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The Impacts of Urban Stormwater in Auckland's Aquatic Receiving Environment

A Review of Information1995 to 2005

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The Impacts of Urban Stormwater in Auckland's Aquatic Receiving Environment: A Review of Information 1995 to 2005

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Context

In 1995 the Auckland Regional Council published TP53, a synthesis of available information on the effects of urban stormwater on Auckland's aquatic receiving environment. Since this report was produced over a decade ago, a large body of research and monitoring information on this topic has been gathered and is stored in a wide range of locations.

In 2005, the Auckland Regional Council sought to provide an update to TP53, which led to the development of this report. The purpose of this report is to provide a single reference document that consolidates and summarises the key findings from a significant amount of monitoring and research information on the impacts of urban stormwater on Auckland's aquatic receiving environment that was produced between 1995 and 2005.

This report is a review of available information and it provides a comprehensive reference list to direct readers to further information. It is intended that this report will facilitate the broader use and application of research outcomes, by making information more accessible to a wider audience in this consolidated format.

The Auckland Regional Council recognises that a considerable amount of work has been undertaken in this field since 2005 and note that this has not been included in this report. The Auckland Regional Council will consider undertaking a further update on the state of knowledge about the impacts of urban stormwater in future.

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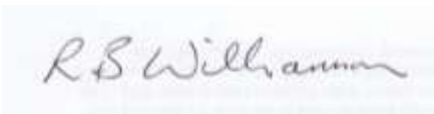

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1 Executive Summary

In 1995, the Auckland Regional Council (ARC) published a report entitled "The Environmental Impacts of Urban Stormwater Run-off". This report was an overview of urban stormwater and its impacts on receiving waters, and summarised the state of knowledge at that time.

Since then, a large amount of scientific research and monitoring has been conducted, significantly furthering our understanding of stormwater impacts. The ARC has therefore commissioned the production of an updated review, covering the decade 1995–2005.

This document provides this updated review. It aims to critically examine and summarise information that was produced between 1995 and 2005 on the impacts of stormwater on the aquatic environment in Auckland. It draws together the large body of published information on stormwater impacts into a single overview report that can be used as a "one-stop" source of information and as a bibliographic reference for those requiring more detailed information.

The following key areas are covered:

- the **Auckland environmental setting** as it affects the impacts of stormwater on aquatic receiving environments – the nature of Auckland's streams, estuaries, and harbours, their response to stormwater contaminants, and in particular how they can be classified to aid resource management;
- **key sources of urban stormwater contaminants** – what is known about where heavy metals, sediment, pathogens, and organic contaminants come from in Auckland, and hence how management can be appropriately targeted;
- **environmental guidelines** – how monitoring data for key stormwater contaminants is interpreted to determine the potential for adverse ecological or human health effects arising from stormwater discharges; and
- **impacts on water quality, sediment contamination, and ecological health** in urban streams and estuaries. This is the major part of the review, and is covered in separate chapters on the freshwater and marine environments.

The review has found that significant advances have been made in a number of areas, which together have greatly improved our understanding of the history, spatial extent, nature, causes, ecological effects, and likely future impacts of stormwater-derived chemical contamination.

Important advances include:

- Development of **classification methods** for freshwater and marine areas that provide a framework for objectively assessing and managing stormwater effects.
- Acquisition of a large **database on chemical contaminant concentrations** in waters, sediments (particularly marine sediments), and biota which have been used to

assess the spatial extent, temporal trends, and potential ecological impacts of stormwater-derived contamination.

- Development of **systematic monitoring methods** (a "blueprint") for measuring contaminants and benthic ecology in marine sediments, improving the reliability of monitoring data and providing a sound basis for testing models that predict the ecological consequences of stormwater contamination.
- Improved understanding of **sediment transport and accumulation processes** in estuaries, which have helped to decipher the history of sediment-related impacts and contributed to the development of models to predict sediment and contaminant accumulation in developing areas.
- Detailed studies of **contaminant sources in urban catchments**, so that source-control measures can be accurately targeted. Refining the data gathered to date remains a challenge to improve the reliability of existing estimates.
- Improved understanding of the **effects of land use** on freshwater receiving environments, and initial investigations to link land use pressure measures with effects in receiving waters.
- Improved understanding of **estuarine ecosystem functioning**, and the **consequences of fine sediment deposition** (an important urban contaminant).
- Development of the **"Healthy Benthic Community Model"** to assess and predict the ecological health of Auckland's estuaries in relation to key stormwater contaminants. This will improve the reliability of sediment quality guidelines, providing greater certainty for resource management.

Since 2005, work has continued in these areas, further improving our knowledge of stormwater impacts in aquatic receiving environments, enabling the development of more reliable tools to predict future impacts of urbanisation and test the effectiveness of resource management scenarios.

The knowledge acquired in the decade 1995–2005 provides a solid foundation for future scientific research and resource management activities, which together should provide the tools required to further improve stormwater management and reduce future impacts of urbanisation.

The information reviewed to date clearly shows that urban stormwater discharges can have serious long-term impacts on the health of receiving waters. Continued efforts are therefore required to prevent or minimise on-going effects and, where possible, restore impacted environments.

2 Introduction

Stormwater is a major hazard affecting the quality of aquatic receiving environments in Auckland's urban areas. It is appropriate, therefore, that a large effort has been directed towards improving urban stormwater management to reduce environmental impacts.

A fundamental component of the stormwater management framework is knowledge – understanding the risk factors associated with stormwater, and how the receiving environment responds to stormwater discharges. Armed with this knowledge, sensible management strategies aimed at reducing impacts and restoring degraded environments can be developed.

A large knowledge base has been built up over the past 20 years or so which has helped to define the nature of the stormwater problem, and hence what must be done to solve it. In 1995, the Auckland Regional Council (ARC) published a report entitled *The Environmental Impacts of Urban Stormwater Run-off* (ARC 1995). This report was an overview of urban stormwater and its impacts, and summarised the state of knowledge at that time. Since then, an even greater body of information has been produced that significantly furthers our understanding of this issue.

2.1 Purpose of this review

The purpose of this document is to update the 1995 review by critically examining and summarising information that was produced between 1995 and 2005[1] on the impacts of stormwater on the aquatic environment in Auckland.

The goal was to produce a document that drew together the large body of published data and information on stormwater impacts into a single report that could be used as a source of up to date information, as an overview of key issues, and as a bibliographic reference for those requiring more detailed information. As a result of the review process, information gaps would be identified for future study.

The scope of the review is limited to studies conducted in the Auckland region (although, where relevant, some reference has been made to other New Zealand studies), and that have been described in publicly available reports.

2.2 Approach

The approach taken was to describe key studies in some detail and provide reference to secondary studies, so interested readers can follow up on these themselves.

[1] A few important studies undertaken during this period, but published in 2006 during the production of this review, have also been included where it was felt they contributed significantly to the knowledge base, or to guide readers towards key on-going research.

Information from a wide variety of published sources was used, including scientific papers, reports, and conference presentations. The aim was to capture the information within a larger narrative so studies are linked and their importance can be seen within a broader context.

While the review has tried to provide comprehensive coverage of key issues, the large number of studies that have been undertaken since 1995 means that there are almost certainly some information sources that have not been included in the review. However, the major issues have been addressed, and much of the information not currently publicly available (eg confidential consultancy reports, or work yet to be formally published) would not greatly change the conclusions drawn at the time of the review.

2.3 Structure and content

The narrative structure used in this review is designed to cover the following key areas:

- the **Auckland environmental setting** as it affects the impacts of stormwater on aquatic receiving environments – the nature of Auckland's streams, estuaries, and harbours, their response to stormwater contaminants, and in particular how they can be classified to aid resource management – this is covered in Chapter 3;
- the **key sources of urban stormwater contaminants** – what is known about where heavy metals, sediment, pathogens, and organic contaminants come from in Auckland, and hence how management can be appropriately targeted – Chapter 4;
- **environmental guidelines** – how monitoring data for key stormwater contaminants is interpreted to determine the potential for adverse ecological or human health effects arising from stormwater discharges – Chapter 5; and
- **impacts on water quality, sediment contamination, and ecological health** in urban streams and estuaries. This is the major part of the review, and is covered in Chapters 6 (freshwaters) and 7 (marine environment).

The ARC (1995) review was used as a starting point. At the beginning of each chapter and major section there is a "state of knowledge in 1995" section in order to "set the scene". Advances made since 1995 are then described, and information gaps identified.

Unlike the 1995 review, this review seeks to provide more than an overview illustrated by suitable examples. As described above, attempts were made to locate and review all publicly available studies – at least all key, new or unique ones. In cases where the information from a particular study is only applicable to a localised area, and there is little further information to be obtained that is not already described from other key or earlier studies, then the information is referenced rather than described in any detail.

2.4 State of knowledge in 1995

In addition to the ARC (1995) review, several other key documents available at that time provided valuable coverage of urban stormwater issues:

1. VANT ET AL., 1993. *Effects of future urbanisation in the catchment of Upper Waitemata Harbour*. NIWA Consultancy Report No. ARC220R. Prepared for Auckland Regional Council, June 1993.
2. WILLIAMSON, 1993. *Urban run-off data book*. Water Quality Centre Publication No. 20. Hamilton, DSIR Marine and Freshwater.
3. ARC, 1994a. *The distribution and fate of contaminants in estuarine sediments: Recommendations for monitoring and environmental assessment*. Auckland Regional Council Technical Publication TP47.

These documents were attempts to draw together information on stormwater quality and its impacts on the environment, and were major factors in determining the direction of management strategies and scientific investigations from the mid-1990s to the early 2000s.

Information from these documents, as well as the 1995 review, have used throughout this document to define the "state of knowledge in 1995".

As well as these earlier reports, an overview of stormwater impacts in the New Zealand aquatic environment (EVA et al. 2002), and the ARC stormwater management manual "TP10" (ARC 2003a) provide useful broader scale references.

3 The Auckland Environmental Setting

The impacts of stormwater are greatly influenced by the nature of the receiving water bodies, which in Auckland range from tiny streams, to sheltered estuaries, and even exposed ocean coastlines.

The close connection between urban areas and the marine environment is a key feature of the Auckland setting, and consequently the impact of stormwater on marine receiving environments, especially estuaries and harbours, is a major issue (see Chapter 7).

Most streams are small and short, and there are no large freshwater bodies receiving urban stormwater. Because streams are small, stormwater discharges and associated urban development can have major impacts (see Chapter 6).

Since 1995, major efforts have been directed towards improving our understanding of how streams and marine receiving waters respond to stormwater discharges, and hence improving our ability to measure and predict the scale, nature, location, and duration of impacts that might occur. This chapter describes this progress.

3.1 Freshwater receiving environments

"Streams and wetlands are often the first aquatic resources to receive urban discharges. Their enhancement and protection are important in their own right, and are necessary to protect and enhance estuarine and marine resources downstream" (ARC 2004a).

State of knowledge in 1995

In 1995, it was known that Auckland's streams, riparian areas, and floodplains had been highly modified to accommodate urban development and efficiently convey flood flows. However, there was no systematic description of streams that allowed managers to understand the key functions, values, or uses associated with the various types of streams, or their susceptibility to stormwater discharge impacts.

Advances since 1995

Major advances in the understanding of Auckland's freshwater resources have been made since that time. We now have a clearer understanding of the types of streams found in the Auckland region and an improved knowledge on how these streams function. This has led to:

- methods to classify stream reaches, which will enable viable stream restoration options to be selected; and
- a clear description of stream functions, uses and values.

This information forms the basis for an objective, structured, approach for managing urban streams.

In addition to these advances in understanding the functions and types of freshwater systems, a large body of information describing the nature of Auckland's streams and the impacts of stormwater on water quality, ecology, habitat etc has also been collected. This is summarised in Chapter 6.

3.1.1 Stream types

The vast majority (approximately 90 per cent) of streams in the Auckland region are short (first or second order), narrow (channel width <2 metres), and contained within small (<100 ha) catchments (ARC 2004a). Managing these small streams to maximise the stream ecological function and human uses presents a major challenge.

A comprehensive study of the ecological conditions in Auckland's urban streams was conducted in 2001 (Allibone et al. 2001), which formed the basis of a stream categorisation method. This method allows managers to classify stream reaches, and by comparing this classification with known stream functions (Section 3.1.2), management strategies can then be devised. Much of this advance has been described in Framework for Assessment and Management of Urban Streams in the Auckland Region (ARC 2004a). North Shore City's "Kokopu Connection" project details work conducted on classifying North Shore streams (NSCC 2003, 2004a; individual stream reports are available at www.northshorecity.govt.nz). Stream classification methods and management objectives are described further in Chapter 6.

Under the ARC's classification system, Auckland stream reaches fall into one of six types (Figure 1). The nature of these stream types is illustrated in Figure 2.

Figure 1

The ARC protocol for assigning urban stream reach types (ARC 2004a).

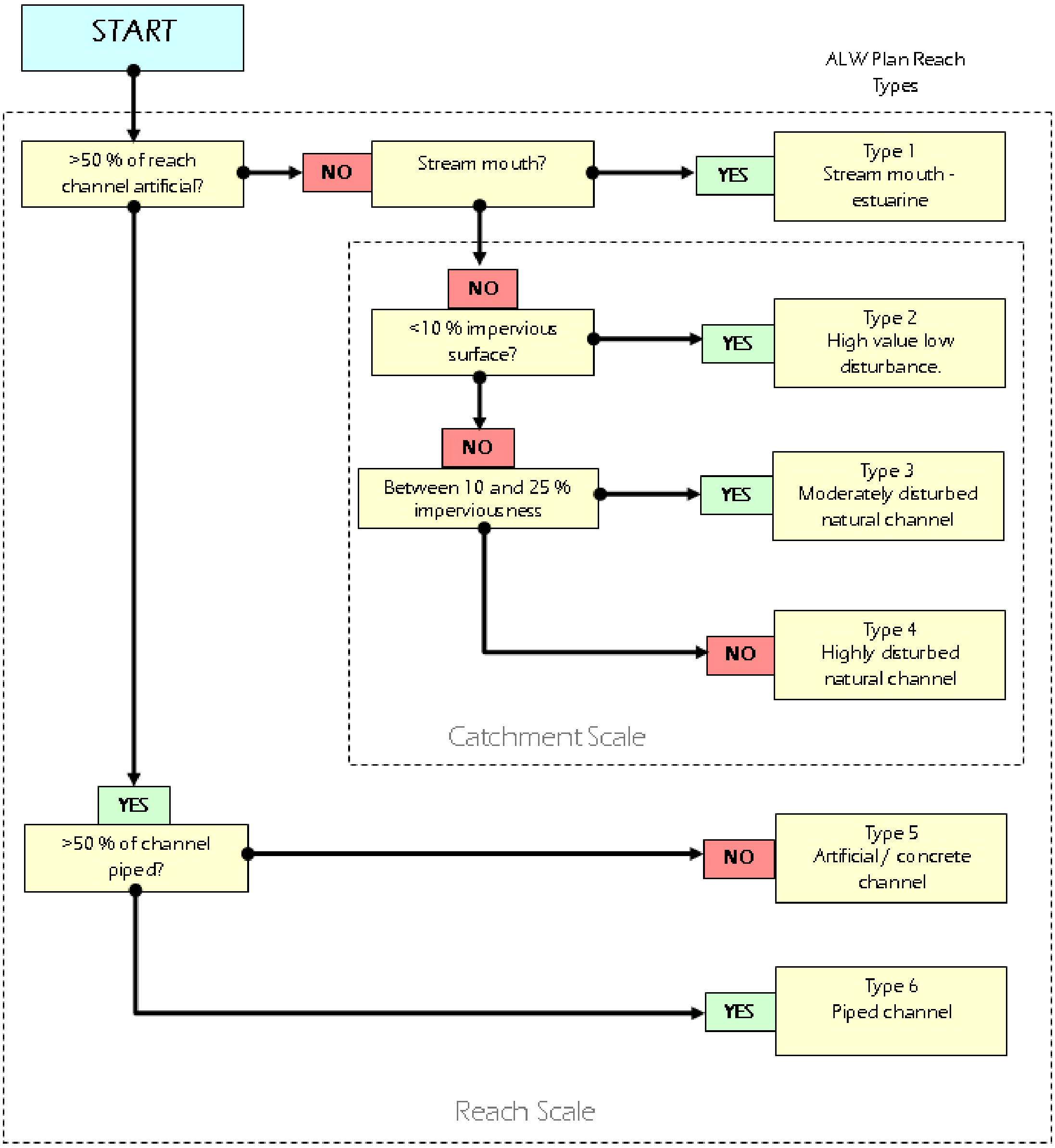

Figure 2

Typical appearance of the six ARC stream reach types.

Type 1: Stream mouth – Kaipatiki Creek
(NSCC 2004b)

Type 2: High value, low disturbance
(ARC 2004a)

Type 3: Moderately disturbed natural channel
(ARC 2004a)

Type 4: Highly disturbed natural channel
(Elliot et al. 2004)

Type 5: Artificial concrete channel
(Elliot et al. 2004)

Type 6: Piped channel – Hillcrest Creek
(NSCC 2004b)

3.1.2 Stream functions, values, and uses

Auckland streams provide many functions and values that can be maintained, restored, and enhanced with improved management. Primary stream functions include the following (summarised from ARC 2004a):

Ecology – urban streams provide living and breeding space for freshwater fish, invertebrates and plants.

Connectivity – stream functions are determined in large measure by their connectivity between reaches from the headwaters to the sea. For example, many native aquatic species (eg fish) require access to the sea to complete their life cycles. Stream functions are also directly affected by connectivity with the riparian zone, the floodplain, and groundwater. For example, insects are an important food source for fish and birds, and have both aquatic (larva) and terrestrial (adults) life stages.

Water Quality – good water quality is necessary to support ecosystem functions and human uses in streams and coastal areas. Streams provide for the physical, chemical, and biological processing of contaminant inputs, including those related to wastewater and stormwater.

Flood Management – streams and their surrounding floodplains are the drainage network for a catchment and are an important part of the flood management system. Many urban streams have been highly modified (piped and channelised) to convey flood flows and protect property from flood damage. Conveying flood flows in upper reaches can lead to more severe flooding downstream.

Amenity/Cultural – streams in urban areas have a number of values related to human use. These values are often interrelated and include:

- Amenity – streams flowing through reserves (or private land) may include walkways or picnic areas and are valued for their visual appeal.
- Recreation – paths along streams are popular for walkers and joggers. Non-contact recreational uses include kayaking in the lower reaches of larger urban streams, although full contact recreation such as swimming and playing are generally discouraged in urban streams to protect public health.
- Cultural/Community – significant cultural and/or community values are attached to a number of Auckland's urban streams. Waicare groups, local Iwi, or other community organisations may have special connections to particular streams, and often participate in protecting and improving their condition.
- Economic – streams provide a variety of indirect economic benefits related to tourism, commercial and business uses, and property values. In addition, streams provide for the treatment, processing, and attenuation of contaminants at no direct cost.

The primary focus of urban stream management is to (1) maintain or enhance the quality of all reach types, (2) optimise the connectivity between reaches within a catchment, (3) provide for public use and amenity values, (4) protect public health and safety, and (5) protect and enhance cultural values. ARC (2004a) describes approaches for prioritising the management of these functions and values in the various classes of urban streams.

3.2 The marine receiving environment

State of knowledge in 1995

By 1995, the physical and ecological characteristics of the Auckland marine receiving environment were generally well understood (eg Cameron et al. 1997). Some knowledge of processes influencing the behaviour and fate of contaminants in the marine environment had also been obtained, largely from studies conducted in the Manukau Harbour (Williamson & Hume 1990; Williamson et al. 1991, 1992a, 1992b; ARC 1994). For example, the processes by which contaminated sediments self-cleanse (termed "sediment recovery") had been the subject of a study funded by the ARC (ARC 1994a, Williamson & Hume 1990). The major processes are summarised in Table 1 below.

Table 1
Key mechanisms of sediment recovery (adapted and updated from ARC 1994a).

Process	Mechanism
Resuspension of fine sediments and dispersal by currents	Wave action on all sediment textures
	Intertidal bank erosion in muds, sandy mud or muddy sand
	Currents in channels
Sediment mixing and dilution	Wave action on all sediment textures
	Bioturbation
Burial	Sedimentation
Solubilisation	Metabolism of organic material containing trace elements
	Sediment phase changes (FeS to FeOOH)
	Complexation (eg dissolved organic matter, S2-, S2O32-)
	Exchange of organic compounds between sediment organic carbon and pore water phases
Exchange with overlying waters and dispersal by currents	Tidal pumping
	Diffusion
	Wave mixing
	Bioturbation
	Burrow irrigation
Degradation	Physical, microbial and chemical processes

The recognition that marine receiving environments could have zones of deposition (accumulation), transportation, and erosion was particularly helpful in describing the fate of sediments and sediment-associated contaminants. The concept of depositional zones was first quantitatively applied by Vant et al. (1993) to predict the effects of urban development on the Upper Waitemata Harbour. They proposed that depositional zones in sheltered tidal creeks, such as Lucas Creek, are typically 4 per cent of the contributing watershed area and accumulate 75 per cent of the fine sediments and associated contaminants that are discharged there. While this was a simplification of a complex situation, it was a major step forward because it allowed managers to link watershed characteristics to the state of contamination in marine sediments.

By 1995, it was also realised that mangroves were spreading in many of Auckland's estuaries. The reason was thought to be a least partly due to higher sedimentation rates and infilling (Roper et al. 1994).

However, as with the freshwater environment, there was no systematic classification of estuaries and harbours that would enable managers to understand or predict how the marine resource would respond to stormwater inputs. To do this, conceptual and mathematical models that would link activities on land to receiving water impacts were required.

Advances since 1995

Most of the advances in our understanding since 1995 have come from four major study areas:

1. Modelling contaminant build-up in estuarine sediments, which included the development of a conceptual classification of Auckland's marine receiving environment (Section 3.2.1).
2. Instituting the Regional Discharges Programme (RDP) to manage stormwater impacts (Section 3.2.2).
3. Deciphering the history of sedimentation from sediment profiles (Section 3.2.3); and
4. Benthic ecology monitoring (Section 3.2.4).

These are summarised in the following sections.

3.2.1 Predictive modelling of urban stormwater impacts in estuaries

Further advances in our understanding of the marine environment and the impacts of urban stormwater-related contaminants came from initiatives to test and improve the modelling developed and described by Vant et al. (1993) and summarised in ARC (1995).

Further information on the models described below is given in Chapter 7. A description of key estuarine processes and classifications underpinning these models is provided in an attachment to this chapter (Sections A3.1 and A3.2).

Urban Stormwater Contaminant Model #1 (USC1 Model)

Because of the desire by resource managers to apply the approach of Vant et al. (1993) to other parts of Auckland, research studies were conducted to see if the predictions of urban contaminant distributions and accumulation made under the model were valid (ARC 1998; Williamson & Morrisey 2000; Morrisey et al 2000; Morrisey et al. 2003).

Sediment contaminant surveys[2] were undertaken in urban estuaries and demonstrated reasonable agreement between predictions and observations. The concept of "deposition zones" – eventually termed "Settling Zones" (SZ) – seemed to be scientifically defensible and had "arrived" as a useful management tool for assessing where, and how much, contaminant accumulation might occur in Auckland's muddy estuaries.

These detailed studies confirmed the picture postulated by Vant et al. (1993) of the processes occurring in Settling Zones. These processes are described in the appendix to this chapter (Section A3.1).

Urban Stormwater Contaminant Model #2 (USC2 Model)

The second step was to try and extend predictions beyond the muddy estuaries to other parts of the marine receiving environment. This led to the development of the USC2 Model (ARC 2002a, ARC 2004b, Green et al. 2005a and b).

The concept of Settling Zones worked well for tidal creeks and some sheltered embayments, but was not useable in the wider harbour, for example the open waters of the middle Waitemata Harbour, because it was unable to handle redistribution processes – eg the remobilisation of contaminants from one part of an estuary to another.

To extend the model beyond the muddy estuaries, a conceptual picture of the key processes influencing the fate of sediments and contaminants in the total marine receiving environment was required. To achieve this, NIWA scientists undertook visual and sampling surveys of Auckland estuaries to describe them in terms of well-known, global, physical and chemical processes that occur in estuaries. The result was a classification of Auckland estuaries (ARC 2002a), which laid the foundation of the USC2 model.

Estuaries were classified in terms of "arms", "main body" and "throat", with different physical processes operating in each area[3]. Sub-estuaries were classified as Primary Deposition Areas (PDA) or Secondary Resuspension Areas (SRA), depending on the dominant physical processes operating in these areas. In the simplest terms, PDAs are muddy, sheltered estuaries where sedimentation/deposition after storm inputs is the primary process affecting the fate of fine sediments (and contaminants). The SRAs are less sheltered, sandy estuaries where there is sufficient wave energy between storm events to move the fine sediments deposited during storms.

Key processes were also summarised as a set of "rules", which physical and contaminant processes that occur in Auckland estuaries obey. The rules are based on both a sound general scientific basis (ie that which can be gleaned from the

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[2] Further information on these surveys and a description of the model are given in Chapter 6. The contaminants studied were the heavy metals copper, lead, and zinc, which are well known urban stormwater contaminants.

[3] The key estuary characteristics and processes are explained and shown diagrammatically in the appendix to this chapter (section A2.2).

international and New Zealand literature) and also on the observations made in Auckland estuaries. Some examples of the rules are summarised in Table 2 below. Examples of the application of the USC2 model are given in Chapter 7.

Table 2

Examples of "estuary rules" (from ARC 2002a).

Rules of accumulation:
Rule 5. Mud is most likely to accumulate where the bed is already muddy. The corollary is that mud is least likely to accumulate where the bed is sandy.
Rule 6. Waves and mud do not co-exist. Muds are hindered from accumulating in exposed areas, but they accumulate in sheltered areas.
Rules governing longitudinal distribution of sediments:
Rule 7. Terrestrial fine sediment mostly deposits at the head of the estuary where streams discharge. Two processes are happening here. Firstly, there is a large current-speed drop where freshwater meets the estuary basin, which rapidly reduces transport capacity of the stream. Secondly, fines suspended in the freshwater flocculate rapidly in the presence of very low salinities. This occurs at the head of the estuary where freshwater sources discharge and first contact seawater and results in the formation of larger particles with increased settling speeds. As a consequence of these two processes, terrestrial sediment is deposited mainly at the head of the estuary.
Rule 8. Channels act as extensions of streams at low tide. The fluvial/estuary transition with accompanying high-deposition zone moves down-estuary at low tide and up-estuary at high tide.

3.2.2 The Regional Discharges Programme (RDP)

The ARC's Regional Discharges Programme (RDP), which provides the technical basis for processing consents for stormwater and wastewater discharges throughout the Auckland region, was the vehicle for a further advance in receiving environment characterisation and classification. The monitoring component of the programme uses a network of sites to assess marine sediment contamination and benthic ecological health. Identifying suitable monitoring sites required mapping of the marine receiving environment to locate the zones where most contaminants would initially be deposited, and also sites beyond these zones that would characterise the wider marine area.

The Tamaki Estuary and the Upper Waitemata Harbour were critically examined in terms of the fate of contaminants discharged from the land in stormwater (Williamson & Green 2002). The concepts described above, and in appended Sections A3.1 and A3.2, were simplified and two main types of receiving environment were defined:

- Settling Zones, which were defined as areas which received the bulk of watershed discharge and where most of the terrigenous sediments and their associated contaminants were deposited. In terms of the processes occurring in estuaries, Settling Zones are Primary Deposition Areas.
- Outer Zones, which are all the other marine receiving environments. They may be Primary Deposition Areas, but are more likely to be Secondary Resuspension Areas. Basically Outer Zones are very complex environments that are generally far less contaminated than Settling Zones because they are remote from the watershed inputs or energy is too high for contaminants to accumulate.

Figure 3: Settling and Outer Zones in Shoal Bay.

There are three types of Outer Zones:

i. Primary Deposition Areas (PDA) that are beyond Settling Zones. While the dominant process is still deposition, they are separated from the watershed by Settling Zones. They are not the primary recipients of watershed discharges, but they will receive sediments and contaminants from the watershed via the Settling Zone.

ii. Secondary Redistribution Areas (SRA) that are beyond Settling Zones. These are similar to i. above, except that the energy (wind, tides waves) is high enough to bring into play secondary resuspension as the dominant sediment fate mechanism.

iii. SRAs that receive run-off directly from watersheds. Here catchment run-off is discharged directly into an open estuary, where the energy is too high for fine sediments and contaminants to accumulate.

These concepts were employed by the RDP to categorise and map receiving environments in Auckland (ARC 2002b). While more precise scientific definitions were available, the simple classification of the receiving environment into Outer Zones and Settling Zones enabled straightforward monitoring strategies to be developed, and captured the idea of Settling Zones being the primary recipients of stormwater contaminants.

Figure 4 illustrates the concepts of SZ and OZ for Hellyers Creek estuary, and Figure 5 and 6 illustrate the processes occurring in the SZ and OZ in the Tamaki estuary.

Figure 4

Settling Zones and Outer Zones in Hellyers Creek estuary.

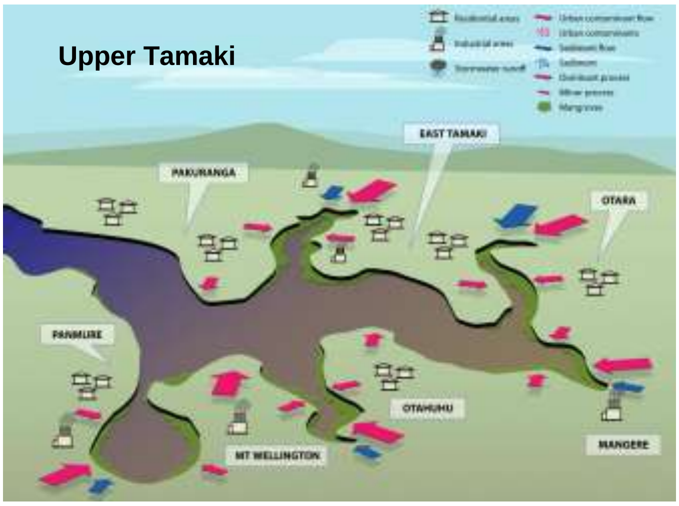
Classification of the Settling and Outer Zones in the Tamaki Estuary

The Tamaki Estuary can be conveniently divided into two parts – the Upper Tamaki lying upstream of the Pakuranga Highway, and the Outer Tamaki lying downstream, from Panmure to the throat.

The Upper Tamaki

This is a mixture of Settling Zones and an Outer Zone. The settling zones are the sub-estuaries or **arms**: Pakuranga, Otara, Middlemore, Otahuhu, and Panmure, while the Outer Zone is the **main body** of the Upper Tamaki. The reason for this classification is that the **arms** receive much of the run-off and trap most of the contaminants. The main body receives little run-off directly from catchments, but receives "overflow" or "leakage" from the **arms.**

Figure 5
Sediment transport processes in the Upper Tamaki Estuary.

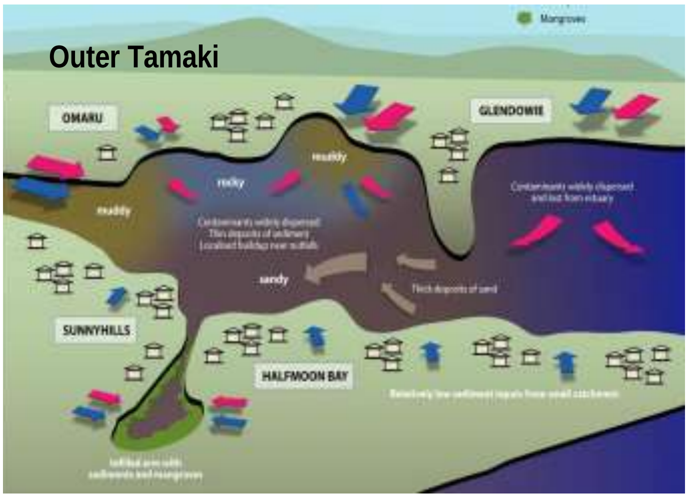


The Outer Tamaki

The Outer Tamaki Estuary illustrates the Outer Zone concept further.

Most of the Outer Tamaki is an Outer Zone and is part of the **main body** of the estuary. Here the estuary receives run-off directly from catchments such as Glendowie but the energy is too high for fine sediments and contaminants to accumulate except in a few sheltered bays. Therefore no Settling Zone is able to form.

Figure 6
Sediment transport processes in the Outer Tamaki Estuary.



The Outer Tamaki also receives some contaminants "leaking" out of the Upper Tamaki and uncontaminated sand from Tamaki Straights through the throat of the estuary.

3.2.3 Deciphering the history of sedimentation from estuarine core profiles

The history of a Settling Zone is often captured in the sediments that have accumulated. By carefully taking samples of sediment profiles – the sediments that have built up over time – and deciphering these profiles, scientists are able to shed light on how the nature of the estuary has changed with urbanisation. They also give indications of how the receiving environment will change in the future.

Pakuranga Estuary studies

Dating of the sediment profiles taken from Pakuranga Creek showed that sedimentation rates in the tidal creeks during urbanisation were much higher (approx. 30 mm/yr) than those occurring when the catchment was in agricultural land use (approx. 1–2 mm/yr) or in the original forest land cover (0.2–0.5 mm/yr; ARC 1998, Swales et al. 2002). The urbanisation in this catchment was largely carried out without erosion controls, and brought about substantial environmental changes in the upper parts of the estuary, including:

- continued infilling of shallow, intertidal areas – up to a 1 metre since 1960;
- changes in sediment texture, from sandy muds to muds; and
- rapid mangrove colonisation of formerly bare intertidal sediments.

Accompanying these physical and ecological changes was an increase in the concentrations of heavy metals and organic contaminants in the estuary sediments – this is described in Chapter 7.

Auckland region-wide studies

A major study entitled Evidence for the Physical Effects of Catchment Run-off Preserved in Estuarine Sediments (ARC 2001, 2002c) provided a comprehensive picture of sediment accumulation and general changes in the marine receiving environment on a regional scale. Because previous studies of sedimentation in Auckland estuaries had been focused on the tidal creeks near catchment outlets, the new research investigated sub-tidal and intertidal flat environments in the main body of estuaries. Most of the study sites were outside the Metropolitan Urban Limit, but the study still provided a valuable context in which to place and assess processes occurring in urbanised estuaries.

Important findings of relevance to urban stormwater impacts were:

- Over the last 50 years, intertidal flats have shoaled by ~0.5 m, the significance of which can be grasped by considering the fact that the average high tide water depth in many Auckland estuaries is <1 m.
- At decadal time scales, sediment infilling of Auckland estuaries will continue at several mm per yr. The effects of catchment sediment loads will be greatest in tidal creeks, at the catchment outlet. Even in largely intertidal estuaries, there is still no evidence that sedimentation rates are slowing down. Sedimentation rates in Auckland estuaries over the last 50 years are 2–3 times higher than the average

rate of sea-level rise at Auckland (1.3 mm/yr) since the early 1900s. Thus, sea level rise has only partially reduced the rate of estuary aging, which has been accelerated by increased catchment sediment run-off.

- All estuaries follow similar evolutionary paths as they infill with sediment. Water areas and depths decrease over time and as a result hydrodynamics, sediment deposition, and biological communities change. The relative dominance of the fluvial system increases as estuaries mature and tidal volumes shrink. As estuaries age and infill, sedimentation rates may increase (even in the absence of catchment sediment load increases) because the available deposition areas reduce in size. However, as Auckland estuaries age, their trapping efficiency will decrease and consequently sediment accumulation rates will decline as more sediment is exported to adjacent coastal waters.
- Auckland estuaries are at different points along this evolutionary cycle.
- Many of Auckland's urbanised estuaries are indeed at an advanced state of infilling. One of the most advanced is Henderson Creek, much of which appears to be approaching dry land (EVA et al. 2003d). Many of the mangrove areas are only flooded at spring high tide, while the low tide channel looks like a riverine channel bounded by a flood plain. Contaminants discharged from the urban areas upstream will therefore be mostly confined to the low tide channel, rather than spread across the estuary. A greater proportion of the contaminated sediments will be transported downstream to the lower estuary and Waitemata Harbour.

Figure 7
Henderson Creek Estuary below State Highway 16.

Clearly, the fate and impacts of stormwater contaminants is strongly influenced by the physical characteristics of the marine receiving environment. The studies referenced above have provided a much clearer understanding of how these environments function, and hence have improved our ability to monitor and predict the nature, scale, and location of stormwater impacts.

3.2.4 Advances from benthic ecology surveys and monitoring

A large number of animal species inhabit the soft sediment intertidal areas of the Auckland region. In a recent survey of six sites in the Manukau, 99 species were found on a single occasion (Auckland Regional Council unpublished data, ARC 2004c).

By 1995, numerous surveys had provided a good understanding of the types and distribution of animals living in Auckland estuaries and harbours. While detailed accounts were available for a few areas in each of these harbours and estuaries (eg Manukau Harbour, Tamaki Estuary), much of the information was descriptive or semi-quantitative, relatively old (ie greater than ten years), and lacked sufficient definition of spatial and temporal variation to allow defensible comment on existing conditions at a given location or the likely effect of a given stormwater discharge (NIWA 2000e). Since that time there have been a number of surveys, and on-going ecological monitoring, that have greatly improved the knowledge base.

Ecological surveys

Of the surveys conducted since 1995, three warrant special mention because they provided significant advances in our broad-scale understanding of ecological characteristics, and in providing benchmarks for on-going monitoring programmes:

- Qualitative description and mapping of the intertidal fauna and habitats of the Waitemata Harbour, during the mid–late 1990s (Hayward et al. 1997; ARC 1999a).
- Detailed quantitative survey of the benthic ecological values of the Upper Waitemata Harbour (ARC 2002d).
- Coastal and estuarine vegetation surveys and mapping (Morrisey et al. 1998 and 1999).

1 Waitemata Harbour ecological mapping

A survey of the ecology of intertidal (between low and high tide) areas of Waitemata Harbour bordering Auckland City – from the Whau Estuary to Bastion Point – was conducted between 1996 and 1998 (ARC 1999a). The report provides a map of intertidal habitats, supported by descriptions of each habitat type (eg mangrove forest, sea grass beds, sandstone reefs, cockle shell-covered flats). Subtidal (below low tide) fauna and sediments within the same area are described from samples collected between 1993 and 1995 from some seventy sites. Two broad habitat types are documented, shallow subtidal flats and subtidal channels, and eight "faunal associations" where the fauna are represented by a dominant animal (eg patches of horse mussel beds with animals that share the habitat created by the horse mussels). Maps are provided in the two reports to show how habitat type and species associations vary spatially in the central harbour region.

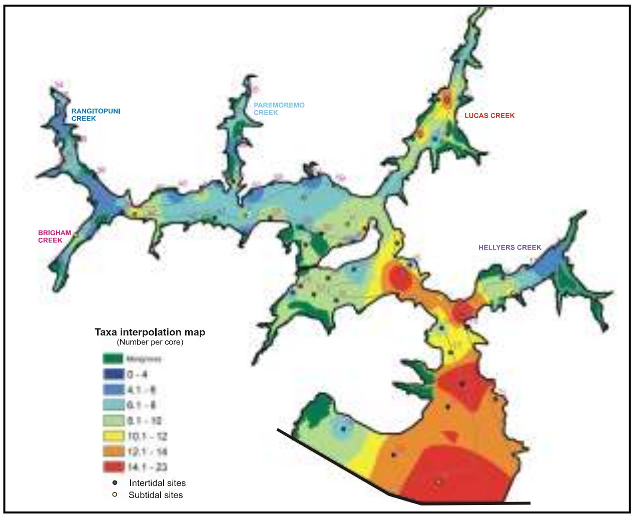
2 Benthic ecological values of the Upper Waitemata Harbour

A detailed survey to define the benthic ecological values of the intertidal and subtidal habitats of the Upper Waitemata Harbour (UWH), in light of plans for increased urban development in the UWH catchment was undertaken (ARC 2002d).

Despite the generally muddy nature of the upper harbour, and the perception that such habitats have low diversity and value, this survey showed that the intertidal and subtidal benthic communities and habitats of the UWH are generally in good condition, and are worthy of careful consideration when development occurs in the near future. Ecologically important areas of the UWH were identified as the main body of the outer UWH, Lucas Creek, Hellyers Creek, the northern side of UWH near the mouth of Paremoremo Creek, and the main body of inner UWH, in the vicinity of Herald Island.

Figure 8

Number of benthic taxa per core in Upper Waitemata Harbour (ARC 2002d).



The benthic community composition was best explained by the sediment mud content, and several benthic taxa were identified that were sensitive to this habitat characteristic. These taxa could therefore be used as "early warning" indicators of degradation related to increased sediment loads from developing catchments.

3 Distribution and changes in coastal vegetation

The margins of parts of the Auckland coastline (concentrating on the Waitemata Harbour) have been mapped to show the distribution of coastal and estuarine vegetation and changes over time in relation to changes in catchment land use (Morrisey et al. 1998 and 1999).

Five Auckland region sites were selected for on-going monitoring – Lucas Creek (UWH), Mangemangeroa (Whitford/Howick), Mullet Creek (Kaipara Harbour), Okura (North Shore), Puhinui Creek (Manukau Harbour). Of these, Lucas, Okura, and Puhinui Creek catchments have significant urban areas.

Along with the maps of the Waitemata Harbour benthic habitats and faunal distribution described above, these are a valuable resource for managers to determine the long-term effects of changing land use on the marine receiving environment.

Long-term ecological monitoring

Long-term benthic ecology monitoring has been carried out to provide an inventory of resources, feedback on estuary management activities, and a baseline upon which to assess future changes, especially those related to land use change or urban stormwater management initiatives.

Monitoring carried out in the Manukau Harbour since 1987 has laid the basis for understanding trends and spatial variability in benthic community composition at sites that are not directly impacted by stormwater (ARC 1994b, 2005a). Various research programmes have extended this understanding by looking at relationships and responses to defined perturbations (eg sediment dumps and chemical contamination – see Chapter 7 for details).

In 1994, a further long-term monitoring programme commenced in the Mahurangi Harbour (ARC 2003b, 2005b), which examined different environments (muddy and sandy subtidal as well as intertidal sites) than were being studied in the Manukau.

While these programmes did not consider urban stormwater impacts, they were fundamental in developing an understanding of the animals that inhabit various ecological niches, natural variability in benthic community composition, and the impact of fine sediment deposition (in the case of the Mahurangi, from pasture land), which is one of the major contaminants from urban development.

However, the main value of the monitoring programmes in respect to urban stormwater is the provision of a wealth of baseline information that can be used in future to assess urban stormwater impacts. They have already supplied information on the ecological effects of sedimentation, a potentially major impact of urban development. The monitoring data from these long-term programmes have also been used to develop the Benthic Community Model, which is a major advance in assessing the impacts of stormwater (see Chapter 7).

In 2000, a third "sentinel" ecological monitoring programme was established in the middle Waitemata Harbour, using five intertidal sites (ARC 2002e, 2006a), while other programmes incorporating different habitats (eg beaches and rocky reefs) have also begun (see Table 3). Monitoring that is directly relevant to urban stormwater effects is discussed in Chapter 7.

Table 3

Benthic ecology monitoring programmes in the Auckland region.

Locations	Habitats	Start dates	No. of sites	Reference
Manukau Harbour	Intertidal flats	1987	3–7	ARC 1994b, 2005a
Mahurangi Estuary	Intertidal flats, subtidal	1994	7	ARC 2003b, 2005b
Middle Waitemata Harbour	Intertidal flats	2000	5	ARC 2002e, ARC 2006a
Long Bay, Mairangi Bay, Torbay, Browns Bay	Intertidal beaches	Long Bay 1998 Others 1999	2 3	ARC 2005c
Waiwera, Stanmore, Little Manly, Long Bay, Torbay, Campbells Bay	Sub-tidal rocky reef communities	Long Bay 1998 Others 1999	24	ARC 2005c
Okura, Orewa, Waiwera, Puhoi, Mangemangeroa	Intertidal flats	Okura 2000 Others 2003	10 10 in each	ARC 2004d, ARC 2005d
Turanga, Waikopua (Whitford)	Intertidal flats	2004	10 in each	ARC 2005d
Upper Waitemata	Intertidal	2006	14	ARC 2006b

Locations	Habitats	Start dates	No. of sites	Reference
Harbour				
ARC Regional Discharges Project (RDP)	Intertidal flats	2002	72	ARC 2003d

3.2.5 Mangrove areas: a special case

Loved or despised, mangroves are a major feature of Auckland's urban estuaries. Because they are so widespread, and have been a topic of great public interest, they warrant special mention.

Figure 9a
Mangroves in the Auckland region.

In 1995, it was recognised that mangroves were spreading in many estuaries, and this was occurring in both urban and rural settings. The spread was perceived to be caused by increased sedimentation and changes in estuary morphology that trap sediment (eg causeways etc), but it was uncertain whether other factors might be at least partly responsible.

The phenomenon is now better understood (Green et al. 2003). Mangrove spreading is due to increased sediment run-off from the catchment, both past and present. Estuaries naturally trap and fill with sediments, and mangroves naturally spread in estuaries where climatic and other growth factors are favourable. However, accelerated erosion in developing and deforested catchments, often coupled with inadequate sediment controls to reduce sediment losses, has greatly increased the amount of fine sediment delivered to estuaries and harbours. Sand flat habitats have become smothered with silt, and in response, mangroves spread rapidly, expanding from the headwaters and sides of the estuary out into areas that were previously floored with clean sand.

Figure 9b
Silt deposition within mangroves.

And the perverse part is that mangrove spread and silt deposition are intimately bound together – mangroves help trap silt, and silt provides the environment that helps mangroves to thrive. The problem here then is an acceleration of what are otherwise natural processes.

There is little information available on how fish use mangroves, but on-going NIWA research is filling in some of the gaps. While it appears that fish species diversity is less than in some other estuarine habitats (eg sea grass beds), there are several species that do use mangrove channels, such as yellow-eyed mullet, grey mullet, smelt, and anchovies.

In other ways, mangroves contribute to the species and habitat diversity of New Zealand estuarine ecosystems. Microscopic bacteria decompose leaves dropped by mangroves, thereby recycling nutrients in the estuary and making them available for other photosynthetic organisms (eg algae). These in turn are significant sources of food to animals that live in the sediment, such as crabs, snails, cockles, and worms. Mangrove trees also provide anchorage surfaces for filter-feeding organisms such as black mussels, small barnacles, rock oysters, and Pacific oysters. Carnivorous scavengers and predators, such as mudflat whelks and snapping shrimp, form another strand in the food web.

The number of species found in mangrove stands and associated sediments is lower than on adjacent intertidal sand flats, but the community composition (the types of animal that make up the community) is quite different. Sand flat communities have a higher proportion of shellfish than muddy areas, but relatively more worms live in mud. The only safe conclusion is that the ecological function of mangroves is different from the ecological function of sand flats. When mangroves spread, they do so at the expense of other habitats, and the value – ecological and human – of those habitats that are consumed is lost. The habitats that yield to the spread are the lower intertidal and subtidal zones, which people prize for kaimoana, recreational opportunities, and aesthetic reasons.

3.3 Summary

Studies conducted since 1995 have added hugely to the database describing both the freshwater and marine environment in the Auckland region. The data has been used to improve our quantitative understanding of how these environments function, both physically and ecologically, and this has enabled the receiving environment to be classified in ways that aid management of these resources.

Freshwater environment

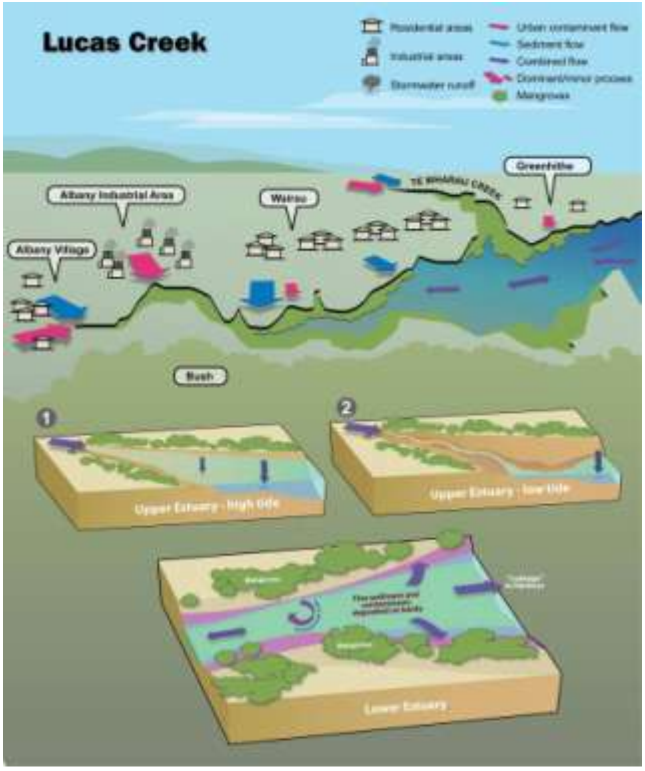
Ecological studies have lead to a stream classification system that provides objective procedures for assessing and managing streams of all conditions, from pristine to highly degraded. Sensible restoration objectives can therefore be established for impacted stream reaches.

Marine environment

Studies conducted since 1995 have provided scientific validity to the general concept of fine sediments and contaminants settling near the discharge point in sheltered estuaries and embayments.

Estuaries can be classified in terms of arms, main body and throat, where different physical processes operate. Sub-estuaries can be classified as Primary Deposition Areas or Secondary Resuspension Areas, depending on the dominant physical processes. This classification, along with estuary rules, underpins much of our understanding of the marine receiving environment and its response to incoming sediments and contaminants.

Figure 10
Tidal flow of Lucas Creek, Albany Village.



Deciphering the history of sedimentation revealed in estuary cores has shown that Auckland estuaries have significantly infilled in recent times (~0.5 m in 50 years). Infilling is proceeding faster than sea level rise. Because many estuaries are already at an advance state of infilling (the average high tide water depth in many Auckland estuaries is < 1m) there will be substantial changes in morphology and ecology in the foreseeable future. Effects may extend beyond the estuary, as sediment trapping efficiency decreases, and sediments (and contaminants) are discharged into the wider harbour or coastal areas.

The Regional Discharges Project has formalised the complex processes occurring in estuaries and harbours into the simple concepts of Settling Zones and Outer Zones. This concept has already proved useful in managing the marine receiving environment.

A3.1 Sediment settling and redistribution processes in estuaries

The settling process

At high tide estuaries are at "zero" gradient and are effectively backwaters or ponds. At low tide they are effectively an extension of the stream or drainage channel because stream flows are carried in low tide channels to the tidal front. At some point the channels widen and the channel takes on more of an estuarine quality, eventually widening out to the main body.

The fate of suspended sediment and attached contaminants depends on the state of the tide, the size of the storm and the morphology of the estuary at the tidal front.

At high tide, within a short distance of the discharge meeting the tidal front, currents will be low enough to provide ideal settling environments (Hume and McGlone 1986). The coarser particles settle by gravity because there is a large drop in the velocity of the water carrying these particles.

For smaller storms, or where stormwater discharges into a large estuarine area, in addition to settling by coarse particles, a proportion of the finer particles will begin to flocculate and the resulting flocs settle to the bed. For small watersheds and small storms, these processes occur in the near-shore areas. Therefore, the immediate fate of a proportion of the contaminants after entering the estuary is deposition by settling in the upper reaches of the estuary (Williamson and Morrisey 2000).

As the tide retreats, stormwater will flow in channels incised within intertidal flats. This may result in scouring of any previously deposited fine sediments (eg during the previous high tide) and deposition of coarse, urban-derived sediments in these channels. On reaching the tidal front, stormwater will be mixed and spread out over lower intertidal and subtidal areas, with the mixing/settling field therefore tending to spread down-estuary. A similar picture holds for the rising tide, except the mixing/settling field moves up-estuary (Williamson and Morrisey 2000).

In larger watersheds or during large storms, greater discharges will push stormwater further down the drainage channel irrespective of tide. During very large storms, fine particles and dissolved contaminants will be carried well out into the Bay as the large volume of fresh water displaces saline water from the estuary arm or channel and/or as higher buoyancy of the large fresh water inflow spreads over the top of the saline waters of the Bay. Coarse particles may still settle in the upper reaches because of the decrease in velocity or turbulence. At mid-high tides, the discharge field will tend to spill out over the top of the adjacent intertidal areas, which provide ideal quiescent settling areas. Stormwater solids tend to build up on the low tide channel margins (ARC 2002a). As the tide ebbs, most of the flow is concentrated in the low tide channel. Settling will occur only where channels widen significantly and the stormwater discharge can spread out over the intertidal and at low tides, subtidal areas.

The redistribution process

Once settled, contaminated sediment is intermittently resuspended and redispersed. Direct observations of the action of small waves (5–20 cm) show very high turbidity in shallow waters behind the tidal front (ARC 1994a; Green and Bell 1995). Very fine sediments (clays and fine silts) can be transported in suspension for large distances (100s m) until reaching quiescent areas, whereas the coarser fraction of the suspended material (medium silts to fine sands) settles within short distances (eg <10 m) of the point of resuspension. The continual advance and retreat of the tide means that contaminants can be spread widely over the intertidal zone.

Some of the sediment resuspended by waves will escape from the estuary on the ebb tide. However, since many estuarine arms discharge into effectively enclosed basins, much of the “escaped” material will return on the next flood tide. Because of estuarine processes termed “settling lag” and “scour lag’, fine sediments tend to march up-estuary (Postma 1967). This process was described for Orewa estuary (Williamson et al. 1998).

A3.2 Characteristics of Auckland estuaries (ARC 2002a)

Figure 11

Plan view of an "archetype" Auckland estuary.

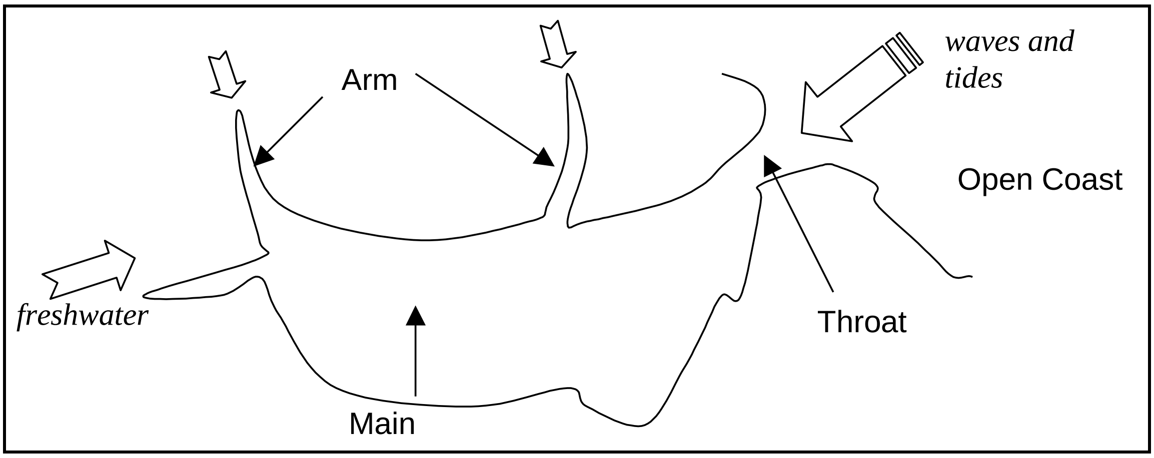


Figure 11 above shows the "archetype" Auckland estuary. Arms are sheltered tidal creeks leading to catchment drainage. The main sediment process is deposition of muds eroded from catchment soils or carried up into the sheltered arm from the main body. The main body is muddy in the upper reaches, where it connects with the arms associated with source streams. The lower reaches, which are connected to the coastal ocean via the throat, tend to be sandy. The main process is secondary redistribution of muds injected from arms during floods and of sands that are pushed through the throat by waves and tides. Secondary redistribution is achieved mainly by tides, internally generated waves, and gravitational flows. The main body is characterised by mixed sediments (mud and sand) and, in places, encroachment of muds on sands and vice versa. Two areas can be differentiated on the basis of processes:

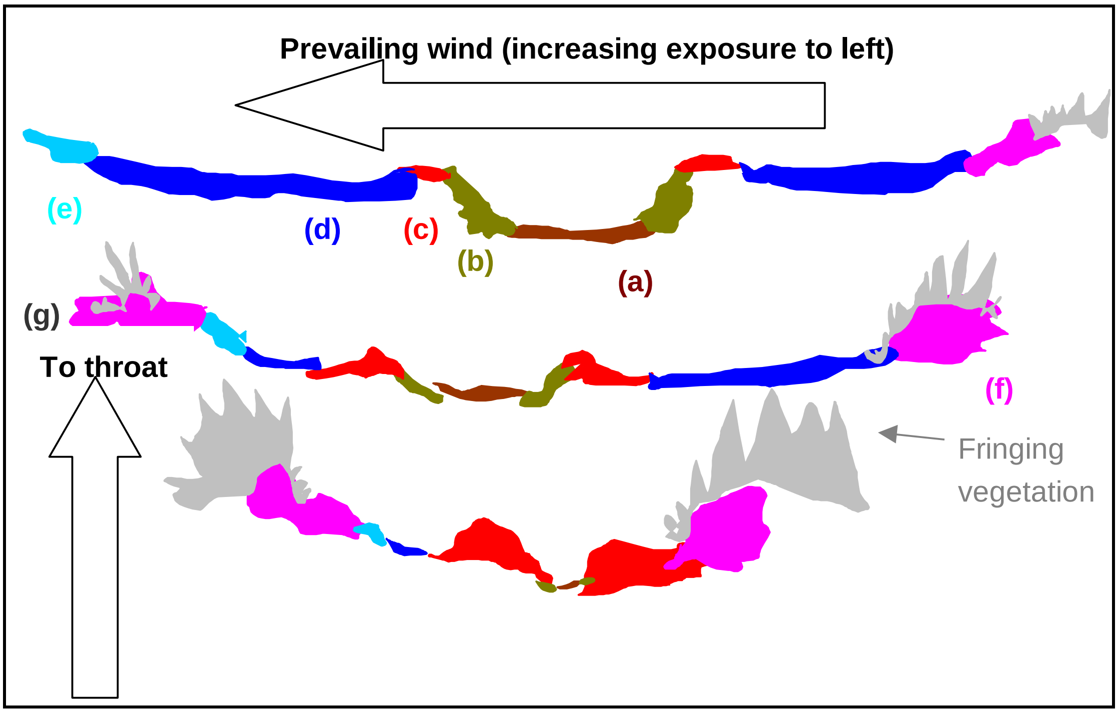
Primary deposition areas (PDA) are muddy sheltered estuaries where sedimentation/deposition is the primary process following storm inputs of fine sediments. Fine sediments and associated contaminants tend to be trapped here

Secondary resuspension areas (SRA) are less sheltered, sandy estuaries where there is sufficient wave energy between floods to move the fine sediments deposited during floods. SRAs are often "downstream" of PDAs, although they can also receive run-off directly from land.

In addition to the characteristic longitudinal gradient in sediment characteristics, there is also a typical cross-estuary gradient: sandy intertidal flats lie between areas of mud along the channel margins and mud or sand/shell on upper intertidal-flat margins.

Figure 12

Cross-sectional view of "archetype" Auckland estuary.



(a) Lagged channel bottom. Typically, a larger area of the channel bottom is lagged in the lower reaches of the estuary, where tidal currents are stronger.

(b) Mud accumulations along subtidal margins of channels.

(c) Mud accumulations along channel intertidal fringes. These typically become thinner in the lower reaches where tidal currents are stronger, exposure to wind is greater, and distance from source of mud is greater.

(d) Sandy intertidal flat, typically covered by wave-induced ripples and littered with shells and shell fragments. Broadest in the lower reaches, which are closer to source of marine sands. May be exposed to ocean swells that penetrate through the estuary mouth.

(e) Wave-built accumulations of sand/shell on the fringes of the upper intertidal flats. These may be thicker and more numerous in the lower reaches, but occurrence throughout an estuary will vary greatly with local exposure and sediment supply.

(f) Mud accumulations on fringes of upper intertidal flats. These are typically stabilised and promoted by fringing vegetation. Typically more extensive in the upper reaches, but, again, occurrence throughout estuary will vary greatly with local exposure and sediment supply.

(g) Mud impoundment behind a wave-built barrier.

4 Sources of Chemical Contaminants

Recently, there has been a considerable amount of effort directed at determining the key sources of contaminants in urban stormwater in Auckland. Contaminant reduction or elimination at source ("source control") is the most effective method for reducing contaminant loads and impacts on downstream receiving environments. Once sources are identified and quantified, the benefits of source control measures can be objectively predicted and prioritised.

For established urban catchments, where retrofitting effective stormwater treatment can be impractical, source control is the key to minimising on-going impacts associated with contaminated stormwater. It also has greater potential for sustainable reductions in contaminant loads than conventional treatment methods, which can slow down, but cannot halt, build-up of contaminants in marine receiving environments (see Chapter 7). It also offers the greatest possible effectiveness for reducing contaminants present in dissolved forms, which are difficult to remove by commonly used treatment methods.

There is a wide range of potential sources of contaminants in urban stormwater, as summarised in Table 4 below.

Table 4
Key sources of urban stormwater contaminants (from ARC 1992a, Kennedy 2003a).

Source	Comment on source
Motor vehicles	Vehicle exhaust emissions, oil and lubricating losses, wear of vehicle body, tyre and brake wear.
Road surface	Wear and breakdown of road surface including aggregate and other material used in base course and surface, bitumen, concrete and other materials used on road surface (eg marking paint).
Litter/organic debris	Wide variety of litter and debris of inorganic origin (metal, glass, plastic) and organic (food, paper, plant, litter and animal waste) origin.
Buildings	Corrosion and other weathering products from adjacent buildings including dry and wet losses. Includes losses such as Cu from Cu facings and gutters, roof run-off (paint fragments etc.,).
Construction	Building maintenance and construction produces a wide variety of inorganic dust and other debris (eg paint waste).
Soil	Wind and rain will transport fines in soils from adjacent areas onto roadways. Materials are typically silts, clays and sands with low organic content.
Rainfall	Atmospheric washout will contribute fine particulate material present in urban air. Sources will be site specific. A portion will be local and vehicle derived.

A wide range of chemical contaminants is potentially present in urban stormwater, originating from both present-day and historical uses. Table 3.2 (ARC 2005e) lists common contaminants and their sources.

Table 5

Contaminants and their primary sources in urban stormwater (ARC 2005e).

Chemical	Primary sources
Zinc	Vehicle tyres, galvanised building materials (eg roofs), paints, industrial activities
Copper	Vehicle brake pads, plumbing, industrial activities
Lead	Residues from historic paint and petrol, industrial activities
Polycyclic aromatic hydrocarbons	Domestic fires, industrial emissions, vehicle exhaust, vehicle lubricating oil.
Hydrocarbons	Vehicle lubricating oil, vehicle exhausts, fuel spills
Plasticisers	Building materials, plumbing
Herbicides, pesticides	Residential and council (mostly historic) use
Fungicides	Paints, residential use

From Tables 4 and 5, the most important potential contaminant sources are:

- Vehicle emissions.
- Road run-off.
- Buildings – mainly roof run-off, but also residues associated with building materials such as paints and plumbing.
- Catchment soils, containing chemicals (eg metals) naturally present ("background" levels) as well as residues associated with historical uses (eg organochlorines, Pb).

The quantities and relative proportions of contaminants associated with these sources in Auckland is the focus of this chapter. Sources of other key "non-chemical" contaminants, such as sediment and pathogens, are reasonably well understood and are discussed in Chapters 6 and 7.

What was known in 1995

In 1995, loads and concentrations of potential contaminants in stormwater had been measured, based largely on extensive monitoring conducted in the Wairua, Pakuranga and Southdown catchments and from overseas literature (ARC 1992a and 1994c; Williamson 1993). There was no information, either locally or from overseas literature, which could be used to attribute the relative importance of different sources. The contribution that vehicle use makes to urban stormwater contaminant loads was the subject of professional judgment (NIWA 1994a and 1996a; MoT 1996). The estimates ranged from 40 per cent for Zn to 100 per cent for Pb.

Major advances have been made since 1995, as described below.

4.1 Vehicle emissions

A substantial body of work aimed at identifying and quantifying the vast number of inorganic and organic contaminants present in vehicular components and emissions, and predicting the contaminant loadings associated with transport sources, has been carried out on behalf of the Ministry of Transport[4].

Contaminants in vehicular sources such as fuels, brake linings, tires, exhaust emissions, oil leaks, and roading materials have been comprehensively characterised, and the international literature has been critically reviewed and complemented by analyses on brake pads, tyres and road bitumen used in New Zealand (Kennedy & Gadd 2003a–c).

Emission factors from vehicles have been estimated from each of the potential contributing vehicle components and a model to predict the contaminant loads from vehicle emission available for run-off to waterways, the "Vehicle Fleet Emissions Model for Water" – VFEM-W – prepared (Kennedy et al., 2002). The data are very complex, and have considerable uncertainty associated with them. This information will become increasingly useful as source identification becomes more refined and if additional contaminants of concern are identified in the future.

Initial quantitative predictions of the contribution of vehicle emissions to potential stormwater contaminant loads have been made in Waitakere City, and the possible links and effects have been reviewed (Kennedy 2003a, Kennedy & Gadd 2003c and d). These assessments indicate that:

- >95 per cent of Cu on the road surface is sourced from brake wear.
- The major source of Zn is tyre wear.
- The amount of Pb found on the road is much larger than is predicted from vehicle emission data.
- PAHs are sourced largely (very approximately 70 per cent) from exhaust emissions, and to a lesser, but still significant, extent (ca. 30 per cent) from tyre wear. Predicted loads were much higher than those measured.
- Improved estimates of tyre and brake composition and wear are required to improve predictions of Cu, Zn, and PAH loadings, while a better understanding of off-road sources of Pb and emissions from Pb-free fuels are required to improve predictions of Pb loads from roads.

Emission factors have also been estimated by back-calculation from measured road run-off loads, which integrate the multiple individual contaminant sources and transport processes associated with stormwater run-off in the road corridor (Timperley et al. 2003).

[4] A compilation of reports is available online at www.transport.govt.nz/stormwater.

Emission factors of common stormwater contaminants measured or estimated for Auckland are reproduced in Table 6. Loads of key contaminants predicted from the two most recent studies (Timperley et al. 2003, Kennedy & Gadd 2003c) generally agree within a factor of 3.

Table 6

Vehicle emission rates for Cu, Pb, Zn and PAH (mg per km of vehicle travel; Kennedy & Gadd 2003c).

Study	Copper	Lead	Zinc	PAH
Timperley et al. (2003)	0.06	0.05	0.45	-
NIWA (1996)	0.16	-	0.7	-
Kennedy & Gadd (2003c)	0.16	0.11	0.18	0.014

4.2 Roads

Roads contribute contaminants to urban stormwater from vehicle deposits (tyre wear, oil leaks, brake wear etc) and also from abrasion and dissolution of road materials (Table 4). Analysis of roading materials, dust from the road surface, and from gutters provides information on the nature of the particulate material produced on, or deposited onto, roads and is *potentially* available to be transported by stormwater run-off into aquatic receiving environments.

However, only a portion of the road- or vehicle-sourced material is transported from the road into receiving waters during storm run-off. Volatile components can be lost by evaporation to the atmosphere, and some organic contaminants may be degraded or transformed by sunlight and microbes, thus limiting their contribution to stormwater contamination[5]. Dense particulates are less likely to be carried into stormwater and are more effectively captured by commonly used stormwater treatment devices such as catchpits and ponds. The water-soluble chemicals and fine particulates are more effectively mobilised by rainfall and are less effectively captured than heavier, coarse, materials, and are therefore more likely to be representative of the contaminants that may actually reach receiving environments. However, fine particulates are also more susceptible to “non-wash off” removal processes; eg removal from the road surface by wind or traffic-induced turbulence.

Analysis of the fine fraction of road and gutter dust provides data on the upper limits of contaminant loads that could eventually be transported into waterways. Analysis of vehicular components and road materials (Kennedy 2003b; Kennedy & Gadd 2003a–d; Kennedy et al. 2002; Depree & Ahrens 2005; Ahrens & Depree 2006) enables an assessment of the range of contaminants that could be an issue to be identified and provides some insight into where these contaminants might be coming from.

[5] A comprehensive assessment of transport-related contaminants, their properties and concentrations, and stormwater issues is provided by reports commissioned by MoT (www.transport.govt.nz/stormwater), in particular Kennedy (2003a).

Analysis of water and suspended sediments in road run-off itself provides better data on the likely impacts of roads and vehicles on receiving waters than analysis of road and gutter dust. This approach has been used to assess the contribution of roads and vehicular sources to stormwater contaminant loads in Auckland City, as described in Sections 4.2.2 and 4.4 (NIWA 2005).

4.2.1 Road dust

There have been a number of investigations of urban gutter dust quality in Auckland. Prior to 1995, one study (ARC 1992a) presented a comprehensive assessment of particle size, PAH, and heavy metals in road gutter dust samples from Pakuranga (low- and high-density traffic residential) and Southdown (industrial) catchments. This data, together with overseas data, showed that road dust can be highly contaminated, with Zn, Pb, and Cu present at high concentrations and a range of organic contaminants (including petroleum hydrocarbons, PAH, PCBs, and DDTs) present at lower, but still elevated, levels.

Since 1995, a wider range of elements have been analysed in road dusts in Auckland urban areas (Ng et al. 2003, Kennedy & Gadd 2003c). Ng et al. (2003) undertook a comprehensive survey of road surface material on streets in Remuera, Hillsborough and Mt Roskill. Kennedy & Gadd (2003c) reported data ranges for samples of material systematically collected from road surfaces in Waitakere City. Data from these studies, and some comparisons with overseas data, are summarised in Kennedy (2003a).

The composition of the particulate matter found on roads or in urban street gutters is dependent on the method of sampling (eg sweeping, vacuuming), and the location of the sample site and its surroundings, but in general can be summarised as follows (Kennedy 2003a).

Particulate composition

Inorganic matter makes up a substantial, but variable, proportion (15–60 per cent) of the material present. Fragments of rubber, and metals (including rust and other ferromagnetic material) are common, as is litter (such as paper, glass, plastic and other debris). Organic debris from plants and lawn clippings may also be present but the amount tends to be seasonal and varies between areas depending upon land use (eg residential streets, motorway grass verges, median strips etc).

Road and gutter dust is generally fairly coarse, with much of it being mostly sand-sized or larger (approximately >80–90 per cent is >0.1 mm). Gravel roads, or those receiving large amounts of vegetative debris may have a substantial proportion of gravel-sized particles; eg some Waitakere City roads had >40 per cent of the road dust particulates in the >2 mm fraction (Kennedy & Gadd 2003c).

The proportion of mud-sized (<63 µm) material is generally relatively small, <10–20 per cent (Kennedy 2003a, Kennedy & Gadd 2003c). The size of particles produced by motor vehicle emissions is small, but they can aggregate to become larger size particles, or can attach to larger particles on the road or in gutters. Loss of fines by wind and wash-off processes is a possible contributing factor.

The low proportion of fines in road dust contrasts with stormwater, which is dominated by fine particles (40–70 per cent <60 m and essentially all <0.3 mm; NIWA 2005), indicating that much of the coarse road dust is not transported through the stormwater system into receiving waters (presumably because it is not as efficiently mobilised and/or is captured en-route – eg most of the >0.5 mm fraction is captured in roadside catch-pits – NIWA 2005).

Contaminants

The concentrations of contaminants in road dust depend on range of factors such as traffic densities, position on the roadway, surrounding land use, analytical methods, and the particle size range that is analysed, so it is difficult to reliably summarise "typical" concentrations. Kennedy (2003a and b) and Kennedy & Gadd (2003c) provide comprehensive reviews of NZ and overseas data.

Metals

Data from the few studies that have been conducted in Auckland indicate that metal concentrations vary over a wide range. Copper (Cu), lead (Pb) and zinc (Zn) are usually the metals that are most elevated over background. Cadmium, (Cd), Fe, Cr, Ni, Sb, Pt and Mo are also often present in higher concentrations than background (Kennedy 2003b).

Copper concentrations may average about 150–400 mg/kg (range approx. 50–1600 mg/kg) and Zn about 300–1500 mg/kg (range 130–2700 mg/kg). Lead concentrations before the removal of Pb from petrol in 1996 were very high, averaging about 1600 mg/kg (ranging up to 10,600 mg/kg). More recent data (Kennedy 2003a&b, Kennedy & Gadd 2003c) suggest typical concentrations may be about 200–500 mg/kg (range 20–2000 mg/kg).

These concentrations are often well above sediment quality guidelines for the protection of aquatic life (see Chapter 5), so discharge of large quantities of road dust directly into aquatic receiving environments could eventually lead to toxic effects on aquatic life[6].

PAH

Total PAH concentrations in gutter dust from Pakuranga and Southdown catchments of 0.96 and 0.77 mg/kg respectively were reported by ARC (1992a), with higher concentrations present in the fine particle size fractions (eg 2.5 mg/kg in the <47 m fraction in Pakuranga). Waitakere City road dusts contained a median total PAH in the <2 mm fraction of approximately 1.5 mg/kg (Kennedy & Gadd 2003c).

These concentrations are similar to, or higher than, those commonly found in the sediments in Auckland's muddy urban estuaries (see Chapter 7), suggesting that roads could contribute significantly to PAH contamination in receiving environments.

The potential significance of road dust and roading materials as sources of PAH has been further investigated by NIWA researchers, who have studied the concentrations,

[6] Accumulation of contaminants in estuarine receiving environments is discussed further in Chapter 6.

composition, and distribution of PAH in roading materials, gutter dusts, urban stream sediments, and estuary sediments in Auckland and Christchurch (Depree 2003; Depree & Ahrens 2003; Ahrens & Depree 2004; Depree & Ahrens 2005; Ahrens & Depree 2006).

The sediments of Motions Creek and its downstream estuary, which has some of the highest PAH concentrations in Auckland estuaries, was studied to evaluate the source and bioavailability of PAH. PAH concentrations in the stream and estuary sediments were higher than those present in motorway road dust (see Table 6), and hence the source of the PAH was unclear.

The highest PAH concentrations in the Motions Creek stream sediments were found in the coarse particulate fraction (1–2 mm). These particles were coated with black material, which was found to be coal tar, a by-product of gas production from coal that was historically used as a road binding material.

Figure 13
Tar-coated particle in Motions Creek sediment (Depree 2003).

It was concluded that the high concentrations of PAH found in Motions Creek receiving water sediments may have been the result of historical inputs to the estuary from erosion of old roading and pavement material or, as has been shown in Christchurch, from present-day wear and/or erosion of older road and pavements (Depree & Ahrens 2005).

Table 7 (below) lists concentrations of PAH found in these studies compared with other possible sources, and shows the large differences between possible sources. The difference between coal tars and modern bitumen are dramatic: coal tars contain about 150,000 mg/kg total PAH, while modern bitumen contains about 30 mg/kg (Depree & Ahrens 2005) or less (<0.2–2.2 mg/kg; Kennedy & Gadd 2003a). Also, the chemical "signature" of the modern day bitumen suggests petrogenic origins (ie sourced from petroleum oils), while the coal tar "signature" is pyrogenic (sourced from combustion).

Table 7

Concentrations of total PAH (mg/kg) in source materials, Motions Creek sediments and road matrix binders in Christchurch road material (from Depree and Ahrens 2003 and 2005).

Source	Total PAH (mg/kg)	Source	Total PAH (mg/kg)
Tyre	21–95	Road dust (Auckland NW Motorway)	3–4
Bitumen	11–19	Atmospheric fallout (dust)	12–27
Used oil	52	Motions stormwater pipe sediments	3–7
Charcoal	23	Motions Estuary sediments	12–17

Coal	5		Motions Stream sediments	23–26
Pitch	72200		Motions – particulate organic matter	954
Christchurch roading			Motions – plant detritus	66
• Old roads, coal tar binder	1200–140000			
• New road, bitumen binder	30–280			

More recent work in Auckland (Ahrens & Depree 2006) shows that while the use of coal tar was phased out over 40 years ago, coal-tar contaminated materials can still be found at the surface of some pavements, and is therefore available for loss to urban waterways. Because the concentrations of PAH in coal tar are so high, the presence of relatively small amounts of historically contaminated pavement materials could greatly affect the concentrations of PAH in receiving water sediments. The significance of this PAH source in Auckland is currently being investigated further for the ARC (M. Ahrens, NIWA, pers. comm.)

Overall, the available information of PAH concentrations in road-related materials indicates that roads are a significant contributor to PAHs in receiving waters. Road dusts from modern roads and pavements appear to contain total PAH concentrations of a few mg/kg. This is above typical "threshold effects" ecosystem protection guideline levels, but is well below the "probable effects" level where ecological effects become more frequent (see Chapter 5). Older roads and pavements may have higher PAH concentrations if coal tars were historically used as binders. Streams and downstream estuaries in these areas may be highly contaminated with PAH as a result of losses from these roading materials. Whether this contamination results in toxicity to aquatic life depends on the PAH bioavailability. Preliminary assessments suggest bioavailability is relatively low in Auckland's estuaries (Ahrens et al. 2005). This is currently the subject of further investigations.

Other chemicals

Gutter dusts from Southdown and Pakuranga catchments were found to contain elevated concentrations of organochlorines, with higher concentrations present in the finer particle size fractions (ARC 1992a):

- PCBs at 64 ng/g (parts per billion) and 28 ng/g in gutter dusts from Southdown and Pakuranga catchments respectively.
- DDTs at concentrations of several parts per billion, ranging from 6–36 ng/g.
- Dieldrin, at 4–45 ng/g.
- Chlordane, at 4–8 ng/g.
- Lower concentrations of lindane and heptachlor (generally <1 ng/g).

Organochlorine pesticides are not sourced from vehicles, so the presence of these contaminants in gutter dusts reflects transport and deposition of off-road material such as local soils and vegetative matter.

The concentrations of the organochlorines is quite high, generally above aquatic ecosystem protection guidelines, and therefore road dust represents a potentially significant source of contamination to receiving waters.

Long-chain alkane hydrocarbons (>C29 compounds) and various natural alkanoic acids and hydrocarbons are found in road dust, but their generally low toxicity means that they are unlikely to cause adverse effects in receiving waters.

A wide range of other urban-derived organic contaminants (eg phthalates, polybrominated diphenyl ethers (PBDE), chlorinated dioxins and furans), which are of concern overseas, is almost certainly present in road dusts, but no quantitative data for Auckland were found during this review.

4.2.2 Road run-off

Road run-off contributes a significant volume of stormwater and mass of contaminants to receiving waters. The composition of road run-off and the contribution it makes to the total contaminant load from a catchment will depend on a number of factors, including the traffic volumes, catchment land use (proportions of commercial, residential, industrial etc), age and condition of roads and buildings, road maintenance regime (eg gutter sweeping) and the types, condition, and numbers of run-off treatment devices (eg catchpits, swales etc).

Recent studies, summarised in ARC (2005e), have attempted to quantify the contribution of major contaminant sources, including road run-off, to total stormwater loads of the key three metals Cu, Pb, and Zn in three study catchments[7] with different land uses.

This work, which is described more fully later (Section 4.4), found that road run-off contributed:

- only a minor fraction of the Zn load from all three catchments, from <1 per cent in the industrial catchment to the maximum of 15 per cent for the residential Mission Bay catchment;
- a largely uncertain proportion of the Cu load. Mass balances were incomplete for Cu, with most of the catchment loads being unaccounted for by the sources investigated (roads, roofs, building walls, soil). However, road run-off contributed significantly to the total identified source load: 24 per cent in the industrial catchment, 46 per cent in the residential, and 79 per cent in the commercial CBD catchment. However, these proportions could change greatly when the "missing" Cu is "discovered" through further work on Cu sources; and
- an uncertain proportion of the Pb, especially in the CBD and industrial catchments, where most of the Pb was unable to be accounted for in the sources evaluated. Of

[7] A residential catchment – Mission Bay, an industrial catchment – Mt Wellington, and a commercial catchment – the Central Business District (CBD).

the *identified* sources, road run-off contributed about 18 per cent in the industrial catchment, 20 per cent in the residential, and 58 per cent in the CBD. As was found for Cu, the Pb loads are quite small, and therefore these proportions could change markedly when loads are refined by addition of new information.

4.3 Roof run-off

Roof areas can account for a large proportion of an urban catchment's impervious area, and can therefore contribute a large amount of stormwater run-off. Roofs can be equivalent to, or greater than, that of trafficked road surfaces and typically can make up about 40 per cent of the total impervious area, and 20 per cent of the total catchment area.

Studies of potable drinking water collected in rainwater tanks (Simmons et al. 2001), found high levels of Zn if the roof was galvanized iron in poor condition. Gadd & Kennedy (2001) reviewed NZ and overseas studies, challenging the perception that roof run-off is clean and proposing that it could be a major source of stormwater contaminants.

There have been historical changes in the predominant materials used on the roofs of New Zealand buildings that are relevant to roof run-off quality. Traditionally used long-run galvanised iron has been largely replaced by Zincalume coated iron. There has also been a shift from in situ application of roofing paints to factory painted steel products. There have been considerable advances in coating technology, with increased durability and decreased breakdown of roof coatings and materials. Accompanying the decrease in the use of long run galvanised iron has been a corresponding increase in the use of steel tiles coated with paint or grit, or concrete tiles. Consequently, there has been a shift from galvanised roofs of highly variable surface quality, and from grit-coated steel tiles of poor durability, to roofing materials with improved durability. These changes in materials can make a significant difference to the quality of roof run-off, and hence its contribution to stormwater contaminant loads.

The ARC commissioned a comprehensive study to examine roof run-off quality in the Auckland region (ARC 2003h). Artificial roofs were used to examine the effects of land use on run-off quality.

Figure 14
Artificial roof setup (ARC 2003h).

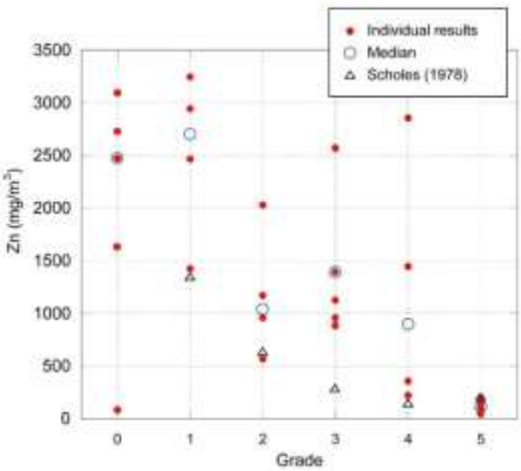
Run-off from roofs in rural areas was collected to assess effects of roof material and condition. Suspended sediments, pH, metals and PAH were analysed. First flush, dissolved, and particulate phases were examined, along with roof paint composition on roofs examined in this study and in a selection of available products.

The study sought to cover as wide a variety of situations as possible and sampling had minimal replication.

Many of the new roofing materials were found to contribute little contamination and were relatively un-reactive. Older galvanised iron roofs and unpainted galvanised Zincalume roofs contributed significant concentrations of Zn as a result of oxidative wear and acidic corrosion.

Examination of the run-off quality from galvanised roofs showed an increase in the concentration of Zn once paint had deteriorated (Figure 15). Sometimes high concentrations could be found with relatively minor deterioration. Figure 15 shows that concentrations of Zn in roof run-off are highly variable and that other factors besides roof condition can affect Zn concentrations.

Figure 15
Zinc in run-off collected from galvanised iron (GI) roofs (KMA & DSL 2003). Grade 0 are unpainted GI roofs; 1–5 are painted in different conditions; 5 = best, 1 = worst. Data are plotted with those from an earlier study by Scholes (1997) who found an approximately linear relationship between condition and Zn concentration.



Most of the Zn in the roof run-off was in the dissolved phase. This has implications for treatment of roof run-off because dissolved metals are more difficult to remove than particulates. Conventional treatment is likely to be less effective than source control for reducing metal loads from catchments with a large proportion of roof run-off.

There was a strong "first flush" effect, with much of the metals being washed off with the first part of the rain event. The first flush concentrations from galvanised roofs increase with the degree of weathering of the paint. This again has implications for treatment – managing the first flush, rather than the total volume of roof run-off, is likely to be an efficient way of reducing metal loads.

Lead concentrations from galvanised roofs were elevated but were poorly correlated to degree of weathering, possibly due to the amount and condition of lead-headed nails and lead flashing.

There was a modest but important land use effect on the water quality of run-off. Local industrial activities appear to contaminate roof run-off with Zn, Cu and Pb. There was a surprisingly small effect of motorway land use, but this may reflect the limited sampling undertaken in this study.

Examination of paints used on roof surfaces has also shown that there have been significant improvements in the composition of paints in relation to potential contribution of metals to stormwater run-off, especially with respect to Pb. Zn concentrations are still high in some modern primers and paints, however.

A survey of roof types within older small sub-catchments within Auckland City (parts of the CBD, Westmere, Pt. Chevalier, Epsom, and Mt. Roskill) found that galvanised iron formed a large proportion of roofs. If these roofs exhibit the same degree of contamination found in run-off from rural roofs, then it was concluded that roofs contribute the major proportion of the stormwater Zn load in older catchments.

4.4 Budgets

In 2004 there were sufficient studies of catchment, roof, and road loads to enable a first attempt to identify and quantify the metal sources in three urban catchments. A study commissioned by the ARC combined the data from three studies to produce mass budgets for zinc, copper and lead in the stormwater from the three urban catchments and thereby to determine the contributions of each known metal source to metal loads in urban stormwater (ARC 2005e).

Zinc

For all three catchments the mass budgets obtained for Zn are probably as complete as it is possible to achieve using the data presently available.

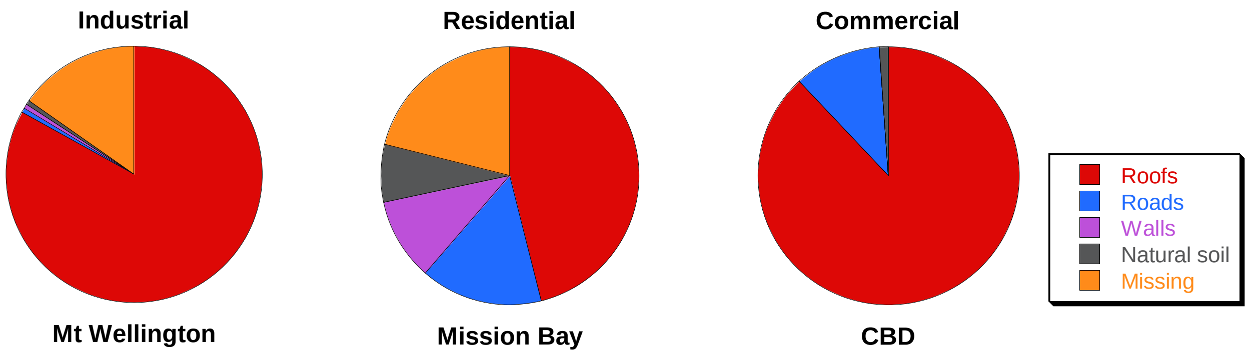
In the commercial (Central Business District) and industrial (Mt Wellington) catchments, roof run-off could account for almost all the Zn in the catchment stormwater. Galvanised iron roofing contributed the major part of the roof run-off Zn load.

For the residential catchment (Mission Bay) roof run-off contributed about 45 per cent of the catchment Zn load. The Zn budget for this catchment has about 20 per cent of the catchment load unexplained but this could be within the error in the load estimates.

As described above, road run-off is a relatively small source of Zn in all catchments.

Figure 16

Mass budgets for Zn in three Auckland catchments (ARC 2005e).

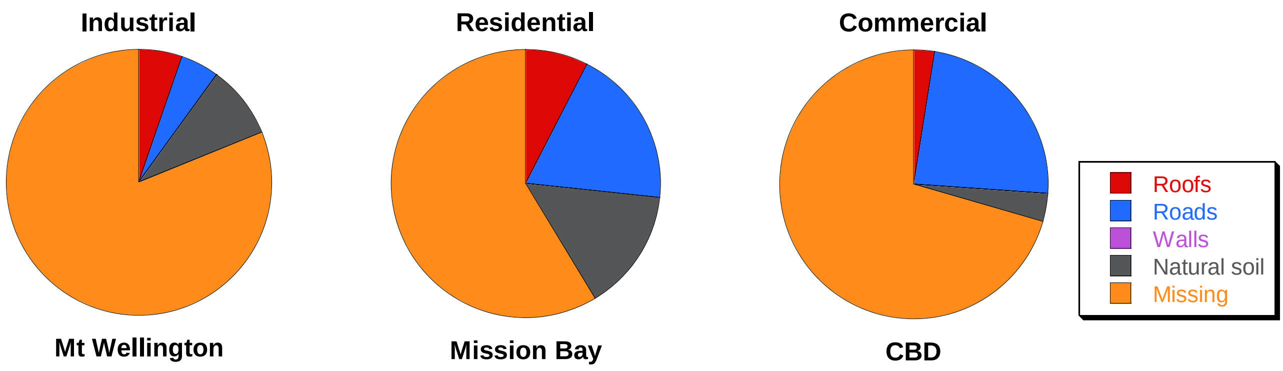


Copper

The mass budgets for Cu for all three catchments are far from complete, with Cu loads from unidentified sources amounting to about 60 per cent of the total catchment load for the residential catchment, 70 per cent for the commercial catchment and 80 per cent for the industrial catchment. The mass budget does not include run-off from building walls and fittings that drain directly onto impervious surfaces then into the stormwater network. However, the data obtained for soils adjacent to buildings in the residential (Mission Bay) and industrial (Mt Wellington) catchments show that run-off from building walls in these catchments does not carry large amounts of Cu. The situation in the commercial (CBD) catchment could be different, however, where possibly more copper is used on buildings. It is also possible that industrial activities deliver Cu (and also Pb) onto impervious surfaces and thereby provide another source of Cu (and Pb) that is not accounted for explicitly in the mass budget. However, it is believed that the largest portion of missing Cu is due to underestimating the Cu load from roads (most probably from brake linings).

Figure 17

Mass budgets for Cu in three Auckland catchments (ARC 2005e).

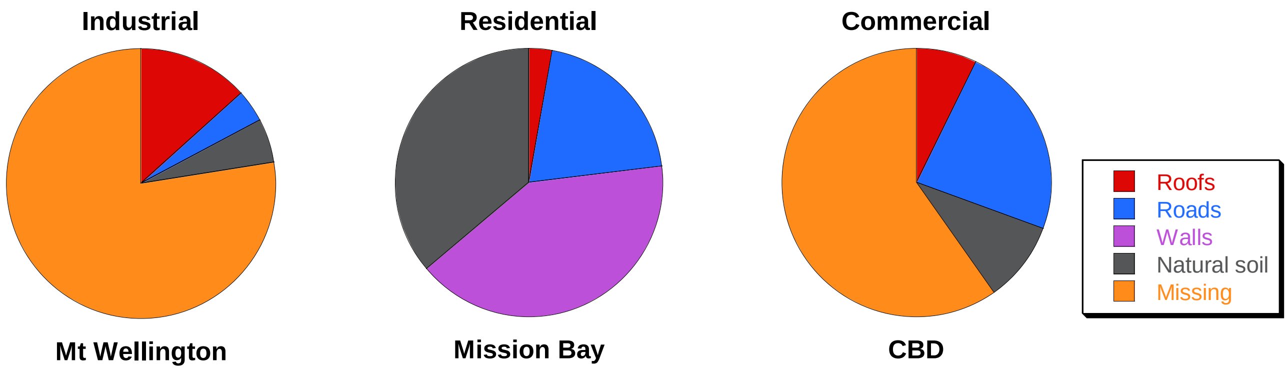


Lead

The mass budgets for Pb in the commercial and industrial catchments are, like those for Cu in these catchments, incomplete. Run-off from building walls and flashings could, however, be the source of some or all of the missing Pb. This would appear to be the case in the residential catchment. A realistic assumed average concentration of 500 mg/kg for Pb in 10 per cent of catchment soils (the "near building" soils) completed the lead mass budget for the residential (Mission Bay) catchment. Overseas, the dominant source of Pb is believed to be topsoils besides roads. The missing Pb could be due to the logistical difficulty in accurately measuring the amount of stored Pb in all catchment soils because of the large sampling effort required.

Figure 18

Mass budgets for Pb in three Auckland catchments (ARC 2005e).



4.5 Summary

Key sources of stormwater contaminants in Auckland have been identified, and attempts to quantify their contributions to catchment loads have been made, using a combination of detailed stormwater monitoring and predictive modelling.

This identification of aging galvanised roofing as a major source of a key urban contaminant, Zn, was a major advance. This discovery, coupled with modelling tools developed in the last couple of years, means that options for Zn management are better understood.

Prediction of contaminant loads from other key stormwater sources remains a challenge. Refinements are required to improve their reliability, especially for other important contaminants such as Cu, Pb, and PAH. This is currently under study.

5 Guidelines for Stormwater Impact Assessment

Determining the effects of stormwater on ecological and human health in receiving waters is a difficult task. Stormwater contains a complex mixture of potentially toxic chemicals (eg heavy metals, hydrocarbons, pesticides, ammonia), pathogenic microorganisms, sediment, nutrients, and organic matter, all of which have the potential to influence environmental effects. Adding to the complexity is the highly variable nature of stormwater composition (both spatially and over time) and the wide range of receiving environments into which stormwater is discharged.

Ideally, evidence for adverse effects on biological receptors (eg aquatic animals, plants, humans) is obtained from a direct assessment of the health of exposed populations at the sites of interest. However, this is usually a major undertaking and, unless contamination levels are very high (a rare occurrence in New Zealand), impacts can be difficult to distinguish from the effects of other environmental variables (eg salinity gradients and sediment textural variations in estuaries, habitat and flow variations in streams etc)[8].

An alternative way to assess potential environmental impacts is to compare the concentrations of key contaminants in the receiving environment with guidelines that aim to protect human and ecological health.

Using guidelines as impact assessment tools is much easier, and far less expensive, than conducting ecological or human health studies, but guidelines may not take into account site-specific factors that could affect the nature and scale of impacts[9]. Guidelines are therefore generally used as "first cut" screening tools to provide indicative, rather than absolute, evidence for adverse effects. Exceedance of guidelines indicates that there is *potential* for an environmental impact and that further investigations are required to determine with greater certainty whether effects are actually occurring (or not). Guidelines can also be used to assess the relative importance of contaminants present in mixtures (eg stormwater), which can help prioritise source control and mitigation strategies[10].

Most of the guidelines currently available have been developed from overseas studies that relate the health of exposed biological receptors to concentrations of environmental contaminants. The guidelines generally define contaminant concentrations ("trigger levels") above which adverse effects might be expected to occur. The exact definition of the kind of effect or the probability of an effect occurring

[8] Progress made on assessing ecological effects are described in Chapter 5 (freshwaters) and 6 (marine).

[9] The ANZECC (2000) guidelines provide comprehensive coverage of how receiving environment characteristics may affect the applicability of guidelines.

[10] As described in Chapters 5 and 6, this approach has been used to identify zinc (Zn) as a priority contaminant in Auckland's receiving waters.

at these trigger levels varies between guidelines, as do the names assigned to trigger levels in the various guideline documents.

The following sections summarise the guidelines that have been used for assessing the potential effects of stormwater on receiving waters, sediments, and human health in Auckland, and the progress that has been made towards improving these guidelines since 1995.

What was known in 1995

In 1995, judgement about the potential effects of stormwater contaminants in receiving waters was made using a range of (largely) overseas guidelines. There were few guidelines derived specifically for New Zealand, notable exceptions being for colour and clarity (MFE 1994), undesirable biological growths in streams (MFE 1992), and microbiological indicators for contact recreation and shellfish gathering waters (McBride et al. 1992).

Chemical contaminant[11] concentrations in water were generally assessed by comparison with North American or Canadian water quality guidelines or criteria (eg USEPA 1986; CCREM 1991), or the ANZECC (1992) guidelines, which were largely based on US and Canadian data (although NZ studies, including colour, clarity, and microbiological, were also incorporated).

The biological impact of contaminated sediments was the subject of considerable attention in the international scientific literature, and in the early 1990s guidelines for chemical contaminants in sediments were proposed. The most notable of these were the guidelines developed by the US National Oceanic and Atmospheric Administration (NOAA; Long & Morgan 1990), and later refined for marine and estuarine sediments by Long et al. (1995), which formed the basis for a number of similar sets of guidelines that were subsequently developed and widely used[12]. These guidelines were derived from the relationship between adverse biological effects (eg toxicity) on benthic animals and contaminant concentrations in sediments from a large database of studies conducted around the USA. Guideline values for nine metals, 13 individual polycyclic aromatic hydrocarbons (PAH), three classes of PAH, and three classes of organochlorines (including DDTs, dieldrin, and PCBs) were derived.

Guidelines for a limited suite of organic contaminants[13] were also developed by a different approach – the “Equilibrium Partitioning – EqP” method – by the USEPA (USEPA 1993a). These have not been as widely used as the NOAA guidelines, partly because they cover a more limited set of organic contaminants. ARC (1994a) gave a comprehensive discussion of the various guidelines, their derivation, and application in Auckland estuaries.

[11] Key chemical contaminants often included in stormwater studies were heavy metals (e.g. Cu, Pb, Zn), organic contaminants (e.g. PAH, organochlorines such as DDTs and PCBs), and ammonia.

[12] The sediment quality guidelines adopted by ANZECC (2000) were largely those proposed by Long & Morgan.

[13] The organochlorine pesticide dieldrin, and the polycyclic aromatic hydrocarbons (PAH) acenaphthene, fluoranthene, and phenanthrene. (USEPA 1993b)

Limits for the safe consumption of fish and shellfish were available in the NZ Food Regulations for a very limited range of chemical contaminants[14] (DoH 1984). The relevance of other chemical contaminants for human consumption of shellfish was occasionally assessed by comparison with USA or Australian residue limits (eg the Tamaki Estuary 1988–91 report, ARC 1992b). Microbiological criteria for food (MoH 1995) provided limits for pathogenic indicator organisms (eg faecal coliforms) in fish and shellfish.

Application of all the above guidelines enabled the potential environmental impacts of stormwater-derived contamination in receiving waters and sediments to be assessed, gave an indication of the relative importance of various contaminants present in stormwater and receiving waters and sediments, and provided information on the degree of contaminant reduction (eg by dilution) that would be required to reduce concentrations to below "adverse effect" levels.

As described in Chapters 5 and 6, these assessments identified a range of contaminants as being of particular concern. Pathogenic microorganisms in stormwater, near-shore receiving waters, and shellfish, and zinc (Zn), copper (Cu), and lead (Pb) in receiving water sediments, were identified as being sufficiently elevated above guideline levels to be of concern in Auckland's urban streams and estuaries. Organochlorines (DDTs, PCBs, dieldrin) and PAH were also commonly present at elevated concentrations in some shellfish and sediments, but were usually below guideline concentrations. They were therefore considered to be of secondary importance.

Advances since 1995

Significant advances have made since 1995 in several areas, which have resulted in the publication of the following key guideline documents:

- The ANZECC (2000) guidelines, covering water and sediment quality for a wide range of chemical contaminants and physical parameters. Comprehensive coverage of guideline derivation, contaminant fate and effects, and application is included. A notable feature for stormwater impact assessment is a "decision tree" scheme for determining whether sediments are likely to be toxic to benthic fauna. The ANZECC guidelines have become the primary reference for assessing receiving environment quality in New Zealand (with the exception of microbiological quality, as described below).
- Microbiological guidelines for recreation and shellfish gathering in marine and fresh waters (MFE 2003). These guidelines present a complete protocol for determining the suitability of waters for recreational use, including a grading system based on catchment characteristics and receiving water quality.
- Guidelines for assessing the effects of stream flow on habitat and ecology (Elliot et al. 2004), and for fish passage (ARC 2000a).

Despite what was known in 1995 and these advances, it is currently still difficult to decide which guidelines to use and how to use them in all situations. These difficulties

[14] Upper limit concentrations were specified for arsenic (As), copper (Cu), mercury (Hg), and dieldrin.

led to a review of guidelines and their use for marine waters in Auckland. As part of the Regional Discharge Project (RDP), an analysis of Auckland receiving water sediment contamination levels and a review of guidelines available at that time were conducted to recommend the most appropriate sediment quality guidelines (SQG) for Auckland's estuaries (DSL 2001a).

Integration of all the above guidelines was then undertaken to devise a set of protocols for monitoring Auckland's marine receiving environments for the RDP stormwater management process – the result was the Blueprint for Monitoring Urban Receiving Environments document (ARC 2004c). The guidelines given in the "blueprint" were based on a combination of ANZECC (2000)[15], MFE (2003), Canadian guidelines (CCME 1999) and guidelines derived in Florida by MacDonald et al. (1996).

The guidelines given in the ARC "blueprint" are called Environmental Response Criteria (ERC), which define three grades of contamination: "green", "amber", or "red". The grades were judged to be conservative (ie protective). This approach was adopted because concentrations in the receiving environment were perceived to be generally increasing and an early management intervention approach seemed prudent. Concentrations in the green zone pose a low risk to environmental health, amber zone concentrations indicate that concentrations had increased to levels where biological impacts were possibly beginning, while concentrations in the red zone indicated that the ecology was likely to be impacted.

The relationship between the ERC guidelines and the health of Auckland's estuarine benthic ecology has been the subject of a considerable amount of recent study, to assess the most appropriate methodology for classifying sites in terms of their biological health and contaminant levels. The recommended methods are detailed in ARC (2006c).

There has not yet been a formal rationalisation or review of guidelines and monitoring procedures for the freshwater receiving environment (as done for marine environments in the "blueprint"), although a framework for assessing stream condition and management strategies has been produced (ARC 2004a). Monitoring can be more complex in streams than in estuaries because of the more variable physical environment (eg flows, sediment textures, effects of short-term rainfall events etc).

Impacts of stormwater contaminants in freshwaters are usually assessed using:

- ANZECC (2000) for water quality (chemical contaminants, nutrients etc);
- MFE (2003) for microbiological contamination effects; and
- for sediment quality, the ANZECC guidelines (which apply to both marine and freshwater sediments). The guidelines for US freshwater sediments published by MacDonald et al. (2000, also provide a comprehensive resource.

[15] The ANZECC (2000) sediment quality guidelines were originally those given by Long et al. (1995) or Long & Morgan (1990).

Current guidelines for stormwater impact assessment

Guidelines are continually being revised as new information on linkages between contaminants and ecological or human health effects becomes available. The guidelines that are currently commonly used for assessing the potential effects of stormwater in Auckland's receiving waters are summarised in Table 8 below.

Table 8
Current guidelines commonly for assessing the potential effects of stormwater in Auckland.

Receiving environment	Protection targets	Primary guidelines	Other useful guides
Freshwater:			
Water quality	Ecological health	ANZECC (2000)	USEPA (2004) criteria can be used to assess acute and chronic effects of dissolved metals
Pathogens	Human health	MFE (2003)	
Nuisance growths	Ecological health and aesthetics	ANZECC (2000)	MFE (1992) provides NZ background information
Colour and clarity	Aesthetics and ecology	ANZECC (2000)	MFE (1994) provides NZ background information
Stream flow	Ecological health	Elliot et al. (2004)	
Physical barriers	Fish passage	ARC (2000a)	
Sediment quality	Ecological health	MacDonald et al. (2000)	ANZECC (2000) SQG also apply to freshwater sediments
Marine and estuarine:			
Water quality	Ecological health	ANZECC (2000)	Blueprint provides guidance on application and interpretation (ARC 2004c)
Pathogens	Human health	MFE (2003)	Also "blueprint" – as above
Sediment quality	Ecological health	ARC (2004c) ARC (2006c)	ANZECC (2000) provides useful background information on SQG. See footnote 1.
Shellfish quality	Human health	FSANZ (2002a and b)	
Sedimentation	Ecological health	ARC (2004e)	See footnote 2

Notes:

1 Detailed evaluation of relationships between contaminant (Cu, Pb, and Zn) concentrations and ecological health in Auckland's estuaries is provided in ARC (2006c).

2 Effects of sedimentation are complex (see Chapter 7), but some guidance has been provided in ARC (2004e) that may aid interpretation of potential effects.

Information gaps

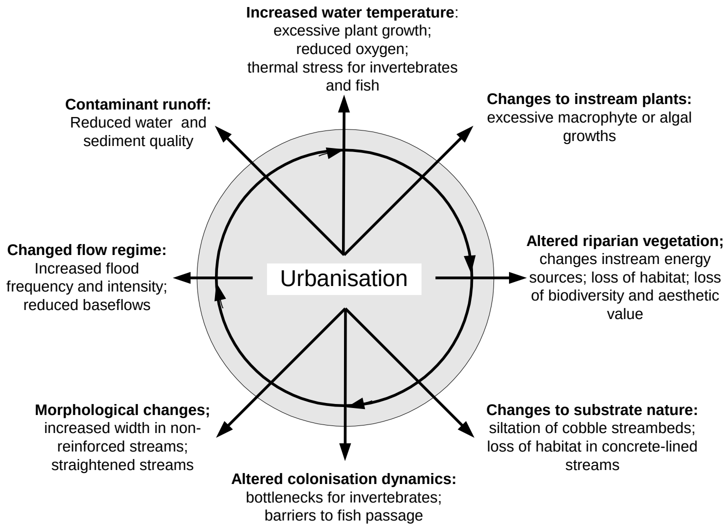
Application of the above guidelines has identified a number of information gaps that, if filled, would significantly improve our ability to reliably assess potential impacts of stormwater using guidelines:

- Viruses – there may be significant human pathogens in stormwater, especially where wastewater overflows are prevalent. Viruses behave differently from bacterial indicator organisms in receiving waters, and therefore guidelines based on bacterial indicators may not reliably reflect human health risks. Guidance on the best viral indicators to use, how they should be monitored, trigger levels, and relationship with other microbiological guidelines for both freshwater and marine environments is required.
- The appropriateness of overseas-derived Sediment Quality Guidelines (eg ANZECC) to the NZ setting – where contaminant concentrations are generally lower, and benthic ecology may have significantly different sensitivity to contaminants of concern (especially Zn and Cu) – needs further testing. Note that the relationship between benthic ecology and key contaminants (Zn, Cu) in Auckland's marine sediments is currently being indirectly assessed through the Healthy Benthic Community Model (described in Chapter 7), and this will provide important data for refining and testing trigger levels. This has been significantly advanced in recent studies (ARC 2006c).

6 Impacts in the Freshwater Environment

Urbanisation causes a wide range of changes to the stream environment, affecting water quality, stream flows, stream channel shape and stability, riparian vegetation, and streambed characteristics. These changes (or "multiple stressors") can adversely affect the quality of aquatic plant and animal communities inhabiting urban streams. The diagram below summarises these potential effects.

Figure 19
The effects of urbanisation on the freshwater environment (reproduced from Suren & Elliot 2004).



State of knowledge in 1995

In 1995 it was well recognised that Auckland's urban streams were often heavily modified and provided poor habitat for aquatic life. Key factors affecting degradation that had been identified at that time included:

- physical modifications of the stream channel – channelisation, artificial waterfalls, concrete lining, sedimentation, and removal of riparian vegetation;
- hydrological changes – more frequent flood flows, reduced groundwater inflows; and
- poor water and sediment quality – low clarity, depressed dissolved oxygen concentrations, elevated levels of microbial indicator organisms, and high concentrations of organic contaminants and heavy metals in sediments.

The small size, narrow channels, low volumes of Auckland streams make them particularly susceptible to stormwater impacts. The 1995 review provided a concise summary of these issues.

Key studies pre-1995

The ARC's regional long-term baseline water quality monitoring programme (LTB-WQ) and the Tamaki catchment monitoring programme included regular monitoring of several streams receiving urban run-off – Oakley Creek, Puhinui Stream, Oteha Stream, Lucas Creek, Pakuranga Creek, Otaki Creek, Omaru Creek, and Otara Stream (ARC 1992b, 1993a). The data from these programmes contributed to the broad understanding of urban stream water quality and how it compared with streams influenced by other (predominantly rural) land uses.

More localised studies included:

- stormwater and stream water quality in Wairau Creek (ARA 1983, Williamson 1986);
- urban stormwater and stream water quality in the Pakuranga catchment (ARC 1992a and 1994c); and
- sediment quality and ecology in Auckland isthmus streams (NIWA 1995a and b).

These studies gave more detailed information on the short-term temporal variability in urban water quality, and the concentrations of key urban contaminants in selected stream waters and sediments. The Pakuranga Stream studies (ARC 1992a and 1994c) provided detailed information on the nature and variability in stream and stormwater quality in a residential catchment over an approximately seven-year period.

Advances since 1995

Since 1995, there have been significant advances in the amount and quality of information on Auckland's urban streams. Studies have:

- refined and expanded the knowledge base, enabling better definition of the nature and extent of impacts on water quality and ecology;
- provided methods for classifying streams and defining realistic management objectives for Auckland streams; and
- begun investigations to link stream condition (environmental "state") with catchment influences ("pressures"), using the Pressure-State-Response (PSR) environmental management framework.

This chapter summarises these advances under the following headings:

6.1 – Impacts on water quality

6.2 – Impacts on sediment quality

6.3 – Impacts on stream ecology

6.4 – Toxicity assessment in the freshwater environment

6.5 – Pressure-State-Response (PSR) models linking environmental condition (state) with catchment pressures and management responses.

6.1 Impacts on water quality

Key impacts of urban development on stream water quality are:

- Decreased water clarity (and increased suspended solids concentrations and turbidity) associated with discharge of fine particulates (eg gutter dust, soil, vegetative debris) and increased stream channel erosion.
- Elevated maximum water temperatures, resulting from reduced summer low-flows and loss of stream shading.
- Changes in dissolved oxygen dynamics, with higher maximum and lower minimum concentrations, which result from greater primary productivity (during the day) and microbial respiration (at night) in nutrient-enriched, poorly shaded streams, especially during summer low flows. Organic enrichment from WWOFs, intermittent spills of organic wastes, and introduced soil and vegetation contribute to this problem at times.
- Contamination by pathogenic microorganisms, as a result of WWOFs, leaking sewers, and animal wastes in stormwater run-off.
- Nutrient enrichment, from fertiliser run-off, sewage overflows, and soil loss. Plant nutrients, including nitrate-nitrogen (NO_3-N) and phosphorus (P), are present in moderate amounts in stormwater. Concentrations are much higher in sewage, and therefore stormwaters contaminated by WWOFs have considerably higher nutrient concentrations.
- Increased concentrations of toxic substances (NH_4-N, heavy metals, PAH), from wastewater overflows (NH_4-N), vehicular use and road run-off (heavy metals, PAH), fuel spills (hydrocarbons and PAH), and greater amounts of exposed metallic surfaces (particularly roofs) in urban areas (Zn, Cu).

Aesthetic impacts can also result from litter, which is common in many urban stream reaches, and the occasional presence of oily films, discolouration, and unsavoury odours.

Change in stream flow regime is another major impact of urbanisation. Higher peak flows, increased total volumes of stream discharge during storms, and reduced summer base flows are well documented consequences of increased catchment imperviousness and channel modifications that often accompany urbanisation.

The nature and severity of these impacts vary greatly, both spatially (between streams and along a given stream) and temporally (over time), depending on a combination of factors including stream characteristics (size, gradient, stream bank and bed stability, riparian cover, neighbouring property types etc), storm size and duration, and catchment characteristics (eg numbers of WWOF and stormwater inputs, number, type and locations of stormwater treatment devices, and the type, history, stage, and intensity of urban development). Impacts are potentially severe in Auckland's urban streams because stormwater can make up a substantial proportion of stream flow.

These factors are important to consider when assessing the water quality of urban streams. The results obtained from water quality sampling provide a measure of the conditions prevailing at the time of sampling, and therefore the results largely reflect the effects of catchment activities and weather conditions at that time. Because of the highly variable nature of urban catchment activities, water quality in an urban stream can vary greatly within short time periods and between locations. A truly representative picture of urban stream water quality can only be obtained by multiple samplings over the time period of interest.

Comparing results obtained in programmes that use different sampling strategies should therefore be done with caution – the results can be highly dependent on how the monitoring was done, and interpretation of the results can be flawed unless details of the programme are understood.

"Integrative" monitoring methods – eg sediment chemistry and biological monitoring – are therefore useful complementary tools to assess the cumulative effects of stormwater in streams. These approaches are described in Sections 6.2 to 6.5.

The following sections summarise the findings of regional water quality monitoring and some localised studies to illustrate the impacts of urban stormwater on stream water quality in Auckland.

6.1.1 Regional monitoring

Two regional monitoring programmes, described below, are being conducted by the ARC. The regional programme data are of great value because they currently represent the only long-term, consistently gathered, information on stream water quality in Auckland that can be confidently used to measure the effects of various land uses and trends over time. They provide a regional context from which to judge the impact of urban stormwater on streams; in particular the fact that many urban streams are already impacted by upstream rural land use.

The regional programmes mostly sample during base flow conditions, and high-flow events are generally not encountered (eg <3 per cent of the time for LTB-WQ over the 1992–2000 period; NIWA 2000a). The data from urban stream sites therefore provide information on the long-term residual impacts of stormwater and shallow groundwater inflows rather than the full impact of short-term storm events.

ARC Long-term baseline water quality monitoring

The ARC long-term baseline water quality (LTB-WQ) programme currently monitors water quality at 16 freshwater stream sites in the Auckland region (ARC 2000b, NIWA 2000a). The key aims of the programme are to determine broad-scale effects of different land uses in the region and to measure trends in water quality over time (temporal trends).

Most of the LTB-WQ stream catchments are dominated by rural land uses, including pastoral farming, horticulture, lifestyle blocks, and native and plantation forestry (ARC

2003e) – urban land is generally a minor component. For most streams, the impacts measured by the LTB-WQ programme therefore reflect effects of mixed land uses rather than solely urban stormwater impacts.

Five of the streams (Oakley Creek, Puhinui Stream, Oteha Stream, Lucas Creek, and Otara Creek) are strongly influenced by urbanisation (and hence stormwater inputs). Of these five streams, only Oakley Creek has a completely urbanised catchment, having over 98 per cent of the catchment in urban land use (including open urban space; ARC 2003e). Oteha Stream is mostly urban (87 per cent), while Puhinui (40 per cent) and Lucas Creek (22 per cent) have significant urban areas. Otara Creek has a smaller proportion of urban land use in its upstream catchment (4.5 per cent, although this has probably grown significantly since the land use assessment was done), but the lower reaches are significantly influenced by urban activities. Other LTB-WQ catchments have <3 per cent urban land use (ARC 2003e).

Water quality at all sites, apart from Oakley Creek and Oteha Stream, is therefore influenced by mixed land uses. Because of these mixed land uses, the LTB-WQ programme cannot give an unambiguous picture of urban stormwater impacts on streams (except for Oakley Creek).

Monitoring at sites above and below the urban boundary is required to fully assess the impact of urbanisation on a stream – an example of this approach, Waitakere City's "Project Twin Streams" (PTS) catchment monitoring programme, is described later in this section.

Tamaki catchment stream monitoring

Regular long-term monitoring has also been carried out from 1987 at six urban stream sites in the Tamaki Estuary catchment – Pakuranga (3 sites), Otaki, Omaru, and Otara Creeks – as part of the ARC "special survey" monitoring of Tamaki, Upper Waitemata and Mahurangi catchments (ARC 1999b, ARC 2003f). The Tamaki streams are mostly urban, being impacted by a mixture of residential, commercial, and industrial land uses.

Land use effects on stream water quality

Water quality in the LTB-WQ programme has been found to be related to catchment land use, and can be broadly ordered as follows (ARC 2000b):

Native bush (best) > exotic forest ≈ market garden ≈ mixed rural > rural/urban > urban (worst).

This relationship was based on the sum of rankings of seven commonly measured water quality variables[16], but didn't include potentially important "urban" indicators such as heavy metals or hydrocarbons[17].

The worst five streams in the LTB-WQ programme were (in order of decreasing water quality) Lucas > Papakura > Puhinui > Oteha > Otara (worst). Oakley Creek, which has

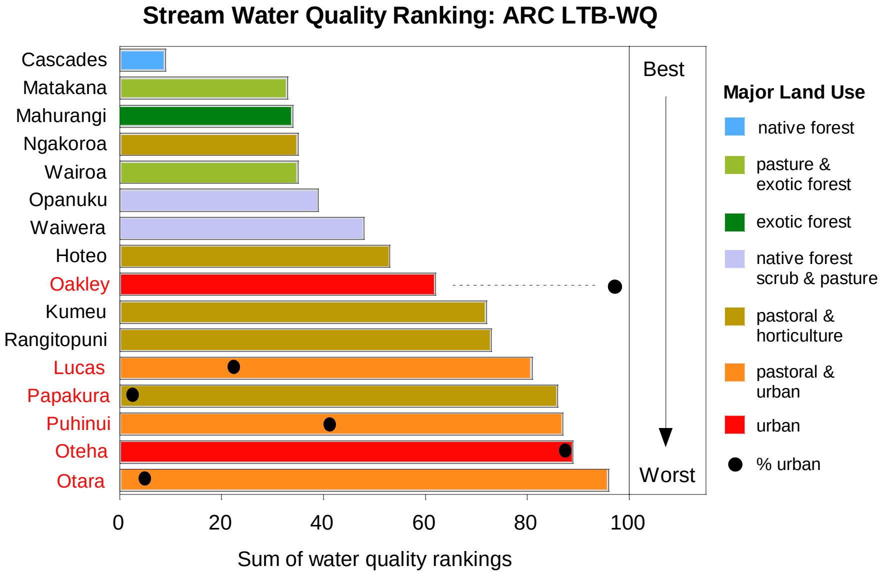
[16] Clarity (black disk), DO (per cent saturation), faecal coliforms, ammonia, nitrate, total phosphorus, suspended solids.

[17] Note that if urban contaminants such as heavy metals, PAH, or hydrocarbons were included in the ranking, urban streams would be ranked even lower in quality than is shown here.

the highest proportion of urban land use in the catchment, was not one of the worst sites, ranking 9th out of the 16 sites. Figure 20 shows the ranking of sites from the ARC LTB-WQ programme (based on 1992–2000 data given in ARC 2000b). A similar ranking for 16 "soft-bottomed" streams was obtained from data on water quality and benthic macroinvertebrate communities obtained between 1992 and 2003 (ARC 2005f).

Figure 20

Ranking of long-term median water quality (for 1992–2000) in Auckland streams monitored in the ARC LTB-WQ programme. Streams that are significantly influenced by urban run-off are labelled in "red". Ranking data are from ARC (2000b), and land use (excluding "urban open space") from ARC (2003f).



Long-term average (median) water quality for the ARC LTB-WQ and Tamaki catchment streams is graphically summarised in Figures 34a and b, which are appended to this chapter (Section A6.1).

These long-term data indicate that, compared with rural or forest streams, urban streams have:

- higher and/or lower dissolved oxygen concentrations; eg Pakuranga Stream at Botany Road, where the shallow, exposed channel allows attached algae to proliferate, has daytime DO which reaches supersaturated levels. In contrast, the Otara Stream LTB-WQ site, has depressed DO, due to higher organic loadings;
- generally higher ammonia concentrations, although on average these are well below those that would be toxic to aquatic life;
- slightly higher (although variable) nitrate, and substantially higher total phosphorus concentrations, which are sufficient to promote excessive in-stream growth of aquatic plants. Wastewater overflows are a possible contributor of these nutrients;

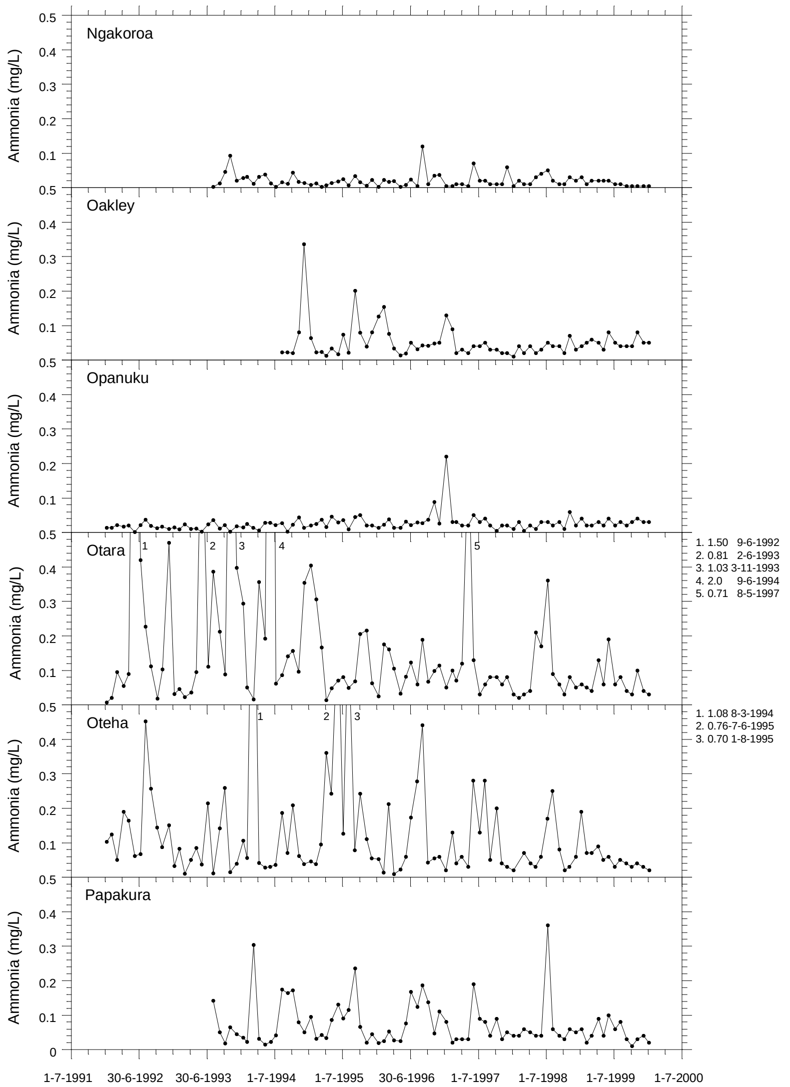
- similar average daytime temperatures, which are below 20°C and therefore should not expose aquatic life to prolonged periods of thermal stress. Note, however, that maximum temperatures can be very high in exposed stream reaches – spot measurements along Oakley Creek in December 2000 found temperatures at some sites approached or exceeded 30°C, which would almost certainly adversely affect sensitive aquatic life forms (ACC/Metrowater 2001); and
- generally higher (and sometimes much higher) faecal coliforms, suspended solids, and turbidity. Microbial contamination is usually well above that considered safe for contact recreation[18].

Figure 21 (taken from ARC 2000b), illustrates the difference between the urban/rural streams (Otara, Oteha, and Papakura Creeks), the purely urban (but better quality) Oakley Creek, and two rural streams (Ngakoroa and Opanuku Streams), using ammonia as an indicator. The higher ammonia concentrations and greater variability in the urban/rural streams, and to a lesser degree the urban stream (Oakley), are evident. The high variability in water quality is a feature of small streams, particularly those receiving urban stormwater and wastewater overflows.

[18] MFE (2003) guidelines use E. coli as the indicator for freshwaters, but these were not measured in the ARC programmes until 2002. ANZECC (2000) primary contact recreational guidelines are 150 FC/100 mL.

Figure 21

Ammonia concentrations in six streams in the ARC LTB-WQ monitoring programme between 1992 and 2000, showing the particularly poor quality and high variability in the urban/rural Otara, Oteha, and Papakura Streams (taken from ARC 2000b).



Trends in water quality over time

Regional stream water quality monitoring data (ARC 2000b and ARC2003f) show that water quality has been very poor at urban and urban/rural sites in the past 10–15 years (as shown in Figure 21 for ammonia). Examples include:

- Pakuranga Creek in the mid to late 1990s, where suspended solids' concentrations up to 1500 mg/L, BOD up to 150 mg/L, DO below 20 per cent (and above 150 per cent saturation), and NH_4-N up to 2.9 mg/L were recorded at times. Elevated nitrite, nitrate, and phosphorus concentrations were also recorded during this period;

- Otara Creek, where DO concentrations well below 50 per cent saturation were regularly recorded throughout the 1990s, BOD levels above 5 mg/L were relatively common, and elevated NH_4-N and TP were also present;
- Oteha Stream, where high turbidity (exceeding 100 NTU) and SS concentrations (up to 230 mg/L) occurred throughout the 1990s. Elevated ammonia and nitrite were also present at times in the mid-1990s (possibly a consequence of the sewage discharge); and
- Lucas Creek, where very high turbidity (>500 NTU) and SS (up to 480 mg/L) were recorded in the mid to late 1990s.

Water quality was apparently even poorer than this in the 1980s, with extremely high concentrations of BOD, SS, and NH_4-N, and low DO concentrations being reported for Tamaki catchment streams (ARC 1999b).

Despite very poor water quality at times in the past, statistical analysis of the long-term ARC monitoring record shows few consistent trends in water quality over time from 1992 (ARC 2000b). This may be a consequence of the high variability in urban stream water quality, which reduces the sensitivity of monitoring for trend detection. The major change in water quality in Auckland's streams between 1992 and 2000 was a general increase in temperature of 0.1–0.5 °C per year (ARC 2000b). This is not solely an urban effect because it has also occurred at non-urban sites. Increasing temperature is important because small streams can be thermally stressed in summer, and, if these trends continue, will become increasingly so in future. This may render some streams unsuitable for aquatic life. Measures such as improving stream shading are required to counter this effect.

Regional monitoring summary

In summary, the ARC regional stream monitoring programmes have, since the mid-1980s, documented the quality of 16 streams in the LTB-WQ programme – 11 of which are significantly impacted by urbanisation – and a further six urban stream sites in the Tamaki catchment.

While it is unlikely that the monitoring data reflects the full impact of stormwater inputs (because storm events were not often sampled), it has shown that urban streams usually, but not always, have poorer water quality than those in other land uses. Very poor water quality has historically been measured, probably as a result of sediment run-off and inputs of organic and nutrient-rich wastes from both established and developing urban areas.

Long-term monitoring since around 1987 has not (to 2000), revealed any significant changes in urban stream water quality (apart from a rise in temperature, which is affecting all monitored streams).

Apart from Oakley Creek, all the streams monitored in the regional programme are influenced not only by urban inputs, but also by upstream rural land use, which is known to contribute to degraded water quality. The monitoring therefore generally shows cumulative impacts from a range of potential sources rather than those associated purely with urban run-off. Unambiguously quantifying the impacts of urban run-off on stream water quality requires monitoring at sites upstream and downstream of the urban boundaries – an example of this approach, being undertaken by Waitakere City in their "Project Twin Streams", is summarised later in this chapter.

6.1.2 Local studies

A number of shorter term water quality studies on urban streams have been conducted in the region since 1995, including:

- North Shore City – wet- and dry-weather sampling of 16 urban streams between September 1999 and March 2000 (Meritec 2000a). Information from these surveys is included in several "Project Care" reports, which summarise the condition of aquatic receiving environments in North Shore City drainage catchments (KMA 2001). These data are also included in the comprehensive physical and ecological assessment reports for North Shore streams, available on the NSCC website[19].
- Otara Creek and stormwater inputs to Otara Lake (Manukau City) – five dry weather and three wet-weather samplings between October 1999 and January 2000 at several sites impacted by residential, commercial, and rural land uses (Worley 2000).
- Waiheke Island – "one off" sampling at six urban sites in February 2001 (ACC 2001).
- Auckland City – a number of short-term studies at approximately 10 stream sites between 1994 and 2000 (summarised in ACC/Metrowater 2001 and NIWA 2001a), more detailed studies of two "large streams" (Oakley Creek and Whau River) carried out between January 2002 and May 2003, and 16 "small stream" sites (two wet and two dry weather samplings in May and June 2002), which are comprehensively summarised in NIWA (2005).
- "mid-Waitemata Harbour" streams – a single sampling of 15 sites on 11 streams draining into the middle Waitemata Harbour (ARC 1999c).
- Waitakere City – a catchment-wide "snapshot" study of stream water quality at 30 sites (14 urban) in relation to land use and stormwater discharges was carried out in October/November 1999 (KMA 2000),
- urban stream research conducted by NIWA at several Auckland stream sites as part of a national research project (NIWA 2000b).
- the Waitakere City "Project Twin Streams" monitoring programme (EVA et al. 2004a–c), is monitoring water quality (and sediment quality and ecology) at 15 sites in the Henderson catchment, enabling the impacts of urbanisation to be quantified[20]. This study is described later in this section.

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[19] www.northshorecity.govt.nz. Go to "Streams" page in the A-Z index to find the "Stream Assessment" reports.

[20] The Project Twin Streams programme also monitors sediment quality, stream ecology, and catchment "pressures" to obtain an integrated assessment of land use effects on the aquatic environment, and to investigate cause-effect relationships using the PSR framework. The number of sites was extended to 19 for the summer of 2005-6 (EVA et al. 2006a&b).

While there is a large amount of information in these studies, it is difficult to provide a detailed objective comparison of their findings because of the different monitoring and data reporting methods used (unlike the regional monitoring, where consistent methodologies are used). Most of these studies were reviewed by EVA et al. (2003a), who found that the general features of the water quality were similar to those summarised above for "regional monitoring".

The localised studies do, however, contribute extra information and provide a wider perspective of water quality across Auckland's streams, and in particular provide additional data on the concentrations of heavy metals (mainly Cu and Zn).

Data from these local studies have been compiled, summarised, and displayed graphically at the end of this chapter (Section A6.2).

Waitakere City – Project Twin Streams

An extensive aquatic environmental monitoring programme has been initiated as part of Waitakere City's "Project Twin Streams" (PTS; EVA et al. 2003b–d, 2004a–c, 2005a and b, 2006a and b). This programme is of particular value in illustrating the effects of urbanisation on stream water quality (as well as sediment quality and aquatic ecology) because it includes sites that encompass the full range of land uses from the bush-clad headwaters to fully urbanised lower-catchment zones.

The PTS stream water quality monitoring involves monthly base flow sampling over a six-month "summer" period (November–April inclusive) at 15 sites in three key stream catchments (Oratia, Opanuku, and Waikumete), with the sites placed to assess the effects of different land uses (Figure 22). An additional four sites on the neighbouring Swanson Stream were added for summer 2005–6 (EVA et al. 2006a and b). Monitoring began in summer 2003–2004 and is planned to be repeated every two years or so to assess trends over time and water quality responses to changing land use and catchment management[21].

Sampling sites are located to reflect the effects of key land uses – native bush headwaters, rural (pastoral, horticultural, and lifestyle blocks), urban fringe (the boundary between urban and rural, or bush and urban[22]), and established urban.

The PTS monitoring from summer 2003–4 clearly shows the effects of urbanisation water quality, even though direct monitoring of storm events is intentionally avoided. Some of the data obtained are presented in Figure 23.

Urban stream reaches show:

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[21] The monitoring follows the Pressure–State–Response (PSR) framework, which aims to link environmental state with catchment pressures and management responses. See section 5.5.

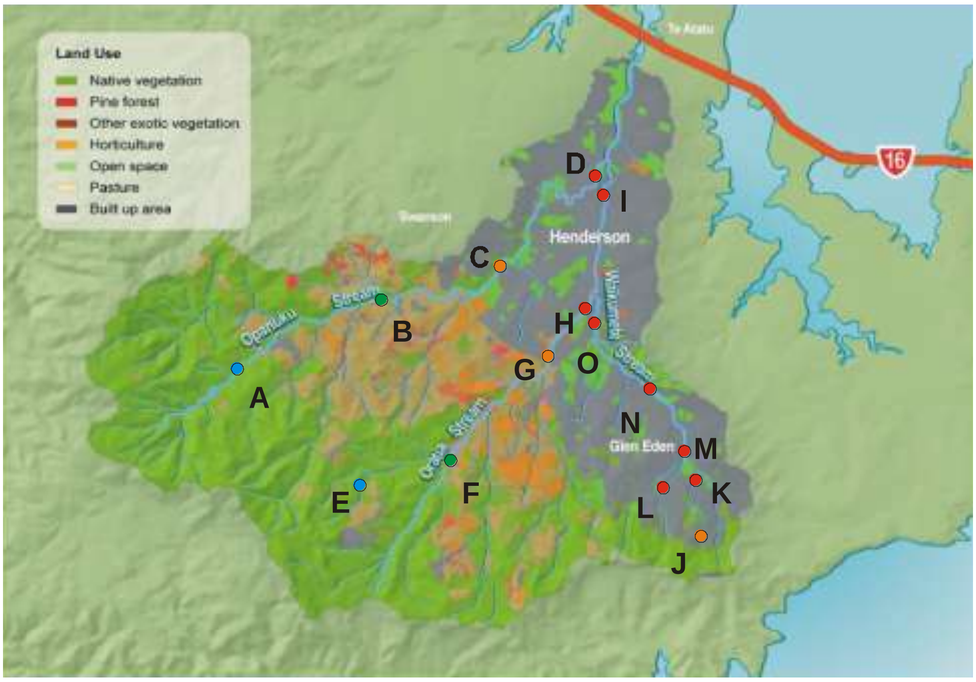
[22] Bush living in Waitakere is well established, with low-density housing sited in established native forest. Therefore headwater sites can experience urban influences. For example, site J is a headwater bush site (and serves as the headwater reference site for the Waikumete Stream system), but is at the start of urban zone and is slightly impacted by urbanisation. Similarly, site E, in the headwaters of the Oratia Stream, appears to be affected by mild nutrient enrichment, possibly as a result of neighbouring low-density development in the surrounding bush-clad hills.

- higher concentrations of dissolved Zn and, to a lesser degree, Cu. Concentrations approach or exceed ANZECC guidelines at urban sites;
- elevated, and highly variable, ammonia concentrations at sites affected by wastewater overflows (eg site K). However, concentrations are below water quality guidelines for aquatic toxicity;
- Higher levels of microbiological contamination (in this case the indicator organism E. coli); and
- Slightly higher turbidity and nitrate-N concentrations, and lower clarity than upstream rural sites.

In addition, continuous temperature monitoring over the summer has shown that urban stream reaches have slightly wider temperature ranges, and higher maximum temperatures, than rural or headwater sites further upstream (EVA et al. 2006a). Temperatures in these generally well-shaded streams do not, however, approach the extremely high values measured in the shallow, unshaded, concrete-lined channels present in some other parts of Auckland (eg Oakley Creek, where temperatures of 30ºC have been recorded; ACC/Metrowater 2001).

Figure 22

Project Twin Streams (PTS) monitoring sites sampled for water and sediment quality and aquatic ecology in summer 2003-4 (EVA et al. 2004a–c). Base map courtesy of Kingett Mitchell Ltd. Four sites on the Swanson Stream (not shown) were added in summer 2005–6.



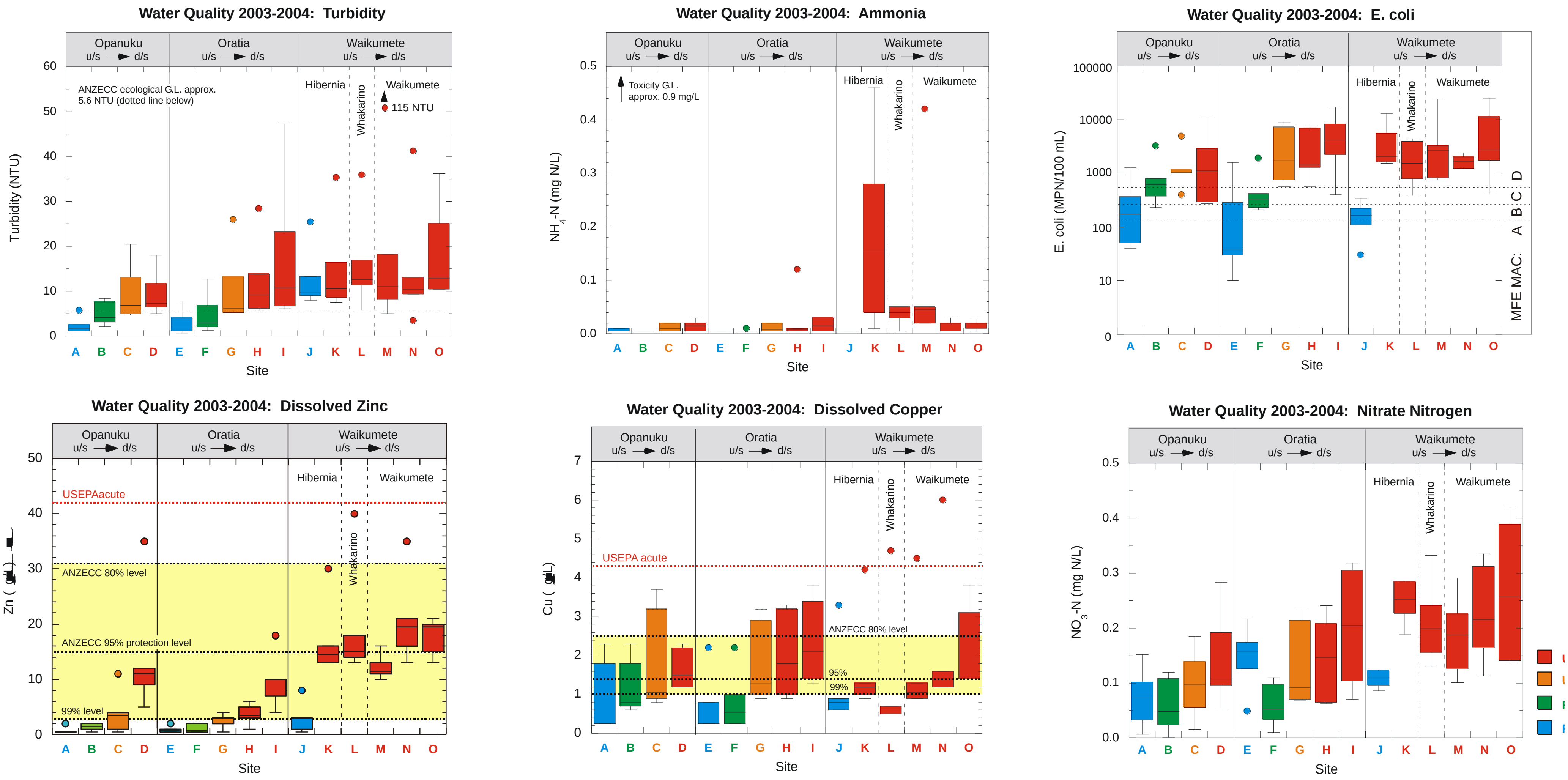
Upstream rural land use contributes to the poor water quality observed in the urban stream reaches, as shown by generally degraded water quality at the rural-urban fringe sites.

These results are consistent with the findings of regional monitoring and other local studies, and highlight that urban streams have water quality characteristics similar in some respects to degraded rural streams. Urban streams can sometimes, but not always, have higher concentrations of ammonia, nutrients (N and P), pathogenic indicator microorganisms, and turbidity than rural streams. Wastewater overflows and earthworks are major sources of these contaminants.

A key difference between urban and rural stream reaches is the higher concentrations of dissolved Zn, and to a lesser degree Cu, at urban sites.

Figure 23

Water quality in Waitakere City's "Project Twin Streams" monitoring programme, summer 2003-4 (adapted from data in EVA et al. 2004a). Sites are shown in Figure 22.



Urban stream water quality – summary

Urban streams share many of the characteristics of degraded rural streams – ie relatively poor water clarity, elevated (and variable) concentrations of suspended solids, turbidity, nutrients, and ammonia, and very poor microbial quality. Temperatures can be high in summer in poorly shaded reaches, and dissolved oxygen concentrations may vary widely in response to diurnal changes and/or organic waste inputs.

Unique to urban streams is the elevated concentrations of heavy metals, principally Zn. Concentrations often exceed chronic, sub-lethal, toxicity guidelines and occasionally (during storm run-off events) increase to concentrations that may be acutely toxic to sensitive aquatic fauna.

The major factor limiting human recreational use of urban streams is the high level of microbiological contamination. Microbiological water quality guidelines to protect human health are almost always exceeded in urban stream reaches, and usually by large amounts. Because of this, direct contact with urban streams is not recommended. The collection of aquatic plants (eg watercress) and animals (eg koura, eels) for human consumption is also probably unwise due to the high potential for exposure to pathogenic microorganisms.

Aquatic life in urban streams is likely to be affected by a combination of stressors, including:

- elevated temperatures and large fluctuations in dissolved oxygen concentrations in poorly shaded reaches of small streams in summer;
- poor water clarity and elevated concentrations of fine suspended particulates (which can be highly contaminated – see discussion on sediment quality, Section 6.2);
- occasional inputs of wastes containing ammonia and biodegradable organic material; and
- elevated concentrations of heavy metals (mainly Zn).

Of these stressors, water clarity most consistently does not meet water quality guidelines. The other contaminants approach, or occasionally exceed, guidelines. Exceedances are usually greater during storm events. Poor clarity is a consequence of excessive inputs of fine suspended particulates, which originate from stream channel erosion as well as catchment inputs. Auckland's clay soils exacerbate the problem, producing relatively low clarity in even only slightly disturbed streams. Stormwater flow management and sediment controls are clearly important to try and improve stream water clarity in Auckland.

Temperatures are almost certainly above those suitable for sensitive aquatic life forms in unshaded reaches of small urban streams in summer – however, there is little data to assess how bad or widespread this problem is.

It is currently difficult to state whether urban stream water quality is getting better or worse over time. So far, increasing temperature is the only consistent change detected at the ARC monitoring sites. However, this is a region-wide, rather than specifically urban effect, and the cause is unknown.

As stormwater and wastewater management improves, resulting in fewer gross contamination events and better quality stormwater inflows, variability in urban stream water quality should reduce. This may permit future monitoring to detect more subtle changes over time.

6.2 Impacts on sediment quality

By 1995, a small number of Auckland studies (ARC 1992a, 1994a; NIWA 1995a) had established that urban stream sediments could be highly contaminated, with concentrations of heavy metals (Cu, Pb, and Zn) and, in some places PAH, well above sediment quality guidelines for the protection of aquatic life. It was also known that levels of contamination could vary greatly within a stream because of factors such as variable sediment texture and contaminant dilution through inputs of cleaner material sourced from stream bank erosion and catchment soils, especially sub-soils released during earth working.

Since then, a number of studies have improved our knowledge base, including:

- a study of 14 streams flowing into the middle-Waitemata Harbour, where Cu, Pb, Zn, and PAH were measured, including analysis of metals in the fine sediment fraction (<63 m) to improve comparability between sites with different sediment textures (NIWA 1998a);
- 14 urban stream sites in Waitakere City, where Cu, Pb, Zn, TPH, PAH, and organochlorines were measured (KMA 2000). Variations in sediment textures between sites complicated interpretation of these data, which have since been superseded by the Waitakere City PTS monitoring programme (described later; EVA et al. 2003c, 2004b, 2006b);
- six urban stream sites on Waiheke Island (ACC 2001);
- eight sites in the Pakuranga, Awaruku, Kahika, and Botany Downs Streams (NIWA 2000b); and
- 15 sites (now increased to 19 sites) in the Opanuku, Oratia, Waikumete and Swanson Stream catchments in Waitakere City Project Twin Streams (PTS) monitoring programme (EVA et al. 2003c, 2004b, 2006b). Systematic, replicated, sampling for Cu, Pb, Zn, PAH, organochlorine pesticides (mainly DDT) was conducted in 2003 (and repeated in 2006) at the same locations described previously for stream water quality (Section 6.1).
- 16 “small stream” sites in Auckland City, which were analysed for Cu, Pb, Zn, and PAH (NIWA 2005). In addition, the NIWA studies also analysed the metals’ concentrations in suspended sediments taken from the water column.

Data collected prior to 2003 were summarised by EVA et al. (2003a) and compared with sediment quality guidelines (in this case the freshwater guidelines of MacDonald et al. 2000)[23]. The streams sampled until that time had a substantial proportion of

[23] The MacDonald et al. (2000) guidelines provide two values, the TEC (Threshold Effects Concentration) and the PEC (Probable Effects Concentration), which are analogous to the ANZECC ISQG-low and ISQG-high

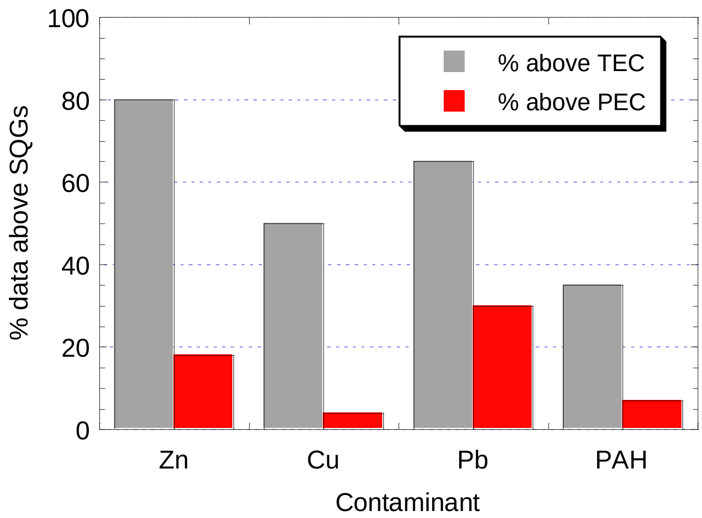
samples above guideline levels, with the greatest proportion of exceedances by Zn and Pb (Figure 24).

Using these data as a guide, it appeared that most urban stream sediments could be expected to have Zn and/or Pb concentrations that are high enough to potentially adversely affect aquatic life. This was also found to be the case in the 16 "small streams" in Auckland City studied by NIWA (2005). In this study, three of the 16 streams had Zn concentrations exceeding the ANZECC ISQG-high and eight of 16 were above the ISQG-low. For Pb, 2/16 and 9/16 exceeded the ISQG-high and ISQG-low guidelines respectively. For Cu, four out of the 16 sites exceeded the ISQG-low, and none exceeded the ISQG-high.

Several of these "small streams" also had very high PAH concentrations (up to 30 mg/kg), which were above ANZECC sediment quality guidelines.

Figure 24

Sediment quality guideline exceedance by key urban contaminants in Auckland's urban stream sediment samples taken prior to 2002. Figure taken from EVA et al. (2003a).



Threshold Effects Level (TEC) and Probable Effects Level (PEC) are freshwater sediment quality guidelines given by MacDonald et al. (2000).

High concentrations of Zn, Cu, and Pb are also found in the fine particulate matter suspended in the water column in urban streams. Concentrations in the suspended solids from 16 "small streams" in Auckland City (NIWA 2005) during normal dry weather flows had median concentrations of Cu of 191 mg/kg (range 36–475), Pb of 136 mg/kg (19–563), and Zn of 1107 mg/kg (235–7114). Concentrations during wet weather flows were similar. Comparably high metal concentrations were also found in stream suspended solids in Oakley Creek and Whau River (NIWA 2005). Stormwater suspended solids have also been well characterised in Auckland City, and found to contain (on average) 270 mg/kg Cu, 380 mg/kg Pb, and 1900 mg/kg Zn (Table 40 in NIWA 2005).

respectively. Concentrations below the TEC are unlikely to have adverse effects on aquatic life, while concentrations above the PEC are likely to have significant, frequent, adverse effects.

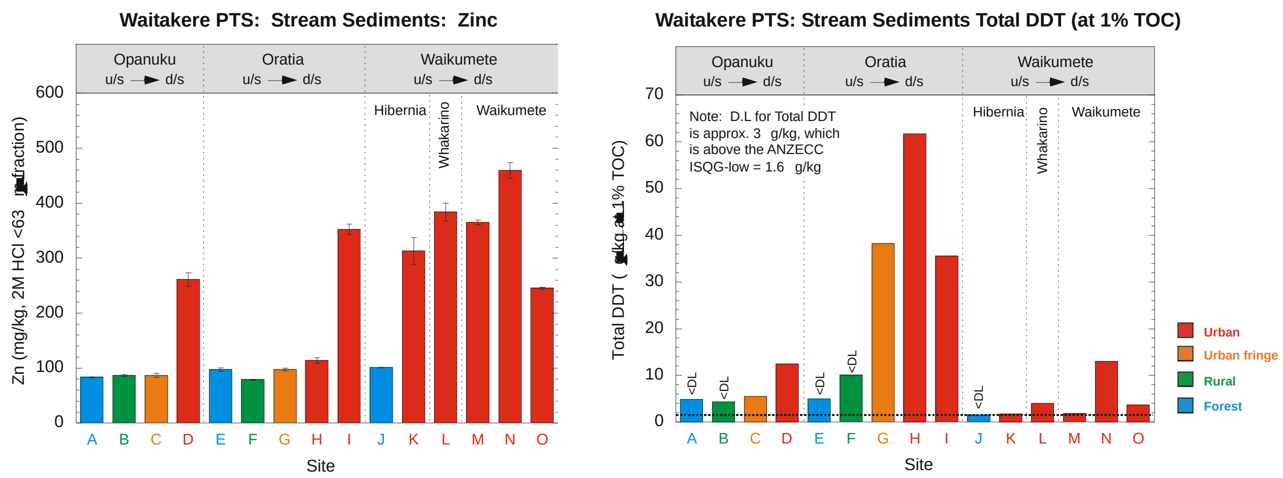
The effect of urbanisation on stream sediment contamination has also been clearly demonstrated in the Waitakere City Project Twin Streams (PTS) monitoring. Concentrations of Zn, Cu, Pb and PAH increase markedly downstream of the rural–urban boundary. Figure 25 shows these changes for Zn. Spatial trends for PAH and Pb were similar to those observed for Zn, while Cu changes were less marked, partly due to higher Cu concentrations in sediments at headwater sites (presumably a reflection of the geology of the Waitakere Ranges) and possibly also as a consequence of inputs into rural stream reaches from horticultural land, which was historically a major land use in parts of this catchment.

Despite large reductions in Pb use following its removal from petrol in 1996, Pb continues to be present in high concentrations in urban stream sediments (and stream water suspended solids). This indicates that substantial reservoirs of Pb remain in urban catchments, probably in soils and/or streambed sediments.

Organochlorine pesticides (OCPs), which are not primarily urban contaminants, may also be present in urban stream sediments. Major OCP contaminants are DDE and DDD, originating from DDT use on horticultural land. This is evident in the Waitakere PTS monitoring (Figure 26), particularly in the Oratia Stream catchment, which historically had a substantial amount of horticulture. Total DDT concentrations exceed sediment quality guidelines in some areas. Relatively high concentrations of DDTs (20 µg/kg and 80 µg/kg) have also been found in stormwater treatment ponds receiving run-off from urban areas that were once in horticultural land use (DSL 2003a).

Figure 25

Concentrations of Zn (in <63 m fraction) and Total DDT (normalised to 1 per cent TOC) in stream sediments in Waitakere City's "Project Twin Streams" monitoring programme in 2003. Adapted from data in EVA et al. (2004b).



Impacts on stream sediment quality – summary

Urban stream sediments can accumulate high concentrations of Cu, Pb, Zn, PAH and, in ex-horticultural areas, DDTs (mainly DDE). Concentrations of all these contaminants frequently exceed sediment quality guidelines to protect aquatic life, and it is therefore reasonable to conclude that stormwater-derived chemical contaminants in stream sediments pose a significant risk to aquatic fauna in urban streams.

Contaminant concentrations may vary greatly within, and between, streams as a result of a range of factors including variable stormwater inputs, land uses (eg roading, stormwater treatment, type and density of buildings), catchment history, and sediment texture.

Trends in stream sediment contamination over time are unknown because there are currently no time-series monitoring data. A trend monitoring programme in Waitakere City was initiated in 2003/4, with a view to determining changes over time in response to catchment management and urban development.

6.3 Impacts on stream ecology

Stream ecology is a direct measure of the health (or "biological quality") of streams, integrating the effects of the multiple physical, chemical, and hydrological changes that accompany urbanisation. Assessing stream health usually involves measuring a range of key ecological indicators and related physical variables including:

- The types and numbers of macroinvertebrates (eg snails, worms, crustaceans, fly larvae) that live in the stream bottom sediments, on woody debris, or on macrophytes (plants).
- Fish populations.
- Species and densities of aquatic plants (macrophytes and algae).
- Stream bed characteristics, such as sediment grain size distribution and composition.
- In-stream and riparian habitat quality.

Spot checks of key water quality parameters (eg dissolved oxygen, pH, and temperature) are also often included in ecological monitoring programmes to more fully characterise stream health and to investigate factors that may help explain the results.

The aquatic fauna of urban streams is often characterised by communities dominated by organisms that are tolerant to stressful conditions, including high- and low-flows, high temperatures, low dissolved oxygen concentrations, sedimentation and scouring, and the presence of chemical contaminants.

Macroinvertebrates are commonly used as ecological indicators, and macroinvertebrate communities in urban streams are often very impoverished when compared with forested, and to a lesser degree, pastoral streams. There is a decline in the range (richness) and density (numbers) of sensitive species such as mayfly, stonefly and caddisfly (ie Ephemeroptera, Plecoptera, Trichoptera – the "EPT" group), and an increase in densities of more pollution-tolerant species such as oligochaetes (worms), chironomids (midges) and snails. Low abundance of EPT taxa generally indicates poor stream health.

What was known in 1995

In 1995, there were relatively few published studies documenting the ecological health of Auckland's streams. However, it was accepted that the ecology of Auckland's urban streams was likely to be impacted in similar ways to that found in overseas studies, in particular the decline in sensitive macroinvertebrates.

Advances since 1995

Since 1995, urban stream ecology has been widely studied, in Auckland and in other NZ urban centres (eg Suren et al. 1998; Suren 2000, 2001). As summarised in the following sections, studies have almost universally found relatively poor ecological health in Auckland's urban streams, resulting from the combination of degraded habitat, modified flow regimes, and poor water quality that is prevalent (to varying degrees) in many streams. Change inflow regime is thought to be the major factor affecting urban stream ecology, because flow affects many aspects of habitat and water quality (Elliot et al. 2004).

Studies undertaken since 1995 that have improved the understanding of the ecological effects of urbanisation in Auckland streams include:

- Analysis of the effects of land use on stream ecology at a regional scale (ARC 2004f and g; ARC 2005f; Moore 2003; EVA et al. 2003a).
- Several major surveys that have detailed the ecological health of a large number of streams in Auckland, North Shore, and Waitakere cities (Wilding 1996 and 1999; ACC/Metrowater 2001; Suren 2001; ARC 1998b &1999c; KMA 2000, NSCC 2004b, EVA et al. 2004c).
- Stream classification studies, which have grouped streams in various ways in order to characterise their condition, establish realistic management objectives for various key types (or classes) of urban streams (Webster et al. 2005, Allibone et al. 2001, EVA et al. 2003a), and produce frameworks for stream assessment and management (ARC 2004a).
- Analysis of relationships between environmental variables and stream invertebrate communities from urban centres throughout NZ, which (among other outcomes) produced protocols for assessing urban stream habitat quality (Suren et al. 1998).
- Developing protocols for the collection and interpretation of macroinvertebrate data in Auckland's "soft-bottomed" streams (Maxted et al. 2003; ARC 2004g).
- Exploration of the relationships between land use "pressure" indicators (eg catchment impervious area) and environmental "state" (eg ecological health, water and sediment quality) in an effort to better understand the effects of urban land use on streams and how better to monitor these effects (eg EVA et al. 2003a, 2005a and b).
- The production of a comprehensive guide to quantifying the effects of urbanisation on stream flows and channel modification, and the resultant effects on in-stream habitat and aquatic life (Elliot et al. 2004).

These studies are summarised in the following sections.

6.3.1 Broad-scale effects of land use on stream ecology

An overall narrative assessment of the ecological state of Auckland's streams, which encapsulates the key features of urban stream ecology, was given by EVA et al. (2003a; reproduced below):

- A particular feature of many of Auckland's urban streams is their short length and low-lying nature of the catchments. Compared with streams in non-urban areas, many Auckland urban streams are narrower, have warmer temperatures, and are slower flowing. This is particularly apparent in Auckland City and North Shore streams.
- Habitat modification occurs with increasing urbanisation. Urban streams often have reinforced banks and channels, a loss of suitable riparian vegetation and cover, and an increase in aquatic plant growth.
- A loss of pollution and habitat-sensitive macroinvertebrates occurs with increasing urbanisation. In particular EPT (Ephemeroptera (mayflies), Trichoptera (caddisflies) and Plecoptera (stoneflies) taxa are lost from streams with increasing habitat loss, warmer temperatures, and modifications to flow and habitat.
- Increase in pollution and habitat-tolerant macroinvertebrate taxa. In particular grazing snails and midge larvae are more abundant in urban streams with increasing habitat homogeneity, warmer temperatures, and modifications to flow and habitat.
- At extreme stream habitat and channel modifications (eg concrete-lined channels), the macroinvertebrate fauna is dominated by only a very few taxa, but often in very large numbers.
- Streams with higher MCI scores are generally greater in length, have greater catchment areas, arise at higher elevations, and distance from the sea is greater. This reflects the lower degree of urbanisation in the upper parts of these larger catchments (eg some Waitakere streams), and where some natural stream and catchment elements still dominate.
- In contrast, streams of lower length, arising in low-lying catchments (eg central Auckland) or with steeply descending upper catchments (eg eastern North Shore streams) and are narrower, are characterised by low diversity and low MCI scores. These conditions imply degraded water quality and highly modified habitat. Examination of remaining unmodified but similar streams in the Auckland region (reference sites), suggest that in the absence of urbanisation, well vegetated, low-lying natural soft bottom streams do retain a high diversity and presence of pollution and habitat-sensitive macroinvertebrate taxa.
- Fish diversity in Auckland streams varies. There is a general absence of galaxid fish in central Auckland city urban streams, while banded kokopu have been recorded in North Shore, Manukau City and Waitakere urban streams.
- Non-natural barriers to fish migration can be indicative of increasing urbanisation. However, current technology and retrofitting of culverts in particular has increased the fish passage potential in many urban streams.

ARC regional surveys

Macroinvertebrate community composition data collected from 41 sites throughout the region[24] (ARC 2004g, 2005f) found biological quality is related to catchment land use, and can be ordered from best to worse as follows:

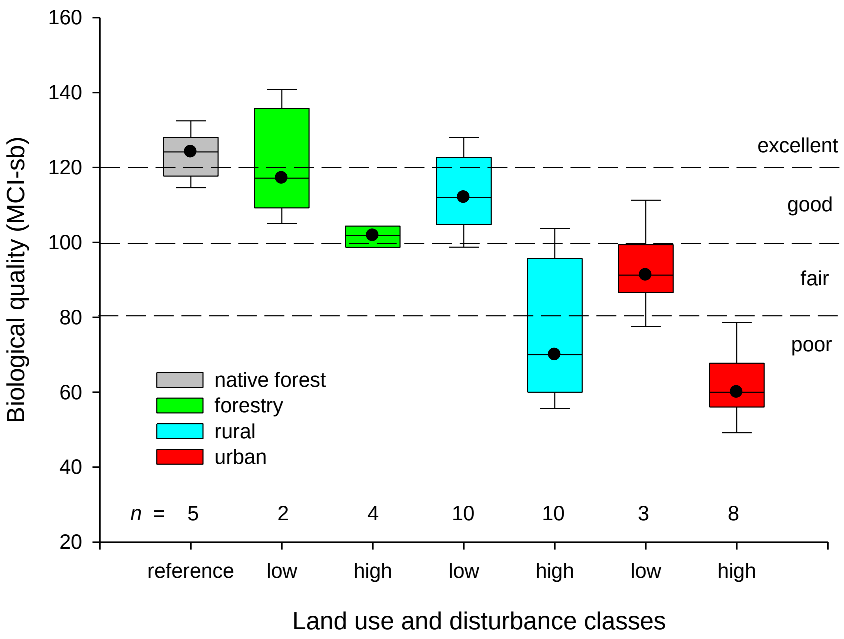
Native bush (best) > exotic forest > rural > urban (worst).

Within these land uses, biological quality was found to be related to the level of land disturbance, with higher quality in streams surrounded by less disturbed land. For urban land, "low disturbance" was defined as having 10–20 per cent of the area in urban use, while "high disturbance" had >40 per cent. Results are summarised in Figure 26 (ARC 2005f).

[24] The majority of Auckland's streams affected by human activities are "soft-bottomed" (SB) – ie in soft rock (clay or sand) geology. A biological quality index, the "MCI-SB" (ARC 2004g), was developed to provide a reliable indicator of the state of these streams, so they could be graded into quality classes (excellent to poor) as shown in Figure 26.

Figure 26

Biological quality of Auckland's "soft-bottomed" streams as a function of land use, and disturbance (ARC 2005f). The "MCI-sb" is an index of macroinvertebrate community health derived for Auckland's soft-bottomed streams (ARC 2004g).



Urban sites with a high degree of disturbance had poor biological quality, while urban areas with a low degree of disturbance were rated as only "fair", indicating that impacts occurred in the early stages of urban development (<10–25 per cent urban land use). Few urban sites were "good" and none were rated as "excellent".

Biological quality was strongly associated with the quality of physical habitat (including stream channel, banks, and riparian zone) in the immediate vicinity of the sites. However, some urban streams had good quality habitat but poor biological quality. This was attributed to the effect of poor water quality. These observations are consistent with the view that urban stream ecology is affected by multiple stressors, in particular habitat modification and degraded water quality.

Also apparent from the regional data is that there is a considerable amount of variability in biological quality within most land use classes, and some overlap between different land uses. Some urban streams have better biological quality than rural (and even some high disturbance forestry) streams.

One management implication of the regional data analysis is that urban stream health could be substantially enhanced with improved riparian management, but full protection of stream ecological health would not necessarily be achieved at all sites due to the effects of poor water quality.

The macroinvertebrate fauna of Auckland's lowland urban and rural streams was also compared by Moore (2003). While the urban and rural surveys used slightly different monitoring methodologies, it was found that the condition of the aquatic communities in the rural streams was generally better than those at the urban stream sites. It was noted that while some urban streams have good habitat conditions, and healthy invertebrate communities with some sensitive taxa being relatively abundant, urban stream invertebrate communities tended to have lower taxonomic richness, lower number of EPT taxa, and lower macroinvertebrate community index (MCI) values than rural streams. This is consistent with the ARC's regional analysis summarised in Figure 27.

Generally therefore, Auckland's urban streams have markedly poorer ecological quality than streams in other land uses.

NZ nationwide survey

Suren et al. (1998) conducted a survey of 59 urban stream sites in nine New Zealand urban centres, including several in Auckland. The study investigated relationships between invertebrate communities and environmental variables at varying scales. This work produced the "Urban Stream Habitat Assessment" (USHA) protocol. Statistical analysis of the data showed that, at a national scale, six macro-scale physical variables (eg geology, rainfall, topography, temperature, stream order and source) were the dominant influences on invertebrate community composition and limit the ultimate biological potential of urban streams. These macro-scale variables were used to group the streams into three types. Within each of these groups, smaller scale "meso-" and "micro-" scale habitat variables (eg riparian vegetation, stream bed type, bank stability etc) that strongly influenced invertebrate communities were identified and incorporated into a habitat scoring system.

The most numerous animals collected in the survey were worms, the snail *Potamopyrgus antipodarum*, and midges. Almost no mayfly, caddisfly or stonefly (EPT) taxa were found above total densities of 1 per cent. The USHA survey also found that invertebrate communities in peri-urban sites (ie on the fringe between urban and rural areas) were also highly degraded because of poor habitat and water quality caused by upstream agricultural land uses (Suren & Elliot 2004, Suren 2000). In this respect the survey data were similar to the results from water quality monitoring (Section 6.1) and from the regional monitoring described above – the stream environment is often degraded by upstream land uses by the time the urban area is reached, and urbanisation further impacts upon this degraded state. The poor biological state of the peri-urban stream sites prevented the survey data from showing a significant relationship between catchment imperviousness and biological health, which is commonly reported for urban streams (eg Herald 2003, Allibone et al. 2001).

6.3.2 Local urban stream ecology surveys

Surveys of stream ecology have been conducted in a large number of sites in Auckland, North Shore, and Waitakere cities (eg Wilding 1996 and 1999; ACC/Metrowater 2001; Suren 2001; ARC 1998b; KMA 2000, EVA et al. 2004c, and NSCC 2004b). A brief summary of these studies is appended to this chapter (Section A6.3). Generally these studies show effects that are consistent with the regional monitoring described above.

6.3.3 Stream classification and management objectives

Managing and restoring urban streams requires an understanding of the biological values that can realistically be achieved given their physical and biological characteristics – ie the stream's biological potential.

Allibone et al. (2001) undertook a comprehensive study of the ecological condition of streams throughout the Auckland region to develop a classification process for the streams so that appropriate management objectives could be applied to Auckland's urban streams.

Initial workshop sessions identified twelve potential biological objectives for fish and invertebrates in Auckland's urban streams. These ranged from objectives that were thought to be easily achievable (eg sustain shortfin eels) to ones that required good to high quality instream conditions (eg retain six EPT species).

Field surveys of 64 urban Auckland streams were carried out to assess the potential for each of the biology objectives to be achieved. The surveys also collected data on a wide range of habitat parameters for each stream. Eight fish species and 78 invertebrate taxa were found during the survey. The fish and invertebrate communities in each stream were analysed in conjunction with the habitat data to determine if different stream types with distinct stream communities could be recognised. Cluster and correlation analyses indicated that fish community structure was strongly related to stream slopes, stream size parameters and riparian vegetation parameters. The strong relationship between stream slope parameters and fish community was attributed to the varying ability of different migratory fish species to penetrate inland up different stream gradients. Water quality appeared to be more important in determining the invertebrate community structure. Habitat variables such as streambed substrate and habitat diversity parameters showed little relation to fish and invertebrate communities.

The stream classification developed from the survey recognised eight stream types: 1. Estuarine area, 2. Concrete channels, 3. Highly disturbed streams, 4. Low gradient coastal streams, 5. Restricted fish passage streams, 6. Steep forested streams, 7. Degraded water quality streams, and 8. Ephemeral streams.

Each of these stream types had biological values associated with it, which could realistically be expected to be maintained, enhanced, or restored if absent. Some stream types met a single biological objective whereas others fulfilled up to six different biological objectives. For each stream type management issues were identified that centred around the provision of the appropriate instream cover for fish and invertebrates, improving fish passage for migratory native fish and reducing water quality problems. A summary of the objectives for each stream class is given in NIWA (2005).

The recommendations from this study formed the basis for the stream management framework incorporated into the Auckland Regional Plan; Air, Land, and Water (ARC 2004h), which is detailed in ARC (2004a). The ARC management framework focuses on "Category 1" (perennial) streams. Management approaches for ephemeral streams and wetlands ("Category 2" streams) are still being reviewed by ARC.

Six classes of Category 1 stream reaches have been defined by ARC, depending on catchment variables (eg location, level of catchment development as defined by imperviousness – see Section 6.6), and stream reach variables (eg riparian condition, degree and type of channel modification). Management options and priorities for each type of stream reach are then given (Figure 27 and Table 9).

Figure 27

ARC's procedure for assigning stream reach types for urban stream management (ARC 2004a).

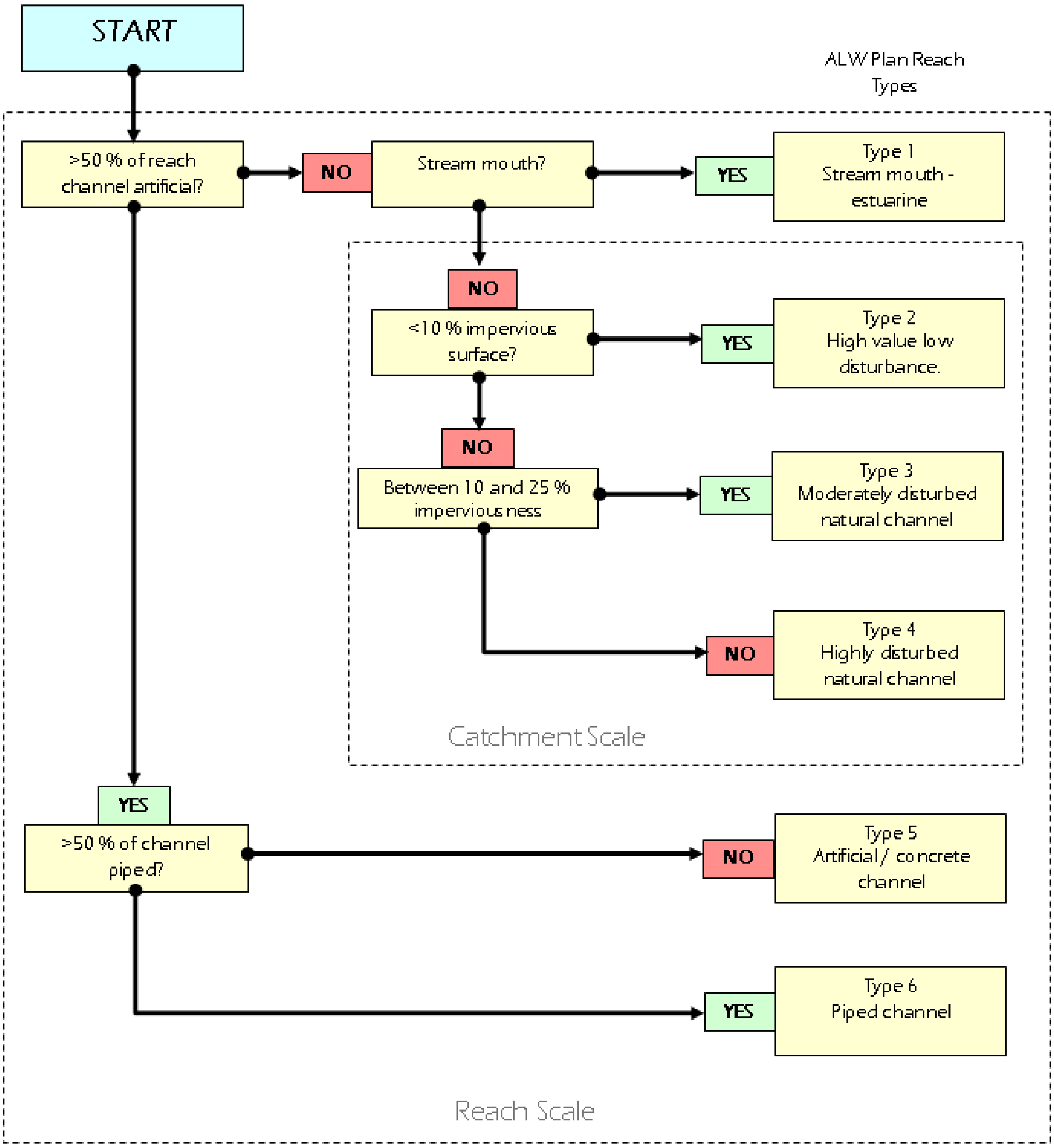


Table 9

ARC management priorities for urban streams, with relative emphasis given as high (H), medium (M), or low (L); not applicable (–). From ARC (2004a).

Urban rivers and stream management priorities	Reach classification type					
	1	2	3	4	5	6
1. Maintain or enhance amenity values (aesthetics, recreation, cultural/community, economic)	L	L	H	H	–	–
2. Maintain or enhance instream values	H	H	H	M	–	–
3. Maintain or enhance public access to and along rivers, lakes and wetlands	L	L	H	H	–	–
4. Maintain high water quality by avoiding, remedying or mitigating contaminant inputs	H	H	H	H	H	H
5. Improve degraded water quality by avoiding remedying or mitigating the adverse effects of contaminant inputs	–	–	H	H	H	H
6. Stabilise and protect stream banks from erosion	L	L	M	M	–	–
7. Restore the pre-development hydrology to the fullest practicable extent	–	M	H	H	–	–
8. Reduce instream temperatures and improve reach connectivity by planting or enhancing riparian vegetation	H	H	H	H	H	–
9. Protect and enhance inanga spawning areas	H	–	–	–	–	–
10. Maintain or enhance fish passage for appropriate species by avoiding, remedying or mitigating effects of artificial barriers	H	H	H	H	H	H
11. Minimise flood risk to humans and property through the application of best management practices	–	L	M	M	H	H
12. Protect human health	H	H	H	H	H	H

North Shore City has also developed a stream classification system. In 2001, thirty North Shore City stream sites were visited in order to gain an understanding of the ecological characteristics, habitat and catchment factors that influence biotic communities. From this work, stream groups based on aquatic macroinvertebrate communities were identified, and a methodology for classifying other streams was developed. Key ecological characteristics (habitat, riparian vegetation and bank erosion) collected during the stream walk, and impervious area and catchment vegetation information, were used to classify stream sections into six stream groups. The methodologies used are given in Kokupu Connection Reports KC1 (NSCC 2004a; detailed methodology) and KC70 (NSCC 2003; overall stream categorisation process).

- Group 1 – Stream mouth: the area identified as the tidal/freshwater interface.
- Group 2 – High value low disturbance: these streams can be typified as flowing through rural or forested areas and have the highest water and habitat quality.
- Group 3a – Urban semi-modified: these streams can be generally described as those that flow through urban areas where there is limited bed scouring and sections of the banks are reinforced.
- Group 3b – Urban modified: these can be typified as streams that flow through urban land use where there is limited bed scouring, sections of the banks are reinforced and there is significant bank erosion or potential for bank erosion.
- Group 4 – Concrete channels sections: where the channel bottom comprises >50 per cent artificial material.
- Group 5 – Piped sections: any piped sections within the stream network.

Another method of stream classification based on ecological quality was given by EVA et al. (2003a). They used five indicators (MCI, EPT, dominant macroinvertebrates, habitat quality, and numbers of fish taxa) to group streams into five ecological condition classes (Table 10):

Table 10
Stream classification based on ecological condition of Auckland urban streams (EVA et al. 2003a).

Condition	Quality	MCI	EPT	Dominance	Habitat	Fish
Natural/low modification	1 Good	>110	>6	EPT	High	>5 native taxa
Natural semi-modified	2 Moderate-good	90–110	4–5	Snails, EPT	High-moderate	3–4 native taxa
Urban semi-modified	3 Moderate	75–90	2–3	Snails	Moderate	Galaxiid + 2 native taxa
Urban modified	4 Poor	60–75	1	Snails, midges	Moderate-low	<2, galaxiid absent
Urban high modification	5 Very poor	<60	0	Midges	Low	Eels present but no other native fish

Note: Quality ranking 1 = high quality/low modification, 5 = low quality/high modification.

A further urban stream classification system for Auckland City streams was developed by NIWA (Webster et al. 2005, summarised in NIWA 2005), based on three key values: drainage, aquatic habitat, and public amenity. Five components were incorporated into the scheme (water quality, physical habitat, riparian quality, aesthetic/recreation, and hydrology) to produce scores for each of the three key values. These scores then reflected the relative importance of drainage, habitat, and amenity value for each stream reach.

Data from 48 stream reaches were classified, and indicated that the primary function for 77 per cent of the Auckland City streams was drainage. However, reaches with moderate to high habitat value and high public amenity value were also identified. Maps in NIWA (2005) show the locations and classifications of these streams.

Stream restoration

While the quality of even the best urban streams is poorer than those in native bush, exotic forest, and many rural catchments, urban streams provide important ecological functions, and in many cases their condition can be improved. The stream categorisation process (Allibone et al. 2001, ARC 2004a) provides the tools for sensible management and restoration options for Auckland streams.

A three-step process is suggested (ARC 2004a):

1. Map resources.
2. For new development, consider using low impact design, especially in respect to erosion control, fish barriers, culverts, connectiveness to the flood plain and riparian zone etc.
3. For existing development, consider restoration priorities and actions, which depend on the type of stream reach under investigation (see Table 9 above).

In summary, a number of studies have classified Auckland's streams into groups based on various combinations of ecology, water quality, habitat, and physical characteristics, in order to rank overall stream condition and determine realistic stream management objectives and restoration options. The ARC has adopted a classification scheme into the regional planning framework to improve urban stream management and restoration.

6.3.4 Assessing urban impacts on stream habitat and benthic ecology

A recent major advance has been in the area of "metrics", or biological quality indices, for the interpretation of macroinvertebrate information from soft-bottomed streams, which dominate in Auckland (ARC 2004g). Until very recently, assessment depended on the use of methods that work well for hard-bottomed streams (eg cobbles), but less well for soft-bottomed streams. These metrics simplify the detailed data describing the macroinvertebrate communities at a different stream sites to simple "indices", the most common being the Macroinvertebrate Community Index or MCI.

The new "soft-bottom" metrics provide greater discrimination – ie there is a greater range of index vales from pristine sites to severely degraded sites – which is helpful when teasing out stream categories and restoration options, as well as in determining the different effects of stormwater. The new metrics are also more likely to correctly assign a new site to the correct ecological condition (excellent, good, fair, poor), so they provide more confidence in their application.

Application of these metrics to Auckland shows that to maintain (or restore) excellent biological communities, it is essential to have both high habitat quality and low catchment development. In fully urbanised catchments, it may not be possible to achieve conditions comparable to reference conditions unless poor water quality is improved, although substantial improvements in the biota can be achieved by improving habitat quality.

The use of the "soft-bottom" macroinvertebrate community index (MCI-sb) in demonstrating the effects of land use on stream ecology was shown previously in Figure 26.

6.3.5 Impacts on fish

Compared with studies of macroinvertebrates, water quality, and stream habitat, relatively few studies specifically addressing stormwater impacts on fish in urban streams have been conducted. However, use of the NZ Fisheries Data Base and local surveys (eg ARC 1998b, Allibone et al. 2001, KMA 2000) has revealed that there is considerable fish diversity within Auckland's urban streams. Shortfin and longfin eels are common, while common bully and inanga are also reasonably widespread. Even relatively uncommon fish such as banded and giant kokupu can be found in urban streams, provided habitat (eg riparian and in-stream cover) is suitable (Suren & Elliot 2004).

Of the 35 indigenous freshwater species currently recognised in New Zealand, 18 are "diadromous" and undergo migrations between fresh and saltwater as a necessary part of their life cycle. Apart from the degradation of in-stream habitat for adult fish, one of the most significant causes of the decline in freshwater fish populations in New Zealand is the construction of structures such as dams and culverts that prevent fish from accessing otherwise suitable habitats.

Fifteen indigenous, and eight introduced, fish species have been recorded in the Auckland region (ARC 2000a). Thirteen of these 15 indigenous species are diadromous and fish migration barriers are therefore expected to have a major influence on fish distribution in the Auckland region. Potential migration barriers such as waterfalls, rapids, chutes and debris jams are natural, however the majority of instream obstructions in urban areas are anthropogenic. These include badly positioned or undersized culverts, fords, dams and diversion structures, weirs (including flow measuring weirs), diversion channels, bed erosion control devices, and streambed modifications.

Fish barriers

Guidelines for fish passage have been developed nationally (Boubée, et al. 1999) and for the Auckland region (ARC 2000a). The guidelines identify height (vertical differential between streambed and structure outlet), water velocity and turbulence, water depth, channel length, light, and climbing medium as important variables that must be managed to ensure successful fish passage.

Figure 28
Waterfall into stream.

Barriers to free upstream passage include any structures, natural or constructed, that cause the water to "free fall", often referred to as "drop structures". Potential barriers include weirs, dams, and culverts. Even a drop of as little as 10–20 mm may be a barrier because most native fish and invertebrates cannot jump out of water to traverse such obstacles. Some fish species have the ability to climb steep wetted surfaces (eg eels, banded kokopu) while others cannot (inanga, giant kokopu).

Other factors that adversely affect fish passage include velocity and depth. Some species have adapted to survival in stream reaches isolated above obstacles such as waterfalls, but are more limited in their ability to expand their populations and recolonise after pollution events. Consequently, the greatest diversity and abundance of native fish are found at low elevations where there is direct access to the sea (Allibone et al. 2001).

Figure 29
Stormwater outlet pipes.

Pipes and culverts are common barriers in the urban environment, and can serve to prevent the movement of fish to suitable habitats in upper catchments. A smooth concrete invert does not provide the protection needed by fish and aquatic biota for breeding and resting. Flows in pipes and culverts often have higher velocities than natural channels, and commonly exceed the swimming ability of many native fish. Channel length can also have adverse effects where there are no resting areas (low velocity zones). Fish passage can thus be restricted through a combination of slope, depth, velocity, and culvert length.

Observations made during the North Shore City Stream Walk Programme suggest that some species can negotiate around what were previously seen as barriers. For example, of 33 culverts inspected in North Shore urban streams, only 25 per cent had potential barriers to fish passage (KMA 2001). Report KC32 Assessment of Constraints to Fish Passage (NSCC 2004c) discusses the issues concerning fish barriers in more detail.

Studies conducted in the Waitakere Ranges (ARC 2005g) found that even in non-urban areas, many structures are unsuitable for fish passage. Fifty structures (mostly culverts) were located and evaluated, of which 76 per cent were barriers to fish passage under most flow conditions. In contrast only 12 per cent allowed unrestricted fish passage. Fish barriers affected approximately 26 per cent of the area within the Auckland water supply catchment. This equated to nearly 30 km of high quality freshwater fish habitat.

Clearly fish barriers are having a major ecological impact in Auckland streams.

Habitat preferences

The importance of habitat for fish was studied in Auckland urban streams (Allibone et al. 2001). Fish community structure was strongly related to stream slopes, stream size parameters and riparian vegetation parameters (canopy cover and ground cover type). The strong relationship between stream slope parameters and fish community was attributed to the varying ability of different migratory fish species to penetrate inland up different stream gradients. Water quality appeared to be more important in determining the invertebrate community structure than the fish community and, unlike for macroinvertebrates, there was a poor relationship with per cent imperviousness. Habitat variables such as streambed substrate and habitat diversity parameters showed little relation to fish communities (Table 11). It is, however, notable that no fish were encountered in streams with concrete channels.

Table 11

Habitat preferences for fish in Auckland's urban streams (from Allibone et al. 2001).

Species	Substrate	Permanent or intermittent flow	Type of instream cover	Riparian shade	Migration ability
Shortfin eel	No preference	No preference	No preference	No preference	Very good
Lonfin eel	No preference	Permanent	No preference	No preference	Very good
Common bully	Little suspendable fines	Permanent	No preference	No preference	Limited
Banded kokopu	No preference	No preference	No preference	Good shade required	Very good
Inanga	No preference	Permanent	No preference	No preference	Limited
Redfin bully	Cobble	Permanent	Substrate/wood debris	No preference	Moderate
Mosquito fish	Shallow water macrpohytes	No preference	No preference	No preference	Very limited

Effects of land use on fish

Fish data examined as part of a study of Waitakere City Streams (KMA 2000) revealed a decline in the average number of native fish species present as land use changed from native forest through to urban. As mentioned above, fish are found in many Auckland streams and their distribution appears to be primarily a function of the stream "connectivity" and barriers to migration.

Fish communities in the Waitakere Project Twin Streams monitoring project (EVA et al. 2004c, 2005b) generally contained low to moderate species diversity. In general, fish species diversity was greatest at mid-catchment sites, with only the relatively strong climbing longfin and shortfin eel species, banded kokopu and redfin bullies recorded at upper catchment sites. This, along with the presence of fish not known for their climbing abilities such as inanga and common bullies recorded only at mid-catchment sites indicates that there are likely to be no instream barriers to migration in the lower catchment but the possibility exists that there may be barriers to migration in the upper catchments. However, the streams in the upper reaches were very small, which would also severely limit habitat.

Analysis of data from the Project Twin Streams survey showed that fish communities were affected by urbanisation differently from macroinvertebrates. While invertebrate communities were clearly related to land use (eg Figure 26), the best predictor of fish taxa richness was a local habitat factor – the proportion of streambed comprised of cobbles, which is a measure of instream fish cover. This contrasts with the survey of 64 streams in and around the Auckland metropolitan area (Allibone et al. 2001), which found that local habitat conditions (including substrate composition, instream cover, and shade) were relatively unimportant factors in determining fish community structure and abundance.

Overall therefore, fish populations are potentially affected by all the physical, chemical, and hydrological stressors introduced by urbanisation. However, current evidence indicates that the major factor influencing the distribution of fish in urban streams is barrier to migration. Local habitat (eg cover, stream substrate, shade) may also be a factor in some areas. Until these key factors are remedied (by restoration of inadequate structures, riparian vegetation, and in-stream habitat), the more subtle effects of other stormwater stressors are unlikely to be evident.

6.3.6 Effects of stormwater ponds

Stormwater treatment ponds and wetlands are commonly used methods for mitigating the adverse effects of stormwater discharges on streams. While they can be effective for reducing sediment and sediment-associated contaminant loads to receiving waters, impacts on downstream water quality and ecology can arise from their use. Research in the Auckland region has found poor water quality conditions in rural and bush ponds (including elevated temperatures and depressed dissolved oxygen), and adverse effects on downstream water quality extended for hundreds of metres below the pond outlets. Macroinvertebrate community composition was also adversely affected (Maxted et al., 2005).

Online ponds (ie ponds sited in the normal stream flow path) are also often a barrier to fish passage. Efforts to control the adverse effects of stormwater on rivers, estuaries, and harbours should endeavour to also protect and enhance the functions, values, and uses of streams within their catchments. The ARC regional plan therefore discourages the placement of ponds, including stormwater treatment ponds, within perennial stream channels and floodplains and encourages their location offline or on ephemeral streams.

6.4 Lakes

Lake Pupuke is the only urban freshwater lake or pond that has been monitored. Its watershed is small and completely urban, although park forms a large proportion of the land use. The lake is monitored for appearance, clarity and enrichment (ARC 2005h). Water quality in Lake Pupuke is stable and has not changed from 1992 to March 2005. The Secchi disk record indicates that while improvements in water clarity and hence water quality from 1966/67 has been significant through to the late 1990s, recent data show these improvements have not continued. This suggests that improvement in lake condition following nutrient load reduction (diversion of domestic sewage and agricultural waste) has stabilised. Further improvement may be limited by remnant point source discharges (stormwater), diffuse inputs (fertiliser, faecal matter) or by internal nutrient recycling, particularly phosphorus (ARC 2005h).

6.5 Toxicity assessment in Auckland's urban streams

Stormwater contains a wide range of potential toxicants including heavy metals (eg Zn, Cu, Pb), PAH, and other organics (eg pesticides, hydrocarbons). These toxicants may be present dissolved in the water, or attached to suspended solids, which may be deposited in streambed sediments and in thin slimy biofilms that coat aquatic plants and in-stream debris. If concentrations are high enough, the toxicants may induce short-term acute lethal effects on stream aquatic life, or more subtle chronic effects through longer-term exposure to lower levels of contamination.

During storm flows a portion of the contaminated sediments are flushed into the downstream estuaries and marine areas where they settle out (see Chapter 7). As the flow recedes, remaining suspended sediment settles onto the stream bed, especially in pools, backwaters and reaches with low-flow velocities.

The toxicity of contaminants in urban streams is usually assessed indirectly by comparing chemical concentrations in water and sediments with ecosystem protection guidelines (see Chapter 5 and Sections 6.1 and 6.2). Relatively little work has been published on direct assessment of toxicity in urban streams in Auckland, partly because of the difficulty in conducting the in-situ toxicity tests in urban streams and separating the effects of contaminants from other stressors such as high storm flows, suspended sediments, high temperature, and low dissolved oxygen that may prevail in varying degrees during the tests. Toxicity studies have been carried out in Auckland on water and sediments collected from stormwater systems and through in situ deployment of stream organisms., The toxicity of urban stormwater and detention pond waters and sediments has been investigated (Nipper et al. 1995, Hickey et al. 1997). The tests included:

- Exposing freshwater mussels in situ and measuring their condition and metal uptake.
- 48-hour acute mortality and 14 day chronic mortality and reproduction using *Daphnia magna*.
- 96-hour reproduction inhibition test using the alga *Senastrum capricornutum*.
- 15-minute bioluminescence reduction test (EC50) using the Microtox system.
- Chemical analyses of urban contaminants, including Cd, Cu, Pb, Zn, and PAH.

The tests showed that acute and chronic toxicity occurred on some occasions and locations but not others, and no clear picture emerged. Enhanced growth of algae and reproduction of daphnia was sometimes measured during base flow testing, possibly because of nutrients in the water. Toxicity was sometimes higher and sometimes lower than expected on the basis of chemical concentrations, possibly indicating low bioavailability on occasions and the presence of other unmeasured toxicants on others. The results are fairly consistent with overseas studies that show that stormwater has variable acute and chronic toxicity and that the causative agent(s) is difficult to determine, which underscores the complexity and variability of urban stormwater quality.

Hickey (1998) provides a review of tests utilised in New Zealand to assess the toxicity of freshwater sediments. One of the most commonly tests used is the amphipod *Chaetocorophium cf. lucasi* acute survival and chronic growth test. Other tests are survival and burying behaviour of the clam *Sphaerium novaezelandiae* and survival, contaminant uptake and reproduction of the worm *Lumbricus variegatus*.

Sediment toxicity tests have been performed on sediments collected from the inlet and outlet of the stormwater ponds and the sand filter at the UNITEC carpark in Mt Albert (Nipper et al. 1995) using these test procedures. Problems with test animal survival in reference sediments compromised interpretation of some of the tests. The amphipod did not survive in the pond sediments, but was able to live in the sand filter sediment, possibly because of the high proportion of original clean sand in the filter. The pond sediment also appeared to be toxic to worm survival and reproduction (Nipper et al. 1995, Hickey 1999).

Timperley et al. (2001) discussed possible issues associated with the presence of high concentrations of contaminants in the fine particulate material in road run-off. Of particular concern was the fine particulate matter that might become associated with biofilms in streams. Biofilms such as filamentous algae or diatoms are common surfaces in streams and form an important food source for many invertebrate grazers in streams. Concentrations of copper, lead and zinc in these films can reach concentrations similar to those observed in the suspended sediments present in streams. The concentrations are higher than in algal films that do not contain fine particulate material. Timperley et al. (2001) showed that *Potamopyrgus* (a common freshwater grazing snail in New Zealand streams) ingests fine particulate material associated with biofilms when grazing and that 99 per cent of the particles (by number) were ≤5 µm in size. Studies conducted in Christchurch streams suggest that metal-contaminated biofilms in urban streams may play a significant role in depletion of sensitive macroinvertebrates (Suren & Elliot 2004).

Golding (2002) caged adult and juvenile snails (*Potamopyrgus antipodarum*) at multiple sites in two Auckland urban streams and compared their survival over a 28-day period with a native bush reference stream. Survival in the urban streams was lower than in the reference stream, and juveniles were more sensitive than adults. 100 per cent juvenile mortality was recorded at half the urban sites, and up to 90 per cent mortality of adults occurred at some sites. The toxicity was attributed to elevated metals' concentrations, as other potential stressors such as dissolved oxygen, temperature, and pH were within the tolerance limits of this organism. *Potamopyrgus* is generally accepted to be a fairly resilient animal, and therefore the marked impacts measured in this experiment indicate that toxicity may be a major factor affecting urban stream invertebrate communities.

On the basis of these studies, it is reasonable to assume that chemical toxicity will occur in streams receiving substantial amounts of urban run-off, especially those in industrial, commercial, or older residential areas that have little stormwater treatment and large areas of exposed metallic and impervious surfaces (and hence higher loadings of metals, and possibly, PAH).

6.6 Land use pressure indicators and integrated monitoring

Effective management of urban stormwater requires an understanding of the relationship between land use pressures and the environmental state (or health) of the receiving waters. One component of this is developing a suite of land use pressure indicators, which can be used to predict how changes in land use or human activities in urban areas may impact the receiving environment. The results from the analysis of "pressure–state" relationships can then be used to develop appropriate management "responses" to mitigate or prevent undesirable changes (eg catchment plans, regulations, community activities, engineering works etc).

This is the so-called "Pressure State Response" (PSR) framework for environmental management and reporting (MFE 1997). It is based on the concept that human activities exert pressures on the environment, changing the quality and quantity of natural resources. These changes alter the state of the environment. The human responses to these changes include organised behaviour, which aims to reduce, prevent or mitigate effects on the environment.

Potentially useful pressure indicators are numerous, and include:

- The area (or percentage) of impervious surface in a catchment (or upstream of a stream reach or monitoring site).
- Numbers and sizes of stormwater outlets (per unit of catchment area) discharging to streams.
- Age and condition of stormwater and sewer systems.
- Numbers and volumes of sewer over flows (per unit of catchment area).

- Numbers and types of contaminated sites in a catchment.
- Traffic densities, or vehicle counts on catchment roads.
- Stormwater treatment devices – types, efficiencies, area treated etc.

A first attempt at using the PSR framework to report on the state of Auckland's urban streams was conducted by EVA et al. (2003a), and further work has since been conducted as part of Waitakere City Council's Project Twin Streams monitoring programme (EVA et al. 2003/6). Application of PSR to urban catchments is relatively new, and therefore much of the data obtained so far is preliminary.

The following briefly outlines some of the key results from studies linking some pressure indicators with stream health in the Auckland region. Many of the above indicators are likely to be correlated, as they reflect the general intensification of pressure as catchment urbanisation proceeds. This was confirmed by analysis of data in the Waitakere City area (EVA et al. 2005a), which found significant correlations between five indicators (imperviousness, numbers of wastewater overflows and stormwater outlets, traffic densities, and riparian margins). Imperviousness was found to be strongly correlated with most other pressure indicators, and it therefore provides a powerful general measure of potential stormwater run-off impacts. It is discussed further below.

Imperviousness

As urbanisation proceeds, there is an increase in impervious area made up from roads, parking lots, roofs etc, all of which contribute to increased volumes of contaminated stormwater run-off, more rapid increase in stream flows and higher peak flows during storms, and lower low flows in catchment streams. "Imperviousness" has been widely used as an effective indicator of urban land use pressure on aquatic receiving environments. Because imperviousness is relatively easily measured, and it affects several factors that may have adverse effects on the aquatic biota in urban streams (including reduced water quality, altered hydrology, and reduced physical habitat quality), it is a useful tool for predicting the level and type of development that may result in unacceptable impacts in receiving waters.

It appears that biological quality, as indicated by macroinvertebrate community composition (eg MCI, EPT), declines rapidly once a critical threshold for the proportion of catchment developed in urban area is exceeded. Studies in the USA have shown progressive degradation of fisheries and benthic ecology occurring as the urban area is expanded from 5 to 30 per cent of the catchment, and this seems to be the case in NZ as well.

A first study by Wilding (1996) found that streams in the Auckland region lost sensitive EPT macroinvertebrates once the area urbanised and connected to the stream exceeded about 20 to 30 per cent of the total catchment area (Table 12).

Table 12

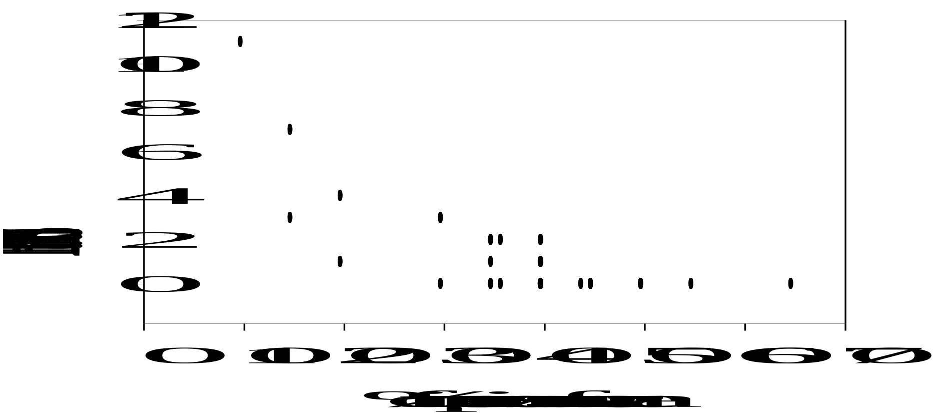
Summary of macroinvertebrate data for Auckland streams (from Wilding, 1996).

Land use	No. of EPT	No. of taxa
Bush	10	23
Bush +7% urban	17	36
Bush + 30% urban not reticulated	13	25
Bush + 30% urban reticulated	5	16
Fully urban + wide bush riparian cover	1–5	8–16
Fully urban + grass	0–1	16–27

The larger-scale study of the ecological conditions in Auckland's urban streams by Allibone et al. (2001) more clearly demonstrates the relationship between imperviousness and condition. Stream quality was highest at <10 per cent imperviousness, declined rapidly between 10–25 per cent, and was consistently poor at greater than about 25–30 per cent (Figure 30). The ARC stream management framework (ARC 2004a) has used <10 per cent to define "Type 2 – High value low disturbance" streams, while stream reaches with 10–25 per cent imperviousness are classed as Type 3 – moderately disturbed".

Figure 30

Bological quality of urban streams (shown here by EPT richness) declines rapidly as the proportion of impervious area in a catchment increases, as shown by data recorded from urban stream sites in Auckland (Allibone et al. 2001).



Similar results were obtained for four urban and peri-urban catchments in Waitakere City (Herald 2003). MCI scores were greater than 100 (indicating high quality macroinvertebrate community composition) when imperviousness was <12 per cent, but fell rapidly to 69–86 at 12–20 per cent imperviousness. Above 20 per cent imperviousness, the MCI values ranged between 60 and 80, and communities were dominated by less tolerant macroinvertebrate taxa. Water quality also declined with increasing imperviousness, as evidenced by decreasing dissolved oxygen and increasing total Zn concentrations.

Further monitoring in Waitakere streams (EVA et al. 2005a and b) has revealed clear patterns of reduced water quality and ecological health associated with increased urban development. Increased catchment imperviousness was associated with increased turbidity, elevated concentrations of heavy metals (dissolved and sediment associated), nitrogen, bacteria, and PAHs, and a reduction in macroinvertebrate community health (a selection of relationships are plotted in Figures 31 and 32). As described previously, there was no relationship between fish populations and imperviousness (or other land use pressure indicators).

One of the key conclusions from the Project Twin Streams study was that stream ecological health was related to the quality of stream habitat for less intensively developed parts of the catchments, but where imperviousness exceeded about 20 per cent the impacts of large-scale catchment disturbance outweighs the benefits of good quality local habitat. Therefore, for areas with intensive development (and poor stormwater treatment), restoration of stream habitat may not markedly improve macroinvertebrate community health. This reinforces the need for effective stormwater treatment and/or source control.

Contrasting with the results of these Auckland studies, a study of urban streams throughout New Zealand found no relationship between catchment imperviousness and stream health (Suren 2000). It was suggested this was because for many of the urban streams studied, the entire catchment was developed, with the streams already degraded by rural land use upstream of the urban boundary. However, this study looked at streams across the whole country and effects, such as the degree of imperviousness, may have been obscured by the effect of large-scale geographical factors.

Figure 31

Relationships between catchment imperviousness and water quality in three Waitakere City stream systems monitored over the summer of 2003-4 (reproduced from EVA et al. 2005a).

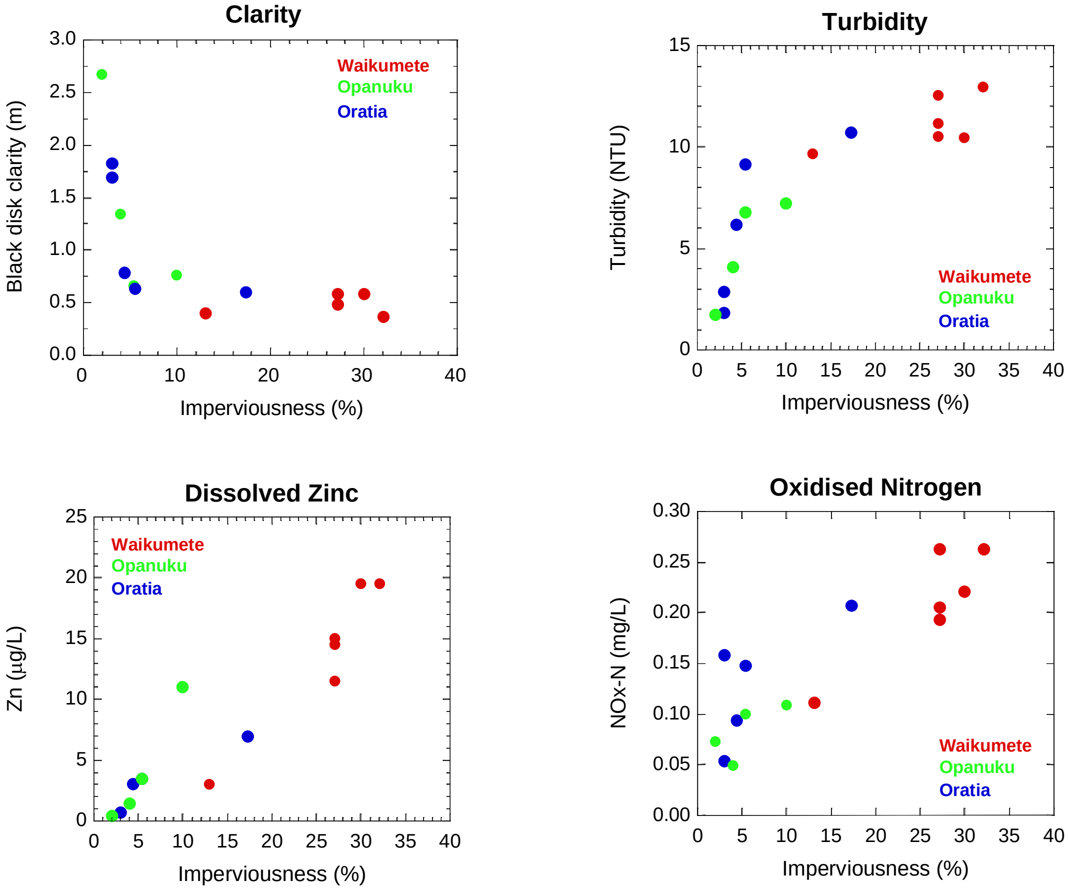
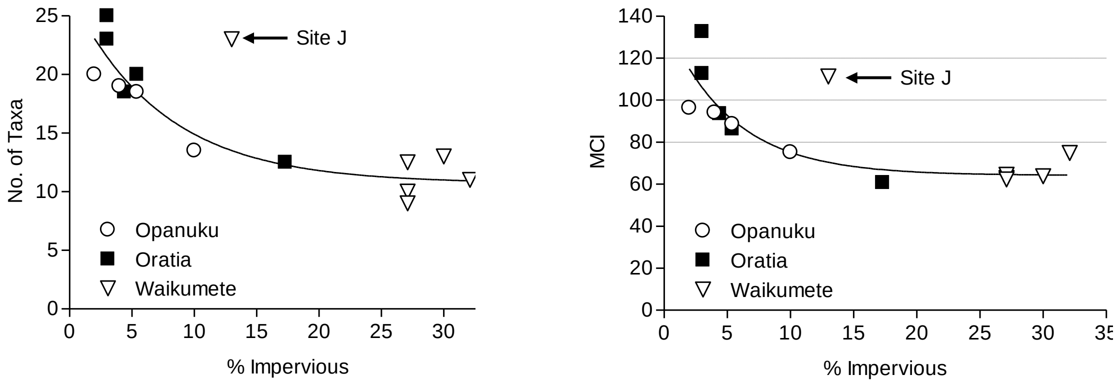


Figure 32
Relationships between catchment imperviousness and macroinvertebrate community indices (biological quality) in three Waitakere City stream systems monitored over the summer of 2003-4 (from EVA et al. 2005b).



Overall, it appears that catchment imperviousness provides a robust indicator for the effects of urbanisation, being related to receiving water quality and ecological health indices. The data collected to date suggest that even small increases in imperviousness can have major adverse effects on stream quality.

Clearly, preventing stormwater run-off into streams without prior treatment, source or flow control is required, even in relatively undeveloped catchments. Reducing the effects in established urban areas with high imperviousness by source control, retrofitting stormwater treatment and using retention strategies to minimise direct discharges of stormwater to streams is also required to maximize the benefits of other stream restoration efforts (eg in-stream and riparian habitat improvement).

6.7 Effects of urbanisation on flow-related stream habitat

Increased catchment imperviousness results in greater volumes of storm run-off, higher peak flows, and (sometimes) lower low flows. Impacts of these changes inflow include increased frequency of flooding, accelerated stream bank erosion, changes in stream channel morphology (stream channel widening, reduced habitat diversity), increased streambed disturbance (and hence disruption of benthic animals and plants), and reduced water quality (increased turbidity, maximum temperatures etc). Because change in stream flow affects many aspects of stream habitat, it has a major influence on biological community health.

Elliot et al. (2004) have produced a comprehensive guide that provides information on the effects of urbanisation on stream flows, the associated effects on stream habitat, and the consequent effects on stream communities. This guide gives methods to assess these effects for a given degree of development, along with measures that can be used to control the flows.

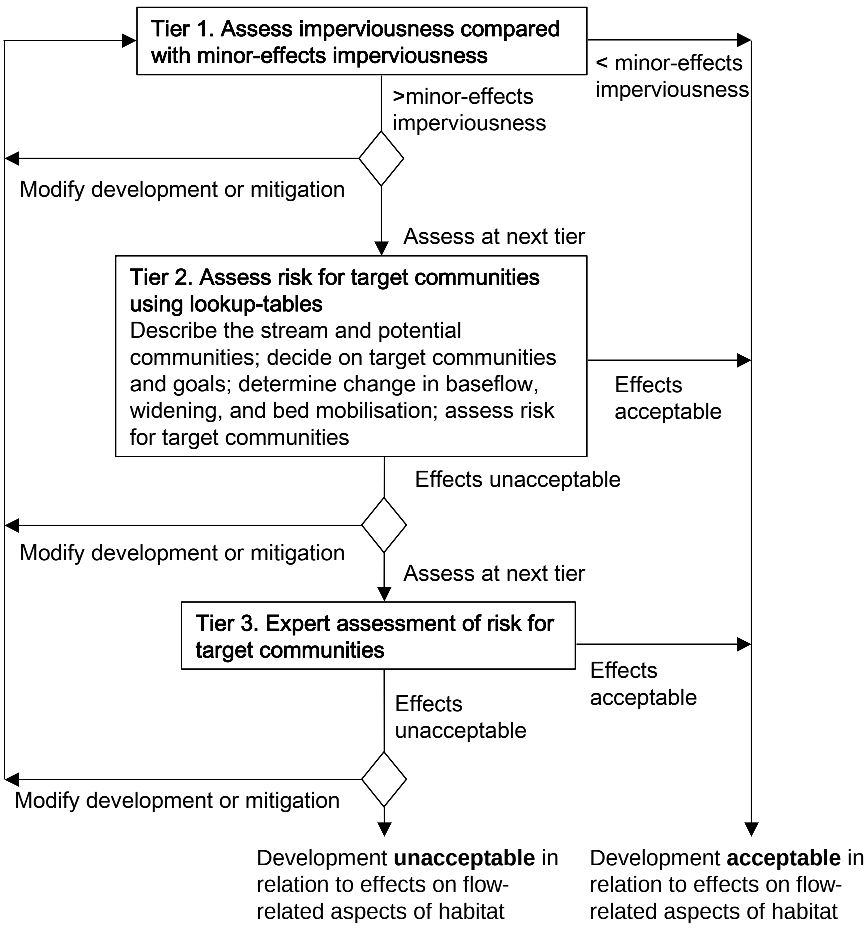
Their approach involves a 3-tiered assessment procedure (Figure 33):

- Tier 1 – calculate or measure the stream catchment imperviousness and compare this with imperviousness value below which "minor effects" are expected. A default value of 10 per cent imperviousness was suggested as this threshold. If imperviousness is below this level, then no further assessment is required (unless the catchment is judged to be particularly sensitive, and a lower imperviousness cutoff is warranted). If imperviousness is above the minor effects level, the degree of development can be reduced or mitigation measures introduced (eg erosion control ponds), or the assessment can move to the more detailed Tier 2.
- Tier 2 – describe the stream, the potential biological communities that could exist in the stream, and the degree of protection required for these communities. Then analyse the change in baseflow, channel widening, and frequency of bed disturbance, associated with the development taking into account any mitigation measures. Finally use "lookup tables" (provided as Table 2 in the guide) to assess the severity of effects for the target communities. If the effects are unacceptable in relation to the target degree of protection, then the degree of development or mitigation measures can be changed and the effects re-assessed, or a more detailed assessment can be performed at the next tier.
- Tier 3 – use expert site-specific assessment, including more detailed modelling to reduce the uncertainty in effects predicted from Tier 2.

Methods for conducting the technical assessments required are detailed in the guide.

Figure 33

Assessing impacts of urbanisation of flow-related aspects of stream habitat (from Elliot et al. 2004).



6.8 Summary of impacts on streams

Since 1995, a very large body of work has been conducted on improving our understanding of the impacts of urbanisation on Auckland's streams. The work has:

- Provided a huge database on water quality, stream ecology, and (to a lesser degree) sediment quality, which have enabled the impacts on streams to be well characterised.
- Clearly demonstrated that urban streams have the poorest water quality, sediment quality, and biological quality of all the streams in the Auckland region. However, in many places, upstream rural land use contributes to the degraded state of urban waterways, at least for water quality and ecology.
- Provided much of the knowledge required to predict how changes in land use, and consequent changes in stream flow, water quality etc, will affect the biological

health of urban streams, and how these changes are likely to develop over time as catchments undergo development and mature.

- Allowed an urban stream management framework based on scientifically-based, realistically achievable, maintenance and restoration objectives to be developed.

The information shows that urban streams have:

- highly variable water quality and flow, due to the intermittent nature of discharges, and their small size relative to the run-off volumes that they receive;
- very poor microbiological quality, making them unsafe for contact recreation or food gathering;
- high turbidity and suspended solids concentrations, and low visual clarity, which decreases the aesthetic appeal and ecological values of these streams (although this is also common to Auckland's rural streams);
- occasionally high ammonia concentrations, probably most often associated with wastewater overflows or leakage;
- moderately elevated plant nutrient concentrations, generally sufficient to support excessive growths of aquatic plants and algae;
- elevated temperatures in shallow, unshaded, stream reaches, particularly open concrete-lined channels where thermally sensitive aquatic animals would be unlikely to survive in summer;
- elevated concentrations of Zn and Cu in stream waters, such that short-term acute effects during storm run-off, and longer-term chronic toxic effects during base flows, probably occur;
- very high concentrations of Zn, Cu, and Pb in streambed sediments, suspended sediments, and biofilms, which are almost certainly going to be toxic to aquatic animals that ingest these materials when feeding;
- a degraded macroinvertebrate community, dominated by animals that are tolerant of poor water quality and habitat and high-flow variability, and lacking sensitive life forms that are indicative of high quality, stable, stream environments; and
- variable fish populations, depending largely on stream location, size, barriers to migration, and to a lesser degree, presence of appropriate habitat.

The degree to which these characteristics are observed in a given stream will vary depending on a range of variables, including the nature and intensity of urbanisation in the catchment, the age and history of development, upstream land use, the amount and type of stormwater (and sewage) reticulation and treatment present, traffic densities etc. It appears that many of these factors are integrated into "catchment imperviousness" as a simple, holistic, indicator (or predictor) of potential impacts on stream quality. Impacts are observable at low levels of imperviousness (<10 per cent), and increase dramatically between 10 and about 20 per cent, above which streams often have very poor quality.

Current knowledge gaps on impacts on streams include:

- Quantifying the real human health risk from high levels of indicator organisms in urban streams. Identifying the dominant sources and forms of pathogenic microbial contamination (eg cows, sheep, dogs, cats, birds, human sewage; are viruses a significant health issue?). What is the effect of land use (especially old sewerage systems versus new developments). Is microbial re-growth or multiplication a major issue?
- Impacts and levels of currently used pesticides, legacy contaminants (organochlorine pesticides, PCBs), and emerging contaminants (PBDEs, plasticizers).
- Detailed guidelines and designs for stream restoration.

A6.1 Regional stream water quality: a graphical summary

Figures 34a and b provide a graphical summary of average (median day time) water quality in streams monitored in ARC's regional monitoring programme. Urban streams have been separated to permit comparisons with "rural" (ie non-urban, including exotic forest, native bush, and farmed land).

Figure 34a

Long-term (1992–2000) median water quality in ARC LTB-WQ and Tamaki catchment streams monitoring. Data are from ARC (2000b), and NIWA (2001b). The red dashed line indicates approximate water quality guideline levels (ANZECC 2000).

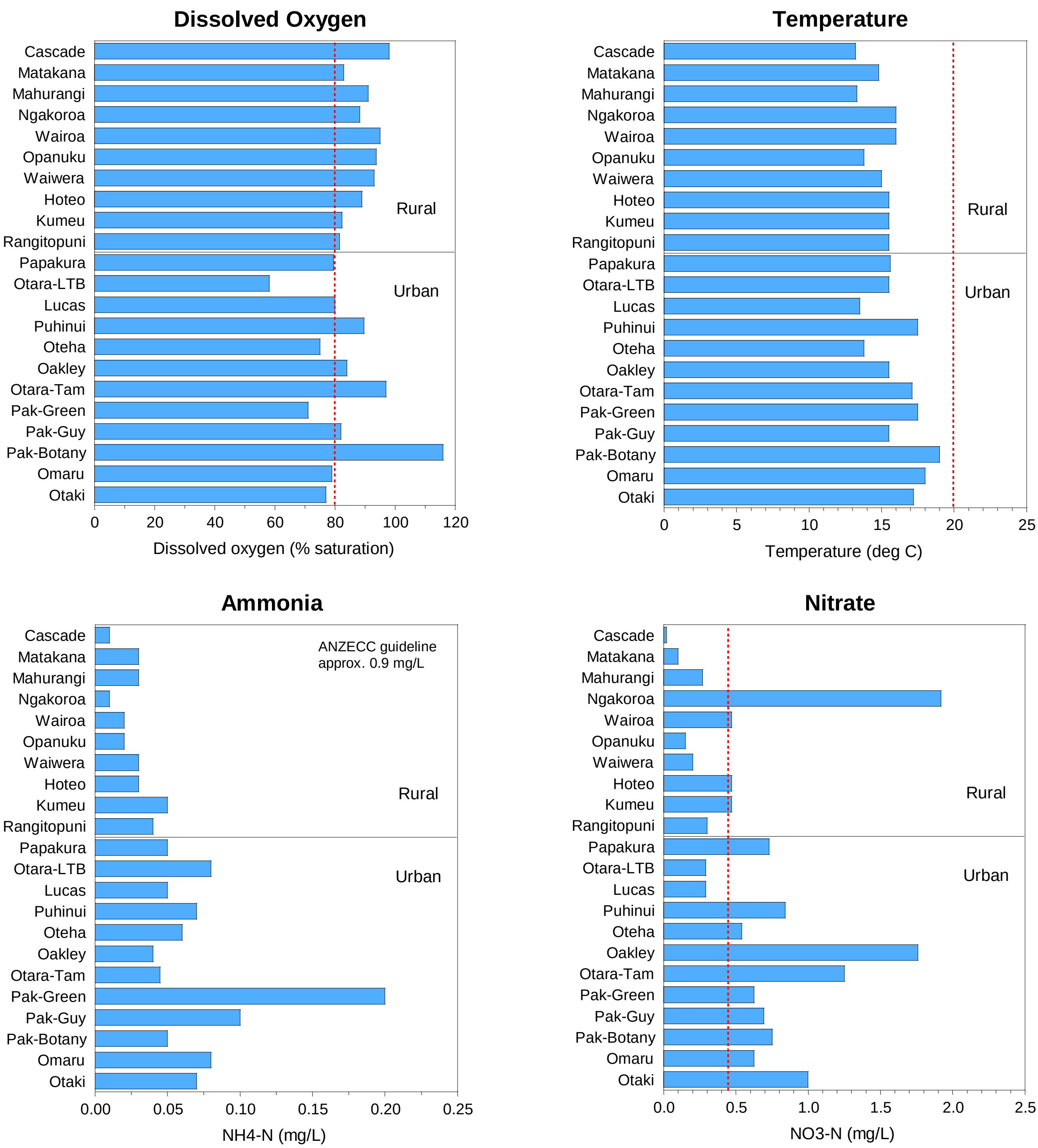
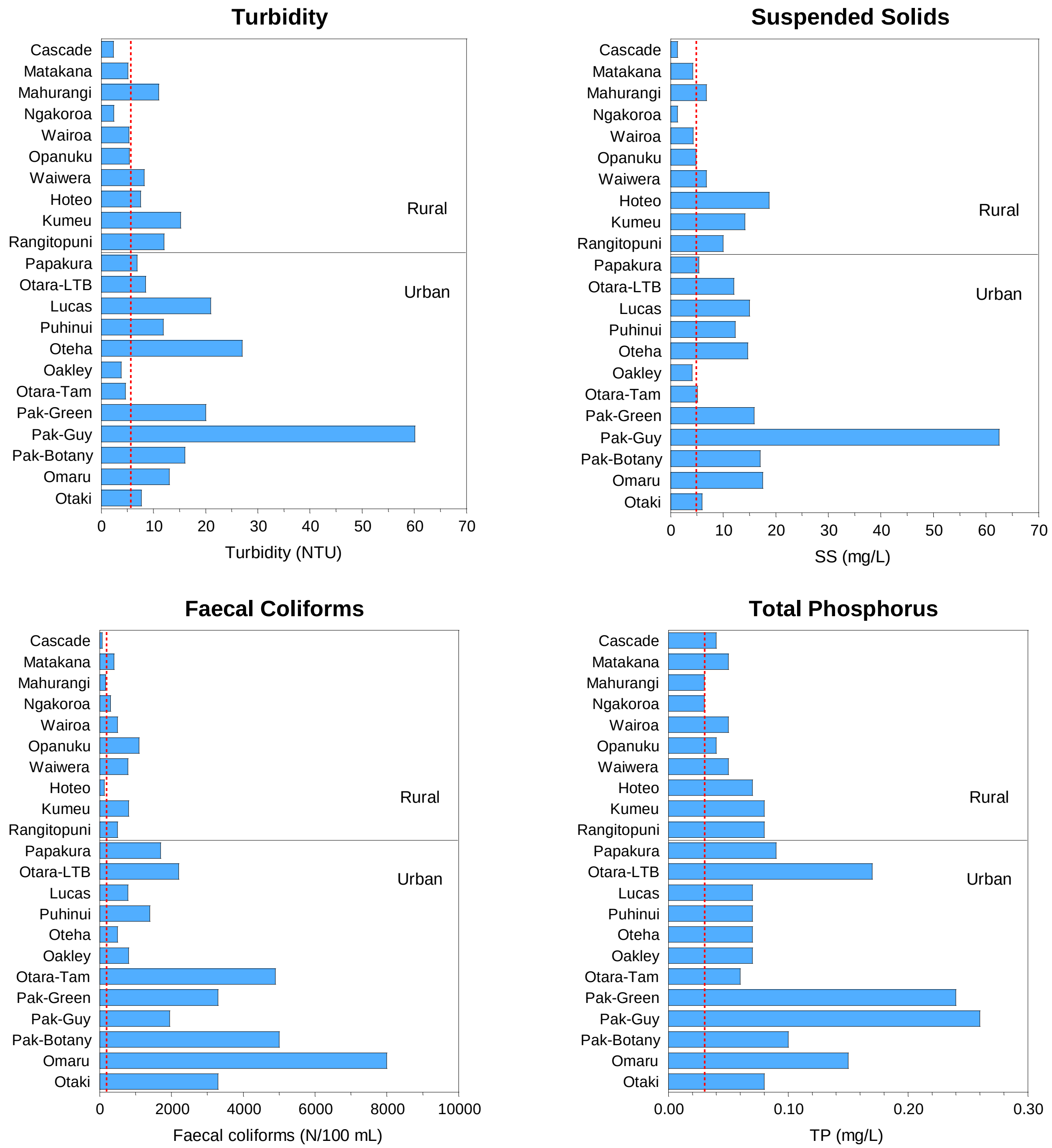


Figure 34b

Long-term (1992–2000) median water quality in ARC LTB-WQ and Tamaki catchment streams monitoring. Data are from ARC (2000b), and NIWA (2001b). The red dashed line indicates approximate water quality guideline levels (ANZECC 2000).



A6.2 A summary of local urban stream water quality studies

As briefly described in Section 6.1.2, a number of local studies of urban stream water quality have been conducted around Auckland in the past decade. Data from the studies listed in Section 6.1.2 (and summarised in Table 13 below) have been collated and summarised here to illustrate the "average" (generally the median) and range of water quality reported for urban streams around Auckland. Where sampling conditions were identified in the studies, normal (or dry weather) and storm (wet weather) flow data have been separated to show how storm run-off may affect water quality. Note the high variability in water quality – the concentrations of the various water quality parameters are shown on a log scale.

Table 13

Local stream water quality studies summarised in the following discussion and plots.

Study	Stream or location	Description of data	Reference
A	Waitakere streams	14 sites, 1–3 samples in spring 1999, normal flows	KMA 2000
B	Waitakere streams	8 urban sites, 6 samples in summer 2003–4, base flows	EVA et al. 2004a
C	North Shore streams	15 sites, 2 samples in each of normal and storm flows, 1999–00	Merritec 2000a
D	Wairau Creek	2 base flow samples, 4 storm events, 1999	Merritec 2000a
E	Wairau Creek	Motorway site, base flow and 11 storm events, 1982–83	Williamson 1986
F	Wairau Creek	Chartwell site, base flow and 4 storm events, 1982–83	Williamson 1986
G	Mid-Waitemata streams	Single sample, 15 sites, 11 streams, base flow	ARC 1999c
H	Otara catchment	4 urban sites, 5 normal and 3 storm flows	Worley 2000
I	Waiheke Island	Single sample, 6 streams, February 2001, base flow	ACC 2001
J	Pakuranga Creek	One site, 1985–1992, base flow and 1988–1992, storm flow	ARC 1994c
K	Pakuranga Creek	20 fortnightly samplings in mostly dry weather flows, 1998–99	NIWA 2000b
L	Awaruku Stream	20 fortnightly samplings in mostly dry weather flows, 1998–99	NIWA 2000b
M	Botany Creek	20 fortnightly samplings in mostly dry weather flows, 1998–99	NIWA 2000b
N	Oakley Creek	Jan–July 2002. 10%ile and 90%ile shown for range	NIWA 2005
O	Whau River	Blockhouse Bay Rd, Jan–May 2003	NIWA 2005
P	Whau River	Wolverton Rd, Jan–July, 2002	NIWA 2005
Q	Auckland City streams	16 small streams, 2 high and 2 normal flow, May–June 2002	NIWA 2005

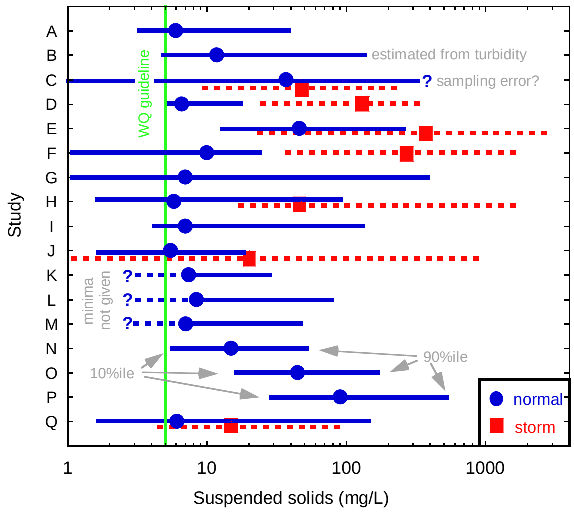
Suspended solids

Streams usually have moderate to very high suspended solids' concentrations, and therefore streams have low visually clarity.

Median TSS concentrations during normal flows lie between approximately 5 and 100 mg/L, and are generally somewhat higher during storm flows (approx. 15–400 mg/L). Maxima of over 1000 mg/L have been recorded during storm flows.

Median SS concentrations are mostly above an ecological health guideline of approximately 5 mg/L, and therefore it would be expected that aquatic life may be adversely affected.

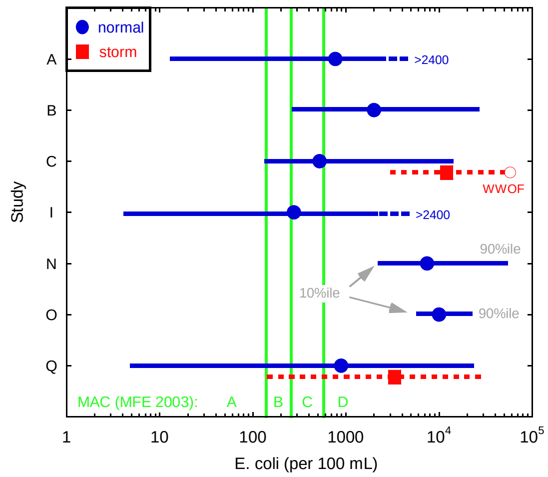
Figure 35
Urban streams: suspended solids



Microbial indicators

Microbial quality is poor. Median E. coli numbers range from approximately 300–10,000 per 100 mL, and can approach 100,000 per 100 mL during WWOF events. The data suggest that urban streams would have a "Microbiological Assessment Category" (MAC; MFE 2003) of D – the worst grading in the MFE recreational water quality guidelines – and therefore they are almost always unsuitable for human contact recreation.

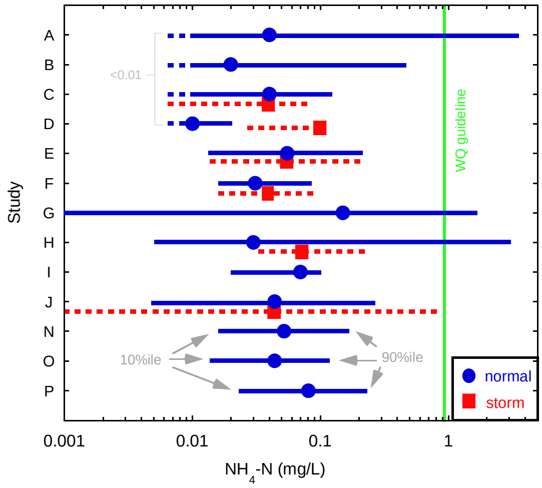
Figure 36
Urban streams: E.coli



Ammonia

Concentrations of ammonia are usually low, with median concentrations about 1/10th of the aquatic toxicity guideline (approximately 0.9 mg/L depending on temperature and pH). Maximum concentrations very occasionally exceed the toxicity threshold, but not by much. WWOFs can contribute to elevated ammonia, as can inflows of anaerobic groundwater or decomposing organic-rich sediments.

Figure 37
Urban streams: ammonia



Nutrients

Urban streams show moderate nutrient enrichment, with median DRP and nitrate-N concentrations below, or close to, water quality guidelines for nuisance aquatic plant growth. Maximum concentrations exceed guidelines.

Figure 38
Urban streams: nitrate-nitrogen

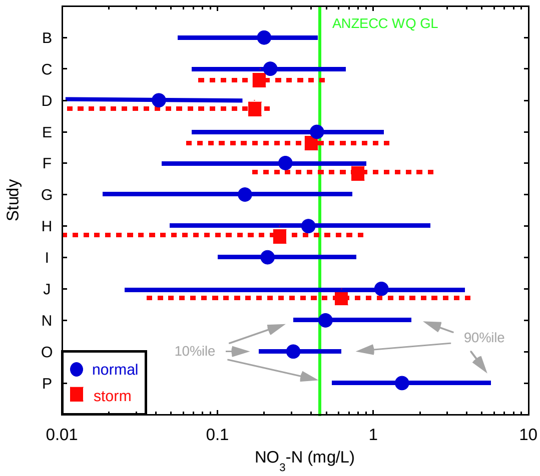
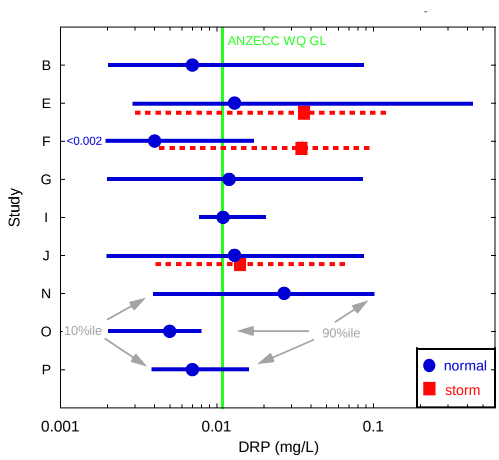


Figure 39
Urban streams: dissolved reactive phosphorus



Heavy metals

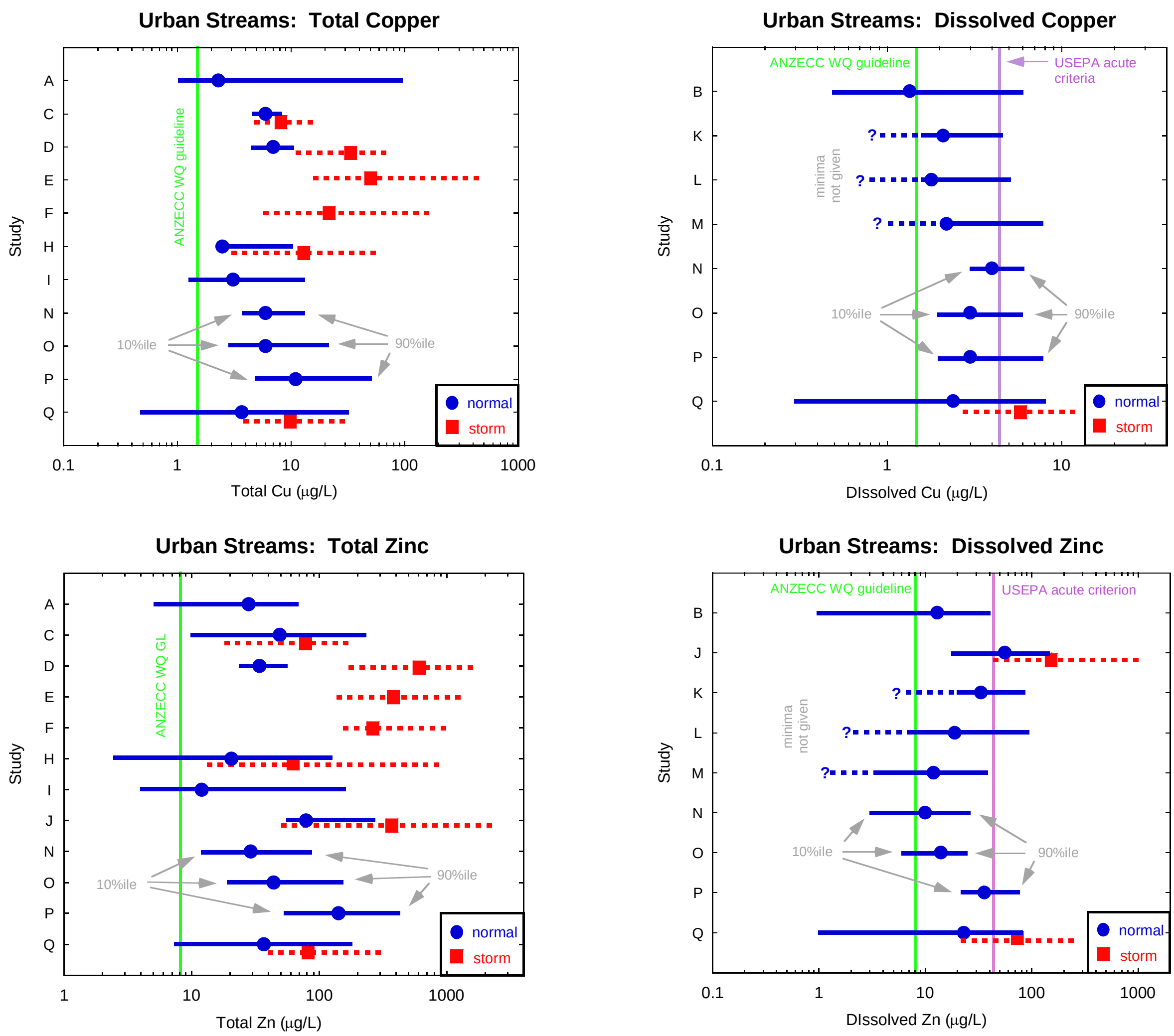
Total Cu and Zn concentrations are usually well above ANZECC water quality guidelines, with concentrations several times higher during storms than at normal flows. Dissolved Cu and Zn concentrations are also usually above ANZECC guidelines, with median concentrations exceeding guidelines by up to about 4–5 times during normal flows. A small amount of data indicates that dissolved Cu and Zn concentrations are also considerably higher during storm flows, again up to several times higher than at normal flows. Median concentrations of dissolved Cu and Zn are usually below USEPA chronic toxicity water quality criteria for dissolved metals at normal flows, but exceed the USEPA acute toxicity thresholds during storm flows.

NIWA (2000b) estimated that under base flow conditions in Pakuranga and Botany Downs streams (studies K and M in the plots below), <5 per cent of aquatic fauna would be adversely affected by dissolved Cu, and <10 per cent by Zn. Data from Waitakere streams show similar results – up to 20 per cent of species might be affected by Cu and/or Zn in urban stream reaches under base flow conditions, with acute toxicity thresholds (USEPA criteria) being exceeded at higher flows.

Metals would therefore be expected to have occasional short-term acute effects on aquatic life, especially during storm events, and long-term chronic impacts on a relatively small proportion (generally <20 per cent) of the more sensitive aquatic species.

Figure 40

Metals in urban streams.



There is relatively little information on other chemical contaminants in urban stream waters. Total PAH concentrations in Pakuranga, Botany Downs, and Awaruku Streams were low (median 0.125 g/L (parts per billion), range 0.093–0.229 g/L; NIWA 2000b), but were considerably higher (up to 3.5 g/L) in the Whau River (NIWA 2005). Total petroleum hydrocarbons (TPH) in 16 small Auckland City streams were below 0.2 mg/L, and up to 1.19 mg/L in the Whau River (NIWA 2005). These concentrations are unlikely to represent a significant risk factor to aquatic life.

A6.3 A summary of local urban stream ecological studies

Wilding (1996 and 1999)

A survey of 12 urban and four reference (forested) streams was conducted by Wilding (1996, 1999) to assess the effects of catchment development, in-stream modifications (eg piping, concrete channelling), and riparian vegetation on Auckland urban stream ecology.

Urban streams were found to have more pollution tolerant invertebrates, with piped and concrete channelled streams having the poorest quality fauna. EPT scores for the urban streams approached zero (indicating the almost complete absence of sensitive macroinvertebrate species), and freshwater crayfish and several species of native fish were also absent from urban sites.

Extensive riparian forests on North Shore streams was found to improve ecological health to some degree, but poor water quality and altered flow regimes in urban streams were thought to limit the benefits of riparian vegetation and contributed to the lower diversity of aquatic life present in these urban streams.

Mid-Waitemata Harbour catchment freshwater ecology survey

In 1998, an extensive survey of 61 sites on 44 streams in central and west Auckland and the North Shore was carried out (the Sides & Bennett study reported in ARC 1998b). Aquatic plants, macroinvertebrates, and fish were sampled and habitats described.

Ecological value of the streams was categorised into five classes (low to high value) according to the presence or absence of three indicators – mayflies (one of the EPT taxa), riparian forest, and galaxid fish species (or >2 fish species).

Most of the central area sites were rated as having low ecological value, having poor macroinvertebrate and fish communities, and sparse riparian vegetation. Many of the streams outside the central area had higher ecological value, mainly because of good habitat conditions associated with forested riparian margins. The most significant influence on aquatic fauna appeared to be channelisation, followed by removal of riparian vegetation, followed by water quality effects.

Further analysis of the information collected in the Sides & Bennett 1998 study was presented by NIWA (2005). Streams were assigned to four groups according to stream size, catchment land cover, maximum altitude, and catchment area.

Invertebrate communities from these four groups were found to differ, especially between groups 1 (small, low-lying, urban streams) and four (longer streams with greater amounts of unmodified forest). Urban streams were dominated by midges, worms, damselfly and a snail (Physa), while the forested streams were characterised by having sensitive species including at least four caddisfly and three mayfly genera, and stoneflies.

The "urban" streams were also characterised by having warmer water temperatures and lower dissolved oxygen concentrations, slower velocities, and concrete or bedrock substrates – conditions that are not suitable for sensitive aquatic invertebrates. The differences in water quality and habitat are in part a consequence of the smaller, low lying, nature of the urban stream group, but also a result of urban modifications, especially removal of riparian vegetation (shade) and channel works.

Ecological and physical state of Auckland City streams

A comprehensive description of the state of Auckland City streams is given in ACC/Metrowater (2001). A significant component of this work was a survey of aquatic habitat quality at 30 sites in nine streams (Suren 2001). Key habitat variables (riparian vegetation, bank modification, channel modification, aquatic plants) and basic water quality (pH, temperature, conductivity, dissolved oxygen) were recorded.

Five classes of habitat quality were identified based on the scores from the four key habitat variables, ranging from 0 (low quality) to 4 (high quality). Of the 30 sites surveyed, 13 were assessed to have low habitat quality – these were mostly from streams sites with concrete culverts, little or no riparian vegetation, and streambed covered in algae. Three sites were "moderate-low" quality, and only seven sites had "high quality" habitat – the latter were usually sited in reserves, where bank and channel were natural and shaded by trees.

North Shore City streams

The urban streams in North Shore City have been extensively surveyed, providing a very comprehensive database on their physical and ecological state. Reports on 26 streams have been presented on the North Shore City Council website[25]. The information in these reports covers catchment features (land use, history of development, imperviousness, geology etc), hydrology, water and sediment quality, stream ecology (including invertebrates, fish, in-stream and riparian plants), and stream channel conditions (including channel erosion, structures, and barriers to fish migration).

As found in other parts of Auckland, the North Shore streams show a wide variation in physical and ecological condition. The streams are generally short and steep, with muddy bottoms and clay banks. Conditions vary hugely between and within streams, from near pristine to almost entirely lined and piped, and the aquatic communities present in the streams reflect the various types of habitats. While the highly modified stream reaches have degraded biological health, most of the longer streams were

[25] www.northshorecity.govt.nz. Go to the "Streams" page in the A-Z index to find the "Stream Assessment" reports.

found to have banded kokopu, long and shortfinned eels and freshwater crayfish. Some of the streams also contain valuable fish spawning habitats and certain streams have up to seven native fish species.

North Shore City has undertaken macroinvertebrate trend monitoring at selected sites to assess changes in ecological health over time. EVA et al. (2003a) summarised findings from 1999–2003:

- Macroinvertebrate taxa number showed a slight increase at over half of the sites sampled. The number of pollution and habitat sensitive EPT taxa remained very low, with essentially the same number of taxa at most sites over the three-year period.
- MCI values had essentially remained the same at most sites. At a small handful of streams, MCI values have shown a slight sustained increase over the three years (eg Kahika, Kaipatiki, and Alexandra streams).
- There was little change in the dominant taxa, with snails dominating the majority of streams, and with either worms or midge larvae (or both) representing a large component of the macroinvertebrate communities.

Waitakere streams

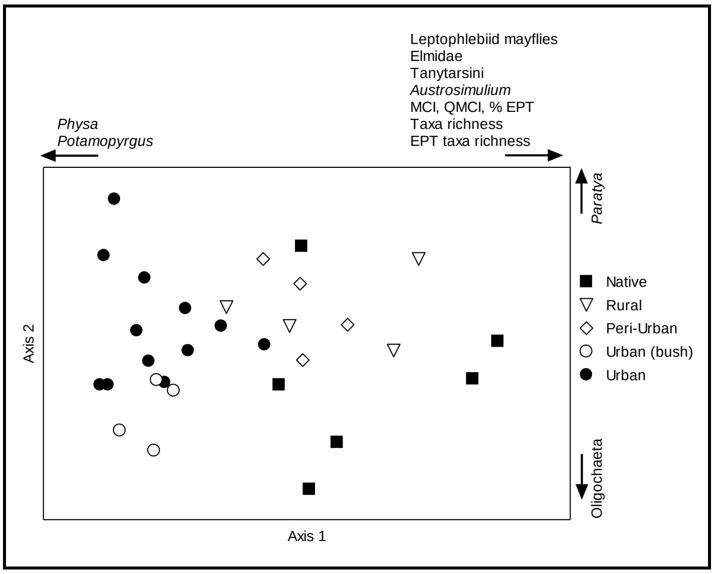
Streams in the Waitakere City area have been well surveyed, with reports on stream ecology by KMA (2000), which includes references to earlier studies undertaken in the area, and EVA et al. (2004c). Studies linking ecological health with water and sediment quality and catchment land use pressures have also been undertaken (EVA et al. 2005b; Herald 2003; van Roon et al. 2004).

Aquatic habitat, macroinvertebrates, and freshwater fish were surveyed at 30 stream sites, including nine urban sites, by KMA (2000). Macroinvertebrate communities were found to be well correlated with land use and with habitat quality. Water quality and sediment quality were also significant influencing factors. Sites from throughout the catchment were scored based on a combination of water quality, sediment quality and macroinvertebrates, then ranked from best to worst quality (best possible score = 0, worst = 7). Urban sites ranked worst, with an average score of approximately five (with sites in recently urbanised areas scoring better than older urban sites), rural sites scored approximately 4, while bush living (score 2.9), and reference sites (1.9) were of markedly better overall quality.

Monitoring conducted at 15 sites (see Figure 22) in summer 2003/4 as part of Waitakere City's "Project Twin Streams" found that invertebrate community composition differed significantly between land uses (EVA et al. 2004c). Urban and native land uses had the most distinct faunas, with rural and peri-urban sites having intermediate communities. Urban sites had a greater abundance of pollution tolerant species (*Potamopyrgus* snails, oligochaete worms and *Paratya* shrimps), while native sites had greater numbers of sensitive invertebrates (leptphlebiid mayflies, *Austrosimulium* blackflies, and Tanytarsini chironomid midges). These differences are shown below (Figure 41).

Figure 41

Ordination analysis of macro-invertebrates at 15 sites in Waitakere streams monitored in summer 2003-4 (EVA et al. 2005b). Invertebrate communities change with changing land use, with lower community richness and fewer sensitive species (EPT) at urban sites.



Habitat quality was found to be related to ecological health in areas with less intensive urban development, but for more urbanised sites (where catchment imperviousness exceeded around 20 per cent) catchment-scale disturbance outweighed the potential benefits of localised good quality habitat. They concluded that for areas with significant upstream urban development, and poor stormwater treatment, restoration of stream habitat is unlikely to greatly improve macroinvertebrate community health.

7 Impacts in the Marine Environment

Auckland's marine areas are highly valued natural environments, but they are also the ultimate receiving water for urban storm water discharges. Understanding the effects of stormwater on the marine receiving environment is therefore critical for sustainable management of this prized resource.

State of knowledge in 1995

By 1995, it was known that parts of Auckland's marine environment, in particular the older industrialised urban areas such as Mangere Inlet, Tamaki Estuary, and parts of the Waitemata Harbour, were contaminated by heavy metals (Zn, Cu and Pb), organochlorines (eg DDT, PCBs), polycyclic aromatic hydrocarbons (PAH) and human pathogens. Studies on stormwater found that a large proportion of most stormwater contaminants was bound to particulate matter, most of which passes through the drainage network into coastal receiving waters. Sediment was known to settle out in sheltered areas – effects were greatest in the upper estuaries and least in the higher energy open coasts and harbour entrances.

In sheltered coastal sediments, there was a link between urban stormwater discharge and build-up of contaminants. This build-up was judged to be potentially harmful to animals living in the sediment because some contaminant concentrations approached or exceeded North American sediment quality criteria, and because there was evidence for chronic toxic effects of contamination on aquatic animals. The evidence was circumstantial, in that the specific causes of the observed effects could not be identified, and it was acknowledged that much work was yet to be done to better understand this potential problem.

Key studies pre-1995

- Extensive surveys of heavy metal levels in sediments of the Waitemata and Manukau Harbours, (Glasby et al. 1988; Williamson et al. 1992b).
- Measurement of metal and organic contaminant concentrations in shellfish from the Manukau Harbour (ARC's shellfish quality survey annual reports, eg ARC 1993b), Tamaki Estuary (eg ARC 1992b), and Waitemata Harbour/East Coast Bays (eg NIWA 1995c) showing spatial patterns of contamination consistent with pollution from industrial/urban run-off.
- A synthesis of information on contaminant distribution, fate processes, guidelines, and monitoring, with particular reference to studies of the Manukau Harbour made between 1989 and 1993 (ARC 1994a; Williamson et al. 1991).
- Studies of the effects of sediment contamination on benthic fauna, which were able to establish adverse behavioural effects of specific chemicals on benthic animals in field or laboratory toxicity tests (Pridmore et al. 1991; ARC 1992a) and effects of general pollution on bivalve health (Roper et al. 1991), but which also highlighted the difficulty in detecting biological effects of run-off at a community level (Roper et al. 1988).

- An assessment of the potential impacts of urbanisation of the Upper Waitemata Harbour catchment on estuary sedimentation, contaminant concentrations in harbour waters, and contaminant accumulation in estuarine sediments (Vant et al. 1993).

Advances since 1995

A huge body of information gathered since 1995 has refined and expanded the knowledge base, enabling better definition of the nature and extent of impacts. These advances have been summarised in this chapter under the following headings:

7.1 – Impacts on marine water quality

7.2 – Chemical contamination of marine sediments

7.3 – Contamination of shellfish and fish

7.4 – Deciphering the history of urban stormwater impacts

7.5 – Stormwater impacts on benthic ecology

7.6 – Sedimentation impacts on benthic ecology

7.7 – Toxicity in the marine environment

7.8 – Modelling the build up of contaminants in estuarine sediments and harbours.

7.1 Impacts on marine water quality

The impact of stormwater on marine water quality is greatly affected by the nature of the receiving water environment in the vicinity of the discharge point. Two key factors influence potential impacts:

- dilution, which ranges from low in estuarine headwaters (especially at low tide during large storm events), to high in more exposed harbour and coastal waters; and
- turbulent mixing by winds, waves, and currents, which promote dilution of contaminant inputs, but also resuspend bottom sediments leading to lower water quality in shallow, near-shore areas of beaches and estuaries during storm events.

Beyond the upper estuarine and near-shore areas, the large dilutions available in the receiving environment reduce contaminant concentrations to very low levels, reducing impacts and making reliable measurements difficult.

Storm-derived inputs are episodic and highly variable, depending on factors such as storm size and duration. This also complicates monitoring and data interpretation.

Because of these difficulties, the focus has been on more "integrative" assessment methods – sediment chemistry and biological monitoring using sediment-dwelling animals. Shellfish, both native and deployed, have also been used as indicators to reflect medium- to long-term water quality conditions for low-level, toxic, accumulative contaminants (eg metals and organics).

Relatively few direct measurements have been made of the impact of stormwater on marine water quality. The exception is for microbiological water quality, for which there is a large amount of data. While there are few direct measurements of marine water quality impacts from stormwater, there is a large body of information that can be used to infer what the effects are likely to be. This section uses a selection of available information from stormwater monitoring data, broad-scale long-term regional monitoring, and short-term special surveys to assess the nature and scale of impacts.

Potential issues

- Human health effects from contamination by pathogenic microorganisms.
- Increased suspended sediment concentrations – effects on visual clarity and aquatic life.
- Low concentrations of persistent, toxic, chemical contaminants – Zn, Cu, Pb, PAH, and organochlorines – in marine waters.
- Cumulative effects of nutrients on harbour enrichment and algal blooms.

A **key issue of immediate concern is contamination by pathogenic microorganisms.** Stormwater contains elevated levels of pathogens. A major source is wastewater overflows into the stormwater system, which occur when sewer pipes become blocked, equipment fails and when heavy rainfall overloads the wastewater network. Other sources are overland run-off and illegal wastewater connections to the stormwater network. Non-human sources (dogs, cats, birds etc) may be significant contributors.

Sediment discharges are of greatest concern from developing catchments, where earthworks and bare ground generate high sediment yields during storms. Yields are much lower from mature catchments, and from modern developments with effective sediment control practices and treatment. The effects of sediment discharges into shallow estuarine and harbour reaches are likely to be difficult to distinguish from naturally high turbidity caused by resuspension of bottom sediments in these areas. Suspended sediment in the water column may therefore be less important than contamination by pathogens, although localised, short-term, aesthetic impacts are likely to occur, especially where stormwater is discharged into open harbour or coastal waters where natural background turbidity is low. The effect of deposited sediment on aquatic life is a potentially more serious issue and is described in Section 7.6.

Chemical contaminants, including heavy metals (Cu, Pb, Zn) and organics (PAH, organochlorine pesticides and PCBs), are present in variable, but generally small, concentrations in urban stormwater. By the time stormwater reaches the coast, most of the contaminants are associated with the suspended sediment fraction, and therefore concentrations dissolved in the water column in the marine receiving waters are very low and are unlikely to cause adverse effects on aquatic life. The contaminants ultimately accumulate in harbour and estuary sediments, and it is here that biological effects are more likely to occur – this is described in Sections 7.2 and 7.5.

Plant nutrients, including nitrate-nitrogen (NO_3-N) and phosphorus (P), are present in moderate amounts in stormwater. Concentrations are much higher in sewage, and therefore stormwaters contaminated by WWOFs have considerably higher nutrient concentrations. It is possible that stormwater discharges cumulatively provide a significant nutrient load to marine receiving waters, and hence may promote algal growths in relatively enclosed harbour and estuarine areas around Auckland.

State of knowledge in 1995

Relatively few measurements had been made of the direct impact of stormwater on marine water quality. Estuaries were observed to be visually impacted by large loads of fine sediment from erosion of soils during earth working. Most studies, both then and more recently, relied on predicting concentrations in estuaries on the basis of average concentrations in stormwater, and assuming complete mixing in the marine receiving water – eg Vant et al. (1993) predicted concentrations of SS, nutrients, and indicator bacteria in the Upper Waitemata Harbour from a range of urbanisation scenarios.

The ARC had been conducting State of the Environment saline water quality surveys since the mid-1980s at a wide range of marine sites, some of which (eg Panmure Basin, Tamaki Estuary, and Middle Waitemata Harbour sites) are significantly affected by urban stormwater. This monitoring provides little direct information on stormwater impacts on water quality, but it does reflect the broad-scale impacts of catchment land use and enables urban impacts to be placed into context with other contaminant sources (eg rural and point-source sewage).

To overcome some of the difficulties associated with monitoring very low concentrations of metals and organic pollutants (OCPs, PAH, PCBs) in marine waters, the ARC had begun to use sediments and shellfish as integrative monitoring tools (see Sections 7.2 and 7.3). Monitoring of contaminant concentrations in shellfish from the Manukau Harbour and Tamaki Estuary began in the 1980s, and showed clearly elevated concentrations of metals and persistent organic pollutants (eg PAH, PCBs, OCPs) at sites receiving urban/industrial run-off. These results indicated that run-off was contaminating receiving waters (and sediments) and that contamination was biologically available to marine life.

Advances since 1995

Relatively few advances on water quality impacts have been made since 1995, except in the area of human health risk assessment from the measurement of pathogenic indicator bacteria. This reflects the importance and public interest in microbiological impacts, and the less acute nature of other water quality issues.

On-going water quality monitoring in ARC programmes initiated in the 1980s–early 1990s, and detailed data reviews of monitoring data (eg Vant & Lee 1998; ARC 2000b) have improved our knowledge of broad-scale spatial patterns and temporal trends in water quality. Similarly, shellfish monitoring in the Manukau, Tamaki, and Waitemata has continued and methodologies have been refined to give improved data on trends in water quality (eg ARC 2004i).

Studies on heavy metal mobilization in Manukau Harbour (Williamson et al. 1996), short-term studies of wet- versus dry-weather water quality at several near-shore sites (Worley 2000, Auckland Healthcare 2000, ACC/Metrowater 2001), and routine bathing beach monitoring conducted by TLAs provide additional information, particularly on microbiological impacts. In addition, very comprehensive data on the composition of Auckland City stormwater (NIWA 2005 and references therein) enables some predictions of potential impacts to be made.

7.1.1 Predicting impacts from stormwater composition

The composition of Auckland City's stormwater has been comprehensively characterised in a series of NIWA studies, which have recently been compiled and summarised (NIWA 2005). These data can be used to predict the likely impacts of stormwater discharges to marine waters, by comparing stormwater quality with receiving water guidelines (eg ANZECC 2000, MFE 2003), as shown in Table 14 below.

Table 14

Comparison of median and 90 percentile concentrations of selected contaminants[a] in Auckland City stormwater with marine water quality guidelines[b]. Stormwater data are taken from NIWA (2005).

Measure	TSS g/m^3	Cu (diss) mg/m^3	Zn (diss) mg/m^3	DRP mg/m^3	NO_3-N mg/m^3	NH_4-N mg/m^3	Enterococci N/100 mL
Median	39	6	160	28	690	60	6800
90%-ile	198	16	740	72	2680	250	29100
Guidelines[b]	note c	1.3	15	10	444	900	140

a. TSS – total suspended solids; dissolved copper (Cu) and zinc (Zn); DRP – dissolved (reactive) phosphorus; NO_3-N – nitrate nitrogen; NH_4-N – ammoniacal nitrogen.

b. Water quality guidelines – ANZECC (2000) at 95 per cent protection in slightly-moderately disturbed ecosystems. Enterococci guideline is the single sample "alert" level (MFE 2003).

c. There is no ANZECC guideline for suspended solids – reference to receiving water conditions is recommended to make a judgment. Data from ARC marine water quality monitoring programmes indicate that average and maximum concentrations (in brackets) of TSS in Auckland marine waters range from approximately 15–50 (up to 500) g/m^3 for estuaries, 10–20 (50) for harbours, and 3–5 (10) in open coastal waters.

The data summarised in Table 14 show that:

- Typical Auckland City stormwaters have fairly low suspended solids concentrations. They are unlikely to have a major impact on suspended solids concentrations in estuarine waters because of the naturally high levels present, especially during storms. There may be a small effect on harbour waters and open coastal waters. However, beach waters are typically turbid during storms because of sediment resuspension in the surf zone, so impacts in these environments are likely to be limited (see Mairangi Bay "case study" later in this chapter).

- Dilution by 5–12 times in clean receiving waters would be required to reduce dissolved Cu to guideline levels, while Zn would need to be diluted by about 10–50 times. These dilutions should be readily achieved, except perhaps in estuary headwaters at low tide, when the freshwater inflows constitute most of the water present. Impacts might then occur, but they are likely to be short-term (dilution by the in-coming tide occurs within six hours) – the guidelines probably overestimate the likely effects under these conditions because they represent longer-term effects on sensitive fauna (generally from chronic toxicity tests lasting several days; ANZECC 2000).
- Ammonia concentrations in raw stormwater are well below toxicity guidelines, and therefore adverse effects should not occur.
- Nitrogen and phosphorus concentrations are above guidelines for promoting nuisance plant growths (eg algal blooms), but not by very much. Dilution by 10-fold would be sufficient to reduce highest levels to below guidelines. Adverse effects are therefore unlikely, although the cumulative impact of the total load of stormwater-sourced nutrients entering Auckland's harbours is unclear. Concentrations are similar to those found in rural streams, which have greater total discharge volumes and therefore probably contribute a greater mass load to the marine environment.
- Enterococci levels are very high, especially where there are significant WWOFs (eg Cox's Bay), which can increase microbial contamination by up to an order of magnitude. Dilutions of approximately 50–200 times are required to reduce median and 90 percentile stormwater concentrations to below MFE single sample "alert" level guidelines, or by 25–100 times to get below the "action" level guideline of 280 enterococci per 100 mL. This is roughly consistent with NIWA's finding that concentrations exceeded the action level when the freshwater component exceeded about 5 per cent (ie 20-fold dilution) at several Auckland beaches (NIWA 2004)[26].

Organic contaminants such as polycyclic aromatic hydrocarbons (PAH) are present in stormwater, but their concentrations are generally low. Auckland City data show median and 90 percentile total PAH concentrations of 4.8 and 28.4 mg/m^3 respectively. The likely impact of these compounds on marine water quality is unknown because of a lack of guidelines.

Higher molecular weight PAH (eg combustion-sourced compounds) and other hydrophobic organic compounds in estuarine environments are associated with, and deposited in, sediments. Impacts of these compounds are therefore more likely to occur through accumulation in estuarine sediments than via the water column. The lower molecular weight, more soluble PAH compounds, are generally shorter-lived in the aquatic environment, being removed by volatilisation and biological degradation. It would therefore seem unlikely that PAHs from normal urban stormwater discharges would have significant impacts in marine waters, but this is unconfirmed.

[26] Modelling by ACC indicates that this assessment may overestimate the exceedances of microbiological guidelines. This was being reviewed at the time this review was conducted.

7.1.2 Broad-scale water quality impacts from ARC regional monitoring

ARC have regularly monitored saline (coastal and harbour) sites since the mid 1980s–early 1990s in two programmes:

- The Long-Term Baseline Water Quality (LTB-WQ) network, which has 18 saline sites including the open northern east coast, Waitemata Harbour, and Manukau Harbour. Data are reviewed annually and reported in ARC Technical Publications and assessed in more detail for statistical analysis of spatial and temporal trends every few years (eg ARC 2000b).
- Three "special surveys", which monitor the Upper Waitemata Harbour (UWH), Tamaki Estuary, and Mahurangi Harbour, have been undertaken because of specific local issues within each of these water bodies. Monitoring sites were chosen to reflect the local influences of each survey area, in contrast to the other LTB sites, which are chosen to be regionally representative. NIWA (2001b) provides details and analysis of data collected between 1992–2001. Data are routinely reported (generally annually) in ARC technical publications (eg ARC 2003f).

Monitoring in both programmes assesses a suite of "general" water quality parameters, including dissolved oxygen (DO), clarity (Secchi depth), turbidity, suspended solids (SS), chlorophyll *a*, nitrogen (N) and phosphorus (P) nutrients, and indicator bacteria (including faecal coliforms and enterococci). Monitoring is conducted monthly and therefore only occasionally captures the direct impacts of storm run-off.

No impacts of urban stormwater have been specifically noted, although many of the sites, especially those located in the Middle Waitemata Harbour (MWH) and Tamaki estuary are likely to be influenced by stormwater discharges. Analysis of the LTB-WQ saline data has shown differences in water quality between 4–6 groups of sites (Vant & Lee 1998, ARC 2000b) – the four main groups were distinguishable by the extent of contamination by land-based activities: Manukau harbour sites impacted and non-impacted by sewage discharges (two separate groups), the urban-affected MWH sites, and the clean East Coast waters. The Tamaki Estuary has the worst overall water quality of the three "special survey" water bodies, with relatively high levels of indicator bacteria, which sometimes exceed contact recreation guidelines (NIWA 2001b).

Water quality in harbours and estuaries is generally poorest at sites closest to freshwater inputs. Concentrations of nutrients, suspended solids, and indicator bacteria are generally higher and more variable at these sites, reflecting catchment inputs from streams (and associated stormwater inputs in urban areas) and a greater influence of particulates suspended by wind and wave action in the shallower estuarine waters.

Open water coastal sites have markedly better water quality, with higher clarity and dissolved oxygen concentrations, and lower concentrations of bacteria and nutrients. Sites located in the harbour channels have water quality intermediate between the estuary headwaters/arms and the open coastal waters. This is illustrated in Figure 42, which shows the influence of freshwater inflow (as indicated by reduced salinity) on two water quality indicators – faecal coliforms and dissolved oxygen.

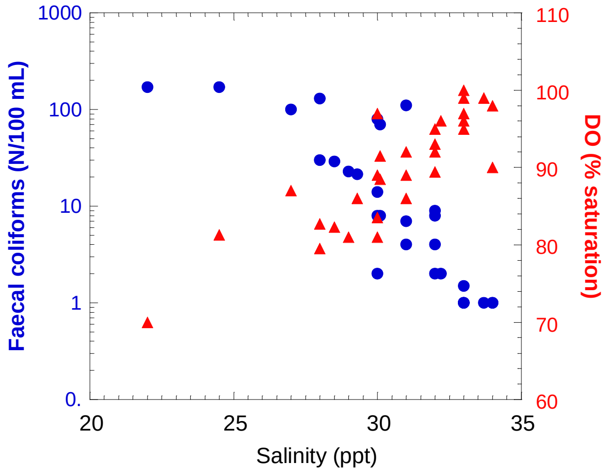


Figure 42
Freshwater inputs to Auckland's coastal waters tend to degrade marine water quality, as shown by lower dissolved oxygen concentrations (▲) and higher numbers of faecal indicator bacteria (●) as salinity decreases.

Data plotted are long-term medians taken from ARC LTB-WQ and "special survey" programmes (ARC 2000b and NIWA 2001b).

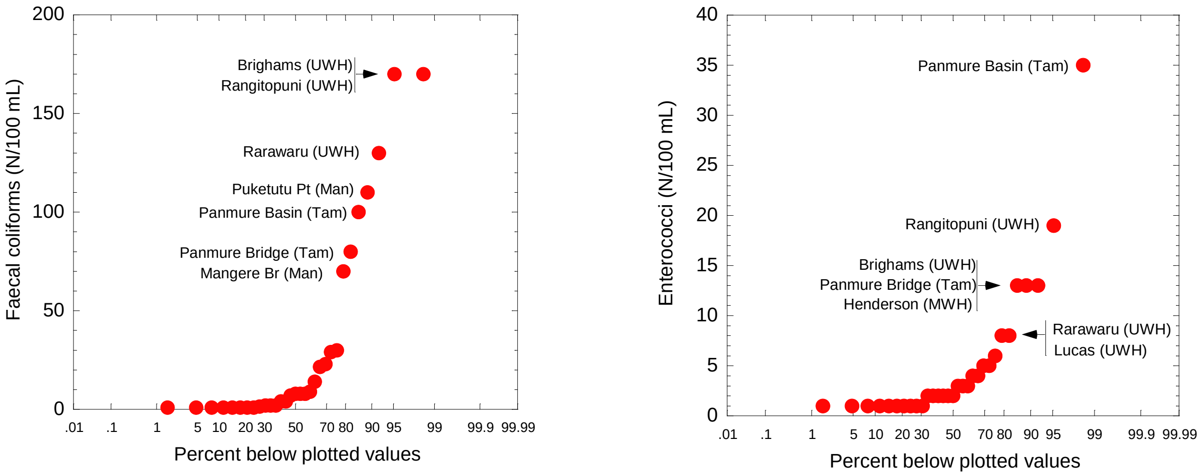
While microbial indicators are clearly elevated at some monitoring sites, long-term average levels do not exceed contact recreation guidelines (MFE 2003). Maximum numbers occasionally exceed guideline levels, with highest levels usually occurring during winter, presumably reflecting the effects of greater amounts of contaminated freshwater inflows (including stormwater in urban areas).

Long-term median faecal coliform levels are highest in the headwaters of the UWH (Brighams, Rangitopuni, and Rarawaru estuaries), reflecting inflows from predominantly rural land. Panmure Basin and Bridge sites in the Tamaki Estuary, and the sewage-influenced sites in the Manukau Harbour (Puketutu Point, Mangere Bridge and Shag Point) are slightly less contaminated, but still markedly elevated[27]. Trends for enterococci are slightly different, although UWH and Tamaki sites are still highest (Figure 43).

[27] Note that water quality in parts of the Manukau Harbour previously directly impacted by the Mangere sewage discharge has improved markedly following the commissioning of the upgraded sewage treatment plant. Ammonia, nitrite, and microbial indicators were lower in 2003 than in 1992–2000 (ARC 2004j).

Figure 43

Long-term median levels of microbial indicators in Auckland's marine waters, showing most sites have low levels of contamination. Highest contamination levels are found at inner estuary sites affected by rural and urban run-off, and the pre-modernised Mangere sewage discharge. Data are from ARC LTB-WQ and "special surveys" programmes, for 1992–2001 (NIWA 2001b) and 1987–2000 (NIWA 2000a).



Sites in the MWH, which might be expected to be affected more by urban than rural land uses, are generally lower than the UWH (mainly rural), Tamaki Estuary (urban), or inner Manukau Harbour (human sewage-affected) sites. This suggests that, apart from enclosed waters such as Panmure Basin, urban effects are less significant than rural inputs as contributors to long-term, broad-scale, microbial contamination.

Some changes in water quality over time have been detected in these programmes, but the changes are generally small, variable, and difficult to relate to any particular potential influence. The exception is the improving water quality in Manukau Harbour as a result of the Mangere WWTP upgrades (ARC 2004j, Watercare 2005).

Despite the generally localised nature of stormwater impacts measured in a number of studies (see next section), ARC's regional monitoring shows that slightly elevated residual levels of indicator bacteria remain in urbanised harbour and estuarine waters. The cumulative effects of multiple stormwater inputs are likely to be significant contributors to these elevated background levels of microbial contamination.

7.1.3 Effects of stormwater discharges on near-shore marine water quality

Most studies of the effects of stormwater discharges on near-shore marine water quality have focused on microbiological quality and consequent potential human health risks. Data for Auckland City up to 2001 were summarised by NIWA (2001a) and ACC/Metrowater (2001), using data from ARC monitoring (1991–1996), and studies conducted by Worley (2000) and Auckland Healthcare (2000). Further studies of enterococci levels and persistence at four Auckland bathing beaches were conducted by NIWA in 2002 – the results of this study, and a collation of earlier study data are summarised in NIWA (2005). North Shore City Council has also conducted comprehensive investigations; eg Mairangi Bay, which is described below (overleaf, page 94) as a case study.

Effects on water quality are variable, depending on location, amount and duration of rainfall, and prevailing weather conditions. However, general conclusions from these studies are that stormwater inflows greatly increase levels of indicator bacteria, especially during and after rainfall. Levels of indicator bacteria commonly exceeded the MFE (2003) guideline "action" level in samples taken during or after rain events at several urban beaches, indicating a significant potential public health risk.

Elevated medians, and extremely high maximum, enterococci levels (maxima up to 39,000 cfu/100 mL) were reported for several Auckland City beaches in studies conducted between 1996 and 2002 (data summarised in NIWA 2005). Measurements conducted by NIWA on three occasions at the four Auckland City Eastern Bays beaches between April and July 2002 recorded highest values (in cfu/100 mL) of 13800 at Okahu Bay, 1900 at Mission Bay, 4600 at Kohimarama Bay, and 17300 at St Heliers Bay, with significant proportions (>20 per cent) of the samples being above MFE "alert" or "action" contact recreation guidelines (NIWA 2004). Guidelines were exceeded when the near-shore marine water contained approximately 5 per cent or more stormwater.

Highest bacterial counts occurred in areas served by combined sewer networks, especially those with high incidences of wastewater overflow events (eg Cox's Bay). High counts were found to be fairly limited in spatial extent – eg high levels of contamination observed in Cox's Bay did not extend to neighbouring coastal sites at Point Chevalier or St Mary's Bay (ACC/Metrowater 2001).

Dilution and microbial die-off (UV-exposure) in marine waters are major contributing factors limiting the extent of high levels of bacterial contamination. Also important is the prevailing weather – strong winds can contain or spread stormwater plumes, and also stir up bottom sediments thereby increasing bacterial levels and persistence in the water column. Elevated bacterial indicator levels at bathing beaches seem to last for about 24 hours after rainfall (NIWA 2004; NSCC 2001). However, poor long-term water quality can persist in some areas due to stormwater inputs during both dry and wet periods, as observed at Mission Bay and Okahu Bay in Auckland Healthcare studies in 1996–1998 (Auckland Healthcare 2000).

Suspended solids' concentrations were also found to increase due to storm events at some sites, but effects were site specific and not necessarily distinguishable from the natural increase in turbidity due to resuspension of harbour sediments (ACC/Metrowater 2001).

In contrast to the generally limited spatial extent of impacts noted in these studies, North Shore City Council estimate that impacts from Wairau Creek may affect an area extending over 2 km from its entry to the marine environment (NSSC 1999). This scale of influence from major urban streams may account for the slight elevation in bacterial levels observed at some harbour and estuary sites in the ARC's regional monitoring (see previous section).

North Shore City Council water quality study: Near-Shore Water Quality Impacts of Stormwater at Mairangi Bay

A detailed study was undertaken at Mairangi Bay beach to characterise stormwater impacts on beach water quality during wet and dry weather and to provide data to calibrate receiving water quality models for East Coast Bay beaches (NSCC 2001).

Surveys measured indicator bacteria (enterococci, E. coli, and faecal coliforms) and suspended solids at 28 sampling points over three days (one fine, two wet), enabling spatial and temporal variations under a wide range of environmental conditions to be assessed.

The study found:

- There was a direct cause-effect relationship between stormwater discharge and bacterial concentrations during both wet and dry weather conditions.
- The zone of influence depends on rainfall, wind, and tidal conditions – effects were considered strong within approximately 100 m of the discharge.
- During dry weather, highest bacterial levels were encountered in shallow waters – levels at 0.5 m depth were approximately 5–15 times higher than those at 1.5 m depth. The "depth effect" was not as prevalent under rough wet weather conditions because of more effective mixing.
- Bacterial numbers remained elevated for at least 12–24 hours after rainfall. Levels were slower to drop during windy and rough conditions, due to bacteria entrained in resuspended sediment and slower die-off. In calm weather numbers remained higher in shallower waters.

Enterococci levels found in the beach waters in this study ranged from <2–4100 cfu/100 mL, depending on the event, sampling position, and time relative to rainfall. The stream water (the major source of stormwater in this location) contained 750–9200 cfu/100 mL. Lowest concentrations in the beach waters were recorded after a period of dry weather, and highest after a rain event (including a WWOF event).

It was also found that high bacterial levels accumulated in the backed-up stream during late summer. Bacteria were released into the beach waters when the stream barrier was breached by the incoming tide, causing localised impacts even in the absence of rainfall-induced stormwater discharges.

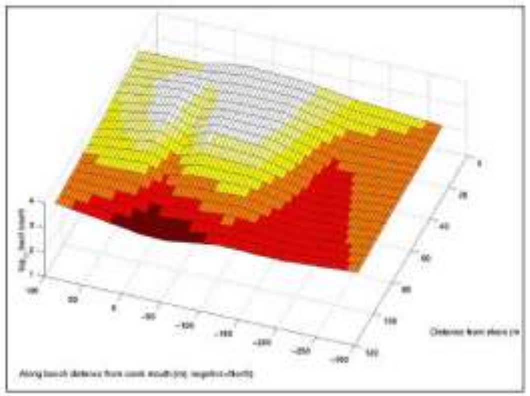
Suspended solids levels were higher in the beach waters (5–538 mg/L) than in the stream/stormwater discharges (approximately 4–114 mg/L), especially during rough weather conditions when sediment resuspension in the surf zone occurred.

Figure 44

Mairangi Bay beach, showing the stream and sampling transects used in the study.

Figure 45

Example of a 3-D plot showing the spatial variation in enterococci concentrations in beach waters around the stream entrance. Highest concentrations (white) are found near the stream mouth, and drop with distance along and away from the beach.



7.1.4 Heavy metals in the waters of Manukau Harbour

Concentrations of metals have been measured in the Manukau Harbour as part of the monitoring of the Wastewater 2000 Project upgrades to the Mangere wastewater treatment plant (Watercare 2005). While this programme is targeted at assessing the effects of the sewage plant upgrade, it provides valuable background information on contaminant concentrations in Auckland's marine waters.

Concentrations of total Cu, Pb, and Zn are summarised in Table 15 below, and show that, even in this historically polluted environment, only Cu approaches the conservative ANZECC water quality guideline. Note that the sampling was done at low tide, when dilution by cleaner outer harbour waters is at a minimum, and therefore these represent "worst case" figures.

Table 15

Concentrations (parts per billion, mg/m3) of total metals in marine waters from Manukau Harbour in the vicinity of Mangere Inlet and wastewater treatment plant. Data are site averages (n = 5), taken at low neap tide, April 2005 (Watercare 2005).

	Site[a]			ANZECC guideline
Metal	**White Bluff**	**Puketutu**	**Purukau Channel**	**95% protection**
Copper	0.91	0.86	<0.5	1.3
Lead	0.28	0.32	<0.1	4.4
Zinc	4.4	7.3	0.64	15

a. White Bluff and Puketutu sites are both approximately 3 km from the treatment plant discharge (White Bluff towards Mangere Bridge, Puketutu towards the wider Manukau Harbour), while the Purakau Channel site lies in the main body of the Manukau Harbour, approximately 8 km from the discharge.

A study more directly relevant to the fate and effects of stormwater-sourced metals in estuaries measured the magnitude of mobilisation of Cd, Cu, Fe, Mn, Pb, Zn and suspended particulate matter (SPM) in Manukau Harbour's Mangere Inlet, to assess the importance of this process in the recovery of contaminated sediments from pollution and as a source of secondary pollution to the water column (Williamson et al. 1996).

Mobilisation of contaminated surface sediments is the main source of contamination to the overlying water column, and the magnitude of mobilisation suggests that this process is very important in contaminant cycling. Mass fluxes of SPM and metals were greater during the flood (incoming) than the ebb (outgoing) tide showing that any mobilised contaminants were largely returned to the inlet, consistent with it acting as a sink for sediments and associated metals. Therefore pollution recovery by the process of mobilisation and dispersal will be slow, a result which presumably applies to the upper reaches of all Auckland's muddy estuaries.

Concentrations of filterable ("dissolved") metals in the water column were low – mostly well below USEPA (2005) water quality criteria for protection of aquatic ecosystems (Table 16), although Cu may approach chronic toxicity levels at low tide when concentrations are at their highest.

Given that Mangere Inlet is one of the more contaminated marine areas in Auckland, it is unlikely that metals in marine waters in other parts of Auckland will approach or exceed concentrations that would adversely affect aquatic life. This is consistent with stormwater quality data that indicates dilution by approximately 3–20 times would be sufficient to reduce Cu and Zn concentrations to below water quality guidelines (NIWA 2005).

Table 16

Concentrations (mg/m^3) of filterable ("dissolved") metals in marine waters from in Mangere Inlet (Williamson et al. 1996). USEPA (2005) criteria are for dissolved metals.

	Mangere inlet	USEPA criteria	ANZECC guideline
Metal	**Range**	**Chronic exposure**	**95% protection**
Cadmium	0.02–0.05	8.8	5.5
Copper	0.10–3.4	3.1	1.3
Lead	0.04–0.08	8.1	4.4
Zinc	0.09–5.4	81	15

The study also showed that the mass of SPM and metals being mobilised within the inlet was orders of magnitude higher than that entering from the catchment. For Mangere Inlet, therefore, the effects of incoming stormwater particulates and metals on water quality are small compared with effects of existing contaminated sediments mobilised by waves (wind-driven) and tidal currents.

7.1.5 Local authority bathing beach surveys

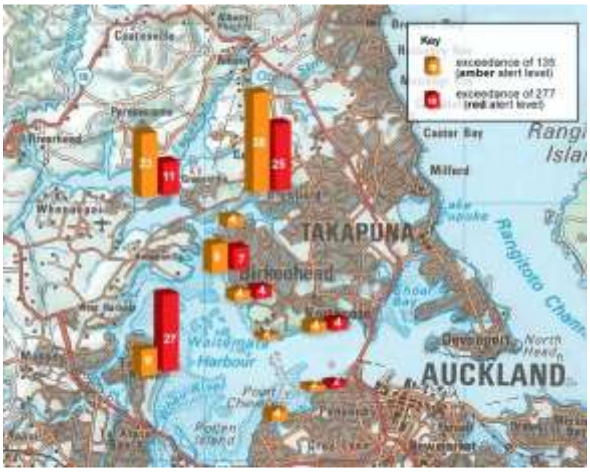
Regular beach quality monitoring is carried out by city and district councils, who warn the public when water quality poses a potential health risk. Rodney District Council, North Shore City Council and Auckland City Council have established a "Safeswim" programme, which provides consistent data on microbiological contaminant levels throughout central and northern beaches. Manukau and Waitakere cities also conduct beach surveys using very similar methodologies. In total, there are over 60 Safeswim sites and approximately 20–30 more in Waitakere (15) and Manukau (8–15). Monitoring is conducted weekly for approximately 25 weeks between October/November and March/April each year. Up to date information on these programmes can be found on the councils' websites.

Enterococci are used in marine waters as an indicator of faecal contamination that can cause diseases such as gastroenteritis and respiratory illness. Concentrations of these indicator organisms are compared with New Zealand microbiological water quality guidelines (MFE 2003) to assess whether the waters are safe for contact recreation. Exceedance of the "alert" threshold (140 enterococci per 100 mL) triggers a requirement for daily monitoring until enterococci levels drop. Beaches should be closed, and public warning signs erected, if the "action" level (280 enterococci per 100 mL) is exceeded on two consecutive days.

Microbiological water quality at most beaches over the summer bathing season is generally good, with relatively few alert or action levels recorded. However, results vary from year to year, depending largely on the number of times sampling coincides with rain events, which increase stormwater inflows and hence indicator levels. For example, Auckland City's Safeswim programme recorded only 1 per cent of 361 samples above the alert level in 2004–5 compared with 4 per cent in the previous summers (ACC 2005). As described previously, problem sites are generally downstream of catchments with significant wastewater overflow problems.

Overflows triggered most beach "closures" in North Shore City over summer 2003–4 (NSCC 2004d) and short periods of rainfall were reported to be responsible for most of the elevated levels in Manukau (MCC 2005). Data extracted from TLA records from the 2002–2004 bathing seasons, illustrates patterns in microbiological levels on beaches in the Middle Waitemata Harbour (Figure 46).

Figure 46



Alerts in the Middle Waitemata Harbour during summer monitoring periods from 2002–2004. Data extracted from TLA records.

This snapshot shows that guideline exceedances are not necessarily only attributable to urban density, as central harbour sites can show fewer alerts than other less densely urbanised upper harbour sites. However, beaches remote from urban areas (eg those north of Whangaparaoa and on the northern side of Waiheke Island) rarely, if ever, exceed the alert thresholds.

7.1.5 Summary of impacts on marine water quality

Data availability

Excellent long-term monitoring data from ARC programmes provides broad-scale information on spatial patterns and temporal trends in water quality at a range of sites across the region. However, this is not targeted at directly measuring the impacts of stormwater, and inferences must be made to assess probable effects.

An extensive bathing beach monitoring network run by TLAs gives the data required to assess potential human health impacts from contact recreation at bathing beaches. Studies by Auckland Healthcare, Auckland City Council/Metrowater (NIWA studies), and NSCC provide detailed data on the impacts of stormwater on microbiological beach water quality.

Excellent data on stormwater composition is available for Auckland City (NIWA studies), which can be used to predict likely effects in receiving waters.

There is little information on the influence of stormwater on the concentrations of heavy metals or other toxic contaminants in Auckland's marine waters. Predictions indicate this is probably of little consequence, because concentrations are low, and our knowledge would not be advanced much by undertaking more measurements.

Key impacts

Degraded microbiological water quality is currently the major identified impact of urban stormwater discharges on Auckland's coastal water quality.

Stormwater contamination restricts contact recreation at a significant number of sites around Auckland, particularly during and after rainfall. Wastewater overflows are a major source of high-level contamination. Severe effects are generally short-term (<24 hours) and localised around stormwater outlets (ca.100 m).

It appears that rural inputs are also significant in elevating microbial levels, especially in estuary headwaters receiving inflows from rural streams.

Key factors affecting the degree, extent, and duration of impacts include the weather (rainfall, wind speed and direction, cloud cover), tides, and catchment characteristics (especially numbers and volumes of wastewater overflows).

Regional monitoring programmes indicate that urban run-off contributes to a broader-scale reduction in marine water quality in Auckland's harbours and estuaries, in particular increased indicator bacterial levels, and show that these effects decline markedly in the wider harbour and coastal zones due to dilution by cleaner oceanic waters.

Other potential adverse effects on marine water quality are likely to be localised and short-lived due to dilution in the receiving waters. Stormwater is unlikely to raise the concentrations of heavy metals in marine waters to concentrations that would adversely affect aquatic life. Only localised zones next to stormwater discharges might be affected, and then only for relatively short periods during storm events and low tides.

Background concentrations of suspended solids and turbidity in estuaries (and on beaches during storms) are high, and Auckland City stormwater concentrations are fairly low. Effects are therefore unlikely to be significant. However, developing catchments, with higher suspended solids loads, may produce impacts in receiving waters. There are no data on these potential impacts to assess the scale or significance of this issue.

Cumulative effects of stormwater-sourced nutrients are unclear, but are probably less than from rural streams and run-off.

7.2 Chemical contamination of marine sediments

As described in Section 7.1, by the time stormwater reaches the coast, the key chemical contaminants are largely associated with the suspended sediments. Their fate is therefore closely linked to the fate of the suspended sediments. Sediments and associated contaminants settle out of the water in the sheltered areas of estuaries and harbours. The muddy sediment builds up over time, and so do the contaminant concentrations. At some point, the contaminant concentrations can reach levels that are toxic to the aquatic life that lives in the sediments. This has the potential to affect the ecology of affected areas, as well as the wider marine environment because sediment-dwelling (benthic) organisms serve as key food sources for animals further up the food chain (eg fish).

Sediment contamination (and consequent effects on aquatic life) is probably the major impact of stormwater in marine receiving environments and offers a robust method of tracking impacts through time and space. It has therefore been comprehensively studied and has become one of the most well defined impacts of urban stormwater in the Auckland region.

Potential issues

Build up of contaminants such as heavy metals (eg Cu, Pb, Zn) and organic pollutants (eg PAH, organochlorines) in marine sediments to levels that are toxic to marine life:

- Elevated levels of heavy metals and toxic organic contaminants in marine sediments.
- Increasing concentrations of Cu and Zn marine sediments through a combination of increasing chemical loads and/or decreasing suspended sediment loads.
- Discharge of soil-bound historical contaminants, in particular DDTs, during urbanisation of former horticultural land.

State of knowledge in 1995

By 1995, there had been extensive surveys of heavy metal levels in the Manukau and Waitemata harbours (Glasby et al. 1988), in the Waiuku Estuary as part of the NZ Steel Mill monitoring (summarised in Williamson et al. 1992a), around ports and port dredging disposal grounds, and the Mangere sewage works (eg HWQTFR 1994). However, much of this data was related to contaminant sources other than urban stormwater, or reflected inputs from industrial yard run-off and discharges.

The relatively small amount of data summarised in the 1995 review that could be attributed to urban stormwater showed very high levels of Pb, Zn, and Cu in the head of estuaries receiving stormwater (eg Wairau Creek, Milford Estuary), while metals and organochlorines (DDT, chlordane, PCBs) were also found at levels of concern in the wider estuary environment, such as the Tamaki Estuary (ARC 1992 a&b). However, these studies included sites influenced by other sources of contamination, including marinas and moored boats, industrial spillages, and port activities.

Based on the information available at the time, it was estimated that about 50 per cent of the 3500 ha marine area receiving most of Auckland's stormwater could have metals' levels exceeding sediment quality guidelines (ARC 1995). While it was acknowledged that accurate predictions were not possible because of insufficient field data, the work demonstrated that large-scale impacts on marine life were possible, and sediment contamination was a major issue.

Therefore at the end of 1995, it was clear that urban stormwater-derived contaminants were contaminating estuaries, but the extent and magnitude of this contamination were not well defined.

Advances since 1995

There have been major advances in the amount of information on sediment contamination, principally for the three major stormwater-derived metals (Cu, Pb, and Zn) and, to a lesser extent, for organic contaminants (PAH, organochlorine pesticides, PCBs).

The advances have been summarised in the following ways:

- Within-estuary distributions – how do the concentrations of contaminants vary spatially within an estuary?
- Regional distribution – how does sediment contamination vary across the region?
- Temporal trends – how rapidly are contaminant concentrations changing over time?

A key development that has enabled these questions to be tackled has been a move towards standardized procedures for undertaking contaminant surveys in estuaries (eg ARC survey protocols, ARC 2004c). The use of different sampling methods and analytical procedures in earlier studies made it difficult to directly compare contaminant data across the region or through time to assess spatial or temporal trends. Much of the earlier data has now been superseded by more consistent data, much of it collected in ARC or TLA sampling programmes since around 1998.

Also advanced since 1995 has been the development of models to predict sediment contaminant accumulation and distribution as a result of urban development. Modelling is described in Section 7.8.

7.2.1 Within-estuary distribution of contamination

Determining the spatial pattern of sediment contamination within estuaries is important for confirming how contaminants behave when discharged into estuaries. It has formed the basis for the conceptual model used in the development of the regional monitoring programme, and has enabled models predicting contaminant accumulation to be developed and tested.

In particular, these studies validated the assumption used in the region-wide monitoring approach – ie one sample location in a settling zone is considered to be reasonably representative of the average concentration within the settling zone. This greatly simplifies, and hence reduces the costs of, estuarine sediment monitoring.

Pakuranga and Hellyers estuaries – contaminant distribution and model validation

The first major study examined the spatial variability of contamination within two urbanised estuaries – Pakuranga and Hellyers/Kaipatiki Creeks (ARC 1998a, Morrisey et al. 2000). This was required in order to test the "USC1" model, which was developed to predict the build up of contaminants in estuarine sediments from urban stormwater inputs and land development scenarios (see "Modelling" section).

The study examined metal levels at different scales and within different compartments throughout these estuaries. The sampling design enabled differences in concentrations among, and between, locations to be distinguished from variations among heights on the shore and from smaller scale variation.

Transects comprising three sites at each of several heights (low tide channel, banks, mudflats, pneumatophores, and mangroves) were sampled at four different points along the longitudinal axis of each estuary. Samples were collected from the surface bioactive layer (15 cm deep) and the heavy metals Zn, Cu and Pb were analysed. In addition, small-scale variability (1m) was also measured at each site. Sampling was conducted between September 1994 and February 1995. Sampling sites for Hellyers Creek are shown in Figure 47, and contaminant data in Figure 48.

Figure 47
Sampling locations a) along and b) across Hellyers and Kaipatiki estuaries used to study contaminant distribution and USC1 model validation (ARC 1998a). Figure 48 shows contaminant distributions from this study.

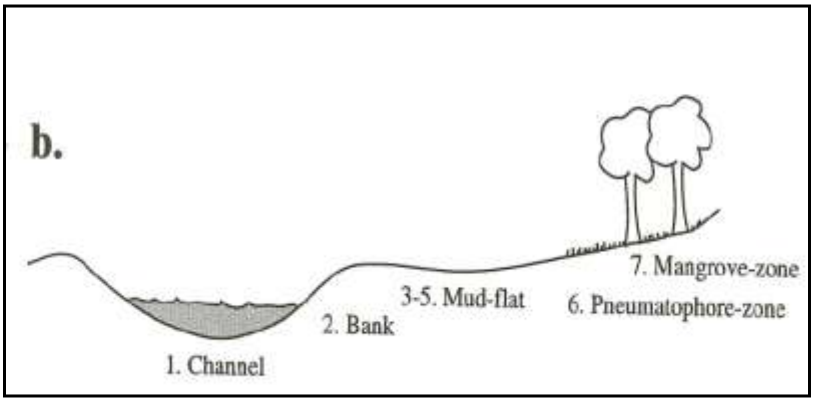
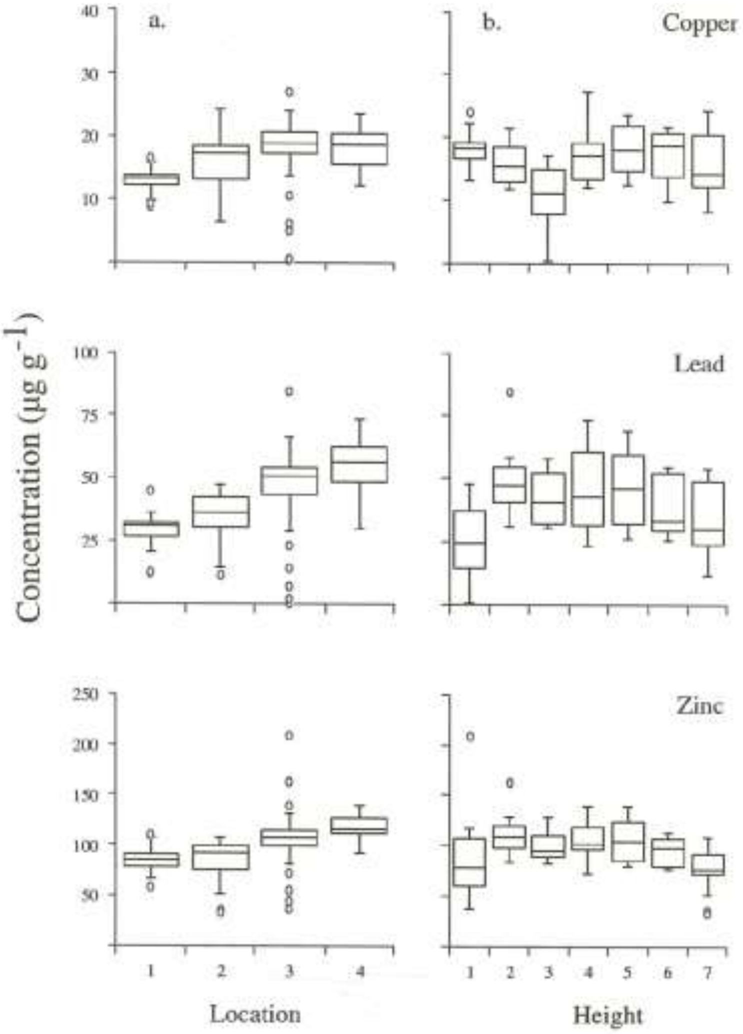


Figure 48

Concentrations of Cu, Pb, and Zn in sediments from various locations in Hellyers Creek estuary (see Figure 47 for locations). The boxes show the 25–75 percentile concentrations, with the median drawn across the box and outliers marked as ο. Figure reproduced from ARC (1998a).



The study confirmed that the model was capable of making good predictions of average concentrations of metals in the upper reaches of urbanised estuaries. It confirmed two key predictions of the USC1 model:

1. concentrations of metals were higher at locations within the settling-zone assumed by the model and decreased with distance down the estuaries; and
2. there were no clear patterns of change in concentrations up the shore from the edge of the low-tide channel to the mangrove-zone. This was consistent with another of the model's predictions – that the continual flooding and draining of the intertidal flats, coupled with wave action, would distribute contaminants evenly through the settling zone.

The sampling also showed that metals did not accumulate as much in the channel sediments as on the mud flats (as was originally predicted to occur), because currents were strong enough to prevent nett accretion of fine sediments. Concentrations of metals varied substantially between sites at any given height or location (Figure 48), and therefore properly designed sampling schemes are required account for this spatial variability.

The "four estuaries" study – contaminant distributions and biological responses

The next major study examined the longitudinal distribution of contaminants in the two urban estuaries (Pakuranga and Hellyers) described above, and in two rural estuaries, Paremoremo in the Upper Waitemata Harbour and Te Matuku Bay on Waiheke Island. Locations of these estuaries and sample plots are shown in Figure 51.

The major aims of these studies were to determine differences in biological communities within, and between, estuaries and their relationships with contaminants and natural variables (eg salinity, sediment texture). The chemical sampling programme reflected these aims.

Figure 49
Sampling in Te Matuku Bay, Waiheke Island.

The sampling strategy consisted of taking four replicates in each of six 50 m x 20 m plots along each estuary, therefore giving 24 samples from each estuary. Sampling was undertaken in May 1995 (Hellyers and Pakuranga) and March 1996 (Te Matuku and Paremoremo).

Metals and organic contaminants (PAH, organochlorine pesticides and PCBs) were analysed. This enabled the relationship between a wide range of contaminants and biological community composition to be assessed, and also filled an information gap on organic contaminant distribution in urban estuaries. Organic contaminant results are summarised in ARC (1998a), and a full account of biological responses to urban contamination in these estuaries is described in Morrissey et al (2003).

The concentrations of urban contaminants are summarised in Figure 25. Concentrations of most contaminants were higher in the urban Pakuranga and Hellyers estuaries than in Paremoremo, which in turn was more contaminated than the relatively undeveloped and remote Te Matuku Bay. Concentrations of the historically used organochlorine pesticides DDT and dieldrin were elevated in the urban estuaries, presumably reflecting inputs from agricultural soils mobilised during catchment development. DDT levels were also elevated in Paremoremo estuary, probably also the result of historical agricultural practices.

Metals and organic contaminants in these estuaries were well correlated with sediment texture (per cent mud) and organic matter (which is also related to mud content). Correlations between contaminants and texture were particularly strong in Pakuranga estuary (r values mostly >0.9 between site mean values, n=6), suggesting common source factors and similar within-estuary transport and depositional processes for metals, organics, mud, and organic matter.

Longitudinal trends in contaminant concentrations varied between estuaries (in part reflecting differences in the spacing of sampling locations) but generally showed lower concentrations at sites close to the estuary mouth than in sites in the headwater reaches (Figure 53).

Sediment texture, which can vary greatly along the estuary but generally becomes less muddy towards the mouth, was a major influencing factor on contaminant concentrations – concentrations were lower in sediments with lower mud content, which reflects settling of contaminated mud in the upper reaches of the estuaries, and/or dilution with less-contaminated sandy sediments near the mouth. Organic contaminant concentrations, especially PAHs, were correlated with organic matter content, indicating that organic matter transport has a major role in distribution of organics in estuaries.

Organic contaminant concentrations were mostly below levels that should adversely affect benthic fauna, although concentrations of organic contaminants in Pakuranga and Hellyers estuaries were relatively high compared with other NZ coastal samples (ARC 1998a). The MFE organochlorines survey (MFE 1999) found that dioxin and organochlorine pesticide concentrations in Hellyers estuary were among the highest encountered in their nationwide survey.

The conclusions from this study related to stormwater impacts were that:

- Urban estuaries had the highest concentrations of contamination, both urban-derived compounds (eg metals, PAH, PCB) and those associated with historical agricultural uses (eg DDT, dieldrin). Urban estuaries can therefore be contaminated by both current, and historical, land use practices.
- While elevated compared with other NZ sites, the organic contaminant concentrations were mostly below sediment quality guidelines (after organic

carbon normalisation[28]), and therefore would not be expected to adversely affect aquatic life.

- Sediment texture and organic matter content are key factors affecting the concentrations of metals and organic contaminants in Auckland estuaries. Comparison between sites and over time therefore needs to take account of this; eg by analysing metals in the mud fraction, as has been adopted in ARC protocols (ARC 2004c) and normalising organic contaminant concentrations to organic matter (or better, organic carbon).

Biological responses were subtle and complex, and are described in Section 7.5, "Stormwater Impacts on Benthic Ecology".

Figure 50

Upper reaches of Pakuranga Creek estuary, showing sampling sites 1 and 2 (see Figure 51).

[28] Comparison of organic contaminants with sediment quality guidelines is often done using organic carbon (OC) normalized concentrations to account for the effect of variable amounts of organic matter and consequent effects on contaminant bioavailability and toxicity – e.g. ARC ERC are specified for concentrations normalized to 1 per cent OC.

Figure: 51 Locations of the "four estuaries", and sampling locations within each estuary.

Figure 52

Concentrations of heavy metals and organic contaminants in two urban estuaries (Hellyers – Hel and Pakuranga – Pak), the primarily rural Paremoremo (Par), and the relatively undeveloped/rural Te Matuku Bay, Waiheke Island (Wai). Data are means ± SE in means (n = 6 per estuary).

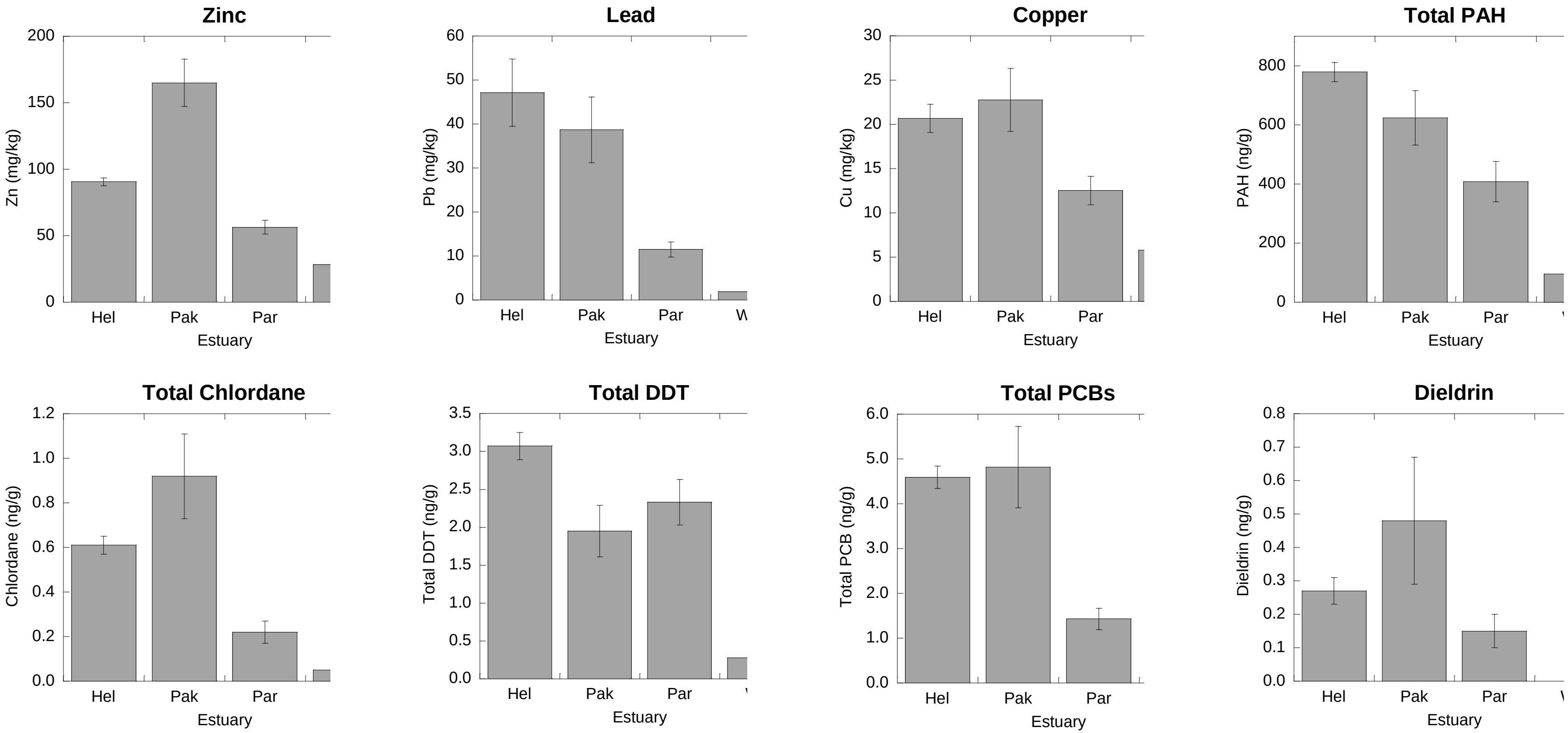
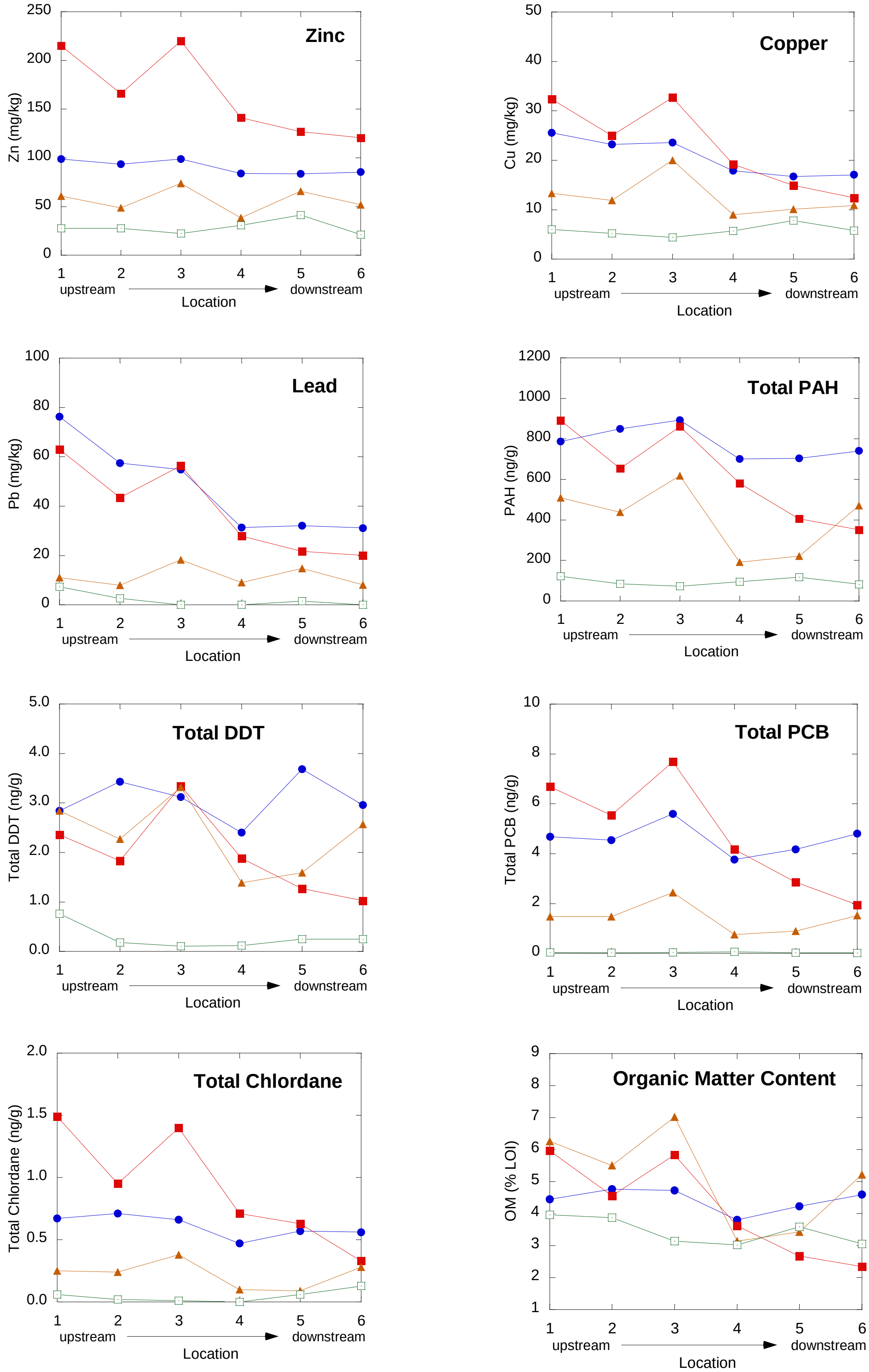


Figure 53 Longitudinal trends in sediment contaminant concentrations and organic matter in the "four estuaries" – Hellyers (●) Te Matuku (◘) Pakuranga (■) and Paremoremo (▲). Data are means at each location (n = 4).



Henderson Creek – monitoring contaminants in mangrove-filled estuaries

The findings described above are applicable to many estuarine arms in Auckland, which have "mudflats"; ie well-defined, large, stable settling zones where mud and contaminants accumulate.

However, Henderson Creek estuary[29] is distinctly different from many other Auckland estuaries. It has very small mudflat zones, and large mangrove flats perched high above the low tide level, which only become inundated at the peak of the spring tide.

Figure 54

Mangroves at Henderson Creek.

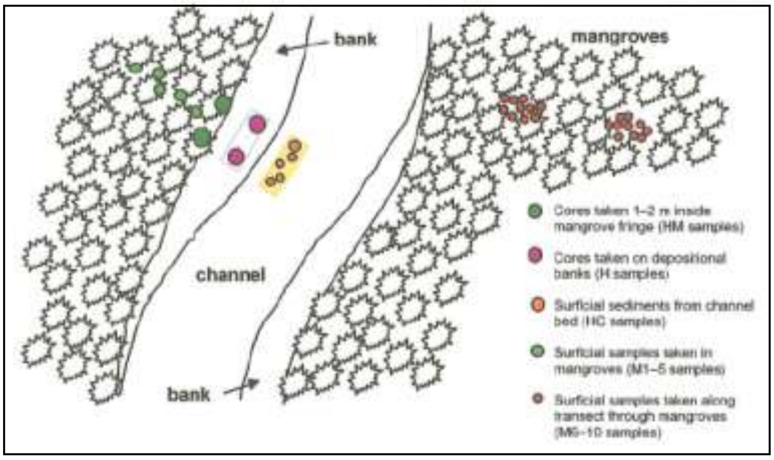
The low tide channel dissects this perched flat, and is "riverine" in appearance with eroding inner bends and sediment bars building up on the outer bends. These are characteristics of "mature" estuaries, at an advanced stage of infilling from catchment-sourced sediment.

The lack of stable, depositional, mudflats presents a challenge for sediment contaminant monitoring. While the mangrove areas are depositional areas, they are appear to receive sediment only near the top of the highest tides, and so may be poor monitoring sites. Diffuse Sources Ltd investigated the upper reaches of Henderson Creek estuary to determine a suitable long-term monitoring strategy for Waitakere City (EVA et al. 2003d).

[29] The nearby Huruhuru Creek branch of the estuary, which drains another part of the wider Henderson catchment, is similar to the upper Henderson Creek estuary described here.

Figure 55

Henderson Creek estuary, showing the five sampling locations along the channel. The schematic shows the range of samples used to investigate heavy metal concentrations in the different parts of the estuary (EVA et al. 2003d).

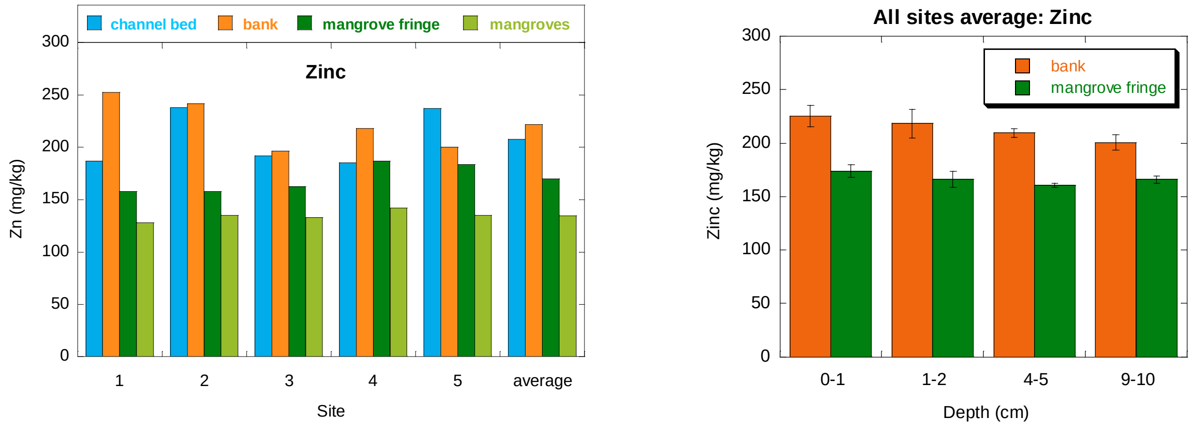


The stream channel bed and banks, the mangrove fringe, and the broader mangrove areas were sampled at five locations along a several hundred meter reach of the estuary (Figure 55). Shallow cores (10 cm) were also taken to determine a suitable sampling depth.

The survey found that concentrations of heavy metals in the stream bank and bed sediments were higher, but slightly more variable, than in the mangroves (Figure 56). The data indicated that the mangrove areas, although a major part of the estuary, would probably not be suitable for long-term trend monitoring of catchment inputs to the estuary. Metal concentrations changed little with depth over the top 10 cm sampled (ca.10 per cent for Zn). A 0–2 cm depth interval was found suitable, and small variations in sampling depth would make little difference in the results.

Figure 56

Zinc concentrations in different physical environments, and in shallow cores, from Henderson Creek estuary (EVA et al. 2003d).



While the bed sediments were coarser textured and more mobile than the bank sediments, analysis of metals in the mud fraction (<63 m) gave consistent data that may provide a useful record of contaminants recently mobilised from the catchment. The estuary bed and bank sediments were selected for on-going monitoring, using multiple replicates and compositing from five sites over a few hundred meter length of the estuary to obtain a robust representation of contaminant status.

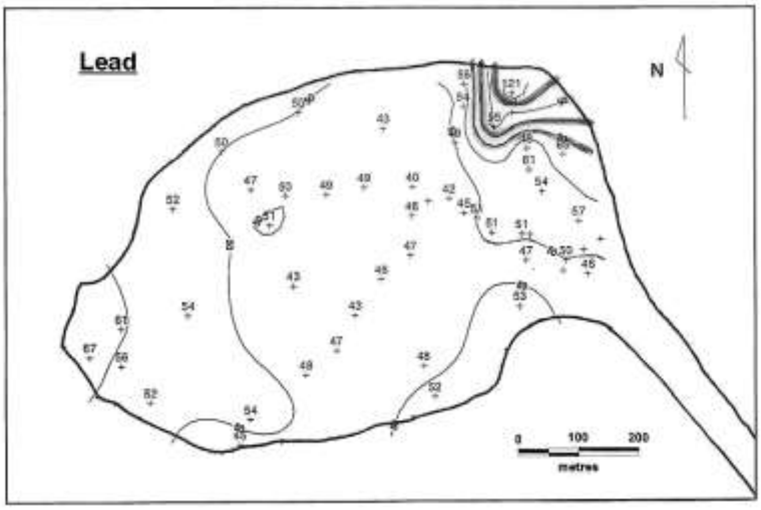
This sampling approach represents a significant variation for Auckland estuaries, brought about the lack of suitable stable mud-flat sites. If successful, it may be useful to apply elsewhere in Auckland, as mangrove encroachment changes the structure of some established estuary monitoring sites.

Contamination of marine sediments near stormwater discharge points

Contaminant levels in marine sediments can be markedly elevated near stormwater outfalls. Concentrations fall with distance away from the discharge points, the nature of the gradient being determined by factors such as the receiving environment exposure and energy (which control contaminant dispersion), and the discharge characteristics (volumes, level of contamination etc). Because of these potentially strong concentration gradients, routine monitoring near stormwater outfalls is not recommended (ARC 2004c).

An example of contamination around an outfall discharging into a sheltered muddy estuarine basin is given in Parker et al. (2002) for Panmure Basin, where stormwater inflows on the northern shore create localised enrichment of heavy metals (shown for Pb in Figure 57. Zn and Cu profiles are similar). In the rest of the basin, Pb contamination is fairly evenly spread, which is probably the result of the resuspension, tidal advection, deposition processes that occur in the settling zone.

Figure 57
The distribution of Pb in Panmure Basin sediments (from Parker et al. 2002). Numbers are concentrations in mg/kg.



Heavy metal contamination of sandy "outer zones" – Motions Bay

As described in previous sections, contaminant accumulation in muddy urban estuaries had been well established. Beyond the sheltered settling zones (or "primary deposition areas" or PDAs; ARC 2002a and b), lies a more variable environment, where contaminated mud can be deposited during storms, but subsequently redistributed by wave and currents to lower energy depositional environments (eg the muddy estuary arms and channel banks).

These areas, termed the "secondary redistribution area" (SRA – ARC 2002a), represent the interface between the contaminated muddy estuary arms and the less contaminated and potentially more sensitive sandy harbour environments. Understanding the distribution of contaminants in the SRA and how best to monitor them was required.

To address this issue, the degree of contamination of Motions Bay was surveyed to determine contaminant variability in a variably textured, but generally sandy, SRA known to be impacted by stormwater run-off (in this case from Motions Creek).

Zinc was measured in the total sediment and in the mud fraction (<63 m) in samples collected over the intertidal area that would be subject to deposition during storm events. Samples were collected between the foreshore and the furthest extent of the intertidal area (which stretches to nearly the end of Te Tokaroa Reef). The outermost sites were about 50 m past the ARC Long-term Monitoring site. Results are appended to ARC (2002a).

The data showed that:

- Zn levels in the sandy sediments were generally low, because of the low level of contamination present in the sand fraction. Total Zn concentrations varied considerably throughout the intertidal area reflecting the variability in sediment texture.

- The concentration of Zn in the mud fraction was far less variable than in the total sediment, reflecting the greater concentration of metals in the mud (rather than the sand) fraction.
- Muddy sediments in the SRA can be highly contaminated, reflecting transport and deposition of highly contaminated muds from the upstream PDA or settling zone during storms.
- The level of contamination in the mud fraction was similar across the SRA, and seemed to be fairly independent of the degree of muddiness of the sediment.

These results indicated that monitoring of the mud fraction is therefore likely to give more consistent results than analysis of the total sediment in a SRA, and that there is the potential for impacts on aquatic life from highly contaminated muds temporarily deposited in storms.

Distribution of PAH within estuarine sediments

Estuarine sediments are comprised of particles covering a wide range of sizes, from <1 µm to over 1 µmm. Most muddy estuaries have median particle sizes <200 µm (ARC 2004b). Transport and deposition are affected by particle size and density – larger particles generally require greater energies to transport, and they are deposited more readily than finer sediments. Fine muds are more easily transported, but once deposited they are more cohesive than coarser sediments and tend to stick to themselves and other particles, forming stable banks and channels. Large particles can be mineral (eg sand, gravel) or organic (plant detritus, coal fragments, bitumen etc). Organic particles are less dense than mineral grains, and can be distributed more widely by currents in the receiving waters.

Determining how contaminants are distributed among the particles making up sediment is useful because this influences contaminant transport and deposition (and hence fate in the receiving environment), treatability in treatment devices, and also the potential bioavailability to aquatic organisms.

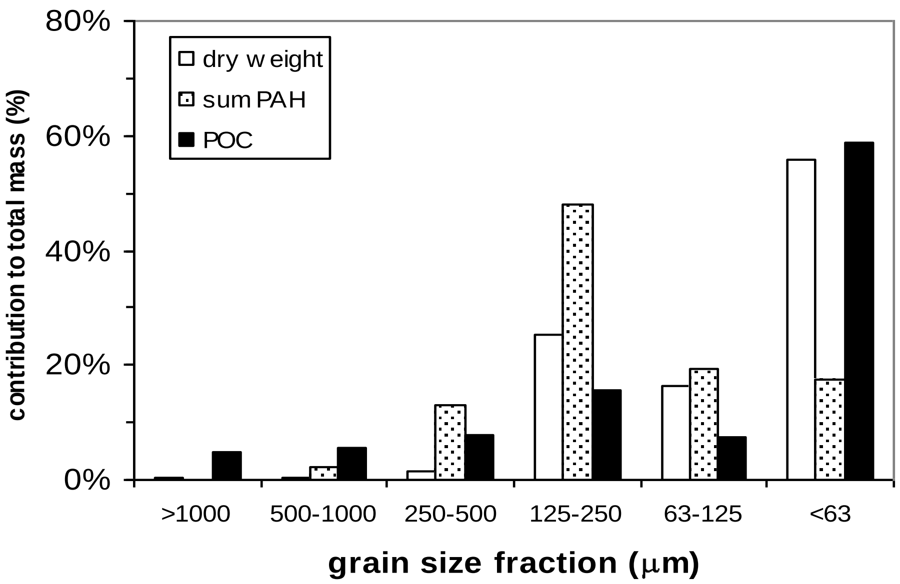
A detailed study of PAH distributions in sediment from Motions Creek (Depree & Ahrens 2003, Ahrens & Depree 2004) found that:

- Highest PAH concentrations were present in the medium size fraction (125–250 µm), and were preferentially enriched in low-density.(<2.15 g/cm^3) organic material which comprised 3 per cent of the sediment mass, but contained over 75 per cent of the total mass of PAH. Figure 58 summarises the distribution of PAH in the sediment.
- PAH concentrations in the low-density material were very high – far higher than common urban PAH sources such as road dust, bitumen, used oil or coal. Coal tar residues were proposed as a plausible source of the contamination.

These findings raise the possibility of PAHs being more readily transported in the aquatic environment than might be predicted assuming PAH was bound to denser mineral fractions. The findings are also relevant to stormwater treatment in that effective PAH removal for this catchment might best be achieved by targeting removal of low-density, relatively coarse materials.

Figure 58

Relative contribution of various particle size fractions to sediment dry mass, total PAH concentrations, and particulate organic carbon (POC) content in sediment from Motions Creek estuary (Ahrens & Depree 2004).



Analysis of marine sediments from five Auckland sites (Motions, Meola, Hobson Bay, Cheltenham, and Pakuranga) with different textures and PAH levels by NIWA (unpublished data, summarised in Table 17 below and Figure 59) also found that PAH concentrations on the coarser sized particles (>0.25 mm) were higher than on the finer particles at all sites, and that distributions differed substantially between the sediments (influenced by particle size distribution at each site).

Clearly, PAH distribution in marine sediments is complex and varies between sites. This may affect fate processes in the receiving environment.

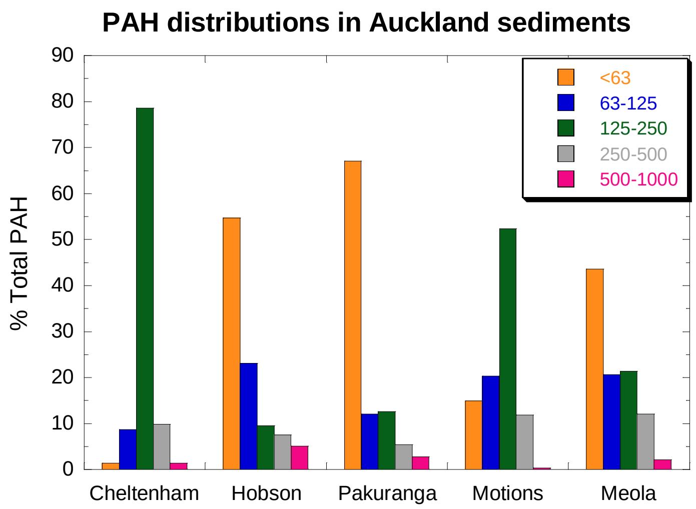
Table 17

PAH distributions among a range of particle size fractions in Auckland marine sediments (NIWA unpublished data).

		Total PAH concentration (ng/g)				
Site	Texture	<63 µm	63–125 µm	125–250 µm	250–500 µm	500–1000 µm
Cheltenham	Sand	480	66	383	1453	6478
Hobson	Muddy sand	1439	114	44	441	6137
Pakuranga	Mud	593	501	670	1830	6455
Motions	Sandy mud	3676	5894	9616	19684	5845
Meola	Mud	2551	10952	18186	72576	100353

Figure 59

Distribution of PAHs between various particle size ranges in marine sediments from five urban Auckland locations (NIWA, unpublished data).



7.2.2 Regional distribution of sediment contamination

How does contamination vary between estuaries across the Auckland region? As noted in the previous section, contaminant concentrations can vary substantially within an estuary, depending onsite location, sampling procedures, and analytical methodologies. Standardised approaches are therefore required to obtain a reliable picture of contaminant status at different places across the region. The introduction of these procedures (documented in ARC's "blueprint for monitoring", ARC 2004c) in 1998 has enabled a reliable picture to be constructed.

Many of the major studies have been incorporated in a review conducted under the Regional Discharges Project (RDP). Contaminant data from throughout Auckland on Pb, Zn, Cu and PAH were reported in tables and maps by ARC (2003d). Data were summarized from:

- ARC State of Environment Monitoring (see "Temporal Trends" below).
- North Shore City Sediment Survey (URS 2002).
- Auckland Strategic Plan (ARC, unpublished "ASP" survey results from the mid-1990s).
- Auckland City Council (Metrowater) sediment surveys conducted in 2002 (NIWA 2004).
- Sampling undertaken as part of the on-going ARC RDP process.

Sites were ranked in terms of the Environmental Response Criteria (ERC) given in the Auckland Regional Plan: Coastal (Table 20.1.A) and graded using the "traffic light" system to assess potential risks to aquatic fauna:

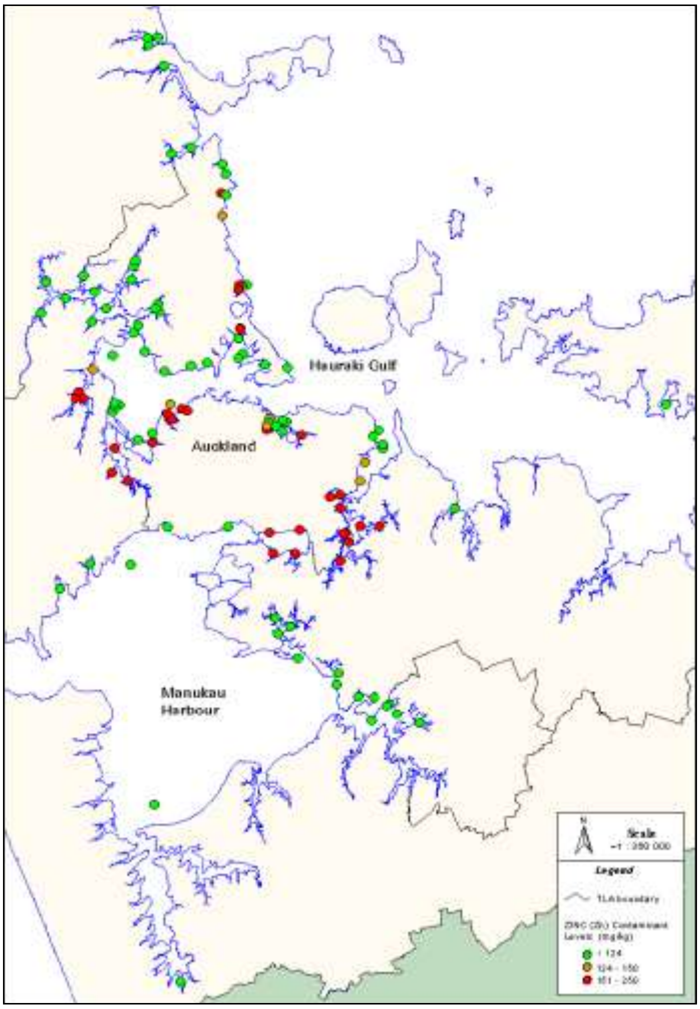
ARC traffic light criteria

Concentration in sediments and water are compared against values reflecting risks to animals that live in the sediments
Green – low risk
Amber – warning of moderate risk, high priority to intervene
Red – high risk, check for adverse effects, investigate "fix-it" options

Of the 72 RDP sites, 32 were ranked "green", 18 "amber" and 22 "red". Most of the red sites were near older, fully urbanised areas. The regional picture that has emerged is described in ARC (2003d), and shown in Figure 55 for Zn, which is currently considered the primary contaminant of concern in Auckland's estuaries. Similar maps for Cu and Pb are given in ARC (2003d). Figures 60 to 64 provide more details for Auckland's main harbours.

Figure 60

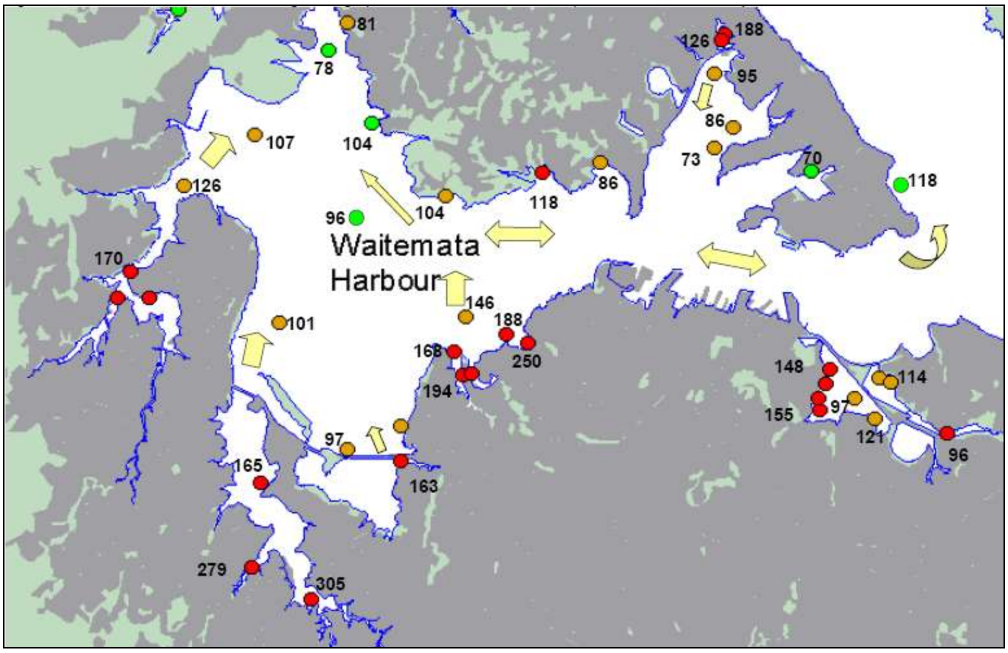
Regional distribution of sediment contamination as reflected by zinc concentrations (mg/kg), using the ERC "traffic light" grading system (ARC 2003d).



The Middle Waitemata Harbour is widely contaminated (Figure 61). Clear contamination gradients extend out from settling zones into adjacent outer zones. As with the other water bodies, the concentrations of the three major contaminants Zn, Pb and Cu are reasonably correlated, which suggests that they are distributed in a similar manner – probably by resuspension and dispersal of fine particulates. There is a slight variation in Zn/Pb ratios and this may be due to reducing Pb levels following the removal of Pb from petrol in 1996. Contamination is probably carried into the wider outer zone such that Zn levels now exceed 100 mg/kg over much of the harbour.

Figure 61

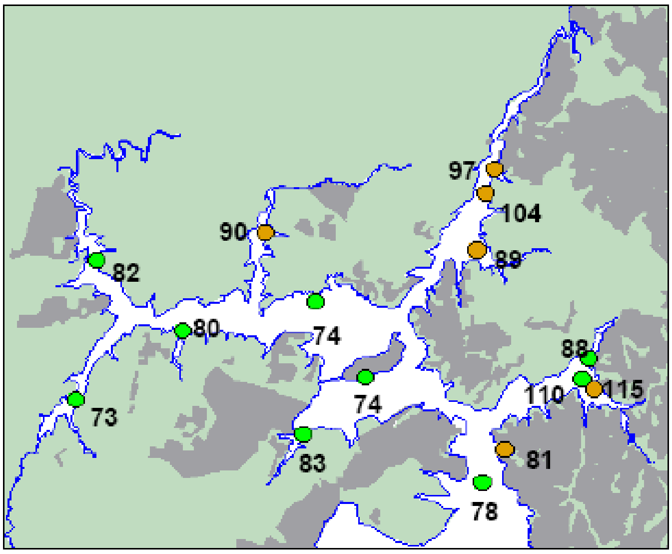
Zinc concentrations (mg/kg) in sediments from the Middle Waitemata Harbour (ARC 2003d).



The Upper Waitemata Harbour (Figure 62) shows a range of contamination. Most of the Upper Harbour remains relatively uncontaminated. Concentrations of heavy metals in the sediments are mostly "green" except in the urbanised branches of Lucas and Hellyers estuaries where they are "amber'. Concentrations of PAH are "green" throughout the estuary. Legacy pesticides (eg DDT and dieldrin) are found at low levels ("green") throughout the whole estuary and derive from use in the horticultural and farming areas in the 1940s–1970s, which has left their residues in the soils. Modern earth working operations associated with urbanisation can result in these soils being eroded and washed into the estuary along with their legacy pesticide contaminants.

Figure 62

Zinc concentrations (mg/kg) in sediments from the Upper Waitemata Harbour (ARC 2003d).



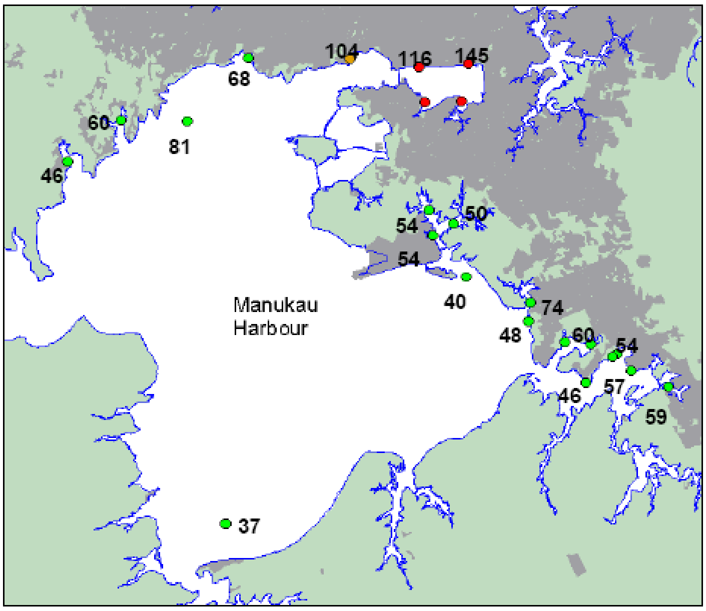
The concentrations of heavy metals in the mud fraction are low throughout the Manukau Harbour (Figure 63), with the exception of Mangere Inlet. Concentrations are much lower than in the Waitemata, even than those parts of the Waitemata that are predominantly rural (eg Upper Waitemata Harbour). Concentrations in the Pahurehure Inlet and along the southern shore are probably close to background. The reasons for the low concentrations are a mixture of the factors:

- large harbour;
- small urban catchments;
- recent urban development; and
- relatively large rural catchments.

Concentrations in the most contaminated area in the Manukau, Mangere Inlet, may be partly due to historical industrial pollution (see Section 7.4, Figure 75), which is being buried by recent sedimentation but will be continued to be mixed upwards to some extent into the surface layers by bioturbation. Another possibility is that it is partly due to high background concentrations, but this requires further investigation. The rate of change of concentrations at the two SoE sites (Cemetery and Anns Creek) is slow, which is consistent with the small catchment and relatively large estuary.

Figure 63

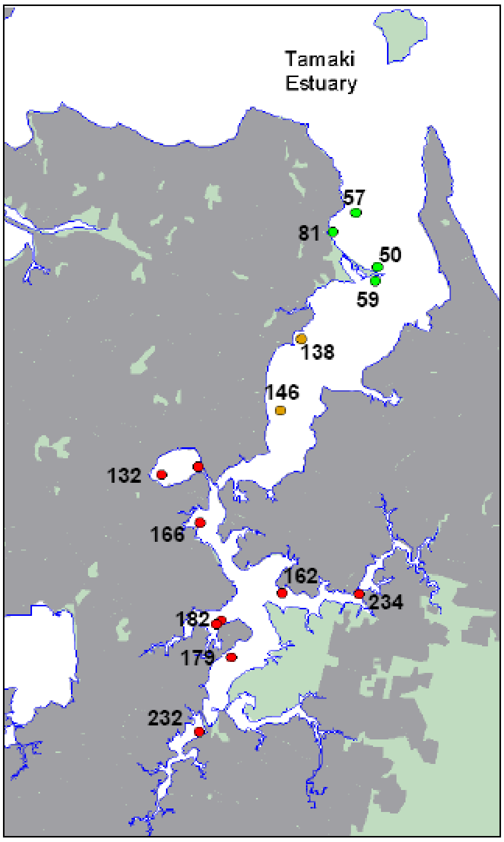
Zinc concentrations (mg/kg) in sediments from Manukau Harbour (ARC 2003d).



The pattern of contamination in **Tamaki Estuary** looks simple – there is a clear gradient from the contaminated urbanised headwaters (Middlemore, Pakuranga, Otahuhu, Panmure) to the uncontaminated mouth (Figure 64). It appears that the middle reaches are becoming contaminated. The evidence suggests that the outer Tamaki (north of Tahuna Torea) is relatively uncontaminated – ie the contamination from the upper estuary is not reaching the outer Tamaki area.

Figure 64

Zinc concentrations (mg/kg) in sediments from Tamaki Estuary (ARC 2003d).



Estuaries to the north of Auckland, including those whose watersheds are within the Metropolitan Urban Limit (MUL) – eg Orewa, Weiti – or have rural centres – eg Mahurangi – are relatively uncontaminated. Parts of Orewa and Weiti estuaries show low levels of contamination by Zn and Cu.

The **East Coast Bays** area discharges into the open coast. Contaminant build up is not regarded as a problem here because of the relatively high wave energy, which tends to disperse fine sediments and their associated contaminants widely. This is confirmed from observations from RDP sites and from studies on metal levels further offshore as part of the Rosedale Outfall study (KMA 1999).

Persistent organic pollutants (POPs)

PAHs have been monitored as part of the ARC SoE sediment-monitoring programme. Concentrations are generally low ("green"), apart from a few scattered sites[30], which may have been affected by historical activities, including a gas works (Little Shoal Bay) and the use of coal tar binders in catchment roading in older urban catchments such as Motions Creek (Depree & Ahrens 2003). In general, the PAH levels in Auckland's

[30] Including Motions, Meola, one site in Hobson Bay, Coxes Bay, Chelsea, Little Shoal Bay, and Wairau Creek.

estuaries are not of as much concern as Zn, as monitoring suggests that if concentrations are changing at all, these changes are slow and variable.

Other POPs that have been monitored at a range of urban sites are organochlorine pesticides (OCPs) and polychlorinated biphenyls (PCBs). Low concentrations of OCPs (mostly DDE and DDD, the degradation products of DDT) and PCBs are commonly found at urban and some rural sites (eg ARC 1992a and b, NIWA 1994b and 1995d; ARC 1998a). Some data, for the "four estuaries" study, are shown in Section 7.2.1 above, and data from a regional review conducted in 2002 are summarised in Figure 65 (EVA 2003a).

Levels of POPs are generally well below ERC, although DDTs can approach (or, in places, even exceed) ERC in some estuaries (eg Hellyers), and localised "hot spots" with high concentrations can be encountered (eg in the Tamaki Estuary; Nipper et al 1998).

A notable exception to the generally low level of POP contamination is the relatively high concentration of DDTs found in Henderson Creek estuary. This is probably the result of soil losses from extensive horticultural areas in this catchment, possibly including during urbanisation. This is described further in Section 7.4, Deciphering the History of Urban Stormwater Impacts.

Much higher concentrations are commonly found close to point source contaminated areas, in particular ports and wharfs (eg Ports of Auckland, Devonport Naval Base). However, these areas are subjected to disturbance from dredging and boating activities, so it is difficult to predict what current levels are from old surveys.

Organochlorines were monitored at the 27 ARC SoE sites in 2003 (ARC 2004k). Most compounds were below analytical detection limits, with occasional detections of DDTs from some sites. The low rate of detection is a reflection of higher analytical detection limits than those used in earlier surveys (eg the "four estuaries" study, and earlier studies of the Tamaki Estuary; ARC 1992b) rather than a decrease in the levels of these contaminants over time.

Figure 65

Probability plots of the concentrations of organic contaminants in urban estuarine sediments. ANZECC Interim Sediment Quality Guideline-Low (ISQG-low) and ARC Sediment Quality Guidelines (ARC SQG-low – now called ERC) are shown for comparison. Note that the comparisons with SQGs assume sediment total organic carbon (TOC) level of 1 per cent, which is likely to be conservative for muddy estuarine sediments. Also note that the distribution of the data, and proportion of sites exceeding the SQGs, depends on the range of sites monitored. There may be a bias towards more highly contaminated sites, because these were generally the focus of most studies before the introduction of the region-wide ARC sediment monitoring programme.

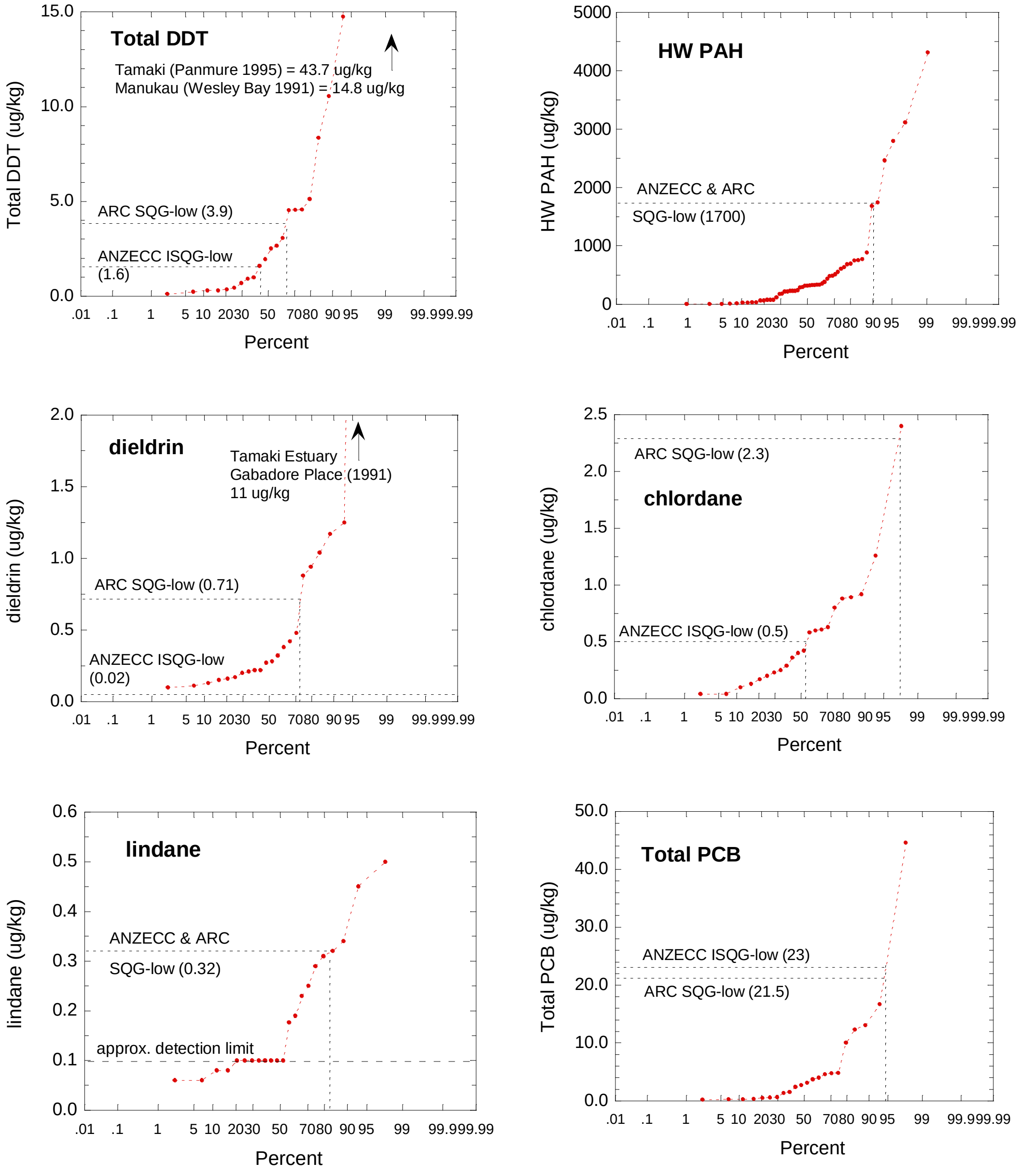


Figure reproduced from EVA et al. (2003a).

7.2.2 Temporal trends in sediment contamination: ARC monitoring

Temporal trends (trends over time) have been assessed in two ways:

- Analysis of contaminant depth profiles from estuarine sediment cores; and
- Regular on-going monitoring of surficial sediments from established monitoring sites.

Information obtained from coring is covered in Section 7.4, Deciphering the History of Urban Impacts. The rest of this section considers on-going trend monitoring by ARC's SoE marine sediment monitoring programme.

The ARC SoE sediment monitoring programme involves collecting composite samples from 27 estuarine and harbour sites, mostly close to urban areas, throughout the Auckland region. These samples are analysed for heavy metals (Cu, Pb, Zn), PAH, and particle size distribution. Samples have been collected in 1998, 1999, 2001, and 2003–2005 samples were being processed at the time of this review. Data are reported in TPs 107, 135, 192, and 246 (ARC 1999d, 2000c, 2002f, and 2004k). A review of the 1998–2001 results and monitoring procedures was conducted in 2002 (ARC 2002g).

While this programme examines the regional distribution of contaminants Zn, Cu, Pb and PAH (and these results have been incorporated into the RDP regional distribution picture described above), the primary focus is to detect changes in concentrations through time, so that informed management decisions can be made, and eventually, to enable the effects of management response to catchment pressures to be measured.

Trend analysis was initially reported for the 1998–2001 samplings (ARC 2002g) and has since been updated with the 2003 data (ARC 2004k). A selection of typical trend plots for the 1998–2001 period are shown in Figure 66.

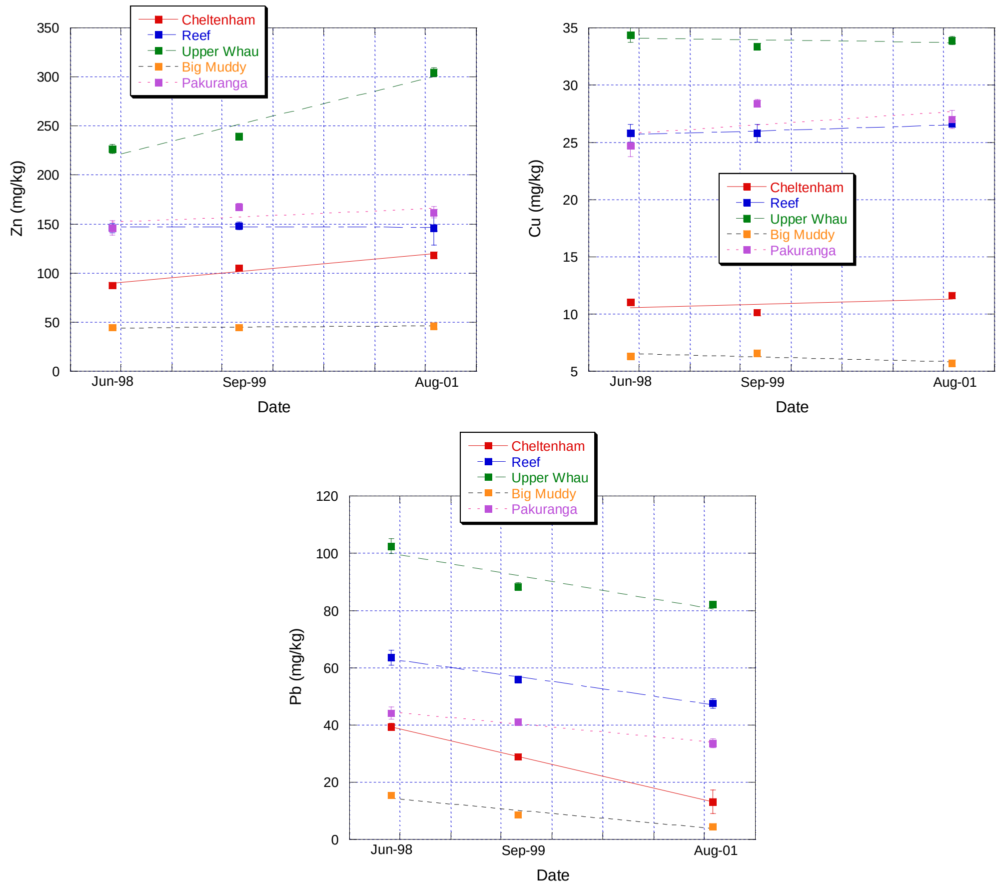
The initial analysis, to 2001, showed that:

- Zn concentrations at most sites were increasing but at a few sites they were not changing. At only one site, Ann's Creek, were zinc concentrations decreasing. The sites with the highest rates of increase, about 24 mg kg^{-1} a^{-1}, were those in the upper Whau River estuary.
- Copper concentrations also generally increased, at rates about 1/8th of those for zinc. This is consistent with the Zn:Cu ratio in stormwater (NIWA 2005). Exceptions to this correlation were the two upper Whau River estuary sites where copper concentrations were either not changing or were increasing only very slowly despite the rapid increase in zinc concentrations. The third exception was the Ann's Creek site where copper concentrations are decreasing.
- Lead concentrations were decreasing at all sites, probably because of the large reduction of lead in petrol in 1996.
- Addition of the 2003 sample data showed that:
- Zn concentrations continued to increase at all sites except Hobson Bay. Increases were most marked at urban settling zone sites. The previously recorded

decreasing trend at Annes Creek was reversed, so that an increasing Zn trend became apparent. Conversely, the Hobson Bay site recorded a decrease.

- Cu continued to increase at all sites except Te Tokoroa and Pakuranga, where concentrations appeared to stabilize, and at Hobson Bay where a decreasing trend emerged.
- Pb, which was previously found to be decreasing at all sites, stabilised so that 2003 concentrations were similar to those recorded in 1998. Only Hobson Bay continued to decrease. The lack of a continuing decreasing trend in Pb was not explained, but it suggested that the removal of Pb from petrol and restrictions on Pb paints may not be primarily responsible.

Figure 66
Trends in heavy metal concentrations at five sites for the period 1998–2001. Data is from the ARC SoE sediment monitoring programme (ARC 2002g).



Note: Cheltenham is a sandy urban beach; Reef is Te Tokoroa Reef, a muddy urban harbour site off Meola Creek estuary in the Waitemata Harbour; Upper Whau is a muddy upper estuarine site in a heavily urbanised/industrialised catchment; Big Muddy is a muddy estuary with an essentially undeveloped bush catchment in the outer Manukau Harbour; and Pakuranga is an urbanised muddy estuary.

Trends in PAH concentrations are much less obvious than those for the metals, with the PAH trends about equally divided between increasing, decreasing and not changing. Because of the high variance of the replicate analysis relative to the low concentrations being measured in most samples, the trends are very small and are significant for only a few sites.

The SoE trend monitoring is still in its infancy, having only a few data points upon which to assess trends. The current results are therefore somewhat sensitive to short-term, year to year, variations that initially can alter the magnitude or significance of emerging trends. Trend analysis will become more robust over time, as more sampling data points are added. The fifth set of data was collected in 2005, but had been reported at the time of this review.

While the magnitude of some trends is still somewhat uncertain, it is quite clear that Zn concentrations are increasing in receiving environment sediments. Reducing Zn run-off is required to slow or halt this trend and prevent irreversible contamination of the marine environment.

Additional temporal trend studies have been commenced in Henderson Creek estuary by Waitakere City, at a site located further upstream from the ARC SoE site (see Section 7.2.1, "Within-estuary distributions"). The first sampling was completed in 2003 (EVA et al. 2004b) and it is due for resampling in summer 2005-06 (reported in EVA et al. 2006b).

Note also that ARC conducts trend monitoring using shellfish as indicators of receiving water quality. This complementary approach is described in Section 7.3 (Contamination of shellfish and fish).

7.2.4 Emerging contaminants

A range of contaminants that have not yet been widely investigated in Auckland are proving to be of concern for potential toxicity and bioaccumulation-related impacts in overseas urban estuaries. Some of these contaminants are known to be present in Auckland's estuaries (eg Hg and PCBs), while the concentrations of others (eg phthalates and PBDEs) are currently unknown. Further investigation of the following is recommended, initially by desk-top review to determine whether they are likely to be important in Auckland, and if so, what the potential effects might be and where further studies should be targeted.

Mercury

Hg levels in Auckland are elevated over background, are close to sediment quality criteria, and seem to be increasing (Kennedy & Gadd 2001). The sources are not obvious, but could include dentistry (eg rinse waters finding their way to combined sewer overflows), fluorescent lights, electrical switches in motor vehicles, and fungicidal paints in building materials.

PBDEs

Polybrominated diphenyl ethers (PBDE) are widely used flame-retardants and are found in many products such as wood products (building materials, furniture), plastics, and fabrics (curtains, carpets, wallpapers). They therefore have a very wide distribution in the environment and have similar physical properties to PCBs – they are persistent, bioaccumulative, and toxic. PBDEs have been detected at high concentrations in people and wildlife in the USA and Europe, leading to their withdrawal from many products. Like PCBs, however, there is a large existing contaminant reservoir in urban areas that provides an on-going source of contamination to the environment.

Phthalate esters

Phthalate esters are used in most plastics, especially in PVC – they aid the extrusion process and provide desirable physical properties such as flexibility. In the city of Tacoma, near Seattle, high levels are found in stormwater run-off from urban areas, especially commercial areas. Phthalates are more biodegradable and less accumulative than organochlorines such as PCBs, but they are also present in much higher quantities.

PCBs

PCBs are a diverse group of environmentally persistent chlorinated aromatic compounds with excellent thermal cooling, non-flammability, and electrical insulating properties that made them widely used in electrical goods. They are also bioaccumulative and toxic to humans and wildlife. Use was banned in NZ in 1995, so levels should be declining over time. While PCB concentrations in Auckland estuaries do not exceed toxicity SQG (eg the ARC ERC), they may exceed clean-up goals set in some overseas estuaries to protect higher food chain organisms such as human consumers and endangered wildlife – eg San Francisco Bay has a fish advisory for PCBs, and there is evidence that the PCB levels are endangering some Bay margin birds. The "clean-up" goal in San Francisco Bay is 2 µg/kg; a level which is often exceeded in parts of the Waitemata Harbour and other urban estuaries (eg Pakuranga, Hellyers). The worst pollution appears to be associated with the manufacture of PCBs, which did not occur in NZ. However, high levels of contamination are associated with spills of transformer oils, and use of transformer oils for painting untreated wood or dust control. Recent detailed work in the USA has found them to be a common contaminant of building materials, such as window caulking.

7.2.5 Summary of chemical contamination of marine sediments

A large regional database on sediment contamination has been acquired since 1995. This data clearly shows that urban stormwater is contaminating Auckland's urban estuaries with heavy metals and, at lower levels, a range of POPs.

Current information indicates that over 50 per cent of regional monitoring sites are contaminated to the point where aquatic life may be beginning to be adversely affected (ranked as amber or red). Worst affected sites are in muddy estuaries receiving run-off from older, fully urbanised catchments.

Zinc is the contaminant of most concern at present. It is increasing in concentration at most sites, and has the greatest proportion of ERC exceedances.

Persistent organic pollutants, such as PAH, OCPs, and PCBs are widely found, but concentrations are low, below ERC at the majority of sites. Organochlorines are unlikely to increase much, if at all, in future because they are no longer legally used. However, urban development can release these contaminants into estuaries from contaminated catchment soils.

Temporal trend monitoring conducted since 1998 indicates increasing concentrations of Zn at most urban sites, smaller increases in Cu, and uncertain trends for Pb and PAH. Trends will become more definite in future as more data is collected.

While the magnitude of some trends is still somewhat uncertain, it is quite clear that Zn concentrations are increasing in receiving environment sediments. Reducing Zn run-off is required to slow or halt this trend and prevent irreversible contamination of the marine environment.

Based on current USA experience, further investigation of the levels and significance of some POPs (including phthalate esters, DBPEs, and PCBs) and Hg seems warranted to assess whether they are likely to be important environmental stormwater-derived contaminants in Auckland.

7.3 Contamination of shellfish and fish

Aquatic organisms such as shellfish and fish can accumulate substantial levels of chemical and microbial contaminants when exposed to polluted water and sediment. In the case of microbial contamination, this can lead to these organisms being unfit for human consumption. While chemical contamination is not generally high enough around Auckland to cause problems for human consumers of fish or shellfish, chronic health effects on the aquatic organisms themselves, or on other animals that feed on them, are possible ecological consequences.

Concentrations of chemical contaminants in the water column are usually very low (Section 7.1), and are unlikely to directly affect aquatic life. However, some chemicals, such as organochlorine pesticides and PCBs, can bioaccumulate in the tissues of some aquatic organisms and may cause chronic, long-term ecological problems. Species that are likely to accumulate highest levels of contaminants are those that live in contaminated environments, particularly when exposed to polluted sediments – eg shellfish, snails, bottom-feeding fish, and worms.

Some chemicals are transferred through the food chain, so higher trophic level organisms, in particular birds that feed on contaminated worms, fish, and shellfish, can accumulate high concentrations, and this can cause serious ecological problems (eg the infamous egg-shell thinning problems for American birds of prey, caused by exposure to organochlorine pesticides such as DDT). It is worth noting that even modest levels of some contaminants in sediments can lead to biological problems – San Francisco Bay is an example, where a cleanup target of 2 ppb total PCBs in sediments has been put in place to protect bird life from PCBs accumulated through the food chain (Hetzel 2004).

Because shellfish bioaccumulate some contaminants, they offer a sensitive method of detecting the presence of even very low concentrations of these chemicals in the water column. This is useful for measuring spatial patterns and temporal trends in contaminant levels over time. This monitoring approach has been widely employed around Auckland, using both resident biota (oysters, mussels, snails, cockles) and deployed species (mussels) in areas where there are no suitable resident populations (eg harbour channels and open waters).

Like the ARC's regional marine water quality monitoring programme, the ARC shellfish contaminant monitoring programme measures the broad-scale impacts of urban run-off and wastewater contamination. It now comprises two strands (ARC 2004l, m):

- the Manukau Oyster Monitoring Programme, which samples oysters (*Crassostrea gigas*) at four sites; and
- the Mussel Monitoring Programme, which provides wider geographical coverage including seven sites in the Manukau and Waitemata Harbours and the Tamaki Estuary, using three-month deployment of mussels (*Perna canaliculus*).

Sampling is conducted annually, and the shellfish analysed for metals and persistent organic pollutants (PAH, organochlorine pesticides, PCBs).

Issues

- Microbiological contamination of shellfish, and consequent human health impacts.
- Potential adverse effects on aquatic fauna, and on higher trophic level animals (birds) as a result of chemical contamination.
- Monitoring spatial patterns and temporal trends in contamination using tissue analysis of shellfish and fish.

State of knowledge in 1995

In 1995, microbial contamination of shellfish due to stormwater and wastewater discharges from urban areas was well known.

Studies of chemical contaminant levels in a variety of resident species (cockles, mussels, oysters, and mud snails), and in transplanted oysters, had been conducted in the Tamaki Estuary (ARC 1992b, NIWA 1994c). Higher levels of contamination were found in transplanted oysters at estuary sites influenced by older, high intensity urban development (Otara, Panmure), and levels declined towards the mouth of the estuary (Bucklands Beach) and open waters of the Hauraki Gulf (Browns Island). Contamination in resident species reflected both the location and the species – eg amphibola (mud snails) accumulated high concentrations of organic contaminants, while Zn levels were highest in oysters.

Long-term monitoring of resident oysters in the Manukau Harbour had been in place since 1987. Initially 11 sites around the harbour were monitored – these were reduced in 1992 to four sites representing the major land uses in the catchment. As found for the Tamaki, highest concentrations of contaminants were found in the heavily urbanised Mangere Inlet, and lowest near the harbour mouth at Cornwallis.

In 1993, the oyster transplantation approach was modified, using commercially grown mussels deployed for three-month periods at several sites in the Tamaki Estuary, Waitemata Harbour, and East Coast Bays (NIWA 1995c,e,f,g, 1996b). This monitoring continued until 1996. The mussel data were consistent with effects of urbanisation – lowest contaminant concentrations were present at most East Coast sites, slightly higher levels at Browns Bay, and highest at urban inner harbour of estuary sites (eg Panmure site in the Tamaki, and Chelsea site in the Waitemata Harbour).

There was clear evidence for chemical contamination of the water column in harbour and estuary waters in urban areas, and contaminants were bioavailable to common shellfish species.

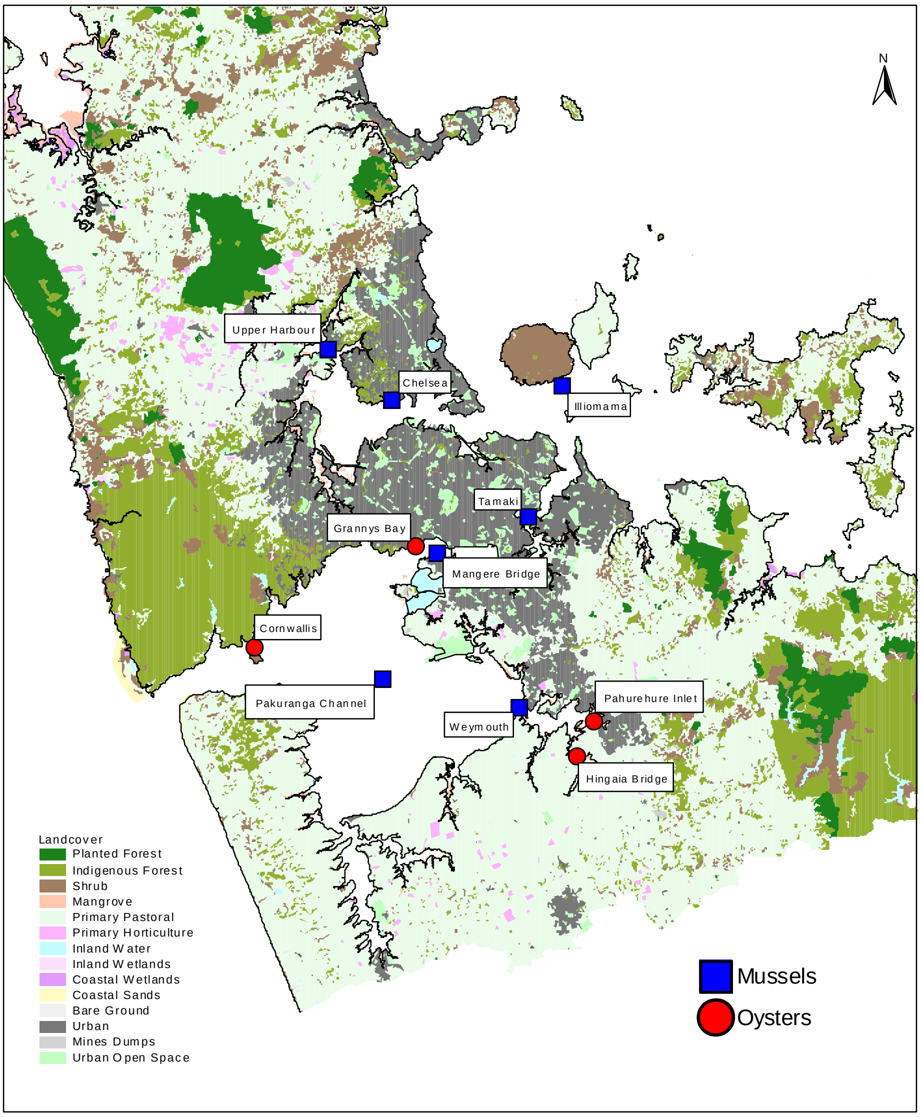
Advances since 1995

There has been an on-going refinement of the oyster and mussel monitoring programmes. The Manukau oyster programme was reviewed in 1998 (NIWA 1998b) to assess temporal trends in contaminant levels and to see whether any changes were required to improve the programme. In 1999, a revised mussel programme was begun, covering seven sites in the Manukau, Tamaki, and Waitemata Harbours (Figure 67). The first data were reported by NIWA (2000d).

Organic contaminants in Tamaki Estuary shellfish were reassessed in 1999 (NIWA 2000c), and in Waitemata Harbour resident oysters in 1998 (NIWA 1998c).

Organic contaminants were measured in flounder from four sites in the Manukau Harbour in 1998, to assess whether there was any relationship between contaminant levels and biochemical response markers (early warning signals of sub-organism biological effects). An extensive project on parasites and pathological lesions was also undertaken at the same sites to assess relationships between contamination and fish health (ARC 2000d).

Figure 67
ARC shellfish contaminant monitoring programme sites (ARC 2004i).

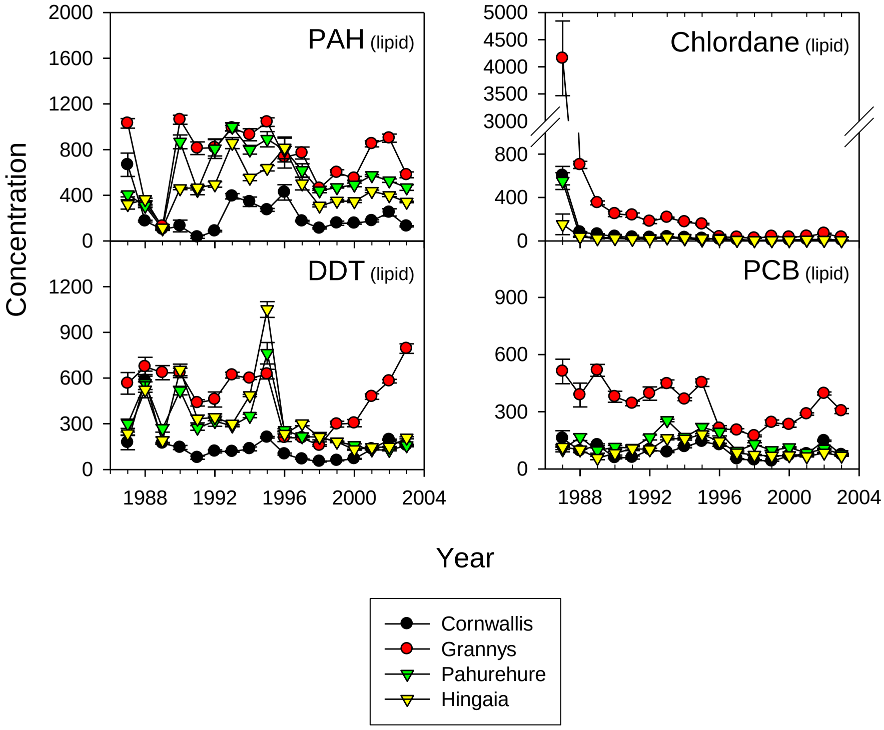


7.3.1 Trends from the ARC Manukau Harbour oyster programme

Changes in contaminant levels over time (temporal trends) in oysters at four sites in the Manukau Harbour have been assessed by NIWA (1998b) and ARC (2004l&m). The strongest trend was observed for chlordane, which reduced markedly following removal of a point source of contamination from a timber treatment plant. Dieldrin has also dropped over time, although less dramatically than chlordane. No other consistent long-term trends over the entire duration of the monitoring have been observed, although some medium-term (eg over a few years) trends are apparent – some examples are shown for selected organic contaminants in Figure 68 (ARC 2004m).

Figure 68

Long-term trends in selected organic contaminants (ng/g lipid) in Manukau Harbour oysters (ARC 2004m). Note the large decrease in chlordane (following removal of a point-source discharge), but smaller and more variable changes in other contaminants (which are more diffuse in origin).



These trends suggest that:

- source control (eg elimination of point-sources of pollution) can have major beneficial impacts on water quality (and hence contaminant levels in shellfish);
- changes related to larger (catchment) scale, diffuse-sourced, contamination are likely to be slow; and
- trends vary over different time-scales – decreasing trends may be observed over a period of a few years, and then increase for the next few years. This is probably due to the combination of natural factors (biological, hydrodynamic, climatic factors) and changes in catchment contaminant loads.

The programme's long-term record enables the short- to medium-term fluctuations to be put into context with the longer term overall trends. Better information on catchment contaminant loads may help interpret some of the changes that have occurred over time.

One potentially major influence on contaminant levels in oysters in the Manukau Harbour has been the decommissioning of the Mangere sewage treatment ponds and treatment plant upgrading. It is possible that the increasing levels of DDT observed in oysters from Granny's Bay may reflect the release of DDT from contaminated sludge associated with the treatment plant works. Similar changes have been measured in the Watercare Manukau Harbour Environment Monitoring Programme, but 2004 data suggest concentrations may be dropping again (Watercare 2005).

7.3.2 Trends from the ARC mussel monitoring programme

Data collected since 1999 in the mussel-monitoring programme show distinct differences between sites for organic contaminants (Figure 70). Differences between sites for metals were smaller.

Trends in organic contaminant levels have begun to emerge, even after only a few years. Increases in DDT, and to a lesser degree PCBs, between 1999 and 2003 at Mangere Inlet are consistent with those observed in the oyster programme. Small, slowly decreasing trends appear to be occurring at most of the east coast sites, with the possible exception of DDT, which has remained fairly constant over time.

Figure 69

Summary of contamination levels at different sites, as determined from the ARC oyster and mussel contamination monitoring programmes (ARC 2004m).

Worst quality → Best quality

<u>Oysters:</u>

Granny's Bay	Pahurehure	Cornwallis
	Hingaia	

<u>Mussels:</u>

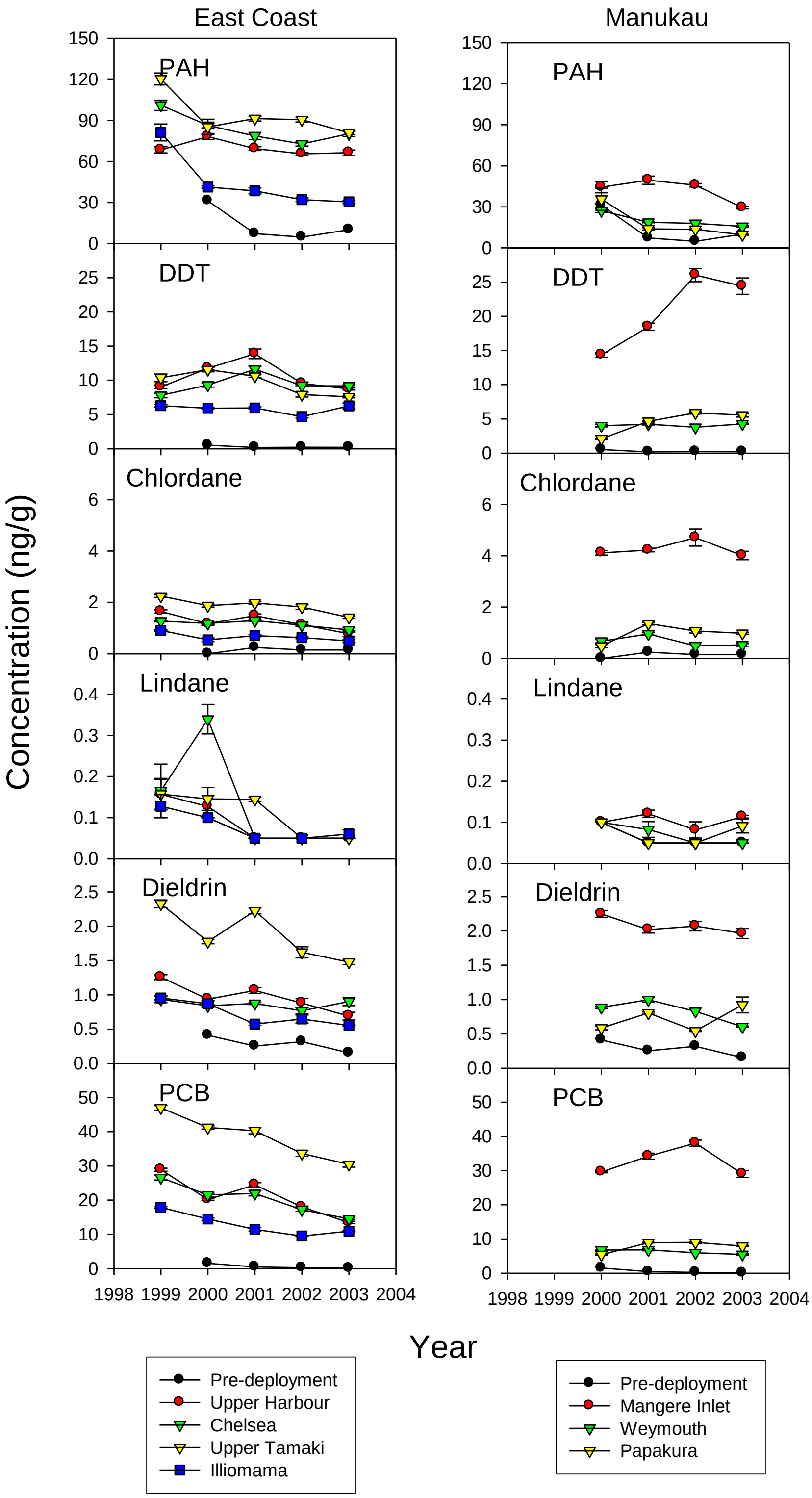
Mangere Inlet	Tamaki Estuary	Upper Waitemata	Weymouth	Pre-deployment
		Chelsea	Papakura Channel	
			Illiomama	

Overall, the shellfish monitoring has provided data that shows that the water quality at urban sites is impacted by low levels of organic contaminants, including historically used organochlorine pesticides that are generally considered to be primarily rural, rather than urban, contaminants. The levels of contamination are summarised in Figure 70.

Trends are complex, with marked short- to medium-term changes altering the form of the longer-term trends every few years. Apart from the large decrease in chlordane in the Manukau Harbour resulting from the removal of the point source in Mangere Inlet, changes are generally slow reflecting the overall effect of diffuse sourced catchment inputs to marine receiving waters.

Figure 70

Trends in selected organic contaminants (ng/g lipid) in mussels deployed in Tamaki Estuary, and the Manukau and Waitemata Harbours (ARC 2004m).



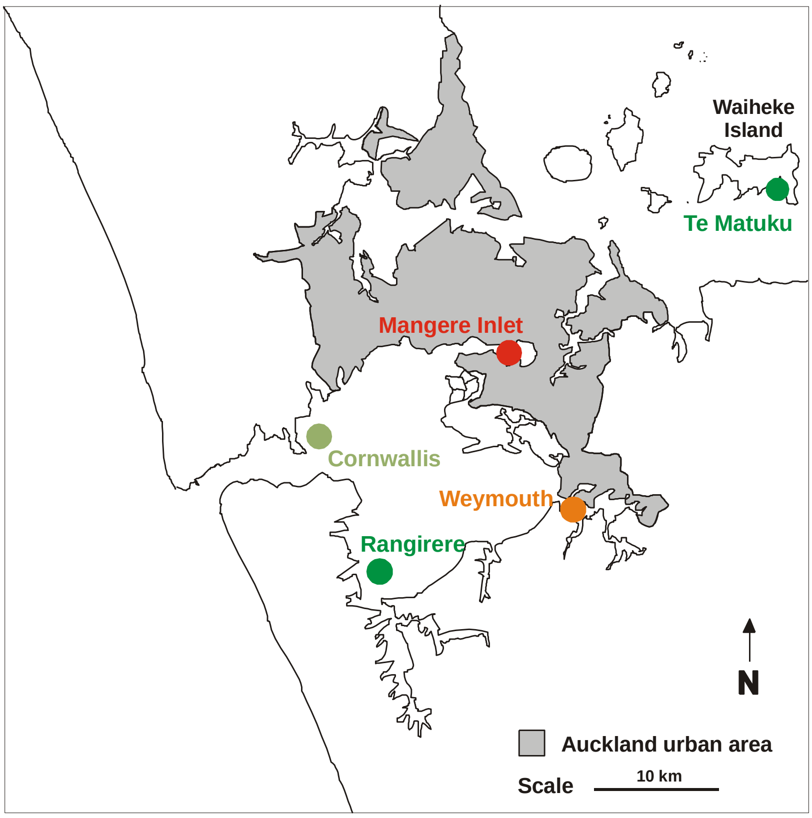
7.3.3 Organic contaminants in Manukau Harbour flounder

Studies carried out in polluted harbours overseas have found links between concentrations of some contaminants (including PAH and PCB) in sediments and a variety of diseases in fish (eg liver lesions). Bottom-dwelling fish, which are directly exposed to sediment-associated pollutants by contact with the sediment, as well as by ingestion of benthic prey and sediment, are particularly susceptible to contaminated sediments.

In 1997/98, PAH, organochlorine pesticides (OCP), and PCB were measured in yellow-belly flounder (*Rhombosolea leporina*) from a range of contaminated and clean locations in Auckland (Figure 71) to determine whether these contaminants were bioavailable, and whether their health was related to contamination. A preliminary study also included a site in Wellington Harbour, where DDT concentrations are known to be high. This work is unpublished, but has been summarised in ARC (2000d), and was presented at the 1998 Marine Sciences Society Conference (Mills et al. 1998).

Figure 71

Flounder sampling sites in the 1997/98 NIWA contamination studies.



The potential biological effects of these contaminants were assessed by determining contaminant burdens (PAH metabolites in the bile, organochlorines in liver) and the response of a biochemical biomarker called EROD[31], which provides a sub-organism, biochemical marker of exposure to some PAH, PCBs, pesticides and dioxins. Exposure to these contaminants results in an increased level of liver EROD. This may serve as an "early warning" signal of more serious adverse effects at higher biological levels (eg organism level).

Measures of fish health, including liver pathology, blood cell abnormalities, and parasites were also examined (ARC 2000d).

Key findings (summarised for the Manukau Harbour study in Figure 72) were:

- Contaminant levels in fish from Te Matuku Bay (Waiheke Island) were far lower than those in Wellington Harbour, which in turn were lower than those found in Mangere Inlet flounder. The exception was DDT, which was higher in Wellington Harbour fish – this was consistent with the high concentrations of DDT present in sediments at some Wellington Harbour sites.
- Contaminants were detectable in fish caught from all Manukau Harbour sites. The spatial trends in contaminant levels generally followed the anticipated environmental contaminant gradient, increasing from lowest at Rangirere to highest at Mangere Inlet. PCBs and chlordane were exceptions to the "smooth" trend in contaminant levels, being either unexpectedly high at Cornwallis, or relatively low at Weymouth. Relatively high levels at Cornwallis may be due to transfer of contaminants and/or fish movement between Mangere Inlet and Cornwallis along the Wairopa and Purakau channels.
- EROD induction followed a similar pattern of increasing response from clean to contaminated environments, indicating that contaminants were eliciting a biological response.
- Comparison with data from the US National Oceanographic and Atmospheric Administration (NOAA) Benthic Surveillance Programme, in which contaminants in a range of bottom-dwelling fish have been measured at approximately 200 sites since 1984, revealed that the Manukau Harbour flounder have moderate contaminant levels. Levels at Mangere Inlet are in the 40–75 percentile range of the US data. These levels are approximately 10–1000 times lower (depending on the contaminant) than the maximum levels recorded in the USA. Examples are shown in Figure 73.

Compared with US levels, contaminants in Manukau Harbour flounder range from very low (at the clean site) to moderately high at Mangere Inlet. The fish health surveys showed that fish from Mangere had higher prevalence of pre-neoplastic liver lesions, consistent with the higher contaminant levels at this site.

[31] EROD is the cytochrome P450 mixed function oxidase isozyme "ethoxyresorufin O-deethylase", which gives a measure of the induction of cytochrome P4501A, a family of proteins that catalyse the oxidation of a range of aromatic and chlorinated hydrocarbons, including some PAH, PCBs, pesticides and dioxins.

The flounder data therefore provided evidence that organic contaminants are entering the aquatic food chain, and suggested that biological effects may result. However, no cause–effect relationship between contaminants and health effects was established.

Figure 72

Concentrations of organochlorine contaminants (ng/g lipid in liver), PAH-metabolites (g/mL of bile), and liver EROD activity (pmol/min/mg protein) in Manukau Harbour flounder. Sites are Rangirere (R), Cornwallis (C), Weymouth (W), and Mangere Inlet (M). Boxes are 25–75 percentiles of the data, with the median shown as a horizontal line within the box. The *'s are outliers.

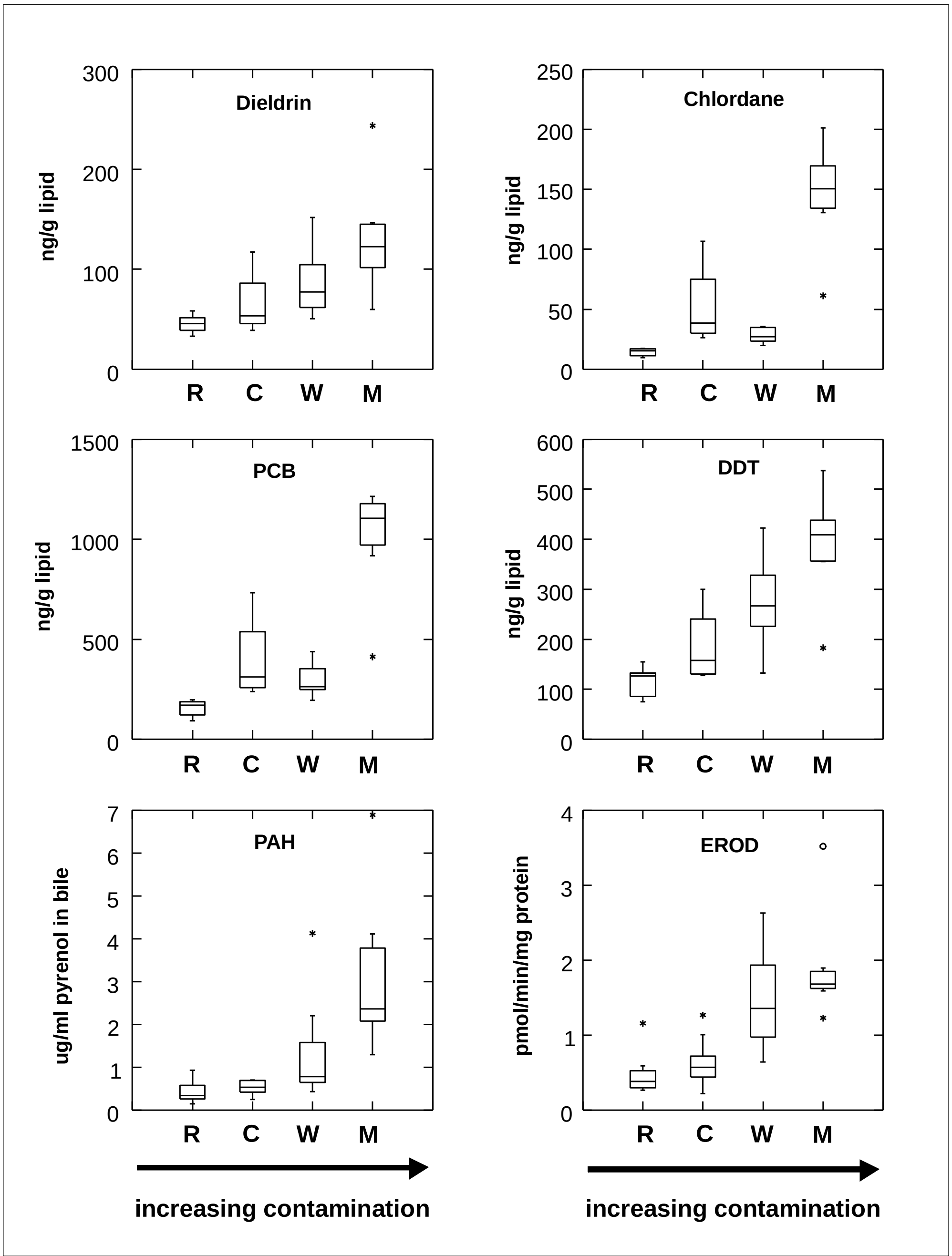
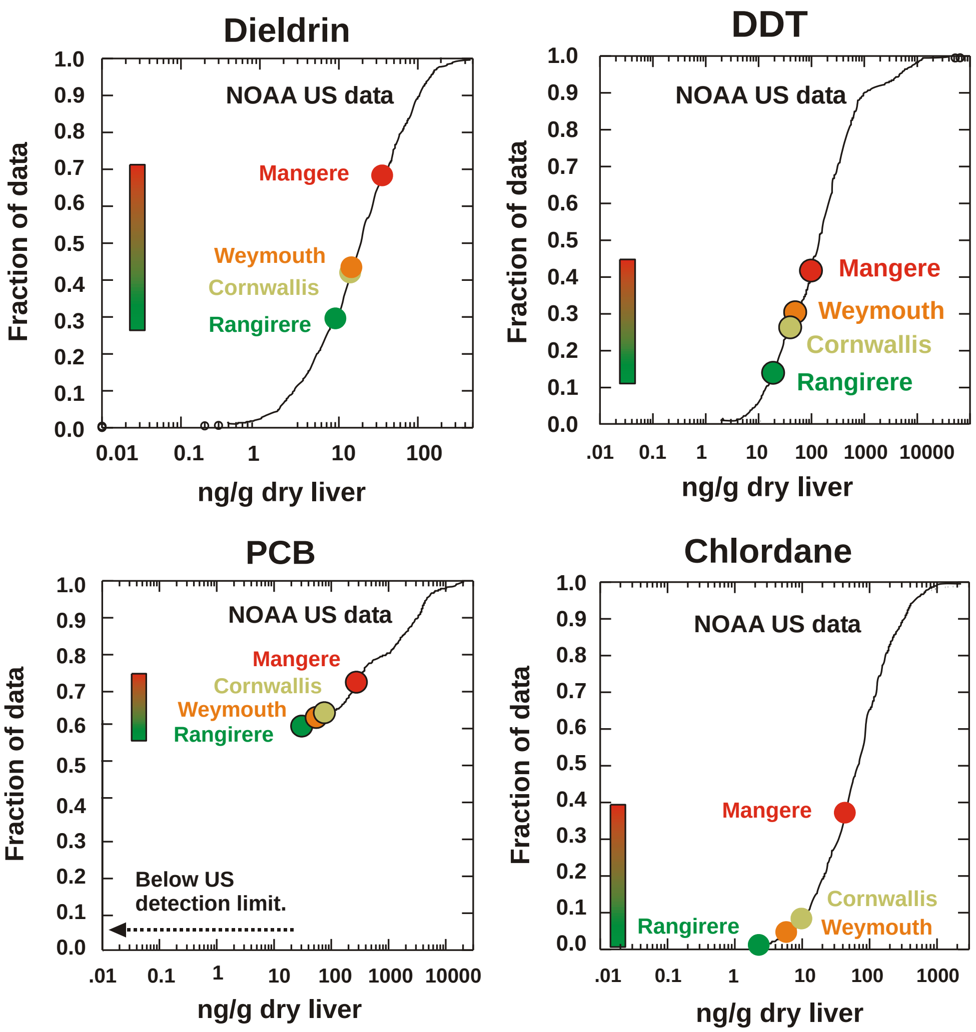


Figure 73

Comparison of concentrations of organochlorines in Manukau Harbour flounder (ng/g lipid in liver) with those from the NOAA database of US estuaries and harbours (Mills et al. 1998).



7.3.4 Summary of chemical contamination of shellfish and fish

ARC monitoring of resident and deployed shellfish show that chemical contaminants, in particular organic compounds including PAH, OCPs, and PCBs, are accumulated by these biota from the water column, enabling spatial patterns and temporal trends in contamination to be measured.

Contaminant levels are highest at sites receiving run-off from the older intensively urbanised areas, in particular Mangere Inlet and the inner Tamaki Estuary. Waitemata Harbour sites have intermediate levels of contamination, and the open East Coast waters are least contaminated.

Trends over time are dependent on the period of interest – the long-term record (from 1987 for the Manukau programme) shows little change apart from large decreases in chlordane associated with the removal of a point source of contamination in Mangere Inlet. Quite large increases and decreases occur over shorter time periods, reflecting the combined effects of biological variability, changes in catchment run-off and, in the case of the Manukau Harbour, the decommissioning of the old sewage treatment ponds.

Limited studies of organic contaminants in flounder in the Manukau Harbour show spatial patterns of contamination consistent with the degree of nearby urban development. Contaminant concentrations are moderate to low compared with USA data. There is some evidence to suggest that fish health is being affected at the most contaminated site, although more study is required to be definitive about what the ecological implications of contamination really are.

7.4 Deciphering the history of urban stormwater impacts

The history of contaminant inputs to an estuary is often captured in the sediments that have accumulated over time. By taking samples of sediment profiles – the sediments that have built up over time – and by deciphering these profiles we are able to shed light on the impact of urban stormwater. They also give indications of future trends.

State of knowledge in 1995

Very little was known in 1995 on the history or sequence of events that occurred in estuaries after catchment urbanisation. Sediment profiles had been studied in several estuaries around Auckland to assess long-term sedimentation patterns and estuary processes. Coring locations included Lucas, Hellyers, Brighams Creeks – all in the Upper Waitemata Harbour (Hume & McGlone 1986, Vant et al. 1993) – Drury Creek in the Pahurehure Inlet in the Manukau Harbour (Hume et al. 1989), and the Manukau Harbour (Murray North 1988).

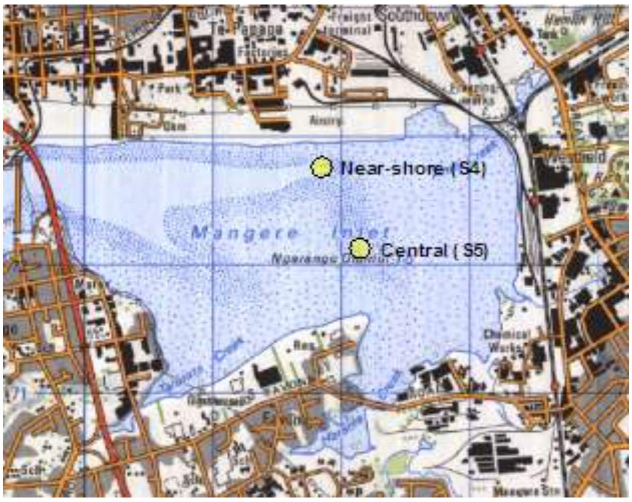
None of these cores showed effects of urban stormwater, but they were not taken for that purpose. The stratigraphy of the cores did depict sediment processes occurring in estuaries, high sedimentation rates in the sheltered tidal creeks, low sedimentation rates on sandy flats in Manukau Harbour, and the long-term changes brought about by catchment land uses since pre-Polynesian times.

Two deep cores were taken in Mangere Inlet in 1991 specifically to reveal the impacts of urbanisation, including the effects of industrial and waste discharges prior to the commissioning of the Manukau Wastewater Treatment Plant in 1962. Historical discharges from the heavy industrial areas of Onehunga, Otahuhu, and Penrose contaminated the sediments in Mangere Inlet with heavy metals (in particular Zn, Cu, Pb, and Cr) and organic contaminants (pesticides, PCBs, and PAH).

As the estuary has filled in with sediment, these high concentrations have been buried and "preserved" in the sediment profile (shown in Figure 75)

Figure 74

Mangere Inlet showing the sites cored in 1991.

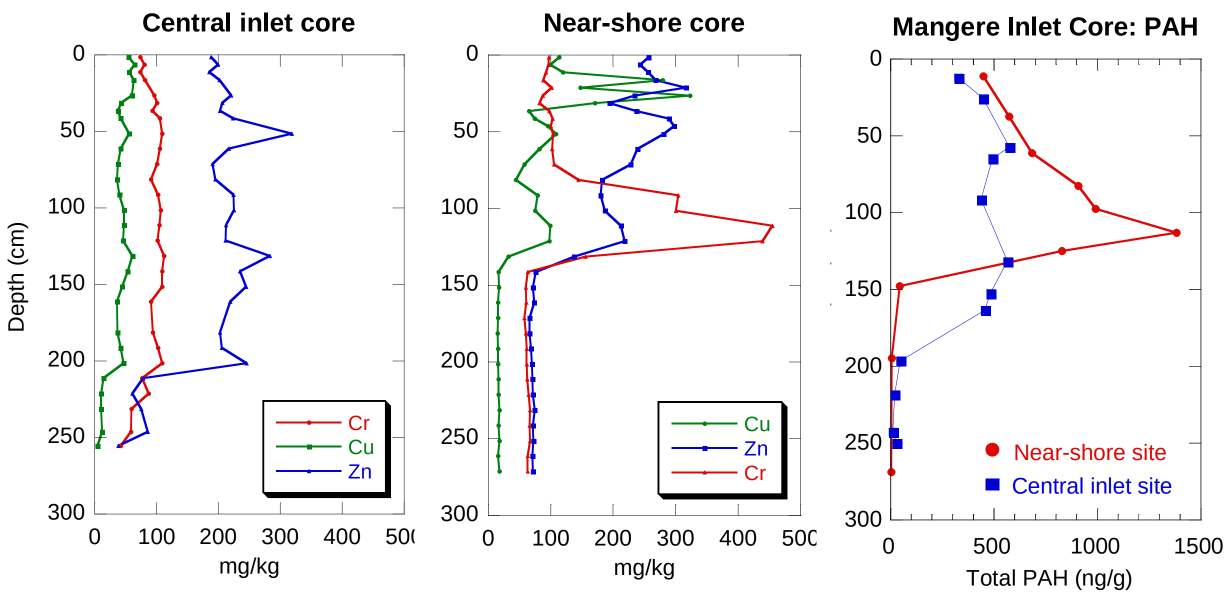


Concentrations of Cu, Cr, and Zn in the sediments were all very high, exceeding the ERC "red" levels at times in the past, as seen in the depth profile of these metals (Williamson et al. 1991; Figure 75). PAH also increased in concentration at around the 1950s, more markedly in the near-shore sediments close to a stormwater outfall than in the central area of the inlet (Wilcock & Northcott 1995; Figure 75).

Concentrations of heavy metals were still very high in parts of the inlet in the 1980s. After the industrial discharges stopped, these contaminated sediments were gradually buried by less contaminated sediments that had been flushed into the inlet from the main body of the harbour or from reclamation activities. However, the concentrations of Zn and Cu in the surface sediments appear to have levelled out or to be slowly increasing again due to on-going urban stormwater inputs.

Figure 75

Concentrations of Cu, Zn, Cr, and total PAH in cores taken from two locations in Mangere Inlet in 1991 (replotted from Williamson 1991 (metals) and Wilcock & Northcott 1995 (PAH).



Advances since 1995

Since 1995, a number of detailed sediment coring studies have been undertaken to decipher the history of urban stormwater impacts on estuaries. These sediment profiles have been used to:

- Reconstruct the history of contamination by urban stormwater-derived contaminants.
- Measure sedimentation rates in different estuarine environments.
- Measure the mixing of sediment by the burrowing activities of benthic animals (bioturbation).

Studies from Pakuranga, Lucas, Henderson, and the Tamaki estuaries are summarised below to illustrate how urbanisation has changed the concentrations of contaminants in estuarine sediments.

7.4.1 Pakuranga Creek

Five cores from the length of Pakuranga Creek estuary (Figure 76) were collected in 1996 to determine sedimentation and contaminant history, and in particular the impact of catchment urbanisation (Swales et al. 2002). Heavy metal concentrations, particle size, ^{137}Cs, pollen, and catchment sediment loads were all needed to disentangle the complex estuarine response to urbanisation and interpret the contaminant profiles.

Dating of the sediment profiles showed that during urbanisation, sedimentation rates in the tidal creek and estuary were higher than sedimentation rates associated with past agricultural land use and the original forest land cover. In pre-Polynesian and pre-European times, estuary infilling was very slow (0.2–0.6 mm/yr). Catchment deforestation and subsequent agricultural land uses from the mid-1800s increased rates about 3-fold (to 0.8–1.6 mm/yr). Urbanisation increased these dramatically – in the mid-lower estuary rates were about doubled (to 1.7–3.8 mm/yr), and at the head of the estuary rates averaged nearly 33 mm/yr (approximately 20 times the pre-urban rate).

Urbanisation brought about substantial environmental changes, especially in the upper estuary, through continued infilling of shallow, intertidal areas, contamination by heavy metals to levels of ecological concern, sediment textural changes, and rapid mangrove colonisation of formerly bare intertidal sediments.

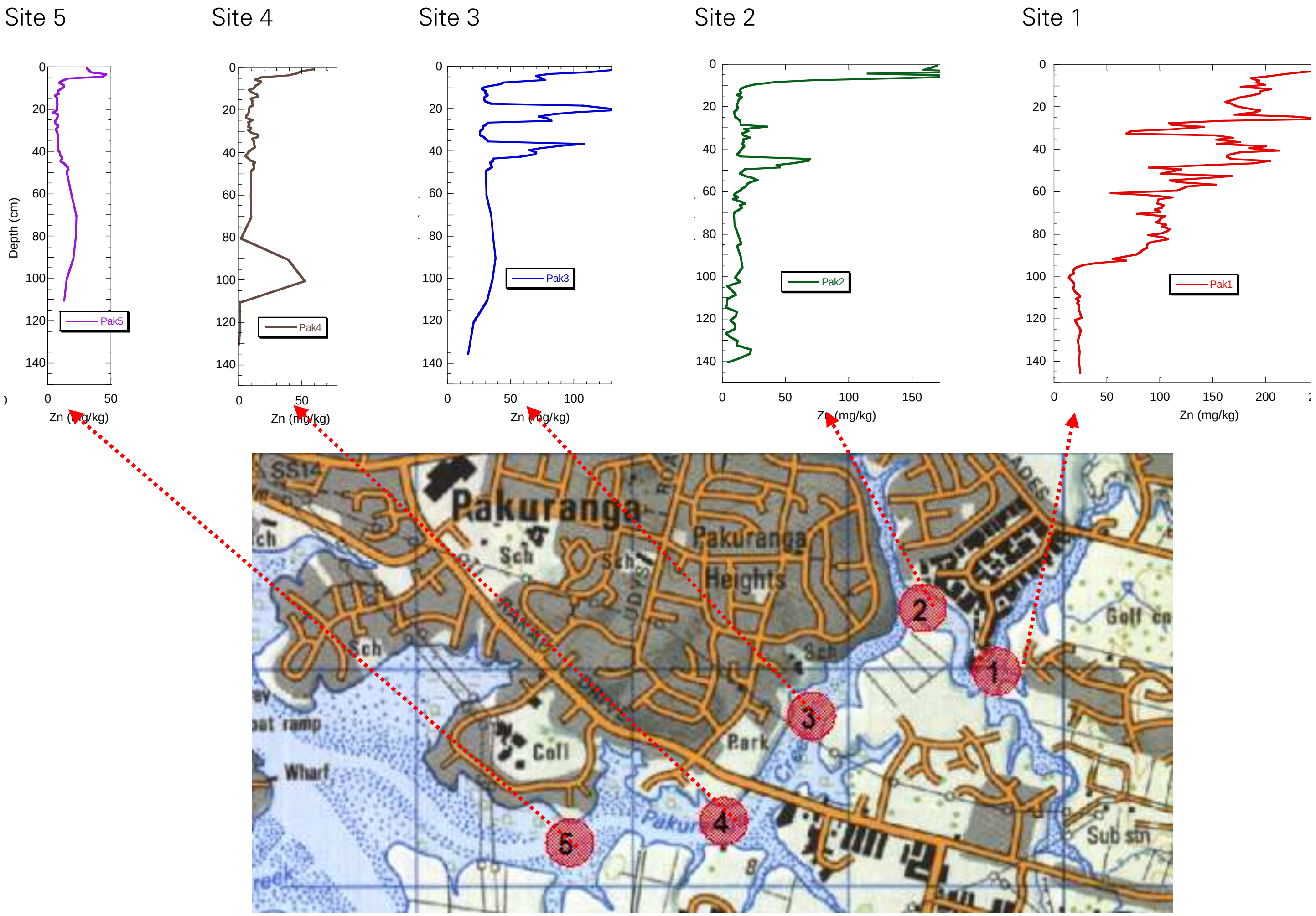
The complex heavy metals concentration profiles obtained from the cores are shown in Figure 76. Key features are:

- Increases in Zn concentrations in near-surface (more recently deposited) sediments, indicating increasing inputs from the catchment over time. Concentrations of Pb and Cu also increased over the same depths. Contamination began to occur in about 1950. Concentrations at the head of the estuary increased approximately 200 mg/kg over ca. 40 years.
- Considerable variability in Zn concentrations, which is linked to the complex combination of time-varying inputs (low metal sub-soil erosion, high metal street run-off, top-soil erosion, periods of earth working, construction activity etc) that contribute to metal loads, as well as within-estuary processing (eg bioturbation). The variability seen in the core profiles reflects the detailed sampling used (narrow time horizons) and the high deposition rate in the head of the estuary, which reduces the smoothing effect of in-situ mixing processes (in particular bioturbation by benthic fauna).
- Higher levels of Zn contamination (extending to greater depths) in the headwater sediments than further down the estuary, reflecting greater accumulation of contaminated sediments in settling zones close to the contaminant source.
- Very high deposition rates in the uppermost estuary (about 1 m since 1960), decreasing with distance down the estuary, until site 5 where contamination barely reaches the lower estuary.

This study showed that the general effect of catchment urbanisation on the build up of metals in estuarine sediments could be readily determined from coring, but interpreting the core profiles required considerable additional information including catchment land use history, sediment load data (modelled in this case), and dating (eg ^{14}C and pollen dating to determine long-term sedimentation rates, and ^{137}Cs for sediments deposited post-1953).

Figure 76

Zn profiles for sediment cores from Pakuranga Creek estuary.

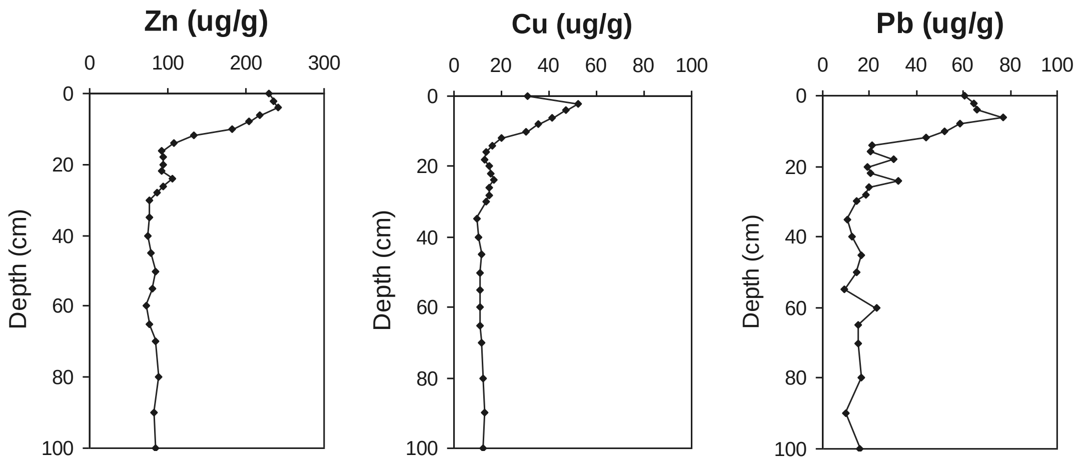


7.4.2 Tamaki Estuary

Eight sediment cores (50–100 cm) recovered from the main body of the Tamaki Estuary in Auckland showed significant heavy metal enrichment towards the surface (Figure 77; Abrahim & Parker 2002, Parker et al. 2002). The trends in Figure 77 all indicate that significant sediment enrichment starts above a depth of 20 cm and attains maximum values above 10 cm. Cs137 data indicate that in this core, depths shallower than 10 cm represent sediments deposited after 1965. The high levels of heavy metal contaminants found in the upper sediment layers of the core are consistent with periods of considerably increased development and disturbance of the Tamaki catchment in the 1960s, 1970s and 1980s.

The data shows that the depth of contamination is relatively shallow in the main body of the estuary, similar to that found in the lower arm of Pakuranga estuary, in marked contrast to the large depth of contamination found in the head of the Pakuranga estuary (Figure 76 previously).

Figure 77
Sediment metals' profiles taken from deep cores in the Tamaki Estuary (Abrahim & Parker 2002, Parker et al. 2002).



7.4.3 Lucas Creek

Sediment contaminant profiles were measured in cores taken from two locations in Lucas Creek estuary in April 2003, primarily to study how bioturbation[32] mixes contamination in the surface layers of estuarine sediments. This information was

[32] Bioturbation is the mixing of sediments by biological activity, principally the burrowing activities of sediment-dwelling animals such as crabs, worms, shrimps etc. This mixing homogenizes the surface sediments to a depth of at least 10–15 cm in Auckland mudflats. It is less effective in areas where sediment deposition is rapid, such as in estuary headwaters, as the bioturbation rate can't always match or exceed the sedimentation rate.

needed for development of models to predict contaminant accumulation in sediments as a result of urban development (ARC 2002a).

Lucas Creek has a similar size estuary and watershed areas as Pakuranga, but urbanisation began about 30 years later than Pakuranga, in the 1990s rather than the 1960s. In this study, the overall average sediment profile was wanted (rather than the detail obtained in the Pakuranga study), so nine cores were taken and combined together across the same depth intervals. This procedure smoothed out any localised effects, such as crab burrows or the presence of debris in the profile.

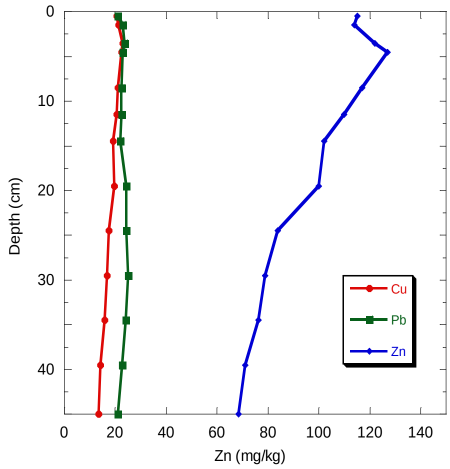
The profile obtained from the lower site (Figure 78) showed that background concentrations were reached at about 30 cm depth. Above this, concentrations of Cu, Pb and Zn increased to a depth of about 13 cm, after which concentrations stabilised. This profile was consistent with intense mixing in the top 13 cm, presumably largely due to bioturbation. Without this mixing, the profile would show a far more rapid increase in metals' concentration, increasing all the way to the sediment surface, and far higher surface concentrations (reflecting the concentration of metals in stormwater suspended sediments, which for Zn is approximately 2000 mg/kg in a mature, fully developed, urban catchment).

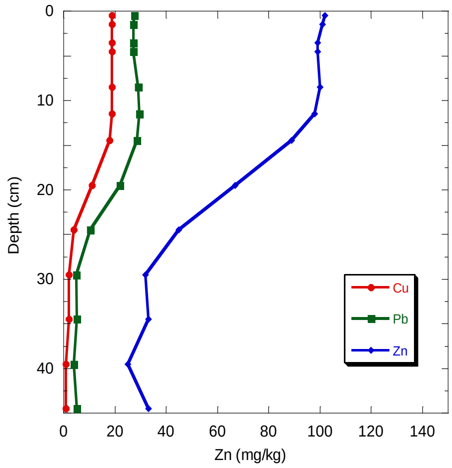
Another profile was collected further up the estuary. Here contamination is deeper, because sedimentation was more rapid. A greater mass of contaminants and sediments have settled here, but the level of contamination is about the same on the surface, however, in line with the estuary acting as a Settling Zone (see Chapter 3).

Figure 78

Cu, Pb and Zn profiles for composite cores from Lucas Creek estuary. The upper profile is taken from site 1, further up the estuary than the lower profile (site 2).

Lucas Creek estuary, showing sediment core sites.



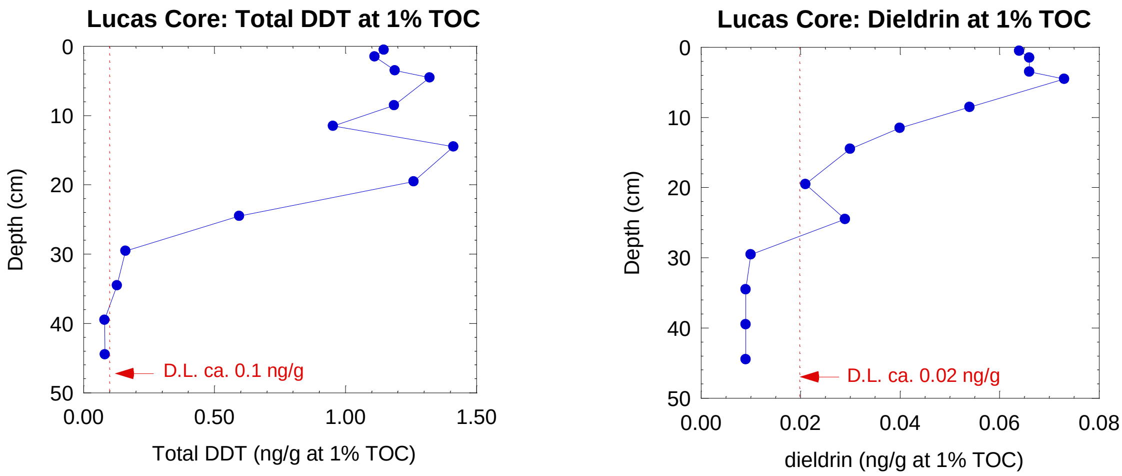


Organochlorine pesticide profiles were also measured in the lower Lucas Creek core (T&T 2005a). DDTs were the major OCPs present in the core sediments. Concentrations were low, below the ARC ERC – the maximum Total DDT concentration in the core was approximately 1.5 ng/g (at 1 per cent TOC), compared with the ARC ERC of 3.9 ng/g. Despite the low concentrations, DDT and dieldrin profiles were quite clear, as shown in Figure 79. The dieldrin profile is different from that for DDT, probably because dieldrin is more degradable in the sediment environment.

The sediment core profile showed that the input of DDT coincided with the onset of urbanisation, as reflected in the increase in Zn, which also started at 30 cm depth. Inputs before urbanisation from horticultural land use were too small to be detected. The results support the hypothesis that a major route for input of legacy pesticides to aquatic receiving waters is the mobilisation of catchment soils during major disturbance such as earth-working operations during urban development. Sediment profiles from Lucas Creek show a reasonably strong DDT signal despite the catchment only having a relatively small proportion of horticultural land.

Figure 79

Depth profiles of dieldrin and Total DDT (in the <0.5 mm sediment fraction) in the composite cores from site 2 (Fig. 6.31), lower Lucas Creek estuary (T&T 2005a).



In summary, the Lucas Creek cores showed that sediment Zn concentrations had increased by about 70 mg/kg since urbanisation began (approximately 15 years) – this is comparable to the increase observed at the head of Pakuranga estuary (200 mg/kg over 40 years, or 5 mg/kg/yr). Bioturbation produced fairly constant metal concentrations to around 13 cm depth. DDT concentrations increased by 1–1.5 ng/g (at 1 per cent TOC), coinciding with the increasing metals' levels, presumably as a result of inputs from catchment soils mobilised during urban development.

7.4.4 Henderson Creek

Henderson Creek catchment has a long history of horticulture and viticulture. Persistent pesticides, including organochlorine pesticides (principally DDT), arsenic, and Cu were widely used in horticulture. Urban expansion in the lower Henderson catchment has included development of significant areas of former horticultural land, and consequently loss of pesticide-contaminated soil to Henderson Creek estuary represents a potentially significant risk to the receiving environment.

Sediment profiles from a composite core taken from the mid to upper reaches of the Henderson Creek estuary (location shown overleaf) were studied to determine the history and long-term trends in OCP exports from the catchment (DSL 2004, T&T 2005b). OCP concentration profiles were compared with Cu, Pb, and Zn profiles to assess whether OCP export from the catchment was related to urbanisation or to pre-urban, horticultural soil losses.

Figure 80
Coring site in Henderson Creek estuary.

The sediment accumulation rate in the estuary was unexpectedly high, so that pre-development sediments were not reached in the 1.5 m deep core, and an incomplete historical contaminant profile was obtained. Figure 81 shows depth profiles of Cu, Pb, Zn, total DDT, and dieldrin obtained from the core.

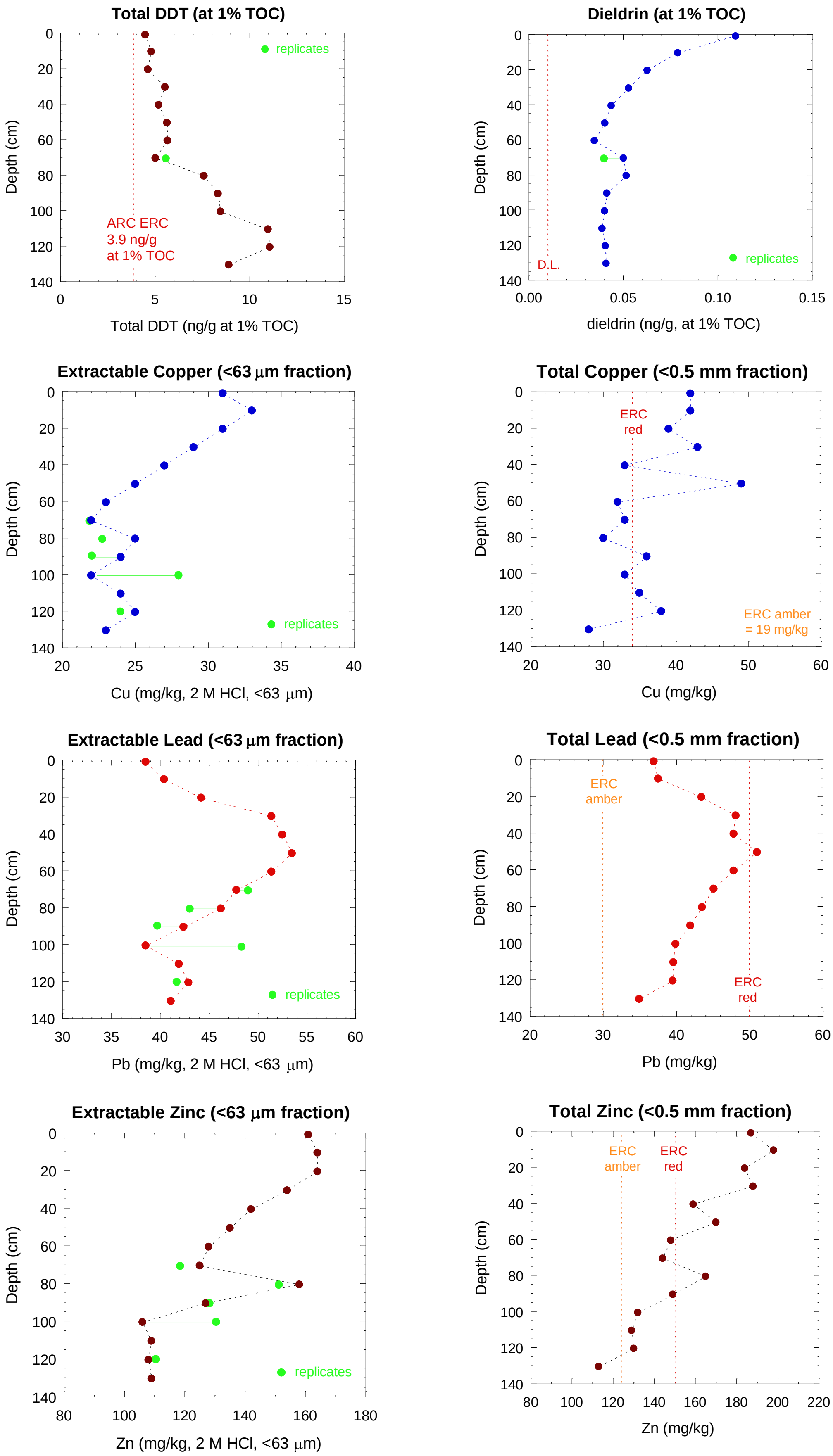
At the bottom of the core, high-level total DDT contamination (mainly present as DDD), of unknown origin, was present at depths possibly corresponding to around 1960–1981. This may reflect urban development of horticultural land, or possibly contamination from an old landfill that was apparently located adjacent to the coring site. DDT concentrations slowly decreased towards the surface, indicating decreasing inputs of DDT over time. If this rate of decrease continues into the future, DDT concentrations will fall below ERC in about nine years.

Despite composite sampling from five cores, metals' concentration profiles were "noisy", mostly attributable to analytical variability, which was unusually high for samples from these cores. They did, however, show increasing Zn and Cu, and decreasing Pb, concentrations in more recently deposited sediments – this is consistent with changes associated with urbanisation.

Henderson Creek estuary is clearly a complex environment, being impacted by large amounts of sediment, historical inputs of pesticides from horticulture, urbanisation (including development of contaminated horticultural land), landfilling alongside the estuary, and historical chemical spills. The profiles obtained from coring reflect this long history of varied catchment land use with multiple potential contamination sources.

Figure 81

Metals and OCP profiles in cores from Henderson Creek estuary (T&T 2005b).



7.4.5 Auckland region-wide studies

A comprehensive regional study of sedimentation processes in estuaries has been carried out by NIWA (ARC 2002c). Key findings are summarised in Chapter 3.

Auckland estuaries have infilled significantly in recent times (~0.5 m in 50 years). Infilling is proceeding faster than sea level rise. Because many estuaries are already at an advance state of infilling (the average high tide water depth in many Auckland estuaries is <1m), there will be substantial changes in morphology and ecology in the foreseeable future.

Analysis of Zn in shallow cores from inter-tidal and sub-tidal locations in seven estuaries or embayments across the Auckland region showed increasing concentrations consistent with increasing urbanisation in the Waitemata Harbour and Waikopua estuary, and possibly in Whitford and Wairoa estuaries. In contrast, Mahurangi, Puhoi, Okura, and Te Matuku estuaries showed no significant increases in Zn levels.

7.4.6 Summary of deciphering urban impacts from sediment cores

Sediment coring clearly shows the accumulation of urban-derived contaminants and sediment within urbanised estuaries. Sedimentation effects are largest in the upper reaches of muddy urban estuaries where sedimentation rates are very high.

Effects reduce below the estuary headwaters; eg Pakuranga Creek shows very high deposition rates in the uppermost estuary (about 1 m since 1960), then a decrease in deposition rates down the estuary, so that contamination barely reaches the mouth. The limited data for Lucas Creek estuary show the same phenomena, with more than 45 cm of contaminated sediment at the uppermost site deposited since about 1990, and 12–15 cm of contaminated sediment at the lower site (after allowing for bioturbation). While effects are less severe beyond the confined estuarine arms, they are still measurable in the sub-tidal areas of the Waitemata Harbour and Tamaki Estuary.

Conversely, and probably less commonly, coring can also show decreasing contamination over time as a result of reducing contaminant loads, burial of historical contamination, and/or increasing loads of less contaminated sediment (eg rural soils or sub-soils). Mangere Inlet (metals, PAH), and possibly Henderson Creek (DDT), are examples.

Detailed cores, such as those taken in Pakuranga Creek estuary, have sufficiently high resolution to show variations in contaminant concentrations due to periods of intense construction (with little erosion controls on earthworking operations), interspersed with periods of lower constructional activity where run-off from mature urban areas predominates.

Lower resolution cores, such as those from Lucas and Henderson Creeks, show increasing levels of Zn and Cu (and sometimes OCPs), and decreasing Pb in more recently deposited sediments, but without the short-term variability or detail shown in cores sampled at finer depth intervals. They also show sediment mixing by animals living within the sediment (bioturbation) is represented by a homogeneous mixing depth of approximately 13 cm, at least for intertidal mudflats in tidal creeks. This is in good agreement with earlier estimates based on ecological studies and modelling.

There is limited evidence, from Lucas Creek, that OCP levels in estuarine sediments may increase as a consequence of urban development. Increases may be relatively small (as in Lucas Creek, where the proportion of horticultural land was small), or potentially large (eg Henderson Creek, where a relatively large amount of horticultural land has been urbanised). Studies are continuing to better understand the risks to aquatic receiving environments from OCP mobilisation during urbanisation.

Region-wide coring has shown that Auckland's estuaries are infilling more rapidly than in the past, and this will bring inevitable changes to ecology, hydrodynamics, sediment transport (including to the wider coastal receiving environment), and contaminant accumulation and dispersion. This infilling is due to sediments from all development and is not only an urban-related phenomenon. However, urban development has greatly accelerated sedimentation, especially in the headwaters of tidal creeks.

7.5 Stormwater impacts on benthic ecology

Contaminants derived from urban run-off have been shown to accumulate in estuarine sediments, reaching concentrations that are potentially capable of causing adverse biological effects (Section 7.2). However, conclusive demonstration of effects is difficult due to the confounding effects of strong natural environmental gradients (eg sediment texture, salinity), multiple past or present point sources of contamination, and natural variation in biological populations.

Issues

The major issue is the reduction in benthic habitat quality and hence benthic community biodiversity and community composition as a result of accumulation of stormwater-derived contaminants and sediment. Measuring the abundance of animals in sediments (benthic ecology) is the most appropriate measure of the effect of contaminants on ecological communities because:

1. Contaminants in stormwater run-off accumulate in sediments within marine receiving environments.
2. A diverse range of organisms live in marine sediments and are continuously exposed to stormwater contaminants.
3. Sediment dwelling organisms are a major component of broader estuarine, harbour and coastal ecosystems, providing food for birds, fish and humans, and affecting water quality, nutrient cycling and productivity.
4. Contaminant levels can reach concentrations that are toxic to sediment dwelling organisms.
5. Some contaminants can bioaccumulate to toxic levels in sediment dwelling organisms even at relatively low concentrations in water and sediment.

A large number of animal species inhabit the soft sediment intertidal areas of the Auckland region. In a recent survey of six sites in the Manukau, 99 species were found on a single occasion (ARC unpublished data).

State of Knowledge in 1995

There had been many studies of impacts of industrial and sewage discharges, which may include effects of stormwater. There were no studies focused specifically on the impacts of urban stormwater.

Advances Since 1995

Advances have come in three areas:

1. Methods of analysis.
2. Specific studies of benthic ecology in urban estuaries.
3. The Healthy Benthic Community Model.

7.5.1 Methods of analysis

The response of individual species to pollution depends upon their tolerance to contaminants, as well as associated changes that may occur to the chemical, physical and/or biological characteristics of their habitat. For instance, tolerant species may benefit from the loss of sensitive species, and actually increase in abundance in moderately polluted habitats. Consequently, in systems with relatively high diversity it can be difficult to disentangle community changes due to pollution from those caused by other factors.

Several methods are commonly used to monitor biological responses to pollution. Diversity indices and indicator species provide very coarse measures of ecological response, because they do not take into account the types of taxa present at a site, or only consider the response of few species. Consequently, there is a risk that biologically important changes may be missed by these techniques. Multivariate techniques, which take into account variations in all of the species, offer a more sensitive alternative for detecting ecological responses.

Advances in computing power mean that multivariate techniques are now the favoured method of assessing and comparing the composition of ecological communities. Without these techniques, community data must be reduced to indices, which do not maintain important information on taxa composition, or data on individual species must be considered independently using univariate techniques. This quickly becomes unwieldy as the number of species increases, making it difficult to detect relationships between biological variables and physico-chemical processes, such as ecological responses to pollution. In multivariate analyses each species (or taxon) within a community is regarded as a dimension in multivariate space. Changes in the abundance of a species from one site to the next can be thought of as a difference between the sites in that particular dimension. Multivariate ordination methods are generally used to reduce the number of dimensions to allow graphical representation of the sites in 2 dimensions, where the relative distances between sample points (sites) in the diagram correspond to the relative differences between communities (see Figures 82 and 83 for examples of these plots).

Canonical, or constrained, ordination techniques have been developed, which reduce dimensionality by projecting communities onto a particular variable or gradient of interest, such as a pollution gradient (eg canonical correlation analysis). These canonical methods can be used to: (a) test for significant relationships between community structure and a pollution gradient, (b) construct a model of community structure by reference to a gradient and (c) place new observations (sites) along the model canonical axis for classification.

A key application of this approach has been the development of the "Healthy Benthic Community Model", which is being used to assess the effects of sediment pollution in Auckland estuaries and harbours on benthic ecological health. This is described later in this section.

Canonical methods of analysis are used to assess impacts in the freshwater environment as well (see Chapter 6).

7.5.2 Impacts of mature urban areas

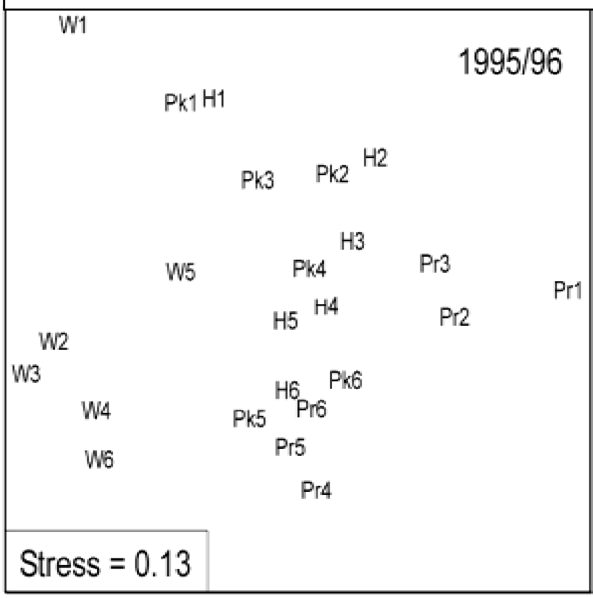
As the urban area matures, the receiving environment impacts shift from an increase in fine sediments to impacts from higher flows and levels of potentially toxic contaminants. As described in the start of this section, such impacts are difficult to detect and define.

Four estuaries study

The first attempt to determine the ecological effects of stormwater in Auckland estuaries compared the benthic invertebrate community in two urban estuaries (Hellyers and Pakuranga) with two rural estuaries of similar morphology (Paremoremo and Te Matuku (Waiheke Island) – the "four estuaries" study described in Section 7.2 (Morrisey et al. 2003).

Concentrations of contaminants were higher in estuaries with urbanised catchments, and concentrations of Cu, Pb, Zn and DDT in some samples exceeded those at which biological effects may be expected to appear.

Figure 82
Graphical presentation of results from canonical correspondence analysis conducted on benthic biota from six sites in two urban estuaries (H1-H6, Pk1-Pk6) and two rural (W1-W6, Pr1-Pr6) estuaries (from Morrisey et al. 2003).
Sites that are similar are grouped together, while sites with different characteristics are further apart. The diagram shows that Waiheke Island sites (W) are different from the two urban sites (Pk & H), which differ from Paremoremo (Pr). There is no difference between the two urban sites. However, there is a degree of overlap between the sites indicating that differences in biota between the sites are not great. This is not surprising because the level of contamination in the urban sites have only just reached levels of concern.



Benthic communities in the two urban estuaries were not significantly different, but the urban estuaries differed from both rural estuaries, which also differed from each other. Distributions of benthic invertebrates were significantly related to environmental variables, both natural environmental variables (nature of the sediment, position in estuary) and contaminants. Differences in faunas between the urban and non-urban estuaries were not, however, clear-cut (Figure 82), nor were relationships between faunal assemblages and environmental variables consistent between two times of sampling, emphasizing the difficulties in detecting and determining the cause of impacts.

Sediment transplant study in the "four estuaries"

Sediments from the two urban estuaries and the rural Paremoremo estuary were defaunated by freezing, then transplanted into the relatively uncontaminated Te Matuku estuary, to see whether the animals that recolonised the sediments were related to the level of contamination in the sediments. Few differences in recolonising fauna were found. Artefacts such as changing pore water ammonia and dissolved organic carbon concentrations, and contaminant dilution by deposition of relatively uncontaminated sediment onto the urban sediments, confounded analysis of effects (D. Morrisey, NIWA, pers. comm.).

Auckland City stormwater study

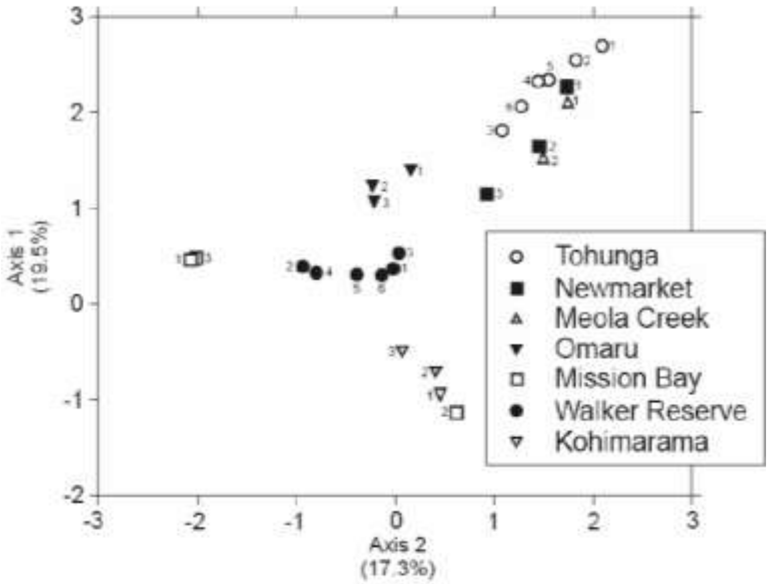
A study of sediment quality and biological community composition at seven sites adjacent to stormwater discharge points around Auckland City (NIWA 2000e) indicated that the greatest impacts of stormwater discharges were likely to be in depositional (low energy) environments.

In high energy environments (Walker Reserve, Mission Bay, Kohimaramara Beach), concentrations of Zn, Cu and Pb were close to background. In the low energy environments, which received run-off from older urban areas, (Hobson Bay, Pt England, Meola Creek) higher concentrations of the three key elements were found.

The benthic communities were different at the lower and higher energy sites (Figure 83), and were correlated with environmental variables including sediment grain size (texture), organic content, and contaminants (Zn). When the effects of variable sediment texture were removed, 77 per cent of the variation in community composition could be explained by organic content and Zn concentration. As with the previous study, this assessment does not "prove" that the biota at depositional sites are affected by these metals, but it suggests that they could be.

Figure 83

Graphical presentation of results from canonical correspondence analysis conducted on benthic biota from adjacent to stormwater discharge points in Auckland City (taken from NIWA 2000e).



7.5.3 Healthy Benthic Community Model

Auckland Regional Council commissioned NIWA and University of Auckland to investigate methods for assessing the health of benthic communities in relation to pollution levels, specifically for Auckland's harbours and estuaries.

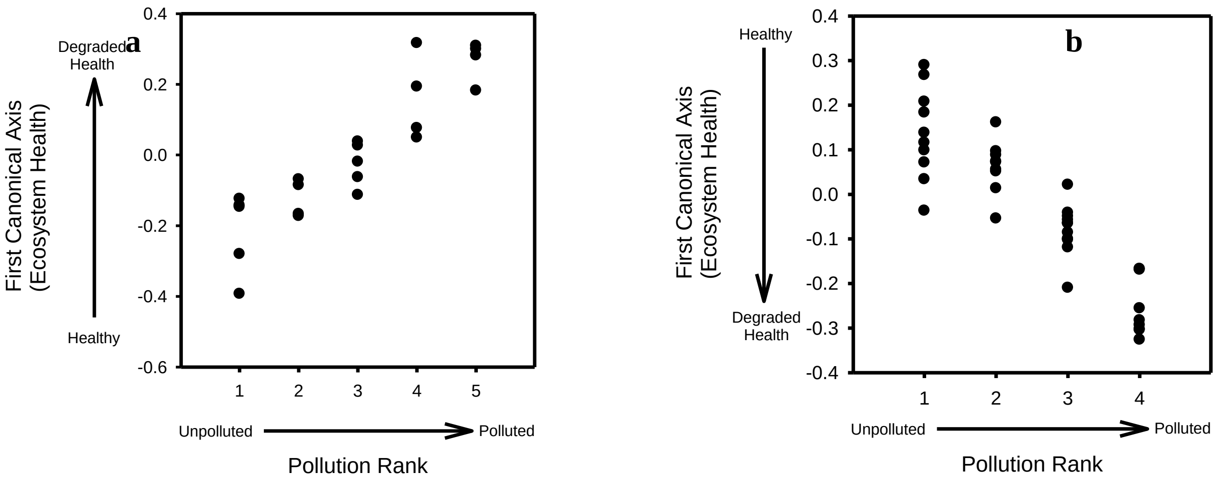
Anderson and Robinson (2003) and Anderson and Willis (2003) have generalised canonical analysis to allow any measure of community similarity to be used as the basis of the analysis. Their "canonical analysis of principle co-ordinates" (CAP) method was adapted by ARC to rank the community structure of new or re-surveyed sites against the community structure of existing sites with known levels of pollution (ARC 2002h, 2003g).

Separate CAP models were required for settling and outer zones because of differences in the biology and sediment dynamics of these areas. Community structure (species counts) and pollution rank data from 22 settling zone and 34 outer zone sites are presently used in the analyses. The settling zone model includes 117 species while the outer zone model 144 species.

Sites were grouped into five "pollution ranks" depending on the level of sediment contamination. The relationship between ecological health and sediment contamination (pollution rank classes) is shown in Figure 84. The change in ecosystem health with changing level of contamination is apparent, even though there is some variation leading to overlap between adjacent pollution rank groups. The effect of potentially confounding factors such as sediment texture and salinity, which may have contributed to the observed relationship, were checked and found not to invalidate the model.

Figure 84

Results of the gradient canonical analysis of principle co-ordinates (CAP) analyses on (a) settling zone and (b) outer zone sites[33]. The plots show the same key feature, that benthic community health declines as pollution levels increase.



The reliability of the model in predicting ecological health from sediment contamination was examined by checking how sites that were ranked as amber or red for sediment quality rated for ecology. All of the new settling zone sites ranked as amber or red for sediment quality were ranked as amber according to ecological health. The lack of concordance between red sediment chemistry and benthic ecology may mean that the amber-red sediment chemistry ERC are not optimal. However, at present it is not sensible to change the amber-red ERC, given criteria are based on widely accepted sediment quality guidelines, and there are insufficient data to set new values.

The ecological results support the conclusion that ecological impacts occur in the amber sediment quality range. Benthic communities of sites with red sediment quality showed signs of ecological stress, but they were not degraded to the point where the ecology was severely impacted. Remedial actions that reduced or reversed the accumulation of contaminants would be expected to delay further degradation, or improve, the ecological health of these sites.

The Healthy Benthic Community Model supports the conclusion that ecological health is affected by contaminant levels present in Auckland's harbours and estuaries. However, a lack of data from highly degraded outer zone sites, and a relatively high degree of overlap between outer zone sites ranked 1 and 2, and 3 and 4 (see Figure 84

[33] Note that the ordination axis represents a constrained axis that does not have a fixed orientation in multivariate space. Consequently, the relationship between pollution rank and the ordination axis (which represents community health) varies between settling and outer zones. This is not unexpected and does not alter the validity of either model.

(b) above), means that it is not sufficiently robust to be used as a ranking tool. Nevertheless, in its present form, the outer zone community health model remains a useful tool for monitoring trends in ecological health. The addition of extra data, particularly from highly contaminated sites, should increase its utility.[34]

7.5.4 Summary

Determining the effects of stormwater on benthic ecological health has not been straightforward, due to the combined effects of:

- relatively low levels of contamination (in most places) compared with known biological effect thresholds (eg ERC);
- multiple contaminants – including known species such as metals, organics, organic matter, and fine sediment, and possibly also chemicals not yet analysed in Auckland's marine sediments;
- natural variations in sediment characteristics;
- spatial and temporal variation in biological community composition, and;
- variation in the way contaminants have been analysed between studies.

It has required the relatively recent development of sophisticated multivariate statistical methods, coupled with consistent contaminant sampling and analysis protocols to tease out the effects and present them in an easily understandable form – the Healthy Benthic Community Model. Work is continuing to improve the model, in particular for Outer Zone areas.

7.6 Sedimentation impacts on benthic ecology

Input of sediment into the marine environment is largely related to land disturbance. This can include earth working during urban development, but can also include other land uses such as forestry operations, farming in erodible landscapes, and as a natural consequence of New Zealand's dynamic landscape and high rainfalls.

Effects can occur in the water column, due to suspended sediment (SS), or after deposition on the floor of estuaries and harbours. Depending on the levels and duration of sediment inputs, effects can be catastrophic (acute), sublethal, and/or cumulative.

Sediment is a key stormwater contaminant, contributing to the infilling of Auckland urban estuaries and changes in benthic ecology. It is therefore important to understand what kinds of effects are caused by increased sediment delivery to the marine environment, and how these effects change with variations in sediment exposure. Sediment type, frequency of delivery, and amounts, as well as receiving environment characteristics, are all potentially important factors contributing to impacts.

[34] Further development of the ecological health models has been made since this review. Refer to ARC 2006c (TP317) for details.

Potential issues

Changes in marine ecology due to increased levels of suspended sediment in the water column and/or degradation of benthic habitat after sediment deposition.

State of knowledge in 1995

In 1995, there were few studies and little information on the effects of terrigenous sediment on soft sediment marine ecosystems, even internationally.

Advances since 1995

Since 1999, a large amount of research, both FRST and ARC-funded, has been undertaken on the effects of sedimentation on macrofaunal communities. The results from many of the studies have been published in the scientific literature.

While this information is all available in various reports and scientific papers, a NIWA report by Gibbs & Hewitt (ARC 2004n) provides a synthesis of the different strands of the research and documents our current understanding of the effects of sedimentation on macrofaunal communities. This section quotes liberally from this review[35], which is recommended reading for more complete coverage of this very important topic.

The research covers catastrophic (acute), sublethal (chronic), and cumulative impacts and is based on field and laboratory experiments, surveys of macrofauna inhabiting Auckland's estuaries and harbours, and sedimentation history.

7.6.1 Effects of suspended sediment

Terrigenous sediment[36] normally enters coastal waters as a suspension, following erosion of a disturbed catchment during rain. Consequently, the first impact on the benthic communities is an increase in turbidity or suspended sediment (SS) concentrations. Increased turbidity reduces light penetration into the water column, reducing primary production of phytoplankton and benthic microphytes (algae that live in or on the sediments) and thus reducing a key food component to suspension feeders, herbivorous benthic grazers and deposit feeders. Other effects include oxygen depletion in the water and underlying sediments, and changes in sediment biogeochemistry, such as reduced nutrient supply and ammonia accumulation.

Many benthic animals can withstand high levels of SS for short periods (typically a few days to a week), but some animals are negatively impacted when exposed for longer times. Effects vary between species, as illustrated in Figure 85 for four common shellfish species.

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[35] Refer to ARC TP264 for references to the individual research studies, which have not been directly quoted here.

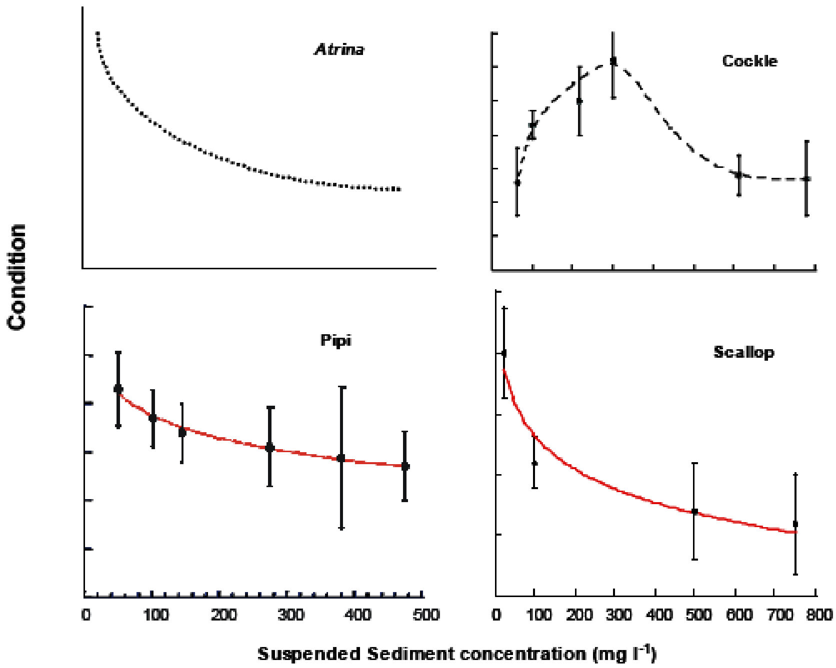
[36] Terrigenous sediment is derived from the land through erosion and, in the Auckland region appears as yellow-orange "clay" – mainly silt and very fine sand sized material. It is often very much finer sediment than the ambient sediments from sandy and shelly intertidal estuarine zones. However, it has less clay content than the mudflat sediments.

Experiments show that horse mussels, which are an important benthic species around Auckland, are increasingly stressed by increasing suspended sediment concentrations. Cockles, a widespread animal and food source, have difficulty with SS concentrations >400 mg/L after long periods (typically >week), while scallops have difficulty when concentrations exceed 100 mg/L for long periods. Experimental results are confirmed in the field – eg growth rates of juvenile cockles and adult reproductive status of cockles and pipis at Whitford Bay were adversely affected by high SS concentrations.

As well as the amount (or concentration) of SS, the type of SS is important in determining effects – eg terrigenous sediment affected cockles more than marine sediment.

Figure 85

Changes in condition of four suspension-feeding bivalves with increasing suspended sediment concentration (ARC 2004n).



Sediment feeders, as well as suspension feeding animals, have also been shown to be affected by high SS concentrations. The heart urchin, a large burrowing deposit feeder common in both sandy and muddy subtidal habitats, was adversely affected after three days at SS concentrations above 80 mg/L. Death rates and times to safely bury itself from predators increased with increasing exposure to SS. The deposit-feeding worm, Boccardia, which lives in tubes that protrude from the sediment surface in intertidal and subtidal habitats, was also adversely affected at SS concentrations above 80 mg/L, but only after nine days. Again the effect of terrigenous material was more pronounced than marine sediments. The surface feeding wedge shell, a common and abundant inhabitant of soft sediments, was adversely affected at SS concentrations above 300 mg/L after nine days. After 14 days of exposure to the highest SS concentrations, most had died or were lying exposed on the surface of the sediment (where they are susceptible to predation and scavenging).

The concentrations of SS required to affect or kill animals can be quite high, and such prolonged high concentrations are unlikely in estuaries, except during large run-off events from unstable landscapes. However, the cumulative inputs of fine sediments and their high concentrations after resuspension by wind and waves in shallow waters can prolong the exposure of animals to high SS on the sea bed.

7.6.2 Effects of sedimentation

Terrigenous sediment discharged to the marine environment can become a subtidal or intertidal deposit, depending on the state of the tide during sediment discharge. During the ebb tide, sediment tends to be carried into sub-tidal zones, whereas during the flood tide the sediment may be deposited on intertidal sand and mud flats. The thickness of the deposited layer and the period it remains determines the impact on the benthic community.

Intertidal situations

Catastrophic deposition events associated with large, sudden, depositional events are fortunately rare. Long-lasting thick layers of terrigenous sediment kill most animals in the intertidal zones and the recovery process is slow. Based on laboratory and field experiment results, the critical burial depth has been defined as 2–3 cm.

Figure 86
Catastrophic sediment dump on a cockle bed after a storm (ARC 2004e).

Unlike subtidal deposition, where the terrigenous sediment remains partially fluid for some time, thick layers of terrigenous sediment on the intertidal zone become dewatered when exposed at low tides. In hot weather the sediment becomes hard and impervious to small macrofauna.

Large crabs can burrow through terrigenous sediment layers up to 9 cm thick and are the first to colonise a new deposit. This burrowing action helps breakdown the terrigenous sediment deposit by mixing it with the natural sediments from below.

Wave action, especially as the deposit becomes immersed on each tidal cycle erodes the terrigenous sediment and continues the breakdown and burial process.

The results of NIWA catastrophic sedimentation studies in Okura and Whitford estuaries showed that:

- In Okura, within 10 days of thick layers of terrigenous sediment being applied to intertidal flats, nearly all the macrofauna in the clay/silt were killed. Monitoring during upstream earth working activities showed that a 1–2 cm deposit of eroded terrestrial sediment lasted >7 days and adversely affected macrofauna.

- There was consistently slow recovery of experimental sediment dump plots from catastrophic deposition lasting up to two years in estuaries, although a single storm event could bury the terrigenous sediment in a coastal situation.
- Recovery of ecology typically lags behind the recovery of the sediment. For example, at an experimental site at Whitianga (Coromandel), recovery of sediment properties occurred after 50 days, ie all symptoms of the applied sediment had disappeared, but biological effects were still observed for some species after 210 days.
- Laboratory experiments found high mortality in six test animals, including the mud snail. Adult bivalves such as pipis, wedge shells and cockles showed stress after 1–6 days, but the mud crab and the snapping shrimp were unaffected.

Thick deposits of terrigenous sediment also adversely affected mangroves by covering their pneumatophores and blocking oxygenation of their roots.

Figure 87
NIWA staff conducts experiments on ecological effects of sediment deposition on intertidal sand flats (ARC 2004e).

Figure 88
Thin, sub-lethal terrigenous mud deposit on a sand flat following light rain in a disturbed catchment.

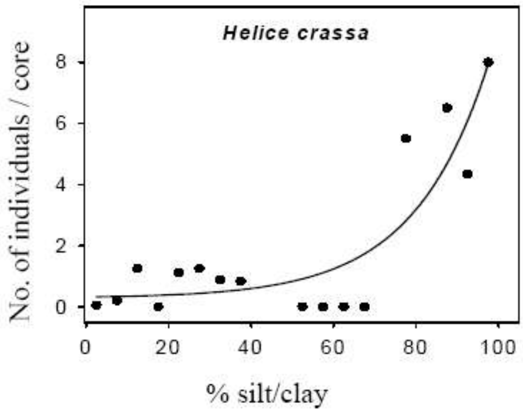
Sub-lethal, non-catastrophic events are the most likely form of terrigenous sedimentation on the intertidal zones. Depositions of thin layers of sediment are unlikely to cause defaunation, but subtle changes, which may be cumulative and ultimately alter the structure of the benthic communities, can occur.

For example, mat-forming benthic microphytes are likely to die beneath even a very thin terrigenous sediment deposit in the finer sediments, while terrigenous sediment may stabilise coarser sands against movement by wave action, thereby allowing other benthic microphyte species to thrive.

Macrofauna generally survive unaffected, although adverse impacts on most taxa occur with terrigenous sediment deposits of >3 mm. The exception was the mud crab *Helice*, which is highly tolerant to silt/clay environments (Figure 89). Benthic macrofauna can remain in their habitats during sediment recovery following deposition of thin layers of sediment, albeit at lower numbers (Figure 90).

Figure 89

The positive effect of increasing silt/clay content of the sediment on populations of the mud crab (ARC 2004n).

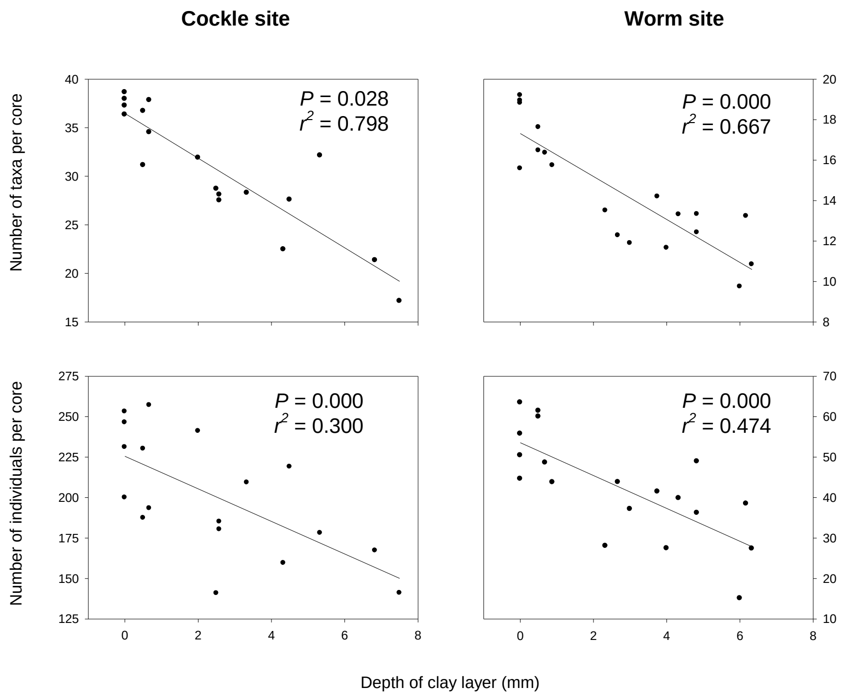


The type of sediment, as well as the amount, can be an important factor. For example, although cockles could selectively reject organic detritus to feed on suspended algae (phytoplankton), they were unable to reject the fine terrigenous sediment particles in the same size range as the phytoplankton, and thus lost their condition.

Studies at Whitford indicate that thin layers of sediment are also likely to aid the spread of mangroves but do not appear to affect salt marsh meadows.

Figure 90

Thin layers of terrigenous sediment reduce the numbers and range of types of benthic animals, which recover as sediment thickness reduces (ARC 2004n).



Subtidal situations

Terrigenous sediment impacts the benthic community by settling on the sea floor and forming a blanketing layer that alters the sediment structure. Underwater, the sediment does not form a hard layer but produces an amorphous ooze through which the benthic macrofauna must burrow to survive. Benthic algae, which form an important part of the marine food web, are trapped beneath the terrigenous sediment and die.

Significant changes in benthic communities occur with additions of 1–2 cm of sediment, and some changes (eg on some worms and shrimps) have been observed with as little as 3 mm. Effects are complex, depending on the animals involved and observation time scales. Larger filter feeders (horse mussels, golf ball sponges and a sea squirt) were generally not killed or buried by the deposits, but there were significant effects on their condition and ability to feed when these were assessed in the laboratory three weeks after sediment addition. Heart urchins lay dead on the surface or were scavenged when they were forced to the surface, probably because their breathing tubes were blocked by the deposit.

Based on studies conducted in Mahurangi Harbour, it appears that harbour environments are less sensitive to additions of sediment than coastal waters, possibly because harbour-dwelling animals are conditioned to cope with a greater degree of sedimentary disturbance.

7.6.3 Relevance to resource management

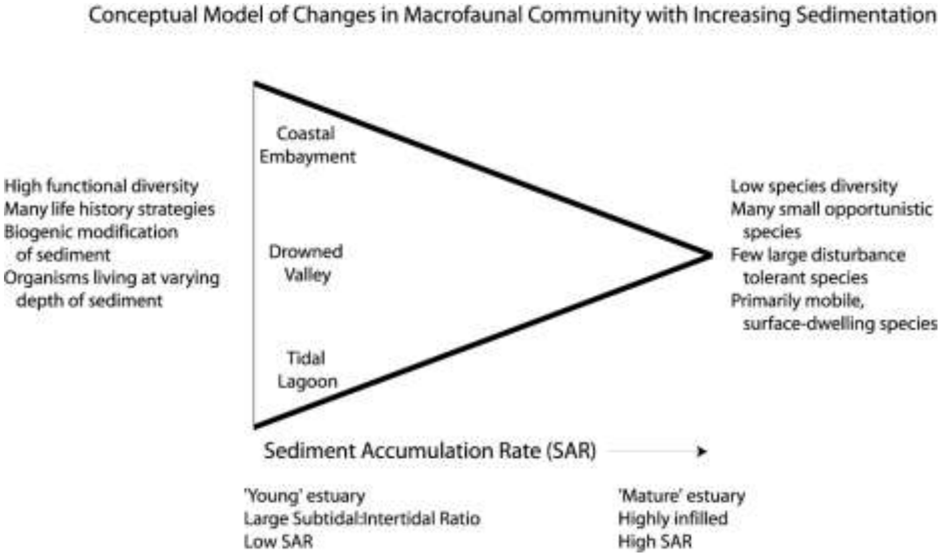
Assessing how changes are likely to influence biodiversity and ecosystem values are key elements of ecological science, but once the magnitude and scope of a particular problem has been identified, it is also important that scientists provide information to help managers minimise risk and reduce threats. Important applications of ecological knowledge relate to defining threshold effects, prioritising actions, and forecasting ecological responses.

In estuaries, multiple habitat types, such as salt marsh, sea grass and unvegetated intertidal flats promote diversity by enhancing recruitment and maintaining species with requirements for multiple resources. The modification of available habitats due to elevated sedimentation has been shown to reduce the overall ecological heterogeneity. The general exclusion of slow-growing, large species frequently results in lower diversity and reduced ecological functioning of ecosystems.

Although muddy areas are generally thought of as being of low diversity, this is not necessarily the case. A study of intertidal and subtidal muddy sites in a number of different estuary types (drowned river valleys, tidal lagoons and coastal embayments) found a range of community types in Auckland estuaries, with diversity and community type responding to rates of sediment accumulation.

All three types of estuaries studied were comprised of diverse, variable communities at early stages of estuarine infilling (corresponding to low or natural rates of sediment accumulation). The three types of estuaries then appeared to converge to a similar low diversity, stress-tolerant community as sediment accumulation rates increase and estuaries became infilled (Figure 91).

Figure 91
Conceptual model of changes in macrobenthic community structure as sediment deposition increases, ranging from "young" estuaries with low sediment accumulation rate (SAR) to "mature', highly infilled estuaries with high SAR (ARC 2004n).



An example of this application can be found from detailed benthic invertebrate surveys of the Upper Waitemata harbour (ARC 2002i). By investigating the relationship between the mud content of the sediment and the abundances of individual taxa in different part of the upper harbour, several taxa were identified that are likely to be negatively impacted by long-term habitat change resulting in increases in per cent mud. Monitoring changes in the abundances of these taxa may give some insight or forewarning of habitat degradation due to increased sediment loads, say from erosion during earth working. The relationships also give credence to the imperative to control soil erosion during urban development.

This type of deterioration in a major estuarine system has been observed in Mahurangi Harbour. Long-term monitoring carried out for the ARC has noted trends in the abundance of several species consistent with increased sediment accumulation rates (ARC 2005i). This was in response to an increase in the amount of fine sand at all sites sometime between April 1996 and April 1997. However, instead of a "pulse" response by macrofauna, gradual declining or increasing trends have been observed. Changes in Atrina (horse mussel) abundance at the subtidal sites occurred prior to this time, but work described earlier has shown that species is particularly sensitive to fine sediment increases.

Changes in fringing vegetation, particularly mangroves, and the distributions of seagrass beds are often associated with increased sedimentation. Mangrove progradation has been observed in many Auckland estuaries (Roper et al., 1994; Morrisey et al. 1999), although it is not always clear whether the progradation is driven by increased sedimentation or causes the increased sedimentation[37]. Ellis et al. (2004) observed decreased diversity of macrofauna inside mangrove areas versus the mudflats surrounding mangroves and this decrease was more apparent in older stands of mangroves.

7.6.4 Management tools

An approach used to provide information on the magnitude and scope of likely sedimentation effects in estuaries that has been used successfully by the ARC is a combination of **modelling, surveying and experimentation**. Experiments on effects of terrestrial sediment deposits on macrobenthic communities in different habitats are combined with models of sediment run-off from the catchment and particle dispersal models within the estuary, to assess the relative ecological risk to different estuarine habitats under different development scenarios. These risk assessments allow managers to contrast the threats posed by various scenarios of land development and thus improve decision-making.

Another approach is the use of **statistical models** that relate ecological variables to environmental factors enabling forecasts of species responses to increased muddiness or changes in the relative proportion of habitats to be made. This approach assumes that the relationships apparent in the observed spatial patterns match those that will

[37] A comprehensive review of mangroves has been published since this review (ARC 2007, TP325).

occur over time. Figure 91 is an example of a model for the effects of increased sedimentation rates over a long-time scale on communities in Auckland mudflats.

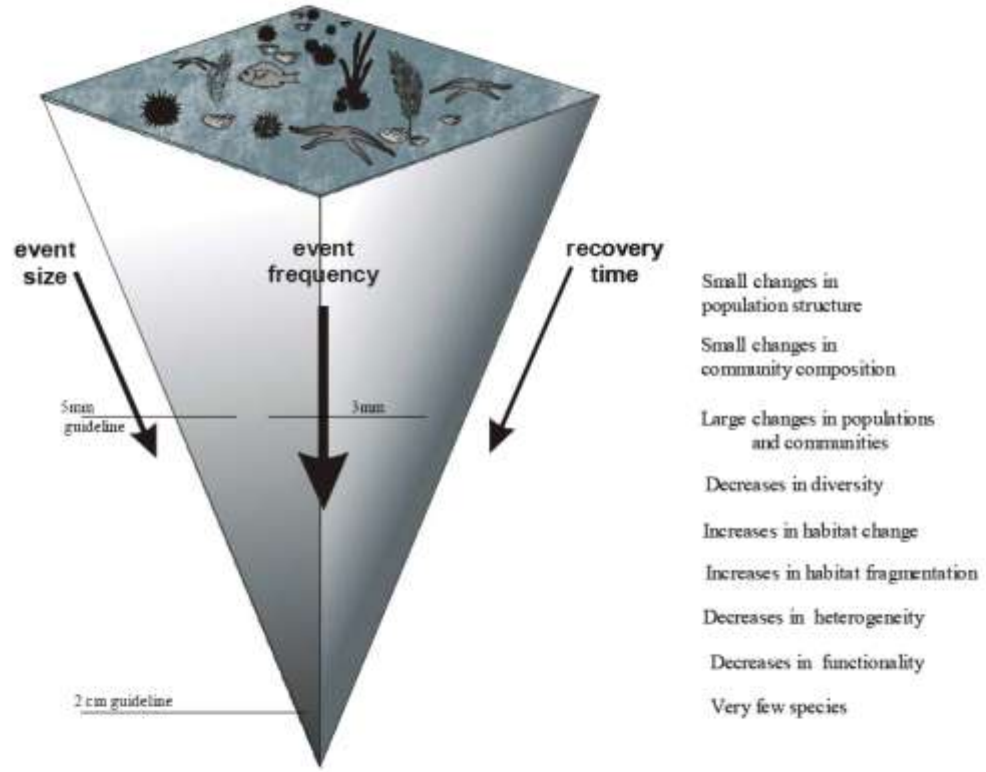
A third approach is to use **species that are sensitive to increased sedimentation.** Such species can be identified both by experiments and the models developed from surveys. The work that has been done on the effects of increased sedimentation rates, suspended sediment concentrations and mud content can be summarised to produce a list of species sensitive to changes in sedimentation rate and increased mud content. Large-scale spatial surveys enable scientists to relate ecological variables to environmental factors so that models of species responses to increased muddiness can be made. For example, a study by Norkko et al. (2001) for the ARC presents an overall compilation of species sensitivity based on their peak abundance, and their range of distribution across different sediment types. Of all the 38 taxa assessed, 26 were found to prefer sandy sites, five were found to be intermediate and seven were found to prefer higher mud content of the sediment.

The results of this and other studies in New Zealand has resulted in a list containing 56 animals categorised in terms of sediment preferences: strong sand preference; sand preference, prefers some mud but not high percentages; mud preference; strong mud preference. This list also contains information on abundance and the roles (functions) that the animal performs in the environment.

Conceptual models are also important tools for environmental management. For example, terrestrial sediment can influence estuarine and coastal ecology, both through increased SS concentrations and sedimentation rates. Increased SS concentrations can influence primary production and the benthic animals that feed by filtering water. These effects can flow through the communities and ecosystem along a variety of pathways. A single conceptual model presented in ARC (2004n) summarises these concepts (Figure 92).

This model assumes that changes in the estuarine sediment-loading regime will, by favoring some species and habitats over others, influence estuarine biodiversity. Event frequency, extent and magnitude will influence the recovery response of the benthic community by affecting habitat suitability and the possibility for undisturbed areas to provide colonists to disturbed areas. The result is that, with increasing frequency, extent and magnitude, recovery time increases and depletion of sensitive species occurs. Finally a depleted community or estuary occurs, with low diversity and low function.

Figure 92
Conceptual model modified showing broad scale changes in biodiversity and recovery time as sedimentation event size and frequency increase (ARC 2004n).



7.6.5 Effects of sedimentation: summary

A huge amount of work has been undertaken in the last several years to gain a better understanding of the effects of sedimentation on benthic aquatic life. This work has shown that sediment has a wide range of impacts over short and long time frames, related to the amounts, frequency, and duration of sedimentation.

As a result of this work, four general guidelines have been produced (ARC 2004n):

1. In general, the thicker the layer of mud, the more animals will be killed and the longer recovery will take. This will affect both the number of species and the number of animals within each species. Some species are more sensitive than others, and these will be affected first/most.
2. If mud that has been washed down a stream to one of the tributary estuaries or the embayment results in a mud layer greater than 2 cm thick, remaining for longer

than five days, then all the resident animals in that area (with the exception of mobile crabs and shrimp) will be killed due to lack of oxygen.

3. A mud thickness of around ½ cm, persisting for longer than 10 days, will reduce the number of animals and the number of species, thereby changing the structure of the animal community.

4. Frequent deposition of mud, less than ½ cm, may still have long-term impacts that can change the animal communities.

It is important that the limitations of knowledge, either in terms of information gaps, underlying assumptions of studies and models and the effect of simplification of results, are understood. The large spatial and temporal scales of community responses in an estuary or coastal area have been extrapolated from the spatial and temporal scale of a manipulative or one-off survey. The underlying assumption is that the processes defined in the small-scale experiments in one part of the system can be applied to the whole system. Interactions between the stress caused by increased sediment inputs and other factors, whether natural (eg predation) or anthropogenic (eg contaminants) are likely to be multiplicative.

Information gaps

There has been little work on sub-lethal effects such as organism health, growth or reproductive output. All of these are likely to be important when making long-term forecasts of population and community changes.

Little knowledge has yet been generated on the effect of duration and frequency of events over long-time scales, or on large-spatial changes, to the distribution of estuarine and coastal habitats.

Interactions between the stress caused by increased sediment inputs and other factors, whether natural (eg predation) or anthropogenic (eg contaminants) are likely to be multiplicative. Further work on this aspect of multiple stressors is required.

7.7 Toxicity of urban stormwater in the marine environment

The toxicity of environmental pollutants can be determined by a variety of approaches including:

- laboratory bioassays, in which test organisms are exposed to polluted sediments or water;
- in-situ tests, where test organisms are exposed to the contaminants in the environment, and
- interpretation of the health of naturally occurring species or communities present in the receiving environment.

Laboratory tests are the most commonly used methods for assessing toxicity to marine test organisms. Interpretation of biological community health has a broader ecological base, and is described in "Impacts on Benthic Ecology".

Toxicity tests are described in two ways:

- **Acute** toxicity, which occurs when the concentration of a pollutant is high enough to cause death or some other adverse effect in a relatively short time (typically, two to 10 days for most test organisms) compared to the organism's life span.
- **Chronic** toxicity occurs when the pollutant(s) are present in concentrations too low to cause an immediate effect but high enough to have detrimental effects on aquatic living organisms in the long-term. Examples are abnormal development of juveniles, retarded growth rates, reduced reproductive capacity, impaired behaviour (eg reduced burrowing rates by shellfish), avoidance behaviour, physical deformities (eg tumours, abnormal shell structure) and physiological stress (which can underlie retarded growth and a reduced capability to reproduce).

An acute toxic response to test organisms indicates a serious problem associated with high levels of one or more chemical contaminants or non-chemical stressors. Chronic toxicity is less immediate, but no less serious, as it indicates stressful conditions that may upset longer-term ecological health.

Chemical analysis may reveal contaminants at high enough levels to be responsible for the observed toxicity, but often this is not the case, and the cause of the toxicity remains unexplained. Toxicity tests are therefore useful "integrative" indicators of the impacts of contaminants or stressors introduced by stormwater contamination, providing "biological effect" information that chemical analysis of suspected causative agents may not be able to provide.

Potential Issues

Reduction in the ecological health of receiving environments due to the combined stressors present in urban stormwater.

State of knowledge in 1995

The 1995 overview of stormwater impacts gave a comprehensive review of observations on toxicity tests in Auckland conducted in the early 1990s and the findings are still relevant. The 1995 overview is recommended reading.

Auckland stormwater pond sediments were shown to be acutely toxic to a common marine amphipod. Estuarine sediments impacted by stormwater (ie had significantly elevated concentrations of urban-derived contaminants) were not acutely toxic, except for one site near an industrial discharge with very high levels of contamination.

Chronic toxicity was observed in a number of interesting and diverse studies. Rock oyster condition deteriorated with distance down urbanised estuaries: along the north shore of Manukau Harbour from Mangere Inlet to the Heads, and down Tamaki Estuary from Otahuhu to Browns Island. The distance down these estuaries correlated with a contamination gradient. In laboratory tests, juveniles of wedge shells and cockles showed avoidance of contaminated sediments by not burrowing or shifting to uncontaminated sediments when placed on the sediment surface. Large scale in-situ doping of sandy sediments with chlordane to levels which would be predicted to cause adverse biological effects to benthic organisms resulted in loss of juvenile wedge shells. Recruitment of juveniles occurred after storms had lowered chlordane concentrations.

These studies showed that urban run-off might cause chronic toxicity to marine benthic animals in some situations, and this could possibly contribute to a decline in ecological health of benthic communities. The studies also showed that understanding the effects of even individual contaminants on benthic ecology required sophisticated field experiments. Toxicity of a complex mixture like stormwater is not easily measured or explained, especially after discharge into the dynamic marine receiving environment.

Advances since 1995

Since 1995, advances have been made in:

- measurement of toxicity of water and sediment from stormwater ponds;
- bioavailability, and hence potential toxicity, of organic contaminants;
- phototoxicity of PAH, and;
- combined approaches to toxicity assessment (eg "triad" method).

However, relatively little work has been done in Auckland since 1995, except in the application of these advances, as summarised below.

7.7.1 Stormwater pond toxicity studies

NIWA assessed the toxicity of stormwater discharged from stormwater detention ponds to the marine environment at an industrial site (Pacific Steel) in 1995, and an urban site (Unitec) in 1997 (NIWA 1997, Hickey 1999). Water and sediment samples from pond inlets and outlets from four storm events were tested to gauge pond performance at removing toxicity and contaminants (metals and ammonia in waters, metals and PAH in sediments) and to assess the likely toxicity of treated stormwater discharged to the receiving environment.

Key findings were:

- Stormwater toxicity was variable from storm to storm, but was generally only slightly-moderately toxic in a range of tests. It was estimated that dilution of about three times was sufficient to ensure that there is no toxicity in the receiving water after discharge.
- Some tests showed a reduction in toxicity of stormwater as it passed through a treatment pond, but nearly as many tests showed similar toxicity at the inlet and outlet of the pond.
- Stormwater toxicity was moderately low compared with that anticipated from the elevated metals concentrations in the water samples, suggesting that bioavailability of these contaminants (Zn, Cu, Pb) was low.
- Sediments that had accumulated in the pond were highly toxic, even at the outlets. This indicated that sediment discharged from the ponds could still have significant toxicity to downstream receiving environments.

- There was a marked difference between the sensitivity of the test organisms – amphipod survival was minimal at all sites, whereas oligochaetes showed improved survival and reproduction through the ponds. Oligochaete toxicity was strongly related to contaminant concentrations in the sediments, both metals and PAH.
- Tests of the toxicity of contaminated suspended sediment from treatment ponds were made on marine amphipods and juvenile bivalves. The animals were exposed to sediment made from mixing suspended sediment from the treatment ponds at UNITEC in Auckland with uncontaminated marine sediments. Acute toxicity was observed in one series of tests but not in another. A threshold response was estimated to be 10 per cent (EC10) of stormwater particulates. The stormwater solids used in the test exceeded the ER-L guideline (where effects might begin to occur) for most urban contaminants, with Zn and dieldrin exceeding the ER-M (a level where significant adverse effects would be expected to occur).

These tests indicated that stormwater from detention ponds was likely to be slightly to moderately toxic – probably chronically toxic, but only occasionally acutely toxic. Only a relatively small dilution (approximately three times) in the receiving environment was required to reduce metals' concentrations to below toxic thresholds. Stormwater sediments were more toxic, and a 10-fold dilution by clean sediment was predicted to be required to get below a toxic threshold.

7.7.2 Triad study: sediment chemistry, toxicity, and community structure

Nipper et al. (1998) investigated the effects of mild contamination in mudflats using the "triad" approach. The sediment triad approach compares the relationship between sediment toxicity, sediment and pore water chemistry, and benthic community structure. Six of the eight study sites were in Auckland (others were at Raglan and Kawhia) – three stations in Tamaki Estuary, two in Manukau Harbour, and one at Okura. Of these six, two were relatively uncontaminated "reference" sites, and four were contaminated.

Chronic whole sediment toxicity tests were conducted with the estuarine amphipod (*Chaetocorophium lucasi*) and a marine bivalve (*Macomona liliana*), and pore-water toxicity tests were conducted with embryos of the sand dollar (*Fellaster zelandiae*). Although concentrations of organic chemicals and heavy metals were much higher at the sites considered to be contaminated (and were amongst the most contaminated in Auckland), levels of contamination were still relatively low compared to internationally based sediment quality guidelines.

No pronounced difference in the benthic community structure between reference and contaminated sites was found. However, multivariate analysis indicated that natural sediment characteristics (ie total organic carbon, acid-volatile sulfide, ammonium) and factors related to contamination (ie DDT, sediment toxicity) may have been affecting community structure.

In general, this study reinforced the view that determining the biological effects of contaminants in Auckland's muddy estuaries was not straightforward. None of the toxicity tests responded more strongly to sediments or pore waters from contaminated sites than from uncontaminated reference sites, indicating that the tests were not sufficiently sensitive to detect effects in these mildly contaminated environments.

7.7.3 Bioavailability

In order to exercise toxicity, contaminants must be able to be adsorbed or taken up by animals either through their gills (aqueous route) or their food (gut route). The fraction of contaminant in the water or sediment that can be taken up by the animal is called the bioavailable fraction. To experience toxicity through the food route, the animal usually has to accumulate the contaminant, often in its fatty tissue (lipid).

Studies have been carried out which have measured the uptake of two of the most important organic contaminants, PAH and PCB. The accumulation of organic contaminants in two sediment-dwelling shellfish species wedge shells (*Macomona liliana*) and cockles (*Austrovenus stutchburyi*) along the northern shore of Manukau Harbour was measured by Hickey et al. (1995).

Both shellfish species showed marked gradients of reducing contamination from the inner harbour to the entrance. Concentrations of contaminants were similar in the two species, with generally slightly higher levels of PCBs and PAHs in wedge shells. The abundance and condition of wedge shells were found to be reduced at the more contaminated sites in the inner harbour. PAH biota to sediment accumulation factors (BSAFs) were found to be low, averaging approximately 0.01–0.1, indicating that most of the PAH in the sediments was not bioavailable to shellfish. By comparison, BSAFs for PCBs were much higher, about 3–4.

To further understand the bioavailability of PAH in New Zealand (including Auckland) sediments, PAH uptake was measured in cockles and wedge shells by Ahrens et al. (2005). Seventeen sediments from urbanised estuaries and streams around NZ were tested; five came from Auckland (two from the Waitemata Harbour, three from Manukau).

Typically, less than 20 per cent of PAH concentration in sediments was bioavailable to the shellfish. While this only a preliminary study, the results are entirely consistent with many overseas findings (and those from the earlier Manukau Harbour survey), which have found that PAH are frequently tightly bound up in the sediments. The reason for this is that many types of sediment, especially those near urban areas, contain a small proportion of particles of "black carbon" (coal, pitch, char, charcoal, soot, tar), which can be an effective binder for PAH.

The low bioavailability of PAH in estuarine sediments suggests that PAH toxicity to benthic organisms will also be relatively low.

7.7.4 PAH phototoxicity

Auckland estuarine sediments generally have PAH concentrations that are well below guidelines for protecting benthic aquatic life. However, the guidelines do not incorporate the effect of UV radiation, which can markedly increase the toxicity of some PAHs. This increased sensitivity is termed phototoxicity. Toxicity measured in standard indoor laboratory tests (such as those used to derive sediment quality guidelines) may therefore underestimate biological risk to organisms living in PAH-contaminated environments.

Small, transparent organisms, such as larvae or juveniles are potentially particularly UV-sensitive, especially if they spend a significant amount of time exposed to the sun.

The degree of phototoxicity is dependent on the PAH concentration in the animal (dose), and the duration and intensity of the UV light exposure. For example, USEPA have estimated that the toxicity of the PAH fluoranthene in water increases by a factor of about 10 times in the presence of UV light (estimated final chronic value of 0.26 g/L with UV compared with 2.96 g/L without).

The acute phototoxicity to six benthic invertebrates was assessed by Ahrens & Hickey (2002) using fluoranthene, a major PAH component of contaminated sediments. In water-only exposures, toxicity to bivalves and amphipods was 5–10 times greater when exposed to UV light. Gastropods and isopods were less sensitive, and anemones showed no signs of phototoxicity. When sediment was available for the animals to bury into, toxicity was greatly reduced. The study concluded that infaunal species such as juvenile bivalves and amphipods are highly sensitive to UV effects, but not when buried in the sediment. Epifaunal organisms such as snails and anemones can endure high UV doses without major effects, because they have inherent photo protection mechanisms such as absorbent pigmentation, or protective shells.

Under normal circumstances therefore, PAH phototoxicity is probably not a major problem for estuarine species. Low risk from phototoxicity, low bioavailability, and generally low concentrations of PAH around Auckland, indicate risks to aquatic life from PAH contamination are likely to be low.

7.7.5 Toxicity in the marine environment: summary

Toxicity testing has continued to show that stormwater is mildly toxic to marine organisms. However, dilution in the marine receiving environment should rapidly reduce this to non-toxic levels.

Stormwater sediments (eg detention pond sediments) have been found to be toxic, reflecting the high levels of contaminants accumulated on the particulate phase. Stormwater ponds did not remove all toxicity. Dilution of urban stormwater sediment by uncontaminated marine sediments by about 10 times, was predicted to be sufficient to render the urban sediments non-toxic.

PAHs discharged in urban stormwater probably pose a low risk to marine aquatic life in Auckland because of low bioavailability, generally low concentrations, and low risk from phototoxicity.

As stormwater sediments are deposited and mixed in estuaries, contaminant concentrations are reduced to only moderate levels, which are possibly too low for consistent responses from toxicity tests. Toxicity therefore becomes ambiguous. The interaction of physical, biological, and chemical processes in marine sediments affects contaminant bioavailability and hence toxicity, and complicates interpreting relationships between toxicity and contaminant concentrations. The toxicity of the generally mildly contaminated estuarine environment around Auckland therefore remains uncertain.

Information gaps

In addition to the advances in our understanding outlined above, there have been considerable advances in increasing the range and reliability of toxicity test procedures. There have also been advances in understanding the causes of toxicity and the factors affecting bioavailability. It is therefore highly appropriate that toxicity testing is re-examined in the Auckland marine receiving environment. A concise but comprehensive summary of toxicity tests currently used, and an assessment of their reliability, sensitivity, and applicability to measuring impacts of stormwater in the receiving environment should be prepared. For useful application of toxicity testing it will be important that test outcomes can be interpreted in a meaningful way that will advance our scientific understanding of the impacts of stormwater discharges to Auckland's receiving environment, and provide reliable guidance to resource managers.

7.8 Modelling contaminants in estuarine sediments

As described in Section 7.2, contaminant build-up in marine sediments is a major concern for the long-term health of the Auckland's marine environment. The present day concentrations exceed sediment quality guidelines to protect aquatic animals, but have not built up to levels where the toxicity is clearly apparent. The chief concerns are the concentrations that are reached in the future, and how quickly these levels are reached.

How do we study future concentrations? ie test when sediment concentrations will reach levels that are toxic or will affect benthic communities. Extrapolating existing conditions into the future with conceptual and mathematical modelling is the only way to gaze into the crystal ball and play "what if" games with different land use management and stormwater treatment scenarios.

Issues

Levels of contaminants in estuarine sediments will increase over time as a result of continued stormwater inputs. Predicting the rate(s) of increase, and when toxic thresholds might be reached, enables the effects of land management and stormwater treatment options to be assessed and prioritised, without waiting until adverse effects are actually observed (which is far too late!).

State of knowledge in 1995

In 1995, a simple 1-box model had been developed for predicting contaminant build up of Zn, Cu and Pb in the Upper Waitemata Harbour under different development scenarios (Vant et al. 1993). The model predicted that sediment quality guidelines would be exceeded in the near future (20–50 years), and that intervention measures (such as urban stormwater treatment) would slow this build up, but not reverse it.

The modelling approach was applied to the greater Auckland area as part of the Auckland Strategic Planning exercise in the mid-1990s, which predicted that many urban development on sheltered harbours and estuaries would likely lead to the loss of some estuarine benthic organisms, based on exceedance of sediment quality guidelines, within a few decades.

Clearly alarm bells were ringing by 1995 about the future state of Auckland estuaries and there were additional concerns about the efficacy of known methods to address these concerns.

Advances since 1995

Advances came from validating and expanding modelling capabilities:

1. Validation of the existing USC1 model.
2. Application of the model and consideration of the implications.
3. Development and application of improved models (USC2).

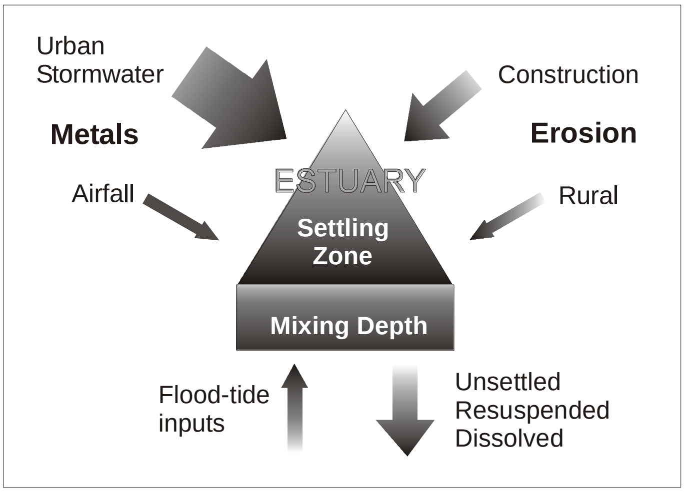
7.7.6 The USC1 model

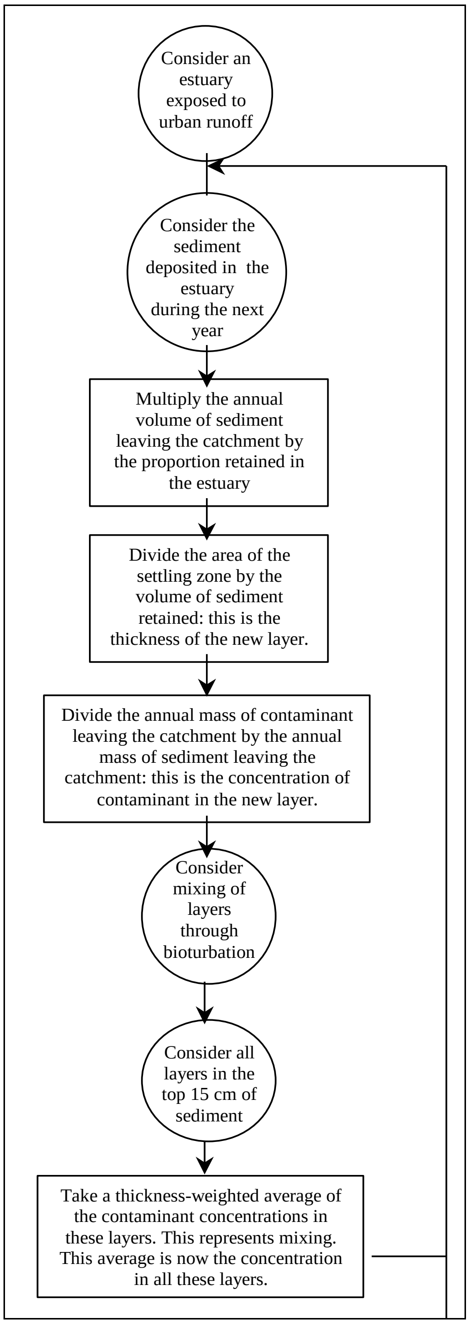
The Urban Stormwater Contaminant "1" (USC1) model a variation of the Auckland Strategic Plan (ASP) model, was designed to predict *average* concentrations of Cu, Pb and Zn in the "settling zone" of estuaries (ARC 1998a, Williamson & Morrisey 2000, Morrisey et al. 2000). The model was developed to address contaminant accumulation in the upper reaches of estuaries where most stormwater is discharged. Figure 93 show schematics of the model, which is described further in ARC 2002a.

The model has been used to predict accumulation of contaminants in estuarine sediments as the result of a range of different urban development scenarios (refer Table 18).

Figure 93

Conceptual underpinning of USC1 model, showing the key processes affecting the build up of contaminants in estuarine sediments.



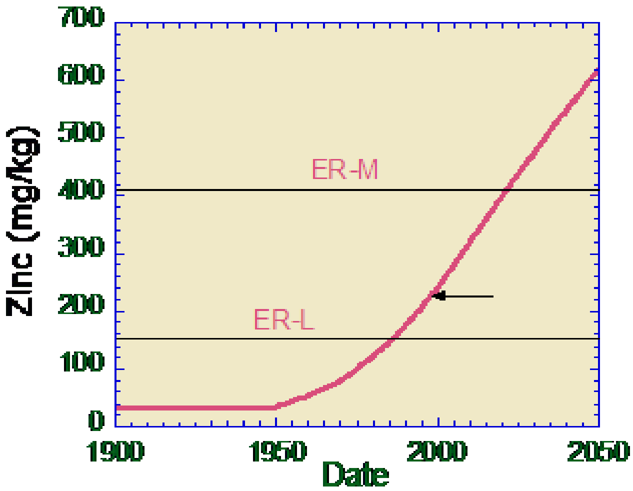


Validation of the USC1 model

By 1995 studies were well underway to test the predictive model, given the long-term implications to Auckland estuaries. This was achieved by formalising the modelling procedure and testing it on two estuaries, Pakuranga Creek and Hellyers Creek, as described in Section 7.2 (ARC 1998a, Williamson & Morrisey 2000).

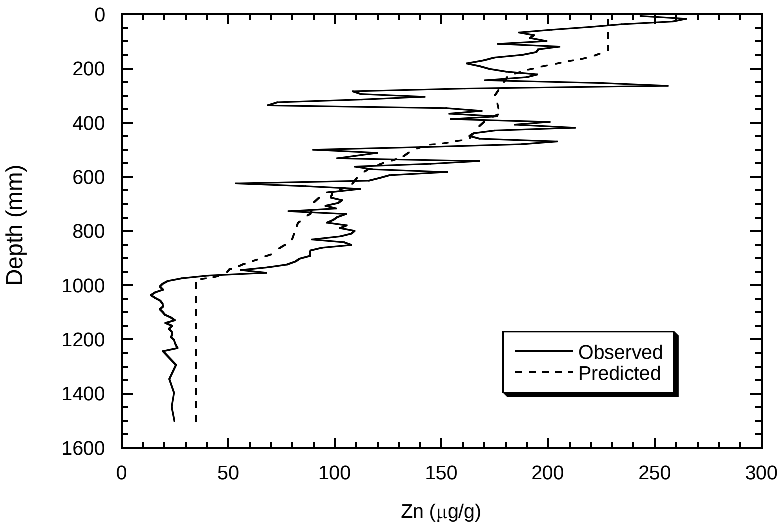
The testing showed that the model was able to predict the average concentration of metals in the Settling Zone, and was therefore a useful management tool for predicting order of magnitude concentrations in sheltered estuaries see Figure 94. The test also confirmed that (providing all the model assumptions were correct), Zn concentration would reach levels of grave concern in a few decades.

Figure 94
Predicted increases in Zn concentrations in the sediments of Pakuranga estuary made with the USC 1 model. The arrow shows the concentration measured in 1995.



When appropriate input parameters were used, the model was able to reproduce with reasonable accuracy a "smoothed" Zn concentration depth profile from a sediment core from Pakuranga estuary (Figure 95).

Figure 95
Comparison of contaminant observations from a sediment core taken from Pakuranga Creek estuary with USC1 model predictions (after observed sedimentation rates are used in the USC model). Taken from ARC (2002a).



Application of USC1 model

The model has been used in a number of locations since its development, including some outside Auckland. These applications have been summarised in Table 18.

Table 18

Application of the USC1 model in predicting heavy metal concentrations from stormwater discharges to sheltered estuaries.

Estuary	Finding	Reference
Lucas, Brighams, Waiarohia	Introduced concept of heavy metal build up in estuarine sediments – predicted SQG will be exceeded within decades under some development scenarios.	Vant et al. 1993
Kaipatiki Creek, Hellyers Creek; Pakuranga, Tamaki	USC1 model tested by predicting present day levels.	ARC 1998a
Okura River	10-20% urbanisation needed for an estuary to exceed SQG by 2150.	ARC, unpublished
Raglan Harbour, Tairua Harbour	Small coastal town unlikely cause estuary sediment contamination unless discharge is to small, enclosed, highly urbanised estuaries.	Williamson 1999
Waiarohia	Development scenario will lead to exceedance of red ERC within about 100 years (depending on treatment).	DSL (2001b)
Lucas, Te Wharau, Upper Hellyers, Kaipatiki, Hellyers, Beachhaven, Soldiers, Island, Kendalls, Chelsea, Little Shoal Bay, Shoal Bay	Scoping study to identify priority estuaries for NSCC and the benefits of management intervention.	DSL 2005
Motions, Kaipatiki, Wairau (Whau)	Investigates the benefit of treatment with stormwater ponds.	ARC 2004o

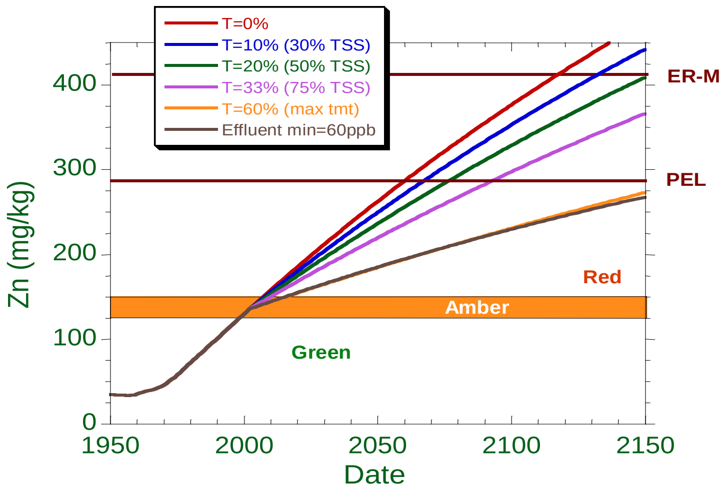
One of the applications of the USC1 model was to examine the benefits of stormwater treatment. This was originally investigated in the Upper Waitemata Urbanisation Impact study (Vant et al. 1993), which raised alarm bells that treatment merely slowed the build up but did not reverse it.

The benefits of treatment were examined more closely in three estuaries of different degrees of contamination – the Kaipatiki arm of Hellyers Creek estuary, Motions Creek, and the Wairau arm of the Whau estuary (ARC 2004o). This exercise also looked closely at the reductions achievable for Zn, Cu and Pb in stormwater ponds, so predictions were realistic. Figure 96 shows an output of the USC1 model, for Kaipatiki Creek, illustrating the relative benefits of different levels of SS removal (0–75 per cent) and the associated Zn removal (T= 0–60 per cent).

These predictions show that treatment reduces the rate of contaminant increase in the estuary sediments, but doesn't stop it – for the Kaipatiki estuary SZ example shown above, severe ecological effects (Zn above the Probable Effects Level - PEL) would be expected to occur within about 60 years with minimal treatment, and about 90 years for a high (but realistically achievable) level of treatment. Only when a very high level of treatment occurs will adverse effects be delayed by significantly longer time frames. This shows the importance of early intervention and effective contaminant source control in minimising receiving environment impacts.

Figure 96

Example of the USC1 model output for Zn in Kaipatiki Creek estuary, showing Zn levels predicted for various treatment scenarios (ARC 2004o).



7.7.7 The USC2 model

The USC1 model is useful for predicting contamination in Settling Zones, but cannot be used in the wider estuary area, such as Outer Zones. The concept of settling zones worked well for tidal creeks and some sheltered embayments, but was not useable in the wider harbour, for example the wider open waters of the Waitemata Harbour. It makes simple assumptions about the amount of contaminant that is trapped in an estuary and where the contaminant is trapped. It is not able to handle redistribution processes such as remobilisation of contaminants from one part of an estuary to another. This deficiency led to the development of the USC2 Model (ARC 2002a, ARC 200p).

The USC2 model utilises sediment transport sub-models nested within 2D and 3D hydrodynamic models. This allows more precise definition of the fate of fine sediments and their associated contaminants during and after storm events, which deliver the sediments and contaminants to the estuary. Exchange between different estuarine environments can occur. The output from this dealt with in much the same way as the USC1 model, building up the concentrations in the sediment profile over time.

The USC2 model incorporates other advances such as two sediment particle sizes, and is able to be coupled to catchments models, such as hydrological and sediment generation models such as GLEAMS.

Application of the USC2 model: Upper Waitemata Harbour contaminant study

The USC2 model was first formally applied to predict the effects of various development scenarios in the Upper Waitemata Harbour catchments on contaminant accumulation in estuarine and harbour sediments. A summary of this project can be found in ARC (2004q).

Spatial patterns of Zn, Cu, PAH, and sediment accumulation throughout the Upper Waitemata Harbour were predicted under two different development scenarios, where each scenario is characterised by a particular land use, sediment controls and stormwater treatment.

Results generated by the model included the times in the future when the concentrations are predicted to breach sediment quality guidelines (a summary is given below in Table 19), sediment deposition rates, the principal sources of sediment/contaminants, and where these sediments/contaminants end up (their fate).

The study showed how rapidly contaminants build up in each sub-estuary, identifies sub-estuaries most at risk, and where management efforts should be directed.

Table 19

Years for Zn to reach 150 mg/kg in the Upper Waitemata Harbour (adapted from ARC 2004q).

Sub-estuary	Land use	Years to red ERC (Zn = 150 mg/kg)		
		No controls	Development scenario #1[a]	Maximum attainable controls
Hellyers	Urban (1960)	10	10	15
Lucas	Urban (1990)	11	12	15
Paremoremo	Rural	>54	>54	>54
Rangitopuni	Rural	>54	>54	>54
Brighams	Future urban	23	29	39
Rarawaru	Future urban	36	46	>54
Waiarohia	Future urban	39	>54	>54
Upper main body	Mixed	17	20	24
Middle main body	Mixed	15	19	21
Lower main body	Mixed	19	24	28

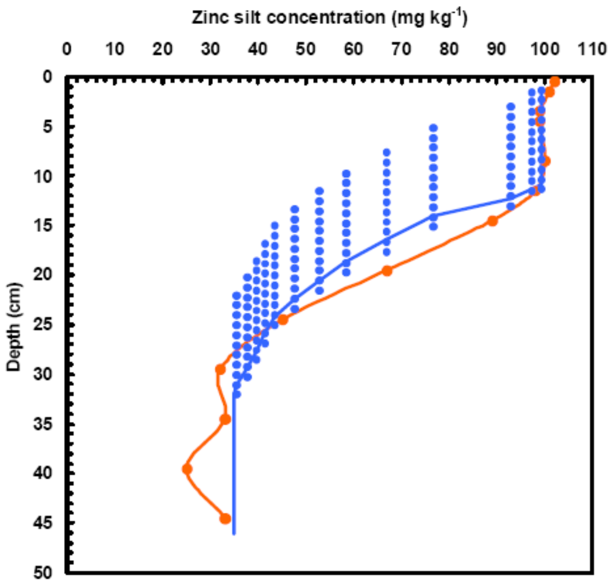
a. This is a realistic scenario based on projected land use changes and sediment controls of 50 per cent of the earthworks having 70 per cent treatment (ARC 2004q).

A number of approaches were taken to model verification. These included comparing predicted and observed annual-average estuarine sedimentation rates, and comparing predicted and observed contaminant-concentration profiles in bed sediments. The latter represented quite a severe test of the model, since concentration profiles in reality develop from the entire suite of interacting processes simulated by the model.

Figure 97 shows good agreement between the predicted and measured sediment core profile for Zn, taken from Lucas Creek estuary (see Section 7.4).

Figure 97

Sediment Zn depth profile in lower Lucas Creek estuary. The red line shows the measured profile. The blue line is that predicted from the model. See ARC (2004p) Appendix F for details of how to interpret the blue points and line.



USC2 model application: Orewa and Weiti estuaries

A simplified version of the USC2 model (or an improved version of the USC1 model) has been applied to the Orewa and Weiti estuaries (Williamson et al. 2005, Green et al. 2005 a and b).

Major findings were that the model did not work well in Orewa estuary and over-predicted contaminant build up, based on present day concentrations. Possible reasons suggested for the differences between predicted and measured values were the effects of estuary processes that were not included in the model and/or the low historical run-off on the sand spit underlying Orewa in early years when there was inefficient storm drainage.

Despite these problems, the modelling exercise was able to show that the rate of build up of contamination in these estuaries is very slow and source control, treatment of stormwater, and other BMPs will result in very low concentrations in the foreseeable future.

7.7.8 Modelling contaminant accumulation in sediments: summary

Modelling, both relatively simple (USC1) and sophisticated (USC2), has enabled the consequences of urban development on contaminant levels in estuarine and harbour sediments to be predicted.

The relatively simple USC1 model is useful for predicting impacts in estuarine settling zones, whereas the USC2 model can be applied to predict effects in more complex settling and transportational environments.

The models have been validated by field measurements of contaminant and sediment accumulation, and therefore offer reliable methods to assess the potential impacts of different land development scenarios and management options.

Modelling highlights the increases in Zn, and to a lesser degree Cu, concentrations in estuarine sediments that occur as a result of urban stormwater discharges. It also demonstrates that treatment alone is often not enough to prevent significant estuarine contamination from occurring in the (relatively) near future, and that, ultimately, the key to managing contaminant impacts in estuaries is source control (particularly for Zn).

Models continue to be improved and applied to even more challenging situations – most recently the modelling of contaminant fate in the Middle Waitemata Harbour, which has just begun and is due to be completed in 2007.

7.8 Impacts of stormwater in the marine environment: summary

7.8.1 Water quality

A large amount of water quality data, ranging in scale from long-term, regional monitoring programmes to detailed studies on localised impacts around stormwater outfalls, has been gathered over the last decade. Stormwater composition has also been well characterised, enabling the impacts on marine water quality to be predicted.

The major identified impact is degraded microbiological water quality, which limits contact recreation at a significant number of sites around Auckland, particularly during and after rainfall. Severe effects are generally short-term (<24 hours) and localised (to within about 100 m of the outfall).

Regional monitoring indicates that urban run-off contributes to a broader-scale reduction in marine water quality in Auckland's harbours and estuaries, in particular increased levels of indicator bacteria. These effects decline markedly in the wider harbour and coastal zones due to dilution by cleaner oceanic waters.

Other potential adverse effects on marine water quality are likely to be minor due to dilution in the receiving waters. Only localised zones next to stormwater discharges should be affected, and then only for relatively short periods during storm events and low tides.

Effects of suspended solids depend on the stormwater source and the nature of the receiving waters. Background levels of suspended solids and turbidity in muddy estuaries (and on beaches during storms) are high, so increases from stormwater inputs may not be noticeable. Stormwater SS concentrations from developed catchments (eg Auckland City) are fairly low, and therefore effects are unlikely to be significant. However, developing catchments, with higher SS loads, may produce marked impacts in receiving waters. There are no known data on these potential impacts to quantitatively assess the scale or significance of this issue.

Cumulative effects of stormwater-sourced nutrients are unclear, but are probably less than from rural streams and run-off.

7.8.1.1 Sediment contamination

A large regional database on sediment contamination has been acquired since 1995, which clearly shows that urban stormwater is contaminating Auckland's urban estuaries with heavy metals and, at lower levels, a range of POPs, including PAHs, DDTs and PCBs.

Current information indicates that over 50 per cent of regional monitoring sites are contaminated to the point where aquatic life may be beginning to be adversely affected. Worst affected sites are those in muddy estuaries receiving run-off from older, fully urbanised catchments.

Zinc is the contaminant of most concern at present. It is increasing in concentration at most sites, and has the greatest proportion of ERC exceedances.

Persistent organic pollutants, such as PAH, OCPs, and PCBs are widely found, but concentrations are low, below ARC ERC at the majority of sites. Organochlorines are unlikely to increase much, if at all, in future because they are no longer legally used. However, contaminated catchment soils are a stored source of these contaminants, which can be released to receiving environments during catchment development. The potential impacts from this release are still being investigated, although initial results suggest that this is not a widespread serious threat to the marine environment.

7.8.1.2 Contamination of shellfish and fish

Although concentrations of chemical contaminants such as organochlorine pesticides and PCBs are very low in marine receiving waters, shellfish and fish from urbanised estuaries show clearly elevated concentrations. Contaminant levels are highest at sites receiving run-off from the older intensively urbanised areas, in particular Mangere Inlet and the inner Tamaki Estuary. Waitemata Harbour sites have intermediate levels of contamination, and the open east coast waters are least contaminated.

Limited studies of organic contaminants in flounder in the Manukau Harbour show spatial patterns of contamination consistent with the degree of nearby urban development. Contaminant concentrations are moderate to low compared with USA data. There is evidence to suggest that fish health is being affected at the most contaminated site, although more study is required to be definitive about what the ecological implications of contamination really are.

7.8.2 Trends over time

Is contamination getting better or worse over time? Water and sediment quality monitoring, sediment coring studies, shellfish monitoring, and contaminant modelling have been used to assess changes in receiving environment quality over time.

Water quality monitoring has detected no significant changes in marine waters related to stormwater discharges. This is at least partly due to the broad-scale nature of the

ARC regional monitoring programme, which was not designed to measure impacts of episodic, localised discharges. Relatively high variability in water quality data also hampers trend detection. Bathing beach water quality data are sensitive to the effects of rainfall on or near to monitoring dates, and therefore assessing trends from these data need to be done with care.

Estuarine sediment monitoring has provided robust data on trends in key stormwater-derived contaminants. ARC monitoring over the past five years shows significant increases in Zn concentrations at most urban sites, smaller increases in Cu, and uncertain trends for Pb and PAH.

Sediment cores and modelling tell the same story – Zn has increased markedly as a result of urbanization, Cu shows smaller increases, and Pb has recently decreased following removal from petrol. Organochlorine pesticides, mainly DDTs, are elevated in urban estuaries as a result of past horticultural practices, and more recently from catchment urbanisation. The recent increases in organochlorines are probably associated with catchment soil losses, and therefore levels would be expected to decrease as catchments mature.

Shellfish monitoring suggests that little change in chemical contaminant levels in estuarine waters has occurred since about 1987, except where a known contaminant source (chlordane) has been removed. Quite large increases and decreases occur over shorter time periods, reflecting the combined effects of biological variability, changes in catchment run-off and, in the case of the Manukau Harbour, contaminated site cleanup and decommissioning of the old sewage treatment ponds.

Overall, sediment monitoring shows the clearest trends, and of these, the most significant is increasing levels of Zn from urbanised catchments. Reducing Zn run-off at source is required to slow or halt this trend and prevent irreversible contamination of the marine environment.

7.8.3 Toxicity to marine organisms

Toxicity testing shows that urban stormwater is mildly toxic to marine organisms. However, dilution in the marine receiving environment should rapidly reduce this to non-toxic levels.

Stormwater sediments (eg detention pond sediments) are more toxic, reflecting the high levels of contaminants accumulated on the particulate phase. Detention ponds do not remove all toxicity, so dilution by uncontaminated marine sediments (by about 10 times), is required to render urban-derived sediments non-toxic.

As stormwater sediments are deposited and mixed in estuaries, contaminant levels drop, and toxicity becomes ambiguous. The interaction of physical, biological, and chemical processes in marine sediments affects contaminant bioavailability and hence toxicity, and complicates interpreting relationships between toxicity and contaminant concentrations.

The toxicity of the generally mildly contaminated estuarine environment around Auckland therefore remains uncertain.

7.8.4 Effects on marine ecology

While the effects of stormwater on sediment quality are quite clear, the effects on benthic ecological health are not so clear-cut. This is due to the combined effects of a number of factors including:

- relatively low levels of chemical contamination (in most places) compared with known biological effect thresholds;
- multiple contaminants – including known species such as metals, organics, organic matter, and fine sediment, and possibly also chemicals not yet analysed in Auckland's marine sediments;
- natural variations in sediment characteristics, and;
- spatial and temporal variation in biological community composition.

It has required the relatively recent development of sophisticated multi-variate statistical methods, coupled with consistent monitoring protocols, to tease out the effects and present them in an easily understandable form – the Healthy Benthic Community Model.

The model is the strongest evidence that estuarine benthic ecology is adversely affected in urbanised estuaries, with a strong correlation between estuarine health and increase in contaminant levels.

A huge amount of work has been undertaken in the last several years to gain a better understanding of the effects of sedimentation on benthic aquatic life. This work has shown that fine sediment has a wide range of impacts related to the amounts, frequency, and duration of sedimentation. Generally, increasing sediment accumulation rates cause a decrease in benthic community diversity, resulting in a convergence of all estuarine types to a single, low diversity community of disturbance-tolerant species with similar functional roles.

Overall, the work to date shows that even though Auckland's urban estuaries are generally only mildly contaminated (by international standards), ecological health is being compromised, probably by the combination of contamination and sedimentation resulting from stormwater run-off. Contamination is a greater issue in more mature developed catchments where sediment exports are lower and contaminant exports are higher, while sedimentation represents a greater risk in developing catchments where contaminant inputs are lower, but sediment inputs from pasture soils are augmented by losses from earthworks and exposed soils. Clearly both contaminant and sediment controls are required to minimise these risks.

Knowledge gaps

Gaps in our current knowledge include:

- Human health risks associated with microbial contamination of urban stormwater. Do the currently monitored indicator bacteria provide a reliable assessment?

- Ecological consequences of contaminated fine sediments in sandy outer zones[38] .
- Sub-lethal, long-term, ecological consequences of stormwater-derived contaminants and sediment.
- Sedimentation rates in estuaries receiving run-off from mature urban catchments. Are they high enough to affect ecology?
- Concentrations and potential biological effects of "emerging contaminants" including plasticisers (phthalate esters) and fire retardants (DBPEs).
- Effects of low levels of bioaccumulative contaminants such as PCBs and Hg on marine food webs.

[38] Significant advances were made in this area in the year following this review – ARC (2006c).

8 Summary and Conclusions

As can be seen from the length of this review, a massive amount of work was carried out between 1995 and 2005 on the impacts of stormwater in Auckland's aquatic receiving environments.

Significant advances have been made in a number of areas, which together have greatly improved our understanding of the history, spatial extent, nature, causes, ecological effects, and likely future impacts of stormwater-derived chemical contamination.

Important advances include:

- Development of classification methods for freshwater and marine areas that provide a framework for objectively assessing and managing stormwater effects.
- Acquisition of a large database on chemical contaminant concentrations in waters, sediments (particularly marine sediments), and biota which have been used to assess the spatial extent, temporal trends, and potential ecological impacts of stormwater-derived contamination.
- Development of systematic monitoring methods (a "blueprint") for measuring contaminants and benthic ecology in marine sediments, improving the reliability of monitoring data and providing a sound basis for testing models that predict the ecological consequences of stormwater contamination.
- Improved understanding of sediment transport and accumulation processes in estuaries, which have helped to decipher the history of sediment-related impacts and contributed to the development of models to predict sediment and contaminant accumulation in developing areas.
- Detailed studies of contaminant sources in urban catchments, so that source-control measures can be accurately targeted. Refining the data gathered to date remains a challenge to improve the reliability of existing estimates.
- Improved understanding of the effects of land use on freshwater receiving environments, and initial investigations to link land use pressure measures with effects in receiving waters.
- Improved understanding of estuarine ecosystem functioning, and the consequences of fine sediment deposition (an important urban contaminant).
- Development of the "Healthy Benthic Community Model" to assess and predict the ecological health of Auckland's estuaries in relation to key stormwater contaminants. This will improve the reliability of sediment quality guidelines, providing greater certainty for resource management.

Since 2005, work has continued in these areas, further improving our knowledge of stormwater impacts in aquatic receiving environments, enabling the development of more reliable tools to predict future impacts of urbanisation and test the effectiveness of resource management scenarios.

The knowledge acquired in the decade 1995–2005 provides a solid foundation for future scientific research and resource management activities, which together should provide the tools required to further improve stormwater management and reduce future impacts of urbanisation.

The information reviewed to date clearly shows that urban stormwater discharges can have serious long-term impacts on the health of receiving waters. Continued efforts are therefore required to prevent or minimise on-going effects and, where possible, restore impacted environments.

9 Acknowledgements

This review summarises a large body of research undertaken by many people. The authors wish to acknowledge the efforts of all these researchers, and the organisations that have supported their work. In particular, the ARC and city councils have been instrumental in initiating, facilitating, funding, and reporting much of the work described in this review.

As evidenced by the large number of reports referenced in the review, NIWA has been a key research provider, with studies supported by regional and local councils and by public good science funding. Their work has provided many of the major advances in our understanding of stormwater impacts.

Thanks to all the agencies and individuals that have made the research possible and the findings available for us (and the wider public) to use. We hope this review does justice to their work.

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