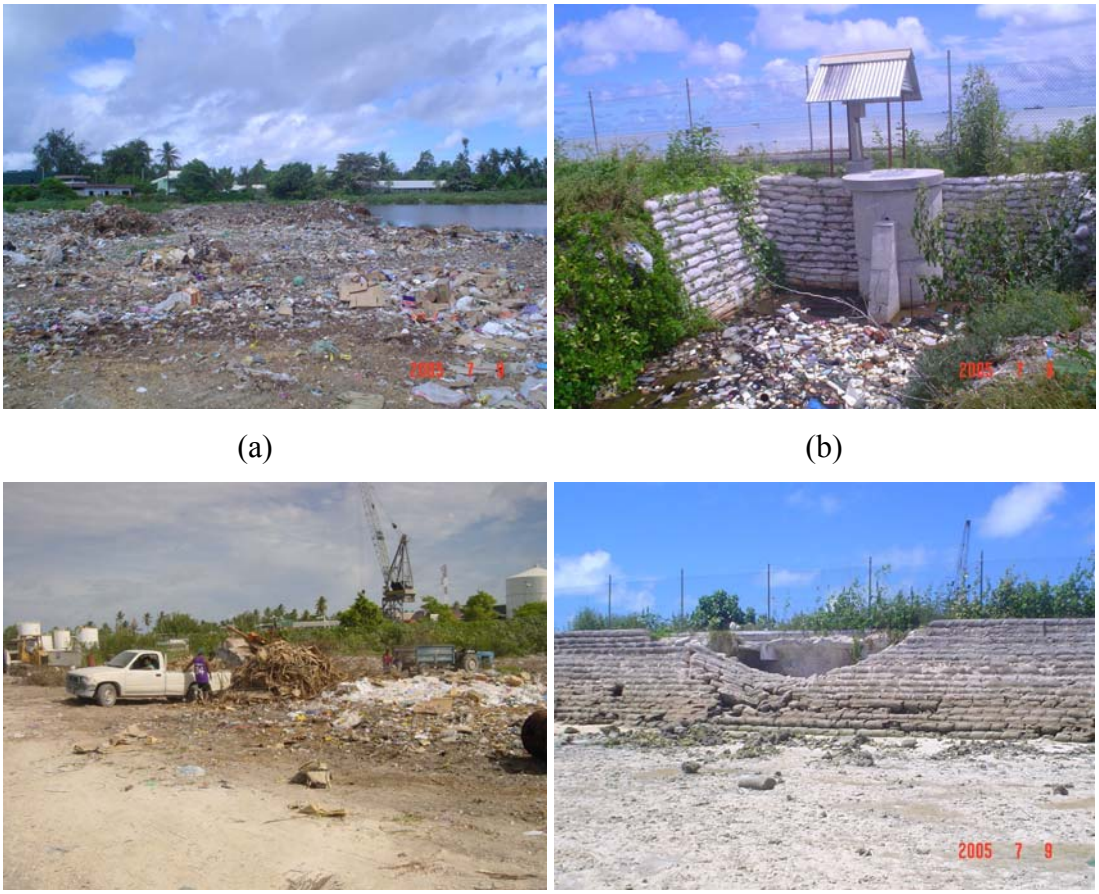


Figure 3.7: Site plan for the Betio Landfill (after Kiribati SAPHE (n.d.))

There is stagnant water inside the landfill, mainly seawater and rainwater, which had been left after the construction of the seawall while the dumping of waste is carried out from the perimeter of the site progressively towards the centre (Figure 3.8). In terms of historical leachate control there is no evidence to indicate that the Betio landfill was developed with an impermeable liner or equivalent, or possesses an engineered leachate collection system (Kiribati SAPHE Project n.d.).

Kaly (1996) a year before the landfill seawall was constructed, documented the contamination of marine communities with tannins and other leachates from water flowing through the dumpsite. The study observed an increase in blue-green algae and reduction in Entermorpha (alga) near the landfill, an indication of local nutrient pollution from the landfill. However, a 2-days observation carried out in July 2005 around the Betio landfill seawall during low tide did not find any obvious leachate seeping through the seawall into the marine environment.



(a) (b)

(c) (d)

Figure 3.8: Existing Betio Landfill;

(a) dumping is carried out from the edges of the dumpsite towards the pond of leachate in the middle; (b) one of the leachate suction pump wells recently installed still not yet up and running; (c) refuse disposal carried out by the BTC and contractors; and (d) part of landfill seawall made up of sand bag and reef mud that collapsed after the storm February 2005 posing a risk to an integrity of the landfill.

The Kiribati Sanitation, Public Health and Environmental Improvement (SAPHE) Project (Loan No. 1648 – KIR (SF)) Solid Waste Management component in 2000 recommended a number of rehabilitation works on the Betio landfill, which include dealing with the leachates issue. Some of these recommendations include installing the leachate suction pump in two places along the inner side of the seawall and one place on the inland side (Figure 3.7). The leachates volume which is estimated to be 123 m^3 per day (0.1 m^3/min) would be connected to the nearest municipal sewer line (Kiribati SAPHE Project n.d.).

There is no confirmed future land-use planning for the landfill after the site is decommissioned though it is possible that the BTC will claim the reclaimed land.

3.3 New Zealand - case study

3.3.1 Site Location

The study site for the New Zealand case study is Taipari Strand, which is situated on the western edge of the Te Atatu Peninsula. It is located north of the western motorway, Auckland and fronted entirely by the tidal estuary of Henderson Creek, which drains the adjacent catchment and intertidal flats into the upper Waitemata Harbour (Figure 3.9) (Worley 1998).



Figure 3.9: Taipari Strand, Te Atatu peninsula, Western Auckland (NZMS260 topographical map)

Taipari Strand is a relatively new region combining foreshore reserves, part of a former Waitemata City Council Depot, a reclaimed area of Henderson Creek, and the Taikata refuse tip (Leong *et al.* n.d.). It is located in a broad 1.5km arc around the Henderson Creek headland. Residential properties (RS 2 Low density) surround the Eastern boundary of Taipari Strand, Matipo School and Aquatic Clubhouses are

located towards the northern end of the area (Leong *et al.* n.d.). To the south also across the Henderson Creek the land is zoned Industrial 2 and while parts are designated for Broadcasting purposes and for Council's Solid Waste Baling and Processing Station, the balance is rapidly being developed with general industrial buildings (Worley 1995).

3.3.2 Regional Geology

Early Miocene subsidence in Northland and Auckland led to the formation of the Waitemata basin and deposition of the Waitemata Group (Edbrooke 2001) over the Northland Allochthon, Te Kuiti Group and Waipapa (composite) terrane basement. A major feature of the regional geology is the prominent cliffs, which are exposed at many sites around the Waitemata Harbour. These cliffs are made up of a sequence of well bedded, flat lying sandstones alternating with generally subordinate siltstones (the Waitemata Group) of Miocene age (Loomb 2001).

Underlying siltstone provided support to the sandstone beds. However, alternate wetting and drying of cliff faces and salt crystallization in voids causes clays to dissipate away over time (Moon & Healy 1994). This eroded geological matrix is easily removed by surface water, wind, and waves (Moon & Healy 1994) and becomes a key source of sediments present in the Waitemata Harbour (Loomb 2001).

Loomb (2001) examined surficial sediments along the Henderson Creek channel and suggested that coarse sand is present in the upper reaches of Henderson Creek as well as in the channel near the main harbour waters. The grain size trend is decreasing with distance down the Henderson Creek channel to fine sand (2.0 – 3.0 phi) in the main harbour entrance.

3.3.3 Climate

The Auckland region experiences a variable daily weather pattern largely due to a product of the passage of eastward migrating anticyclones and low-pressure troughs. These weather systems normally last for 3-5 days and the passage of each front is usually associated with an intensification of rainfalls over the large Auckland area (Hunard 1980).

The average annual rainfall over most of Auckland is 1200-1600 mm, though there is considerable inter-annual variation (Hunard 1980). The amount of rainfall also varies depending on location due to the Waitakere Ranges positioned to the west of the study site. These interfere with the prevailing westerly wind flow over Auckland, promoting an increase in rainfall in the western suburbs of Auckland, near Taipari Strand, compared to the east (Hunard 1980).

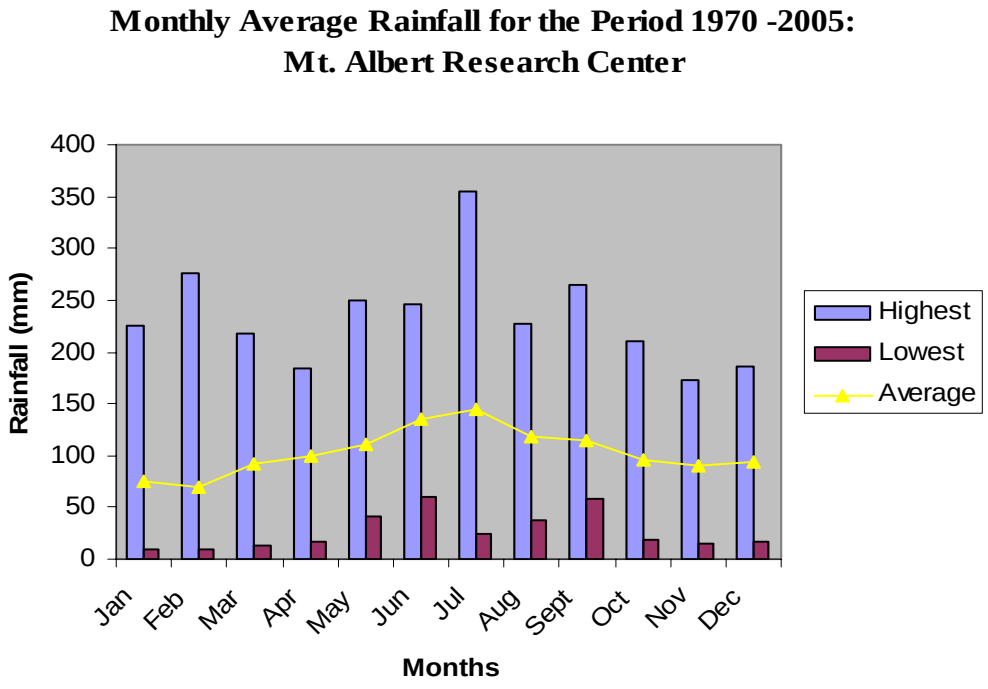


Figure 3.10: Average monthly rainfall for Auckland (1970-2005): Mt. Albert Research Center (Source: NIWA Climate Database).

Auckland long term rainfall patterns (1970-2005) suggest that the most rainfall occurs in winter and least amount falls in summer (Figure 3.10).

3.3.4 Hydrodynamics

Wind speed and direction are important mechanisms in sediment transport studies as they drive the circulation pattern of currents and determine the wave climate (Environment Consultants Kiribati 2002). In turn, waves tend to become an important agent of re-working the sediments and making it available for transport (Loomb 2001).

The Auckland region experiences a seasonally prevailing wind flow from the west or southwest (Hunard 1980). Whenuapai Airport measurements indicate westerly and southwesterly wind dominant in the summer months reaching speeds of 50m/s. Normally these westerly winds range between 5.5m/s and 10.5m/s (Hunard 1980).

Waves and currents are known mechanism in transporting sediments (Consulting Environmental Engineers 1985). The importance of waves is their orbital motion, which interacts with the bed causing sediment motion (Loomb 2001). Therefore, waves play a key role in sedimentation by suspending particles and making them available for transport by agents like water currents.

A study undertaken in 1998 to investigate the wave heights and periods that different fetch lengths and wind speeds can generate off the Te Atatu Peninsula, indicate waves approach the general Westpark Marina, west of Te Atatu Peninsula, from the north between 50º and 80º and are generated by a 10-16 knot wind (Loomb 2001). These waves are predicted to have a height of 0.11 to 0.15 metres and periods of 1.3 to 1.5 s (Loomb 2001).

Currents in the Henderson Creek channel are largely confined to the tidal currents that are continuation of flow down the Henderson Creek channel, with wind induced currents becoming important in the shallow areas the tidal flats (Loomb 2001). Ebb

currents appear to dominate the Henderson Creek channel with faster and more prolonged currents that the flood tidal flows (Loomb 2001).

In the Henderson Creek channel, the ebb tidal currents can reach speeds of 0.5 to 0.6 m/s and limited to about 0.45m/s on the flood tides. The relatively faster currents on the ebb tide are due to freshwater input from Henderson Creek (Loomb 2001).

3.3.5 Ecology

An important ecological feature of the Taipari Strand is the presence of mangrove along the coastal edge (Figure 3.11). The mangrove swamp directly abuts the strand, and pukeko, kingfishers, herons, tui, ducks, gulls and many other birds inhabit the foreshore area (Worley 1994).

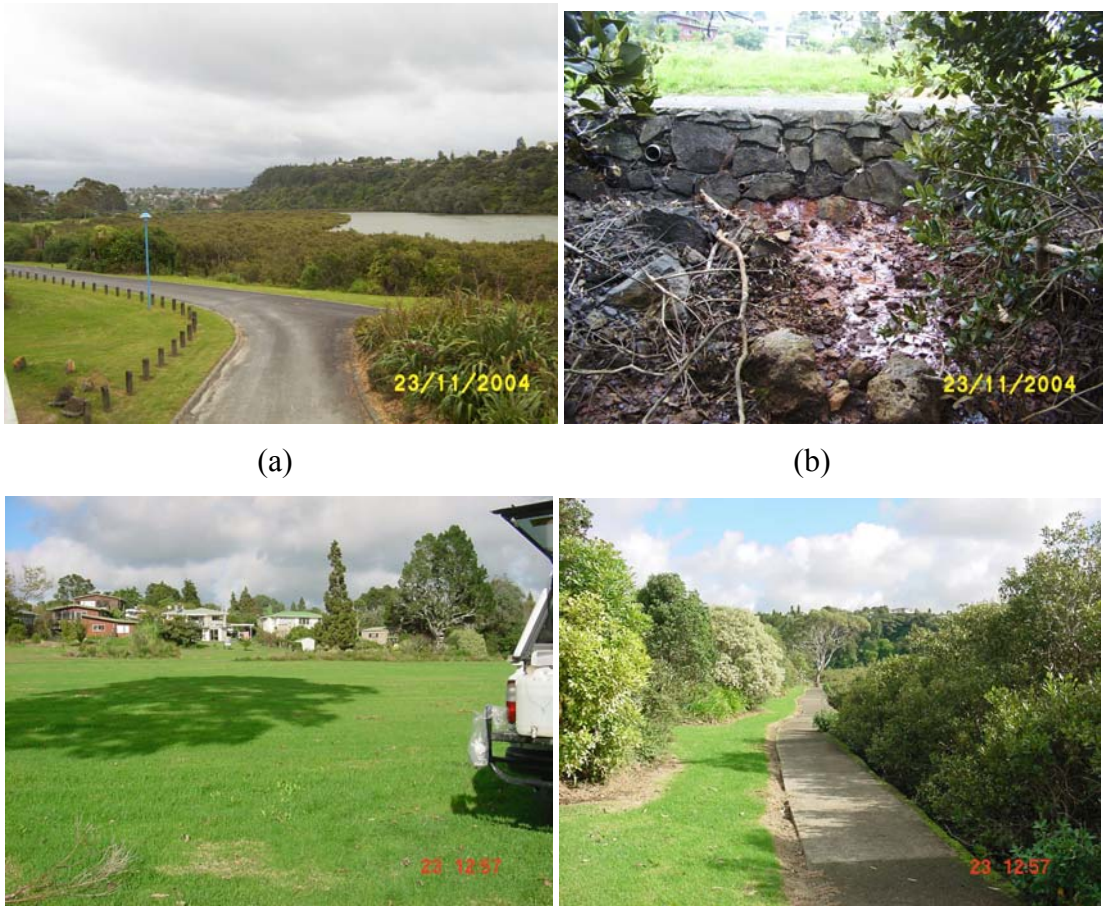
A study done in 1994 to investigate the effects of leachate on estuarine benthos in Whangateau showed that near the landfill there were low macrofauna abundances, biomass, species richness, and biodiversity (Polychaeta, Amphipoda, Oligochaeta, Decopoda, Bivalvia, and Gastropoda) (Leong *et al.* n.d.). This was assumed to be the same for the Taipari Strand region.

However, the intertidal flat sediment was observed to contain large biological components, particularly mud crabs and nematodes. These organisms may play a key role in stabilizing the sediment in the mud flats when they secrete mucus that acts as a "glue" between particles (Black 1997).

3.3.6 Background on Taipari Strand Landfill

The Taipari Strand Landfill was operated by the Waitakere City Council (WCC) as a sanitary landfill from 1963 to 1968 and 1970 to 1975 when it was closed down. During the periods of operation tipping proceeded at the rate of approximately 1 ha per year (Worley 1995). Refuse materials consisted of domestic, commercial and

industrial wastes were placed directly onto the marine sediments which underlies most of the site. The lack of a relatively impermeable clay blanket beneath the refuse allows migration of leachate into the marine sediments beneath. No groundwater or surface water protection measures were installed at the time of landfilling and capping at the site was reported to be poor in places (Worley 1995; Worley 1998).



(a) (b)

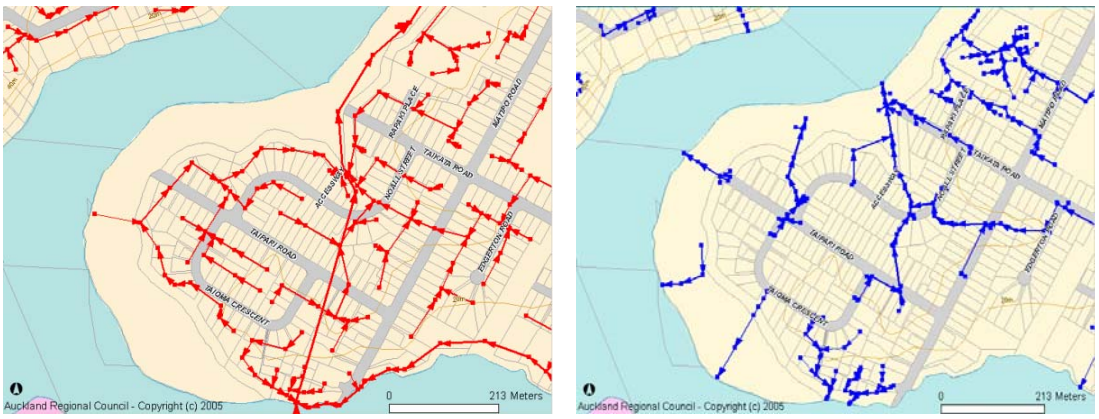
(c) (d)

Figure 3.11: Existing Taipari Strand landfill;
(a) Looking south, the mangrove swamp along the coast of the low-lying Taipari Strand landfill on the left and the Henderson Creek channel on the right; and (b) leachate seeping through the stone seawall into the mangrove swamp; (c) ground surface of the landfill site; and (d) a walking path along the edge of the stone seawall.

The Taipari Strand is approximately 1.5 km in length, covering 15 hectares (37 acres), of which 11 hectares was created by landfilling with an average depth of 2-3 metres

(Leong *et al.* n.d.; Worley 1995). Topography of the land is gently sloping (typical grades 1 in 20 to 1 in 70), though the southern end of the landfill (near Taioma Cresent) is still un-leveled and ponding is evident in some places. A stone seawall has been built along approximately two-thirds of the current foreshore. An observation along the seawall found in some parts leachate seeping through into the mangrove swamp (Figure 3.11). The low-lying nature of the site makes it highly visible to surrounding residences on Te Atatu Peninsula and also parts of Massey across the estuary.

The Waitakere City Council was issued with two resource consents by the Auckland Regional Council to address the issues of historical landfilling activities at the site and the effects of this activity on the environment. The first resource consent (Consent No. 16342) permits the discharge of leachate into the ground and groundwater beneath the site and the diversion of leachate and groundwater into the collection system (Worley 1998).



(a) (b)

Figure 3.12: Taipari Strand Landfill drainage site plan;
(a) landfill leachate collection system connected to the local municipal sewer network; and (b) stormwater lines and outfalls location in the local area (Source: Auckland Regional Council).

Rehabilitation works have already taken place and completed in 1998 to mitigate the discharge of contaminants from the site to surrounding environment which includes

recapping the site and rehabilitation of the stormwater system to eliminate leachate ingress and retrofitting a leachate trench on the landfills western margin. Part of this remedial work includes installing the leachate pump which pumps generated leachate from the landfill into the municipal sewer line network (Figure 3.10) (Worley 1998).

The Taipari Strand is currently vested as a reserve and is zoned Recreation 1 (passive recreation) in the Waitakere City Council District plan. Proposed development for the site includes an educational and visual boardwalk through the mangroves, further tracks for jogging and walking, picnic tables and barbecues, a playground structure (Figure 3.11) (Worley 1994).

Chapter Four: Methodology

4.1 Introduction

This chapter describes the sampling methodology undertaken for this study including sample treatment and storage; and the laboratory procedure for the physical and chemical analysis. Different sampling strategies were used in collecting samples and measuring the pH at the two study sites due to their different sediment types.

The laboratory analysis characterizes the sediment physical and chemical properties, including organic matter, particle size distribution, and the heavy metal content. Organic carbon and particle size aid an understanding of the movement and distribution of the heavy metals. The chapter will start with the Kiribati case study, which includes sampling, pH measurement, particle size analysis and total organic matter analysis, followed by the New Zealand case study, and finally the heavy metal analysis.

4.2 Kiribati study site (medium coral sand)

4.2.1 Sampling method

A total of 96 samples were collected from the site during low tide in July 2005. Sampling was carried out using a geometric sample spacing along transects of; 0 (A), 2 (B), 5 (C), 10 (D), 20 (E) and 40 m (F) starting from the landfill seawall. There were 4 transects with 6 sampling points, 3 from the landfill and 1 at the control site, which was 2 km to the east (Fig. 4.1). The locations of the sampling points for both the landfill and control site transects were determined using GPS (Garmin E-Trex, Appendix 1).

At each sampling point of a transect 4 samples were collected at different depths i.e. surface (1), 5 (2), 10 (3) and 15 cm (4) depths. At the landfill site, the transects were randomly placed in the intertidal zone in an arc around the semicircular landfill seawall approximately 100 m apart (Figure 4.2). However, at the control site, the transect was placed 65 metres from the shoreline, perpendicular to the beach towards the lagoon (Figure 4.1), in a similar sediment environment to the landfill transects.

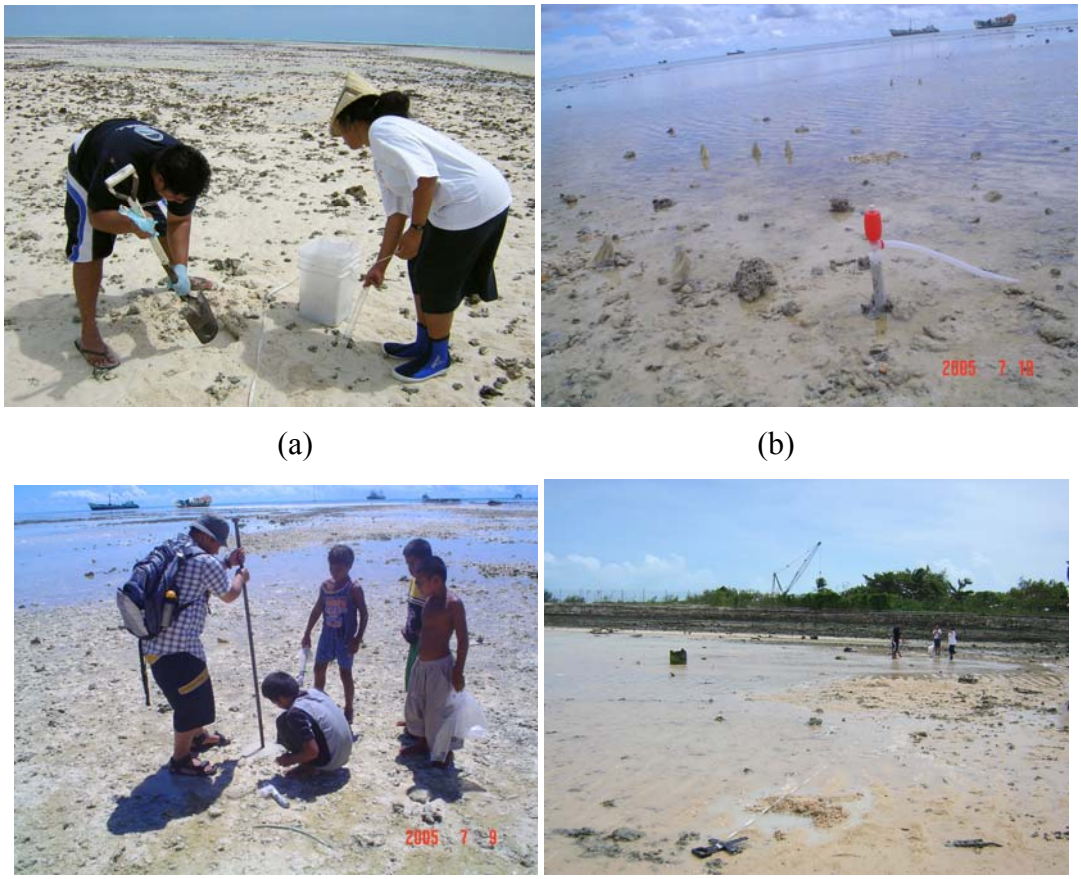


Figure 4.1: Betio islet showing the landfill and control site transects.
T1 – Transect 1, T2 – Transect 2, T3 – Transect 3 and TC – Transect Control.

Sediment samples were obtained by hand, with the help of a spade digging the loosely consolidated reef mud surface (Figure 4.2). A plastic ruler was used to measure the depth of the profile. A handful of sediment (approximately 120 g) was collected from the 4 different depths in each sampling point and placed into labeled plastic zip-lock bags. The samples were drained of water, dried under the sun for 1 day, and then sent back by DHL courier to the University of Waikato, New Zealand and stored at the MAF registered 'transition facility' until they were analyzed.

4.2.2 pH measurement

Piezometers of different lengths 5, 10 and 15 cm were used for the field measurement of the pH. The piezometers were made with 10 cm long above the surface and had small holes at the bottom covered with fine gauze (Figure 4.2). During low tide, holes were made into the mudflat using a crowbar to a specific depth matching the length of the piezometers (Figure 4.2).



(a) (b)

(c) (d)

Figure 4.2: Sample collection at Betio study site;
(a) digging the loosely consolidated reef mud during sediment sample collection at the control site; (b) installed piezometers during low tide with the hand pump used to collect water samples for pH measurement; (c) making holes into the mudflat using a crowbar for the piezometers and; (d) laying a transect on the landfill foreshore during low tide.

The top of the piezometers were covered with fine gauze secured with a rubber band, except when sampling, to prevent sediment and organic matter but only water to entering when the tide comes in. The piezometers were installed and left overnight to let the water stabilize before taking samples.

Water samples were collected into 100 ml plastic vials with a hand pump after draining the piezometer twice to ensure that interstitial water was sampled (Figure 4.2). The sample pH was determined using a pH meter (YSI Model 63). The pH meter was calibrated using a 2-point calibration (pH 7 and 4) according to the manual for the YSI Model 63, before measuring the water samples on different days.

4.2.3 Particle size analysis

The samples were air dried in an oven (45º C) for 3 days, and then put through a splitter to equally divide and obtain a good representative of the samples for particle size analysis and organic matter content, separate from the sample used for heavy metal analysis to avoid cross contamination by the sieves. The 2 replicates of each sample were then placed into labeled plastic zip-lock bags ready for physical and chemical analysis.

Particle size analysis was undertaken by sieving through 5 sieves with mesh sizes of -2.5 Ø (5.60mm), -2.0 Ø (4.00mm), -1.5 Ø (2.80mm), -1 Ø (2.00mm) and 0 Ø (1mm). Approximately 50 g of air-dried sediment were gently crushed to break up dessicated clumps. The samples were then sieved through the 5 sieves manually for 10 min. The weights for each fraction were weighed and recorded. After sieving each sample, the sieved fractions were then added back together for the total organic matter analysis.

The sediment samples were analyzed in terms of medium gravel and coarser (> 5.60 mm), fine gravel (<5.60 to > 2.00 mm), very coarse sand (<2.00 to >1.00 mm) and sand, silt and clay (<1.00 mm) according to Milne *et al.* (1991).

4.2.4 Total organic matter

Total organic matter content was analyzed by the Loss on Ignition (LOI) method according to Blakemore et al. (1987). Dry and clean crucibles were weighed and the weights were recorded – this is CRUCIBLE WEIGHT. Approximately 5 g of samples were added and placed in an oven at 105º C overnight to dry. The samples were then removed and placed in a dessicator for 1 hour to cool down before the weights were recorded – this is DRY WEIGHT. The samples were then placed in the furnace at 375º C for 16 hours (Ball 1964). The samples were then removed and placed in a dessicator for 1 hr to cool down. The samples were weighed and the weights were recorded – this is ASHED WEIGHT.

The crucibles were soaked with Decon 90™ overnight, and thoroughly cleaned and dried in a drying oven before reuse.

Calculation: i) % Ash Residue = ((ASHED WEIGHT – CRUCIBLE WEIGHT) / (DRY WEIGHT – CRUCIBLE WEIGHT)) x 100

ii) % Organic Matter = 100 - % Ash Residue

All the remaining Kiribati samples left after the physical and chemical analysis were stored at the MAF registered ‘transition facility’ for proper disposal.

4.3: New Zealand – study site (muddy fine sand)

4.3.1: Sampling method

A total of 23 short marine surficial sediment cores (top 15cm) were collected from the site during low tide in November 2004. Samples were obtained to a depth of 20 cm. Each core was obtained with a PVC corer of 30 cm length and 11 cm in diameter, pushed into the sediment by hand, plugged and pulled out. There were 4 transects with 6 sampling points at 5 m intervals (0 (Transect base), 5 (A), 10 (B), 15 (C), 20 (D), 25 (E) and 30 metres (F)) except for Transect 3. There were 3 transects from the landfill and 1 control site, which was approximately 2 km north of the landfill (Figure 4.3). The 3 transects at the landfill site were placed away from the stormwater outfalls (Figure 3.12) but close to the places where leachates were found seeping through the stone seawall into the mangrove swamp approximately 200 m apart (Figure 4.2).



Figure 4.3: Te Atatu peninsula and the location of the landfill and control transects. T1 – Transect 1, T2 – Transect 2, T3 – Transect 3 and TC – Transect Control.

The cores were carefully taken out and placed in a plastic bag in a way they were not disturbed and broken up. The core samples were kept in a labeled plastic bag, stored

on ice and transported to the University of Waikato and stored in a freezer at 0 - (-5)º C until they were analyzed for physical and chemical properties.

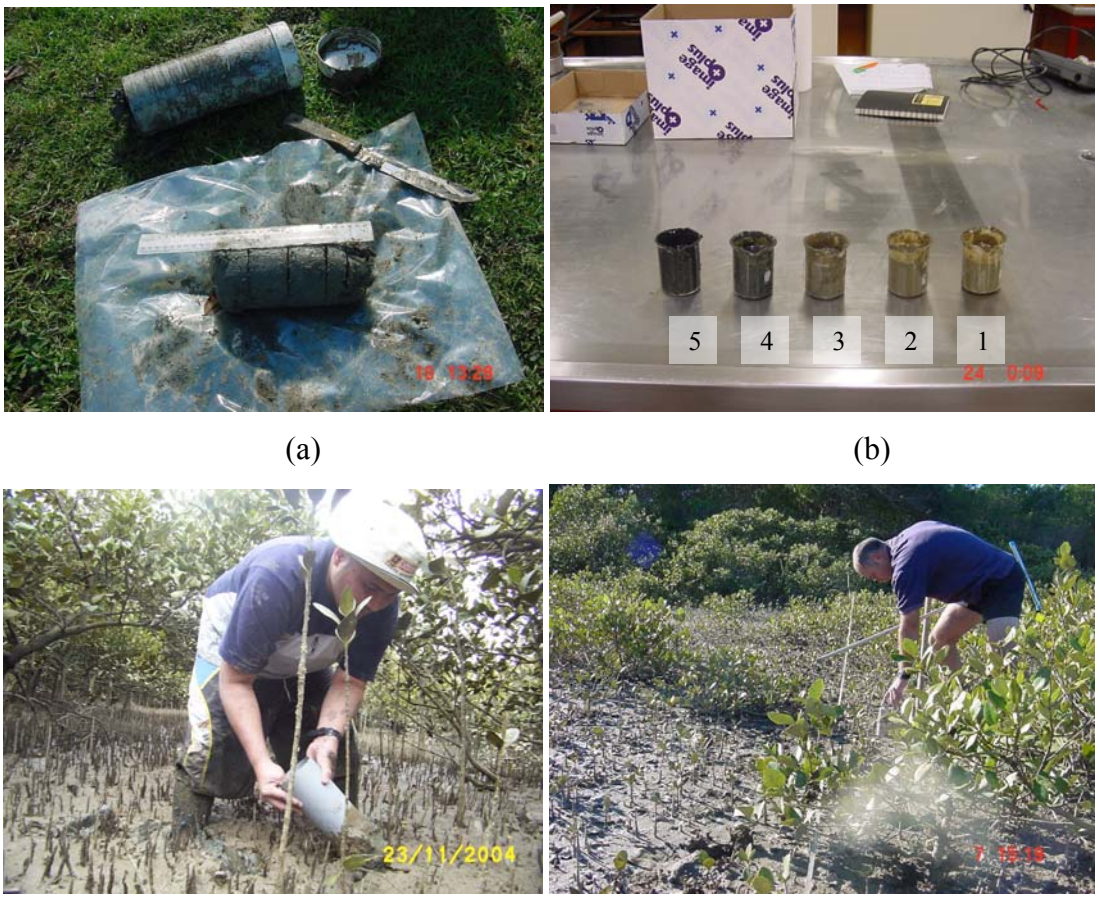
In between sampling each point, the corer was washed in three buckets, containing tap water and Decon 90™, plain tap water, and distilled water respectively. The water in each bucket was changed between samples. Blue nitrile gloves were worn throughout sampling and changed between samples. A brush was also used to ensure that all the mud was removed from the corer during cleaning.

The location of the transect bases and all the sampling points for both the landfill and control sites were recorded using GPS (Garmin E-Trex, Appendix 1).

4.3.2: pH measurement

The cores from the landfill and control sites were stored on ice and transported to the University of Waikato, Earth Sciences Soil Lab, for measuring pH, which was undertaken on the same day as the sediment sampling.

There were 5 pH measurements carried out for each core i.e. the surface, 2.5, 5, 10 and 15 cm depth (Figure 4.4). Distilled water was used with a ratio of soil to water of 1:2.5 (Hodges, *pers comm*). Approximately 10 g of the wet sample was placed in a 100 ml beaker and 25 ml of distilled water was added. The samples were stirred thoroughly using a glass rod for 20 s and left to stand for 3 mins. Sample pH was measured using a pH meter (Mettler Toledo).



(a) (b)

Figure 4.4: Preparing samples for pH measurement;

(a) a core marked at different depths to extract samples for pH measurement; (b) samples prepared in the soil lab ready for measuring pH (1 – surface, 2 – 2.5, 3 – 5, 4 – 10, and 5 – 15 cm depth); (c) a PVC corer used to obtain sediment samples; and (d) placing a transect in the mangrove muddy sediment.

4.3.3 Particle size analysis

Sediment samples were analyzed using a Malvern 'Mastersizer-S' S long bed Version 2.14 laser particle sizer that calculates particle size distribution using laser diffraction. The operation of this instrument is based on the principle that particles of a certain size diffracts light through a given angle, with the angle increasing as particle size decreases (Cooke 1998). On passing through a suspension of a sample, a narrow beam of monochromatic light (the laser) is diffracted onto a multi-element ring

detector which analyses the angular distribution of scattered light intensity (Figure 4.5)

The sediment cores were cut into 6 equal layers (15 divided by 2.5 cm). 0-2.5 (1), 2.5-5 (2), 5-7.5 (3), 7.5-10 (4), 10-12.5 (5) and 12.5-15 cm (6) using a stainless steel knife. Each layer was then divided into 3 and analyzed for particle size, organic matter and heavy metals. The 3 sub-samples for each layer were analyzed for particle size and organic matter and the results were averaged and recorded. The knife was soaked with Decon 90™ and thoroughly cleaned and dried between cutting every layer.

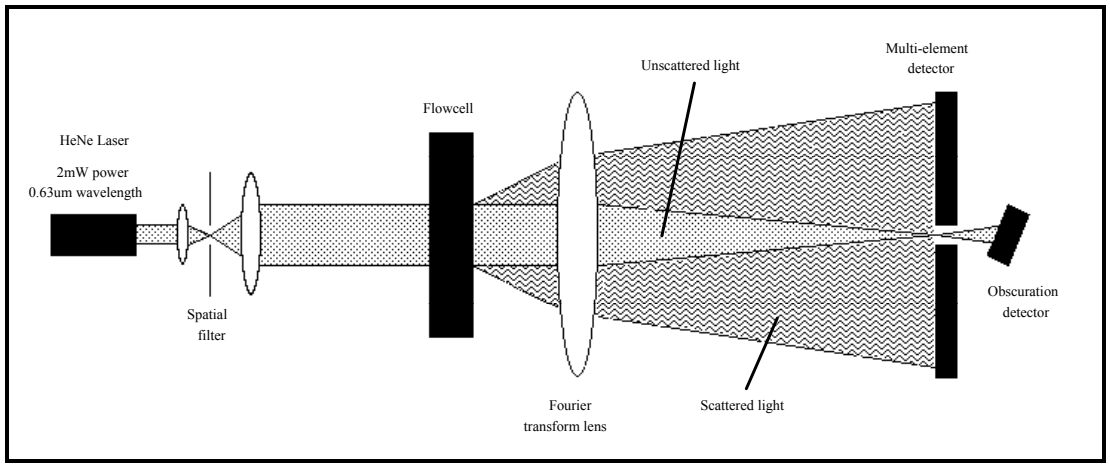


Figure 4.5: Schematic representation of the Laser Particle Sizer instrument (after Cooke 1998)

Pretreatment of samples was carried out in order to remove organic matter and to break up any aggregates according to the method of Cooke (1998). This was achieved by taking 5g of the wet samples and placing it in a 100 ml glass beaker, and adding 10 ml of 10% hydrogen peroxide to remove organic matter and break up aggregates. The samples were left overnight in the fume-hood before being heated on a hotplate to remove excess hydrogen peroxide and water formed from the reaction of hydrogen peroxide and organic matter. The samples were stirred thoroughly with a glass rod for 30 s every 5-6 hrs to help break up aggregates. Another 5 ml of 10% hydrogen hydroxide was added to the sample to ensure the samples were completely covered

with hydrogen peroxide and heated on a hotplate if necessary until all the effervescences subsided (Hodges *per. comm.*).

Approximately 1ml of sample was placed in the lasersizer and analyzed in terms of sand (2000 – 63 μm), silt (63 – 2 μm) and clay (<2 μm) according to the Wentworth classification of sediment grain sizes.

4.3.4 Total organic matter

After the 15 cm cores were cut into 6 equal layers and each layer divided into 3, the samples were placed in an air dried oven at 38º C for at least three days. The samples were then hand ground using the crucible and grinder and placed into labeled plastic zip-lock bags ready for organic matter and heavy metal analysis.

The organic matter content was determined by the Loss on Ignition (LOI) method similar to the one carried out for the Kiribati samples, except the samples were placed in the furnace of 500º C for at least 4.5 hours (Ballinger 2003).

4.4: Heavy metals analysis

Heavy metal (Cu, Pb, Zn, Cd, Ni, Cr, Hg, As) concentrations for all samples for both Kiribati and New Zealand were determined by EDXRF (Energy Dispersive X-Ray Fluorescence) using a SPECTRO X-LAB 2000 instrument, calibrated against international rock standards.

After oven drying at 110º C for at least 24 hours, approximately 40g of samples were ground to a fine powder using a ring mill to allow the samples to be homogenized before chemical analysis. 5 g of samples were hydraulically pressed into pellets, using a hydraulic compressor for insertion into the instruments and placed in an oven at 70º C overnight. The samples were then removed and placed in a dessicator for 1 hour to cool down before the weights were recorded and run through the XRF instrument overnight.

Chapter Five: Results & Discussion

5.1 Introduction

This chapter discusses the results of the Kiribati and the New Zealand case studies. The Kiribati study site comprised medium coral sand, and the New Zealand site had muddy fine sand.

The Kiribati case study investigates the heavy metal contamination in the coastal marine sediments in terms of the total metal content at a landfill currently in operation. While the New Zealand case study involves a comparative analysis of metal contamination in the marine sediments at a coastal landfill decommissioned in the mid 1970s.

5.2 Kiribati case study

5.2.1 Sample description

A visual characterization of the air-dried sediment samples was used in determining the composition and color of the sediments. The sediments composition appeared to be mainly of biogenic carbonates derived directly from the reef platform, reflecting the geological formation of the island. They are poorly sorted and made up of calcium carbonate resulting from bio-erosion and physical abrasion of reef carbonates. The sediments (e.g. pebbles) appeared to be fresh and not greatly abraded. The main sediment compositions are; coral detritus, shell (bivalves, gastropods) and fines of calcium carbonate (muds) which may possibly contain halimeda and foraminifera.

The dominant sediment composition in the Tarawa lagoon are corals, mollusks, halimeda, coralline algae and minor foraminifera (Smith & Biribo 1995).

The color of sediments which is primarily derived from the reef carbonate reflects the pale white of pure calcium carbonate minerals, calcite and aragonite (Denton *et al.* 2001).

5.2.2 pH

The pH of the waters sampled from the Kiribati study site was quite high, ranging between 7.14 and 8.85, which is expected in coral atoll sediments (Figure 5.1, Appendix 2). The trend indicates a relatively lower pH closer to the landfill seawall and slightly increasing towards the lagoon in transects 1 and 2, which were closer to the land. However, transect 3 and the control transect hardly show any change in the surface pH along the transects. The surface pH was higher than 8 which was close to seawater pH (8.98) except for sampling point A in transect 1.

All the 4 transects show the pH down the profile decreasing as it is getting deeper into the sediment which is away from seawater. There was a decrease in pH in the top 5 cm but below it, there was hardly any change. This can be seen in all transects except in transect 3, which shows a constant decrease in pH throughout the profile.

pH is one of the important parameters controlling the mobility and retention of heavy metals in soils. Retention of heavy metals in soils is directly or indirectly affected by the pH. Precipitation and dissolution phenomena of heavy metal-bearing minerals are examples of direct effects. Adsorption sites and reactions also pH dependent. Therefore, element retention in soils will also be affected indirectly by changes in pH, e.g. hydrolysed metal species ($MeOH^+$) are preferential adsorbed compared to free metal ions (Herreweghe *et al.* 2002). The high pH for the Kiribati sediment means it has the potential to retain metals in sediments making metals become immobile.

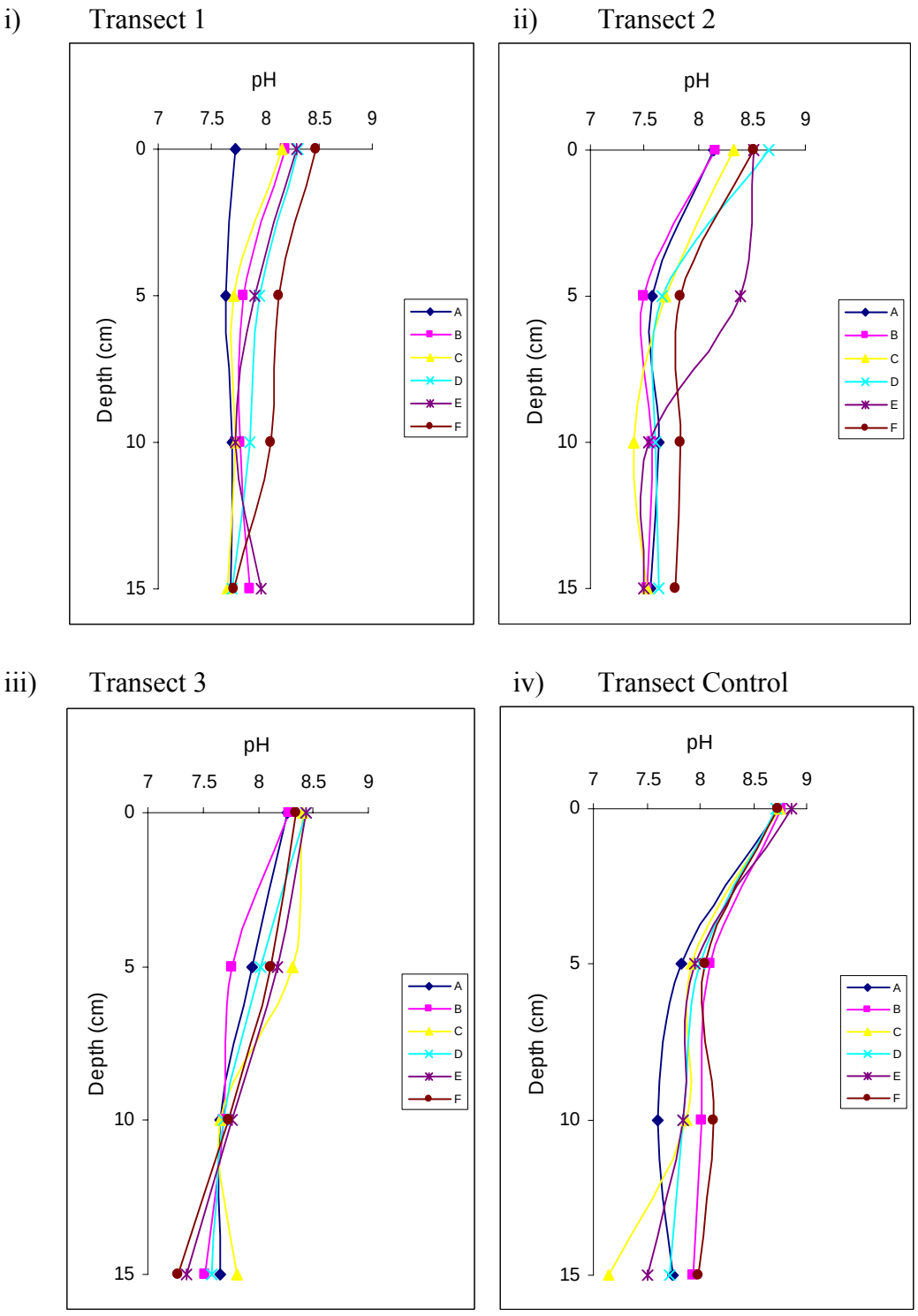


Figure 5.1: Groundwater pH vs. depth for the Kiribati study site.

Graph points A to F are positions along the transect, where A = 0, B = 2.5, C = 5, D = 10, E = 20, F = 40 m along the transect, starting with A at the seawall.

5.2.3 Particle size distribution

The study site is a good example of a transportation zone which is characterized by diverse sediments ranging from sand, silt and clay (<1 mm) to sand and gravels coarser than 5.6 mm (Appendix 3). Figures 5.2 to 5.5 present the sediment size distribution along the transects and down the profile. For all the transects, the bulk of the sediments appear to be made up of finer sediments (sand, silt and clay) which made up at least 40 % and over 80 % of some samples.

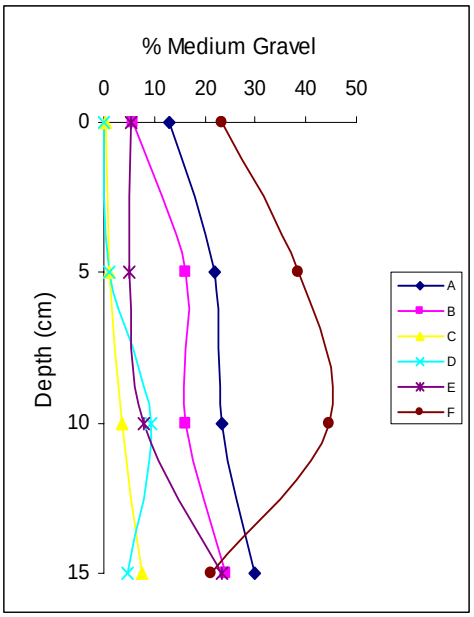
There was a small difference between the landfill and control transects in terms of sediment size distribution. The 3 landfill transects generally show a similar particle size distribution with low finer sediments closer to the landfill seawall and increasing away from the seawall.

In addition, the finer sediments component is high in the surface and tends to become decrease down the profile accompanied by the increase in coarser sediments fraction (sand and gravel). The control shows a similar trend, though the finer sediment component is higher making up at least 65 % to more than 80 % and coarser sediments become low as less than 5 %.

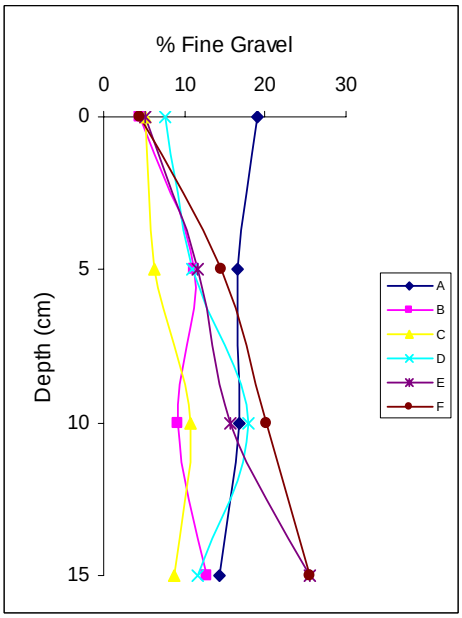
The slightly high accumulation of finer sediments at the control site compared to that of the landfill site can be attributed to their positions. The landfill is positioned to the west, making it exposed to the incoming tidal currents (i.e. high local hydrodynamic energy) and therefore fine sediments are 'moved on' (Williamson *et al.* 2003) whereas the control site is located to the east of the Betio Ports, sheltering the area hence becoming a favorable area of deposition for finer sediments.

Small sediment grains generally have higher sorptive surface than coarse sediments and as a result, they are important as they serve as major repositories for heavy metals compared to larger grained materials (Denton *et al.* 2001).

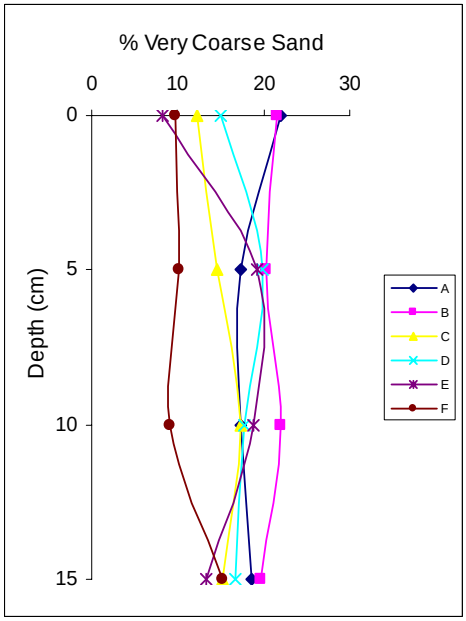
i) Medium gravel



ii) Fine gravel



ii) Very coarse sand



iv) Sand, silt and clay

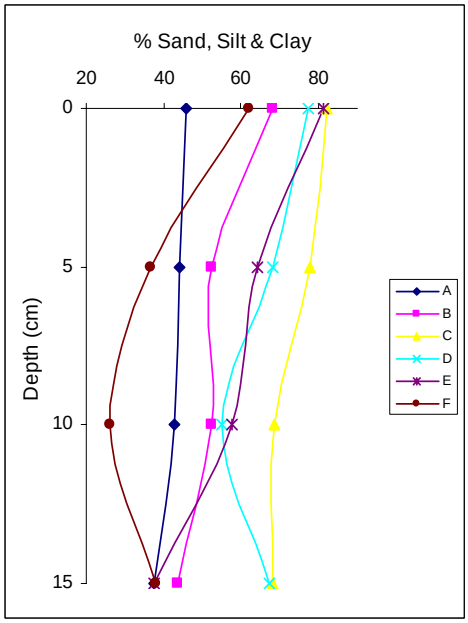
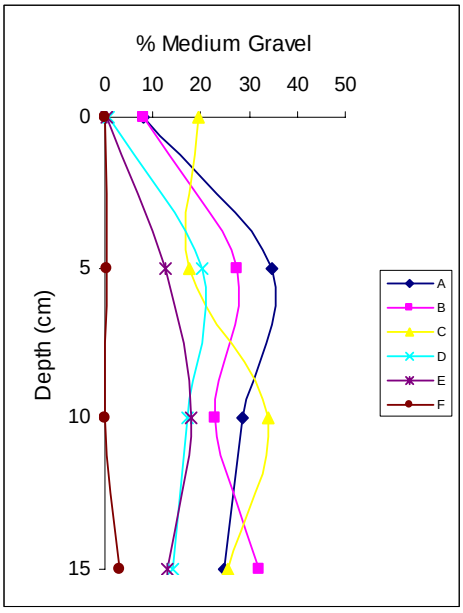
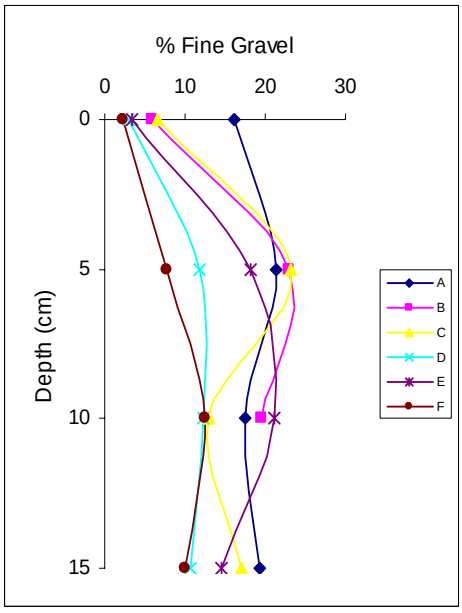


Figure 5.2: Particle size distribution vs. depth for the Kiribati study site;
i) medium gravel and coarser (>5.60 mm); ii) fine gravel (<5.60 and >2.00 mm); iii) very coarse sand (<2.00 and > 1.00 mm); and iv) sand, silt and clay (<1.00 mm) for Transect 1. Graph points A to F are positions along the transect, where A = 0, B = 2.5, C = 5, D = 10, E = 20, F = 40 m along the transect, starting with A at the seawall.

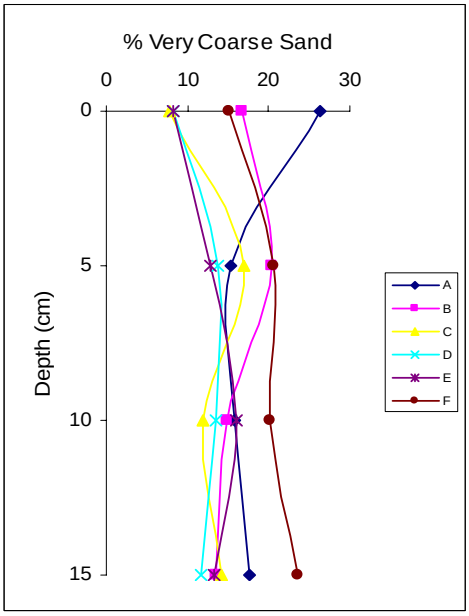
i) Medium gravel



ii) Fine gravel



ii) Very coarse sand



iv) Sand, silt and clay

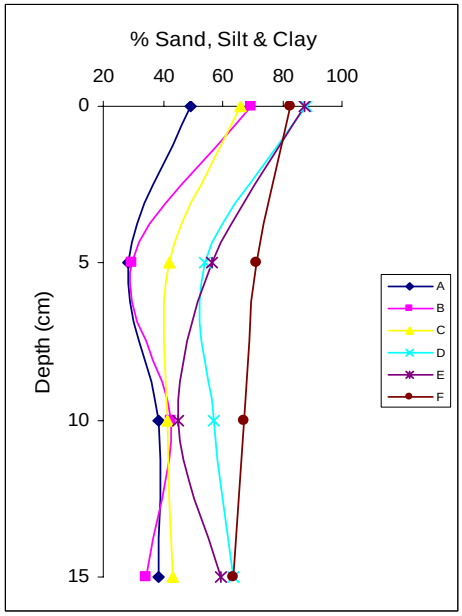
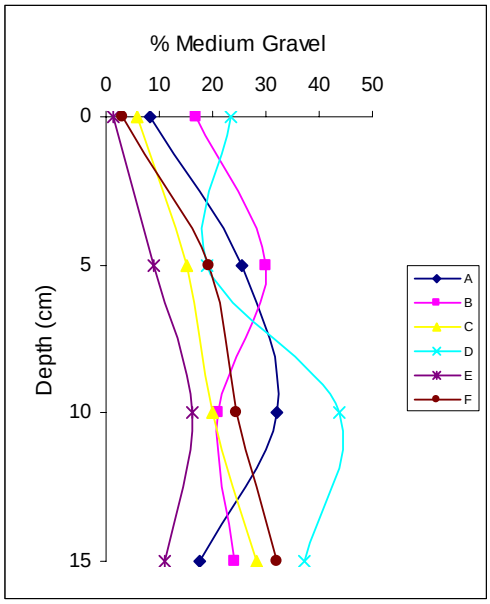
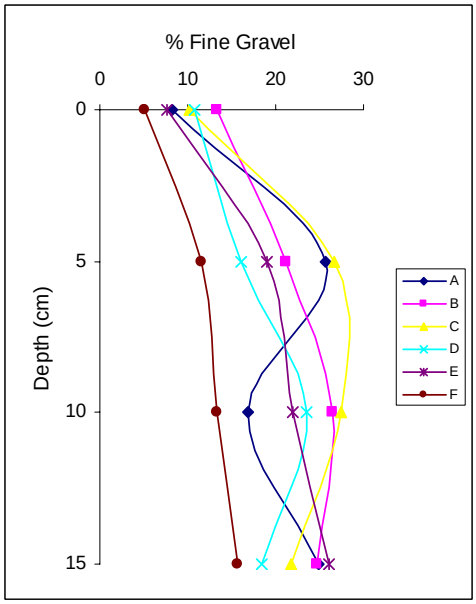


Figure 5.3: Particle size distribution vs. depth for the Kiribati study site;
i) medium gravel and coarser (>5.60 mm); ii) fine gravel (<5.60 and >2.00 mm); iii) very coarse sand (<2.00 and > 1.00 mm); and iv) sand, silt and clay (<1.00 mm) for Transect 2. Graph points A to F are positions along the transect, where A = 0, B = 2.5, C = 5, D = 10, E = 20, F = 40 m along the transect, starting with A at the seawall.

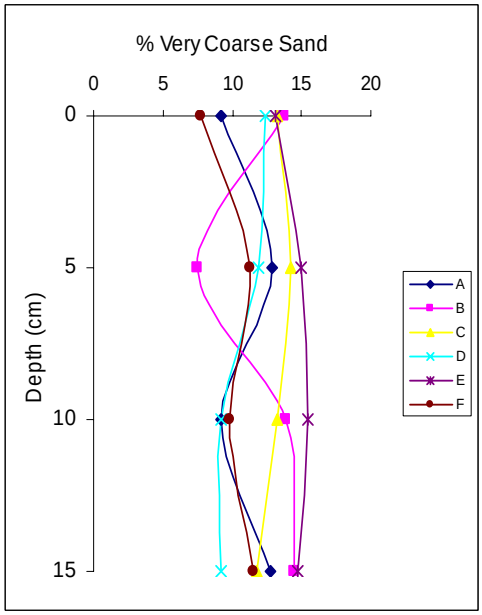
i) Medium gravel



ii) Fine gravel



ii) Very coarse sand



iv) Sand, silt and clay

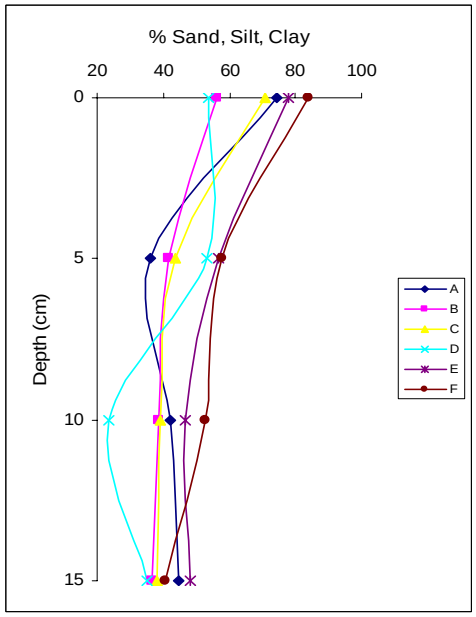
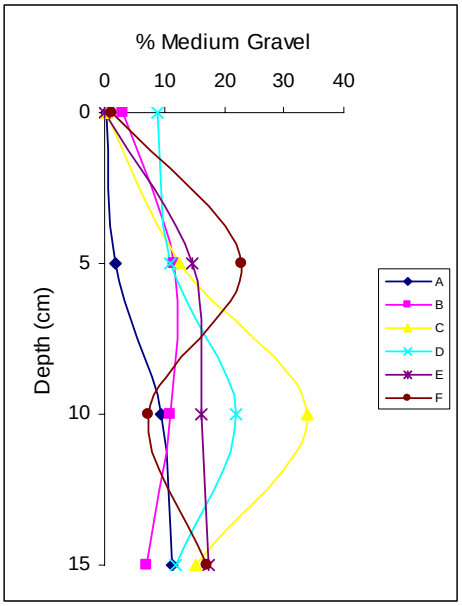
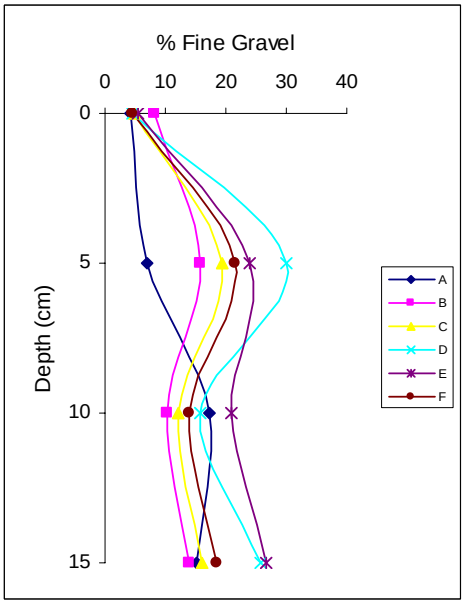


Figure 5.4: Particle size distribution vs. depth for the Kiribati study site;
i) medium gravel and coarser (>5.60 mm); ii) fine gravel (<5.60 and >2.00 mm); iii) very coarse sand (<2.00 and > 1.00 mm); and iv) sand, silt and clay (<1.00 mm) for Transect 3. Graph points A to F are positions along the transect, where A = 0, B = 2.5, C = 5, D = 10, E = 20, F = 40 m along the transect, starting with A at the seawall.

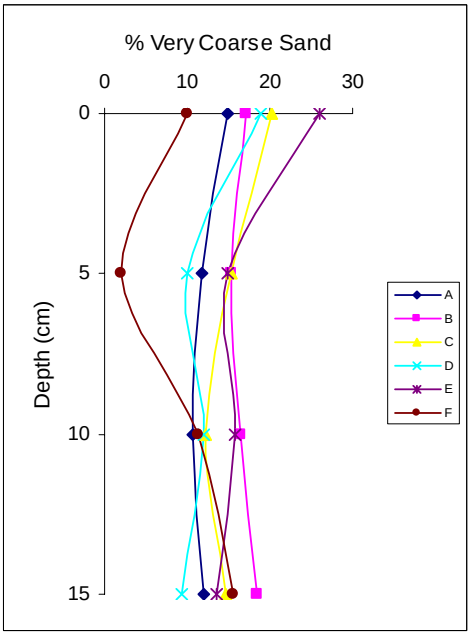
i) Medium gravel



ii) Fine gravel



iii) Very coarse sand



iv) Sand, silt and clay

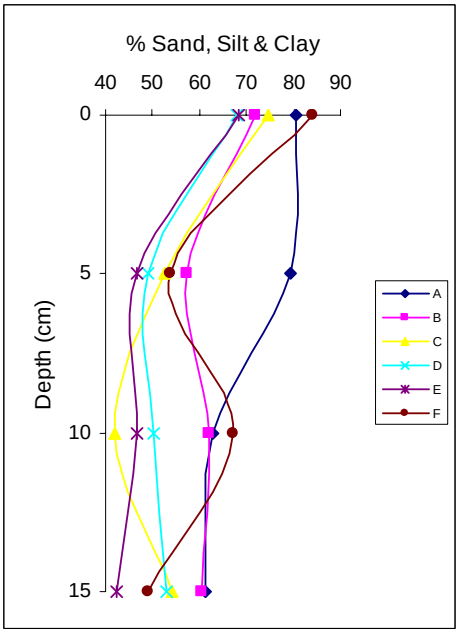


Figure 5.5: Particle size distribution vs. depth for the Kiribati study site;
i) medium gravel and coarser (>5.60 mm); ii) fine gravel (<5.60 and >2.00 mm); iii) very coarse sand (<2.00 and > 1.00 mm); and iv) sand, silt and clay (<1.00 mm) for Transect Control. Graph points A to F are positions along the transect, where A = 0, B = 2.5, C = 5, D = 10, E = 20, F = 40 m along the transect, starting with A at the seawall.

Also at the control, there was no clear pattern of sediment distribution away from the shoreline progressively towards the lagoon. Betio has an extensive mud flat (< 200 m) and using a longer transect may have shown the trend in sediment size distribution at the control site.

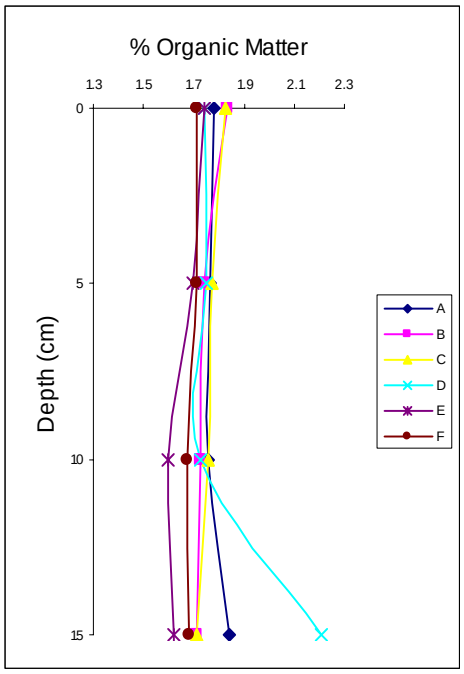
5.2.4 Percentage organic matter

The percentage organic matter in the sediments was very low, ranging between 1.60 and 2.21 % (figure 5.6, Appendix 4). The organic matter content especially for the landfill transects appeared to slightly decrease down the profile except a spike in transect 1, which represent the highest organic matter content. There was no explanation for this. There was little variability within or between transects.

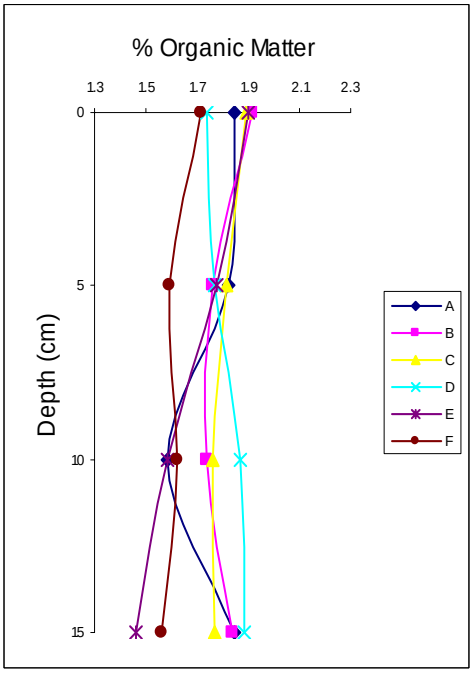
Generally, the organic matter content of sediment increased as the sediment texture become finer. There was a weak correlation between sediment texture and organic matter content reflected in the results. Organic matter has a net abundance of negatively charged sites that attract and bond with heavy metal hence sediments containing high organic matter tend to retain higher concentrations of heavy metals than those with low organic matter content. A summary table presents the results for the particle size analysis, pH and percentage organic matter (Appendix 5).

The extensive foreshore area has patches of sea-grass beds especially towards the lagoon deeper edge. This may be the major contributor of organic carbon to the nearshore environment.

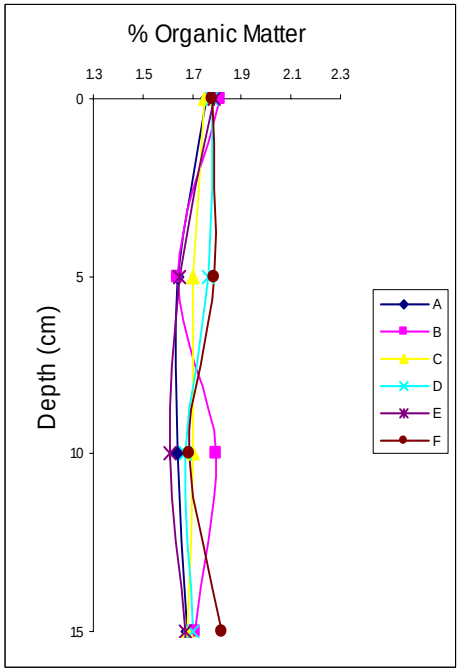
i) Transect 1



ii) Transect 2



iii) Transect 3



iv) Transect Control

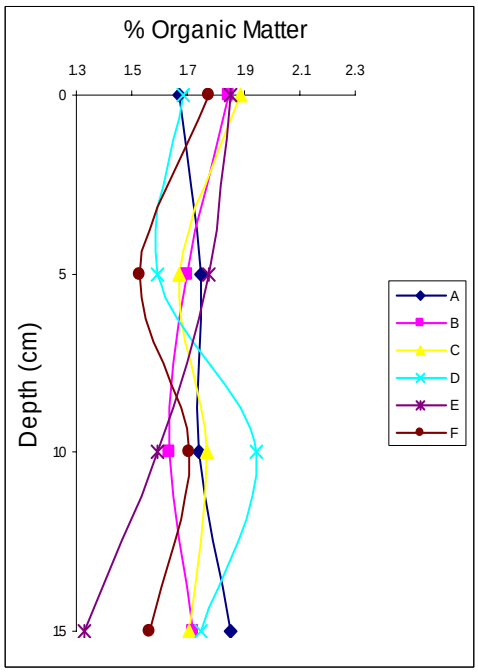


Figure 5.6: Percentage organic matter vs. depth for the Kiribati study site.

Graph points A to F are positions along the transect, where A = 0, B = 2.5, C = 5, D = 10, E = 20, F = 40 m along the transect, starting with A at the seawall.

5.2.5 Total metal concentration

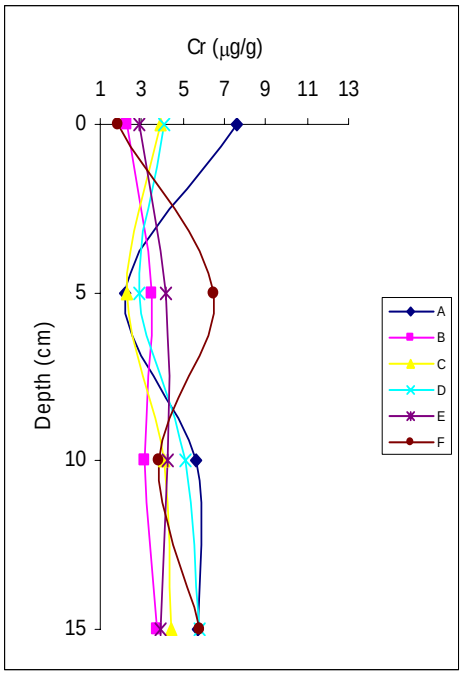
5.2.5.1 Chromium (Cr)

The chromium concentration varied between (0.9 and 12.6 µg/g), where most of the landfill transect sediments generally contained 3-6 µg/g chromium and the control transect had 1-4 µg/g chromium, indicating a higher level of chromium at the landfill compared to the control site (Appendix 6). Transects 1 and 2 had a trend of high levels of chromium at the landfill seawall, decreasing away from the landfill, indicating the landfill as a possible source of chromium. There was a spike of chromium in transect 3 and there is no apparent explanation for it. There was no clear pattern of chromium level changes down the profile but transects 1 and 4 show a slight increase in concentration down the profile (Figure 5.7).

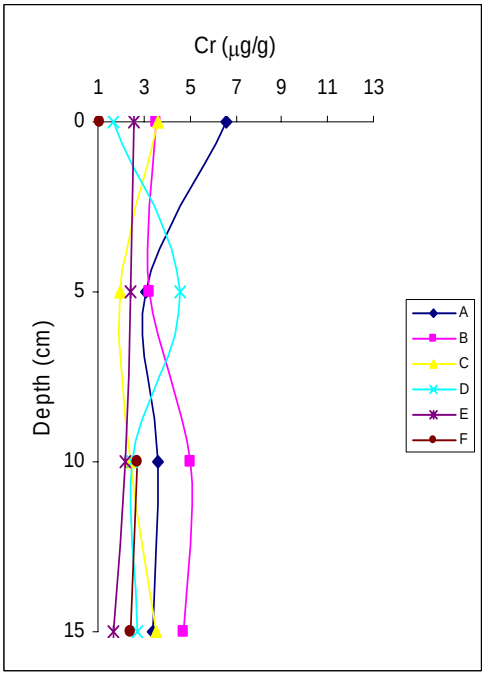
Comparing the concentrations of chromium in this study with the sediment quality guidelines (SQGs) (Table 5.1), chromium concentrations were well below the Effects Range-Low and Threshold Effects Level for the NOAA National Status and Trends Program and the Florida Department of Environmental Protection SQGs respectively, which indicates there is no potential adverse ecological effects on the benthic communities.

However, chromium levels at the landfill transects did exceed the suggested background levels for clean reef sediments, whereas the control site was well within the expected background ranges (Table 5.2).

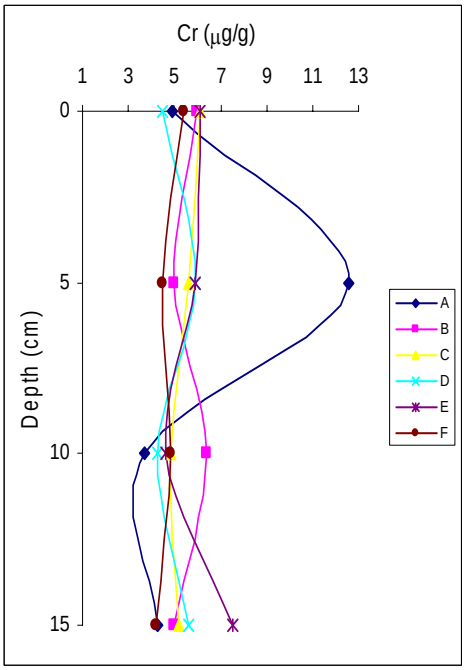
i) Transect 1



ii) Transect 2



iii) Transect 3



iv) Transect Control

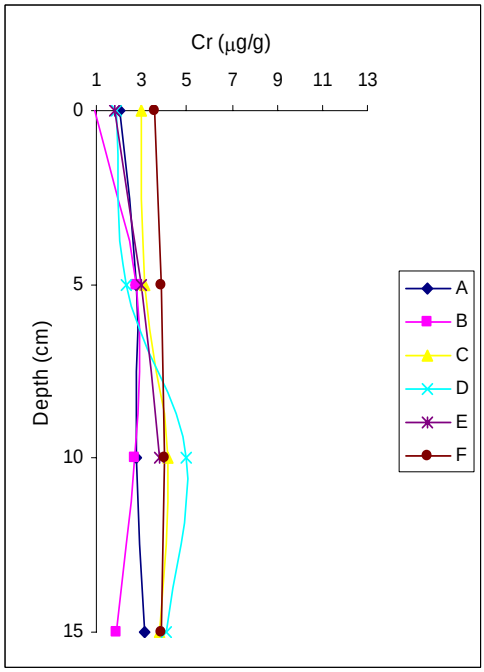


Figure 5.7: Levels of Chromium (Cr) vs. depth for the Kiribati study site.

Graph points A to F are positions along the transect, where A = 0, B = 2.5, C = 5, D = 10, E = 20, F = 40 m along the transect, starting with A at the seawall.

Table 5.1: Biological effects-based sediment quality guidelines (SQGs) (Unit µg/g dry weight).

	NOAA[a]		Florida Department[b]	
Contaminant	**ERL**	**ERM**	**TEL**	**PEL**
Arsenic	8.2	70	7.24	41.6
Cadmium	1.2	9.6	0.68	4.21
Chromium	81	370	52.3	160
Copper	34	270	18.7	108
Lead	46.7	218	30.2	112
Mercury	0.15	0.71	0.13	0.7
Nickel	20.9	51.6	15.9	42.8
Zinc	150	410	124	271

[a] NOAA (National Oceanographic and Atmospheric Administration) National Status and Trends Program (Effects Range-Low (ERL) and Effects Range-Median (ERM)); [b] Florida Department of Environmental Protection (Threshold Effects Level (TEL) and Probable Effects Level (PEL)) (adopted from Denton *et al.* (2001)).

5.2.5.2 Copper (Cu)

The surficial sediments contain from 0.3 to 13.9 µg/g copper and in most sediment, the copper concentrations are generally between 0.3 and 4 µg/g (Appendix 6). Figure 5.8 shows the levels of copper concentrations without the control transect due to the levels in the control being below the detection limit of 1.2 µg/g for the instrument used for metal analysis indicating lower levels at the control site.

Transect 1 had higher copper concentrations than the other 2 landfill transects, ranging between 2.1 and 13.9 µg/g. There was no clear trend of copper concentrations down the profile, only showing slightly decreasing down the profile. Interestingly, transect 1 has only an enrichment of copper concentrations at the 5 cm

depth for almost all the 6 positions along the transect. Increased sedimentation at the site in the past might be the explanation for the pollution history of this site, however, this is not reflected in other contaminants levels found in this study.

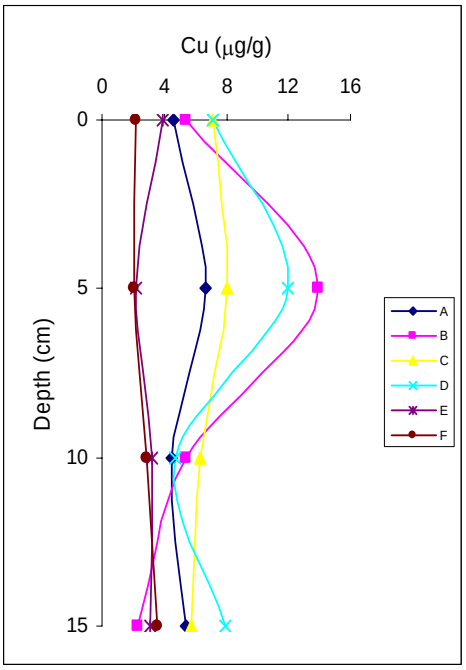
The copper concentrations found in this study, when compared with the sediment quality guidelines, were below the Effects Range-Low and Threshold Effects Level indicating no potential adverse ecological effects on the local communities. Comparing cooper concentrations with levels suggested for clean reef sediments, the concentrations for the landfill transects were above the ranges whereas the control site was within the suggested ranges.

Table 5.2: Suggested concentration ranges for heavy metals in clean reefs from nearshore and offshore locations (Unit µg/g dry weight).

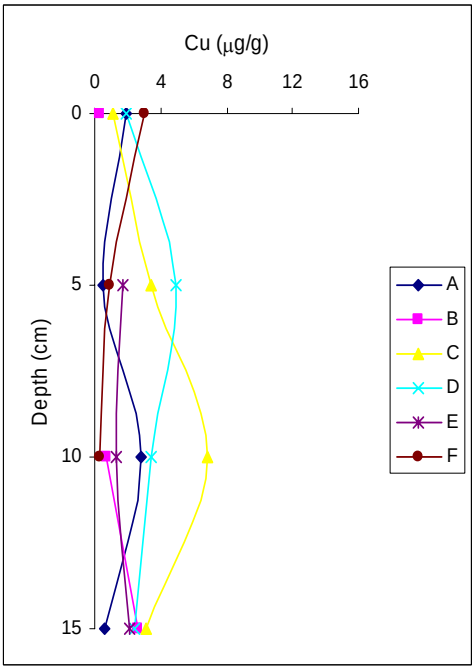
Contaminant	Nearshore	Offshore
Arsenic	1 – 3	0.5 – 2.0
Cadmium	<0.1	<0.1
Chromium	3 – 5	1 – 3
Copper	1 – 3	<0.1
Lead	<1	<1
Mercury	0.005 – 0.01	0.002 – 0.005
Nickel	1 – 3	<0.2
Zinc	3 – 5	<1

(adopted from Denton *et al.* (2001))

i) Transect 1



ii) Transect 2



iii) Transect 3

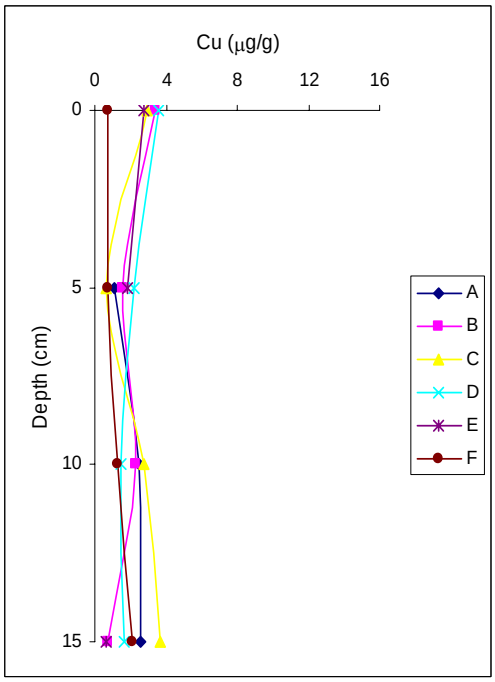


Figure 5.8: Levels of Copper (Cu) vs. depth for the Kiribati study site.

Graph points A to F are positions along the transect, where A = 0, B = 2.5, C = 5, D = 10, E = 20, F = 40 m along the transect, starting with A at the seawall.

5.2.5.3 Lead (Pb)

The lead concentrations found in this study varied between 3.4 and 13.3 µg/g. The sediments at the landfill site mostly contain between 7 and 13 µg/g whereas at the control, the levels are between 3 and 6 µg/g, indicating a higher Pb concentration at the landfill site. At the landfill, transects 2 and 3 the levels ranged between 6 and 11 µg/g whereas transect 1, the levels were relatively higher ranging between 10 and 14 µg/g. Lead concentration did not show a clear trend along the transects or down the profile for either sites (Figure 5.9).

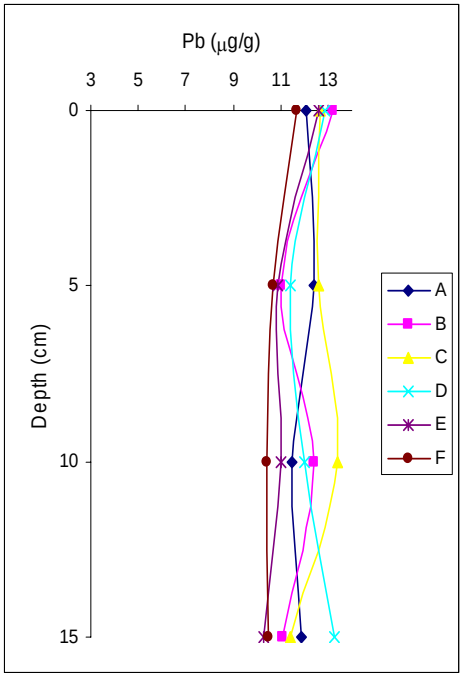
The concentrations for lead were well below the below the Effects Range-Low and Threshold Effects Levels for the 2 sediment quality guidelines used in this study. However, when lead concentrations found in this study are compared with the suggested background concentration ranges for clean reef (Table 5.2), the levels are above the background ranges at both the landfill and control site.

5.2.5.4 Zinc (Zn)

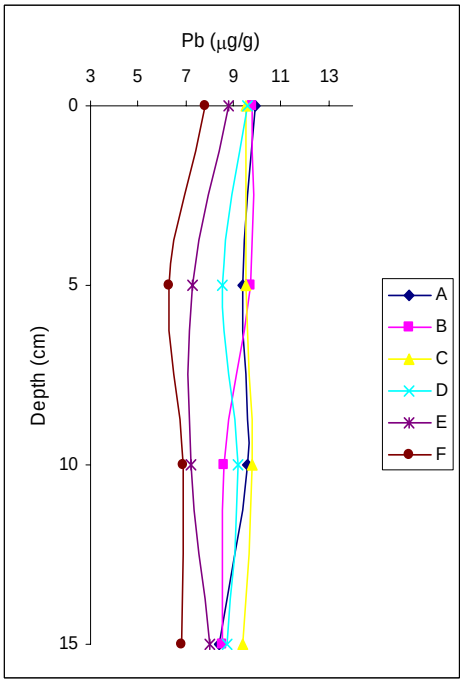
Zinc was found to have a relatively high concentration (more abundant) in this study compared with the other contaminants investigated. The surficial sediment concentrations for Zn varied between 1.2 and 77 µg/g. The Zn concentrations at the landfill ranged between 7 and 77 µg/g whereas the control had low surfical sediment concentrations ranging between 1 and 4 µg/g.

At the landfill site, transects 1 and 2 had decreasing Zn levels away from the landfill seawall (Figure 5.10). There is no obvious explanation of a spike in Zn concentration in transect 2 at a 10 cm depth. The concentration down the profile for the 4 transects generally appeared to be stable.

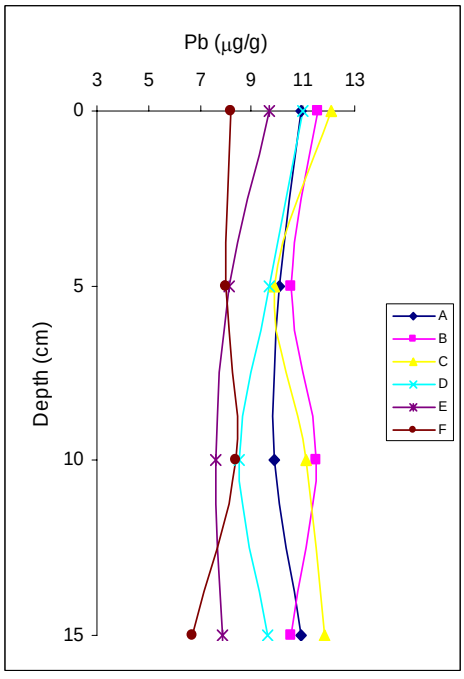
i) Transect 1



ii) Transect 2



iii) Transect 3



iv) Transect Control

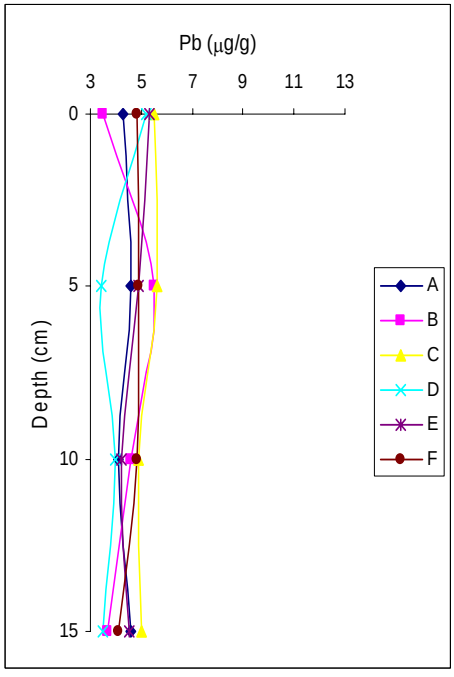
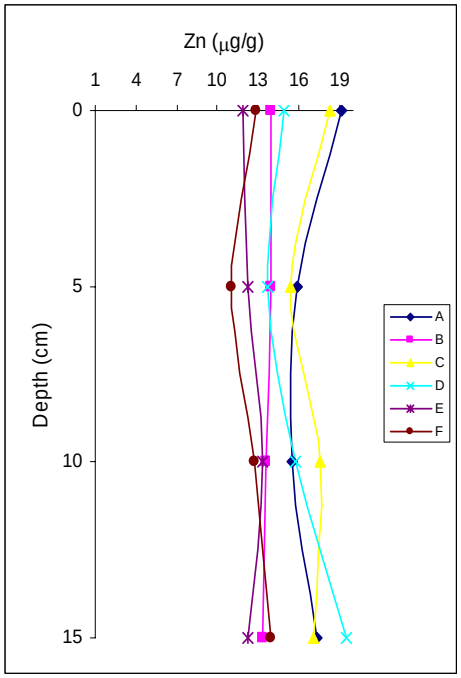


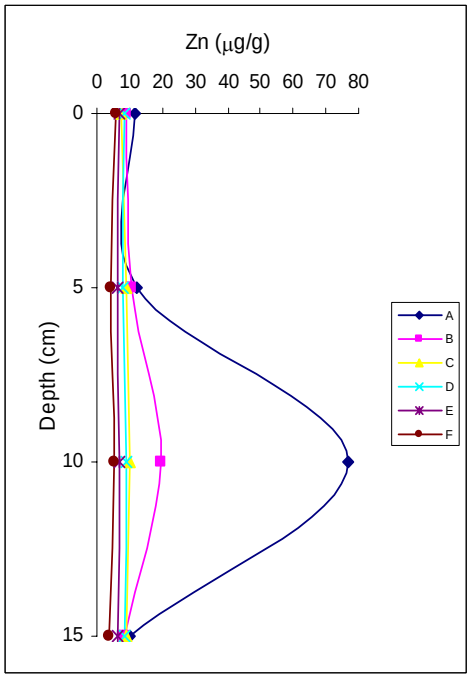
Figure 5.9: Levels of Lead (Pb) vs. depth for the Kiribati study site.

Graph points A to F are positions along the transect, where A = 0, B = 2.5, C = 5, D = 10, E = 20, F = 40 m along the transect, starting with A at the seawall.

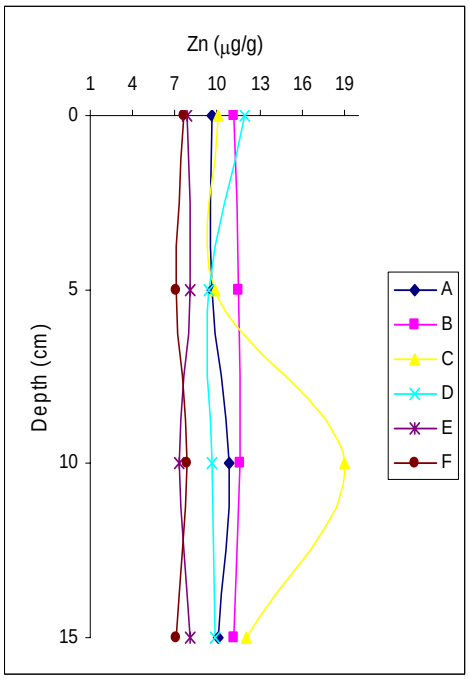
i) Transect 1



ii) Transect 2



iii) Transect 3



iv) Transect Control

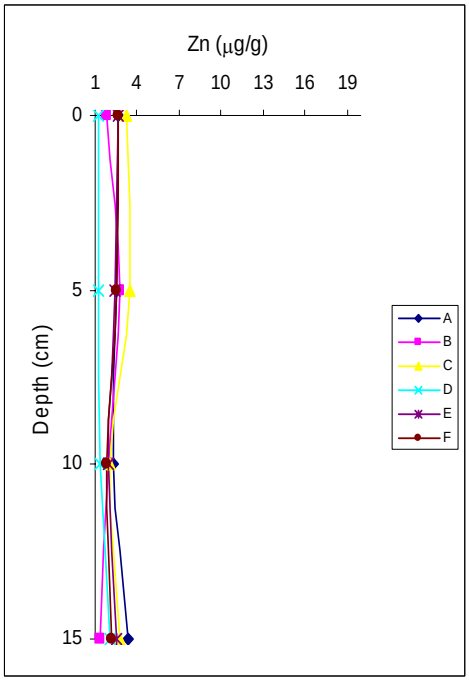


Figure 5.10: Levels of Zinc (Zn) vs. depth for the Kiribati study site.

Graph points A to F are positions along the transect, where A = 0, B = 2.5, C = 5, D = 10, E = 20, F = 40 m along the transect, starting with A at the seawall.

The concentrations for Zn, when compared to the sediment quality guidelines used in this study were below the Effects Range-Low (ERL) and Threshold Effects Levels (TEL). In terms of the suggested concentration ranges for clean reef sediments, the concentration for Zn found in this study was within the range for the control while the landfill site like Cr, Cu and Pb was found to exceed the suggested ranges.

5.2.5.5 Mercury (Hg)

The concentrations for mercury varied between 0.3 to 1.3 µg/g (Appendix 6). 76 percent of the samples were below the detection limit. The concentrations at the landfill transects ranged between 0.3 to 1.3 µg/g and the control had the mercury concentrations ranges between 0.4 to 1.3 µg/g (Table 5.3). The relatively high concentration of mercury at the control site suggests that there were potential sources of mercury other than the landfill in the lagoon.

The highest mercury concentration exceeded both the Effects Range-Low (ERL) and Effects Range-Median (ERM) for the NOAA and the Threshold Effects Levels (TEL) and Probable Effects Level (PEL) of the sediment quality guidelines (Table 2.1). Sediment concentrations higher than the ERM are considered highly contaminated, and are associated with frequent negative biological effects.

Comparing the mercury concentrations with the suggested concentration ranges for clean reef sediments, the concentrations for Hg found in this study were well above the background ranges. The relatively high mercury concentration poses particular concern for human health and the environment knowing mercury toxicity and its ability to bio-accumulate in the food chain even in small amount.

Denton *et al.* (2001) reported that mercury concentrations of 5-10 ng/g were typical of biogenic carbonates in uncontaminated environments. Sediments with mercury levels of 20-100 ng/g are classified as lightly enriched whereas those with values of up to 300 ng/g were considered moderately contaminated and sediments with

mercury concentrations between 300-1000 ng/g were indicative of grossly contaminated sediments.

5.2.5.6 Arsenic (As), Cadmium (Cd) and Nickel (Ni)

The levels for As, Cd, and Ni were all below the detection limit and therefore the concentrations are not in the results table (Appendix 6). Arsenic has a detection limit of 0.4, cadmium 0.5, 1.5 and nickel 1.9 µg/g respectively.

Table 5.3: Mercury concentration in the sediment found in the present study (Unit µg/g)

Site	Lowest	Highest	No. of samples above detection limit
T1	1.1	1.1	1
T2	0.6	1.2	7
T3	0.3	1.1	9
TC	0.4	1.3	6

T1 – Transect 1; T2 – Transect 2; T3 – Transect 3 and TC Transect Control

5.3 New Zealand case study

5.3.1 Sample description

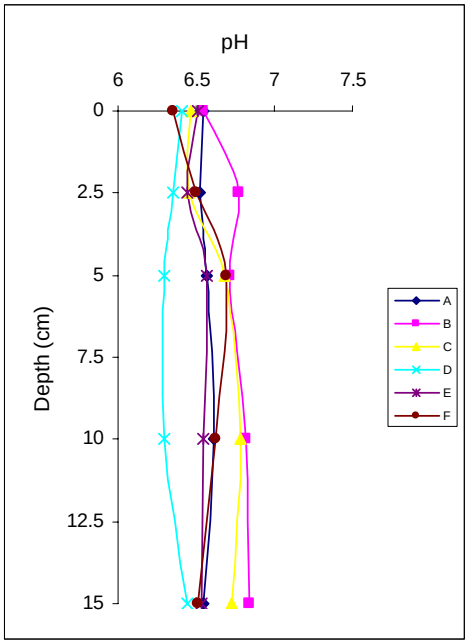
The 15 cm deep sediment cores from the Taipari Strand, Waitemata Habour were made up of fine muddy sand. The sediment load appeared to be dominated by land run-off (terrigenous material).

There was a slight difference in appearance of the samples from the landfill and the control site. The cores from the landfill area were 'soft' and the color appeared to be brownish in the top few millimetres (subsurface) but became dark/black with depth. The black color indicates black ferrous sulphide and pyrites (FeS_2) produced when sulphide in the sediment reacts with iron (Fe) (Williamson *et al.* 2003). The control cores appeared more homogeneous in color, dark grey, relatively more solid and intact compared to the cores from the landfill site.

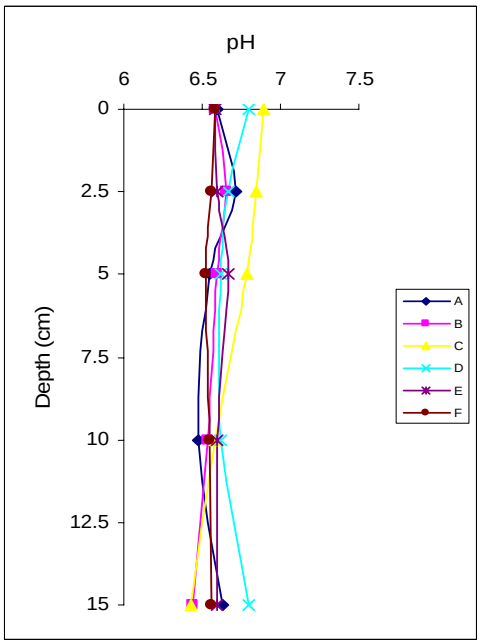
5.3.2 pH

The pH of the sediments sampled from the New Zealand study site was quite low, ranging between 6.22 and 7.24 (Figure 5.11, Appendix 2). The pH was at the landfill seawall and increased towards the channel (Figure 5.11). However, there was no clear pattern of pH changes with depth though it is expected to decrease down the profile. Both the landfill and control pH were in the same range indicating no significant difference in pH between the landfill and control site.

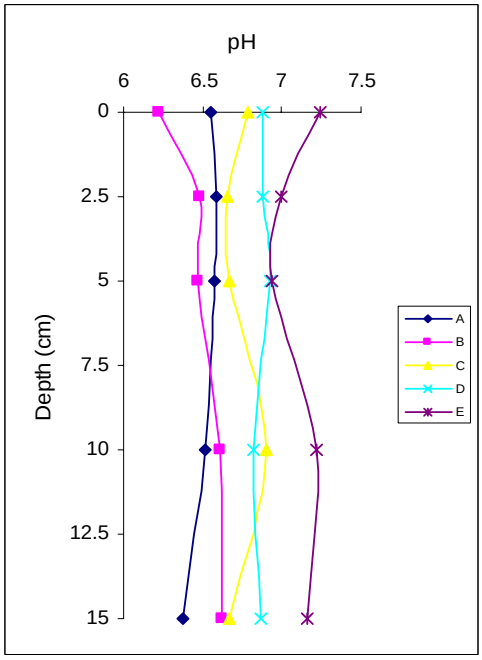
i) Transect 1



ii) Transect 2



iii) Transect 3



iv) Transect Control

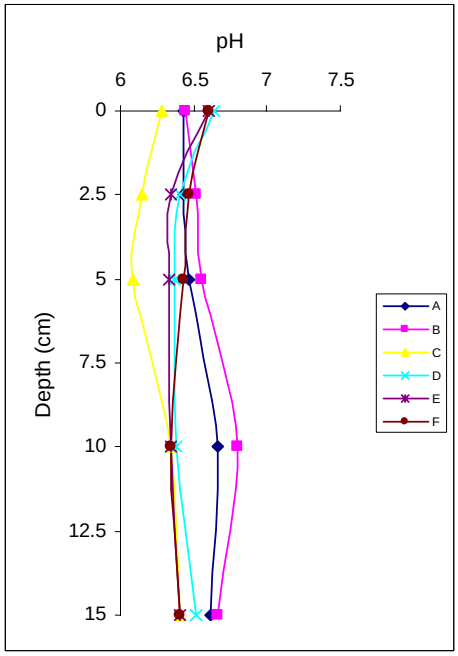


Figure 5.11: Sediment pH vs. depth for the New Zealand study site.

Graph points A to F are positions along the transect at 5 m intervals with A at the seawall.

5.3.3 Particle size distribution

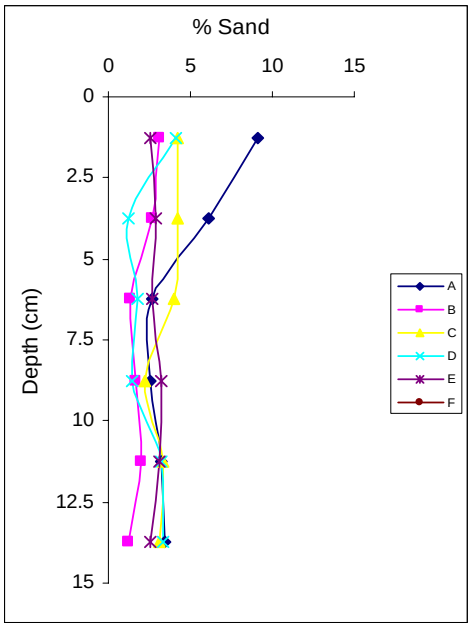
The New Zealand study site is a good example of a zone of accumulation, which is characterized by a high percentage of fine sediments. The sediment was made up of sand, silt and clay with the bulk made up of silt/clay. Figures 5.11, 5.12 and 5.13 show that at the landfill site, transect 1 and 2 had less than 10% sand with silt/clay comprising more than 90% of the sediment fraction whereas transect 3 was slightly coarser made up of less than 15% sand (Appendix 3). Transect control showed a much higher percentage of sand but only in sampling point A which accounted for more than 50% sand but the other 5 sampling points were all made up of less than 20% sand (Figure 5.14).

At the landfill, the percentage sand was high at the seawall and decreased away from the landfill towards the channel accompanied by the silt/clay percentage content increasing. Down the profile, the percentage sand decreasing while the percentage clay increased. Silt content did not appear to change with depth. This same pattern was also observed at the control site except for position A.

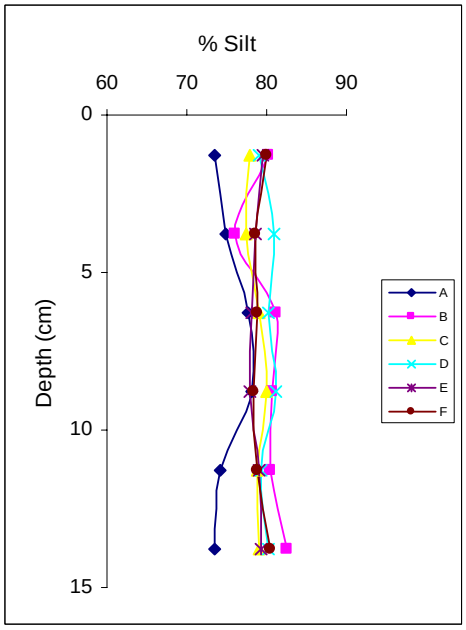
Taipari Strand is boarded by mangroves like all of the Henderson Creek. Mangrove forests are known to act as efficient binders and traps of sediment. They offer an ideal environment for fine sediment deposition by providing a shelter from wave and tidal energy and by doing so, act as settling basins for suspended material. Taipari Strand is an excellent example of an area of accumulation (Leong *et al.* n.d.).

The strong correlation between decreasing sediment size and increasing heavy metal concentration is well documented and suggests that adsorption is the main mechanism by which trace metals accumulate on particles, given that small particle size has a much higher surface area relative to their volume than large particles (Guerra-Garcia & Garcia-Gomez 2005). It is generally accepted that trace metals are mainly concentrated in the clay/silt sediment fraction, consisting of particles of grain sizes <63 μm (Krumgalz *et al.* 1992).

i) Sand



ii) Silt



iii) Clay

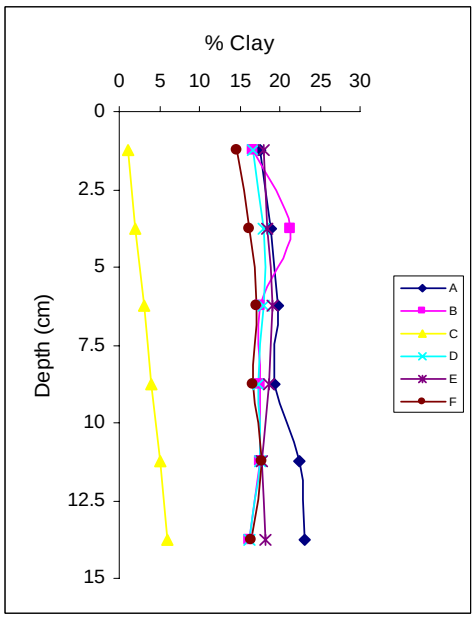
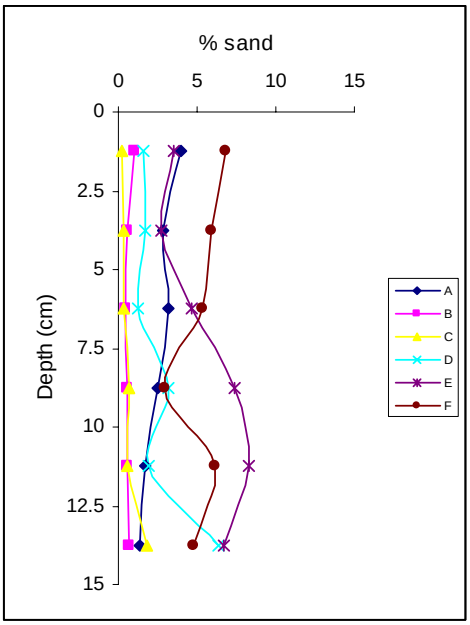


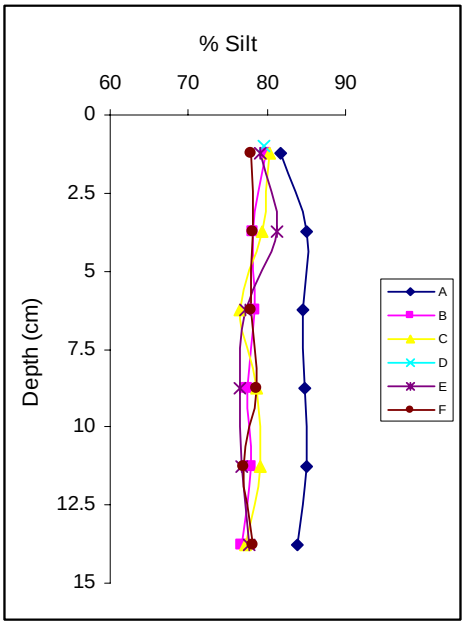
Figure 5.12: Particle size distribution vs. depth for the New Zealand study site for the < 2 mm fraction;

i) sand (2000 – 63 µm), ii) Silt (63 – 2 µm), and iii) clay (<2 µm) contents in the sediment for Transect 1. Graph points A to F are positions along the transect at 5 m intervals with A at the seawall. (Each point represents a sample taken from the depth of the point ± 1.25 cm)

i) Sand



ii) Silt



iii) Clay

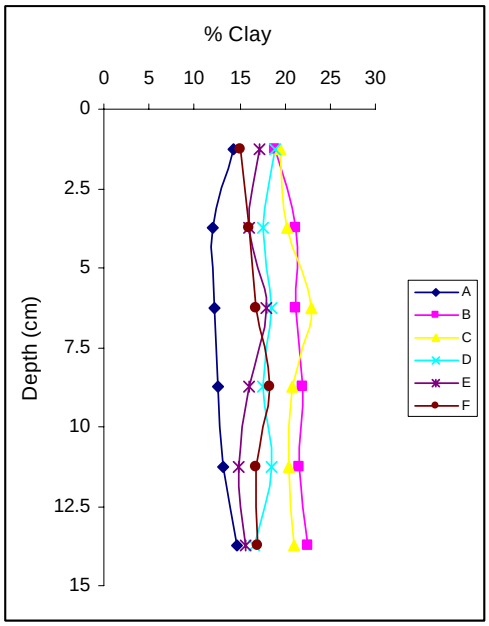
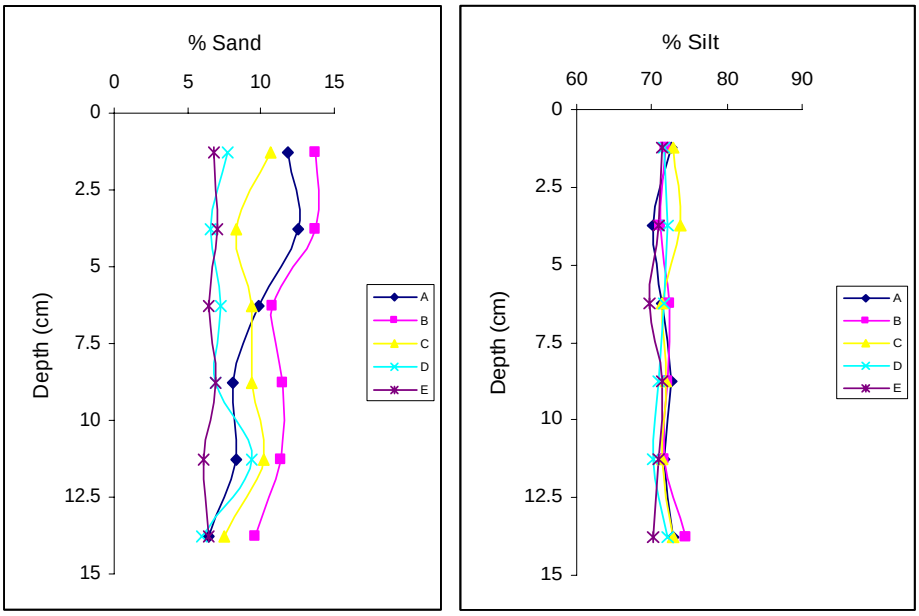


Figure 5.13: Particle size distribution vs. depth for the New Zealand study site for the < 2 mm fraction;

i) sand (2000 – 63 µm), ii) Silt (63 – 2 µm), and iii) clay (<2 µm) contents in the sediment for Transect 2. Graph points A to F are positions along the transect at 5 m intervals with A at the seawall. (Each point represents a sample taken from the depth of the point ± 1.25 cm)

i) Sand ii) Silt



iii) Clay

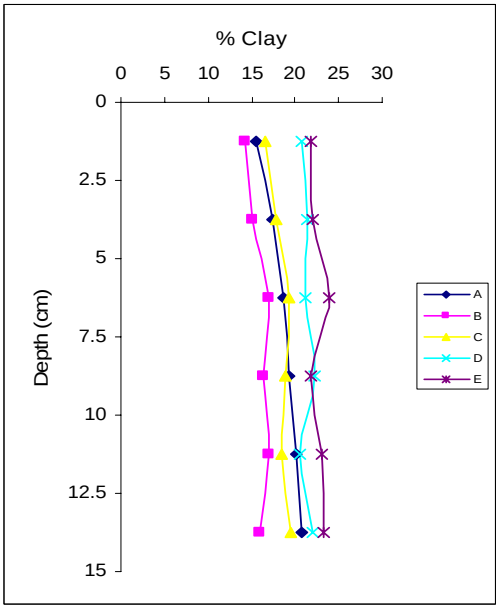
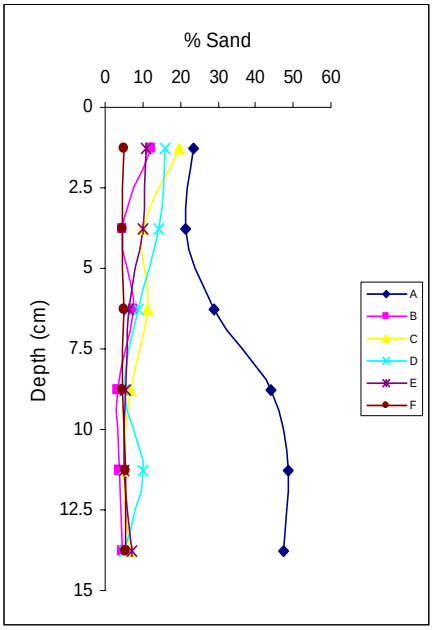


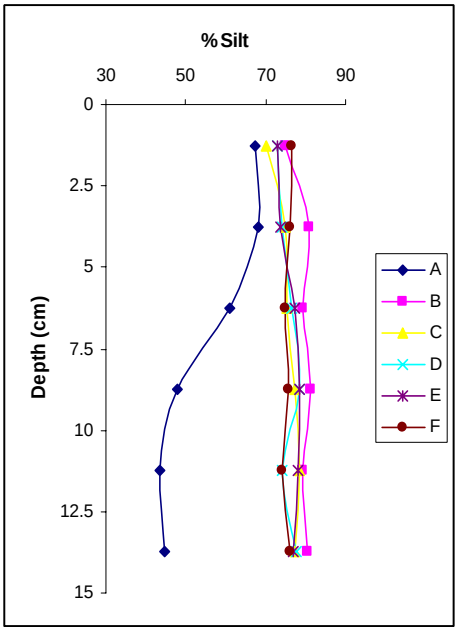
Figure 5.14: Particle size distribution vs. depth for the New Zealand study site for the < 2 mm fraction;

i) sand (2000 – 63 µm), ii) Silt (63 – 2 µm), and iii) clay (<2 µm) contents in the sediment for Transect 3. Graph points A to F are positions along the transect at 5 m intervals with A at the seawall. (Each point represents a sample taken from the depth of the point ± 1.25 cm)

i) Sand



ii) Silt



iii) Clay

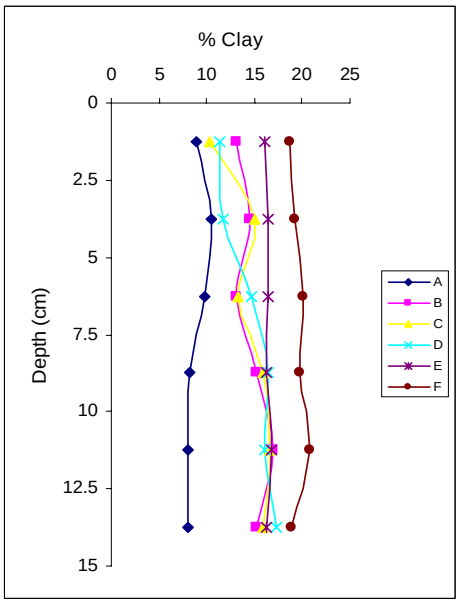


Figure 5.15: Particle size distribution vs. depth for the New Zealand study site for the < 2 mm fraction;

i) sand (2000 – 63 µm), ii) Silt (63 – 2 µm), and iii) clay (<2 µm) contents in the sediment for Transect Control. Graph points A to F are positions along the transect at 5 m intervals with A at the seawall. (Each point represents a sample taken from the depth of the point ± 1.25 cm)

The high silt/clay sediment fraction of the New Zealand sediment (up to 90%) means there would be a high capacity to adsorb trace metals

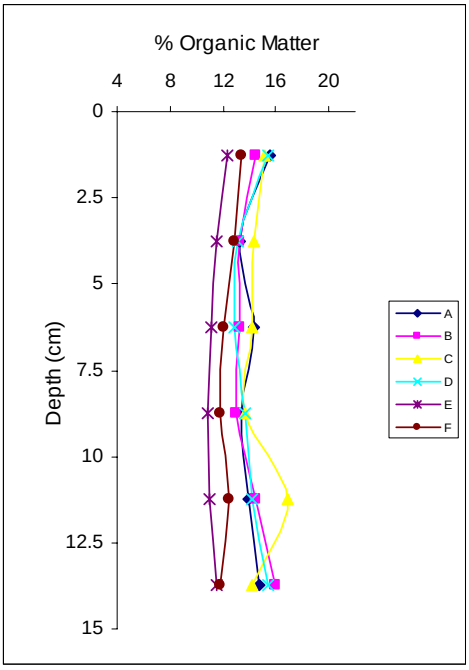
5.3.4 Percentage organic matter

The percentage organic matter was high, ranging between 4.84 and 22 % (Figure 15.6, Appendix 4). The organic matter content at the landfill ranged between 8.93 and 17.9 % however, the control in particular at position A, had the lowest and highest organic matter content. All the transects both landfill and control had a high organic matter content closer to the land and decreasing towards the channel. The organic matter content did not vary greatly with depth, except for one sample at position A in the control site.

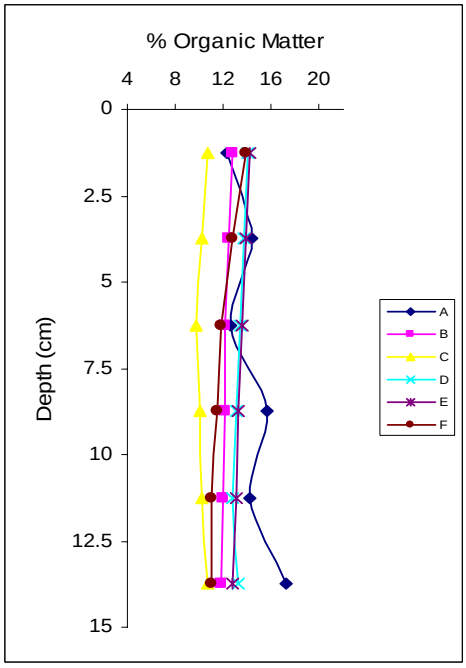
There was a weak correlation between sediment size distribution and organic matter content found in the sediments for both sites. The only clear correlation was shown by the abrupt decrease in organic matter content displayed in position A in the control which was correlated to the high percentage sand content. The increase in percentage silt/clay fraction down the profile was not reflected with an increase in percentage organic matter. However, the decrease in organic matter content with depth was consistent with a classical situation in sediment diagenesis of decreasing organic carbon with depth (Williamson & Wilcock 1994).

The high organic matter content in the sediments was attributed to the inputs by the mangrove swamp bordering the Taipari Strand landfill. Other potential contributors of organic matter to sediments at the study site include restricted tide range, biogenic productivity and the nearby industrial area with organic rich sewage (Loomb 2001).

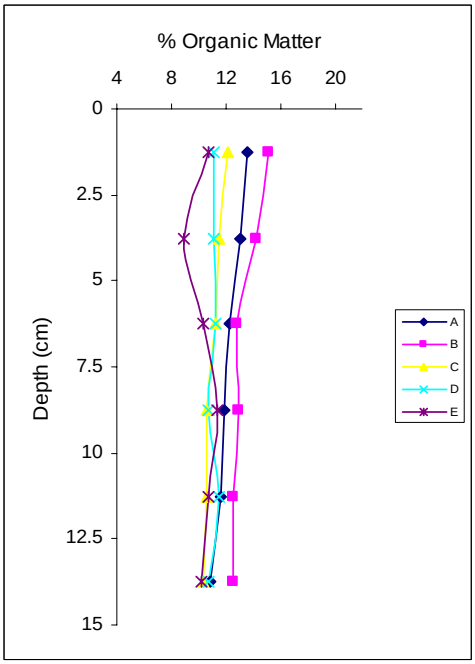
i) Transect 1



ii) Transect 2



iii) Transect 3



iv) Transcect Control

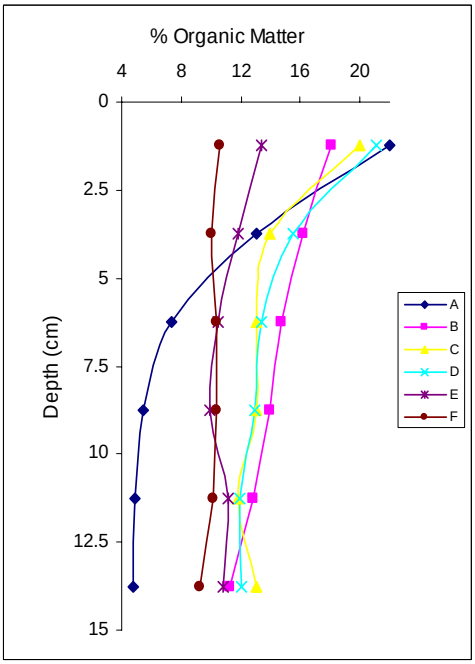


Figure 5.16: Percentage organic matter vs. depth for the New Zealand study site. Graph points A to F are positions along the transect at 5 m intervals with A at the seawall. (Each point represents a sample taken from the depth of the point ± 1.25 cm)

5.3.5 Total metal concentration

5.3.5.1 Arsenic (As)

The arsenic concentration in surficial sediments varied between 2.6-36 µg/g arsenic (Appendix 6), where most sediments contained 10-20 µg/g (Figure 5.17) in both the control and landfill transects. At the landfill site, arsenic concentration was high at the seawall and slowly decreased towards the channel. The concentrations appeared not changed with depth for most of the positions at the landfill transects except position A in transect 2, which clearly showed an increase in level down the profile. At the control, there was no clear trend in concentration both along the transect or with depth, however, position A was found to have lower concentrations than the other positions.

Comparing arsenic concentrations found in this study with the sediment quality guidelines (SQGs) (Table 5.1), arsenic concentrations exceeded both the Effects Range-Low (ERL) and Threshold Effects Levels (TEL) values indicating the potential for adverse ecological effects on the benthic communities. Arsenic highest concentration found in this study did not approach the Effects Range-Medium (ERM) and the Probable Effects Level (PEL) of the respective SQGs.

5.3.5.2 Chromium (Cr)

The surficial sediments contain from 2.4 to 94 µg/g chromium and in most sediments the chromium concentrations were generally between 40 and 80 µg/g (Appendix 6). The chromium concentrations for both the landfill and control transects did not appear to show a general pattern. Transects 2 and 3 (figure 5.18), had a lower chromium concentration at the seawall and increased away from the landfill however, transect 1 showed the opposite with position A at the seawall haviing an enrichment of chromium level.

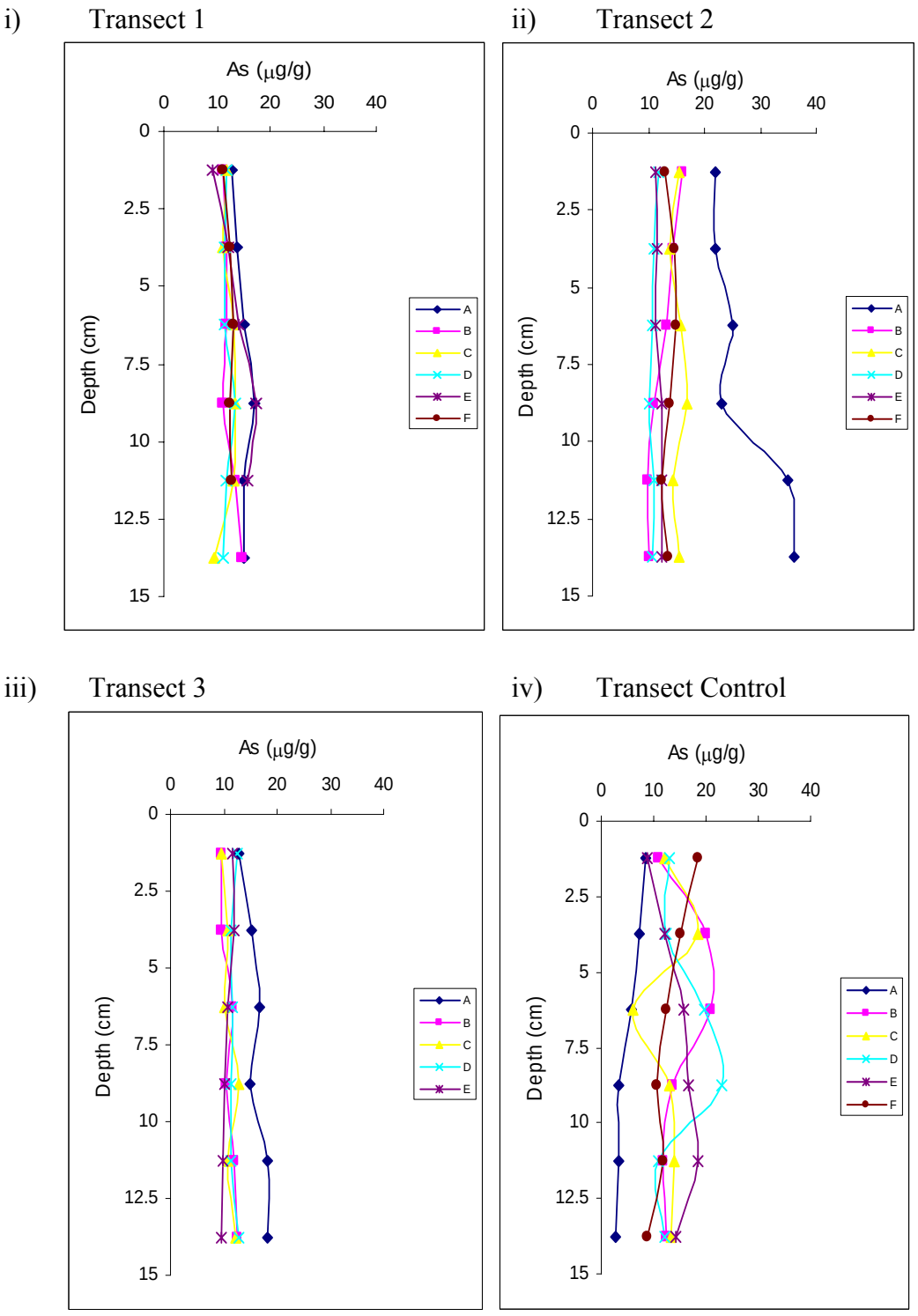
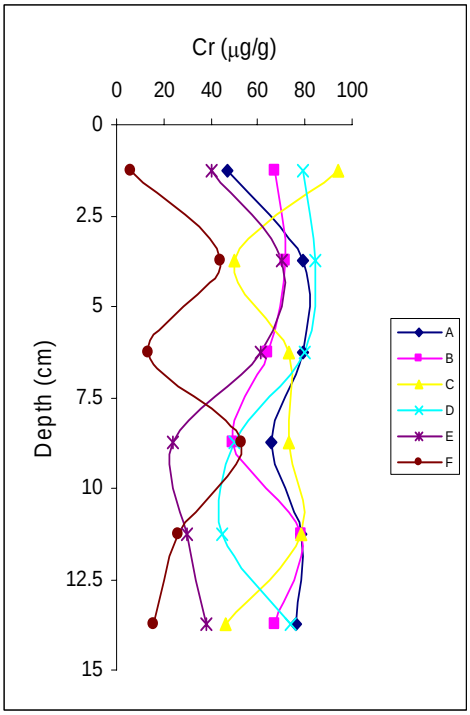
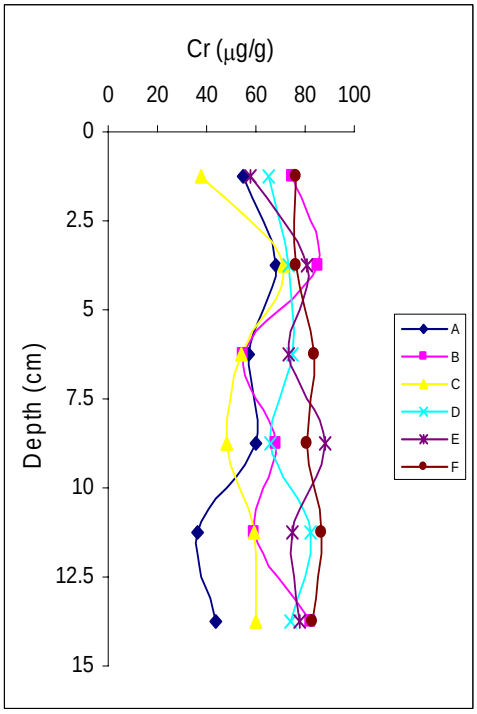


Figure 5.17: Levels of Arsenic (As) vs. depth for the New Zealand study site. Graph points A to F are positions along the transect at 5 m intervals with A at the seawall. (Each point represents a sample taken from the depth of the point ± 1.25 cm).

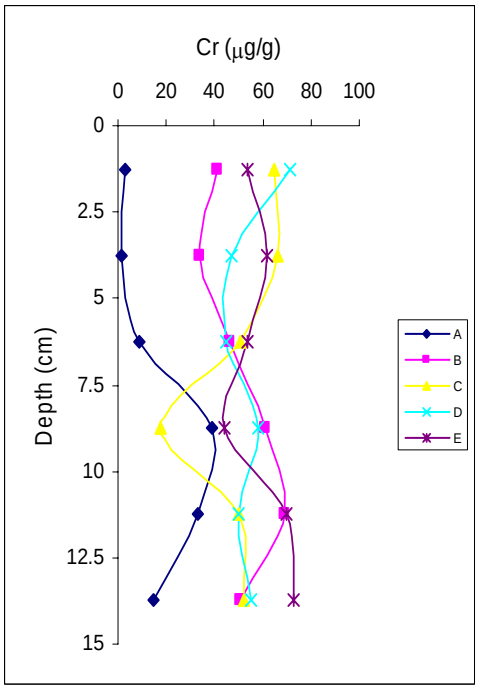
i) Transect 1



ii) Transect 2



iii) Transect 3



iv) Transect Control

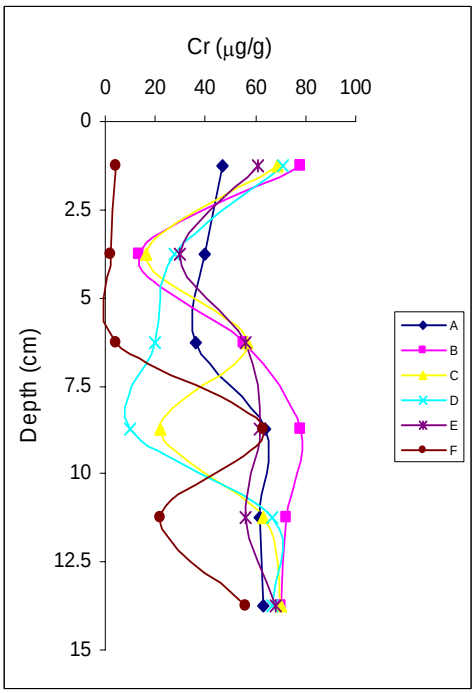


Figure 5.18: Levels of Chromium (Cr) vs. depth for the New Zealand study site. Graph points A to F are positions along the transect at 5 m intervals with A at the seawall. (Each point represents a sample taken from the depth of the point ± 1.25 cm).

The control transect had a low chromium concentration close to the channel and high level close to the land suggesting that chromium source comes from the land. There were no clear changes in chromium concentrations with depth showing a slight variation both within and between transects.

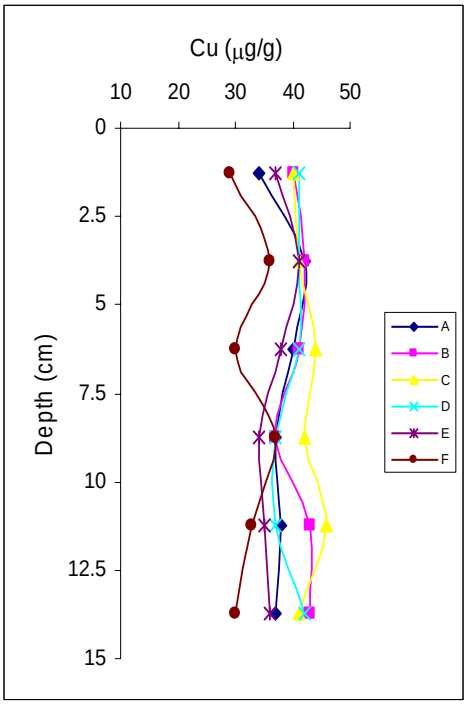
The chromium concentrations exceeded both the Effects Range-Low (ERL) and Threshold Effects Levels (TEL) but below the Effects Range-Median (ERM) and Probable effects Level (PEL) values. Sediments with concentrations above the ERL but below the ERM are considered moderately contaminated and are associated with frequent negative (Guerra-Garcia & Garcia-Gomez 2005).

5.3.5.3 Copper (Cu)

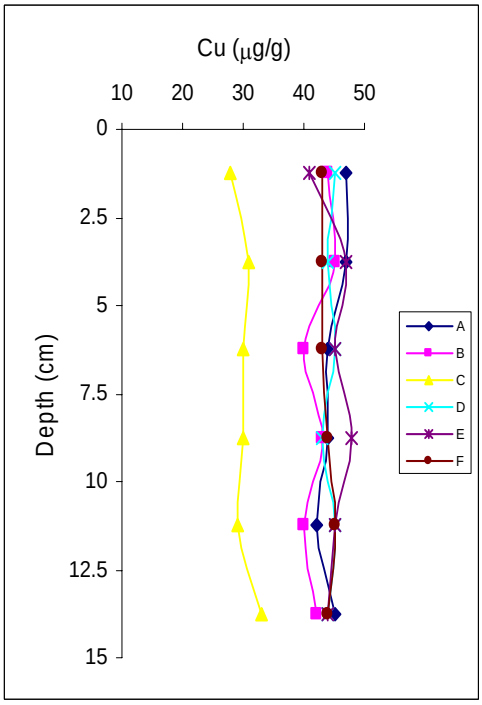
The surficial sediments contain from 14 to 48 µg/g copper (Figure 5.19, Appendix 6), however, in most cases, copper ranges from 25 to 45 µg/g. The landfill transects appeared to have minimal variation within and between transects for copper concentrations, however, the control transect was more variable (Figure 5.19). The copper concentration varied from 30 to 45 µg/g in landfills and 5 to 45 µg/g in the control.

Transect 1 shows relatively high concentration at the seawall whereas transect 3 had the opposite. Transect 2 did not show a clear trend as the concentrations for the 6 positions were clustered together. The copper concentration showed a drop in concentration for control and in transect 1 down the profile, however, transects 2 and 3 didn't show a concentration change with depth. Transect control had a high concentration of copper at the surface but decreased to low concentrations with the profile.

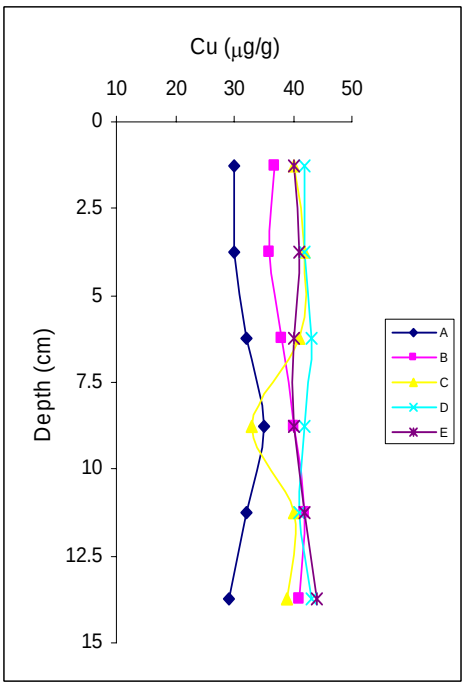
i) Transect 1



ii) Transect 2



iii) Transect 3



iv) Transect Control

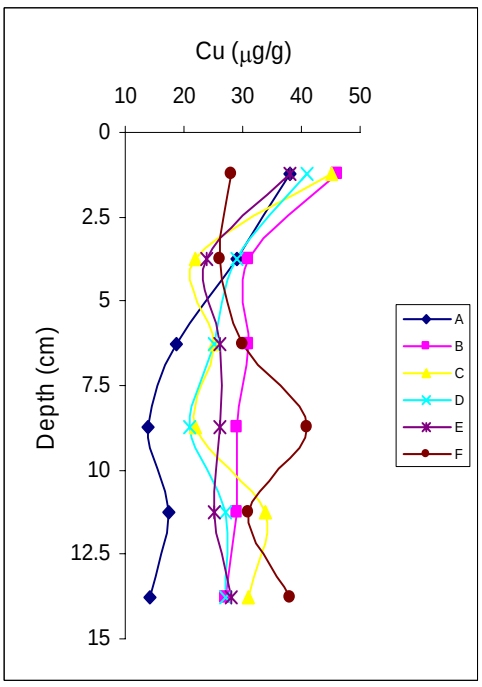


Figure 5.19: Levels of Copper (Cu) vs. depth for the New Zealand study site. Graph points A to F are positions along the transect at 5 m intervals with A at the seawall. (Each point represents a sample taken from the depth of the point ± 1.25 cm).

The concentrations for copper in this study were found to exceed the Effects Range-Low (ERL) and Threshold Effects Levels (TEL) values for the sediment quality guidelines but was below the Effects Range-Median (ERM) and Probable Effects Level (PEL).

5.2.5.4 Lead (Pb)

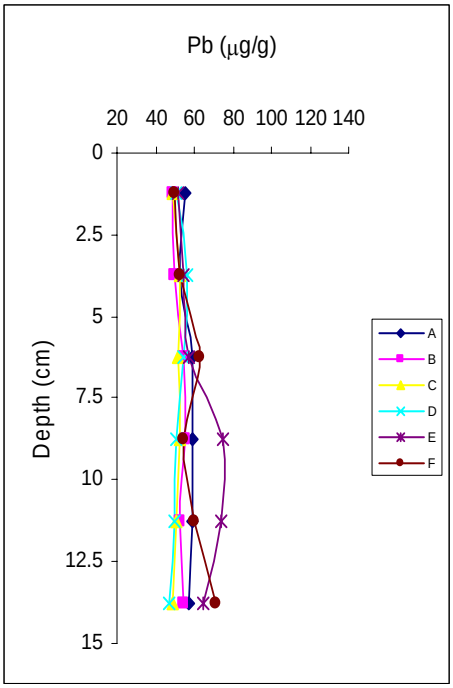
The lead concentration varied from 23 to 130 µg/g (Figure 5.20, Appendix 6), where most of the sediments generally contained 50 – 80 µg/g (Figure 5.20). There was little variability between the landfill transects while the control shows a more variation with higher overall lead concentrations. The landfill transects had a high concentration at the seawall and decreased away from the landfill. There was a spike in concentration in the control transect which account for the highest concentration and there is no apparent explanation for this.

The concentration for lead found in this study when compared with the sediment quality guidelines was found to exceed the Effects Range-Low (ERL) and Threshold Effects Levels (TEL) but well below the Effects Range-Median (ERM) and Probable Effects Level (PEL).

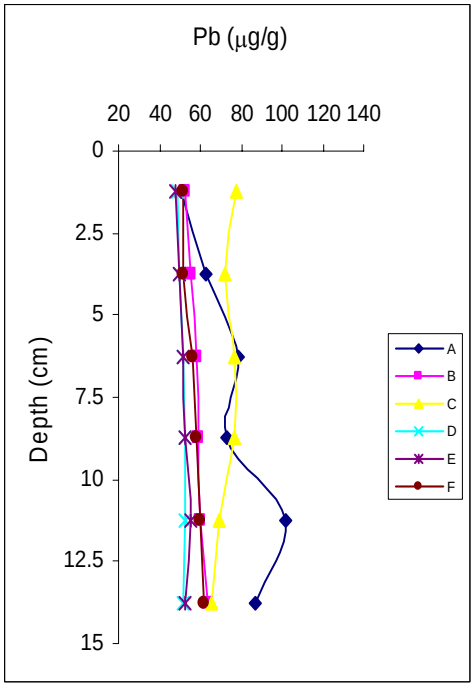
5.3.5.5 Nickel (Ni)

The concentration for nickel varied between 7.6 to 17.1 µg/g (Appendix 6). Most of the sediments contained 10 to 15 µg/g of nickel (figure 5.21). The landfill transects showed low concentration away from the seawall though transect 3 showed the opposite. The concentrations down the profile tended to be fairly uniform. There was no clear trend that could be established with depth. In the control, which was expected to have lower concentrations, the nickel levels were in the same ranges as the landfill concentrations suggesting not only landfill as a source of nickel in the proximity.

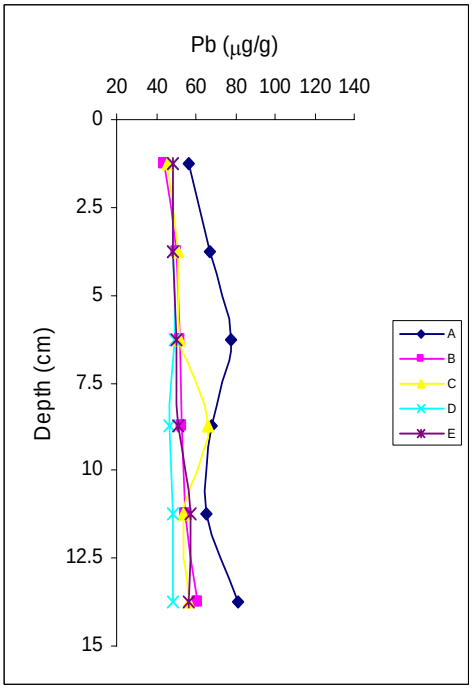
i) Transect 1



ii) Transect 2



iii) Transect 3



iv) Transect Control

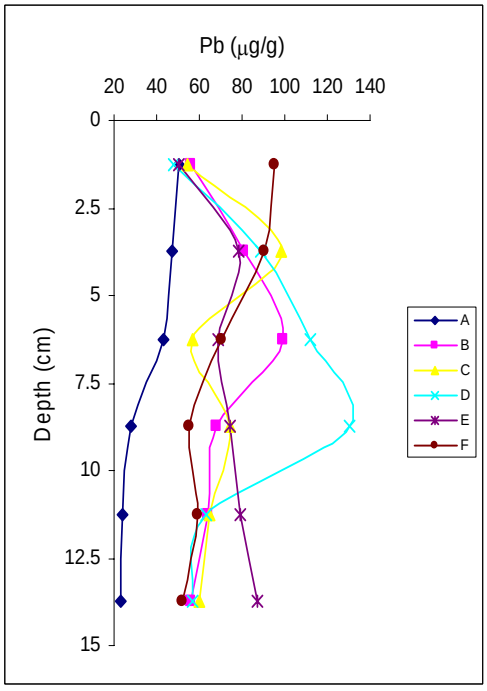
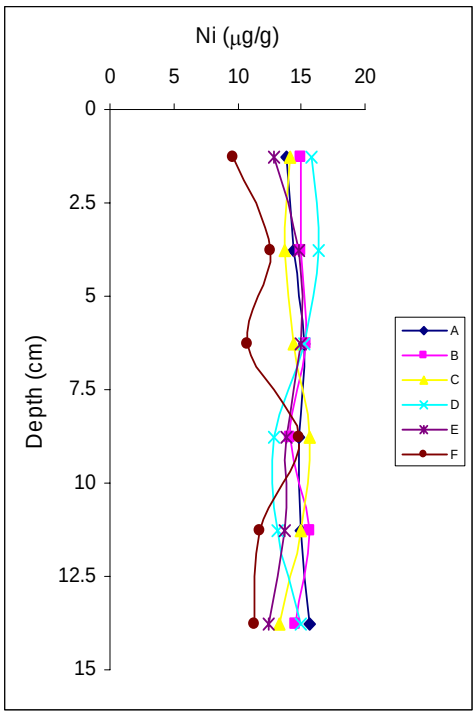
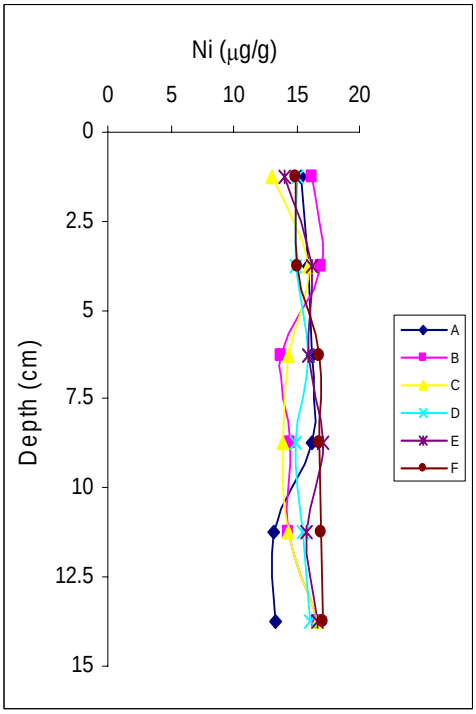


Figure 5.20: Levels of Lead (Pb) vs. depth for the New Zealand study site.
Graph points A to F are positions along the transect at 5 m intervals with A at the seawall. (Each point represents a sample taken from the depth of the point ± 1.25 cm).

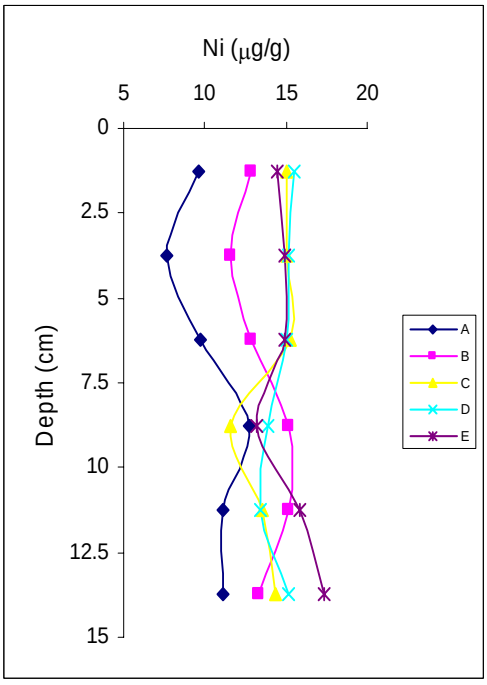
i) Transect 1



ii) Transect 2



iii) Transect 3



iv) Transect Control

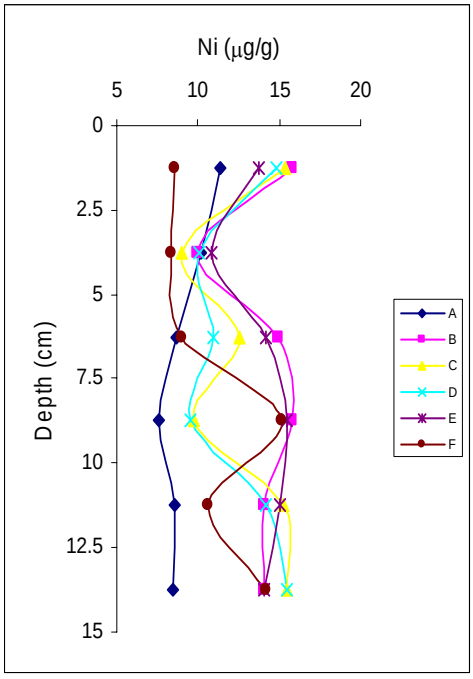


Figure 5.21: Levels of Nickel (Ni) vs. depth for the New Zealand study site. Graph points A to F are positions along the transect at 5 m intervals with A at the seawall. (Each point represents a sample taken from the depth of the point ± 1.25 cm).

The concentrations for nickel were compared with the sediment quality guidelines and it was found to fall below the Effects Range-Low (ERL) but just exceed the Threshold Effects Levels (TEL) value indicating a possible ecological effects on the benthic communities.

5.3.5.6 Zinc (Zn)

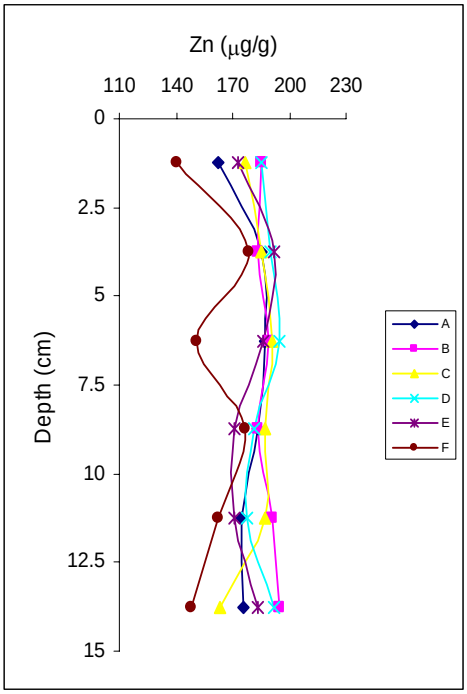
The zinc concentration was higher than the other metals. The zinc concentration in surficial sediments had from 97 to 896 µg/g (Appendix 6). Most sediment concentrations range between 170 to 210 µg/g of zinc (Figure 5.22). The zinc concentrations at the landfill showed a low concentration at the seawall for transects 2 and 3. There was a slight increase in concentration with depth. At the control, there was a spike which made up the highest concentration for zinc found in this study. There was no apparent explanation for this. The relative abundance of zinc found at the control site imply that there are other potential sources of zinc in the area causing the enrichment of zinc in marine sediments at the control.

The zinc concentration in the spike in the control transect exceeded both the Effects Range-Low (ERL) and Effects Range-Median (ERM) for the NOAA guidelines and Threshold Effects Levels (TEL) and Probable Effects Level (PEL) for the Florida Department of Environment guidelines. Consequently, the sediment was considered grossly contaminated associated with frequent negative biological effects (Guerra-Garcia & Garcia-Gomez 2005). For the remaining samples, the zinc concentrations only exceeded the ERL and TEL of the respective SQGs.

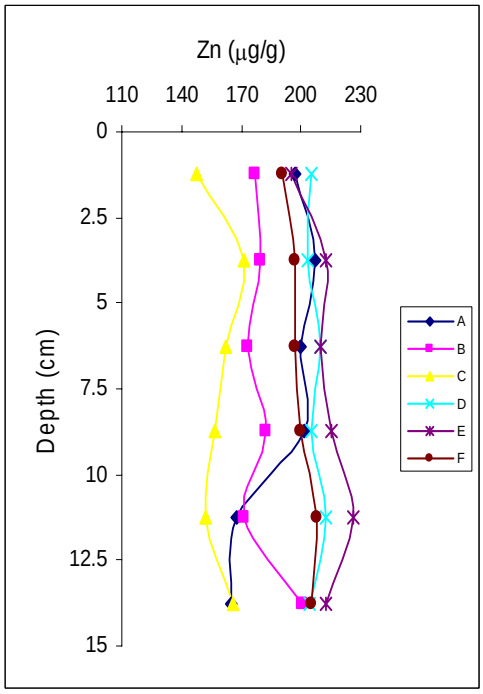
5.3.5.7 Mercury (Hg)

The concentrations for mercury varied between 0.3 to 1 µg/g. Only 24 percent of the samples had measurable mercury concentrations and the rest were undetectable.

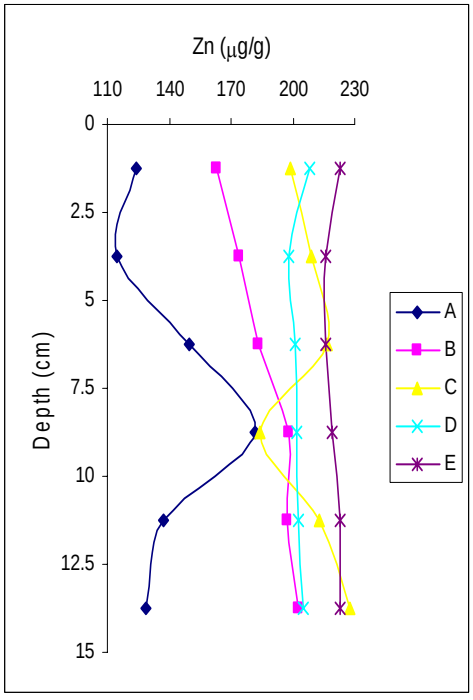
i) Transect 1



ii) Transect 2



iii) Transect 3



iv) Transect Control

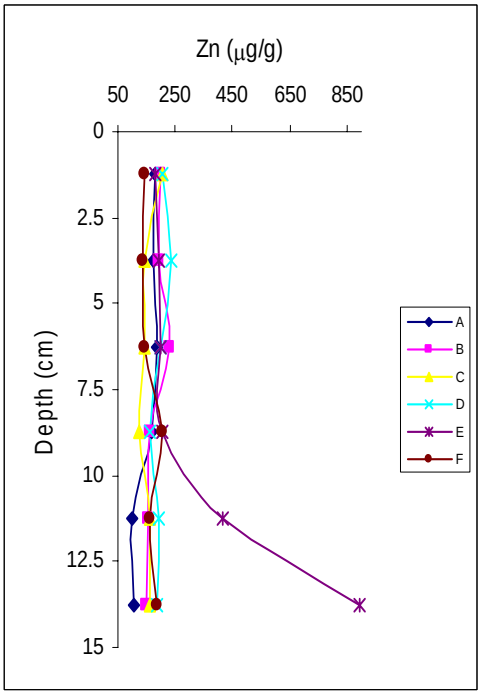


Figure 5.22: Levels of Zinc (Zn) vs. depth for the New Zealand study site.
Graph points A to F are positions along the transect at 5 m intervals with A at the seawall. (Each point represents a sample taken from the depth of the point ± 1.25 cm).

The highest mercury concentration exceeded both the Effects Range-Low (ERL) and Effects Range-Median (ERM) for the NOAA and the Threshold Effects Levels (TEL) and Probable Effects Level (PEL) of the sediment quality guidelines (Table 2.1). Sediment concentrations higher than the ERM are considered highly contaminated, which and are associated with frequent adverse ecological impacts on benthic communities.

5.3.5.8 Cadmium (Cd)

Levels of cadmium were not detectable (below the detection limit). The detection limit for cadmium was 0.5 µg/g.

5.4 Comparison between the two study sites

The two study sites appear to differ in a number of ways.

5.4.1 Sediment types

One of the most important differences between the two study sites that was evident was the sediment type. The Kiribati sediments were comprised mainly of biogenic carbonates (calcium carbonate) derived directly from the reef flatform, reflecting the geological formation of the island. The New Zealand sediments were fine muddy sediments that appeared to be dominated by land run-off (terrigenous material). Sediment type is important as it can also determine sediment texture and other important sediment compositions such as metal-reactive components of sediment (Fe and Mn hydrous oxides, and organic materials) that are important in determining the fate of metals in sediments (Williamson & Wilcock 1994).

5.4.2 Hydrodynamics

The Kiribati study site was a good example of a transportation zone which is characterized by diverse sediments ranging from <1 mm (sand, silt and clay) to > 5.6 mm (coarse sand and gravel). The site was exposed to tidal and coastal currents and therefore has high hydrodynamic energy and fine particulates, which were associated with metals, are moved on (Williamson *et al.* 2003). Metal concentrations were low in areas with coarse-grained sediment and in areas that are non-depositional. These areas have high bottom stress due to the local hydrodynamic regime and therefore the sediment is continually reworked (Loomb 2001).

In contrast to Kiribati, the New Zealand study site was a good example of a zone of accumulation, which was characterized by a high percentage of fine sediments (up to >90 % clay/silt (<63 μm) fraction). The mangroves, which were prominent at the site, provide shelter from wave and tidal energy and therefore act as settling basins for suspended materials (Leong *et al.* n.d.). The control site for the New Zealand study site was located approximately 2 km north-east of the landfill towards the incoming currents (Figure 4.3). The currents at the Henderson Creek channel are largely confined to the tidal currents that are a continuation of flow down the channel. Because of this and the close proximity of the landfill, it was possible that the control site was affected by the discharge from landfill.

5.4.3 pH

There was a clear difference in terms of pH between the two study sites. Kiribati ground water pH was quite high, ranging between 7.14 and 8.85, which is expected in coral atoll sediments. The pH was low at the seawall and increased towards the lagoon but decreased with depth. The high pH for the sediment means the metals are immobile.

The New Zealand sediment pH varied between 6.22 and 7.24. It was low at the seawall and increased towards the channel though it did not show a trend down the profile.

5.4.4 Particle size distribution and organic matter content

The general relationship between particle size distribution and organic matter content is that as the sediment texture becomes finer, the organic matter content of the sediment increases (Denton *et al.* 2001). This pattern was evident at the two study sites. Samples from Kiribati, which were comprised of diverse sediments ranging from fine sand to gravel, contained very low organic matter content (between 1.60 and 2.21%).

However, the New Zealand study site, which was made up of fine sediments (up to >90 % clay/silt (<63 μm) fraction) had very high organic matter content ranging between 4.84 and 22 %. The high organic matter content was correlated to the high clay/silt fraction. The mangroves bordering the area were contributing organic matter directly to the sediments whereas for the Kiribati study site, only patches of seagrass beds were a likely contributor.

Organic matter is decomposed by bacteria in the surface sediments using oxygen. In fine sediments like the New Zealand study site, decomposition occurs faster than the rate of diffusion of oxygen into the sediments and as a result, most sediments are anaerobic just below the surface. However, for coarse sediments and other high hydrodynamic energy areas like the Kiribati study site where there is rapid advection of oxygenated water, oxygen was abundant in the sediments (Williamson *et al.* 2003).

5.4.5 Total metal concentration

The concentrations of contaminants of interest were generally higher in the New Zealand study site than at the Kiribati study site. In Kiribati the concentrations for the metals, except mercury were all below the Sediment Quality Guidelines (SQGs) values used and are rarely associated with biological effects. In the Kiribati samples Mercury exceeded the Effects Range-Low (ERL) and Effects Range-Median (ERM) and the Threshold Effects Levels (TEL) and Probable Effects Level (PEL) of the SQGs at both the control and the landfill transect sites. The high levels of mercury were considered highly contaminated, and are associated with frequent adverse ecological impacts on benthic communities.

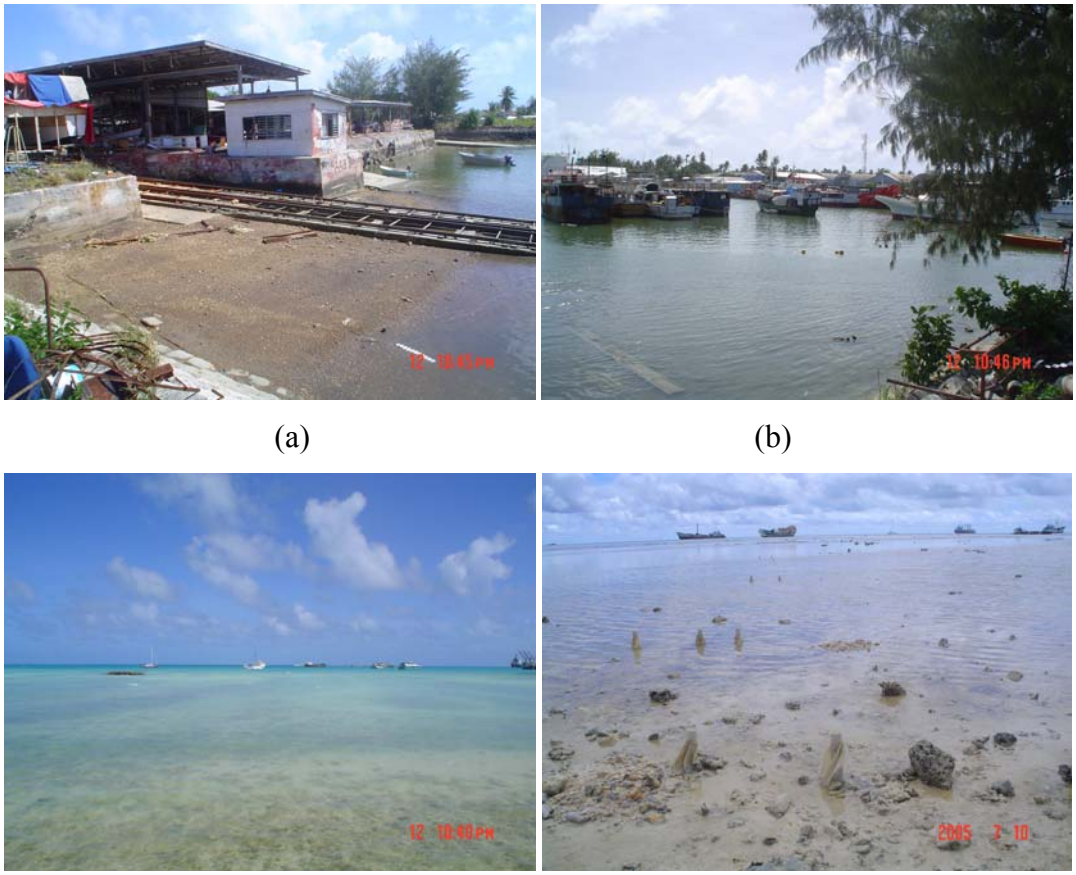
At the New Zealand study site, all metal concentrations exceeded the ERL and TEL of the SQGs except mercury and zinc, which both exceeded the TEL and PEL. The data indicate seriously contaminated sediments.

At the Kiribati study site, the metals concentration were higher at the landfill than the control transects whereas, in the New Zealand study site there was no marked differences in concentrations of metals between the landfill and the control transects.

5.4.6 Potential pollution sources

The two study sites had a few potential pollution sources other than the landfill investigated that could contribute to the elevated concentrations of metals in the sediments.

Betio Islet, the Kiribati study site, was where most of the industrial developments were undertaken on the island. Some of the prominent potential sources identified at the site include the main port (Betio Port), shipyard (boat repairing shop), shipwrecks and the harbour (Figure 5.23) adjacent to the landfill. This site was where most of the anthropogenic disturbances were concentrated.



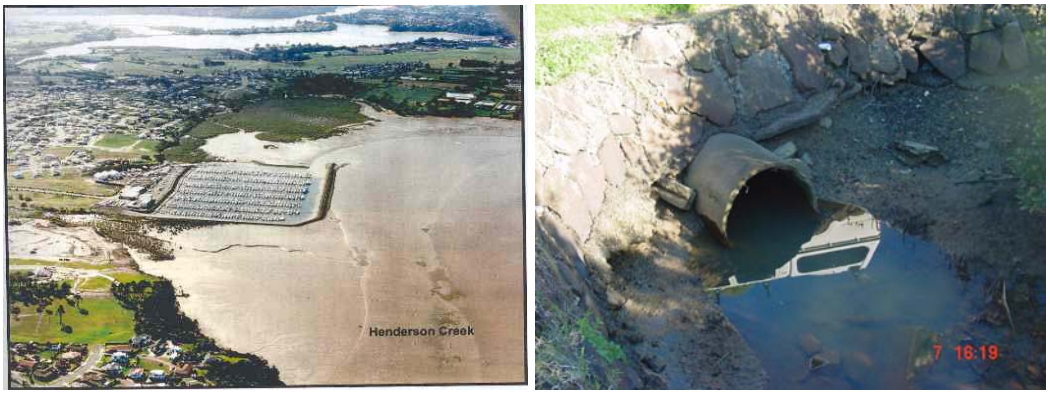
(a) (b)

(c) (d)

Figure 5.23: Potential sources of metals at the Kiribati study site:
(a) shipyard (boat repairing shop); (b) harbour; (c) adjacent lagoon used by mooring ships; and (d) shipwrecks lying on the lagoon reefs

The New Zealand study site potential sources of metals include the storm water discharge, the Westpark marina and urban industrial run-off (Figure 5.24)

The use of anti-fouling paints to cover the underwater parts of the boats, yachts and vessels to protect them from the development of fouling organisms such as algae and barnacles contain metals such as Cu, Pb, Cr, As and Zn which are toxic to organisms (Guerra-Garcia & Garcia-Gomez 2005). The use of anti-fouling paints has left a legacy of highly localized contamination around boating repair shops – mainly from paint removal before repainting. In the past, paints have included Hg and DDT (Williamson *et al.* 2003).



(a) (b)

Figure 5.24: Potential sources of metals at the New Zealand study site:
(a) Westpark marina ~ <5 km north of the Taipari Strand landfill; and (b) one of the storm water drainage outfalls at the landfill site.

The continual discharge of Zn, Cu (and Pb) in urban storm water is a significant source of anthropogenic metal to the coast. The source of Zn is tyre wear and galvanized plumbing and Cu mainly comes from brake linings (Williamson *et al.* 2003). Besides these contributions, atmospheric deposition of Pb from vehicle emissions is well documented. Emissions from automobiles consuming leaded petrol are the major source of atmospheric Pb and the atmospheric pathway may be a major source of Pb to marine sediments (Guerra-Garcia & Garcia-Gomez 2005).

Chapter Six: Conclusion & Recommendations

6.1 Summary

The aim of the present study was to investigate heavy metal contamination from landfills in coastal marine sediments. There were two study sites; Betio, Tarawa, Kiribati and Te Atatu, Auckland, New Zealand. Surficial sediments (top 15 cm) were collected using transects from both study sites and analyzed for physical (particle size analysis and organic matter content) and chemical (Cd, As, Cr, Cu, Pb, Hg, Ni and Zn concentrations) properties. pH was also measured. At each study site, there were 4 transects used with 6 sampling positions along each transects; 3 in the landfill and 1 in the control site.

6.1.1 Kiribati

Sampling was undertaken by using a geometric sample spacing along transects of; 0, 2.5, 5, 10, 20 and 40 m starting from the seawall. At each sampling point along the transect sediments were collected by hand from the surface, 5, 10 and 15 cm depth for analysis for both the landfill and control transects. Groundwater pH at the Kiribati site was high (7.14 - 8.85), a typical characteristic of coral atolls sediment. High pH means heavy metals are potentially retained in the sediments and therefore metals become immobile. The bulk (at least 40 % and over 80 %) of the samples were made up of finer sediments sand, silt and clay (<1.00 mm).

The sediment at the landfill transects were coarser close to the seawall and become finer away from the land and coarser with depth. The control transect had a higher proportion of fine sediment fractions compared to the landfill transects which was

attributed to the relatively sheltered location of the control transect, which favored deposition of finer sediments.

Organic matter content was low (1.60 – 2.21 %) and there was a weak correlation between sediment size distribution, organic matter content, and heavy metal concentrations for both landfill and control transects.

Metals investigated include As, Cd, Cr, Cu, Pb, Hg, Ni and Zn. Only concentrations of Cr, Cu, Pb, Hg and Zn were detected and the rest were below the detection limit in the Kiribati samples. At the landfill transects the distribution of some metals such as Cr and Zn were elevated close to the landfill and gradually decreased towards the lagoon. This may be indicating the landfill as a possible source of metals in the sediments. However, some metals such as Cu did not show a general distribution pattern and this suggests that the landfill was not the only source of such metals but other potential sources close by such as ports and shipwrecks may also have contributed to the elevated levels. The levels of metals were lower at the control transect than the landfill transects except Hg.

The 2 widely used international sediment quality guidelines (SQGs) and reported concentration ranges for clean reef sediments were applied to evaluate the sediment contamination and potential eco-toxicological effects associated with the present concentrations at the Kiribati study site. All the contaminants of interest (As, Cr, Cd, Cu, Pb, Ni, and Zn) except Hg had concentrations below the Effects Range-Low (ERL) and the Threshold Effects Levels (TEL) indicating no potential adverse ecological effects on the local benthic communities. All metals measured (As, Cr, Cd, Cu, Pb, Ni, and Zn), except Hg, were within the ranges for the clean reef sediment for the control site. The landfill concentrations exceeded the published background ranges of Denton *et al.* (2001). The concentration of mercury exceeded the Effects Range-Median (ERM) and the Probable Effects Level (PEL) of the guidelines. The sediment is considered highly contaminated by mercury so may have adverse ecological impacts on benthic communities. However, the data of Naidu *et al*. (1991) showed Hg at low levels in shellfish from Teaoraereke, Bikenibeu and Eita. The

mercury concentration also far exceeded the reported range for clean reef sediments, indicating a contaminated sediment. The mercury concentration at the control transect and the landfill transects was similar. This suggests that the mercury in the lagoon sediments may have come from a source other than the landfill.

6.1.2 New Zealand

Sampling was carried out using transects of 5 m intervals starting from the seawall at the landfill and land at the control site. The top 15 cm sediments were collected using a PVC corer. The sediment pH was 6.22-7.24 which is typical of estuarine muddy sediments. The sediments comprised up to 90 % clay/silt. Sand was high near the seawall and at the surface and decreased towards the channel, and with depth. The same pattern was observed at both the landfill and the control transects.

Organic matter content was high (4.84 - 22 %) and there was a weak correlation between organic matter and particle size distribution at both sites. The organic matter content was generally high at the seawall, decreased towards the channel, and did not vary greatly with depth at either the landfill or the control site.

From the metals investigated, only Cd concentration was not detectable. At the landfill site some metals such as As had an elevated concentration close to the seawall. This probably indicates the landfill as a source. However, some of the metals like Cr did not show a pattern of distribution along the transects and with depth suggesting that the landfill not the only source of such metals. The metal concentrations at the control varied along the transect and down the profile. There was a weak correlation between organic matter content, particle size distribution, and metal enrichment.

The levels of metals both at the landfill and control site were similar suggesting that either the landfill was not the only source for these metals but the wider urban or industrial run-off may have contributed; or, the control site , though 2 km away, was too close and also received inputs from the landfill.

The Sediment Quality Guidelines (SQGs) of NOAA and the Florida Department of Environmental Protection, (Denton *et al.* 2001) (Table 5.1) were used to evaluate the sediment contamination. All the metal concentrations except Hg and Zn exceeded the Effects Range-Low (ERL) and the Threshold Effects Level (TEL) of the NOAA and Florida Department guidelines respectively. This indicates the potential for adverse ecological effects of metals on the benthic communities. Hg and Zn exceeded both the Effects Range-Median of the NOAA and Probable Effects Level of the Florida Department guidelines and consequently this is considered contaminated sediments by Hg and Zn.

The New Zealand study site is an excellent example of a zone of accumulation characterized by a high percentage of fine sediments, which was comprised up to 90 % clay/silt. Mangroves bordering the Taipari Strand act as efficient binders and traps of sediments (Leong *et al* n.d.). They also provide shelter from wave and tidal energy hence act as settling basins for suspended material. These conditions of low hydrodynamism, high sedimentation rate, dominance of silt and clay, which retain more pollutants due to the higher surface/volume, high organic matter content (Guerra-Garcia & Garcia-Gomez 2005), make Taipari Strand an ideal environment for metal deposition hence reflected in the elevated metal concentrations.

6.2 Errors and areas of improvement

A better way to collecting sediments samples down the profile for the Kiribati study site is warranted. The loose nature of the sediments and the low groundwater table (just a few cm depth) during low tide make it hard to extract the sediment samples from a particular depth. Therefore, the contamination of the lower profile was always likely when digging the sediments to collect samples. It would have been ideal to run 3 replicates for metal analysis for both study sites to get a better idea of the concentrations of the metals present in the sediments.

There is a need for better technology to measure the low concentrations of metals e.g. mercury and cadmium especially those that occur in low concentrations. This is of major concern because these metals are known to become toxic even in low concentrations and therefore it is necessary to know the concentrations present. The detection limit for Cd, Pb and Hg were above the suggested concentrations ranges for clean reef sediments from nearshore. It would be ideal to have the detection limit below the suggested ranges.

Due to the generally accepted relationship of clay/silt fraction, consisting of particles with a sediment size <0.063 mm and metals enrichment, analyzing <1 mm sediment fraction of the Kiribati samples in terms of sand, silt and clay fractions would be necessary to better understand the distribution of metals.

6.3 Conclusions and findings

The Kiribati study site was a good example of a transportation zone which was characterized by diverse sediments ranging from sand, silt and clay (<1 mm) to sand and gravels coarser than 5.6 mm and low organic matter content (1.60 - 2.21 %). The site has high hydrodynamic energy due to its exposure to tidal and coastal currents and therefore fine particulates associated with metals tend to move on. In contrast, the New Zealand study site had high percentage of fine sediments (comprised up to 90 % clay/silt) and high organic matter content of 5 - 22 %, a characteristic of areas of low hydrodynamic energy. Mangroves bordering the Taipari Strand landfill act as efficient traps of sediments and also provide shelter from wave and tidal energy hence act as settling basins for suspended material making it an excellent example of a zone of accumulation. Mangroves also contribute directly to organic matter input. These conditions of low hydrodynamism, high sedimentation rate, high clay/silt fraction, which retain more pollutants due to the higher surface/volume and high organic matter content, at the New Zealand study site, serve as a favorable environment for metal enrichment.

The ground water pH for the Kiribati study site was high (7.14 - 8.85), a typical pH for coral atoll sediments. High pH means decreasing in the solubility of metals by adsorption and precipitation and therefore metals become immobile. The sediment pH for the New Zealand study site was 6.22 - 7.24.

At the Kiribati sites Cd, As and Ni were below the detection limit. The Cr, Cu, Pb, and Zn concentrations were higher at the landfill than at the control transects. Cr level decreased away from the landfill indicating the landfill as a possible source. However, Cu and Zn did not show any distribution pattern suggesting other potential sources (port and shipwrecks) may have contributed to the elevated levels. Comparing the metal concentrations with the SQGs they were below the Effects Range-Low (ERL) and the Threshold Effects Levels (TEL) of the Sediment Quality Guidelines (SQGs) of the National Oceanographic and Atmospheric Administration (NOAA) and the Florida Department of Environmental Protection, indicating no potential adverse ecological effects on the benthic community. The levels of metals at the landfill transects were above the suggested background range for clean reef sediments. The metal concentrations at the control transect were within the background ranges. Hg exceeded both the Effects Range-Median (ERM) and the Probable Effects Level (PEL) values indicating potential adverse ecological effects on the benthic communities. High levels of mercury at both the control and landfill site suggested that the landfill was not the source of mercury.

At the New Zealand study site, only Cd was below the detection limit. At the New Zealand landfill transects, Arsenic concentrations decreased away from the landfill indicating the landfill as a possible source. Other metals such as Cr did not show a pattern of distribution along the transects, or with depth, suggesting that the landfill was not the only source of these metals. There was a weak correlation between organic matter content, particle size distribution, and metal enrichment. There were no marked differences in metal concentrations between the landfill and the control transects suggesting the landfill was not the only source of metals and that the wider urban or industrial run-off may have contributed. All the metal concentrations, except Hg and Zn, exceeded the ERL and the TEL values indicating the potential for adverse

ecological effects of metals on the benthic communities. At the New Zealand study site the Hg and Zn concentrations exceeded both the ERM and PEL values and are considered highly contaminated.

Given the low level of metals at the landfill transects at the Kiribati study site found in the present study, it is recommended not to spend money on the remediation but to improve the containment of the leachate with less permeable seawall until a better way to deal with the leachate is identified. The high percentage of organic matter in Kiribati can effectively mobilize metals by increasing their concentrations in the leachate. However, it is suspected that the high pH of the sediments enhances both sorption and precipitation of metals, resulting in the immobilization of metals in the adjacent sediments.

The proposal to hook up the leachate to the local sewer network needs to be reviewed thoroughly. Since the sewage system is not treated, discharging the leachate into the ocean simply means transferring the problem from the site to another. It would be easier to deal with the heavy metals at the landfill but once it is discharged through the outfalls it would escape into the ocean environment.

Environment assessments of the discharges from existing dumpsites had been identified by previous studies in the Pacific Region as one of the most significant areas where new or additional data is required. The findings of the present study helped to address this information gap for the region and for Kiribati in particular in terms of chemical contaminant sources in Kiribati in the marine environment.

6.4 Future direction for coastal contamination research in Kiribati (Recommendations)

On the basis of the work done in the present study, the following recommendations are made for future coastal contamination investigations in the Tarawa lagoon;

- A need exists for an immediate wider study to determine the cause of high mercury concentrations and to identify potential sources. Investigations should be undertaken to evaluate the bioaccumulation potential of mercury in the Tarawa lagoon. This will provide important data both from the public health and ecological point of view. Also necessary to identify promising bio-indicators for future monitoring programmes.
- Given that to date this is the first study to investigate metal contamination in the Tarawa lagoon sediments, there is a need to broaden the coverage of investigation to include the rest of the southern arm of the atoll. Emphasis should be given to those areas impacted or potentially impacted by domestic and industrial discharges.
- Need to establish baseline contaminant levels in sediments and biota for the cleaner relatively unimpacted stretches of coastlines on the northern arm of the island. This data is also needed for the Tarawa Lagoon Management Plan and without such information, the effects of future developments in the area will be difficult to assess.
- Use of appropriate Sediment Quality Guidelines (SQGs) for calcareous sediments. Metals are relatively weakly sorbed to biogenic carbonates and as a result, tend to be more biologically available. The SQGs in the present study used were developed in the U.S and cannot be expected to apply equally to all sediment types and sets of environmental conditions, and therefore, should be utilized for guideline purposes only. However, they are useful in providing a point of reference for the identification of contaminants and sites of potential concern, and are of value from the point of view of prioritizing actions and management decisions.
- There is a need to review the Kiribati Environmental Regulation guidelines and to develop a separate guideline specific for marine sediments appropriate for calcareous sediments.
- Need to expand the investigation to include other organic compounds including TBT, dioxins, the persistent organic pollutants (POPs) and the alkylated PAHs.

- Need to investigate the bioavailability of metals in the sediments. This will help to alleviate the problem to regulators of how to distinguish between background lithogenic trace metals that are relatively unavailable to marine biota.
- Regular monitoring and using longer sediment cores to determine pollution history and sedimentation of the sites.

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Appendices