CAWTHRON

Estuarine Environmental Assessment and Monitoring:

A National Protocol

Prepared for Supporting Councils and
the Ministry for the Environment
Sustainable Management Fund Contract No. 5096

Ministry for the Environment
Manatū Mō Te Taiao
Sustainable Management Fund

December 2002

Estuarine Environmental Assessment and Monitoring:

A National Protocol

by

Barry Robertson, Paul Gillespie, Rod Asher,
Sinnet Frisk, Nigel Keeley, Grant Hopkins,
Stephanie Thompson, and Ben Tuckey

Cawthron Institute
98 Halifax Street East
Private Bag 2
NELSON
NEW ZEALAND

Phone: +64.3.548.2319
Fax: +64.3.546.9464
Email: info@cawthron.org.nz

Report reviewed by:	Approved for release by:
Barrie Forrest Senior Coastal Scientist	Dr Barry Robertson Coastal and Estuarine Group- Manager

Recommended Citation:

Robertson, B.M.; Gillespie, P.A.; Asher, R.A.; Frisk, S.; Keeley, N.B.; Hopkins, G.A.; Thompson, S.J.; Tuckey, B.J. 2002. Estuarine Environmental Assessment and Monitoring: A National Protocol. Part A. Development, Part B. Appendices, and Part C. Application. Prepared for supporting Councils and the Ministry for the Environment, Sustainable Management Fund Contract No. 5096. Part A. 93p. Part B. 159p. Part C. 40p plus field sheets.

PREFACE

In July 1999, Cawthron Institute received support from 11 regional/local authorities and the Ministry for the Environment Sustainable Management Fund (SMF) to develop a standardised, cost-effective and defensible methodology for assessing and monitoring the condition of New Zealand estuaries. In order to achieve this, over a three-year project duration, a case-study approach was adopted. Coordinated surveys of eight New Zealand estuaries, representing different latitudes/ecotypes, were designed and carried out. The objective was to provide a robust database, suitable, both for addressing a variety of management questions, and developing the standardised National protocol.

The surveys combined an initial broad-scale mapping of the spatial distribution of intertidal habitats, followed by fine-scale analyses of one dominant habitat shared by all the reference estuaries; the sand/mud habitat at mid-low tidal elevation.

The Structure of the Report

The project output contains three integral components:

Part A: Development of the Monitoring Protocol for New Zealand Estuaries: Introduction, Rationale and Methodology.

This component provides background regarding the importance of estuaries, the problems often associated with their management, and why the project was undertaken. It describes the methods, rationale and the development of the Estuary Monitoring Protocol (EMP). Summary boxes are included at the end of each main section and numerous technical boxes (for the definition of scientific terms) are interspersed throughout the document. Our intention was to give readers the option of reading the document quickly and obtaining the "essence" of the summary information or delving into the detail of the results and protocol development.

Part B: Development of the Monitoring Protocol for New Zealand Estuaries: Appendices to the Introduction, Rationale and Methodology.

Many of the procedures involved with development of the EMP, and much of the data collected, are of a highly technical nature. For this reason, we have transferred much of the detailed information into a series of appendices with numerous cross references. The Appendices include individual estuary descriptions, details of the broad-scale classification system, results of individual estuary broad and fine-scale analyses, and fine-scale data processing methods. Background information about the individual reference estuaries varies in completeness depending on whether or not the information existed and whether or not we were able to access it. We encourage managers to add information, where possible, that may be relevant to assessment of estuarine condition.

Part C: Application of the Monitoring Protocol for New Zealand Estuaries

This document provides a condensed step by step guide for application of the monitoring protocol (where, what, how and when to monitor), based on the background, rationale and initial case study results described in Part A and Part B.

The "Living Document" concept

All three components of this report should be updated periodically. The individual estuary results provide potentially valuable datasets for managers that can be further evaluated and/or expanded as additional data becomes available. As the protocol is applied to additional estuaries, the expanded database will most likely extend the range of conditions comprising the continuum (pristine to highly modified). It will also extend the range of estuary types and habitat types compared. The expanded data sets will improve the interpretive value of assessment and monitoring surveys. The overall database was also designed to provide opportunities for future development of various indices of estuarine condition. For example, as the data base expands, the species and abundance of animal communities may be used to develop biotic indices, while physico-chemical characteristics could lead to development of companion indices (*e.g.* of nutrient enrichment). Examples of how this may be accomplished are provided in Gibson *et al* (2000) and Wilson and Jeffrey (1994). Similarly, methodologies can be refined/improved over time (*e.g.* taxonomic precision) and new tools may become available (*e.g.* satellite imagery, GIS software capabilities). Thus the Protocol and supporting data should be viewed as a "living document" that will improve with use and technological advancement.

Continued technical support

Cawthron's Coastal and Estuarine Group are dedicated to continued support of the National protocol initiative. In some instances, councils may wish to develop and carry out their own monitoring programmes with minimum consultation (*i.e.* advice only). In others, they may elect to contract some or all of the work to an independent science provider. Cawthron would be pleased to provide support in either capacity.

Estuarine Environmental Assessment and Monitoring:

A National Protocol

PART A:

Development of the Monitoring Protocol for New Zealand Estuaries:

Introduction, Rationale and Methodology

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1. INTRODUCTION

1.1 Background

An estuary can be defined as a partially enclosed body of water which is open to the sea (permanently or periodically) and within which there are variations in salinity due to the dilution of seawater with freshwater from land drainage (Pritchard 1967). Although estuaries are considered short-term features of the landscape on a geological timescale, they are often highly productive areas that play important roles at the boundary between land and sea. They provide a link between terrestrial and marine ecosystems and nourish the marine food web (Gillespie 1983). Due to their position at the foot of watersheds on the coastal interface, estuaries are dynamic, complex and variable environments. New Zealand estuaries, in particular, are generally characterised by extensive intertidal zones that provide productive, high-value habitat for a variety of plant (*e.g.* mangrove, salt marsh, eelgrass) and animal (*e.g.* fish, shellfish, waterfowl) species. Estuaries and their resources are also highly valued in human terms. They often provide transportation arteries and accessible locations for a wide variety of recreational pursuits. When properly managed, they can have high aesthetic/scenic values, particularly in populated areas, and commercial ecotourism use of estuaries is growing rapidly.


Waimea Estuary, New Zealand.

Globally, the coastal zone is under increasing pressure from human activities, and multi-use estuarine environments are reflecting the increase in human impacts by a modification, and sometimes, deterioration in their condition (Knox 1986). Because they are convenient receiving bodies for the wastes of cities, industries and farms, many New Zealand estuaries are considered to be at risk from contaminant impacts. Thus the development of management techniques, to assess estuarine habitat status and change, is currently a major resource management priority within

New Zealand. The localised effects of point source discharges (*e.g.* treated sewage, industrial wastewater, dairy and landfill effluents) have generally been adequately handled through consent procedures but managing and monitoring overall estuary condition, particularly for State of Environment (SOE) reporting, has largely been inadequate. Hence, the overall health of many New Zealand estuaries, and differences between estuaries subject to different pressures from human activities, is poorly understood. In part, this is attributable to the lack of a standard and affordable monitoring approach.

Monitoring within Regional Councils is generally classified as either consent monitoring or SOE monitoring. Consent monitoring is the relatively specific assessment of compliance with conditions relating to a resource consent. It is generally carried out at predetermined intervals after commencement of a development/activity. Monitoring results are compared with a pre-development baseline to assess change. SOE monitoring has a broader focus, is generally long-term and spread over a wide geographical area. It therefore provides the Council with a broad information base that is useful when considering the issues of individual resource consents and the sustainability of resources.

Successful management of estuaries and their catchments for sustainable use in the future requires us to focus our knowledge on developing simple, defensible and cost-effective strategies to assess and monitor estuary condition and predict the results of management actions. However, despite the large extent of research on estuaries, our ability to predict the consequences of change or even develop a set of cost-effective monitoring indicators of estuary condition is limited. Reasons for this include a lack of funding to determine these monitoring indicators, the complexity of estuaries, and the fact that most research has focused on local estuary problems and is difficult to transfer to other sites. The current study provides an opportunity to rectify this situation.

In 1999 the Cawthron Institute received support from eleven New Zealand regional/local authorities and the Ministry for the Environment (MfE) Sustainable Management Fund (SMF) to initiate the development of a defensible and cost-effective monitoring protocol to assess the condition of New Zealand estuaries. The primary aims of the study were to: (1) design and undertake baseline monitoring of a suite of characteristics in nine New Zealand estuaries representing a wide range of latitudes, (2) evaluate these characteristics as potential indicators of estuarine environmental health, and (3) use the results to develop a standardised monitoring protocol. Three potential assessment tools, representing different scales of investigation, were developed to form the Estuary Monitoring Protocol (EMP):

1. **Preliminary assessment of estuary condition:** development of a decision matrix that allows managers to prioritise estuaries for monitoring and provide a defensible basis for their long-term planning decisions.
2. **Broad-scale habitat mapping:** development of a robust GIS-based methodology for mapping the spatial distribution of intertidal estuarine habitats.
3. **Fine-scale environmental monitoring:** development of a methodology to measure the spatial variation and inter-relationships of a suite of commonly measured indicators.

As a management tool, the value of the standardised monitoring approach we propose (particularly the ability to assess change) will evolve as the database expands through application of the approach to additional estuaries or through repeated monitoring in the same estuaries. The assessment parameters used for monitoring, or a subset thereof, may emerge as a suitable suite of environmental performance indicators for estuarine habitats. The final result will be a cost-effective and defensible estuary monitoring protocol that is primarily designed to help meet coastal managers' requirements for SOE-type monitoring, but will also be useful in the design of consent monitoring programmes.

Consent monitoring often lacks the broader perspective provided by estuary-scale or inter-estuary comparison. SOE monitoring programmes generally focus on 'representative' rather than localised, high impact sites. Linking the two through the application of consistent or comparable methodologies will benefit both greatly. Consent monitoring surveys will obtain the broader interpretive base necessary for assessing the significance of localised impacts (refer Technical Box 1.1 for monitoring definitions). Applying the methodology to high impact situations will extend the range of conditions for assessment of estuarine health.

The protocol development described here represents a 'first step' in this direction, and as the database expands, it will become more and more valuable. It is anticipated that additional work will be required to ensure that its full potential is achieved. This work should include:

- testing the optimised sampling design on a wider range of New Zealand estuaries,
- studies in temporal variability,

- extending the fine-scale approach to other intertidal habitats (*e.g.* eelgrass beds, saltmarsh, mangroves),
- extending the protocol to include subtidal habitats and,
- developing guideline standards against which measured benthic characteristics can be compared and the estuary condition categorised (*e.g.* enrichment indices, biotic indices).

Technical Box 1.1: Monitoring definitions

Consent monitoring: monitoring carried out in accordance with conditions pertaining to a resource consent; *e.g.* to determine the effects of a wastewater discharge.

State of the environment (SOE) monitoring: long-term monitoring carried out at sites representative of a region; *e.g.* to identify bay or estuary-scale changes.

2. STUDY RATIONALE

It is impractical and cost-prohibitive to undertake intensive and long-term monitoring of all the types of estuaries and their habitats in New Zealand. Therefore it was necessary to select a manageable number of estuaries, habitats and indicators based on physical and practical considerations, and then develop monitoring approaches that can have a wide application to similar estuary types.

A summary flow diagram of the key decisions and processes involved in the development of the estuary monitoring protocol (EMP) is presented in Figure 1. A brief discussion of the five key stages shown in Figure 1 is given below.

2.1 Estuary selection in the current study

Eight New Zealand estuaries were selected to develop initial comparative data sets, in order to trial the monitoring techniques and develop the monitoring protocol. The estuaries were chosen based on Council nomination and support, geographical location (*i.e.* latitude) and specific issues that were involved. Each reference estuary has been attributed high value by the respective regulatory agencies and interest organisations. All are likely candidates for inclusion (or are already included) in long-term SOE monitoring programmes. Those chosen represent the most common estuary types found in New Zealand in terms of size, how they were formed (*i.e.* geomorphology), flushing time, catchment landuse, catchment area, freshwater and marine inputs, catchment contaminant loadings, and resource uses and values. Funding for one of the case study locations, Whangamata Estuary, was discontinued after completion of the broad-scale mapping component, however an additional location, Ruataniwha Estuary, was subsequently added. The locations of the nine reference estuaries are summarised in Figure 2 and Table 1.

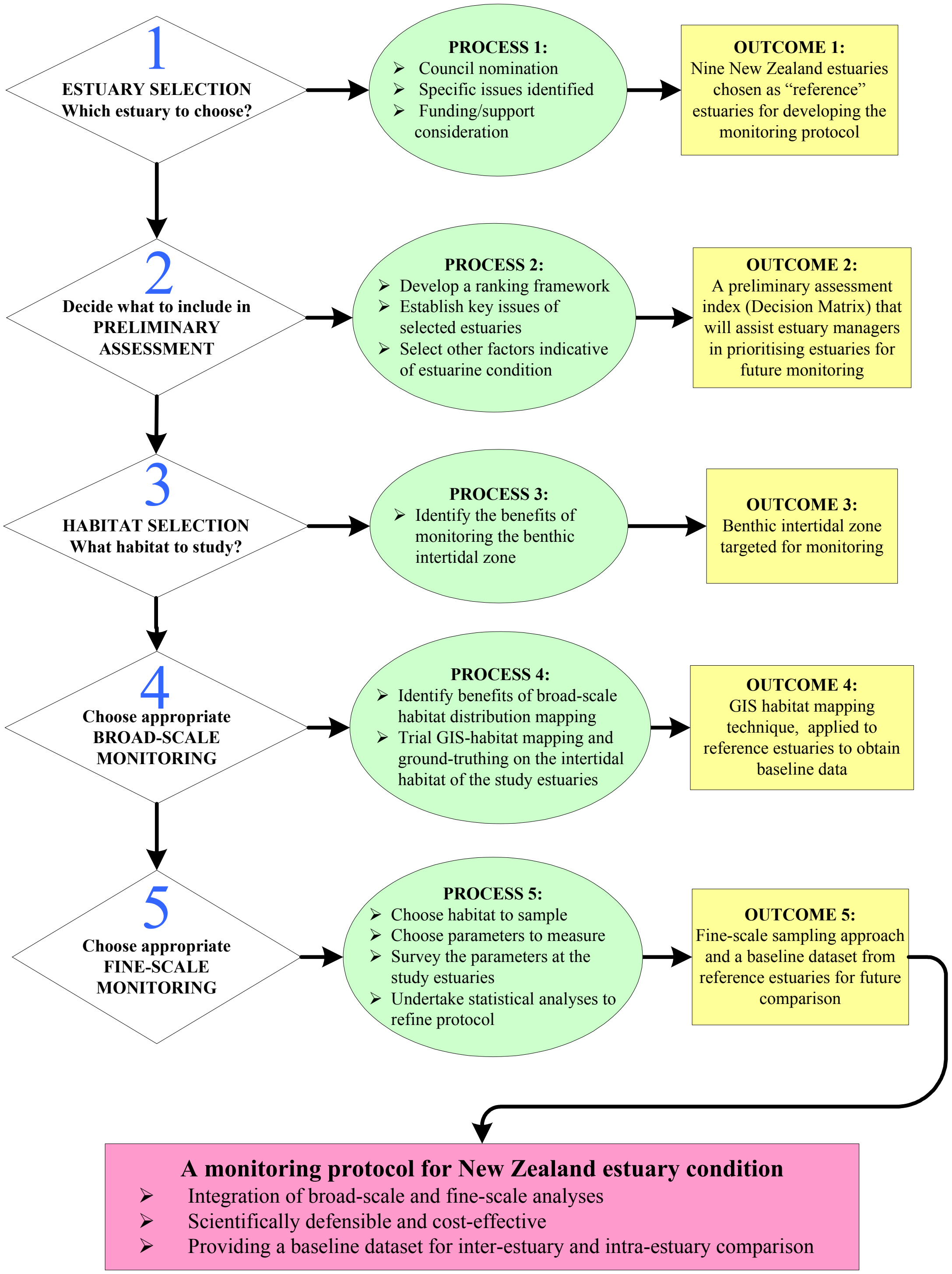


Figure 1: Key decisions and processes in the development of the Estuary Monitoring Protocol

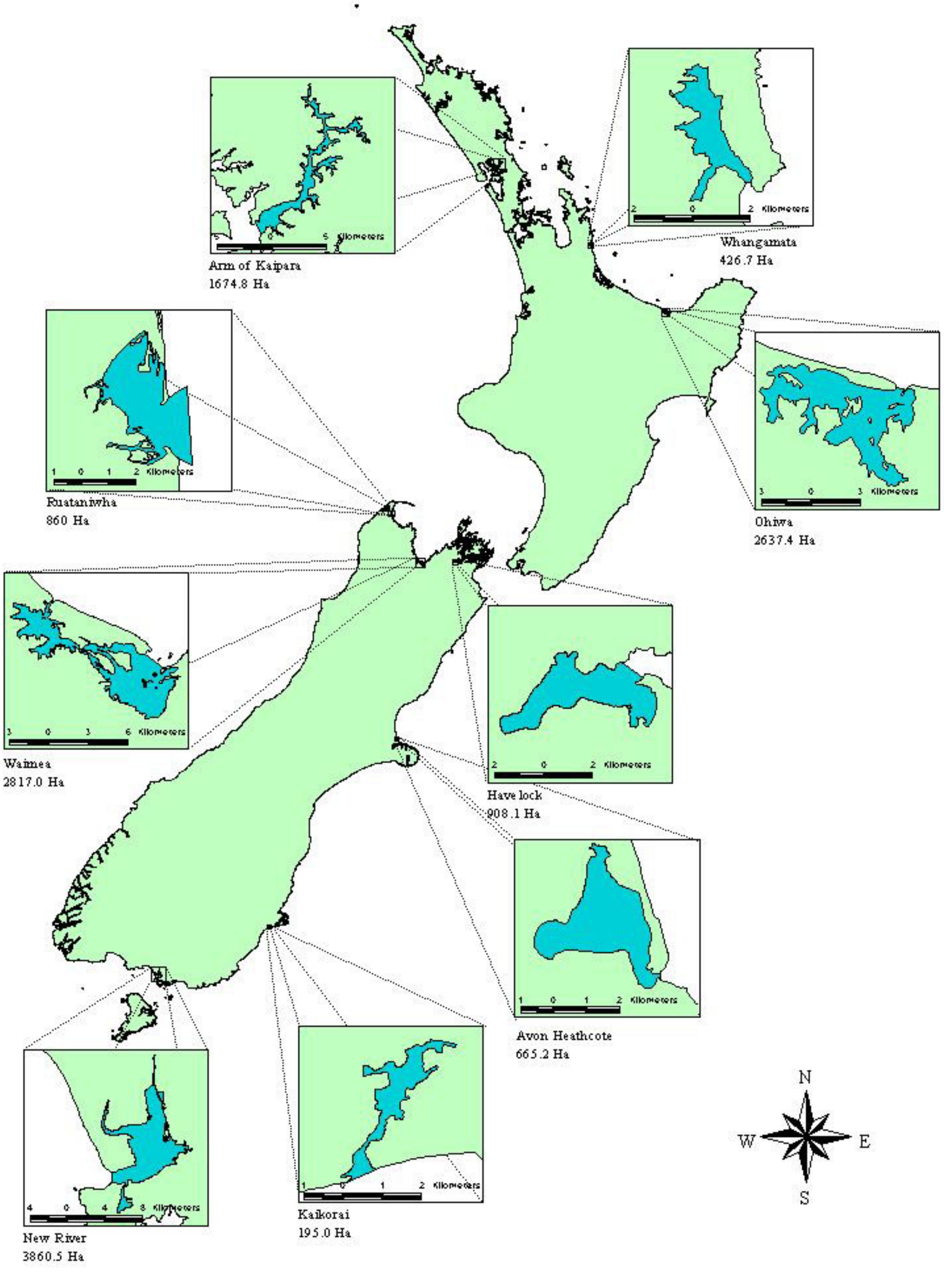


Figure 2: Locations of the nine selected estuaries with expanded inserts showing a magnified view of each estuary.

Table 1: Estuary locations and associated Councils

Estuary	**Location**	**Associated Councils**
Otamatea Arm (Kaipara Harbour)	Northland	NRC
Ohiwa	Bay of Plenty	EBOP
Whangamata[1]	Coromandel	EW
Ruataniwha[2]	Golden Bay	TDC
Waimea	Nelson	TDC, NCC, NRSBU
Havelock	Havelock	MDC
Avon-Heathcote	Christchurch	EC, CCC
Kaikorai	Dunedin	ORC
New River	Invercargill	SRC

[1] The Whangamata Estuary study discontinued after the broad-scale habitat mapping at the request of EW.
[2] The Ruataniwha Estuary was added in 2001 at the request of the TDC.

2.2 What the Councils wanted

The initial phase of the study involved meeting with Regional Council staff and other interested parties from each of the nine reference estuary areas, to ascertain what they perceived were the major issues affecting the condition of the estuary within their region. The key issues that were raised by Council staff and interested parties are summarised in Table 2. This phase also served as a preliminary familiarisation visit and as an opportunity to collect any relevant background information on each of the estuaries. The information was then used to help characterise the estuaries and aid in further refining the monitoring program, particularly in the development of relevant estuary characteristics that would form the initial ranking framework of the preliminary assessment protocol (Section 2.3).

Table 2: Key potential issues relating to estuarine condition for the nine reference estuaries

	Otamatea Arm	**Whangamata**	**Ohiwa**	**Ruataniwha**	**Waimea**	**Havelock**	**Avon-Heathcote**	**Kaikorai**	**New River**
Sedimentation	X	X	X	X	X	X	X	X	X
Nutrient enrichment	X	X	X	X	X	X	X	X	X
Shellfish health/condition	X	X	X		X	X	X		X
Extent of sediment toxicity					X			X	X
Waterborne disease risk		X		X			X		
Expansion of mangrove habitat		X							
Threat of Pacific oyster invasion/spread	X			X	X	X			
Spartina invasion	X					X	X		X
Effects of *Spartina* eradication					X				
Vegetation health/condition				X		X	X	X	X
Ecological status		X	X	X	X	X	X	X	X
Potential for rehabilitation								X	
Wastewater discharge impacts	X			X	X		X	X	X
Reclamation		X		X					X
Influence of residential development			X				X		
Influence of rural development			X	X					
Impacts of boating activities		X				X			

A number of key requirements relating to the monitoring of estuarine condition were identified from these discussions (listed in Table 3 as a 'Council Wish List'). Most of these arose in relation to the Councils' need for a cost-effective and defensible estuary monitoring programme to fit within their SOE requirements. Nine of the 15 requirements identified (Table 3) were addressed during the present study, however two of those have only been partly achieved. Achievement of No. 1, "the ideal tool", will require further incorporation of fine-scale assessment of other habitat types and an expanded comparative data base for assessment of change over time. Although the protocol addresses issues of enrichment status (No. 15) and wastewater discharge (No. 9) impacts in a general sense, the field assessments did not target high impact sites. However, the protocol will provide a context for focussed environmental impact assessments concerning those issues.

Table 3: List of key requirements relating to monitoring of estuary condition (not necessarily in order of perceived importance). +, +/-, and – refer to those requirements that were addressed, partially addressed, and not addressed in the present EMP, respectively.

Council Wish List	Source	This Study
1. A rapid and cost-effective monitoring tool that i) gives a defensible indication of whole estuary condition that is readily transferable to similar types of estuaries, and ii) identifies condition of priority habitats in NZ estuaries. It should be robust enough to be able to measure change over approximately 5 year intervals (*i.e.* the ideal monitoring tool).	EW, EBOP, NRC, TDC, MDC, EC, ORC, ES.	+/-
2. A standardised methodology for mapping and classifying dominant habitats in estuaries (with minimum error in habitat boundary definition)	EW, EBOP, NRC, TDC, MDC, EC, ORC, ES	+
3. A standardised methodology for mapping the extent of mud intrusion (*i.e.* is mud habitat expanding, contracting?)	EW, EBOP, NRC, TDC, MDC, EC, ORC, ES	+
4. A defensible means of choosing which estuaries to monitor in a region.	ES, ORC	+
5. A standardised methodology for assessing mud habitat condition (ecological health, biodiversity, contamination)	EW, EBOP, NRC, TDC, MDC, EC, ORC, ES	+
6. A standardised methodology for assessing mangrove habitat condition (a cost effective indicator)	EW, NRC, EBOP	-
7. A standardised methodology for mapping mangrove habitat expansion	EW, NRC, EBOP	+
8. Assessing whether seagrass habitat is a reliable indicator of overall estuary condition	EW	-
9. A rapid monitoring tool to assess the impact of wastewater discharges on overall estuary condition (issues include metal toxicity, stormwater, treated sewage, industrial)	NRC, TDC, EC, ORC, ES, CCC, NRSA, NCC	+/-
10. A standardised methodology for mapping the habitat of the Pacific oyster (*Crassostrea gigas*)	NRC	+
11. A standardised methodology for assessing shellfish habitat loss and condition of existing beds	Iwi (Whangamata), ES	-
12. A standardised methodology for assessing condition of vegetation habitat (a standardised scale)	EBOP	-
13. Practical guidance on vegetation rehabilitation	ORC	-
14. A rapid monitoring tool to assess the waterborne pathogen risk to humans using the estuary	EC, ES, Whangamata community	-
15. A rapid monitoring tool to assess the enrichment status of an estuary	All	+/-

2.3 Preliminary estuary assessment

The current study aimed to design a preliminary assessment tool that combined a ‘decision matrix’ of a wide range of estuarine characteristics that would be ranked for each estuary within a region. Indices of estuarine condition are valuable as they condense a broad range of often complicated information into a simple, comprehensible index. This ranking system would allow estuaries to be prioritised for long-term environmental monitoring, and provide a fast, simplified way of comparing and evaluating estuaries that is easily communicated to coastal managers and the public. The key decisions and rationale for this approach are outlined in Table 4.

Table 4: Rationale for the preliminary assessment protocol

Key Design Decisions	Reasons
1. Decision Matrix Utility It was decided that the study should develop an index of estuary condition as the initial part of the estuary monitoring protocol.	• To assist coastal managers in the initial decision of which estuaries to prioritise in their region. • Preliminary assessment ranking is a tool that condenses a large amount of information into a simple, comprehensible index. • There needs to be a rapid, first-cut assessment of environmental conditions and relevant issues of estuaries in a region. • A decision matrix provides a holistic, multidisciplinary framework from which preliminary management decisions can be made. • A decision matrix can be revisited periodically for adjustment regarding changing values/issues/priorities, and for evaluating the effectiveness of management decisions.
2. Select Estuary Characteristics to form the Decision Matrix Consultation with Councils and interest groups provided a number of key issues relating to estuaries. Other important physical and biological characteristics and indicators of condition were included as assessment factors to be ranked.	• It was considered important to include current, relevant issues to New Zealand estuaries as some of the ranking criteria. • Priority was given to features that relied on current knowledge, historical data or easily accessible information. • Effort was made to select characteristics that did not require intensive sampling. • A broad range of characteristics were chosen in order to encompass the biological, physical and aesthetic factors as well as catchment processes and risk assessment. • The matrix covered the complete range of estuarine habitats, rather than focussing on a single habitat like other monitoring procedures.
3. Include Weighting Factors A further choice was made to include weighting factors that could be assigned to each estuary characteristic to reflect the important issues and concerns of the region.	• Weighting factors allow the decision matrix to be personalised by the end-user, as they can allocate relative weightings to particular estuary features perceived as more important to determining management decisions.

2.4 Focus on the benthic intertidal habitat

It is important to define and standardise the estuarine habitat targeted for monitoring for a number of reasons. The physical characteristics of sediment (*e.g.* grain size, sediment type) are often identified as important factors governing community structure and distribution of benthic fauna (Probert 1984). Additionally, the physical characteristics of a habitat can influence the relative accumulation of contaminants such as heavy metals (Kennish 1997, Koppelman & Dillard 1975).

Therefore, confining the monitoring to a single, characteristic estuarine habitat allows a simplified, cost-effective approach that avoids the confounding problems of comparing between different habitat types.

The bed of the estuary in the intertidal zone was chosen as the most appropriate area for monitoring, and the rationale for this selection is outlined in Table 5. Due to the large dilution effect of tidal flushing in most New Zealand estuaries, studies of surface waters are often not particularly useful as indicators of estuarine condition (Updegraff *et al*. 1977). Benthic characteristics, however, are more stable (Turner *et al*. 1995) and are generally considered to be better integrators of condition in well-flushed estuaries that may have been periodically exposed to contaminants over a long period of time (Gillespie & MacKenzie 1990, Roper *et al.* 1988). Thus it was decided to develop and test a monitoring protocol for New Zealand estuaries based on comparison of benthic characteristics.

Table 5: Rationale for choosing the benthic intertidal habitat for estuary monitoring.

Key Design Decisions	Reasons
1. Intertidal Habitat Within the nine estuaries it was decided that the study would concentrate on intertidal areas only.	• Intertidal habitats are known to be functionally important ('high value') components of coastal ecosystems. • They are the most accessible. • Data can be obtained quickly and more cost-effectively than subtidal habitat. • They are particularly vulnerable to human impact.
2. Benthic Intertidal Habitat A further choice was made to limit the assessment to the benthic intertidal environment.	• There is evidence that various contaminants, including nutrients, organic matter, metals, synthetic organic toxicants and pathogens, accumulate in estuarine sediments at greater concentrations than the overlying water. As such, estuarine sediments serve as sinks for contaminants with the potential to affect benthic communities and overlying water quality (Church 1975). • Benthic plants and animals are intimately involved in the wider estuary ecosystem through the food web and as agents of bioturbation and nutrient regeneration (Bilyard 1987). • Benthic organisms often contribute to the bioaccumulation of pollutants in estuarine food webs, especially heavy metals. • Compared with overlying water (which exhibits large short term fluctuations in physical, chemical and biological characteristics), the benthic environment is much more stable as it effectively integrates these fluctuations over time. • In shallow, well-flushed, bar-built estuaries that are typical of NZ, it is more promising to characterise the sediments rather than overlying waters to determine the enrichment and toxicity status of estuaries. • Existing consent monitoring information provides useful comparison for benthic habitats.

2.5 Broad-scale intertidal habitat mapping

The next step in developing the EMP was to establish a suitable method for monitoring the dominant characteristics of estuarine benthic intertidal habitat at a broad scale. It was decided that this could be best achieved by developing a standardised approach for classifying and mapping dominant vegetation and unvegetated areas of intertidal habitat. The rationale for this approach is outlined in Table 6.

Table 6: Rationale for broad-scale mapping of intertidal habitat

Key Design Decisions	Reasons
1. Map Intertidal Habitat Develop a method for mapping the distribution of estuarine intertidal habitat and test this on at least selected portions of estuaries. For each estuary, aerial photographs will be used to identify habitat and vegetation at a broad-scale. Digital maps will then be produced from the aerial photographs and verified with field studies to provide baselines for historical comparison, and for detecting change over different spatio-temporal scales.	• To develop a methodology and provide baseline information on the spatial distribution of broad habitat groupings within each estuary. • To provide an overview of habitat distribution in an estuary as a framework for risk assessment and design of subsequent finer scale monitoring programmes. • To allow similar habitats within different estuaries to be compared in general terms. • To help provide a broad picture of the key productive components and ecological processes associated with each estuary. • To progress MfE's confirmed environmental performance indicator ME6 – Percentage change in extent of selected marine habitats.
2. Coordinate with Classification Group In the process of the broad mapping of intertidal habitats, it was decided to coordinate the procedures, techniques and outcome with the general approach being adopted in the SMF funded "Coordinated Monitoring of New Zealand Wetlands".	• Many issues were similar between the two SMF projects. • The wetlands SMF project is producing a national classification system based on the Atkinson system (Atkinson 1985), a defensible, simple and cost-effective approach. • Ensures the use of one National approach. • Expertise shared and overlap avoided.

2.6 Fine-scale benthic intertidal monitoring

The next step in developing the EMP was to choose an appropriate intertidal benthic habitat that would reflect important aspects of overall estuary condition, and select fine-scale variables to monitor. Fine-scale monitoring measures the variation and inter-relationships of a suite of benthic indicators in a habitat that is likely to reflect current estuarine condition and subsequent changes. It represents the appropriate scale to investigate aspects of estuarine health, such as biodiversity, contamination, toxicity and enrichment. The rationale for the fine-scale monitoring approach is outlined in Table 7.

Table 7: Rationale for fine-scale assessment of intertidal habitat

Key Design Decisions	Reasons
Step 1 Choose a commonly impacted estuarine intertidal habitat. A decision was made to target one commonly impacted intertidal habitat. Muddy sand habitat in the mid-low tidal range was selected; *i.e.* where mean salinities of overlying water were greater than 20 ppt,.	• Mud/sand habitat is a common in New Zealand estuaries. • To build on other successful long-term estuary monitoring programmes in use (*e.g.* Manukau and Bay of Plenty estuaries). • Fine-grained (muddy) sediments are characteristic of sedimentary environments. These environments are known to be depositories of particulate contaminants (*e.g.* particle bound organic materials, nutrients, metals, *etc*). Fine-grained sediments can also take up dissolved contaminants from the water column, either by physical sorption or biological uptake. • Mid to low tidal elevations are inundated with potentially contaminated overlying water for a larger percentage of time than higher elevations. • The biology of the mid-low tide area is frequently more diverse and abundant and therefore has a high potential for indicating change. • Biological characteristics of habitats of widely variant salinities are not directly comparable.
Step 2 Choose a suite of benthic characteristics that have potential as indicators of estuary condition **Physical and Chemical** • Grain size • Nutrients (nitrogen and phosphorus) • Organic matter (AFDW) • Depth of redox discontinuity layer (RDL) • Trace metals (copper, chromium, cadmium, lead , nickel, zinc) **Biological** • Species abundance for both infauna and epifauna • Chlorophyll *a* as an indicator of micro-algal mat cover. • Microalgal species dominance	• Common issues associated with estuary condition in New Zealand are: muddiness, nutrient and organic enrichment, clarity, toxicity and human waterborne disease risk. Waterborne disease is best monitored through water column and shellfish monitoring. The other issues are addressed in the proposed suite of benthic indicators. • The selected indicators will reflect the types of human-induced pressures that commonly affect many New Zealand estuaries. • The chosen physical and chemical indicators integrate point in time conditions related to past history of exposure, and therefore require a low frequency of monitoring. • Nutrients, organic matter and heavy metals tend to sorb to small sediment particles and settle onto muddy areas of estuaries. Grain size analysis will provide a record of the relative proportion of fine grained sediments. • The structure of the biological community living on and within the sediment is affected by the condition of both the surrounding sediment and the overlying water.
Step 3 Design and undertake a survey to establish the spatial distribution of these benthic characteristics in the chosen habitat for all the reference estuaries.	• Provide a statistically robust benchmark so that the data can be used to optimise sampling design (precision versus effort), identify differences between estuaries and identify potential benthic indicators. • There have been various low cost indicator studies undertaken on overseas estuaries using minimal replication. However, such studies are unsuitable for general application. In particular, they lack the ability, in terms of sampling design, to adequately account for spatial and temporal distributions. • The general sampling design and methodology is expected to be readily transferable to other intertidal estuarine habitats. • This is an important step in developing a link between mudflat physical and chemical variables and biota distribution in a range of NZ estuaries.
Step 4 Undertake optimisation analyses to balance precision against effort for each of the variables and produce a defensible cost-effective monitoring programme, including the identification of any necessary further work (*e.g.* temporal studies).	• The baseline study will be too costly for widespread use by Councils. • It is expected that sufficient data to establish the general condition of an estuarine habitat can be generated from fewer replicates and less variables than used in the baseline study. This will be achieved primarily by determining the influence of physical and chemical variables on biological species, feeding guilds, abundance and biomass data.

3. ESTUARY CHARACTERISTICS

This section outlines the general characteristics of New Zealand estuaries and how these align with the characteristics of the nine reference estuaries (REs).

3.1 General characteristics of New Zealand estuaries

New Zealand has over 300 estuaries (McLay *et al*. 1975) that vary in size from small coastal creeks and lagoons (as small as 1 ha) to large harbours, sounds and fiords (up to > 15,000 ha), but a majority of those identified are > 500 ha.

New Zealand estuaries have developed in a variety of geomorphological situations. These include coastal former river or glacial valleys, coastal plains, rocky shores (fiords) and tectonically active areas. The majority have developed as incised valleys eroded by river and glacier action during the late glacial maximum. These were filled with water as the sea level rose approximately 10,000 years ago. Subsequently, they began to accumulate deposits and have been doing so ever since. The rate of infilling will vary among estuaries, depending on factors such as wave, tide and river flow energies, biological characteristics and input sediment loads. This infilling has often been encouraged by bar or spit formation near the estuary mouth. A survey of New Zealand estuaries (McLay *et al*. 1975) identified 78% as either bar-built estuaries or lagoon environments.

Extensive sand/mud flats at low tide on Bell Island in Waimea Estuary, Nelson.

Because the majority of New Zealand rivers carry high sediment loads and enter the ocean in broad, coastal plain areas, their estuaries are wide and relatively shallow. This means that they are largely drained at low water exposing extensive areas of sand/mud flats and peripheral salt marsh. The combination of the rapid tidal flushing, the broad expanse of intertidal area, and wind and wave turbulence generally results in most of the input sediments and contaminants being flushed out to sea.

The remainder is dispersed relatively widely within the estuary, but accumulates in localised areas of poor flushing. Despite favourable flushing characteristics, sediment deposition (probably accelerated due to past landcover clearance/disturbance and wetland drainage) has caused the majority of New Zealand's estuaries to become progressively muddier. Over the last century, the depositing sediments have, in many cases, become contaminated with elevated nutrients, organic matter, potentially disease-causing organisms and potentially toxic chemicals (*e.g.* metals and hydrocarbons), as the inputs of domestic, industrial and agricultural wastewaters, landfill leachates and stormwater have increased.

Catchment landuse, both past and present, is intimately linked with estuary condition (Dauer *et al.* 2000, Harris 2001). Within New Zealand, most of the land is used for agriculture and exotic and native forestry, but around some estuaries urban development can be particularly intense (*e.g.* Christchurch, Auckland, Wellington and Dunedin). Because human settlements are often concentrated in coastal regions and attracted by coastal resources, sediment and contaminant loadings to estuaries from urban and agricultural catchments are frequently elevated.

The tidal, sheltered waters and sediments of New Zealand estuaries support diverse communities of plants and animals, specially adapted for life at the land/sea interface (Bradstock 1985). These estuarine environments can be highly productive (Knox 1986). They contain a wide range of different habitats, including shallow open water and/or tidal pools, salt marsh, sandy and rocky shore, mud and sand flats, biogenic reefs (*e.g.* reefs containing oysters, polychaete worms or mussels), mangrove forests, sea grass and kelp beds. Freshwater wetlands and offshore delta regions often adjoin estuaries on the land and seaward ends, respectively. Over the past 150 years, many of these habitats have changed as a result of human activities (*e.g.* infilling, wetland drainage, exotic infestations, grazing, dredging and fishing).

Mangroves in Whangamata Estuary, Coromandel

3.2 Characteristics of the reference estuaries

The nine REs chosen for the study (refer Section 2.1) were classed according to the nomenclature of Hume & Herdendorf (1988). Background information describing each of the REs is provided in Appendix A (Part B of this report). This information is summarised and compared in Sections 3.2.1-3.2.7 in order to demonstrate their similarities/contrasts with respect to the present study.

3.2.1 Estuary location, shape and hydraulics

The REs are located throughout New Zealand from New River, in the lower South Island, to the Otamatea Arm of the Kaipara Harbour in Northland. The estuaries vary in size from relatively small (200 ha for the Kaikorai Estuary) to large (3500 ha for the New River Estuary).

All the REs were formed when the basin was originally cut by river action, generally when sea level was lower than at present. The landform has since been inundated by a rise in sea level, and modified by sediment deposition of both fluvial and marine origin. The majority of the estuaries have barrier spits or islands near their mouths. In almost all cases, the barrier provides no major restriction to drainage from the estuary to the sea. The exception is the Kaikorai Estuary where a barrier beach restricts the ponded drainage of the relatively small input streams to the sea. The Kaikorai estuary is 'perched,' in that the sea crosses the barrier beach only near high tide resulting in a lagoonal situation with restricted tidal exchange. Often the barrier beach builds up and prevents drainage to the sea, or tidal input of marine water, until the barrier is purposely breached by machinery. Such lagoonal estuaries are common and are generally important for wildlife and recreation, but they often present flooding problems for surrounding low-lying land. They are also more prone to nutrient enrichment and/or contaminant build-up. The Havelock estuary is the only RE enclosed by a headland rather than a barrier spit.

All the REs are shallow (mean depths <2m at high tide) and most are well-flushed. Although residence time has not been accurately measured for most of these estuaries, the fact that a large proportion of their water volume drains out on each tidal cycle indicates that residence times are likely to be within the 0.5 to 5 day range. The exception is the Kaikorai, whose residence time will vary depending on the extent to which the mouth is blocked at any time. The estuaries are generally expected to be well-mixed systems with little stratification outside localised freshwater discharge zones.

3.2.2 Catchment geology and landuse

The New River Estuary has the largest catchment (350,000 ha) and the Whangamata and Kaikorai have the smallest (5,188 and 5,467 ha, respectively). Geologically the catchments are relatively diverse, including the muddy limestones and mudstones that overlay basement rock in the Kaipara Harbour catchment (causing the harbour arms to be especially muddy), the recent loess deposits of the catchments of the Avon-Heathcote, Kaikorai and New River estuaries, and the volcanic influenced soils of the Ohiwa Estuary catchment.

Catchment development varies among the estuaries, but all include agriculture as a key landuse category (Table 8). The Otamatea Arm of the Kaipara Harbour and the New River estuaries have the greatest proportion of agricultural development (79 and 65% respectively). Intensive use of part of the catchment for urban development has occurred, to the greatest extent, in the Avon-Heathcote (56%), but urbanisation is also significant in the Kaikorai, Waimea, and New River estuary catchments. The REs also vary in the extent of undeveloped land within their catchments.

Table 8: Summary of catchment landuses for the nine reference estuaries

Land Use	**Avon-Heathcote**	**Havelock**	**Kaikorai**	**Kaipara (Otamatea Arm)**	**New River**	**Ohiwa**	**Ruataniwha**	**Waimea**	**Whangamata**
Percent Cover (%)[1]									
Indigenous forest	0.1	65.2	7.8	7.3	12.0	25.2	65.6	31.2	33.4
Prime pastoral	26.9	12.9	53.5	79.8	64.5	48.8	11.3	25.6	15.4
Planted forest	3.1	12.3	4.8	5.9	5.7	18.4	0.1	31.9	46.4
Scrub	1.5	8.3	9.5	6.0	4.4	5.2	14.7	4.9	1.4
Bare ground	0.1	0.5	1.1	0.0	2.8	0.1	0.7	0.7	0.0
Tussock	3.1	0.3	0.3	0.0	9.5	0.0	7.0	3.0	0.0
Inland water	0.0	0.2	0.0	0.2	0.1	0.0	0.5	0.0	0.0
Coastal wetlands	0.0	0.1	2.1	0.0	0.0	0.8	0.0	0.1	0.0
Inland wetlands	0.0	0.1	0.2	0.0	0.7	0.0	0.0	0.0	0.0
Urban	56.0	0.1	0.0	0.1	0.2	1.1	0.0	0.8	3.4
Urban open space	6.5	0.0	20.8	0.0	0.0	0.3	0.0	0.1	0.0
Prime horticultural	1.0	0.0	0.0	0.2	0.0	0.2	0.0	1.8	0.0
Inland water	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mangrove	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0
Total catchment area (ha)[2]	18783	104660	5468	61414	350289	18559	76681	81170	5188
Population[3]	280000	549	13999	1263	61491	500	120	7023	3700[4]

---

[1] Source: Land Cover Database 2001 (LCDB1)

[2] Source: Regional and local Council's

[3] Source: Census data

[4] Current residential population but increases to 20,000 over the Christmas/New Year period.

Indigenous forest land cover varies from 65% for the more isolated (and therefore relatively pristine) Ruataniwha and Havelock estuaries to 0.1 % for the urban-dominated Avon-Heathcote (Land Cover Database 2001).

3.2.3 Freshwater inflows

Although all the REs are dominated by marine inflow as the major source of water, some of the estuaries have relatively large freshwater inputs (*e.g.* Ruataniwha, New River and Havelock). This causes a diluting effect in these estuaries which becomes more noticeable with decreasing estuary area (Figure 3) and can be further emphasised during flood events. Other estuaries have very small inputs of freshwater and consequently are not influenced by large salinity variations (*e.g.* Ohiwa, Whangamata and Otamatea Arm, Kaipara Estuary). Figure 3 shows that the Havelock and Ruataniwha Estuaries have the smallest area to freshwater inflow ratio, and the Ohiwa Estuary has the greatest. Once again the Kaikorai is an outlier. Although it receives a relatively low freshwater input, it experiences low salinity (brackish) conditions for extended periods due to restricted tidal exchange.

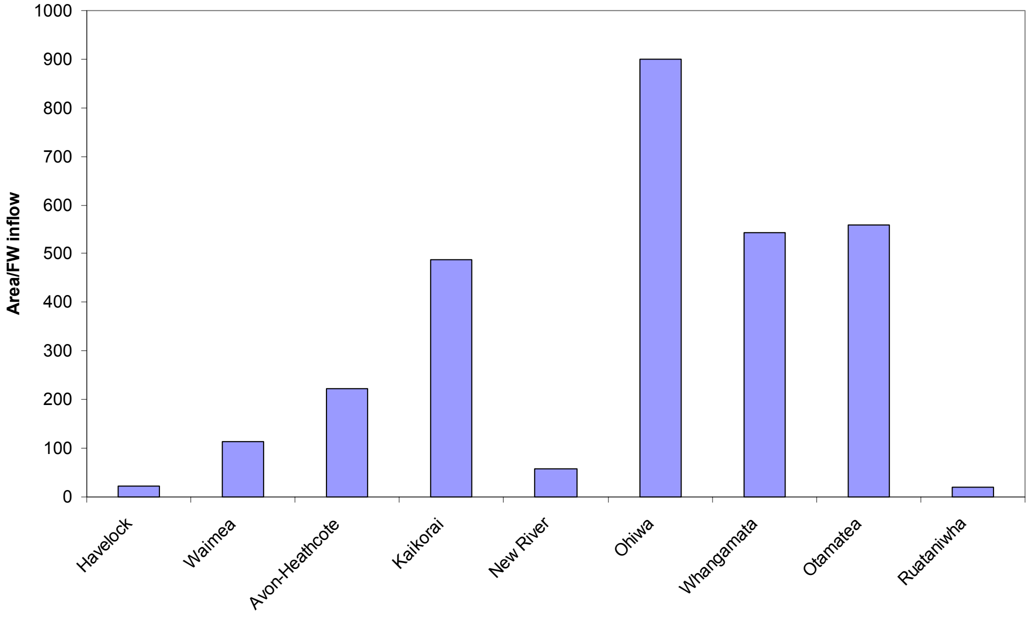


Figure 3: Ratio of estimated estuary area (ha) to mean annual freshwater inflow (m^3/s) for the REs

3.2.4 Contaminant inputs

Contaminant entry from point source discharges to the reference estuaries varies from minor input for some, to relatively large for others (Table 9). For example, treated municipal sewage is the largest point source input to some estuaries, and varies from 145,000 m^3/day for the Avon-Heathcote Estuary to zero for the Ohiwa and Kaikorai estuaries (Table 9). Other point contaminant sources include urban stormwater, dairy shed wastewater, and various small industrial inputs, but flows are not easily quantifiable.

In the past, industrial and municipal contaminant discharges to estuaries like the Avon-Heathcote, Kaikorai and, to a lesser extent, the Waimea were more numerous, with the majority being untreated (*e.g.* those from tanneries, glue factories, metal and gas works, timber and woollen mills, wool scours and untreated sewage). With the growing public awareness of estuary deterioration and health risks, legislation has been tightened up and now there are greater controls/restrictions, and improved treatment systems for most domestic and industrial wastewater.

Table 9: Point source discharge information for the reference estuaries

Estuary	Point Source Discharges	Municipal Wastewater Flow (m^3/d)	Historical Point Source Input
New River	Invercargill treated wastewater, landfill leachate, fertiliser plant wastewater, dairy shed wastewater.	20 000	Moderate
Kaikorai	Urban stormwater.	0	High inputs
Avon Heathcote	Christchurch oxidation ponds, urban stormwater.	145 000	High inputs
Havelock	Havelock oxidation ponds, fish processing wastewater	100	Low
Ruataniwha	Collingwood treated wastewater, dairy shed wastewater upstream.	60	Very Low
Waimea	Bells Is oxidation ponds, dairy shed wastewater upstream.	12 050	Moderate
Ohiwa	Dairy shed wastewater upstream.	0	Zero
Whangamata	Whangamata oxidation ponds, urban stormwater.	1600	Low
Otamatea Arm	Treated sewage, Dairy Factory Discharge	3300[1]	Low

[1] includes maximum of 3000 m^3 per day from the Maungaturoto Milk processing plant

3.2.5 Biology

In general, the plant and animal life of the reference estuaries has not been well studied. In particular, the intricate web of ecosystem dynamics and the major factors that drive the productivity of each of the estuaries (or habitats within estuaries) has received little attention. In addition, the more mobile and temporary inhabitants that may use estuaries as nursery and breeding grounds are poorly understood. However, studies of fish populations in a variety of New Zealand estuaries have identified more than 20 resident or migratory species (Morrison & Francis 2000), and Bradstock (1983) suggests that more than 40 species may use estuarine habitat during some stage of their life cycle. In general, where studies have been undertaken, they have been initiated in response to existing estuary contamination problems.

The available information on benthic animal life in New Zealand estuaries indicates that biota is similar to that found in other estuaries throughout the temperate world. This includes bivalves (*e.g*. pipi, cockle, nut shell,), gastropods (*e.g*. whelks, topshells, mud snails), polychaetes (burrowing worms), crustaceans (*e.g*. crabs, amphipods), anemones and fish. Likewise, the submerged and emergent vegetation include species from the common groupings of rushes, reeds, scrub, grasses and tussocks, herbfields, seagrasses and macroalgae. Because of the generally large expanses of otherwise unvegetated sand and mud flats, benthic microalgal communities are important contributors to estuarine productivity. Estuarine benthic environments are also sites of intensive microbial activities that are important for controlling nutrient and oxygen dynamics.

The mud snail, *Amphibola crenata* on the mud flats of Waimea Estuary.

Table 10: Comparison of general characteristics (where known) of reference estuaries. Many of the values are 'best guess' estimates, however blank spaces were left where the necessary information was not available. For the detailed descriptions of each estuary, refer to Appendix A.

Estuary/ Location	**Type (refer key)**	**Volume HWS (10^3 m^3)***	**Residence Time (days)***	**Mean FW Inflow (m^3/s) ***	**Total Area (ha)**	**Area/FW Inflow**	**Intertidal area (ha)**	**% Intertidal**	**Max. Tidal Range (m)**	**Shoreline length (km)**	**Catchment Area (km^2)**	**Catchment pop'n/km²**	**Major Landuse**	**Values/Uses (refer key)**	**Threats (refer key)**
Otamatea Arm of Kaipara Harbour – Northland	FE, BE (DS)		< 3	3	1675	558			2.7		614		P	a-h	a, d
Ohiwa – Bay of Plenty	FE,BE (DS)		<2	3	2700	900	1890	70	1.5	85	186		P, NF, EF	a-h	a-d
Ruataniwha - Collingwood	FE, BE (MS)		< 1	80	1610	20			4.2		702	0.2	NF, S, P	a-d, f-h	a
Waimea - Nelson	FE, BE (I)	62	0.6	25	3455	166	2870	83	4.2		812		EF, NF, P	a-f	a-c
Havelock - Havelock	FE, HE		<3	40	908	23			4.2		105		NF, P, EF	a-g	a-d
Avon- Heathcote - Christchurch	FE, BE (SS)	8.3	<1	3	800	267		85	2.2		188		U, P	a-e	a-d
Kaikorai - Dunedin	FE, BE (B)		Variable	0.4	195	488			1		55		P, U	d	b-d
New River - Invercargill	FE, BE (DS)	85	3	61	3,500	57					3502		P, NF	a-e	a-d

Estuary Type: FE = Fluvial Erosion; B-E = Barrier-Enclosed; H-E = Headland-Enclosed; SS = Single Spit; DS = Double Spit; MS = Multiple Spit; I = Island; B = Beach

Values/Uses: a = Historical/Cultural; b = Recreational; c = Ecological; d = Fish/Wildlife habitat; e = Wastewater discharge; f = Commercial (e.g. aquaculture, ecotourism); g = Harbour; i = Shellfish harvesting

Threats: a = Spread of exotic species; b = Contamination (e.g. chemical, microbiological); c = Nutrient enrichment; d = Sediment input

Landuse: Major uses (>10% of total) in decreasing order; U = Urban; P = Pastoral; EF = Exotic Forestry; NF = Native Forestry; S = Scrub

3.2.6 Conclusions

How useful were these nine reference estuaries for development of a monitoring protocol? The following points summarise the main considerations:

- A broad range of estuaries, in terms of latitude and size, was trialled. This was necessary in order to make the protocol Nationally applicable.
- The types of estuaries compared (*i.e.* origins, morphological structure, flushing characteristics) were typical of most estuaries in New Zealand. Less common estuary types such as those with deep mixing basins or embayments and low flushing rates (*e.g.* sounds and fiords) were not represented. Nor were river delta systems, that are not barrier enclosed, included. Only one lagoon system with restricted flushing was included.
- Most of the major intertidal habitat classes were represented in all the estuaries, although some (*e.g.* mangroves and Pacific oyster beds) were restricted to northern latitudes.
- The mud/sand habitat, selected for fine-scale analyses, was dominant in all the estuaries, and this is typical of estuaries in general in New Zealand.
- The estuaries were representative of a variety of different states of modification or condition. Some were relatively pristine while others had been subjected to significant impacts from various stresses/uses.
- The reference selection did not include estuaries (or sites within estuaries) that are highly impacted (*i.e.* those with highly enriched, anoxic, or highly contaminated sediments). The intention was to include ‘representative’ sites within the estuaries rather than ‘hot spots’ that would normally be subject to consent monitoring.

Conclusions

We conclude that the choice of reference estuaries provided a useful basis of comparison for trialling/developing the protocol. It is expected that eventual assessment of pristine and highly impacted sites, and estuaries of contrasting morphological characteristics (using the protocol), will build on the database provided in this report. This will broaden the applicability of the protocol and improve confidence in interpretation of assessment results.

4. PRELIMINARY ASSESSMENT OF ESTUARY CONDITION

4.1 Introduction

Estuaries represent a transitional ecosystem between land and sea, influenced by a diverse range of factors. As such, an integrated, multidisciplinary approach to assessing estuarine condition is central to effective environmental monitoring and decision-making. An index of estuarine status that ranks estuary characteristics is a valuable tool for making a rapid, first-cut assessment of environmental conditions and issues of estuaries in a region, and gives a holistic framework within which to make preliminary management decisions.

One impetus behind developing indices of environmental condition is that they allow a considerable volume of environmental information to be conveyed in a concise and meaningful way. This can provide coastal managers, and the general public, a simplified yet comprehensive overview of the estuaries in their region. An index gives regional perspective by condensing a broad range of information without undue sacrifice of individual detail. A preliminary ranking tool for estuaries within a particular region can be useful for assigning relative ecological values and providing a defensible mechanism for setting priorities with regard to long-term, state of environment monitoring.

There have been a number of indices of estuarine health and classification recently developed and applied to estuaries, both overseas and within New Zealand (Hume and Herdendorf 1988; Cooper *et al*. 1994). These have focussed on classifying one or more aspects of the estuary such as geomorphology, water quality, biology and aesthetics. The ‘pressure-state-response’ model is also commonly used as a framework for developing environmental indicators and reporting (MfE Environmental Performance Indicators report series). These indices have provided a framework for comparison, and allowed estuaries, or regions within estuaries, to be assigned particular value or status, and managed and monitored accordingly.

An initial ranking protocol for estuaries was developed in the current project, in order to help New Zealand coastal managers with the decision over which estuaries to monitor in a region. The aim was NOT to provide a 'magic' number that would represent the state of health of an estuary. The aim was to provide a flexible tool (the 'decision matrix') to give a rapid, broad overview of the condition/status of an estuary in order to assist managers in prioritising candidate locations for monitoring, and hence, provide a defensible basis for long-term planning decisions.

4.2 Approach

The first step in developing an initial assessment ranking tool (the decision matrix) was to define the key issues relating to the condition of New Zealand estuaries. This was achieved by consultation and feedback from Regional Council staff and other interested groups from the reference estuary regions (summarised in Table 2). This background information was then used to decide on a number of the estuarine characteristics that would be included in the decision matrix to classify and prioritise estuaries in their regions. It was considered important to include the current issues, as perceived by coastal managers and the public, in the preliminary assessment procedure.

The second step was to identify other estuarine characteristics that should be included in the decision matrix. Effort was made to include a wide range of estuarine habitats, so the matrix was not focussed on a single habitat (*e.g.* the intertidal area that is the focus of the fine-scale monitoring in this study).

Some of the characteristics/features identified are easily assessed (*e.g.* level of aquaculture risk). However, some characteristics are not so easily assessed and may require consultation (*e.g.* community perception, cultural values), or scientific or ecological knowledge/investigation (*e.g.* hydrodynamic characteristics, indicators of habitat condition/health).

A number of characteristics chosen were similar to those from a 'pressure-state-response' model. The matrix included characteristics that were pressures on the environment (*e.g.* point source effluent discharges) and states describing estuary condition (*e.g.* habitat extent). Responses (management actions that may be remedial, *e.g.* applying consent conditions, rehabilitation efforts) were not directly included as characteristics in the matrix, but they may be considered in the scoring (*e.g.* the scoring of '*Aquaculture Licences*' as a '*3: no current or likely future aquaculture activities*' could relate to the restriction of aquaculture activities by council legislation).

The estuary assessment factors that make up the decision matrix were separated into four broad themes:

a) Existing physical and biological characteristics
b) Natural character and values
c) Characteristics that indicate a potential for an adverse impact
d) Characteristics that indicate an existing impact

The final stage of the process was to assign a weighting factor to the assessment criteria, to place emphasis on those characteristics of particular relevance to a region, community or manager (allowing the ranking procedure to be 'personalised'). This means that the ranking is applied, to a limited extent, at the discretion of the end-user (through the application of a relative weighting factor to particular characteristics of interest). For example, if the estuary is of important cultural significance as a traditional food-gathering site, then the assessment factors '*Cultural significance*' and '*Extent of fish/shellfish resource*' may be allocated a greater weighting relative to other factors. Ultimately, the decision on what estuary to monitor, based on the final score, is at the discretion of the matrix-user. By using a risk-assessment approach, they may decide to prioritise estuarine monitoring to high-risk, highly impacted estuaries or to estuaries of high natural (or other) value that are not showing the same level of degradation.

The decision matrix was developed to assist managers in the preliminary characterisation of estuaries in their region. In completing the table for each of their estuaries, it is envisaged that managers will:

a) become more familiar with their estuaries,
b) identify knowledge gaps about their estuaries,
c) identify the significant values within their estuaries,
d) identify potential threats to estuarine values,
e) prioritise estuary monitoring based on the current condition, potential threats, or values of significance (*e.g.* ecological, cultural, recreational, and economic).

It is accepted that the decision matrix does have limitations, including:

- there is some loss of individual detail as it condenses and simplifies a large bulk of information about each estuary,

- the decision matrix can not be applied to a broad comparison of estuaries outside a particular region. The ranking factors allocated in the examples are subjective and discretionary. They can be modified or replaced to emphasise particular features that are considered more relevant to estuaries in a region. Although this allows the ranking process to be tailored to the concerns and issues of the region, community or manager, it precludes its use for ranking estuaries against those in other regions,
- the ranking result is only as good as the information used in its application. *This could also be seen as a strength as it will allow improvement of the result with the application of more or higher quality information about the estuary* and,
- in the case of relatively undisturbed estuaries, particularly, further consideration will be required of the potential for future degradation of existing values; *e.g.* high natural freshwater (nutrient or sediment) inflows, low flushing rate, *etc*.

4.3 Results

The decision matrix is presented in Table 12. Weighting factors are applied to estuarine criteria that are considered to be more important/ significant to that region or estuary, and a rank is applied to that estuary based on the preliminary assessment of the criteria. For example, an estuary that is remote, with no large residential communities nearby might allocate a lower weighting (1, 3, or 5) to '*Extent of water clarity problems*' and '*Extent of nuisance odour problems*' as those factors are not considered as important to the region (*i.e.* they are not 'problems'). Therefore, even if that estuary scores a high rank (1 to 3) for those factors (maybe because there is extensive agricultural effluent causing eutrophication, excess algae, odours *etc*,) those criteria will not contribute as much to the final score as other (more heavily weighted) criteria (Table 11).

Table 11: An example of the application of weighting factors and ranking of the Decision Matrix

Scenario	Assessment Factor	Weighting Factor (5, 3 or 1)	Rank (1,2 or 3)	Total Score
A	Extent of nuisance algal blooms	1	3	**3**
B	Extent of nuisance algal blooms	5	3	**15**
A	Wetland and bird status	5	3	**15**
B	Wetland and bird status	5	1	**5**

Scenario A = low/no residential communities, large extent of algal blooms, low use of the estuary for recreation etc, but high international bird community

Scenario B = high extent of residential communities surrounding estuary, large extent of algal blooms, with a high level of recreation on and around the estuary etc…

The decisions based on the results of the preliminary assessment ranking are also at the discretion of the matrix-user. A low final score indicates that the estuary condition is at risk or already degraded. If the users of the matrix are prioritising estuaries for monitoring that are impacted or are at risk by disturbances or contaminants from urban development, catchment landuse practices, pollution or other disturbances, then a low final score indicates a high priority estuary. If the users of the matrix are prioritising estuaries for monitoring that are near 'pristine', with high natural values, then a high final score indicates a high priority estuary. In this case, further consideration will be required of potential or perceived risks. This could involve some crystal ball gazing to predict the likelihood for future development within the catchment. However existing physical and biological characteristics can also indicate the potential for the future decline of natural values; *e.g.* freshwater and sediment inflow rates, flushing rate, *etc*. Thus fine-tuning of priorities among relatively undisturbed estuaries can be achieved by revisiting the matrix and adjusting weightings accordingly.

Table 12: The decision matrix developed for a preliminary estuary assessment to assist with prioritising estuaries for state of environment monitoring.

DECISION MATRIX FOR PRIORITISING ESTUARIES FOR STATE OF ENVIRONMENT MONITORING

	Estuary Assessment Factor	Explanation	Scoring Schedule	Estuary 1			Estuary 2		
				Score	Weighting factor	Total	Score	Weighting factor	Total
A.	**Existing Estuary Physical and Biological Characteristics**								
1	Area of Estuary (ha)	Value of an estuary increases with the area of the resource.	1 = <500 ha, 2 = 500-2500 ha, 3 =>2500 ha.						
2	Diversity of intertidal habitat	Estuaries with the broadest array of intertidal habitats have the greatest potential for high intertidal biodiversity and therefore have greatest ecological value to a region. Habitats include: rushes, reeds, seagrasses, tussocks, herbfields, scrub, rock, cobble, gravel, mobile sand, sand, shell, muddy sand, soft muds, shellfish beds, sabellid beds.	1 = limited array of habitats, 2 = moderate array of habitats, 3 = most common habitats present and in good condition						
3	Diversity of subtidal habitat	Estuaries with the broadest array of subtidal habitats over a wide depth range have the greatest potential for high subtidal biodiversity and therefore have greatest ecological value to a region. Habitats include: macroalgal beds, seagrass beds, rock, cobble, gravel, mobile sand, sand, shell, muddy sand, soft muds, shellfish beds.	1 = limited array of habitats, 2 = moderate array of habitats, 3 = most common habitats present and in good condition						
4	Flushing time (days)	Flushing time is the average period during which a quantity of freshwater derived from a stream or seepage remains in the estuary. The very well-flushed estuaries will be least at risk from build-up of contaminants.	1 = >10 days, 2 = 3-10 days, 3 = < 3 days						
5	Freshwater input (m^3/s)/Area of estuary (ha) ratio	Estuaries with a high FW/A ratio have a large freshwater influence and often result in a relatively harsh environment for aquatic life (*i.e.* biodiversity tends to be less).	1 = >100, 2 = 10-100, 3 = <10.						
6	Extent of mangrove and saltmarsh habitat	Estuaries where mangrove and/or saltmarsh habitats have been reduced or reclaimed have lower ecological value, fewer feeding and nursery habitat for other species, and a decreased ability to assimilate contaminant and sediment entry. These habitats act as coastal buffers.	1 = low or severely reduced, 2 = moderately reduced, 3 = habitat present in unaltered extent and in good condition (For regions outside the range of mangroves, use saltmarsh habitat as the single assessment factor)						
7	Extent of fish/shellfish resources	Occurrence of fish and shellfish resources in an estuary enhances the value. A drop in abundance and diversity could result from an increase in nutrients and pollutants to an estuary.	1 = low or no fish and shellfish resources, 2 = medium abundance/diversity, 3 = High abundance and/or diversity						
B.	**Natural Character and Values**								
8	Wetland and wildlife status	Estuaries are often important habitat for coastal fisheries and international migratory birds, and may be recognised as having significant conservation value. Estuaries with high wetland and wildlife status have a high perceived value.	1 = low, 2 = medium, 3 = high wetland and wildlife status						
9	Recreational use	An estuary can be a significant social resource, used for water sports, food gathering, sightseeing, exercising *etc.*	1 = low utilisation for recreation, 2 = moderate, 3 = high utilisation for recreation						
10	Cultural significance	The values of tangata whenua, including the issue of mana whenua (customary authority) may be significant to an estuary. Estuaries may have a high cultural value if they are or were a traditional food-gathering site, papa taakoro or of other cultural importance.	1 = low perceived cultural significance, 2 = medium, 3 = high perceived cultural significance						
11	Commercial use	An estuary can be a commercial resource with economic importance, for example through shellfish/fish harvesting, aquaculture, ecotourism *etc.*	1 = low commerical use, 2 = moderate, 3 = high commercial use						
12	Perceived value by the communities in the region	Estuaries may have high aesthetic and amenity value to surrounding residential communities. They may also be important for education, tourism, or significant to the communities' natural character or identity.	1 = low perceived value by communities, 2 = medium, 3 = high perceived value by communities						
13	Potential for rehabilitation	Historically impacted estuaries may have a greater potential for rehabilitation of estuary condition than currently impacted estuaries.	1 = low potential for rehabilitation, 2 = medium, 3 = high potential for rehabilitation						
C.	**Characteristics that Indicate a Potential for an Adverse Impact**								
14	Proportion of urban/industrial landuse in the estuary catchment	Modified catchments are likely to pose greatest risk to each estuary from contaminant entry. Urban and industrial contaminants include heavy metals, nutrients, organochloride pesticides *etc.*	1 = high extent of urban/industrial landuse, 2 = medium, 3 = low extent of urban/industrial landuse						
15	Proportion of agricultrual landuse in the estuary catchment	Modified catchments are likely to pose greatest risk to each estuary from contaminant entry. Agricultural run-off has been attributed to increased sedimentation, nutrients and contaminants in estuaries.	1 = high extent of agricultural landuse, 2 = medium, 3 = low extent of agricultural landuse						
16	Proportion of exotic forest landuse in the estuary catchment	Modified catchments are likely to pose greatest risk to each estuary from contaminant entry. Exotic forestry can impact on estuaries by causing increased erosion of the catchment, increased sedimentation and nutrients in the estuaries.	1 = high extent of exotic forest landuse, 2 = medium, 3 = low extent of exotic forest landuse						
17	Proportion of unmodified estuary catchment	The least modified catchments are likely to pose least risk to each estuary from contaminant entry. Unmodified land may also include parks, reserves and other protected areas on the estuary margin.	1 = low extent of unmodified catchment, 2 = medium, 3 = high extent of unmodified catchment						
18	Estuary margin alteration (*e.g.* reclamation)	Estuaries where margins have been altered and/or reclamation has been undertaken have less value and a decreased ability to assimilate contaminant entry and increased erosion and sedimentation processes.	1 = high extent, 2 = medium extent, 3 = low extent of margin alteration						
19	Point Source effluents	Presence of point source discharges of wastewater (municipal, industrial and/or agricultural) into an estuary poses a high risk of contaminant entry.	1 = extensive discharges, 2 = moderate discharges, 3 = very low or no discharges.						
20	Aquaculture licences	Presence of aquaculture activities in an estuary provides a greater risk of contaminant entry and other impacts (*e.g.* biosecurity risk and impingement on the natural and aesthetic values of an estuary).	1 = aquaculture licences exist in estuary, 2 = estuary is at risk from aquaculture developments, 3 = estuary has no current or likely future aquaculture activities.						
21	Extent of biosecurity risk	Infiltration of an estuary by foreign plants and/or animals poses risks to the existing habitat and community structure. Risk assessment should include such factors as: likelihood of entry (*e.g.* high risk for ports, areas with extensive aquaculture or areas which attract boats), likelihood of invaders surviving, and risk of impacts on perceived estuary values.	1 = high risk, 2 = medium risk, 3 = low biosecurity risk						
22	Extent of risk of accidental spills	Accidental spillage of hazardous wastes (*e.g* . oil) lowers values in an estuary.	1 = high risk, 2 = medium risk, 3 = low risk of accidental spills						
D.	**Characteristics that Indicate an Existing Impact**								
23	Extent of nuisance macro and micro-algal blooms	Algal blooms (*e.g. Ulva* sp.) indicate nutrient enrichment. Estuaries with algal bloom problems often have widespread adverse ecological and aesthetic effects. Additionally, there may be health risks associated with eating contaminated shellfish during bloom events.	1 = frequent algal bloom problems and/or large areas of nuisance macroalgae, 2 = occasional algal bloom problems 3 = rare algal bloom problems						
24	Extent of invasive species	Occurrence of exotic invasive species can threaten the natural character and biodiversity of an estuary (*e.g.* Pacific oyster, *Spartina* sp.)	1 = large colonisation of invasive species, 2 = low extent of invasive species, 3 = no known invasive species						
25	Extent of modification of estuary hydrodynamic characteristics	The hydrodynamic processes of an estuary can be altered by gravel or sand extraction, roading, reclamation and structures, creating modified water circulation patterns, increased sedimentation, less flushing and an increase in contaminant loading.	1 = large extent, 2 = moderate extent, 3 = low extent of modification of hydrodynamic characteristics						
26	Extent of water clarity problems	Widespread water clarity problems (*e.g.* after heavy rain and/or wind events) lower the perceived value of an estuary, have an adverse social effect and adversely effect aquatic ecosystems.	1 = frequent, 2 = occasional, 3 = rare water clarity problems						
27	Suitability for human contact	Water that people would not swim in or wade in has low value. Waters that are appealing to swim or wade in have highest value. Water quality problems include water-borne disease risks.	1 = water frequently not suitable for human contact, 2 = water on occasions not suitable for human contact, 3 = water always suitable for human contact						
28	Extent of faecal contamination problems	Widespread faecal contamination problems lower estuary values. Problems are indicated by high faecal coliforms and enterococci in the water column and shellfish, illness or perceived health risk.	1 = High extent, 2 = moderate extent, 3 = low or no extent of faecal contamination problems						
29	Extent of nuisance odour problems	Widespread nuisance odour problems lower estuary values, *e.g.* from effluent, decomposing macroalgae, anaerobic sediments.	1 = frequent problems, 2 = occasional problems, 3 = rare or no nuisance odour problems						
30	Extent of toxicity problems	Widespread toxicity problems or perceived problems (*e.g.* metals, organics, sulphide, ammonia) lower estuary values. Toxicity problems can be both in the water column and sediment, and may have extensive adverse effects for the biological communities within the estuary.	1 = High extent, 2 = moderate extent, 3 = low or no extent of toxicity problems						
31	Solid waste	The presence of solid waste (*e.g.* refuse) lowers estuary values.	1 = High occurrence, 2 = medium occurrence, 3 = low occurrence of solid waste						
	Total Score	**If estuaries with existing and potential adverse effects and currently degraded estuary condition are prioritised for monitoring, then the lower the final score the higher the priority for state of environment monitoring. If the estuaries with near to pristine condition, high natural values and low potential for adverse effects are prioritised for monitoring, then the higher the final score the higher the priority for state of the environment monitoring.**							

Summary

The Decision Matrix was developed using a combination of specialist input, council feedback and experience gained during the present study. The Matrix was not trialed as a preliminary assessment tool to select the reference estuaries in the current study. These were chosen based on other factors beyond the consideration of 'estuary condition', such as Council nomination, logistics of funding and support, and ensuring a latitudinal spread within New Zealand. However, the Decision Matrix is a relevant step in the EMP as a tool for end-users to choose an appropriate estuary to monitor.

The Decision Matrix provides a framework that the EMP-users can work through to determine and rank specific estuarine issues and characteristics, related to condition, that are relevant to the management of estuaries within their region. The ranking system can be personalized to incorporate the specific issues or priorities of the matrix-user. This is accomplished through the application of weighting factors that reflect the importance of estuary values for that region. By re-addressing the Matrix, it is possible to evaluate different scenarios; *e.g.* What would be the implications of different (or changing) usage and value priorities? Changes that may have occurred subsequent to particular management actions (*e.g.* consent decisions, habitat restoration, etc. may also be assessed by re-addressing the matrix.

5. BROAD-SCALE HABITAT MAPPING (USING GIS)

5.1 Introduction

Intertidal physical and biological habitats are integral to the structure and function of the estuarine environment and the way that they interact with adjoining terrestrial and marine ecosystems. Key steps in understanding the functional importance of the nine REs (and managing them accordingly) were to design an intertidal habitat classification scheme, and, subsequently, to map their different habitat types. Once the dominant habitats are defined, and their boundaries established, this information becomes a valuable benchmark indicator for measuring change (MfE Confirmed Indicators for the Marine Environment, ME6 2001). Within this framework, finer-scale monitoring priorities can be better defined in order to address issues of habitat or whole estuary condition (see Section 6).

The boundaries of ecologically important (functional) habitats (*e.g.* mud flat, sand flat, seagrass, salt marsh, mangrove) although relatively stable over the short term, (*i.e.* weeks to months) have the potential to shift over the longer term (*i.e.* one to five years or longer). Some habitats may shrink in area while others expand or merely relocate. In order to address questions regarding shorter term changes (*e.g.* seasonal variation in macroalgal bed development) repetitive aerial surveys will be required along with particular ground-truthing attention to establishing the threshold for change in the habitat coverage. Gross changes in habitat areas may reflect natural perturbations or human impacts that simultaneously affect benthic physical, chemical and biological characteristics relating to environmental quality. For example, if sandy areas become inundated with mud, it follows that the infaunal biological community living within the substrate will change. While it is possible to monitor the infaunal community directly, it is much more cost-effective, in the first instance, to look for broader habitat changes that will allow specific studies to be focused on areas where change is most likely to be significant.

The aim of broad-scale habitat mapping was to define each RE according to the dominant intertidal habitats, based on surface features (*e.g.* substrate and vegetation type), and develop baseline maps. This procedure involved the use of aerial photography coupled with ground-truthing of the images, and digital mapping using GIS technology. The first stage was to develop a methodology appropriate for translating aerial photographic information into a GIS format suitable for the project's requirements. Wilton & Saintilan (2000) provide a detailed discussion of mapping methodologies.

5.2 Approach

5.2.1 Allocating mapping scales

The ability to detect habitat change is directly related to scale. Scale in this context, is the relationship between a distance on a map and a corresponding distance on the ground. For example, we refer here to a scale of 1:10,000 as large in comparison to a scale of 1:100,000, however this can be confusing as it is sometimes defined (perhaps more correctly) in the opposite way.

In order to detect small changes in habitat area, the habitat boundaries must first be accurately mapped. At large spatial scales, only large changes may be detectable (even when using the most up-to-date mapping procedures) or it may take many years for a number of small changes to become apparent. At smaller scales, small changes are likely to be more readily detectable. The scale used must also reflect the precision or resolution required (*i.e.* the size of the smallest feature that can be represented on a boundary). For example, if it is important to know the total area of sand compared to mud within an estuary, broad mapping can provide such information. However, if it is important to know specifically where the mud and sand areas are, a higher level of precision would be required. Therefore, it is important to define an appropriate spatial scale for baseline mapping of estuarine habitat to enable detection of change at an appropriate scale to assist in management. The scale selected will depend on the logistical constraints of data collection and management, the purpose of the monitoring, and the time frame being investigated.

It was anticipated that the broad-scale habitat mapping technique would be capable of detecting a shift in a habitat boundary of < 5 m for a single rectified image (an image in GIS format that has had certain points on the image matched, or 'rectified', to correspond with the same point on the aerial map). To achieve this, an appropriate mapping scale would be a maximum of 1:10,000. The detection of changes at larger spatial scales than this is likely to be too broad to serve a useful purpose. This is particularly so for salt marsh vegetation where the spatial change may be due to the summation of the expansion of a large number of small patches. The detection of change at smaller spatial scales than this is difficult to achieve using relatively inexpensive methods such as aerial photography and field verified digital mapping. The selected scale of 1:10,000 is also likely to be sufficient to track changes over a reasonably short time interval (2-5 years), which would aid in the ecological management of estuaries and would provide a relatively simple, inexpensive and effective basis upon which to assign management priorities.

5.2.2 Precision on habitat boundaries

The process of broad-scale habitat mapping at 1:10,000 using aerial photography carries with it an error at any point of approximately 2-15 m. This is due to the fact that overlapping aerial photographs will be taken from slightly different angles and, when overlaid and scanned into a computer on a flat plane, single points in space will vary. The direction and magnitude of the error at any point is going to be very similar to that for other points nearby. For example, a circular patch of *Zostera* (eelgrass), which is measured at 5 m diameter in the field, will be shown at around 5 m diameter on the digitised mosaic map. Thus, the area of the habitat patch may remain accurate, and the only difference will be that the map position may show the patch to be 2 m to 15 m away from where it actually is in the estuary.

There are techniques for limiting such errors to < 5 m for a single rectified image. This is achieved by field verification of a number of points within each photo, and subsequent rectification of the aerial photo with the field measurements using computer software. In the process of combining all the rectified images of an estuary into a mosaic of the whole estuary, the total error broadens to around 10-15 m for larger estuaries (*i.e.* those which require a large number of images) and 5-10 m for smaller estuaries.

When using such methods it is critical to ascertain the actual error for each estuary. It was proposed to check this by physically measuring the boundary of representative areas in the field using a Geographical Positioning System (GPS), and checking the extent to which they coincided with their positions on the digitised map. The actual error is also important to determine in situations where there is a need for precise mapping of a habitat boundary. For example, plotting the encroachment by mangroves or *Spartina* into an estuary may require monitoring to be able to differentiate a change of 2-5 m. The monitoring protocol will address this requirement to obtain greater precision by providing single rectified images with an error < 5 m. To put this into perspective, a line drawn on a map with a sharp ‘00’ pencil can produce an on-the-ground thickness of 3 m at a scale of 1:10,000.

5.3 Methods

5.3.1 Production of GIS maps

The GIS maps were created in two stages;

- **Stage 1:** Aerial photography to generate base maps of vegetation and substrate.
- **Stage 2:** Field surveys to verify photography, and identify and map features not distinguishable through aerial photography alone.

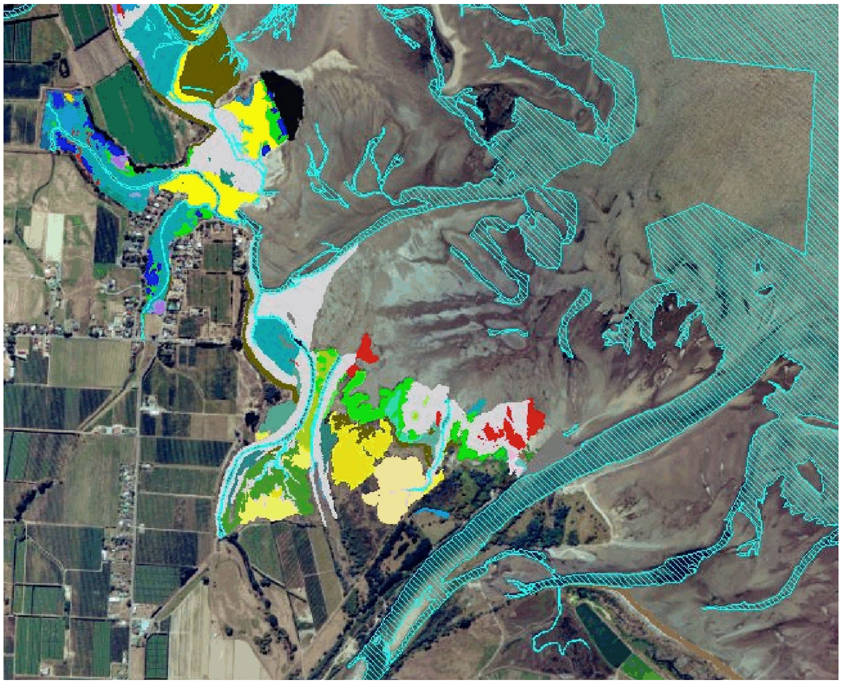
Colour aerial photographs were taken at low tide at a maximum scale of 1:10,000. In some cases, recent exixting photographs, held by councils, were found to be suitable for mapping. In other cases, appropriate aerial survey firms (*e.g.* New Zealand Aerial Mapping Ltd., Auckland/Hastings) were contracted to provide new photographs using standard aerial survey equipment and procedures. Survey dates are provided in Table 13.

Table 13: Aerial survey flight schedule and sources.

Estuary	**Date Flown**	**Source**
Otamatea Arm (Kaipara)	20 October 2000	Emap, Auckland
Whangamata	27 February 1998	Environment Waikato
Ohiwa	unknown	Airmaps NZ Ltd, Tauranga
Ruataniwha	9 December 2000	Aerial Surveys Ltd, Nelson
Waimea	31 March 1999	Aerial Surveys Ltd, Nelson
Havelock	19 March 1999	Aerial Surveys Ltd, Nelson
Avon-Heathcote	9 January 2000	Canterbury Regional Council
Kaikorai	21 March 2000	Aerial Surveys Ltd, Nelson
New River	9 September 2000	Environment Southland

Individual photos were then scanned at a resolution of 508 dpi (dots per inch) yielding an image resolution of 0.5 m per pixel. Prominent landmarks (sometimes referred to as ground control points) were identified on each photo, and during field verification, differential GPS positions were collected for each landmark using a Trimble Pathfinder Pro GPS. Photos were then rectified using a minimum of six GPS landmarks per photo. The landmarks were converted to Arcview shapefiles using Trimble Pathfinder software. ERDAS image analysis software, running under Arcview (v 3.1), was used to register, rectify, and mosaic the scanned photos.

Although this method could not achieve ortho-rectification of the photos, where the GIS image has had the distortion due to tilt and 3-dimensional relief removed (Wilton & Saintilan 2000), the lack of vertical features within the estuaries meant that camera lens distortion was minimised. Positional accuracy was recorded by calculating and documenting the root mean square (RMS) error for each landmark. In general, RMS error was kept to within ± 5 m using this procedure, however much greater accuracy could be achieved for many of the photos. On some occasions, image-to-image rectification could also be used to further improve accuracy. Each landmark, and the associated RMS error, was saved for future reference. Vegetation and substrate features were then digitally mapped on-screen from the rectified photos using the Arcview 'image analysis' extension. This procedure required using the mouse to draw as precisely as possible around the features identified from the field surveys on the computer screen and saving each drawing to a shape file or GIS layer associated with each specific vegetation or substrate feature. To calculate the area cover for a chosen habit type, the Arcview 'X-tools' extension was used. This gave the area of any selected features in hectares. These GIS layers, along with supplemental field information, were then combined with the image mosaic and written to CD-ROM as part of the final GIS output.



An example of broad-scale habitat mapping using GIS, showing an aerial photograph of Motueka Estuary with some of the dominant substrates/habitats overlaid by ground-truthing.

The classification of the features followed the proposed National classification system (with adaptations), which is currently being developed under another SMF program (Monitoring Changes in Wetland Extent: An Environmental Performance Indicator For Wetlands) by Lincoln Environmental, Lincoln. The classification system for wetland types is based on the Atkinson System (Atkinson 1985) and covers 4 levels, ranging from broad to fine-scale;

- Level I: Hydrosystem (*e.g.* intertidal estuary)
- Level II: Wetland Class (*e.g.* saltmarsh)
- Level III: Structural Class (*e.g.* marshland)
- Level IV: Dominant Cover (*e.g. Leptocarpus similis*)

For this project, Level III (Structural Class) and Level IV (Dominant Cover) were used. A list of all the classification types used in the study and their codes are given in Appendix B, Table A21 and A22.

5.4 Results

5.4.1 Summary of the reference estuaries

The broad-scale results for the REs are summarised individually in Appendix B.1. Included for each estuary, are:

- a map describing the general structural class distribution,
- a map representing the pattern of dominant cover,
- a summary figure comparing the areas of major habitats, and
- a table providing the area and relative proportions of the habitat groupings.

The complete data set, provided on the accompanying compact disc, will allow any combinations of habitat characteristics to be identified and compared in a similar way.

These results provide an overview of the broad-scale characteristics of a range of New Zealand estuaries. Considerable inter-estuary variation occurred in the habitat categories represented and their proportional coverage. Contrasting structural patterns can be seen that relate to location (*i.e.* latitude), estuary background characteristics (*e.g.* morphology, hydrology, *etc*), and degree of estuary modification (*e.g.* infilling, catchment development characteristics).

Table 14 provides an example of the application of broad-scale habitat mapping to determine the change in habitat types and area over time. The narrow range of habitats found in the Whangamata Estuary is dominated by unvegetated substrate, mangroves (scrubland) and seagrass. The changes in relative proportions of mangrove and eelgrass habitat are evident over time, as the mangrove habitat has expanded and the eelgrass meadows have declined. Since such changes in habitat structure can significantly alter estuarine function in terms of productivity, sedimentation, nutrient flux, *etc*., they can have important management implications.

Table 14: An historical comparison of the dominant habitat of Whangamata estuary using data from 1944, 1965 and 2001.

Habitat Type	**1944**	**1965**	**2001**
Scrubland (mangroves)	31	46	103
Rushland	17	17	10
Tussockland	-	-	4
Herbfield	-	-	3
Seagrass meadow	81	103	60
Unvegetated	-	-	256
Water	-	-	83
Total area of estuary (ha)			**460**

Some of the REs (*e.g.* Otamatea Arm, Ohiwa, Havelock, Kaikorai) contained a relatively large subtidal area (Table 15). This highlights a potential deficiency of the broad-scale mapping protocol for some estuaries as only intertidal habitats were surveyed. Estuaries that contain large areas that are not exposed during spring low tides should ideally be considered for the inclusion of additional subtidal investigation. This would require different sampling procedures than the ones described in the present project.

Table 15: Summary of the dominant vegetated and unvegetated habitats in the nine reference estuaries.

Habitat Type	Otamatea	Whangamata	Ohiwa	Ruataniwha	Waimea	Havelock	Avon-Heathcote	Kaikorai	New River
Scrubland	19.4%	22.3%	3.7%	1.6%	0.1%	3.5%	0.0%	3.1%	0.0%
Rushland	0.3%	2.3%	1.7%	13.4%	3.1%	22.9%	1.0%	10.5%	7.5%
Reedland	0.0%	0.0%	0.0%	0.0%	0.0%	6.2%	0.0%	1.3%	2.7%
Tussockland	0.0%	1.0%	0.0%	0.1%	0.3%	0.0%	0.0%	0.4%	0.4%
Grassland	0.2%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	7.9%	0.4%
Sedgeland	0.0%	0.0%	0.1%	0.0%	0.0%	0.0%	0.0%	0.2%	0.1%
Herbfield	0.1%	0.6%	0.0%	0.4%	3.8%	0.2%	0.0%	29.8%	0.1%
Seagrass meadow	0.0%	13.0%	4.0%	1.4%	0.9%	0.1%	1.9%	0.0%	2.2%
Unvegetated	39.7%	55.7%	67.6%	68.7%	77.4%	36.8%	66.3%	22.6%	58.4%
Macroalgal bed	0.1%	0.0%	2.1%	0.0%	2.1%	0.3%	6.2%	0.0%	1.3%
Shellfish field	9.6%	0.0%	0.0%	0.0%	1.0%	2.4%	0.0%	0.0%	0.0%
Worm field	0.0%	0.0%	0.0%	0.0%	0.1%	0.0%	0.0%	0.0%	0.0%
Water	40.2%	18.1%	26.8%	15.9%	14.3%	27.9%	32.7%	24.3%	26.9%
Total area of estuary (ha)	**1716**	**460**	**2683**	**863**	**3206**	**817**	**707**	**113**	**4225**

Summary

Baseline habitat maps of the nine REs were prepared, against which future (or historical) comparisons can be made. The techniques, trialed, were suitable for the identification and mapping of boundaries of key intertidal habitats at a scale and precision appropriate for addressing subsequent management questions. The methods description has been targeted for the non-specialist scientist level in order to facilitate its use among coastal managers throughout New Zealand. As a rule of thumb, the cost to survey the broad-scale habitat of an estuary can range from $15,000 to $30,000, depending on its size. We suggest that a monitoring frequency of five years would be suitable to provide input for addressing most medium to long-term, management-related questions/strategies. Shorter term questions, such as the rate of invasion of an exotic species, or the effects of a major hydrological modification, may require more frequent (*e.g.* yearly) surveys.

The described methodology provides a simple and defensible, long-term monitoring tool suitable for incorporation into a National protocol for the assessment and monitoring of New Zealand estuaries. We recognize, however, that the technologies available for broad-scale habitat mapping are advancing rapidly (*e.g.* satellite imagery. GIS software, *etc*.). For this reason, it is essential that the resulting protocol be viewed as an evolving document that can be updated as new and better methods become available.

6. FINE-SCALE ENVIRONMENTAL MONITORING

6.1 Introduction

Once an estuary has been classified according to its main distinguishing features, and the dominant habitats have been described and mapped on a broad scale, suitable habitats may be selected and targeted for fine-scale monitoring. An appropriately designed monitoring protocol will enable many of the key issues (*e.g.* nutrient enrichment, extent of sediment toxicity) affecting estuary condition to be addressed at an appropriate level of investigation. A typical fine-scale monitoring programme involves measuring one or more environmental characteristics that are known to be indicative of estuary condition, and are likely provide a means for detecting subsequent change. For the purpose of this study, the range of environmental characteristics was restricted to a suite of commonly used benthic indicators (see section 2.4 for justification). Decisions regarding which of these analyses are most appropriate and how many samples are needed in order to get reliable estimates, are critical, and will ultimately determine the usefulness of the data.

The main aims of this component of the programme were to undertake a fine-scale survey of the eight reference estuaries, and examine the results to develop and refine a scientifically defensible, cost-effective and repeatable fine-scale methodology for incorporation into the Estuary Monitoring Protocol (EMP) for New Zealand estuaries. The specific goals of this section were to:

1. Describe each reference estuary in terms of a suite of physical, chemical and biological characteristics of a standardised benthic habitat (muddy sand in the mid-low intertidal zone).
2. Utilise data from the standardised habitat to develop a baseline of characteristics relating to environmental condition.
3. Examine and compare environmental characteristics among estuaries and sites within estuaries.
4. Examine the inter-relationships between environmental characteristics in order to determine the key variables to be included in the monitoring protocol.
5. Examine the variability that is associated with each measurement in relation to the precision and sample number that is required to detect the desired level of change.

Additionally, the study has generated a preliminary baseline data set that can act as a benchmark of estuarine condition for future monitoring. This data set will become more valuable over time as repeat surveys are carried out and additional estuaries are included.

The first step was to identify the key issues that needed to be addressed for this standardised programme to be effective, and to highlight any likely limitations.

Key points are:

- how to choose a benthic monitoring site,
- how many sites are needed per estuary,
- which characteristics to monitor,
- what is the appropriate methodology for each parameter (*e.g.* how many replicates to take from each site and where from, how often to monitor, mesh size for sorting macro-invertebrates), and
- how to interpret the results.

6.2 Approach

The fine-scale approach adopted for this study evolved from an initial proposal to assess a suite of benthic characteristics of a variety of dominant or ecologically important habitats within each estuary. However, after evaluation of a variety of potential sampling designs, it became clear that the sample replication required to enable a statistically robust comparison over time (say for four habitats in each of the eight estuaries) was outside the funding allowance available. For this reason, it was decided to focus on a single ‘key’ habitat that was a major component of all the reference estuaries, and indeed most, if not all, estuaries in New Zealand; *i.e.* the mud/sand habitat at mid to low tidal elevation.

Once the general approach had been decided, it became evident that the most effective method was to simply build on the few existing long-term monitoring approaches already adopted in regions such as Auckland and the Bay of Plenty. These regions assess estuary benthic condition by monitoring sediment (primarily macro-invertebrate) characteristics at representative mid-low water sandflat sites using a relatively large number of replicates. Although the Auckland Regional Council does not have an integrated and standardised approach for estuary benthic monitoring in the whole region, it does have a long-term (13 years), defensible ecological monitoring programme operating in the Manukau Harbour (Pridmore *et al*. 1990). On the other hand, the Environment Bay of Plenty estuary monitoring programme (operating for 11 years) is undertaken on a region-wide basis and in a relatively standard and defensible fashion (Park 1995). Monitoring programmes using a suite of benthic indicators of enrichment have also been developed and applied to a number of estuaries in the Nelson region (Gillespie *et al*. 1992, 1995, 2001; Gillespie & Asher 1996). These were based on descriptions of physico-chemical characteristics and related biological processes (Gillespie & MacKenzie 1981, 1990).

These approaches and others have provided Councils with defensible means of assessing estuary condition, particularly when focused on localised impact sites, but the approaches have not been widely adopted throughout the country. Possible reasons for this include:

- A lack of National standards against which the results can be compared;
- Confusion as to the need for a large number of replicate samples at each site and the consequent elevated cost;
- Lack of an integrated and standardised monitoring design.

A brief examination of the macro-invertebrate survey approaches used in two of the most comprehensive and long-term monitoring programmes (Auckland and Bay of Plenty regions) is presented in Table 16. In summary, the Auckland approach consisted of an initial intensive study which was used to design a defensible and relatively cost-effective long-term programme that included sampling 12 replicate cores at a number of representative sites on two occasions per year. The sampling design for Auckland estuaries has focused on detecting changes over relatively large spatial scales (5-30 m) and includes the area between low water and 90 m shoreward of low water in its sampling strategy. The Bay of Plenty approach also evolved out of the Manukau intensive study but in a different way. In the Bay of Plenty approach, it was decided to instigate a programme that included sufficient replication at each site to include all the likely species present at that site. The results of Pridmore *et al*. (1990) showed that for each site, few new taxa were likely to be found in each additional core after 16-24 cores had been analysed. From this evaluation, a soft-shore sample replication number of 30 was chosen for the Bay of Plenty estuary programme. Other differences in the Bay of Plenty approach are that: the frequency of sampling was set at once per year; the sieve size was increased to 1 mm to reduce handling times and cost; the focus has been on detecting changes over both small (1-5 m) and large (5-60 m) spatial scales; and the area of mud/sand habitat covered has been limited to that bordering the channel at low water.

Table 16: Summary of long-term monitoring approaches used to sample sediment macro-invertebrate characteristics at two New Zealand estuaries.

Estuary	**Size (ha)**	**Summary of sampling approach**	**Author(s)**
Manukau	36800	Six sites on the mid-tide sand flats (each site = 100 x 90 m), and each site divided into 12 equally-sized sectors. Every two months, 12 cores were collected from each site (one randomly from each sector). Invertebrates were separated from the sediment using a 0.5 mm mesh sieve.	Pridmore *et al.* (1990)
Ohiwa	2700	A total of six sites in the mud/sand habitat. At each site, a total of 30 benthic cores were collected, 6 cores randomly sampled from 5 x 5 m blocks located at 15 m intervals along a 60 m transect (*i.e.* 5 blocks in total at each site) at mean low tide level, parallel to the shoreline. A 1 mm mesh used to sieve invertebrates.	Park (1995)

The two respective approaches are still in operation and provide Councils with long-term monitoring data which they use to assess the state of their estuaries. Given the apparent success of both approaches, it was logical to build on them and develop a standardised fine-scale macro-invertebrate monitoring methodology that could be used in estuaries throughout New Zealand. It was also decided to incorporate a companion suite of physico-chemical and biological measurements, into the sampling design, that has been successfully used for estuarine impact assessment in a number of locations in New Zealand.

It is important to point out that the proposed approach does have certain clear limitations:

- It was designed as a means of assessing the condition of a dominant and relatively vulnerable intertidal habitat. Its relationship with the condition of other estuary habitat is uncertain and therefore its use as an indicator of whole estuary condition must be approached with caution. It is envisaged that further work will need to be undertaken to quantify such relationships and increase the power of the method.

- It has been designed for use in the dominant New Zealand estuary type (*i.e.* shallow, short residence time, barrier-enclosed estuaries with broad areas of sand/mudflat).

- Monitoring the condition of an estuary is complicated by the high natural spatial and temporal variability frequently associated with complex and dynamic estuarine environments (Pridmore *et al*. 1990). It can be difficult to accurately detect changes in a habitat if natural variation is high, and may lead to uncertainties in results. Therefore, fine-scale monitoring of benthic characteristics is a balance between gathering a sufficiently robust dataset to detect and explain trends, and cost-efficiency/ease of sampling. This trade-off between scientific defensibility (sampling accuracy and repeatability) and cost efficiency/ease of use, dictates the monitoring design and places some limitations on its application and ability to detect change.

- Turner *et al*. 1995 provide evidence of stability in sandflat macro-invertebrate communities in some situations. However, further research is needed to resolve short-term temporal variability in New Zealand estuaries. In order to minimise interference due to seasonal variation, the protocol recommends that the sampling be carried out during the mid- to late summer period.

Technical Box 6.1: Sediment Habitat Definitions

Abundance- number of individuals of a particular plant or animal species occurring within a specific area of seabed.
Benthic- associated with the seabed.
Biota- plants and animals.
Diversity/richness- number of taxa found within a specific area of the seabed.
Epifauna- animals living on the surface of the sediments
Infauna- animals living buried within the sediments
Macro-invertebrates- refers to animals without backbones that will not pass through a 0.5 mm mesh sieve.
Redox Discontinuity Layer (RDL) – transitional zone between aerobic (oxygenated) sediments and anaerobic (deoxygenated) sediments.
Taxa- plural of taxon.
Taxon- refers to a taxonomic category (*e.g.* species, family or class)

6.3 Methods

6.3.1 Sampling design

The sampling approach adopted for the fine-scale monitoring of the reference estuaries incorporated a combination of the methods of Pridmore *et al*. (1990) and some solicited advice of a specialist biometrician, David Baird (AgResearch, Christchurch). A summary of his recommendations is provided in Table 17. Three points to note are:

- Emphasis was placed on getting adequate spatial coverage of New Zealand (*i.e.* sampling several estuaries across New Zealand), rather than attempting to resolve spatio-temporal variation in only a few estuaries,
- Kaikorai Estuary was only sampled at one site due to the fact that the estuary was blocked at the time of sampling and access was limited, (the Kaikorai is also small in size and lacking in shoreline complexity) and,
- Whangamata Estuary was not included in the fine-scale study because Environment Waikato elected to carry out this component independently.

Table 17: Summary of sample design recommendations by the specialist biometrician David Baird (AgResearch, Christchurch)

Aim: Detect temporal change in estuarine health
Recommendations
• The use of static sites avoids adding extra spatial variation into future temporal comparisons. The analyses of changes in parameters over time should use either a paired comparison technique, or a repeated measures analysis.
• To minimise spatial variation in the initial set of samples, as many plots as possible should be chosen (a minimum of ten), with only one sample per plot.
• The fixed sampling location within each plot should be chosen at random and recorded as accurately as possible by GPS.
• Subsequent samples should be taken as closely as possible to the original sampling location, but without re-sampling the same area.
• The number of sites within estuaries does not need to be uniform, however, a minimum of three is recommended.
• The size of a site does not need to be fixed, but should be chosen to reflect the variation in local environment (*i.e.* account for variations in environmental gradients). Habitats within sites should be as homogeneous as possible.

6.3.2 Site selection within estuaries

The choice of sites was made using a combination of the knowledge collected through the broad-scale habitat mapping and on-site, specialist expertise as follows:

- broad-scale habitat maps and local knowledge were used to determine broad areas of unvegetated, mid-low water, mud/sand habitat located away from river mouths (mean salinity of overlying water > 20 ppt),
- a representative position within each of the broad areas was chosen to locate each site. Areas of significant vegetation and channel areas were avoided,
- the number of sites selected within each of the reference estuaries was allocated proportionately, based on estuary size, extent of the mud/sandflat habitat, and the number of isolated arms. Large and/or highly branched estuaries were allocated more sites (a maximum of 4), while those that were small or had a single arm, were allocated fewer (Table 18). The reference estuaries ranged in size from 195 to 3500 ha, which is considerably smaller than the Manukau Harbour at 36800 ha studied by Pridmore *et al*. (1990). In that study, six relatively isolated (un-contaminated) sites were chosen, spaced at distances of between 4 and 20 km apart.

Table 18: Number of sites allocated to each reference estuary. Refer to Appendix C for the site GPS coordinates.

Estuary	**Sampling Date**	**Number of Sites**
Otamatea Arm (Kaipara)	20-21/3/01	3
Ohiwa	27/2-1/3/01	4
Ruataniwha	14-15/3/01	3
Havelock	7/3/01	2
Waimea	5-8//01	4
Avon-Heathcote	17-18/2/01	3
New River	12-15/2/01	4
Kaikorai	16/2/01	1

6.3.3 Sampling design within sites

The size of each site was set at 30 x 60 m. Each site was divided into 12 ‘plots’ of equal size (*i.e.* in a grid-like fashion). One sampling station was randomly positioned within each plot and the co-ordinates recorded. Figure 4 is a schematic diagram representing the sampling design within an estuary, using the Avon-Heathcote Estuary as an example.

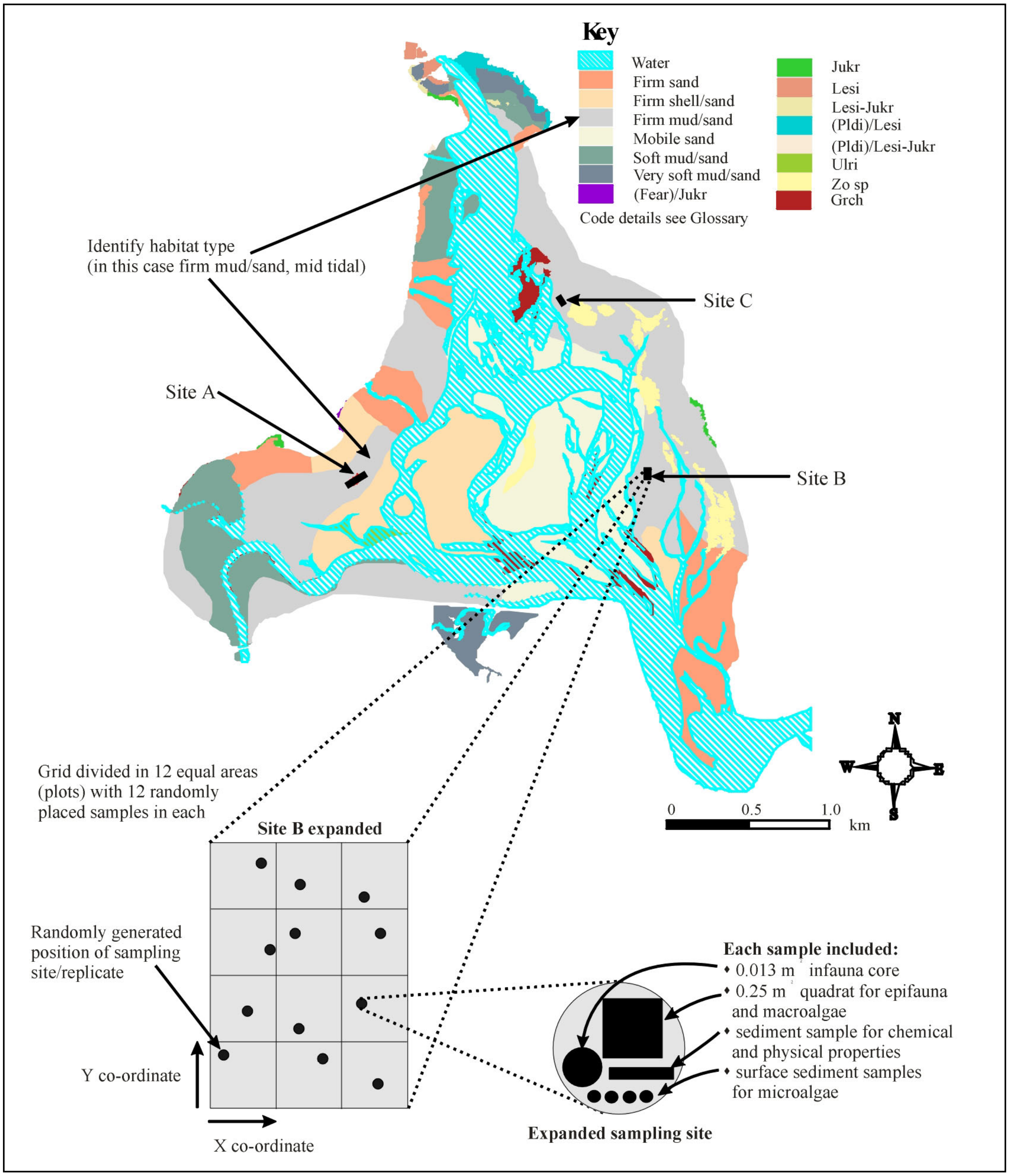
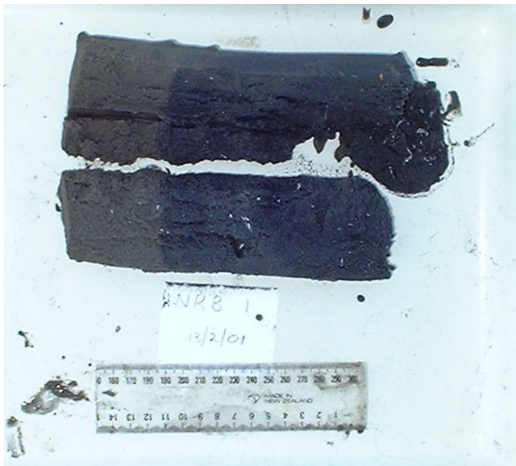


Figure 4: Summary of the sampling strategy applied to each estuary, with a sampling site and station expanded for clarity. The Avon-Heathcote Estuary is used as the example.

The sample collection and analytical procedures adopted for the present study were adapted from those used successfully in a number of previous studies (*e.g.* Gillespie *et al*. 2001a,b). These procedures were as follows:

Photographic Record: Photographs were taken to provide a record of the general site overview and a close-up (~1.5 m) of each plot preferably including a quadrat and label for reference.

Sediment Core Profiles: Representative 62 mm Perspex cores were extruded onto a white plastic tray and split lengthwise (vertically) into two halves along side a ruler. The stratification of colour and texture were described with particular attention to the occurrence of any black (anoxic) zones. Where these occurred, the average depth of the lighter-coloured surface layer was recorded as the Redox Discontinuity Layer (RDL).



An example of a sediment core profile, showing distinct stratification of colour.

Epifauna: (surface-dwelling animals): Epifauna were assessed from twelve replicate 0.25 m^2 quadrats within each site (one randomly placed within each plot). All animals observed on the sediment surface were identified and recorded, and any visible microalgal mat development was noted. Crab burrows were counted as a relative indicator of mud crab populations. Photographs of representative quadrats were also taken.

Infauna: Twelve sediment cores (one randomly placed within each plot) were collected from each site using 130 mm diameter (area = 0.0133 m^2) PVC tubes with 0.5 mm nylon mesh bags affixed to the top to act as a sieve. The tubes were manually driven 150 mm into the sediments, removed with core intact, and the contents were washed through the sieve using seawater from a nearby source. The remaining contents were carefully emptied into a plastic container, preserved in 95% ethanol and transported back to the laboratory for sorting, identification and counting.

Benthic macroalgae: Where a significant macroalgal cover existed, the percent coverage was estimated from the same quadrats but with gridlines dividing it into 36 equally-spaced squares. The number of grid intersections (49 in total, including the outer frame) that overlapped vegetation were counted and the result converted to percent (*i.e.* No. x 2 = %). The method was found to be reasonably consistent with visual estimations of % cover, but less prone to variation among different field personnel.

Benthic microalgae: The primary objective was to identify any major bloom occurrences that could be indicative of eutrophic (highly enriched) conditions. Sediment chlorophyll *a* (chl *a*) and phaeopigment concentrations were analysed as an indicator of the degree of mat development (refer to Technical Box 6.2). Cut-off 10 cc syringe barrels (15 mm internal diameter) were used to collect sediment cores (four per plot). The top 5 mm of the sediment cores were sliced off and mixed in a 50 cc centrifuge tube to obtain one sediment composite sample per plot. Samples were stored on ice and frozen (-20^{o}C) upon return to the laboratory and later analysed as described in Table 19. Additional samples were collected and preserved with Lugol's iodine solution for later microscopic examination to identify dominant taxa.

Technical Box 6.2: Benthic algae

- Microalgae or microscopic algae growing on intertidal flats can often take advantage of excess nutrients under conditions which do not favour macroalgae (e.g. soft mud). In extreme cases of nutrient loading, dense green to orange films or mats can be seen covering the sediment surface.
- Macroalgae or seaweeds (*e.g.* sea lettuce, agar weed *etc.*) can take advantage of excess nutrients resulting in problem accumulations of rotting vegetation.
- Chlorophyll *a* is a primary photosynthetic pigment contained in microalgae (as well as other plants). It is often used as a relative measure of microalgal biomass.
- Phaeopigments (or phaeophytin) refer to a variety of pigments formed as breakdown products of chlorophyll as cells die and decompose.

Chemical and physical analyses: Twelve replicate samples (one 250 ml sample from each plot) were scraped from the top 20 mm of the sediment surface within 300 mm of the infauna cores and analysed for:

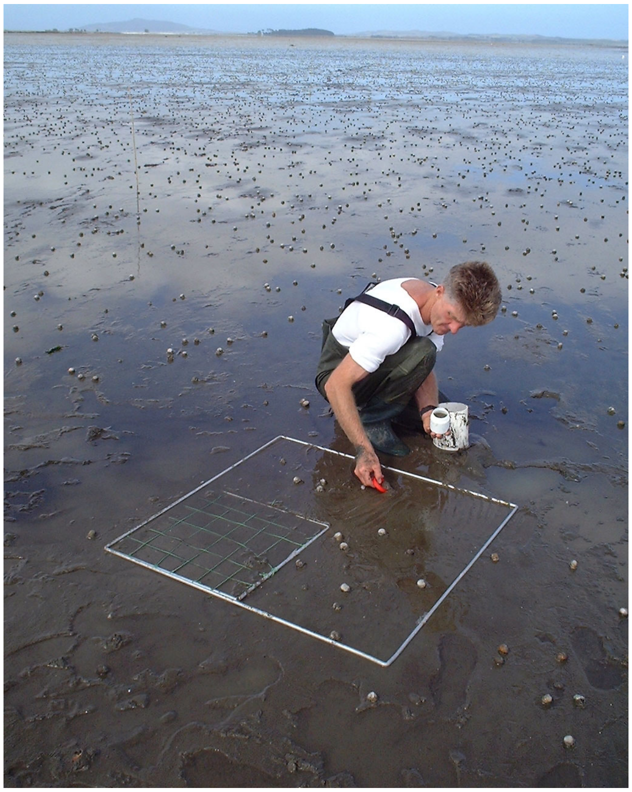
- common trace metal contaminants (copper, cadmium, nickel, lead, zinc, and chromium),
- nutrients (total phosphorus and total Kjeldahl nitrogen),
- ash free dry weight (AFDW, a measure of organic content) and
- particle size distribution (percent gravel, sand, mud).

The methods for analysing each variable are briefly outlined or referenced in Table 19. The six trace metals were chosen because they are the most ubiquitous and commonly used indicators of contaminant sources in New Zealand coastal sediments. Total organic carbon (TOC) may be calculated from the ash-free dry weight (AFDW) according to the following relationship described by Craft *et al*. (1991) for soils from ten salt and brackish water marshes:

$$\text{TOC} = 0.40\ (\text{AFDW}) + 0.0025\ (\text{AFDW})^2$$

Table 19: Methods used for laboratory physical and chemical analyses

Parameter	**Analytical method**
Metals	Perchloric/nitric acid digestion and flame atomic absoption spectrometry (ASTM 3974 Digestion Practice A; AOAC 1995 950.46 modified)
Total Kjeldahl N	Distillation, colourimetric (APHA, 19th Edn. 1995, Method 4500-N Org C)
Total Phosphorus	Colourimetric (APHA, 20th Edn. 1999, Method 4500-P. A, B, E)
AFDW	Weight loss from dry sediment after combustion at 550 °C (APHA 1999, 20th Edn, modified 2540D + E).
Grain size	Wet sieving and calculation of percentage fractions according to dry weight
Chl *a* and Phaeopigments	Extraction with 90% acetone and analysis according to Strickland & Parsons (1968) and Lorenzen (1967)



Dr Barry Robertson sampling sediments for physical and chemical analyses at a sampling station in the New River Estuary.

6.3.4 Data Analysis

Analysis of the environmental data obtained from the fine-scale surveys of the estuaries was approached in three main steps (refer to Figure 5 for a flowchart of the process):

Step 1: Obtaining summary statistics for each variable within each estuary and comparing environmental variables among sites and estuaries.

Step 2: Examining the relationships between environmental variables to determine whether suitable surrogates can replace the measurement of certain variables.

Step 3: Determining the optimum sample size for future surveys to increase the cost-efficiency of monitoring, while ensuring adequate sample collection.

A detailed description of the methodology employed for the data analysis of each step can be found in Appendix C. A summary of the data analyses is presented below.

Data analyses were undertaken using the statistical software SYSTAT® (v.10), PRIMER (v.5.1.2), Excel (Microsoft® 2002) and various statistical tests described in Zar (1999). Differences were considered to be significant when probabilities were <0.05 ($\alpha = 0.05$) unless otherwise specified. In order to facilitate interpretation, the chemical data were presented in two ways. Firstly, as concentration per mass of whole sediment, and secondly they were normalised by assigning the mass of each of the constituents (organic matter, nutrients or metals) to the mud fraction alone. This was undertaken because it is recognised that these sediment constituents are generally higher in muddy sediments. Metals in particular are known to be closely associated with clay minerals in the fine sediment fraction through sorption. Thus differences in contaminant concentrations in sediments of different textures can be a reflection of sediment grain size rather than the extent of contamination (Grant & Middleton 1998). Normalising assumes that 100% of the organic matter, nutrients, and trace metals are associated with the mud fraction of the sediments and the results are interpreted as the concentration of the constituent that would be present if the sample was comprised of 100% mud.

The normalised results can be used as an ‘alarm bell’ to indicate the extent to which the main vehicle of contaminant entry to the estuary (*i.e.* mud <63 μm) may be contaminated compared with other estuaries and other sites in the same estuary. In cases where the mud fraction appears to be contaminated, direct analyses of that fraction may be required for confirmation, along with further investigation to identify the source(s). It is important to note that **normalised** data should **not** be used to determine whether sediment quality guidelines have been exceeded.

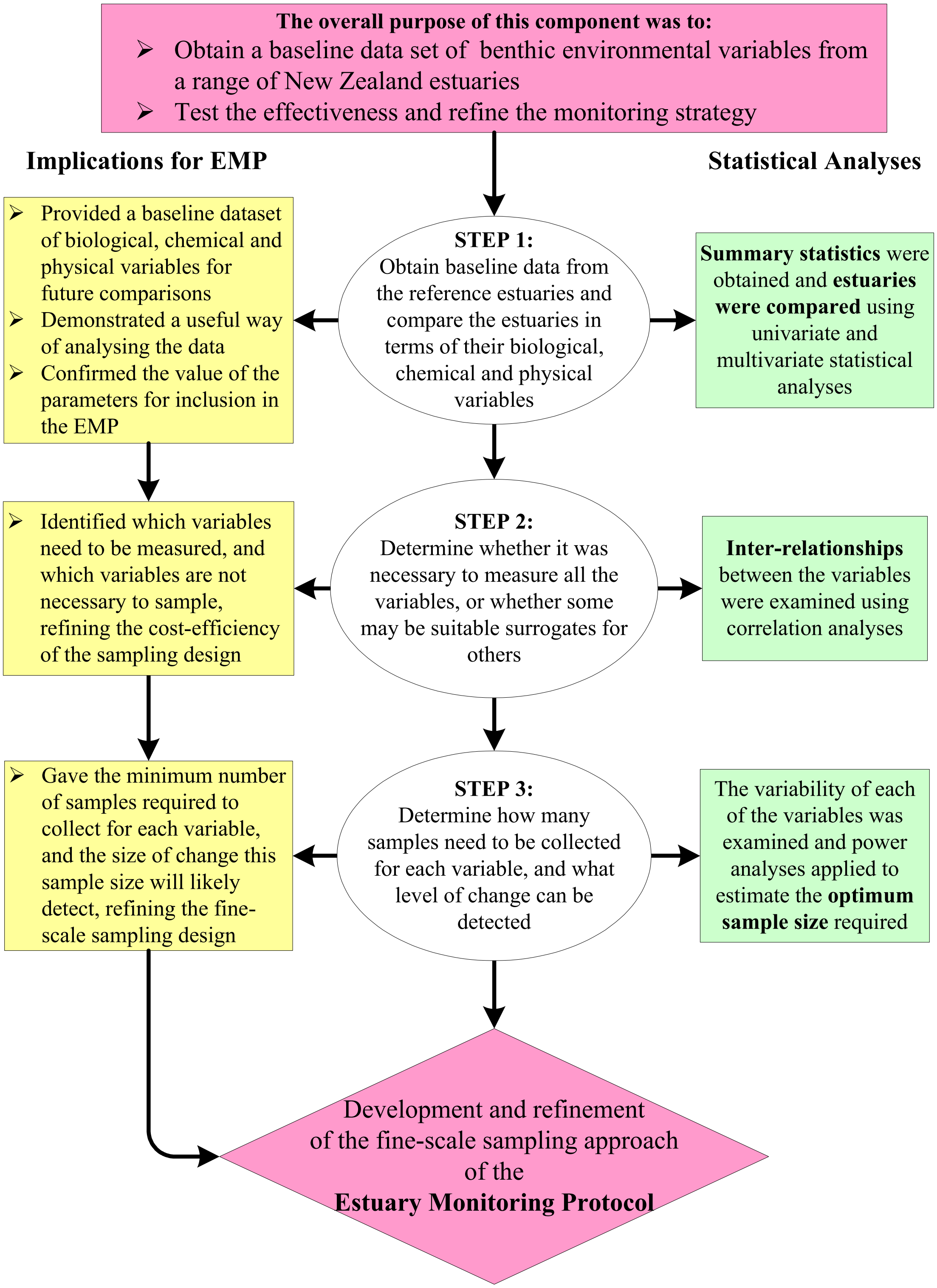


Figure 5: Approach to data analysis for the development of the fine-scale component of the EMP.

6.3.5 Step 1: Comparison of reference estuaries

A comparison of the environmental variables measured in the eight REs provides a preliminary baseline dataset for future comparisons, demonstrates a useful way of analysing and displaying these data, as well as indicating the usefulness of the chosen variables as indicators of estuarine health/condition. For the complete methodology adopted in Step 1, refer to Appendix C.

Step 1 Methodology: Summary of analyses

- Key summary statistics were obtained for each environmental variable (mean, variance, and 95% confidence intervals) and displayed using bar graphs.
- Infauna and epifauna abundance and richness were expressed as the number of taxa encountered per core (0.013 m^2) and per quadrat (0.25 m^2), respectively.
- The biological communities were grouped according to their taxonomic levels. Groups that represented < 1% of the total number of animals encountered were pooled to form a composite group, termed ‘Others’.

Univariate analyses

- Data were examined for normality and for homoscedasticity and, when necessary, were appropriately transformed to satisfy the assumptions of analysis of variance (ANOVA).
- A mixed model, nested ANOVA was used to compare the degree of variation among estuaries with the degree of variation within estuaries for each parameter. This approach was limited to estuaries with two or more sites, therefore excluding Kaikorai from this analysis.
- Multiple, one-way ANOVAs were used to determine which estuaries contained the most site to site variation, and for which variables.

Multivariate analyses

- Environmental data were appropriately transformed, and examined in a site-averaged form.
- Non-parametric multidimensional scaling (MDS) was used to produce dendrograms and ordination plots to depict any similarities among sites and estuaries in their macro-invertebrate composition.
- The significance of differences was tested using an analysis of similarities (a 2-way nested ANOSIM).
- Ordinations of the environmental data were produced using principal components analysis (PCA) for both normalised and un-normalised mean site data.

6.3.6 Step Two: Examining the relationships between environmental characteristics

Inter-relationships between the environmental variables were investigated to determine if it was necessary to measure all those included in the present study, or whether some were strongly correlated enabling one to be a surrogate for the other in future surveys. For the complete methodology adopted in Step Two, refer to Appendix C.

Step 2 Methodology: Summary of analyses:

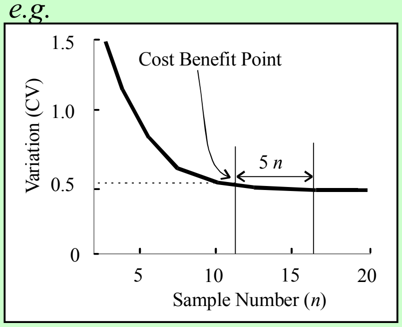
- Data were appropriately transformed and examined both prior to, and following normalisation to mud content.
- The inter-relationships between the environmental and biological variables were initially examined using Pearson product-moment correlation coefficients, compared at both the replicate and site-average levels, identifying highly correlated variables ($r > 0.85$).
- The environmental data were also examined using a principle component analysis (PCA) to explore the similarity of the sites based on their environmental characteristics. (For an explanation of ordination procedures see Technical Box A8, Appendix C).
- Relationships were further examined using the BIOENV procedure, which identified the combination of environmental variables that best grouped the sites, in a manner consistent with the arrangement of sites according to the biological assemblages (MDS ordinations).
- Prior to the BIOENV procedure, one of each pair of the highly correlated variables was removed from the analysis under the assumption that the other was a suitable surrogate. The variable to be removed was selected based on the relative loss of interpretative power if removed, the ease of sampling, and the cost of sample analysis.

6.3.7 Step 3: Determining the optimum sample size

Optimising sample numbers using a 'cost-benefit' approach allows the sampling regime to be designed around the variability of each individual characteristic, providing the minimum number of samples that will enable confident conclusions to be drawn from future surveys. For the complete methodology adopted in Step 3, refer to Appendix C.

Step 3 Methodology: Summary of analyses:

- Optimum sample size analyses were carried out using combinations of the 12-replicate data sets (*i.e.* 12 samples per variable per site) that were collected during the present study.
- Coefficient of variation (CV) was used as the measure of variance.
- The CV was calculated for randomly chosen combinations of the data from each of the estuary data sets. Sites within an estuary were pooled prior to drawing the combinants if the means and CVs for a variable were not significantly different (one way ANOVA). If the sites were significantly different, the site most dissimilar was removed from the analyses and the remaining sites were re-tested.
- A maximum of 1000 randomly generated combinations of the data was drawn from the replicate pool of each variable using Matlab according to the methods of Bros & Cowell (1987). The mean CV of the combinations was calculated along with the 5th and 95th percentiles, which were used to estimate the variability of CV for each variable.
- The optimum sample number for each variable was explored by examining estimates of sample variation with increasing sample number. The Cost Benefit Point (CBP), defined as the point (*n*) when significant gains in CV are not made with a further 5*n* increase in sample size, was identified: *e.g.*



- The level of change able to be detected using sample sizes based on CBP was investigated using two power analysis models.

6.4 Results

The results section of the fine-scale monitoring contains three parts, corresponding to the three steps outlined in Figure 5. Due to the highly technical nature of much of the results, the majority of the statistical outputs and detailed results (*i.e.* tables and graphs) are presented in Appendix C. Summary results are provided in this section (with some cross-referencing to the results in the Appendix), followed by a Summary Box with the major findings of each step and the implications for the EMP. It is envisaged that the appendix will be used for clarification of statistical analyses applied, and to provide defensibility to the major findings and implications for the EMP.

Step 1: Comparison of estuaries

A comparison of the reference estuary (RE) data sets provided in Step 1 allows the reader to compare the current state of some New Zealand estuaries with overseas estuaries, and with the existing environmental quality guidelines (*e.g.* ANZECC & ARMCANZ 2000). It also demonstrates a common way of using, displaying and interpreting the data obtained during estuary monitoring. The findings from the REs allow the usefulness of the variables measured in depicting the overall condition of an estuary to be assessed. Additionally, it provides a baseline data set that can be used as a benchmark, against which future monitoring surveys can be compared.

Summary results for each RE are presented individually in Appendix C. A comparison of the estuary characteristics determined in the present study, as well as comparisons with other New Zealand and overseas estuary studies are presented below. The summary results are described and displayed in the first part of this section, after which, the major findings of the statistical analyses are presented followed by a *Summary Box* outlining the major findings and recommendations.

6.4.1 Physical, chemical and microalgal characteristics

The sediment particle size distributions, organic contents and nutrient and photosynthetic pigment concentrations were variable between the eight reference estuaries, as well as within the estuaries themselves (Figure 6). Sand was the dominant substrate size in all of the estuaries, except in the Otamatea Arm of the Kaipara Harbour, which was dominated by mud at two of the three sites. The sediments at most estuary sites had an organic content (AFDW) of 1 to 2 %. However, the Kaikorai site and the Otamatea sites contained an average of 5.1 and 5.7 % organic content, respectively.

The link between a high proportion of mud and high organic and nutrient concentrations in the sediment was evident (Figure 6). An elevated mud fraction coincided with elevated organic and nutrient contents, particularly for the Otamatea Arm and the Kaikorai sites. Normalising the data to 100 % mud content typically resulted in an inverse pattern to the raw data, where the estuary sediments that contained a small proportion of mud had higher relative concentrations of organics and nutrients within that fraction. Manipulating the data in this way provides some insights into the potential for the accumulation of contaminants into the mud fraction thereby suggesting the possibility of some source(s) of contaminated sediments entering the estuary. It should be stressed, however, that no conclusions can be drawn from the 'normalised' data without verification through direct analyses of the mud fraction.

Sediment chlorophyll *a* and phaeophytin concentrations were analysed as relative measures of photosynthetically active and senescent microalgal biomass, respectively. The primary objective was to identify any major bloom occurrences that could be indicative of eutrophic (highly enriched) conditions. Considerable variation in pigment concentrations was observed, both among and within estuaries, however conditions of extreme enrichment (*e.g.* chlorophyll *a* ≥200 mg m^{-3}) were not observed (Figure 6). Pigment concentrations indicated low to moderate microalgal mat development within a majority of the reference estuaries. The densest microalgal coverage, as indicated by pigment concentrations and sediment colouration, was observed at a site in the Avon-Heathcote estuary that was likely to be affected by the discharge from the Christchurch Wastewater Treatment Plant.

With one notable exception, the microalgal species composition of the surface sediments was generally similar among all estuaries and sites (Table 20). The exception was a site in the Avon-Heathcote estuary that contained an assemblage influenced by an oxidation pond discharge (*i.e.* dominated by freshwater taxa characteristic of oxidation pond communities). Microalgal community structure is a good example of a potential indicator of estuarine health that requires considerably more development. Although it clearly identified the effects of an oxidation pond discharge at one location, it was not generally useful in most situations. The present survey does provide one of the first comparisons of microalgal community structure in a variety of estuary environments in New Zealand. In one other comparative survey, the potentially toxic dinoflagellate species, *Pfeisteria shumwayae* was identified at three of the REs; Kaipara, Havelock and New River estuaries (Rhodes *et al.* 2002).

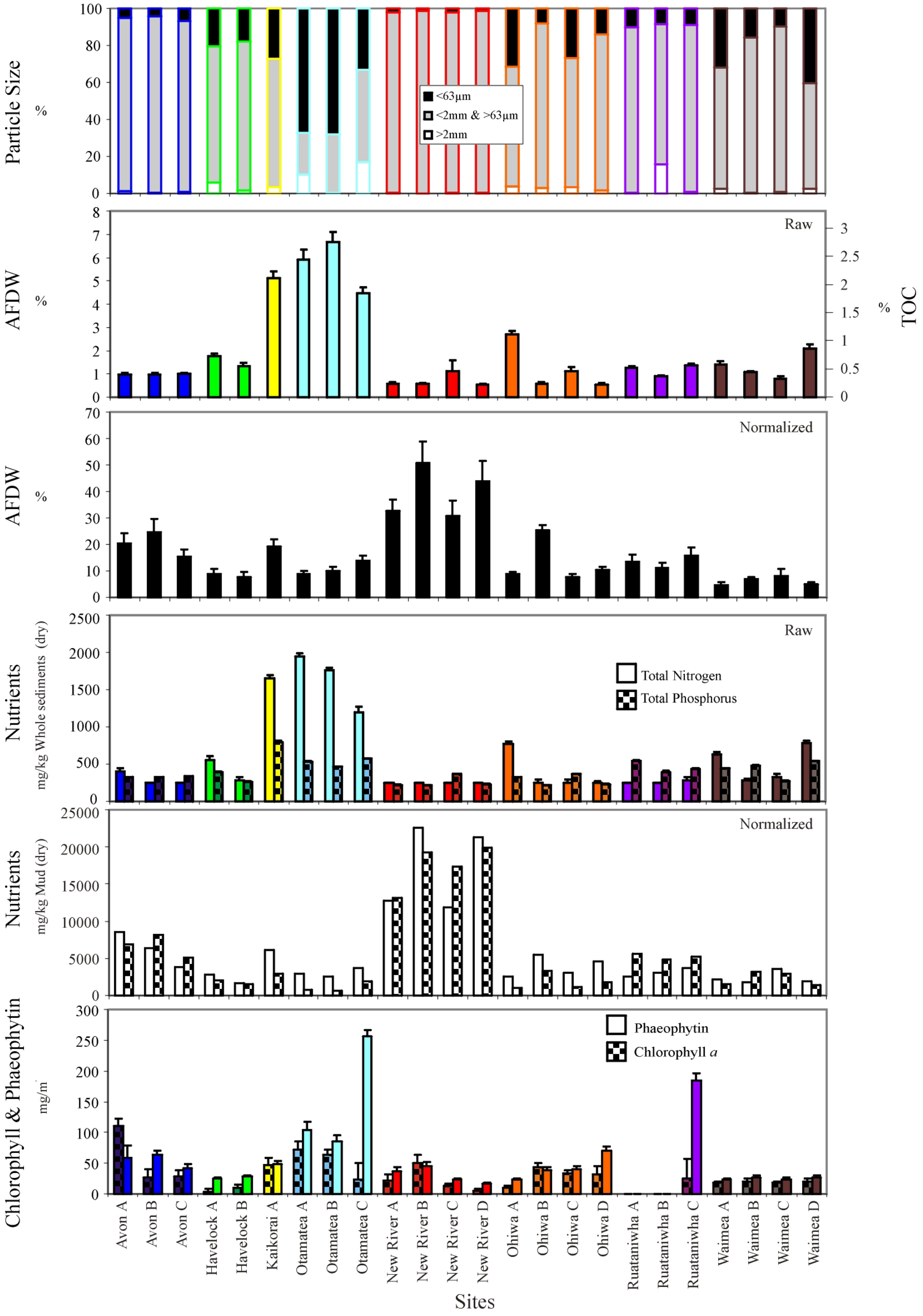


Figure 6: Site-averages of sediment particle sizes, ash-free dry weight (AFDW) and TN, TP, chlorophyll *a* and phaeophytin concentrations in sediments from the eight reference estuaries. Nutrient and AFDW data, normalised to 100% mud content, are included for comparison.

Table 20: Microalgal genera observed in reference estuary sediments (all sites). PD = pennate diatom, CD = centric diatom, Cy = cyanobacteria, Eu = euglenoid, Gr = green alga.

Genus	**Group**	**(Number of Sites)**	
		Dominant or Co-dominant	**Present**
Achnanthes	PD	1	11
Amphora	PD	0	4
Bacillaria	PD	0	1
Chaetoceros[1]	CD	0	1
Chlorococcum[2]	Gr	0	1
Entomoneis	PD	0	6
Euglena	Eu	2	4
Licmophora	PD	0	1
Melosira	CD	1	6
Navicula	PD	0	14
Nitzschia	PD	1	9
Oscillatoria[2]	Cy	1	1
Pleurosigma/Gyrosigma	PD	9	12
Scenedesmus[2]	Gr	0	1
Thalassionema	PD	0	5
Thalassiosira	CD	0	1

1. Generally found in the water column (phytoplanktic).
2. Freshwater taxa typically found in oxidation pond environments.

The large sigmoid-shaped pennate diatom, tentatively identified as *Pleurosigma* or *Gyrosigma* sp., appeared to be dominant or co-dominant in nine of 18 sediments tested. Other genera, such as *Achnanthes*, *Euglena*, *Melosira* and *Nitzschia,* were also occasionally dominant or co-dominant.

The levels of nutrients, organic content and photosynthetic pigment concentrations of sediments of comparable mud content at the eight REs, were within a range reported for other estuarine sites in New Zealand (Table 21). Comparison of molar N:P ratios amongst the REs and other New Zealand estuaries suggest that nitrogen was, as a rule, relatively more limiting for photosynthetic production than phosphorous. This was particularly the case at the sandier sites. Table 21 provides an overview of where any particular site is positioned within an enrichment continuum of sediment characteristics extending from the relatively natural Delaware Estuary (largely native and exotic forestry catchment) through moderately enriched sites affected by a variety of nutrient sources, to a highly enriched site affected by a freezing works waste discharge. Sites within the REs ranged from unenriched to moderately enriched in comparison to the other sites. Kaikorai and Kaipara (Otamatea Arm) sediments (for example) appeared to be moderately enriched, particularly with respect to nitrogen, in comparison to the other locations.

Table 21: Comparison of average physico-chemical characteristics of sediments from the eight estuaries examined in this study and some other New Zealand estuarine sites.

	%Mud %	TN mg kg^{-1}	TP mg kg^{-1}	Molar N:P	AFDW %	Chl *a* mg m^{-2}	Phaeo mg m^{-2}
Present study							
Otamatea Arm (Kaipara)	56	1630	526	6.8	7	53	149
Ohiwa	20	650	278	5.1	3	30	43
Ruataniwha	9	263	458	1.3	1	26	185[i]
Waimea	25	506	433	2.6	2	19	25
Havelock	19	421	330	2.8	2	6	27
Avon-Heathcote	5	301	327	2.0	1	56	55
Kaikorai	27	1650	799	4.6	5	46	49
New River	2	250[g]	268	2.1	1	24	31
Other NZ sites							
Tamaki A (E1) [a]	48	110					
Tamaki B (E2) [a]	86	200					
Tamaki C (E3) [a]	54	250					
Tamaki D (E4)[a]	67	520					
Tauranga Hbr (10 m from outfall)[b]	15	650[h]	275	5.2			
Tauranga Hbr (1 km from outfall)[b]	15	460 [h]	175	5.9			
Delaware Inlet (4 sites) [c]	7	303	540	1.2	2	33	19
Delaware Inlet (5 sites) [c]	73	1260	716	3.9	6	29	20
Nelson Haven (6 sites) [d]	23	347	403	1.9	2	39	25
Moutere Inlet (5 sites) [e]	>50	1305	648	4.5	6		
Moutere Inlet (13 sites) [e]	<50	546	419	2.9	2		
Waimea (enriched site) [f]	83	4340	1063	9.0	9	155	134

a Sites positioned from inner (E4) to outer (E1) estuary locations in heavily urbanised area (Thompson 1987)
b Subtidal on open coast (Roper 1990)
c Largely undisturbed estuary near Nelson (Gillespie & MacKenzie 1990)
d Slightly modified estuary near Nelson; affected by urban stormwater runoff, roading, marina development (Gillespie & MacKenzie 1990)
e Slightly modified estuary near Motueka; affected by food processing industry wastes, urban runoff (Gillespie *et al.* 1995)
f Site affected by a high nutrient freezing works discharge (Gillespie & MacKenzie 1990)
g Below detection limit (250 mg kg^{-1})
h Total Kjeldahl Nitrogen (does not include nitrate/nitrite)
i Probable artifact of decomposing terrestrial plant debris

The trace metal concentrations found in the sediments of the reference estuaries are presented in Figure 7. The concentrations were generally lower than the ANZECC ISQG-Low guideline values, with the exception of nickel and chromium in sites from the Waimea and Havelock estuaries. Since both of these estuaries receive catchment runoff from the mineral-rich Dun Mountain region, this is likely to be a natural condition. Kaikorai estuary sediments contained elevated levels of zinc, lead and chromium relative to the other estuaries, although the concentrations did not exceed the ANZECC ISQG-Low limits.

When normalised to 100% mud content, the trace metal concentrations at the estuary sites showed a different pattern to the raw data (Figure 8). All four sites at the New River estuary had high trace metal concentrations within the mud fraction compared to the other estuaries in the study. Lead and zinc were also elevated at the Avon-Heathcote and Kaikorai estuaries. The patterns of trace metal concentration were clearly related to the proportion of mud in the sediment (Figure 6 & Appendix C2 Table A40/A41). Further investigation would be required to confirm that the metal loadings in the three estuaries in question are indeed concentrated within the mud fraction. Once this has been established, the question of contaminant source(s) could be addressed.

Technical Box 6.3 Sediment Quality Guidelines

Sediment quality guidelines aim to predict 'acceptable' levels of contaminants in sediment, above which adverse ecological effects are possible. New Zealand has recently published national guidelines for sediment quality (ANZECC & ARMCANZ 2000) based on international guidelines (*eg.* PSDDA & U.S. Army Corps of Engineers 1989, Long & Morgan 1991). The criteria are listed as Interim Sediment Quality Guideline- Low (ISQG-Low) and Interim Sediment Quality Guideline- High (ISQG-High) and have two distinct threshold levels under which biological effects are predicted. The criteria use statistical models to determine the levels at which effects can be predicted with a degree of certainty. The lower threshold (ISQG-Low) indicates a possible biological effect while the upper threshold (ISQG-High) indicates a probable biological effect. It should be noted however, that the guidelines are limited to certain individual analytes and do not take into account the synergistic effects of combined contaminants within the sediment. Guidelines are used as part of the risk assessment, and if exceeded, additional testing may be required.

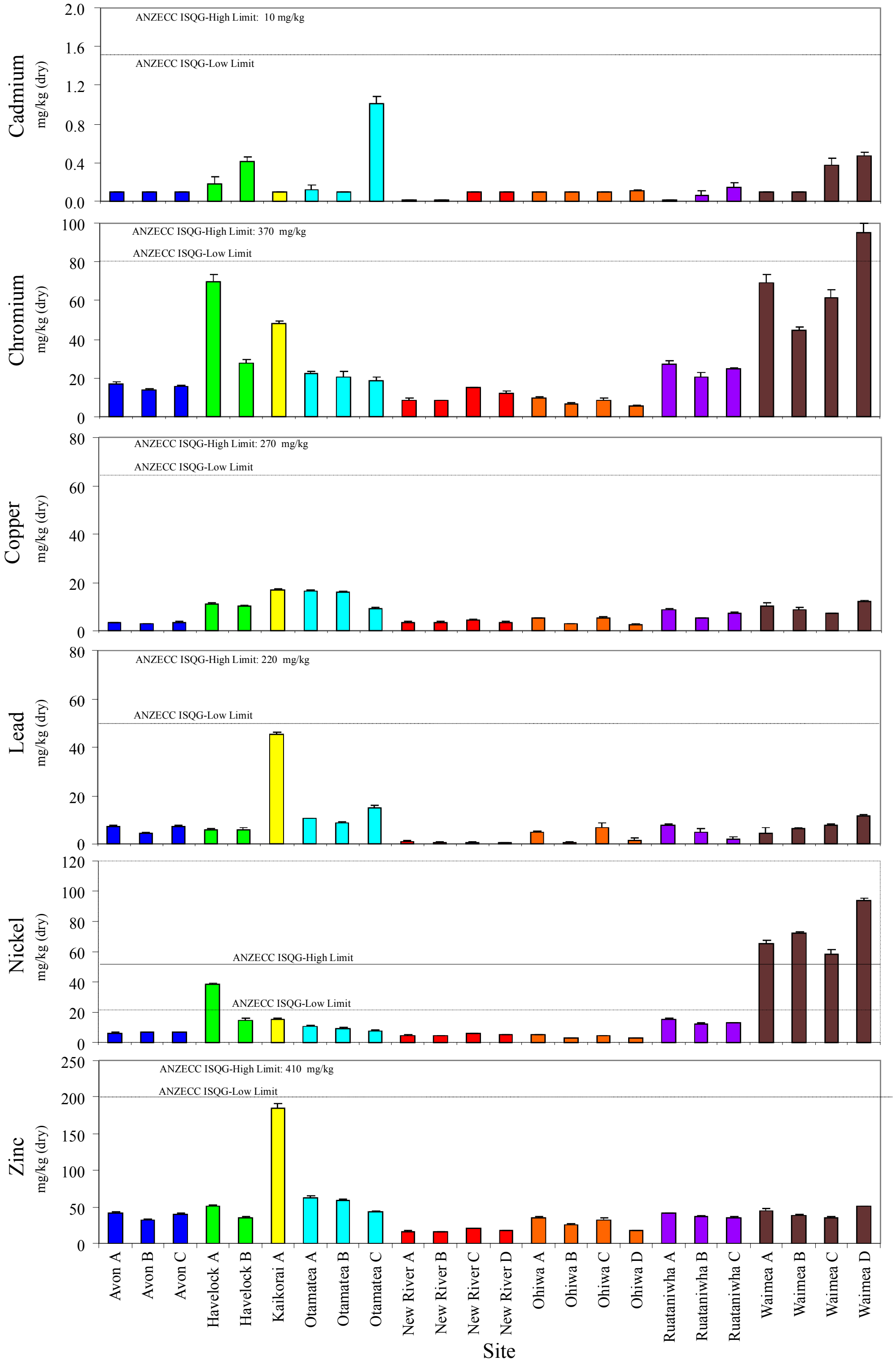


Figure 7: Mean (+ 95 % CI) of cadmium, chromium, copper, lead, nickel and zinc concentrations in sediments at sites within the eight reference estuaries. ANZECC ISQG-High and ISQG -low guideline values are given for each.

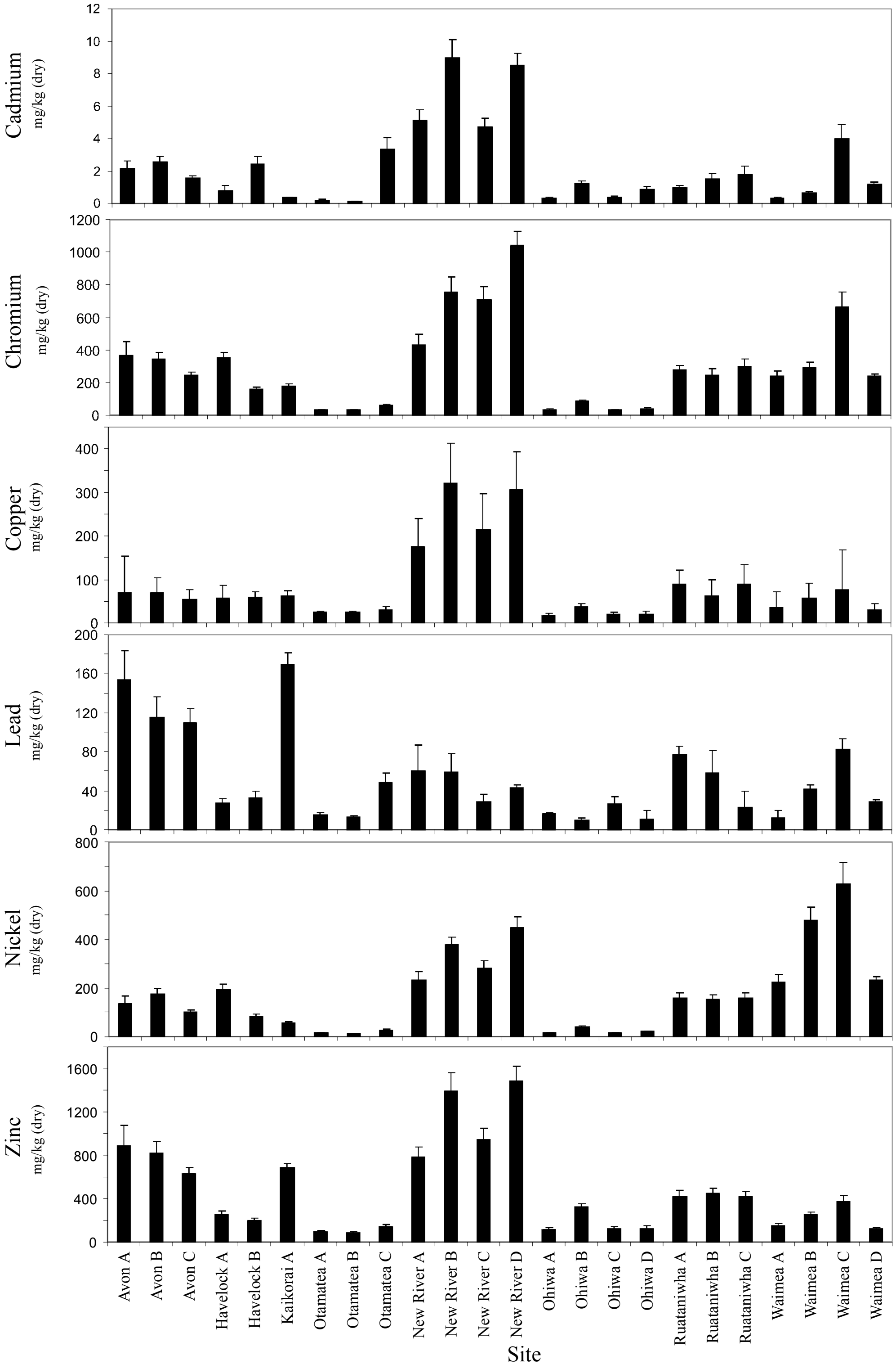


Figure 8: Mean (± 95 % CI) of cadmium, chromium, copper, lead, nickel and zinc concentrations in sediments at sites within the eight reference estuaries, normalised to 100% mud content.

The eight REs in the present study contained sediment trace metal concentrations that are, in most cases, similar to (or lower than) the range reported for a variety of other New Zealand estuaries (Table 22). The only exception was the elevated average nickel concentration observed in Waimea Estuary sediments and attiributed to a mineral belt within the catchment. In a comparison with reported values for some overseas estuaries, however, the New Zealand estuaries often had much lower sediment trace metal concentrations. For all the metals measured, there was at least one overseas estuary with reported sediment concentrations far exceeding the ANZECC ISQG- high guidelines.

Table 22: Average concentrations of heavy metals in sediments from the eight reference estuaries compared to other New Zealand estuaries, a selection of overseas estuaries that have been contaminated to varying degrees, and ANZECC ISQG-High and ISQG -low guideline values. Some values drawn from other studies are approximate as they were estimated from figures.

		Cd mg kg^{-1}	**Cr** mg kg^{-1}	**Cu** mg kg^{-1}	**Pb** mg kg^{-1}	**Ni** mg kg^{-1}	**Zn** mg kg^{-1}
Present study	ANZECC ISQG-Low	*1.5*	*80*	*65*	*50*	*21*	*200*
	ANZECC ISQG-High	*10*	*370*	*270*	*220*	*52*	*410*
	Otamatea Arm	0.4	20.5	13.8	11.4	9.4	54.5
	Ohiwa	0.1	7.4	4	3.4	3.9	27.7
	Ruataniwha	0.1	24	7.1	4.7	13.7	37.5
	Waimea	0.3	67.6	9.6	7.4	72.5	41.8
	Havelock	0.3	48.8	10.7	5.6	26.5	43
	Avon-Heathcote	0.1	15.6	3.2	6.3	6.6	38.3
	Kaikorai	0.1	48.4	16.8	45.3	15.6	184.2
	New River	0.1	11.1	3.8	0.7	5	17.1
Other NZ sites	Tamaki A (E1) [a]		14.5	27.8	132.1	56.9	136.1
	Tamaki B (E2) [a]		20.6	26.1	72.9	6.6	167
	Tamaki C (E3) [a]		17.3	29.4	69.7	9.3	173
	Tamaki D (E4)[a]		35.9	38.5	145.2	12.8	233
	Manukau (rural catch)[b]	0.03		20	9	15	114
	Manukau (industrial catch) [b]	0.25		90	58	14	285
	Waitemata Harbour [h]	<0.5	52	60	65	28	161
	Otago (mid-upper harbour)[c]	0.26	21	17	19	9.7	110
	Lampton Harbour, Wellington [d]		91	68	183	21	249
	Poriora Harbour, Wellington [e]		20	48	93	20	259
	Aparima Estuary[f]	0.067	15	12	11	10	49
	Mataura Estuary[f]	0.024	7.1	6.6	6.2	6	27
Overseas sites	Delaware Bay, USA [g]	0.24	27.8	8.3	15		49.7
	Lower Chesapeake Bay, USA[g]	0.38	58.5	11.3	15.7		66.2
	San Diego Harbour, USA [g]	0.99	178	218.7	51		327.7
	Salem Harbour, USA [g]	5.87	2296.7	95.1	186.3		238
	Rio Tinto Estuary, Spain[f]	4.1		1400	1600		3100
	Restronguet Estuary, UK[f]	12	1060	4500	1620		3000
	Nervión Estuary, Spain[i]	0.2-15	50-300	50-350	50-400	20-100	200-2000
	Sorfjord, Norway[f]	850		12000	30500		118000

Sources: a Thompson (1987), b Roper *et al.* (1988), c ORC. 1998, d Stoffers *et al.* (1986), e Glasby *et al.* (1990) , f Robertson (1995), g Kennish (1997), h Jezus Belzunce *et al.* (2001).

6.4.2 Biological characteristics

Infaunal species richness and abundance were dominated by polychaetes and bivalves at six of the eight estuaries in the current study, with polychaetes being the most dominant in many of the study sites (Figure 9). Avon-Heathcote estuary was largely dominated by the polychaete *Aonides* sp. (approximately 86% of the total abundance of the Avon-Heathcote sites). By contrast, the infauna at Kaikorai estuary, and at one of the four New River estuary sites, was dominated by amphipods and gastropods, respectively.

Infaunal species richness was lowest at the Kaikorai estuary (13 species present in total) and highest at Ohiwa estuary (53 species), with the remaining estuaries having close to the total estuary average of 37 species (Figure 9B). In addition to the dominant polychaetes and bivalves, several other species were commonly found at the reference estuaries; including gastropods (marine snails), nemertea (ribbon worms), and oligochaetes (worms).

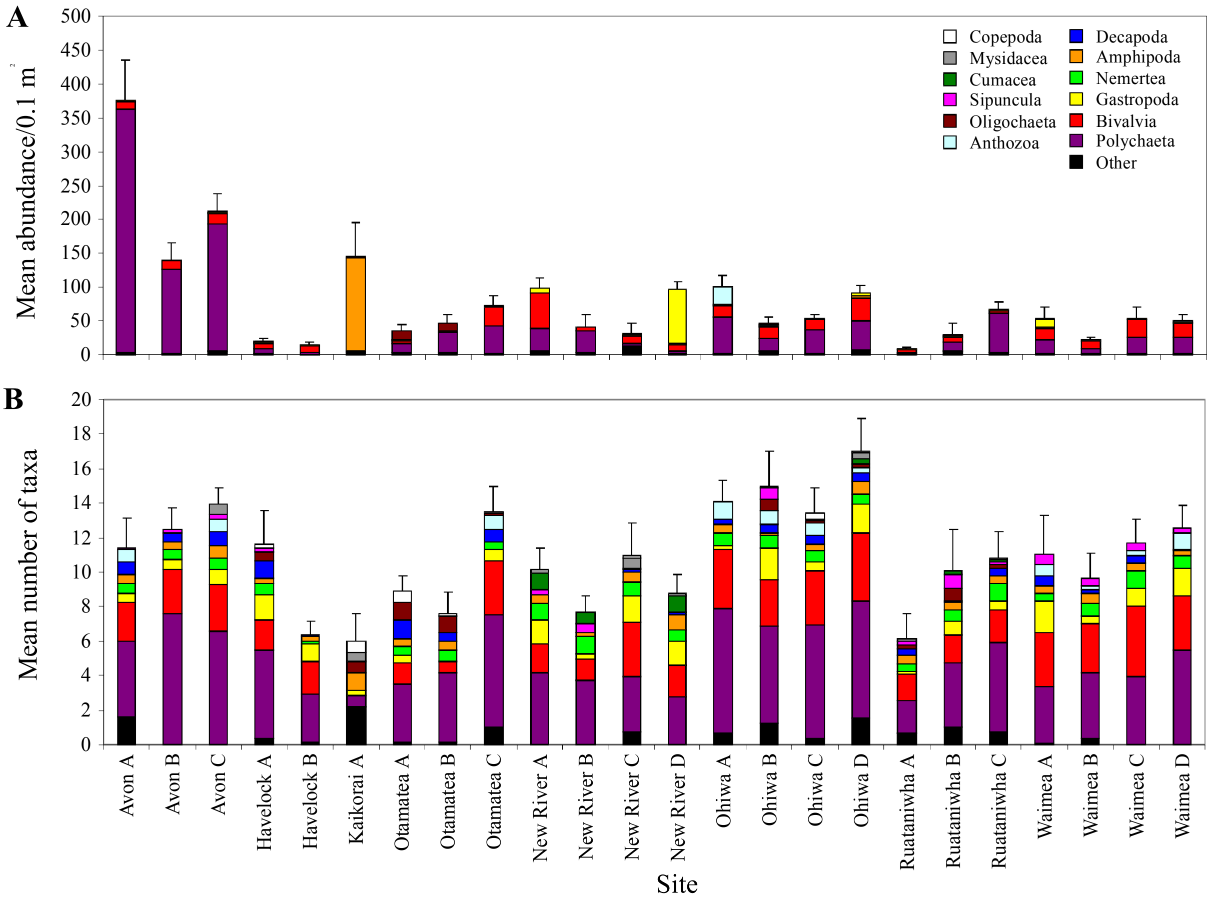


Figure 9: Mean (± 95 % CI) of infauna abundance (A) and species richness (B) at sites within the eight reference estuaries.

The infaunal species richness of the REs was very similar to other New Zealand estuaries (Table 23). Infaunal abundance was also similar to other estuaries around New Zealand, although the average abundances from sites within the Avon-Heathcote and Kaikorai estuaries were higher than most others.

Table 23: A comparison of estuary infauna studies, showing the number of replicates, core size, mean number of individuals, mean number of taxa per core and total number of taxa per site for the eight estuaries examined in the current study and from other New Zealand nearshore sediments.

	n	**Core size m^2**	**No. Individuals / 0.1 m^2**	**No. taxa / core**	**No. taxa /site**
Present study					
Kaipara (Otamatea Arm)	36	0.013	51.4	10.0	24.0
Ohiwa	48	0.013	73.1	14.9	34.3
Ruataniwha	36	0.013	34.8	9.0	26.3
Waimea	48	0.013	44.7	11.2	25.0
Havelock	24	0.013	17.3	9.0	28.5
Avon-Heathcote	36	0.013	242.7	12.6	27.3
Kaikorai	12	0.013	249.5	6.0	14.0
New River	48	0.013	67.0	9.4	20.5
Other NZ sites					
Tamaki A (E1)[a]	3	0.05	13.7	6.30	8.0
Tamaki B (E2)[a]	3	0.05	7.94	4.70	4.0
Tamaki C (E3)[a]	3	0.05	2.0	2.70	2.0
Tamaki D (E4)[a]	3	0.05	8.9	4.00	4.0
Tauranga 0-200 m from dc[b]	3	0.10	25.0	33.0	
Tauranga 600-1000 m[b]	3	0.10	25.0	38.0	
Gisborne 100 m from dc (Coastal)[c]	3	0.10	45.0	15.0	
Hastings 100 m from dc (Coastal)[c]	3	0.10	120.0	20.0	
Gisborne 1600 m from dc[c]	3	0.10	5.0	22.0	
Hastings 3200 m from dc[c]	3	0.10	26.0	32.0	
Manukau (rural catch)[d]	3	0.017	73.4	10.0	
Manukau (Indust. Catch)[d]	3	0.017	11.2	5.0	
Manukau (DSIR ave. 1988)[e]	12	0.013	33.8	12.0	
Tauranga (Welcome Bay)[f]	30	0.013		10.33	
Tauranga (Blue Gum Bay)[f]	30	0.013		6.47	
Tauranga (Katikati)[f]	30	0.013		10.13	
Tauranga (Waikareao estuary)[f]	30	0.013		10.33	
Maketu (Site 1)[f]	30	0.013		8.17	
Maketu (Site 3)[f]	30	0.013		3.67	
Waihi (Site 1)[f]	30	0.013		9.13	
Ohiwa Harbour (Site 1)[f]	30	0.013		10.97	
Ohiwa Harbour (Site 4)[f]	30	0.013		3.50	
Whakatane (site 1)[f]	30	0.013		4.00	
Opotiki (Site 1)[f]	30	0.013		4.30	
Waiotaha[f]	30	0.013		6.33	
Otago (mid upper harbour)	3	0.008	292.8		12

Sources: [a]Thompson (1987), [b]Roper (1990), [c]Roper *et al.* (1989), [d]Roper *et al.* (1988), [e]Water Quality Centre (1988), [f]Park (1995)

The epifauna and macroalgal cover at the eight REs are presented in Figure 10. Epifauna and macroalgae were not sampled from the Kaikorai site and epifauna were not sampled from Otamatea Arm sites A and B. The mean abundance of epifauna was highly variable, both between and within estuaries. For example, the four New River sites ranged from 6 to > 50 individuals per quadrat. All estuaries sampled were dominated by gastropod and bivalve species. Most sites contained a moderate range of taxa, from 7 to 13 species. Refer to Box A.7, Appendix C for further detail on the use of epifauna in estuary monitoring.

Macroalgal distribution was highly variable between estuaries and between sites within estuaries in the present study (Figure 10B). Waimea Estuary Site D contained the greatest percentage cover of macroalgae, with *Gracilaria chilensis* and *Ulva lactuca* covering an estimated 29% of the substratum within quadrats. Site C of the New River Estuary contained the widest range of species compared to the other estuaries in this study (a combination of *Enteromorpha* sp., *G. chilensis*, *U. lactuca* and unidentified red macroalgae). Havelock Estuary site A contained *G. chilensis* at approximately 5% cover while site B contained no algal cover, consistent with the greater abundance and diversity of invertebrates at site A (refer to Tables A42 and A43 in Appendix C).

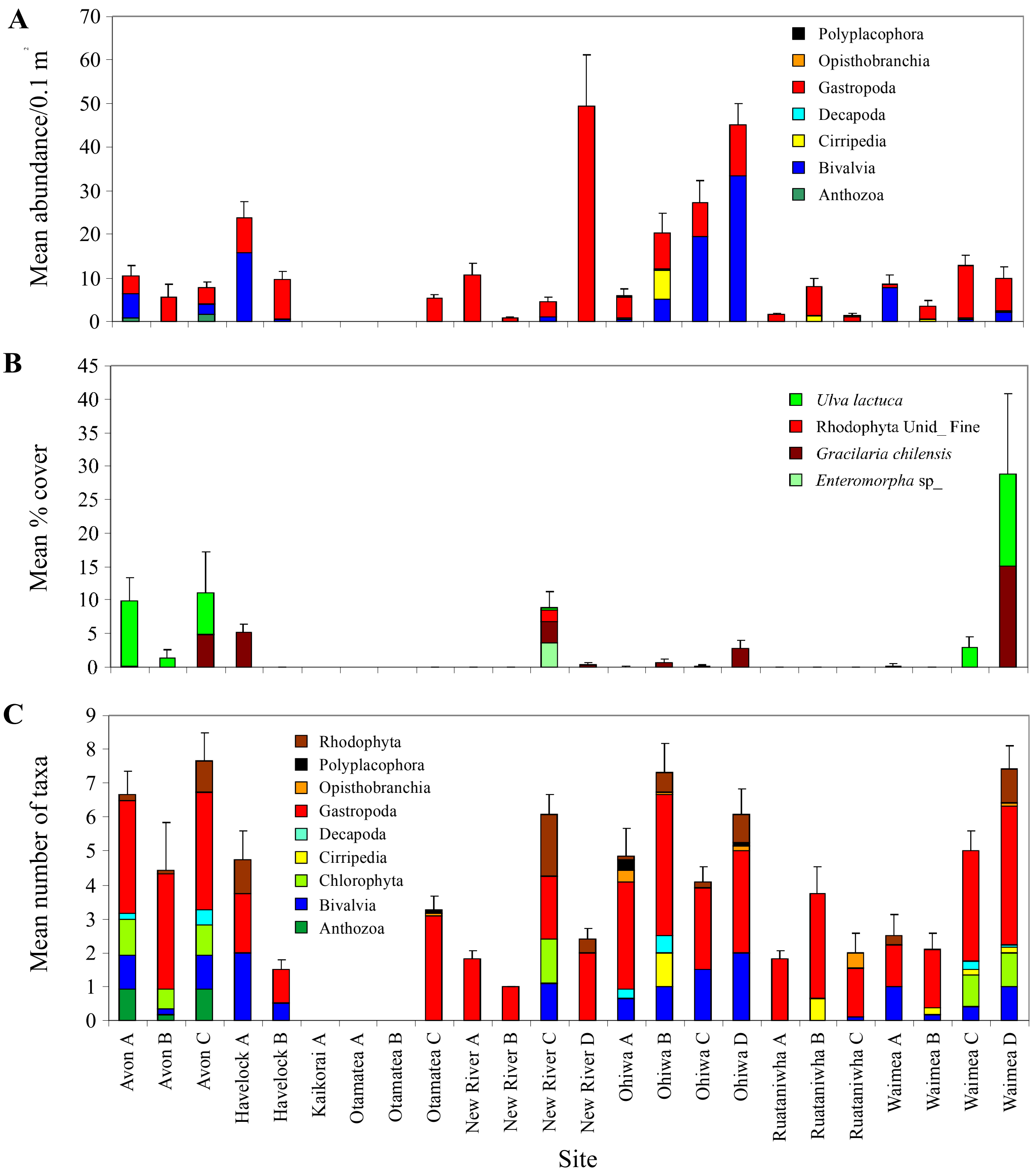


Figure 10: Mean (+ 95 % CI) of epifauna abundance (A) and species richness (C) and macroalgal cover (B) at sites within the eight reference estuaries.

6.4.3 Summary of statistical comparisons

Univariate and multivariate statistical analyses were applied to determine whether within-estuary and between-estuary differences in indicator levels existed. The complete statistical analyses are presented in Appendix C. A summary of the key findings is provided below.

The key findings of the univariate analyses were as follows:

- The nested ANOVA, using normalised data, identified both the estuary, and sites within an estuary as significant sources of variation for all of the indicators (Table A55, Appendix C).
- The multiple one-way ANOVAs for each variable in each estuary indicated significant variation among sites for most of the estuaries (Table A56, Appendix C).
- All environmental characteristics within sites from the Ohiwa and Otamatea estuaries were highly variable. A majority of these also varied significantly amongst sites at the remaining reference estuaries. Cadmium levels and epifauna abundance and richness were the only characteristics to consistently exhibit significant variation amongst sites at all estuaries.

The key findings of the multivariate analyses were as follows:

- The analysis of similarities (ANOSIM) indicated that significant variation existed both among sites (within-estuary) and among different estuaries (inter-estuary) for all biological, chemical and physical characteristics (Table A57, Appendix C).
- Spatial representation by MDS and PCA analyses indicated that infaunal communities at the Kaikorai estuary were distinct from all other sites and estuaries (Figure 11). All four New River estuary sites, and one of the Havelock sites (Site A) were placed in a separate group (Group A) from the remaining sites (Group B) (Figure 11).
- Infauna assemblages appeared to characterise estuaries well, exhibiting strong similarities among sites within estuaries. However, the bulk of the infauna assemblages were similar, with the exception of New River, Kaikorai and one Havelock site.
- Epifaunal communities did not characterise individual estuaries well, although some significant groupings did exist, suggesting that intra-estuary variations in epifauna were comparable to the observed inter-estuary variations (Figure 12).

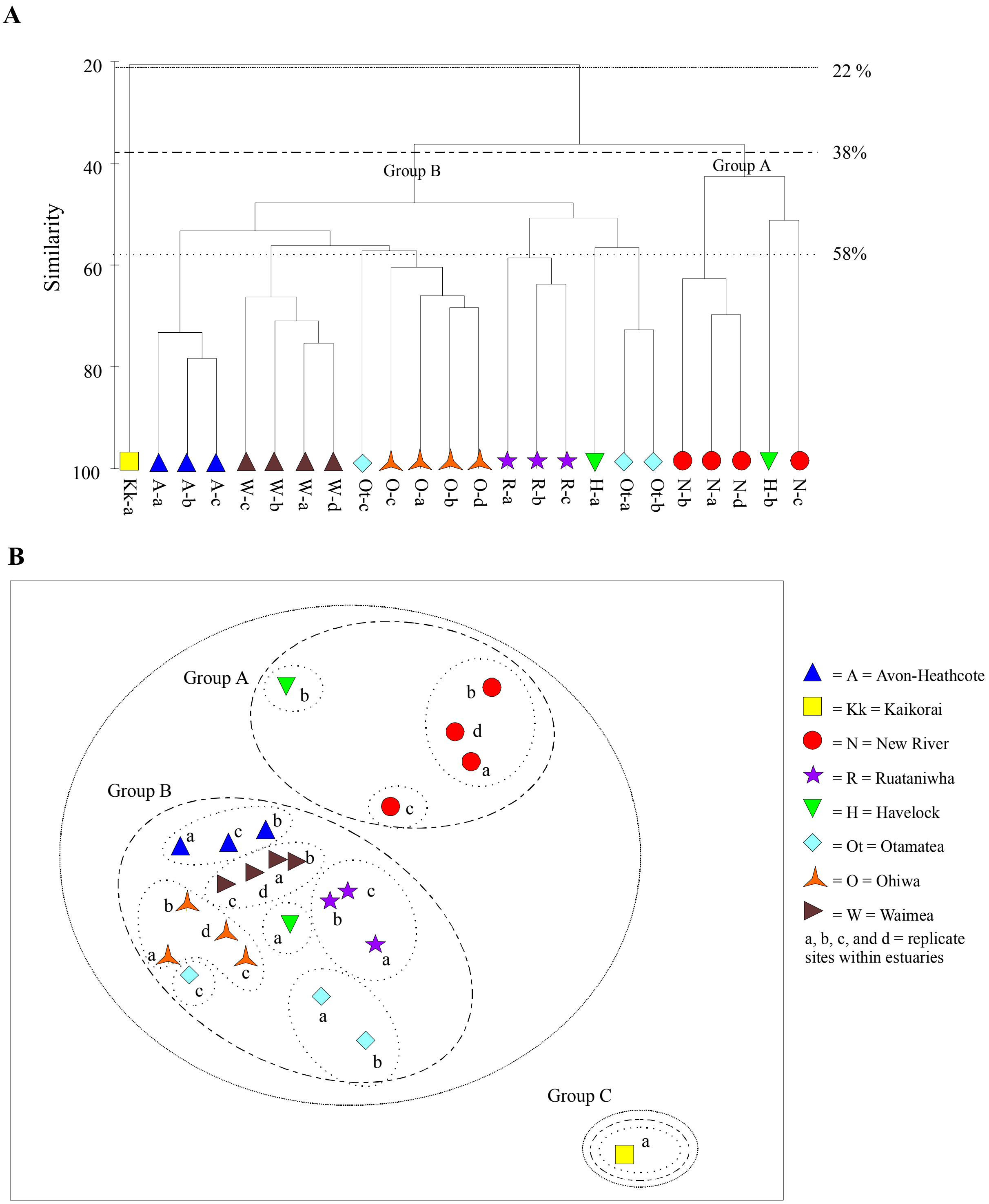


Figure 11: Bray-Curtis similarity dendrogram and corresponding two-dimensional MDS plot for average abundance of infauna species found in cores (grouped by site). Clusters superimposed on MDS plots are at similarity levels of 22 (solid), 36 (dashed) and 50 % (dotted). (2D Stress = 0.14).

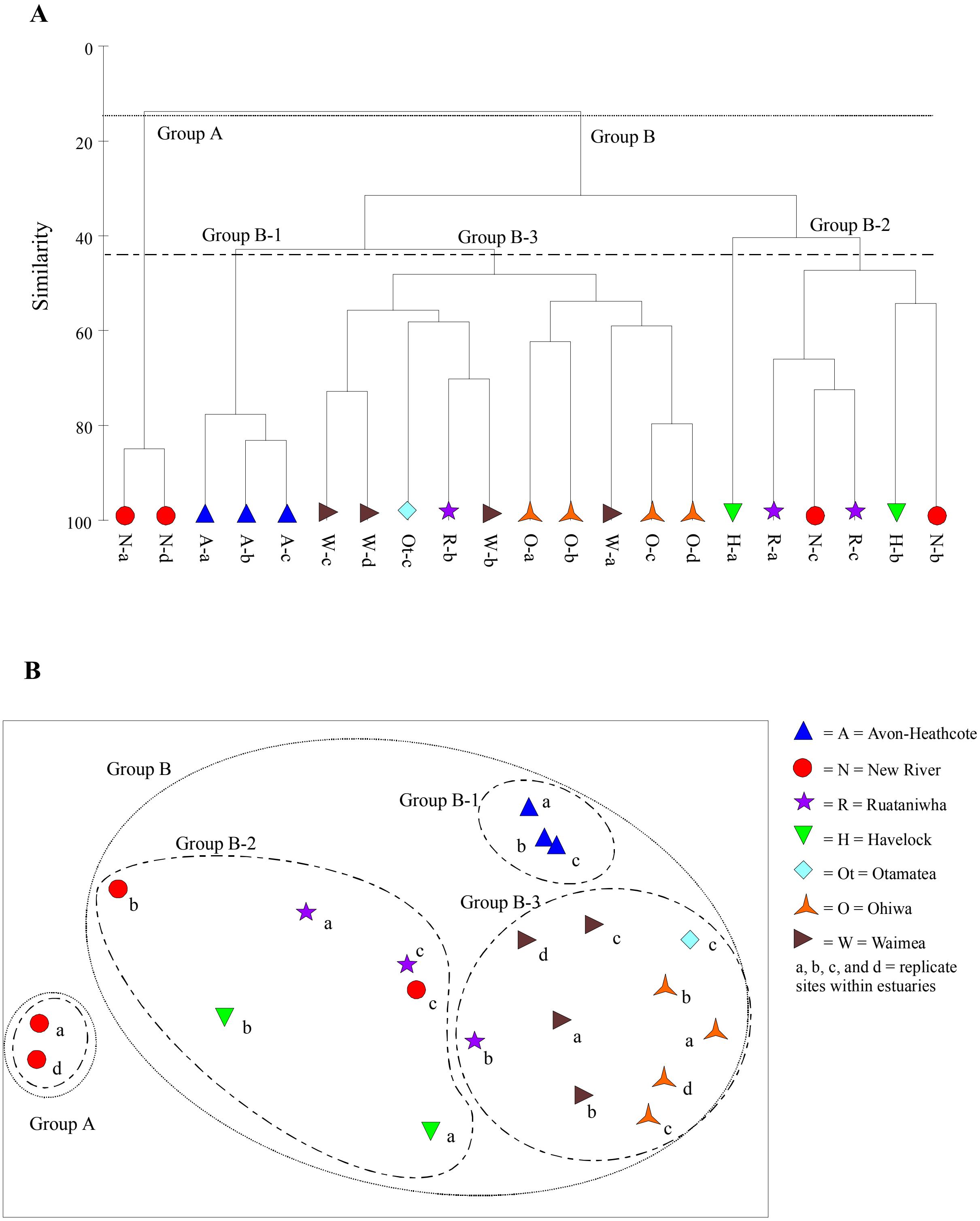


Figure 12: Bray-Curtis similarity dendrogram and corresponding two-dimensional MDS plot for average abundance of epifauna species found in quadrats (grouped by site). Clusters superimposed on MDS plots are at similarity levels of 20 (solid) and 50 % (dashed). (2D Stress = 0.13).

***Step 1 Results*: Summary and applications to the EMP**

- Sediment particle size was variable among estuaries.
- Sediment nutrient, organic matter and photosynthetic pigment concentrations were consistent with sediments of similar textures from other New Zealand estuaries, with a range from unenriched to moderately enriched.
- Sediment trace metal concentrations were within ANZECC trigger guidelines at all reference estuaries, except for the concentrations of nickel and chromium in the Waimea and Havelock estuaries, which were attributable to catchment geology.
- The relative concentrations of organic matter, nutrients and trace metals in the estuary sediments, following standardisation to 100% mud content, were useful in indicating the need for follow-up investigation of contaminant loads in the mud fraction and their origins.
- The fine-scale monitoring provided a record of the commonly found epibiota and infaunal communities of the intertidal habitat sampled at the reference estuaries.
- Species richness, abundance and composition observed in the present study were comparable to those reported for a number of other estuaries throughout New Zealand.
- Environmental parameter levels were significantly different between sites (within-estuary) and between estuaries (inter-estuary). The present sampling strategy (of 12 replicates per site) was sufficient to detect within-estuary differences (*i.e.* suitably accounted for variation on a scale of 10s of metres. Differences among mean values for different sites within the same estuary indicate that they can not be treated as true replicates without undertaking a preliminary (pilot) investigation.
- Such site heterogeneity within a single ‘habitat’ type highlights the difficulties associated with selecting the number and location of monitoring sites and the need for precise relocation for subsequent samplings.
- Infauna assemblages tended to characterise estuaries better than epifauna assemblages and generally, the majority of infaunal assemblages were similar across estuaries in the present study. This may facilitate the development and use of a national biotic health index to assess biological condition of New Zealand estuaries.

***Step 1 Results*: Summary and Applications to the EMP (continued…)**

- Although epibiota communities did not characterise individual estuaries well, some significant groupings did exist. Epibiota has historically been included in estuary monitoring programmes, and are worthy of inclusion in the EMP based on the valuable comparative data available.

The aims of this step of the fine-scale monitoring development were successfully met. The groups of variables chosen to assess estuary condition were appropriate, as the summary statistics describe the reference estuaries in terms of key environmental variables relevant to the intertidal muddy sand habitat. The results of this section provide a baseline dataset for future comparisons of fine-scale characteristics relating to estuary condition, which will be included in the EMP along with the dataset used in the statistical analyses for the following steps of the fine-scale monitoring protocol development.

The techniques of comparison (*i.e.* statistical analyses) used in this study were effective in determining whether the environmental characteristics differed among sites and estuaries.

Step 2: Examining the relationships between environmental characteristics

This section examined the relationships between all measured environmental variables in order to identify those which were closely related, and therefore may have suitable surrogates. The benefit of identifying suitable surrogates is that the number of variables to be measured may be reduced (thereby reducing sampling/analytical costs) without losing interpretive power on the health of the estuary. Particular emphasis was placed on examining the relationships between biological assemblages (infauna and epifauna) and the other environmental variables to determine which abiotic measurements may be the better indicators of biological health. The complete statistical analyses of this section are presented in Appendix C to remove the technical details from the main document. A summary of the key findings is provided below. At the end of this step, a summary box outlining the major findings and recommendations for the development of the EMP is also presented.

Caution:

It is important to note that correlations with TN, cadmium or lead may be partially confounded due to the arbitrarily assigned value (0.5 x the detection limit) to results below the detection limit.

Univariate analysis: correlation matrix

Inter-relationships between the environmental variables were explored using correlation matrices. The Pearson correlation matrix for un-normalised data identified a number of notable relationships and non-relationships, outlined below (refer Table A61, Appendix C for the complete correlation matrix and Table 62 for the correlation matrix using site-averaged data).

- Sediment mud content was strongly ($\rho > 0.8$) positively correlated with organic content (AFDW) and total nitrogen (TN), and to a lesser degree, total phosphorus (TP), copper (Cu), lead (Pb) and zinc (Zn). Relationships with chlorophyll *a*, phaeophytin and the other trace metals were weak.
- Chlorophyll *a* and phaeophytin were not strongly correlated with any of the other variables.
- AFDW was strongly positively correlated with the TN and TP, and with some trace metals (Cu, Pb and Zn).

- Both TN and TP were positively correlated with the trace metals Cu, Pb and Zn, while TP was also correlated with chromium and nickel.
- Apart from cadmium (Cd), each of the trace metals typically correlated well with other trace metals and all significant correlations were positive. In particular, Cu, Cr, Pb, Ni, and Zn were well correlated, although the relationships between Pb and Cr, Pb and Ni and Zn and Ni were not strong ($\rho < 0.5$). Ni was particularly strongly correlated with Cr ($\rho = 0.94$), as was Pb to Zn ($\rho = 0.83$).
- Infauna species richness and abundance did not correlate well to any other characteristics (physical, biological or chemical).
- Epifauna abundance correlated only with epifauna species richness, which was also correlated to infauna species richness.

Standardising chlorophyll *a*, phaeophytin, nutrient and chemical data to 100 % mud content resulted in several important changes in the relationships between parameters (Table A61 and Table A62, Appendix C):

- Relationships of nutrients and trace metals with AFDW were intensified, with some correlations exceeding $\rho = 0.8$.
- Stronger positive correlations were identified between all of the trace metals and nutrients, with many correlating very strongly ($\rho > 0.9$). An exception to this trend was Pb, for which normalisation resulted in weaker relationships with the other metals.
- No strong correlations existed between any of the physical and chemical characteristics and the epifauna and infauna analyses. However, the general trend was for weak negative correlations between epifauna abundance and species richness and all nutrient and trace metal characteristics.

Multivariate analysis: Environmental characteristics (PCA)

PCA plots generated from normalised (to mud content) and non-normalised environmental data are presented in Figure 13A and B. For un-normalised data, PC1 (x-axis) and PC2 (y-axis) together account for 75.5 % of the total sample variability (Table A63, Appendix C). On the PC1 axis, the influence of most of the (transformed) characteristics was roughly equally weighted, with the strongest coefficients being for mud content, AFDW, TN, TP, Cu, Pb and Zn. The characteristics that barely featured on PC1 (Chl a, Phaeo, Cd and Ni) were the dominant influencing factors on the PC2 axis (Table A64, Appendix C, refer to Technical Box A.8 for definitions).

The distribution of sites resulting from the PCA of the non-normalised environmental data formed four approximate groupings:

→ Group 1: the single site at the Kaikorai Estuary.
→ Group 2: the three sites from the Otamatea arm of the Kaipara Estuary.
→ Group 3: all sites from the Avon-Heathcote, Ohiwa, Ruataniwha and New River estuaries.
→ Group 4: all sites from the Havelock and Waimea Estuary.

The amount of variance accounted for by a 2-dimensional ordination (PC1 and PC2) for the 100% mud-normalised values was similar to that for the un-normalised data, at 75.3 % (Table A63, Appendix C). Normalisation of the environmental data to 100% mud content altered the PCA ordinations (Figure 13B):

→ the Kaikorai site was no longer considered distinct from the Otamatea sites.
→ the Ohiwa sites shifted from being more closely related to the New River and Avon-Heathcote estuaries, to being more aligned with the Havelock and Waimea estuary sites.

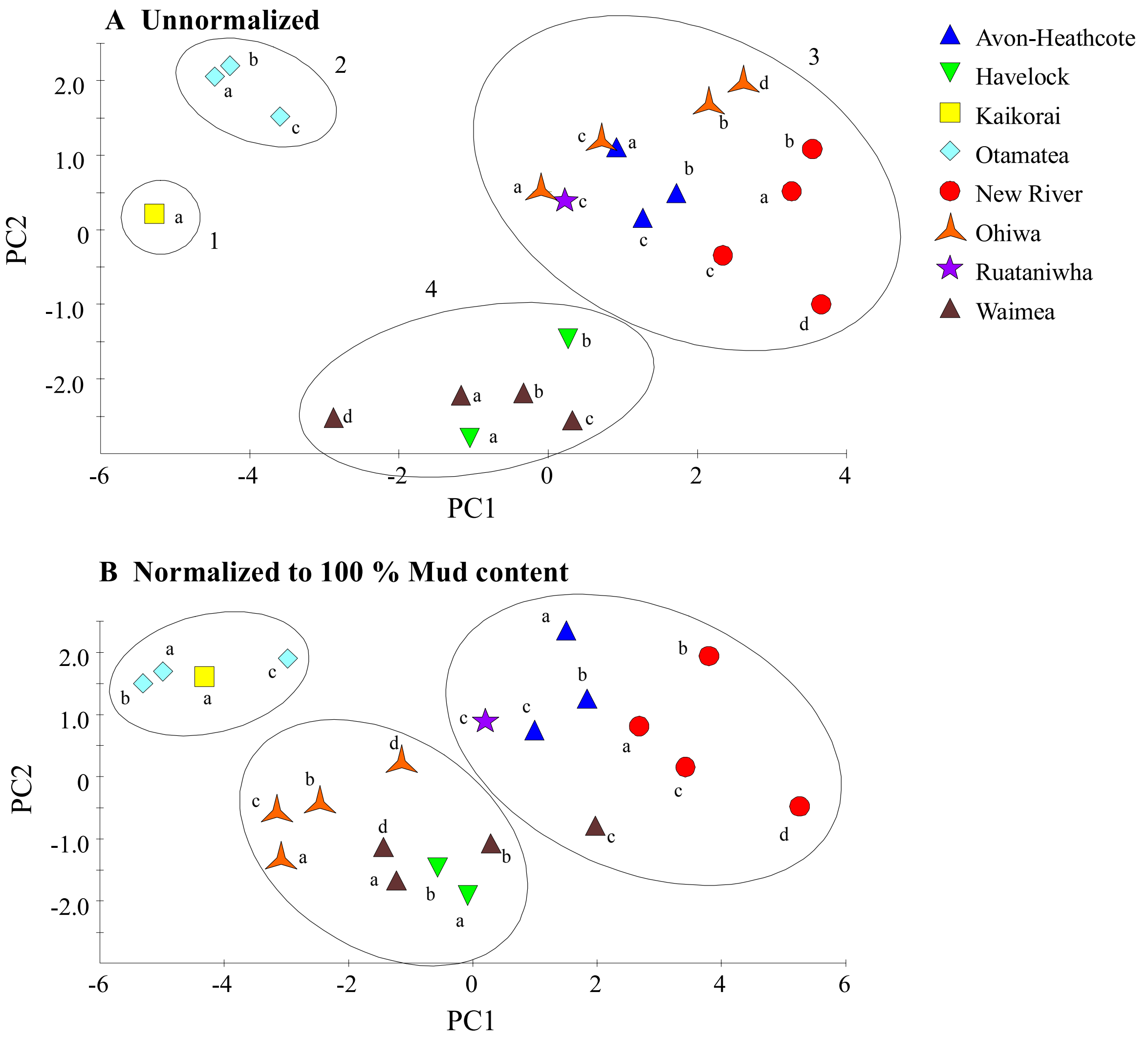


Figure 13: Two-dimensional PCA ordination of (a) transformed un-normalised, and (b) normalised physical and chemical data.

Multivariate analysis: BIOENV Procedure

Based on the strong correlations observed between some variables; AFDW, TN, Cr and Zn were omitted from the first BIOENV procedure comparing biotic assemblages to non-normalised environmental data. AFDW, TN, TP, Cr and Zn were omitted from the normalised data set. The omitted variables and the corresponding potential surrogates and correlation coefficients are listed below in the Summary Box. In general, grouping of the environmental variables did not explain the ordination of the biotic variables well, with no correlations (ρ) > 0.52. However, mud content in combination with Pb concentration did explain approximately 50 % of the pattern. The second best combinations included TP and the third also included Chlorophyll *a*.

The infauna assemblages observed in this study are most closely linked to sediment texture, nutrients and one of a few contaminant concentrations (Cr, Cu or Pb). Normalised data (indicative of the concentrations in the mud fraction) explained a similar amount of the pattern, but included copper in the explanatory combination and omitted TP. Epifauna data could not be well predicted from any combinations of the un-normalised environmental parameters (correlations < 0.23).

Normalised data accounted for up to 50% of the observed pattern. Trace metal concentrations (Cu and Cd), and to a lesser degree Chlorophyll *a*, were the most relevant characteristics. Concentrations of contaminants in the mud fraction of the sediment seemed to be more relevant to the type of epifauna assemblages that were present than concentrations in the whole sediments.

Step 2 Results: Summary and Applications to the EMP

- Many strong correlations ($\rho > 0.8$) existed between nutrients, trace metals, organic and mud content. However, no strong correlations existed between any of the physical and chemical variables with the biological assemblages.
- Sediment type (mud content) played a dominant explanatory role in determining the environmental patterns in the present study. Therefore, sediment particle size analysis is considered an essential part of an EMP.
- The strong correlations ($\rho > 0.8$) determined in Step 2 identified several variables that could serve as surrogates (or proxies) for others. These are outlined in the table below, based on the univariate correlation matrix (refer to Table A61-B, Appendix C) with data standardised to mud content:

Variable 1	Variable 2	Correlation (ρ)	Surrogate	Rationale
Zinc	TP	0.951	No	Loss of interp.
Nickel	Chromium	0.943	?	?
Copper	TP	0.918	Cu	Prelim. screening
Copper	Zinc	0.889	Cu	Prelim. screening
Copper	Chromium	0.865	Cu	Prelim. screening
AFDW	TN	0.847	AFDW	Prelim. screening
Copper	Cadmium	0.836	Cu	Prelim. screening
Zinc	Chromium	0.824	?	?
Chromium	TP	0.816	No	Loss of interp.
Copper	TN	0.813	No	Loss of interp.

- The key issue regarding the use of surrogate analyses is that the interpretive value of the data will always be reduced. This must be weighed against the saving in cost.
- The data indicates that copper may be used as a surrogate for other metals. This may be appropriate as long as it is recognised that observations of high copper concentrations may result in the need for follow-up analyses of other metals that (by association) approach levels of environmental concern.
- AFDW could be used as a surrogate for TN (in some monitoring situations), as it is less expensive to analyse. However, we would suggest, for a general estuarine monitoring programme such as the one under development here, that the loss of interpretive value would outweigh the cost saving.

Step 3: Determining the optimum sample size

Determining the optimum sample size for the environmental variables ensures that an adequate number of samples are collected to provide an accurate assessment of estuarine health, while maintaining cost-efficiency. Determining the optimum sample size in the present study involved:

a) Comparing the variability (using the coefficient of variation (CV)) of each variable amongst sites and amongst the eight REs. Highly variable measures will require larger sample sizes to reliably estimate the mean value.

b) Comparing the relative variation (CV) within estuaries for each variable to identify sites (within an estuary) that could be pooled. Pooling the data from more than one site served to increase the available number of samples from which the estimated mean CV was generated, and thereby also increased the maximum *n* that could be assessed (see Appendix C4 for details).

c) Conducting power analyses using the estimated mean CVs (from random combinations of pooled sites) for each variable in each estuary to determine the optimum sample sizes (by cost benefit point (CBP) analysis).

The complete statistical analyses of this section are presented in Appendix C. A summary of the key findings, followed by a Summary Box is provided below.

Analysis of precision and pooling of sites

Differences in CVs among estuaries were significant (using one-way ANOVAs) for 8 out of the 16 variables (Table A67, Appendix C). The characteristics that demonstrated significantly different inter-estuary variation in CV included: nutrients, TN and TP, and the trace metals, Cd, Cr, Cu, Pb, Ni and Zn. The differences in CVs among estuaries was not significant for mud content, phaeophytin, organic content (AFDW), or infauna and epifauna species richness and abundance. With half of the variables exhibiting differences attributable to estuary, it was considered necessary to address sample size on a estuary by estuary basis.

Tests for homogeneity of CVs identified significant differences among sites (Table A68, Appendix C), meaning that pooling of sites was also not valid in some instances. As with inter-estuary variations in precision, significant variations among sites were also more common with chemical variables. Significant variation in precision was more common for chemical variables than for physical and biological variables. The Avon-Heathcote, Havelock, New River, Ruataniwha and the Waimea data sets contained variables with similar CVs that allowed pooling of the sampling sites within those estuaries. In some cases however, up to two sites needed to be removed to achieve the desired similarity between CVs.

Power analyses to determine optimum sample sizes

The estimated mean CVs (derived from computer-generated plots, refer Appendix C for methods used), approximating the 'actual variation' for that characteristic in a particular estuary, ranged from 0.10 to 0.75, with the majority lying between 0.20 and 0.40. Lead typically exhibited the highest variation (lowest precision), possibly due to the inclusion of samples below the method detection limit (as these were assigned a level of 0.5 x the detection limit). Biological variables also exhibited high variation, with CVs ranging between 0.25 and 0.50. Chlorophyll *a,* TN and Cd had mean CVs of approximately 0.20 to 0.40, while the organic content (AFDW), TP, and the trace metals Cr, Cu, Ni and Zn showed the highest precision, typically between 0.15 and 0.30. The point (*n*) at which substantial gains in CV were no longer made for a further 5*n* increase in sample size (termed cost benefit point, CBP) was estimated for each variable and these are summarised in Figure A32, Appendix C.

The mean number of samples required to reach the CBP was 8.0 (SD = 2.0), averaged across all variables and all estuaries. For individual variables within each estuary, the maximum sample size was 14 (for lead in the Ohiwa estuary), whereas the minimum was 4 (recorded on one occasion for each of: Zn, Ni, Cu and mud content). Based on this analysis, Cu and Ni consistently required 7 samples to reach the optimal level of precision, represented by the CBP. The CBP for Pb was, on average, notably higher than for the other physical and chemical characteristics, at approximately 11 (due, in part, to some sample concentrations of Pb being lower than the analytical detection limit).

Chlorophyll *a* concentrations and epifauna richness required 8-12, and 7-12 samples, respectively. Infaunal abundance required 8-9 samples, while infauna diversity required 7-8 samples to reach the CBP. Figure 55 shows the relationships between sample size (*n*) and the size of the measurable change for four variables with differing CVs from the Waimea Estuary, and provides the CVs used in the two power analysis models.

Based on the G*Power model (refer Appendix C for details), estimates of environmental parameter levels with CVs of greater than 0.6 were only capable of detecting changes greater than a 2-fold change in the given variable. For a sample size of 10, the measurable change able to be detected for a variable with a low CV (0.08) was approximately 15 %, and up to a 98 % with a variable with a higher CV (0.58) (Power = 0.8, $\alpha = 0.05$) (Figure A33a, Appendix C). The iterative process described in Zar (1999) was less conservative, calculating CV values from 0.08 to 0.58 translating to an expected measurable change of 10 to 57 %, despite using a slightly higher power of 0.9 ($\alpha = 0.05$) (Figure A33b, Appendix C).

Figures A33c and d, Appendix C illustrate the sample size required at varying levels of precision (CV), for a range of detectable changes. For example, with a CV of 0.3, the G*Power model calculated that approximately five samples were required to detect a 100 % change in the mean, and as many as 63 samples to detect a 20 % change. However, the less conservative estimate from the t-test model (Zar, 1999) suggested < 3 samples were necessary to detect a 100 % change and 25 samples to detect a 20 % change.

Estimates of mean CV (± 95% CIs) are presented in Table A69, Appendix C. Estimates of the minimum detectable differences for each parameter in each of the estuaries are presented in Table A70, Appendix C. The relative minimum levels of detectable change associated with each parameter are compared in Figure 54C. The size of the detectable difference between means using the CBP sample number according to Zar (1999) with $\alpha = 0.05$ and $\beta = 0.1$ was between 25 and 50 % for most parameters. The less variable parameters such as AFDW, TP, Cu, Ni and Zn had slightly lower ranges of estimated minimum detectable differences of 25 to 35 %. Characteristics that were inherently more variable, such as Chlorophyll *a*, TN, Cd, Pb and abundance of infauna, would require a change in the order of 30-65 % before a difference could be confidently detected.

In some instances (*e.g.* Ni and Zn in the Avon-Heathcote) this method resulted in selecting a low CBP (4) with a relatively high mean CV (> 0.3), which translated into a large relative level of detectable change. This occurred when the plots showed little discernable reduction in CV_{95} with increasing sample number, suggesting additional sampling was not warranted, but at the same time demonstrated considerable variability about the mean (CV_{mean}). In such cases, it is suggested that sample size should be increased slightly in order to increase the power of detection, despite the fact the precision is unlikely to be improved.

Step 3 Results: Summary and Applications to the EMP

- The variability of the environmental characteristics measured in this study (indicated by the estimated actual CVs) was high, typically around 20-40%, and ranged from 10-75%. Variability also differed among estuaries. This means that the estimated optimum sampling size for each variable will fluctuate, depending on its inherent variability and the particular estuary.
- In most instances, taking more than eight samples did not result in any substantial increase in precision. The minimum cost benefit point (CBP) for sample number ranged from 6 to 12.
- The high variability associated with many of the measured variables dictates that impractically large numbers of samples would be needed to achieve a detectable change of less than 30%.
- A recommended minimum sample size of 10 would encompass most of the minimum CBPs determined in this study for the measured variables. However, it is recommended that CBP analyses for a particular estuary are undertaken, once monitoring data were available, in order to determine the estuary-specific optimum sample numbers for measured variables.
- Using the recommended minimum sample size of 10, the detectable change in parameter levels would be between 25 and 65%, depending on the variability of the parameters measured. If a particular variable is considered a higher monitoring priority than others, a greater proportion of sampling effort could be allocated to it (*i.e.* increase the sample number greater than 12, but only for that variable), thereby increasing the ability to detect change.
- In view of the summary statistics described above, it is possible to develop a fine-scale monitoring design that: (1) will enable assessment of change (at an environmentally meaningful level), and (2) is both cost-efficient and scientifically defensible in terms of sampling effort required.

Fine-Scale Environmental Monitoring Summary

The fine-scale sampling approach adopted in this study was successful in obtaining a baseline data set of benthic intertidal variables from the eight reference estuaries. Statistical analyses were then applied to investigate the variability of the data, both among estuaries and at sites within estuaries. Those variables that were closely correlated were also identified, enabling some to be considered as surrogates for others in a monitoring programme. The optimum number of samples was determined for each variable to accommodate the established spatial variability, as well as the expected level of change able to be detected with different levels of variability (~sampling precision).

These findings have refined the fine-scale environmental monitoring approach, particularly with regard to the cost-efficiency, ease of use, and sampling precision. Although we apply this information to the development of a standardised National Estuarine Monitoring Protocol, it could also provide guidance for the design of more specific consent monitoring programmes.

It is important to note that the fine-scale approach adopted in this study focused on middle-of-the-range estuaries, and did not include sampling at highly impacted estuaries. Therefore, the baseline dataset generated from this research (against which future studies will be compared) does not include the full spectrum of estuary conditions. Further research that expands the range of estuaries included in the study may adjust the benchmark relationships to estuarine condition.

The aim of this study was to investigate the spatial distribution of a number of estuarine variables. Although it is well established that temporal variation is often a feature of dynamic coastal environments such as estuaries, it is important to note that the temporal variation has not been described in this study. Further research is needed to describe short-term and seasonal variation of the characteristics measured.

The statistical analyses suggest that it is feasible to design a fine-scale environmental monitoring programme that: (1) will enable assessment of change (at an environmentally meaningful level), and (2) is both cost-efficient and scientifically defensible in terms of sampling effort required.

7. REFERENCES

ANZECC & ARMCANZ. 2000. *Australian and New Zealand guidelines for fresh and marine water quality 2000 Volume 1.* National Water Quality Management Strategy Paper No 4. Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

AOAC 1995. Official Methods of Analysis of the Association of Official Agricultural Chemists. 16th ed. Vol. II. Method 950.46 modified.

APHA 1999. Standard Methods for the Examination of Water and Wastewater 20th ed. American Public Health Association, Washington, D.C.

ASTM 1994. Method D3974- Dig A. Standard Practices for Extraction of Trace Elements from Sediments, American Society for Testing and Materials 3p.

Atkinson, I.A.E. 1985. Derivation of vegetation mapping units for an ecological survey of Tongariro National Park North Island, New Zealand. New *Zealand Journal of Botany* 23: 361-378.

Ballance, P.F.; McCarthy, J.A. 1975. Geology of Okahukura Peninsula, Kaipara Harbour, New Zealand. *New Zealand Journal of Geology and Geophysics 18*(5):721-743.

Beca Carter Hollings & Ferner. 1992. Coastal/Sounds Plan Issues and Options: Picton/Havelock. Prepared for Marlborough District Council. Beca Carter Hollings & Ferner Ltd., Wellington. 48p plus appendices.

Beca Steven. 1993. Fairfield (Maxwells) Landfill–Assessment of Environmental Effects. Prepared for Fulton Hogan Ltd.

Biosearches. 1993. Mapua Site – Sediment and Shellfish Quality. Appraisal Report for Mintech (NZ) Ltd.

Bilyard, G.R. 1987. The value of benthic infauna in marine pollution monitoring studies. *Marine Pollution Bulletin 18:* 581-585.

Boffa Miskell. 1992. Coastal/Sounds Plan: Recreation and Tourism Issues. Prepared for Marlborough District Council. Boffa Miskell Partners Ltd. 58p.

Bradstock, M. 1985. *Between the Tides: New Zealand Shore and Estuary Life*. Reed Methuen, Auckland. 158p.

Bradstock, M. 1983. Fish utilisation of estuary habitats. *In:* Gillespie P. (Ed) Nutrient Processing and Biomass Production in New Zealand Estuaries. *Water and Soil Miscellaneous Publication 60.* Wellington, Ministry of Works and Development. 28.

Brockbank, A.S. 1983. Kaipara Harbour: sedimentation and shoreline environments. Unpublished MA thesis, University of Auckland. 88p.

Bros, W.E.; Cowell, B.C. 1987. A technique for optimising sample size (replication). *Journal of Experimental Marine Biology and Ecology 114*: 63-71.

Bull, M. 1981. Pacific oysters now in Tasman Bay. *Catch 8:* 12.

Burridge, B.E. 1976. Soil conservation survey of the northern Kaipara Harbour and Mangawhai estuary catchments. Prepared for Northland Catchment Commission. 27p.

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

Chittenden, E.T.; Hodgson, L; Dodson, K.J. 1966. Soils and Agriculture of Waimea County New Zealand. *Soil Bureau Bulletin, 30.* New Zealand Department of Scientific and Industrial Research.

Christchurch City Council. 1992. The Estuary – where our rivers meet the sea. Owen, S-J. *Ed.* Christchurch City Council, Christchurch.

Church, T.M. (*Ed*.). 1975. *Marine Chemistry in the Coastal Environment.* ACS Symposium Series 18. American Chemical Society, Washington D.C. 710 p.

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology 18*:117-143.

Clarke, K.R.; Warwick, R.M. 1994. *Changes in marine communities: an approach to statistical analysis and interpretation.* Natural Environment Research Council, UK. 144p.

Cooper, J.A.G.; Ramm, A.E.L.; Harrison, T.D. 1994. The estuarine health index: a new approach to scientific information transfer. *Ocean and Coastal Management 25* (2): 103-141.

Craft, C.B.; Seneca, E.D.; Broome, S.W. 1991. Loss on ignition and Kjeldahl digestion for estimating organic carbon and total nitrogen in estuarine marsh soils: calibration with dry combustion. Estuaries 14: 175-179.

Daniel, L.J. 1984. Mangroves and Saltmarshes of Ohiwa Harbour. Department of Lands and Survey, Gisborne. 71p.

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

Davidson, R.J.; Brown, D.A. 2000. A report on the ecology of Havelock Estuary, Pelorus Sound, Marlborough Sounds. Prepared for Marlborough District Council. Survey and Monitoring Report No. 342. Davidson Environmental Ltd., Nelson. 40p.

Davidson, R.J.; Moffat, C.R. 1990. A report on the ecology of Waimea Inlet, Nelson. Department of Conservation, Nelson/Marlborough Conservancy, Occasional Publication No. 1. 133p plus appendices.

Davidson, R..; Preece, J.; Rich, L.; Brown, D.; Stark, K.; Cash, W.; Waghorn, E.; Rennison, G. 1990. *Coastal Resource Inventory First Order Survey Nelson/Marlborough Conservancy.* Davidson, R.; Preece, J. *Eds.* Department of Conservation, Wellington. 261p plus maps.

Estcourt, I.N. 1962. An ecological survey of the burrowing polychaetes of the Avon-Heathcote Estuary. Unpublished M.Sc thesis (zoology), University of Canterbury.

Forrest, B. 1995. Ecological effects of Havelock sewage discharge (second monitoring survey, February 1995). Report prepared for Marlborough District Council. *Cawthron Report No. 296.* 10p.

Gibbs, H.S. 1968. *Volcanic-ash soils in New Zealand.* Information Series No. 65, New Zealand Department of Scientific and Industrial Research, Wellington. 39p.

Gibson, G. R.; Bowman, M. L.; Gerritsen, J.; Snyder, B. D. 2000. Estuarine and Coastal Marine Waters: Bioassessment and Biocriteria Technical Guidance. EPA 822-B-00-024. U. S. Environmental Protection Agency, Washington, DC.

Gillespie, P. A. 1983. Tidal nutrient exchange in Delaware Inlet. *In* Gillespie P. (Ed) Nutrient Processing and Biomass Production in New Zealand Estuaries. *Water and Soil Miscellaneous Publication 60.* Wellington, Ministry of Works and Development. 4-6.

Gillespie, P.A.; Asher, R.A. 1996. Land disposal of sewage sludge on Bells Island: Assessment of estuarine impacts, December 1994-January 1996. Prepared for the Nelson Regional Sewerage Authority. *Cawthron Report No. 359*. 41p.

Gillespie, P.; Asher, R. 1999. Impacts of the Bell Island Regional Sewerage Scheme on Waimea Inlet: Bacteriological Water Quality. *Cawthron Report No. 460*. Prepared for Nelson Regional Sewerage Authority. 11p.

Gillespie, P.; Asher, R. 2001. Impact of the Nelson (Bell Island) regional sewerage discharge on Waimea Inlet: April 2001 receiving water survey. *Cawthron Report No. 695*. Prepared for Nelson Regional Sewerage Business Unit and Tasman District Council. 13p.

Gillespie, P.; Asher, R.D.; Doyle, M. 2001a. Impact of the Nelson (Bell Island) regional sewerage discharge on Waimea Inlet Part II: Nutrient enrichment potential of receiving waters. *Cawthron Report No. 680*. Prepared for Nelson Regional Sewerage Business Unit and Tasman District Council. 30p.

Gillespie, P. A.; Asher, R.A.; Franko, G.D.; Stark, J.D. 1990. Environmental impact of the use of amitrole/dalapon herbicide sprays to control *Spartina* in Waimea Inlet, Nelson: Part II. Environmental monitoring programme. Cawthron Institute Report to the Nelson-Marlborough Regional Council. 42p plus plates and appendices.

Gillespie, P.A.; Asher, R.A.; Frisk, S. 2001b. Impacts of the Nelson (Bell Island) regional sewerage discharge on Waimea Inlet: Part 1. Benthic enrichment (April-May 2001 survey). Prepared for the Nelson Regional Sewerage Business Unit. *Cawthron Report No. 677.* 62p.

Gillespie, P.A.; MacKenzie, A.L. 1981. Autotrophic and heterotrophic processes on an intertidal mud-sand flat, Delaware Inlet, Nelson, New Zealand. *Bulletin of. Marine Science 31*: 648-657.

Gillespie P.A.; MacKenzie, A.L. 1990. Microbial activity in natural and organically enriched intertidal sediments near Nelson, New Zealand. *New Zealand Journal of Marine and Freshwater Research 24*: 471-480.

Gillespie, P.A.; Stark, J.D.; Asher, R.D. 1992. Investigation of the potential impacts of the Bells Island regional sewerage outfall on Waimea Inlet. Prepared for the Nelson City Council. *Cawthron Report No. 191*. 46p plus plates and appendices.

Gillespie, P.A.; Stark, J.D.; Asher, R.A.; Fenemor, A.D. 1995. Moutere Inlet ecosystem investigation. Prepared for the Tasman District Council. *Cawthron Report No. 273*. 37p plus plates and appendices.

Glasby, G.P.; Moss, R.L.; Stoffers, P. 1990. Heavy metal pollution in Porirua Harbour, New Zealand. *New Zealand Journal of Marine and Freshwater Research 24*:233-237.

Grant, A.; Middleton, R. 1998. Contaminants in sediments: using robust regression for grainsize normalisation. *Estuaries 21* (2): 197-203.

Grindley, G.W.; Watters, W.A. 1965. *Geology of Nelson*. New Zealand Geological Survey, New Zealand Department of Scientific and Industrial Research, Wellington. 21p.

Hare, J.; Hayes, S.; King, S. 1990 *Coastal Resource Inventory First Order Survey Southland Conservancy*. Department of Conservation, Wellington. 276p plus maps.

Harris, G.P. 2001. Biogeochemistry of nitrogen and phosphorus in Australian catchments, rivers and estuaries: effects of land use and flow regulation and comparisons with global patterns. *Marine and Freshwater Research 52*: 139-149.

Healy, J.; Scofield, J.C.; Thompson, B.N. 1964. Sheet 5 Rotorua (1st Ed.). Geological Map of New Zealand 1: 250 000. New Zealand Department of Scientific and Industrial Research, Wellington.

Heath, R. A. 1976. Broad classification of New Zealand inlets with emphasis on residence times. *New Zealand Journal of Marine and Freshwater Research 10*: 469-483.

Hewitt, J.E.; McBride, G.B.; Pridmore, R.D.; Thrush, S.F. 1993. Patchy distributions: optimizing sample size. *Environmental Monitoring and Assessment 27*: 95-105.

Hewitt, J.E.; Thrush, S.F.; Pridmore, R.D.; Cummings, V.J. 1994. Ecological monitoring programme for Manukau Harbour: analysis and interpretation of data collected October 1987 to February 1993. A.R.C. Environment Technical Publication No. 36, Auckland Regional Council, Auckland. 81p.

Hume, T.M.; Herdendorf, C.E. 1988. A geomorphic classification of estuaries and its application to coastal resource management- a New Zealand example. *Ocean and Shoreline Management 11*: 249-274.

James, G. 1999. Fish resources of the Avon-Heathcote Estuary and inshore Pegasus Bay: current knowledge and assessment of effects of treated effluent discharges. NIWA Client Report CHC99/36. 19p.

Jezus Belzunce, M.; Solaun, O.; Franco, J.; Valencia, V.; Borja, A. 2001. Accumulation of organic matter, heavy metals and organic compounds in surface sediments along the Nervion Estuary (northern Spain). *Marine Pollution Bulletin 42*:1407-1411.

Kennish, M. J. 1997. *Pollution Impacts on Marine Biotic Communities.* CRC Press, New York. 310p.

Knox, G. A. 1986. *Estuarine Ecosystems: A Systems Approach. Vol. 1*. CRC Press, Boca Raton, Florida. 271 p.

Knox, G.A.; Kilner, A.R. 1973. The ecology of the Avon-Heathcote Estuary. Unpublished report for the Christchurch Drainage Board. Estuarine Research Unit, University of Canterbury, Chirstchurch. 327p plus appendices.

Koppelman, M. H.; Dillard, J. G. 1975. An ESCA study of sorbed metal ions on clay minerals. pp. 186-201. *In: Marine Chemistry in the Coastal Environment.* Church, T. M. *Ed*. American Chemical Society, ACS Symposium Series 18, Washington D.C.

Kruskal, J.B.; Wish, M. 1978. *Multidimensional Scaling*. Sage Publications, California.

Land Cover Database. 2001. LCDB1 (on compact disc). Terralink International, Wellington. Refer www.terralinkltd.com/tech/data/lcdb/lcdb.htm for product information.

Lee, W. G.; Partridge T. R. 1983. Rates of spread of *Spartina anglica* and sediment accretion in the New River Estuary, Invercargill, New Zealand. *New Zealand Journal of Botany 21*: 231-236.

Lincoln Environmental, 1994. Avon-Heathcote Estuary Dye Study. Lincoln Environmental, a division of Lincoln Ventures Ltd. 2481/2. March 1994.

Long, E.R.; Morgan, L.G. 1991. The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. *NOAA Technical Memorandum NOS OMA 52*

Lorenzen, C.J. 1967. Determination of chlorophyll and phaeopigments: spectrophotometric equations. *Limnology and Oceanography 12*: 343-346.

Marsden, I. 2000. Benthic macro-invertebrates of the Avon-Heathcote Estuary, December 1997. Part 2- Total fauna including polychaetes. Unpublished report prepared for the Canterbury Regional Council.

McBride, G.B. 1993. Time series modelling of effect of Maungaturoto dairy factory waste discharge on dissolved oxygen concentrations in the Otamatea Estuary. Prepared for Northland Dairy Company. NIWA Consultancy Report No. NDC001. 51p.

McLay *et al*. 1976. An inventory of the status and origin of New Zealand estuarine systems. *Proceedings of the New Zealand Ecological Society 23*: 8-26.

McMillan, S.G. 1994. Mineralogy and geochemistry of the Abbotsford Formation. Unpublished BSc (Hons) thesis, Otago University, Dunedin.

Ministry for the Environment. 2001. Environmental performance indicators: Confirmed indicators for the marine environment. ME No. 398, Ministry for the Environment, Wellington. 65p.

Morrison, M.; Francis, M. 2000. Are estuaries important for fish? *Seafood New Zealand 8*: 34-35.

Morton, J.E.; Miller, M.C. 1968. *The New Zealand Sea Shore.* Collins, London.653p.

NZDSIR, 1968. Soils of New Zealand Part 1. *Soil Bureau Bulletin, 26* (1). New Zealand Department of Scientific and Industrial Research, Wellington. 142p.

New Zealand Geological Survey. 1964. *Geology of Northland.* Department of Scientific and Industrial Research, Wellington. 24p.

Nottage, R.A.C. 2001. Water quality of the Aorere River and its tributaries, Golden Bay. Unpublished MSc thesis (geography), University of Otago. 133p.

Owen, K. L.; M. G. Sell. 1985. The birds of Waimea Inlet. *Notornis* 32: 271-309.

Park, S. 1995. Natural Environment Regional Monitoring Network Coastal and Estuarine Ecology Programme 1994/95. Environment B.O.P Environmental Report 95/20. Environment BOP, Whakatane. 102p plus appendices.

Park, S. 2000. Benthic Macrofauna Monitoring. Environment B.O.P Environmental Report 2000/15. Environment B.O.P, Whakatane. 28p.

Pridmore, R. D.; Thrush, S. F; Hewitt, J. E.; Roper, D. S. 1990. Macrobenthic community composition of six intertidal sandflats in Manukau Harbour, New Zealand. *New Zealand Journal of Marine and Freshwater Research 24*:81-96.

Pritchard, D. W. 1967. What is an estuary, physical viewpoint? p 37. *In*: Estuaries. Lauff, G. *Ed.* American Association for the Advancement of Science Publication No. 85, Washington DC.

Probert, P.K. 1984. Disturbance, sediment stability, and trophic structure of soft bottom communities. *Journal of Marine Research 42*:893-921.

PSDDA; US Army Corps of Engineers 1989. Unconfined open water disposal of dredged material, Phase II. Anonymous. AD-A213 372.

Rhodes, L.L.; Burkholder, J. M.; Glasgow, H. B.; Rublee, P. A.; Allen, C.; Adamson, J. E. 2002. *Pfeisteria shumwayae* (Pfeisteriaceae) in New Zealand. *New Zealand Journal of Marine and Freshwater Research 36*: 621-630.

Robertson, B.M. 1993. Southland coastal water quality review. Report prepared for the Southland Regional Council. Robertson Ryder & Associates, Dunedin. 157p.

Robertson, B.M. 1995. Southland Estuaries: heavy metal monitoring. Report prepared for the Southland Regional Council. Robertson Ryder & Associates, Dunedin. 35p.

Robertson, B.; Jensen, S. 2000. Assessment of benthic impacts from the Clifton wastewater treatment plant discharge. Prepared for Beca Steven. *Cawthron Report Number 580.* 28p.

Robertson, B. 2001. Review of trophic status of New River Estuary in relation to Clifton wastewater nutrient contribution. Prepared for Beca Steven. *Cawthron Report Number 618*. 17p.

Roper, D.S.; Smith, D.G.; Read, G.B. 1989. Benthos associated with two New Zealand coastal outfalls. *New Zealand Journal of Marine and Freshwater Research 23*: 295-309.

Roper D. S.; Thrush, S.F.; Smith, D.G. 1988. The influence of runoff on intertidal mudflat benthic communities. *Marine Environmental Research 26*: 1-18.

Roper, D. 1990. Benthos associated with an estuarine outfall, Tauranga Harbour, New Zealand. *New Zealand Journal of Marine and Freshwater Research 24*: 487-498.

Ryder, G.I. 1993. Invercargill Stormwater Parts 2 and 3: impact assessment and management options. Report prepared for the Southland Regional Council. Robertson Ryder & Associates, Dunedin.

Ryder, G.I. 1994. Kaikorai Catchment – sources of water quality contaminants. Report prepared for Otago Regional Council. 43p.

Ryder, G. 1995. Oreti Catchment: Aquatic biota/chemistry assessment. Report prepared for the Southland Regional council. 133p.

Sagar, P. 2000. Christchurch wastewater treatment plant: assessment of environmental effects: birdlife. NIWA Client Report CHOO/72.

Schuckard, R. 2002. Wader distribution at Farewell Spit, Golden Bay and Tasman Bay. Department of Conservation. Nelson/Marlborough Conservancy, Nelson. 39p.

Shearer, J.J. 1989. Pelorus river water quality during low to medium flows. Catchment and Regional Water Board Report.

Stevens, L. 1997. Preliminary assessment of leachate effects from the Havelock Landfill. Prepared for Marlborough District Council. *Cawthron Report No. 430*. 13p.

Stoffers, P.; Glasby, G.P.; Wilson, C.J.; Davis, K.R.; Walter, P. 1986. Heavy metal pollution in Wellington Harbour. *New Zealand Journal of Marine and Freshwater Research 20*: 495-512.

Strickland, J.D.H.; Parsons, T.R. 1968. *A Practical Book of Seawater Analyses.* Fisheries Research Board of Canada Bulletin 167. 311p.

Sutherland, R.D.; Kirk, R.M.; Bell, D. 1992. Natural Processes and Environmental Hazards in the Marlborough Sounds: Issues and Options. Prepared for Marlborough District Council. 39p plus appendices.

Sutton, R.R. 1977. Bird life and fisheries. *In: A History of New River Estuary and Environs*. Compiled by P. Chandler for New River Estuary Technical Advisory Committee 1973.

Tasman District Council. 1998 (updated May 2002). Environmental Information on Waimea Inlet and Environs. *Annotated list of references relevant to assessment of the environmental health of Waimea Inlet*. 7p.

Taylor, N.H.; Sutherland, C.F. 1953. *Soils of North Auckland.* Department of Scientific and Industrial Research Publication No. 44, Wellington. 16p.

Thompson, B.A. 1987. Tamaki Estuary water quality guidelines 1985-86. Report Number TP42. Auckland Regional Water Board, Auckland. 30p.

Thoms, M.C. 1981. Sedimentation in the New River Estuary, Southland. Unpublished MSc thesis (geography), University of Canterbury.

Turner, S.J.; Thrush, S.F.; Pridmore, R.D.; Hewitt, J.E.; Cummings, V.J.; Maskery, M. 1995. Are soft-sediment communities stable? An example from a windy harbour. *Marine Ecology Progress Series 120*: 219-230.

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

URS. 2001. Assessment of environmental effects for Christchurch wastewater discharge. Report prepared for Christchurch City Council. URS New Zealand Ltd, Christchurch.

Water Quality Centre. 1988. Design of an ecological monitoring programme for Manukau Harbour. Technical Publication No. 79. DSIR, Hamilton.

Wilson, D. 1989. Geochemistry of estuarine sediments – the Kaikorai Estuary. Unpublished Dip. Sci. (Geology) thesis, Otago University, Dunedin.

Wilson, J. G.; Jeffrey, D. W. 1994. Benthic biological pollution indices in estuaries. *In* Kramer, K. M. J. (Ed), *Biomonitoring of Coastal Waters and Estuaries*. CRC Press, Boca Raton, FL, USA. 311-327.

Wilton, K. M.; Saintilan, N. 2000. Protocols for mangrove and saltmarsh habitat mapping. ACU Coastal Wetlands Unit technical report 2000/01. NSW Department of land and Water Conservation, Sydney. 27p.

Woodward-Clyde (NZ) Ltd. 1996. Mapua Site – Assessment of Environmental Effects. Report for Tasman District Council.

Zar, J.H. 1999. Biostatistical Analysis, 3rd Ed. Prentice-Hall International, New Jersey. 662p.