The Impacts of Stormwater in Auckland's Aquatic Receiving Environment

A Review of Information 2005-2008

September TR 2009/111

Auckland Regional Council
Technical Report No.111 September 2009
ISSN 1179-0504 (Print)
ISSN 1179-0512 (Online)
ISBN 978-1-877540-26-4

Technical Report, first edition.

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Date: 12 January 2010	Date: 21 January 2010

Recommended Citation:

WILLIAMSON, B., MILLS, G., 2009. The Impacts of Stormwater in Auckland's Aquatic Receiving Environment. A Review of Information 2005-2008. Prepared by Diffuse Sources Ltd for Auckland Regional Council. Auckland Regional Council Technical Report 2009/111.

The Impacts of Stormwater in Auckland's Aquatic Receiving Environment. A Review of Information 2005-2008

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1 Executive Summary

In 1995, the Auckland Regional Council (ARC) published a report TP 53 entitled **"The Environmental Impacts of Urban Stormwater Run-off"** (Macaskill et al. 1995). 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. The ARC therefore commissioned an updated review, covering the decade 1995–2005 entitled "**The Environmental Impacts of Urban Stormwater Run-off. A Review of Information 1995 to 2005"** (ARC 2008 – referred in the following as the 2005 Review).

The Auckland Regional Council recognised that there were a number of key studies underway in this field during 2005, which were not included in the 2005 Review. This present report is a review of this ongoing and new work from 2005 to 2008, with the aim to update the review of 1995–2005 information.

This present update should be read along with the 2005 Review, which has a more comprehensive description of the effects of stormwater on the environment. The 1995–2005 and 2005–2008 reviews critically examine and summarise information produced on the impacts of stormwater on the aquatic environment in Auckland. They draw together the large body of published information on stormwater impacts into two overview reports that can be used as a "one-stop" source of information and as a bibliographic reference for those requiring more detailed information.

The following is a summary of the key advances made during 2005–2008 in understanding the effects of stormwater in receiving environments.

1.1 Sources of chemical contaminants

The sources of the contaminants copper (Cu), polynuclear aromatic hydrocarbons (PAH), lead (Pb), Total Suspended solids (TSS) and zinc (Zn) in Auckland's stormwater run-off are better understood, although further work is needed to quantify some potential sources. Some of this understanding has been encapsulated in the Contaminant Load Model (CLM), which now has the potential to be a primary driver for stormwater management. The model estimates TSS, Cu and Zn loads from roofs, roads, motorways, pavements, parks, construction sites, stream channels, and rural areas. The model can be used for hindcasting loads and is available for public use. The derivative model USC3-CLM[1] is more sophisticated and dynamic, because it incorporates the projected future changes in population, infilling, traffic density and building materials, thus providing realistic predictions of future loads.

A major review of a new "class" of chemical contaminants – the Chemicals of Potential Environmental Concern (CPEC) – has provided an excellent primary source

[1] USC3 or USC-3 = Urban Stormwater Contaminant model version 3.

document for further work in Auckland. As well as a comprehensive list of chemicals, sources, and occurrence in overseas studies, the review provides a relative risk assessment of these CPEC in urban aquatic receiving environments.

1.2 Stormwater effects in the freshwater environment

Water quality of urban streams appears to be improving, and this may be partly due to improvements in stormwater. The ARC Regional Stream Monitoring Programme has shown decreases in TSS, nitrate and faecal coliform levels over the past 20 years. These, in particular TSS and faecal coliforms, have been identified as important factors limiting the potential use of urban streams and their receiving waters, and hence their improvement is a significant step forward. The programme could benefit from pressure-state-response (PSR) analysis to better link the improvement to changes in catchment conditions and hence direct management towards actions that will continue the trends.

The ecological functions of regional streams (including urban streams) are now able to be described and measured through the Stream Environmental Valuation (SEV) methods. This provides a key tool for effective management of urban streams, which, if implemented, should result in real improvements in stream ecology and address many of the impacts of urbanisation.

Toxicological understanding has been advanced on the effects of short-term, high concentrations of TSS, which can occur during major storms in urban streams draining catchments which are undergoing development. The work reported to date has found that New Zealand stream biota are quite resilient in the most part to TSS, so that infrequent, very high TSS levels are not especially toxic. A new initiative studying long-term toxicity across several generations of stream biota commenced during 2008.

A number of studies have measured the reduction in low flows brought about by urbanisation. This effect can be very significant in some catchments, but not in others, and further work is needed if a regional perspective is to be gained.

Overall, from 1995 to 2008, there was enormous progress in understanding stormwater effects in urban streams, to the point that many of the tools for measuring stream health and effective management are now available. Most of the advances address the effects of flow, habitat modification and changes in ecological functioning. Of these, the SEV methodology is the latest, and potentially most powerful, development.

1.3 Impacts on marine water quality

The Coastal Water Quality Programme (27 sites) monitors contaminants associated with erosion, nutrients and biological wastes (organic material and faecal contaminants) in the water column. While not directly monitoring the effect of stormwater, the programme does provide consistent, long-term information on the environmental quality of Auckland's marine environment, and the effect of land use.

Marine water quality is poorest at inner harbour or estuary sites subject to freshwater inflows from urban (and rural) areas. The improvements in stream quality described above (reductions in TSS, nitrate and faecal coliforms) for freshwater have been linked to slow and gradual improvements in marine quality at some sites. Some of the improvement may be due to conversion of rural land (high indicator bacteria, nutrients) to urban land (lower indicator bacteria, nutrients), and some to the reduction in wastewater overflows. Most important, despite the large increase in urbanisation since monitoring programs begun in 1987, the general water quality in the marine environment has not deteriorated and may even have improved slightly. As with the Freshwater Water Quality Programme, the cause and effect linkages are not well quantified and would benefit with PSR modelling.

New work has challenged some paradigms on stormwater in Auckland's marine environment. Zinc concentrations in water in the Whau Estuary (one of Auckland's most impacted estuaries) can be relatively high during low flow conditions and exceed water quality guidelines in saline waters. This is at odds with the previous views, which proposed that Zn (and Cu) concentrations would be low and below water quality guidelines in the water column in the wider marine receiving environment (see the 2005 Review). This proposition was based on dilution calculations and actual measurement of Zn and Cu levels in Mangere Inlet. The new work proposes that Zn desorbs from particulate matter. High Zn concentrations were also measured in pore waters. The implications of these findings and propositions could be important, and therefore worth listing:

1. They point to the significance of the dissolved phase for Zn fate, effects and transportation in the marine environment, while previous work has emphasized the particulate phase.
2. As described above, Zn concentrations are high enough in the saline water column to exceed water quality guidelines or criteria, and hence potentially cause adverse effects on estuarine biota.
3. Zn concentrations can also exceed water quality guidelines in pore water; which provides a mechanism for toxicity in sediments.
4. Mobilisation of Zn from sediments into the water column, along with dispersal in tidal flushing could be a significant redistribution mechanism to other parts of the marine environment, including losses to the Hauraki Gulf and the Tasman Sea.

However, the Whau Estuary work is only one study, so further investigations are warranted, especially because the implications of the initial work are so far reaching.

Overall, advances in our understanding on trends in general marine water have been made, with general water quality improving despite the continued expansion of urban land use. There is new evidence for the greater importance of dissolved phase for the fate, effects and transport of Zn in marine receiving waters.

1.4 Chemical contamination of marine sediments

There is much more comprehensive information on heavy metal concentrations in marine sediments from existing programmes (State of the Environment (SoE), Regional Discharges Project (RDP)) and from new studies. The understanding of spatial patterns has not changed significantly from those described in the 2005 Review, except that there is much better detail in Outer Zones and in areas which are undergoing urbanisation (Upper Waitemata Harbour, South East Manukau Harbour).

The continuing assessment of trends in heavy metal concentrations has uncovered some significant changes from previous findings. The previously described linear trends have not been maintained and trend profiles are more complex. Nevertheless, substantial increases in Zn (and to a lesser degree Cu) have definitely occurred in many urbanised estuaries over the past decade. New developments in contaminant load models and receiving environment models (eg, CLM and USC3) may be useful in gaining a better understanding of the monitoring data, and will provide insight into future trends.

A very recent survey of Chemicals of Potential Environmental Concern (CPEC) in marine sediments is an important preliminary step in developing an understanding of the importance of these chemicals in the marine environment. It is too early to assess the significance of these initial data.

Overall, the picture of the distribution of contaminants in the marine environment that emerged from studies described in the 2005 Review has been "filled in" with more detail. The view that sediment quality is largely determined by inputs of sediment and contaminants and by the hydrodynamic energy in the receiving water is generally supported. Major advances have been made on understanding other factors that control levels of contamination (besides proximity to sources), such as sediment chemistry and texture (particle size) and hydrodynamic forces (tides, currents, waves – see "Predictions" below) and sediment supply (also summarised below in "Predictions"). The simple picture of linear increases in Zn and Cu and decreases in Pb obtained from early trend monitoring (as described in the 2005 Review) has not prevailed, and the recent data show that changes over time are more variable than this. Reasons are, as yet, unknown and are currently being investigated.

1.5 History of urban stormwater impacts

Various studies have produced a great number of results on contaminant profiles in sediments and/or sedimentation and hence the history on sediment contamination and deposition. Studies have published information from cores collected from the Rangitopuni (2 cores), Lucas (1), Central Waitemata (21), Tamaki (12), Mangere Inlet (6), SE Manukau (6), and region-wide (12). Many of these cores have been dated or have other relevant information on stormwater impacts. There is a wealth of information embodied within these studies that, if integrated and rationalised with other sediment profile findings described in the 2005 Review, would greatly enhance our understanding of land use impacts.

The classical picture of heavy metal accumulation close to the sediment surface is generally supported, with the deepest and greatest build up in older, more sheltered urban areas. This accumulation is smaller and close to the surface in newer urban areas. Deposition rates are available for a much greater area than before and in greater detail in some areas. A study specifically designed to determine whether some estuaries have higher background concentrations from volcanic rocks did not find supporting evidence. However, it and another detailed study on heavy metal profiles, did find elevated concentrations as the base of cores at some sites. In some cases, this was probably due to cores not being deep enough to penetrate pre-urban sediment. At other sites, it may be due to cores penetrating underlying rock. These, and other studies, would bear further appraisal of these baseline concentrations, although comparisons are challenging because methodologies varied between studies.

Studies of fossil foraminifera have shown that the major driver for these communities may have been an increase in freshwater run-off from development and was unlikely to be heavy metal pollution. The studies have also postulated that increased freshwater run-off may have also affected benthic macroinvertebrate communities. The understanding of stormwater effects would benefit from further assessment of these studies.

Overall, the information on history of stormwater has been greatly enriched by many studies and conclusions, and it may be timely to integrate, synthesize, and hence clarify a regional picture from this wealth of information.

1.6 Predicting contaminant fate in the marine environment

Two new suites of models have been developed to predict the fate of contaminants (heavy metals, TSS and bacteria) in the marine environment: (1) CREA[2] and (2) USC-3. The models take information on stormwater inputs and through hydrodynamic and particle tracking models, determine the fate of stormwater particulate solids including attached Zn and Cu. The CREA model also tracks the fate of indicator bacteria.

In the Central Waitemata and South East Manukau Harbours, the USC-3 models predict that contaminants increase in concentration only slowly in the sediments and do not reach the high concentrations predicted (and found) in tidal creeks and embayments. Sediment supply has a major influence on the fate and concentrations of Zn and Cu in the marine sediments through dilution (SE Manukau) and destabilising sediment beds. Overall, contaminants are widely dispersed in harbours but can be concentrated within parts of the harbour by hydrodynamic conditions (eg, Shoal Bay receives much of its contamination from catchments west of the Auckland Harbour Bridge).

The USC-3 model is able to take into account inputs from all catchments, and the exchange of contaminants between different areas within the harbours. It shows the links between sources and fate, so that contamination can be traced back to its discharge (and catchment). Therefore, the fate of all contaminants from a specific

[2] CREA = Coastal Receiving Environment Assessment model.

catchment discharge can be tracked. The models can assess the effect of management intervention, such as land use change, source control and stormwater treatment. Overall, the USC-3 and CREA models are good PSR models, linking Pressure (catchment loads) – State (concentrations in sediments) – Response (management intervention).

The CREA model predictions of future concentrations were made by assuming present-day catchment loads persist into the future. They do not take into account the likely changes in loads associated with changes in building materials and traffic densities, population growth and density, and infilling, that have been recently encapsulated in the USC3-CLM catchment models (see above "Sources"). Further work is thus needed on the CREA model using the more accurate and sophisticated USC3-CLM model for stormwater inputs. CREA predictions of bacterial contamination of bathing beaches provides an additional tool to investigate stormwater impacts and management because it provides the Pressure (stormflows) – State (Bathing beach quality) linkages to explain and investigate exceedance of recreational water quality (eg, Ministry for the Environment (MfE)) guidelines.

1.7 Impacts on the marine benthic ecology

The **Benthic Health Model (BHM)** measures the community structure of animals in the sediments (benthic ecology) as an indicator of the effect of contaminants and stormwater on ecological communities. The model ranks the community structure of sites across the Auckland region into five groups. While it is based on correlations with Cu, Pb and Zn contamination, it does not necessarily establish that these are the causative agents. The degree of contamination reflected by the metals may reflect the levels of other contaminants (which may be the responsible agents) or possibly some other correlated environmental parameter (not necessarily a contaminant) not covered by the model. Therefore, the causative agents "driving" the observed ecological differences remain uncertain. This is being addressed to a limited extent in other research that is described in this review. This research suggests that the cause is unlikely to be a single stressor (Section 9 – Ecology), and unlikely to be PAH (Section 10 – Toxicity) but could include Zn (Section 5 – Marine Water Quality).

At present the BHM shows very strong spatial trends but no trends over time. Recently (2008), the first steps have been undertaken to understanding the implication of the pollution ranking to the wider ecosystem. The magnitude of the differences between the five pollution groups and the changes in the types of animals has been described. Many large, relatively rare, species disappear when the degree of pollution increases. While relatively rare, some of these animals provide important functions to the ecosystem.

The many regional monitoring programmes outside urban areas are supplying valuable information on long-term and short-term cycles in biotic communities, as well describing the effects of sediment texture. Regional programmes that monitor urbanised estuaries and harbours have not yet distinguished time trends due to urban stormwater effects. Future work could integrate the BHM findings with other regional programs, some of which ascribe variation in benthic communities and animals in terms of sediment texture (particle size).

There are robust research programmes on understanding benthic community health and structure that are being carried out under Public Good Science Funding. These programmes are expected to provide scientific information to help interpret regional monitoring programmes.

Overall, the BHM is one of the most robust sources of evidence for stormwater effects in the marine environment. Its continual improvement, testing and interpretation should have a high priority. Benthic community monitoring provides a key indicator of environmental health, and the regional programmes, as well as Public Good research, are providing a significant information base from which to interpret stormwater effects.

1.8 Toxicity of urban stormwater in the marine environment

A state-of-the-science review of PAH sources, distribution and effects in Auckland's marine and freshwaters has been prepared. PAHs in the receiving environment are tightly bound to particulate organic matter and only a small fraction (typically less than 5 per cent) can be extracted under conditions that mimic bioavailability to marine biota. On the basis of this and other evidence, the authors concluded that PAH concentrations in marine sediments are unlikely to be having adverse effects on marine biota at present and are unlikely to in the near future.

Laboratory-based toxicity tests were used to improve the understanding of the potential ecological effects of stormwater-contaminated sediments. Testing of sediments from ten estuarine RDP sites, spanning a wide range of contamination levels, showed only a weak toxicological response. Thus contaminants in Auckland sediments are, at most, only weakly toxic under the conditions of the toxicity test procedures. In contrast, the benthic ecology surveys described earlier have shown a wide gradient of ecological changes associated with these sites, presumably reflecting a long-term, integrative response to all environmental perturbations. Combining toxicity test results with other measures of contamination–bioaccumulation of metals in oysters collected at the sites, and a simple measure of biotic diversity/abundance – in a weight of evidence (WoE) approach – gave a clearer picture of effects, which was somewhat comparable with the range shown by the BHM. As found in previous toxicological studies of Auckland's marine sediments, these results indicate that current laboratory toxicity tests alone are unlikely to provide a clear measure of potential ecological effects of stormwater. Combinations of tests, however, may provide useful information.

Studies on cockles have demonstrated that their health, abundance and size distribution is adversely affected in urban estuaries, confirming their presumed susceptibility to urban stormwater. These comprehensive studies provide considerable detail on the nature of the effects on a key species.

Overall, these studies have added significantly to our understanding of stormwater toxicity in marine environments. Sediment toxicity testing showed the advantages and limitations of the latest methodology, providing better information on the most appropriate methods to be selected in future to address specific questions. The review of PAH brought together most of the information currently available to obtain a comprehensive assessment of risks to marine ecology. It would be timely to repeat this type of exercise for other major contaminants (eg, Zn, Pb, Cu, DDT, PCB). The cockle studies provide much greater depth of understanding of the effects of urbanisation on this important species. The BHM (see Benthic Ecology) remains the most powerful tool measuring the effect of stormwater but cannot identify the direct causal links between stressors and benthic animals. These links may be best provided by controlled toxicity test with specific potential stressors such as Zn.

1.9 The key advance in 2005-2008

All the work reported in this review has made important contributions to understanding stormwater effects. However, there are a number of key studies that have a major impact on scientific understanding, management implications, and represent a significant advance on the state of knowledge from the 2005 Review:

- The CLM models for stormwater loads of TSS, Zn and Cu.
- The USC-3 model which predicts the fate of contaminants in Auckland harbours.
- The Stream Ecosystem Valuation methodology, which measures stream ecological functions and provides simple means for guiding the effective remediation of urban stream health.
- The improvement and further validation of the Benthic Health Model.

These studies provide a powerful set of tools for improved stormwater effects assessment and management. However all models and methods would benefit from further refinement.

The development of these tools to the point where they can realistically be used to reduce future effects and aid the restoration and rehabilitation of already affected areas (in particular streams) illustrates a maturing of our understanding of stormwater and its effects, and how these effects could be managed. The initial scoping studies conducted in the mid-late 1990s helped identify and define the nature and extent of the stormwater problem. The next 10 years (as summarised in the 2005 Review) provided additional detail and scientific rigour to this information base and also the creation of management tools (eg, models, classification systems etc.). The most recent work, as reviewed here from 2005–2008, shows that the next stage has built on all this earlier work, and the most significant advances have been in building more sophisticated, but still useful and robust, management tools.

A key future challenge is to continue this progression by using the available tools to identify what can realistically be done to reduce stormwater effects and to implement the most effective measures identified. Refined monitoring (for example using the PSR framework) to link management interventions with environmental responses, must also continue to ensure the predicted outcomes are realised, and to track changes (hopefully improvements) in the state of the Auckland's aquatic receiving environments.

2 Introduction

In 1995, the Auckland Regional Council (ARC) published a report entitled **The Environmental Impacts of Urban Stormwater Run-off** (Macaskill et al. 1995). 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 entitled **The Environmental Impacts of Urban Stormwater Run-off. A Review of Information 1995 to 2005** (ARC 2008).

The Auckland Regional Council recognised that there were a number of key studies underway in this field during 2005, which has not been included in the 2005 Review. This present report is a review of this ongoing and new work from 2005 to 2008, with the aim to update the review of 1995–2005 information. It should be read along with the 2005 Review, which has a more comprehensive picture of impacts.

The 2005 Review and this update critically examine and summarise the impacts of stormwater. They draw together the body of published information on stormwater impacts into two overview reports that can be used as a "one-stop" source of information and as a bibliographic reference for reader requiring more detailed information.

This 2008 Update addresses advances under the following categories, which are similar to those used in the 2005 Review:

- Sources of Chemical Contaminants – Section 3
- Impacts on the Freshwater Environment – Section 4
- Impacts on Marine Water Quality – Section 5
- Chemical Contamination of Marine Sediments – Section 6
- History of Urban Stormwater Impacts – Section 7
- Predicting Contaminant Fate in the Marine Environment – Section 8
- Impacts on Marine Benthic Ecology – Section 9
- Toxicity in the Marine Environment – Section 10.

3 Sources of Chemicals Contaminants

Sources of the contaminants Cu, PAH, Pb, TSS and Zn in Auckland's stormwater run-off are better understood, although further work is needed to quantify some potential sources. Some of this understanding has been encapsulated in the Contaminant Load Model (CLM), which now has the potential to be a primary driver for stormwater management. The model estimates TSS, Cu and Zn loads from roofs, roads, motorways, pavements, parks, construction sites, stream channels, and rural areas. The model can be used for hindcasting loads and is available for public use. The derivative model USC3-CLM[3] is more sophisticated and dynamic, because it incorporates projected future changes in population, infilling, traffic density and building materials, thus providing more realistic predictions of future loads. Improvements have also made in estimating rural sediment loads, which are important sources of sediment and natural metals and are diluents for urban metal loads.

A major review of a new "class" of chemical contaminants – the Chemicals of Potential Environmental Concern (CPEC) – has provided an excellent primary source document for further work in Auckland. As well as a comprehensive list of chemicals, sources, and occurrence in overseas studies, the review provides a relative risk assessment of these CPEC in the urban aquatic receiving environments.

3.1 What was known in 2005

Up to 2005, there had been a considerable amount of effort directed at determining the key sources of contaminants in urban stormwater in Auckland. This was because contaminant reduction or elimination at source ("source control") was perceived to be the most effective method for reducing contaminant loads and impacts on downstream receiving environments. Once sources were identified and quantified, the options for, and the benefits of source control measures could be objectively predicted and prioritised.

The key sources studied and characterised to 2005 were:

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

Attempts to quantify their contributions to catchment loads had been made, using a combination of detailed stormwater monitoring and predictive modelling. In matching

[3] USC3 or USC-3 = Urban Stormwater Contaminant model version 3.

sources of copper, lead and zinc with their loads measured in stormwater, there appeared to be large deficiencies in the source inventory, with relatively large unknown sources of Cu and Pb (Timperley et al. 2005). There were either additional unknown sources, or the contribution of known sources to stormwater had been underestimated.

3.2 Outline of new work 2005-2008

New work addressed the source inventory problem through a reassessment of sources in three studies: identifying sources of Cu, Pb and Zn in the urban landscape (Section 3.3); through improved methods for estimating loads from urban sources with the Contaminant Load Model (Section 3.4); and from estimating loads of sediment and metal from rural sources with the GLEAMS model (Section 3.5).

A comprehensive review of PAH sources, fate and effects has been published which has shown the relative importance of historical coal tar residues, modern road run-off, petrol station and atmospheric inputs (Section 3.6).

One area not covered in the 2005 Review was sources of chemicals from industrial and commercial operations. This has been briefly addressed in this review ("Industrial Sources") through reference to findings from the ARC's Industrial Pollution Prevention Programme (IP3) (Section 3.7).

A new initiative addressed was "emerging chemicals of concern", which was identified in the 2005 Review (ARC 2008) as a major information gap. These represent a whole new spectrum of chemical sources of potential contamination (Section 3.8).

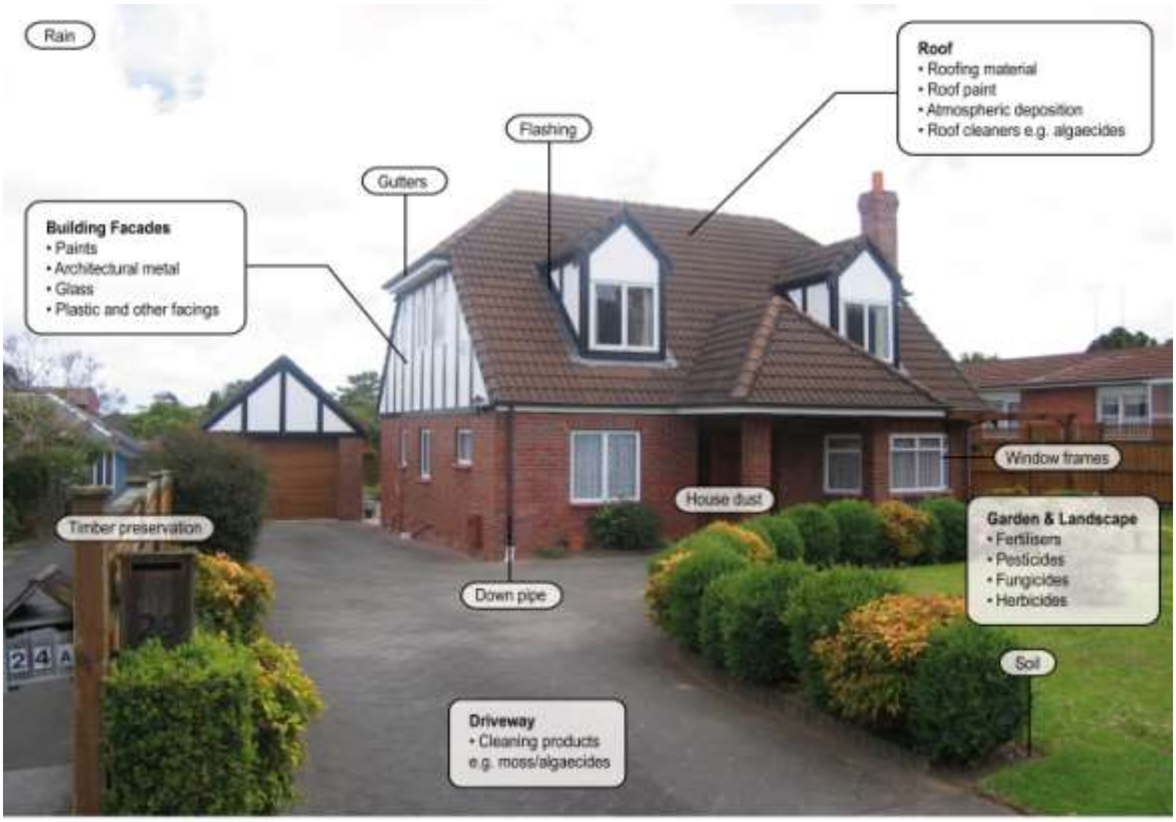
Another area that was not covered in the 2005 Review was stormwater quality itself. It was outside the scope of the review because it was the subject of another major review (Griffiths & Timperley 2005). Since the Griffiths & Timperley (2005) review, significant additional stormwater quality information has been obtained in the Meola and Motions catchments (Moores et al. 2005) and in the Henderson area (Trowsdale et al. 2005). The Meola-Motions study is a useful update in that it compares observed stormwater quality with the other catchment studies covered in NIWA (2005). The Henderson (or "Twin Streams") study compares stormwater quality between lifestyle, peri-urban and urban catchments. These references are given for the sake of completeness but stormwater quality and treatment is not addressed any further.

3.3 Sources of Cu, Pb and Zn in the urban landscape

As described above, the initial budgets (Timperley et al. 2005) found that the sources of metals did not match with their loads measured in stormwater. There were large contributions from "unknown" sources in this budgeting exercise. While the mass budgets for zinc appeared to be relatively well understood, the mass budgets for copper and lead were incomplete. Loads from unidentified sources totalled approximately 60 per cent of the copper load for the residential, 70 per cent for the commercial, and 80 per cent for the industrial catchments. The mass budgets for lead were somewhat better for the residential catchment, with unidentified sources only 8 per cent of the catchment load; however unidentified sources accounted for 60 per cent of the commercial, and 80 per cent of the industrial catchment load.

The ARC commissioned a major review of metal sources within the Auckland urban landscape (Kennedy & Pennington 2008). New Zealand and overseas data were reviewed and estimates made of likely contributions from a wide variety of sources (the picture of the house from Kennedy & Pennington 2008 show many of the sources considered). While some sources could be well characterised and estimated, many could not, although most of these were arguably negligible. The review updated the metal budget (Table 1 and 2), accounting for more of the metals, and pointing the way for future refinements.

Figure 1
Example to show sources of metal sources in the urban Auckland landscape.



There is still a great deal of uncertainty in estimates and these are described in the report. Significant sources for copper were precipitation (wet and dry), brake linings, natural soils, garden soils, road wear, roof material and building walls. Minor sources were tyres and potable water.

Significant sources for lead were precipitation (wet and dry), soils (natural, roadside, garden), roof material, lead head nails and building walls. Minor sources (<1 per cent) were tyres, brake linings, wheel weights and potable water.

The sum of the contributions still falls short of the measured catchment load, although the budget between sources and catchment exports are much closer than in the 2005 study (Timperley et al. 2005). However, one of the significant differences between the estimates of Timperley et al. (2005) and Kennedy & Pennington (2008) is the inclusion of a significant precipitation component in the later estimates. This is unlikely to be an additional independent source (eg, air pollution resulting from activities outside the region, localised industrial pollution) because metals in precipitation are probably largely sourced from the catchment activities (eg, vehicle emissions, dust resuspension), which are already accounted for in both budgets. This uncertainty needs to be resolved. Other uncertainties in contribution for key sources, such as Cu from brake linings, should be resolved by more measurements. The very comprehensive report of Kennedy & Pennington (2008) will form a useful starting point and provide a format to refine these estimates, and should be updated when further measurements are made.

Table 1

Copper budget for three urban catchments (adapted from Kennedy & Pennington 2008). The estimated contribution is presented as the proportion (%) of the measured total catchment load (Timperley et al. 2005). Significant sources are highlighted. Confidence in estimate are given by the uncertainty levels where 1= high certainty to = 5 greatest uncertainty. CBD = Auckland Central Business District (commercial), Mission = residential catchment, Mt Wellington = industrial catchment.

Primary source	Secondary source	Estimated contribution (%)			Uncertainty level
		CBD	Mission	Mt Wellington	
Precipitation	Wet	13.9	2.1	10	3
	Dry	20.5	20.3	12.4	4-5
Vehicles	Exterior paint	NA	NA	NA	5
	Wheel weights	NA	NA	NA	3
	Brake linings	29.8-74.6	7.3-18.7	13.8-35.5	3-5
	Tyres	<0.1	<0.1	0.6-1.2	3
	Oil & grease	<0.1	<0.1	<0.1	3
	Accidents	NA	NA	NA	5
Soils	Natural	1.7	7.4	8.5	3
	Roadside	0.1	<0.1	<0.1	4
	Garden	2.8	9.4	0.4	3
Road wear	Pavement	2.2	1.8	0.4	4
	Marking paint	NA	NA	NA	4
Buildings	Roof material	<2.6	13.1	5.2	4
	Roof	NA	NA	NA	4

Primary source	Secondary source	Estimated contribution (%)			Uncertainty level
		CBD	Mission	Mt Wellington	
	infrastructure				
	Lead head nails	<0.1	<0.1	<0.1	4
	Walls	4.0	0.2	0.3	4-5
	Site infrastructure	NA	NA	NA	4
Potable water	-	1.1	1.4	1.1	5
Garden & household products		NA	NA	NA	5
Litter	-	NA	NA	NA	5
Street infrastructure	Galvanised poles	NA	NA	NA	5
	Cable & wiring	NA	NA	NA	5
	CCA treated timber	NA	NA	NA	5
Total	**-**	**79-124**	**63-75**	**53-74**	**-**
Total 2005	**Timperley et al. 2005**	**40**	**30**	**20**	**-**

Table 2
Lead budget for three urban catchments (adopted from Kennedy & Pennington 2008). The estimated contribution is presented as the proportion (%) of the measured total catchment load (Timperley et al. 2005). Significant sources are highlighted. Confidence in estimate are given by the uncertainty levels where 1=low (ie, high certainty) to = 5 (high). CBD = Auckland Central Business District (commercial), Mission = residential catchment, Mt Wellington = industrial catchment.

Primary source	Secondary source	Estimated contribution (%)			Uncertainty level
		CBD	Mission	Mt Wellington	
Precipitation	Wet	19.8	3.6	9.9	3
	Dry	15	17.2	7.3	4-5
Vehicles	Exterior paint	NA	NA	NA	5
	Wheel weights	0.2	1	0.2	3
	Brake linings	0.6	0.7	0.1	3
	Tyres	0.17	0.25	0.11	3
	Oil & grease	<0.1	<0.1	<0.1	3
	Accidents	NA	NA	NA	5
Soils	Natural	2.5	12.7	2.3	3
	Roadside	4.1	1.9	0.8	4

Primary source	Secondary source	Estimated contribution (%)			Uncertainty level
		CBD	Mission	Mt Wellington	
	Garden	7.9	28	1	3
Road wear	Pavement	0.5	0.4	<0.1	4
	Marking paint	NA	NA	NA	4
Buildings	Roof material	7.2	<2.9	3.3	2
	Roof infrastructure	NA	NA	NA	4
	Lead head nails	3.9	4.6	2.1	4
	Walls	1.8	0.8	0.6	5
	Site infrastructure	NA	NA	NA	5
Potable water	-	0.3	0.4	0.2	5
Garden & household products		NA	NA	NA	5
Litter	-	NA	NA	NA	5
Street infrastructure	Galvanised poles	NA	NA	NA	5
	Cable & wiring	NA	NA	NA	5
	CCA treated timber	NA	NA	NA	5
Total	-	64.1	76.8	28.5	-
Total 2005	Timperley et al. 2005	40	92	20	-

3.4 The catchment load models

Predicting stormwater contaminant loads

There have been major improvements in the ways urban loads are estimated. Until recently, loads have been calculated using specific loads or flow weighted means generic to a particular urban land use (eg, residential, commercial etc.) (Seyb & Williamson 2004, URS 2001, URS 2004) and population density and road areas (Green et al. 2004). These types of models are almost universally used throughout the world for stormwater management, and they can include some source differentiation (eg, roads, permeable surfaces, other impermeable surfaces). Since 2005, the Contaminant Load Model has been developed by the ARC. This model predicts loads of TSS, Cu, Zn and Total Petroleum Hydrocarbons (TPH) for a single year for a wide range of surfaces in the urban landscape, and is available through the ARC website (www.arc.govt.nz/environment/water/stormwater/contaminants-in-auckland-stormwater.cfm).

The above models do not allow for the dynamic variation in the urban landscape from changes in the intensity of sources (eg, traffic density) or changes in source materials (eg, roof types) or source controls. A more recent advance, the Urban Stormwater Contaminant 3 - Contaminant Load Model (USC3-CLM) (Timperley & Reed 2008a, b) provides this dynamic facility, and a more detailed breakdown of sources.

The USC3-CLM is a relatively detailed model in the sense that it calculates loads from many different distinct surfaces such as roads, motorways, other pavement, urban parks, stream channels, construction sites and roofs. It does this for residential, commercial and industrial land uses. Roads are split into four categories depending on traffic density; motorways into two categories; roofs into eight categories depending in roof material (Figure 2). The model uses specific yields of contaminants from surfaces ($g/m^2/yr$), which draw on direct measurements from roof run-off, road run-off, solids trapped in catchpits, stream erosion studies, soils data, reviews and overseas information. Sediment specific yields for non-urban (rural) areas were obtained from GLEAMS modelling (see next section). Rural loads can also be dealt with simply in the model, but it is also more usual to use GLEAMS.

Historical predictions are made based on an assessment of past land use, traffic numbers and roof types. Future predictions are more dynamic, and take into account changes in roofing materials, population trends and traffic densities. These are in turn based on industry roof material sales figures and roof longevity, and on urban growth models. Urban expansion and population growth is allowed to occur within the model through infilling and more-dense housing (ie, apartment building). These changes are allowed to take place in realistic ways, with appropriate losses in pervious area, increase in impervious areas and the short-term appearance and disappearance of construction (bare soil) sites. The model incorporates various means for addressing source controls and stormwater treatment controls, as well as ways to vary the rate of (time line for) implementation.

Figure 3 shows examples of the change in roof and road types into the future in Auckland City. These changes illustrate the dynamic nature of the model, allowing for realistic projected changes in the cityscape into the future.

Figure 2
Schematic breakdown of land use and urban "surfaces" included in the CLM (Adapted from TImperley & Reed 2008 a).

The CLM model sums the loads from individual sub-catchments (Stormwater Management Units or SMUs) to produce the total load leaving the catchment. There is no attenuation of contaminants in the stream channels of flood plains. Figure 4 shows the predicted total TSS, zinc and copper loads to the whole Central Waitemata Harbour (Timperley & Reed 2008b).

The model has been applied to the catchments draining to the Central Waitemata Harbour (Timperley & Reed 2008b) and South East Manukau Harbour (Moore &

Timperley 2008a,b) to provide input to the USC-3 model predicting sediment and metal fate in these estuaries.

Figure 3

A) Trends in roof area (m^2) of different materials in the residential part of Auckland City. B) Trends in areas (m^2) of local roads in the commercial area of Auckland City. vpd = vehicles per day, units of Y-Axis are m^2. (Reproduced from Timperley & Reed 2008a).

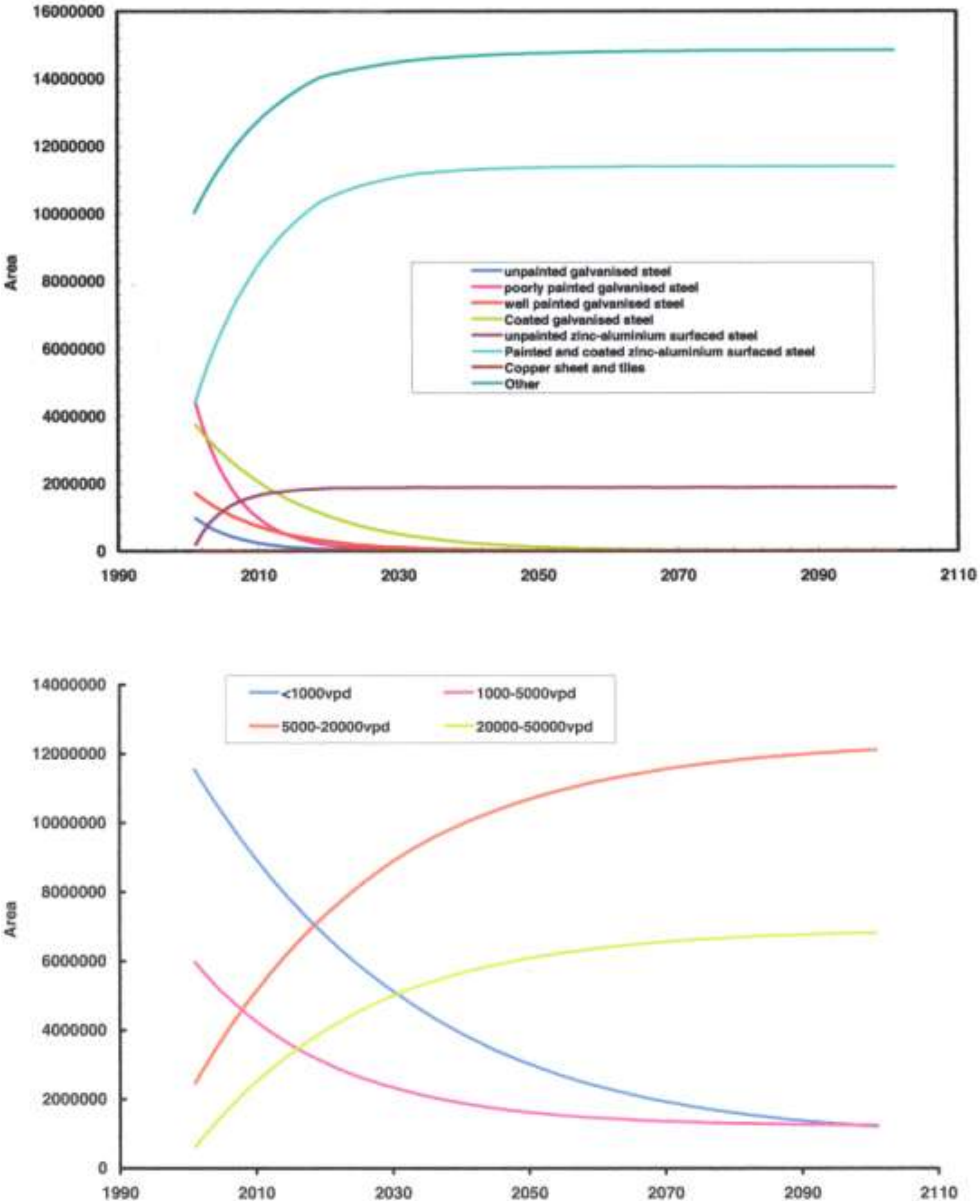
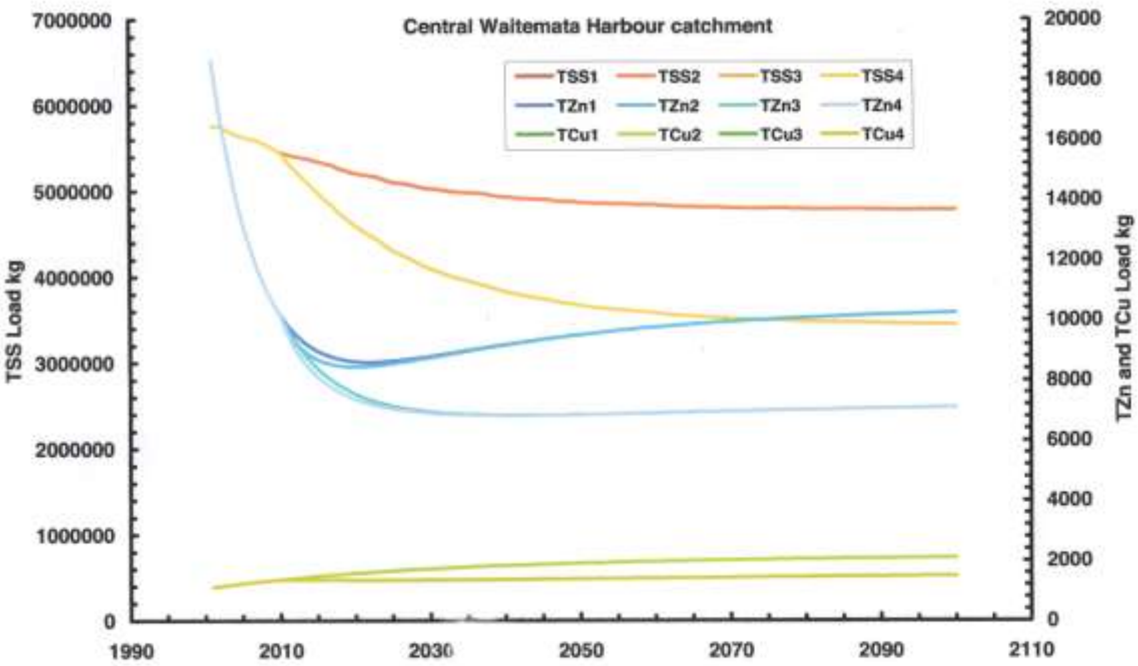


Figure 4

Predicted total loads of TSS, total Zinc (TZn) and total Cu (TCu) to the Central Waitemata Harbour (excluding inputs from the Upper Waitemata Harbour) for 4 different source control and stormwater treatment scenarios (Reproduced from Timperley & Reed 2008b).



This model improves on previously used predictive tools and load estimates by estimating contributions from all the major surfaces (roof, roads, motorways, pervious areas, stream channel, construction sites) in urban areas, taking into account roof materials, and traffic densities. Unknown contributions (mostly from other paved surfaces) were used as calibration factors. The model was calibrated to loads measured in three representative catchments in Auckland (described in Timperley et al. 2005), and specific yields from other paved surfaces were adjusted to get a good fit. The shortcomings of the previous attempt to conduct a source budget for Auckland (Timperley et al. 2005) was overcome in the CLM with a more complete source inventory, improved data to inform the model, use of overseas data where NZ data was lacking and the model calibration.

Predicted changes in roofing materials, based on sales figures and expert assessment of the life of existing galvanised steel roofs, has major implications for Zn loads and concentrations in the receiving environment in the future. There is a relatively rapid decrease in Zn loads at the present time and the near future (Figure 4). A major consequence is that there is a large change between historical trends (where there is a gradual increase in Zn loads to 2001) and future predictions, where there is an almost exponential decrease in Zn loads over the next 20 years.

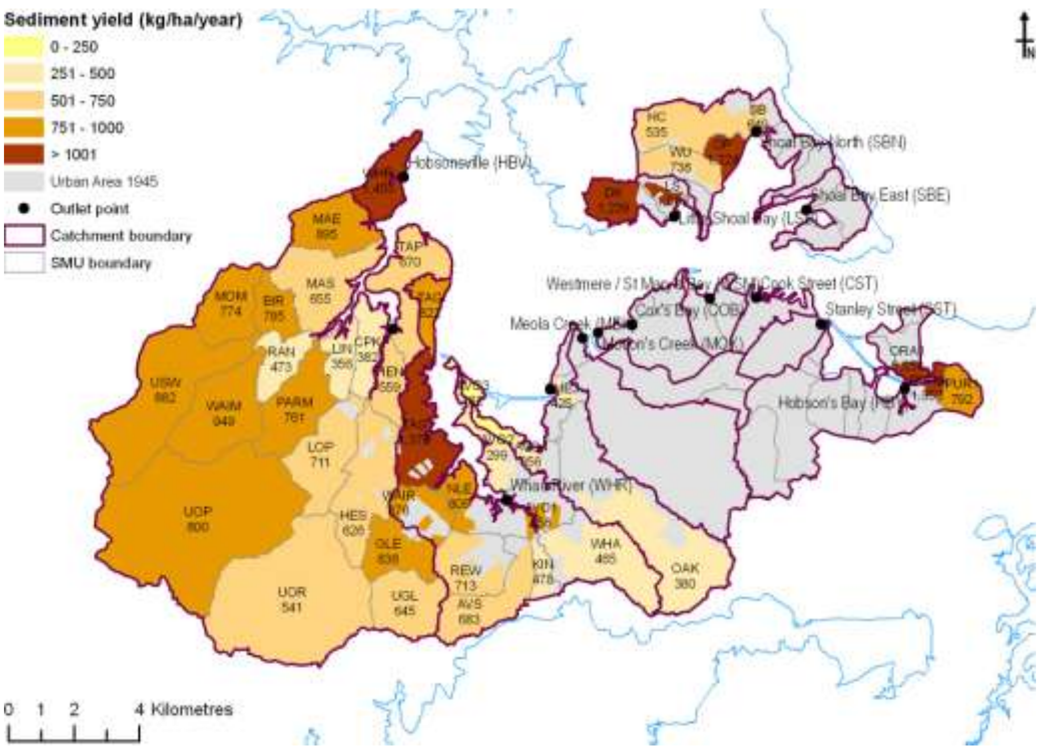
3.5 Rural suspended sediment loads

A major source of sediments, natural metals and a diluent for urban contaminants

Rural areas are an important source of sediments to their receiving waters. Where significant rural areas exist upstream of urban areas or as part of the overall watershed, the rural sediment load is an important component of the total sediment load. This affects sedimentation rates and also the concentrations of contaminants in the receiving environment sediments, largely because the rural sediments act as a diluent to contaminant inputs from urban areas. GLEAMS was described in the 2005 Review (ARC 2008) where it was used in modelling the Upper Waitemata Harbour, Orewa and Weiti estuaries. The GLEAMS model has been improved and tested, and extended for use in hindcasting (Parshotam & Wadhwa 2008a-c, Parshotam 2008, Parshotam et al. 2008a-c). The outputs from the model were used in USC3 modelling in the Central Waitemata harbour and South East Manukau harbour (see Section 8). An example of model output is shown in Figure 5.

Figure 5

Sediment yields (kg/ha/yr) predicted for 1945 by GLEAMS for the Central Waitemata Harbour watershed. SMU – stormwater management unit. Loads are averaged over each SMU (Parshotam & Wadhwa 2008b).



Improvements in the prediction of natural or soils metal loads were made by collecting additional information on Auckland's soils (eg, Reed 2007), which could be applied in the GLEAMS outputs for rural areas, or incorporated in the USC3-CLM model for urban areas.

3.6 Polycyclic aromatic hydrocarbons in Auckland's aquatic environment

A comprehensive summary of sources, fate and effects of PAH

In 2005, the ARC commissioned a major review of sources, concentrations and environmental risk posed by Polycyclic Aromatic Hydrocarbons (PAH) (Depree & Ahrens 2007). The environmental risk assessment described by these authors is summarised in Section 10 – Toxicity.

PAHs are a well-documented class of persistent organic pollutants that are characteristic of most urbanised and industrial areas. They derive from a variety of sources, both ongoing and historic, the majority of which are incomplete combustion of organic matter (such as from residential heating and vehicle emissions) or distillation of fossil fuels. Because PAHs are poorly water-soluble and degrade only slowly under anaerobic conditions, they tend to accumulate in sediments and in biological tissues to concentrations that, if high enough, may have adverse effects on resident biota.

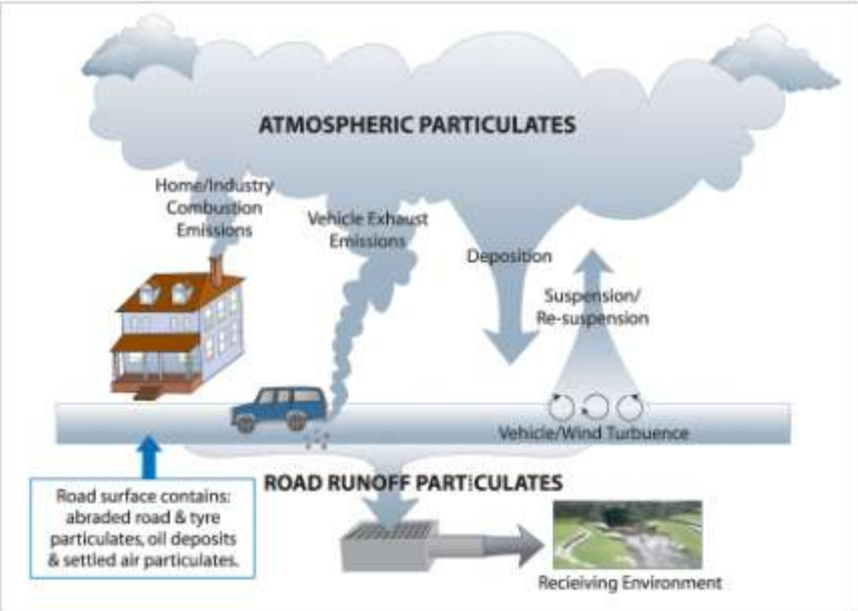
The majority of sediment PAH concentrations in Auckland's estuaries are well below ARC's environmental response criteria (ERC Red) of 1.7 mg/kg (for high molecular weight PAHs)[4]. In a small number of estuarine locations and freshwater creeks (such as Meola, Motions and Oakley Creek, the Whau River and the upper Tamaki Estuary), sediment PAH concentrations are markedly higher.

PAH concentrations have not increased significantly in Auckland estuaries over the last ten years, and the authors suggest that levels are at steady-state. Estimates of PAH mass loadings from likely sources rule against atmospheric deposition and run-off from petrol stations as major PAH contributors and point to road run-off as the primary source (Figure 6).

[4] The ARC's PAH ERC are for high molecular weight PAH. ANZECC guidelines are given for both high molecular weight and total PAH.

Figure 6

Schematic showing the connectivity and inter-relationships between different anthropogenic sources, atmospheric particulates and road run-off particles. Adapted from Depree & Ahrens (2007).



Analyses of deep sediment cores in the Manukau Harbour have demonstrated that sediment PAH levels in Auckland were historically higher than they are today (ARC 2008), probably due to the past widespread use of coal for heating and power, as well as PAH-rich coal-tar for road construction. While the use of coal tar largely ceased by the late 1960s, recent coring of Auckland city streets and footpaths has demonstrated that there are still considerable quantities of PAHs locked in pavements in the form of coal-tar, such as found in older residential catchments upstream of the most PAH-contaminated monitoring sites. Combining advanced source identification techniques ("PAH fingerprinting") and characterisation of PAH distribution in sediment sub-fractions has provided further evidence that the majority of PAHs in high PAH catchments (eg, Motions Creek) are not derived from current vehicle or residential emissions, but rather from roading coal tar. Input of PAH-rich, coal-tar laced pavement material is still ongoing, explaining the high PAH levels in the adjacent creek and downstream estuary.

Source identification using diagnostic PAH ratios confirmed that composition of PAHs in the most contaminated receiving environment sediments is different to that of typical road run-off particulates. These sediments were characterised by pyrogenic PAH compositions with strong similarity to pyrogenic source materials, such as wood soot, coal soot and coal tar. In contrast, road run-off particulates had a mixed or "intermediate" PAH composition. The "intermediate" composition observed for run-off and air particulates is presumably due to a mixture of petrogenic (eg, road/tyre abrasion and engine oil) and pyrogenic (eg, combustion soot) source particles (Figure 6).

Figure 7

Auckland air particulates, sampled three times throughout the year, had a very similar PAH composition to road run-off. This is consistent with these two particulate phases being linked by deposition and suspension/resuspension processes (Figure 6). Analysis of coal soot and wood soot marker compounds showed that whereas these combustion sources are relevant in other cities (ie, Christchurch) there is little indication that they constitute a major source of PAHs in Auckland air particulates, run-off or sediments. Concentrations and composition of PAHs in high PAH aquatic sediments are consistent with coal tar constituting over 80 per cent of the PAHs.

Despite the strong weighting due to historic coal tar inputs in some sediments, run-off from modern bituminous-based roads is likely to be an additional important source of PAHs in Auckland, especially in catchments that were developed after the coal tar era of roading (ie, post 1960s). Modern road particulates collected in a stormwater treatment device in Grafton Gully contained PAH concentrations of 10-13 µg/g.

The proportion of PAHs attributable to run-off particulates was estimated for Auckland receiving environment sediments. Modern road run-off particulates contributed between 4-62 per cent of the total concentration of PAHs in the receiving environment sediments examined. For sediments with higher PAH levels, the proportional contribution from modern road run-off was lower. For example, for stream sediments containing more the 4 mg/kg total PAHs (and therefore of possible concern because this exceeds ANZECC guidelines) modern run-off particulates contributed only 4-14 per cent of the total PAHs. At these sites, coal tar inputs from historic road construction practices are the major source of sediment PAHs. For sediments containing approximately 1 µg/g of PAHs, modern run-off contributed 24-62 per cent of the PAHs. Based on these estimates, and the assumption that the existing "dilution" by inorganic and organic material continues, it seems unlikely that PAH inputs from modern road run-off have the capacity to raise PAH sediment concentrations above the ARC's ERC red criterion of 1.7 mg/kg. However, if the sediment inputs to estuaries were to become more dominated by road run-off particulates (as is predicted for the Central

Waitemata Harbour – see Section 8), then the concentrations could theoretically exceed the ARC red guideline.

3.7 Industrial sources

Management of site operations to minimize stormwater contamination

Industrial and commercial activities can be an additional source of contamination to urban stormwater. While it is difficult to quantify this source because it is site specific compared with generic sources such as road run-off, its management is of utmost importance. This is covered under the ARC's Industrial Pollution Prevention Programme (IP3). This is an ongoing programme that primarily focuses on the high priority (high risk) industries (eg, electroplating).

Figure 8

The following Table 3 is an example of the sources uncovered in an intensive survey ("blitz") of the industrial areas that drain to Wairau Creek in the Whau Estuary (ARC 2000, 2001). The purpose of a blitz is to visit all industrial and commercial premises in each area not categorised as high priority, to investigate actual and potential stormwater and/or groundwater contamination issues associated with activities carried out on each site. [Specific high priority sites are targeted individually and regionally in separate occasions]. Where actual pollution problems are identified, advice is given regarding any remedial work required. Educational material is provided to all site owners/occupiers to raise awareness of potential sources of water pollution from each site and to advise on what actions should be taken to minimise these risks.

Table 3
Actual and potential issues identified during a "blitz" of 99 commercial and industrial premises in the Kelston and McLeod Rd Areas in 2001 (ARC 2001).

Business category	No. of premises with problems		Total no. of problems		Contaminated sites
	Potential	Actual	Potential	Actual	
Motor vehicle	15	8	31	12	4

Business category	No. of premises with problems		Total no. of problems		Contaminated sites
	Potential	Actual	Potential	Actual	
Engineering	7	4	13	14	3
Transport	1	-	1	-	-
Chemical	5	-	7	-	-
Manufacturing	6	3	12	7	2
Miscellaneous	5	4	9	8	1
Totals	39	19	73	41	10

The distribution of actual and potential problems identified during blitzes reflects the type of industry in the study area. The main types of businesses found polluting were in the motor vehicle, manufacturing and engineering categories, which are largely considered to have a low or medium potential pollution risk.

Typical problems identified during the McLeod Road and Kelston area blitzes are listed below:

- washing of equipment, used parts, floors and vehicles drained to the yard or stormwater system;
- compressor condensate discharging to ground or stormwater system;
- contaminants leaking onto the ground from waste bins stored outside;
- poor housekeeping such as old/empty drums left outdoors;
- companies lacking a spill plan or staff education on environmental issues;
- drums, tanks and containers of chemicals and oils stored in an inappropriate manner;
- inadequate bunding;
- spillages onto the yard and into the stormwater system;
- uncovered or uncontained waste bins; and
- ground contamination.

In most cases, it was relatively simple to carry out the remedial works or action required to prevent the discharge of pollutants to the stormwater system or onto the ground. Sometimes clean up requires significant expenditure by the company, so the IP3 impress on company staff and management the importance of preventative action. Industrial and commercial sites that handle large quantities of substances that may be toxic in receiving waters have the potential to grossly contaminate the environment. Therefore the IP3 programme is important for managing the quality of urban stormwater. Over the last 10 years there has been a great deal of effort quantifying diffuse source loads of TSS, Cu, Pb and Zn (eg, see Section 3.3), but there has been little effort in quantifying potential point source loads from industrial spillage etc. This is an important information gap, especially for other potential contaminants such as Hg and CPEC (see Section 3.8). Quantification of sources is not aimed to inform the IP3 programme (although this will also be achieved) but to better understand potential impacts in the environment.

3.8 Emerging chemicals of potential concern

Chemicals that are found at increasing frequency and increasing concentrations in the receiving environment but have as yet uncertain significance

The 2005 Review (Mills & Willamson 2008) concluded that further investigation of the levels and significance of some Persistent Organic Pollutants (POPs, including phthalate esters and PCBs) and Hg was warranted to assess whether they are likely to be important stormwater-derived contaminants in Auckland's receiving environment. Some of these are Priority Pollutants – contaminants with high environmental persistence, high bioaccumulation and high acute toxicity. ARC has monitored many of the traditional priority pollutants (eg, PAH, organochlorine pesticides, PCBs) in recent times.

A new group of chemicals are emerging as being of potential environmental concern, based on their toxicity, persistence, and widespread or on-going use. These have been termed Chemicals of Potential Environmental Concern (CPEC)[5]. In contrast to the "priority pollutants" which have consistently high environmental persistence, high bioaccumulation and high acute toxicity, many CPECs have a lower environmental hazard profile. Notably, many CPECs have lower acute toxicity than Priority Pollutants (PP). Nevertheless, some CPECs have a potential to exert chronic effects by being neuroactive or acting as hormone mimics (endocrine disrupting chemicals). Some are associated with high production volumes, so there is a potential for accumulation of these chemicals in Auckland's receiving environment, with unknown consequences. The differences between PP and CPEC are summarised in Table 4.

Table 4
Comparison of risk profile of priority pollutants and emerging chemicals of potential environmental concern (adapted from Ahrens 2008).

Property	Priority pollutants	CPEC
Toxic effects & mode of action	Acute & chronic.	Not likely to be acutely toxic at environmental doses, but potentially bioactive (eg, estrogenic, neuro-active).
Environmental concentrations	Frequently monitored; stable or decreasing (except Zn, Cu, PAH in urban stormwater).	Not frequently monitored, assumed to be increasing.
Persistence	High.	Variable: low, medium, high.

[5] Also termed Emerging Chemicals of Concern (ECC), or emerging contaminants.

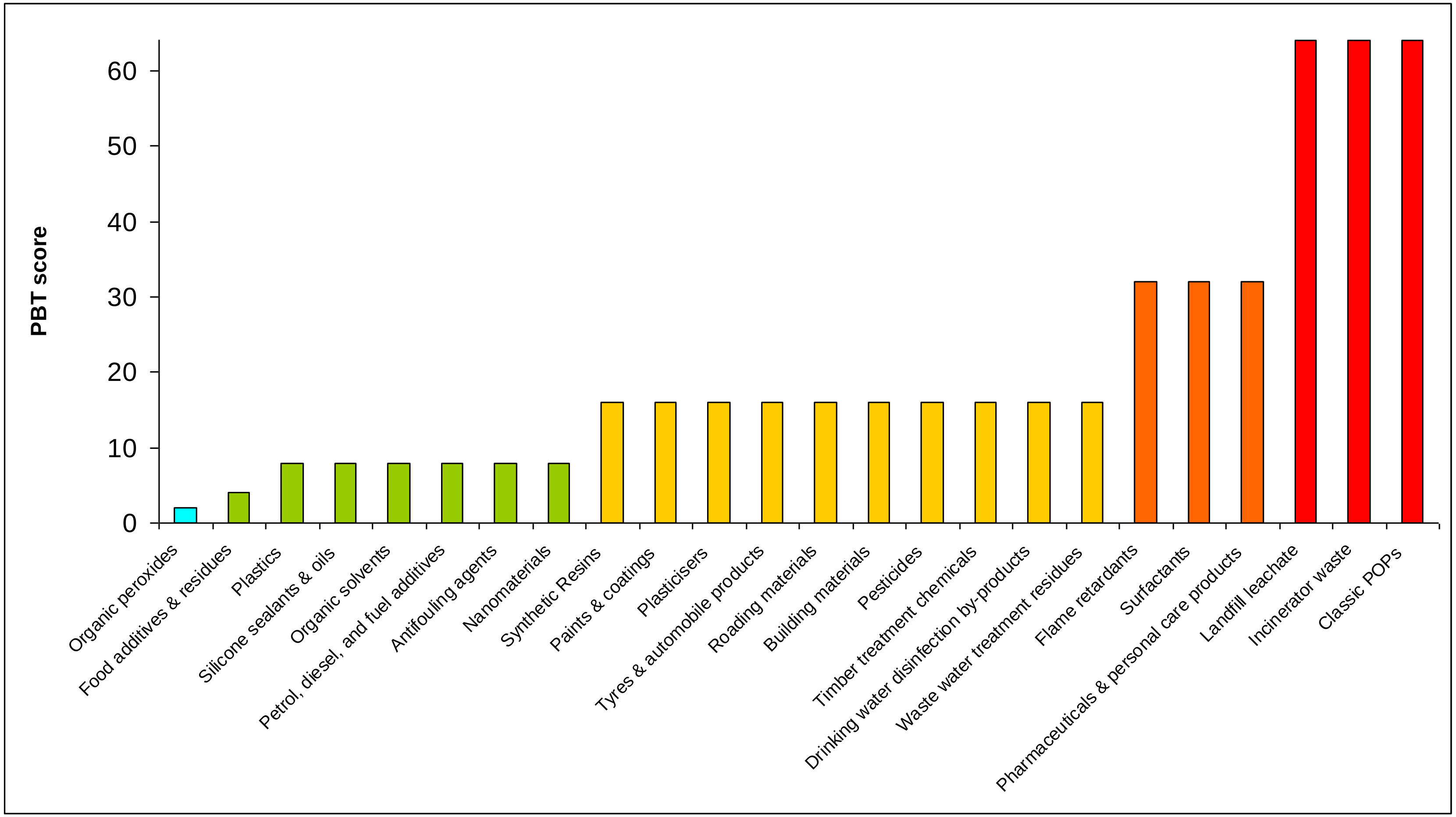
Property	Priority pollutants	CPEC
Bioaccumulation potential	High.	Variable: low, medium, high.
Sources	Mainly industrial & agricultural; building materials and vehicles; few domestic (ie, sewage).	Some industry & agriculture; mostly domestic (via sewer overflows, wastewater discharges).
Existing water quality guideline	Yes.	No.
Discharge regulated	Often (but not in diffuse run-off in NZ).	Rarely.
Detection and quantification	Relatively easy; methods are well established.	Difficult and expensive to measure.
Examples	Mercury, lead, arsenic, DDT, PCB, PAH, dioxin, Cu, Zn.	Surfactants, plasticizers, disinfectants, pesticides, flame retardants, hormones, cosmetics.

Ahrens (2008) conducted a very comprehensive review of CPEC that are emerging in the world's literature. Based on this review, CPEC do not appear to reach environmental concentrations able to exert **acute** toxicity effects on biota. However, if moderately elevated concentrations are present, or bioavailability is enhanced with long-term exposure, there is the possibility of chronic effects on organism health. Because they are likely to occur in mixtures, there is the possibility of additivity of toxicity of chemicals with a common mode of action, such as endocrine disruptors. Thus while the environmental concentrations may fall below the levels where a specific chemical is known to effect organisms, these chemicals may act in concert, producing an additive adverse effect.

As well as urban stormwater as a potential source for such CPEC as pesticides, plasticizers, petroleum products, Ahrens (2008) identified many other potential sources in the urban landscape including marinas, sewage outfalls, combined sewage overflows, landfill leachate, and agricultural run-off. To summarize and compare the potential hazard profile of these chemicals in the Auckland environment, he used the qualitative persistence-bioaccumulation-toxicity (PBT) classification scheme. This scheme assigns a hazard rank based on the persistence of the chemical in the environment, its tendency to accumulate in biota and its toxicity to aquatic organisms. A summary of the environmental hazard rankings, as PBT scores, is shown in Figure 9 for 24 product categories.

Figure 9

Summary of qualitative environmental hazard ratings (Persistence-Bioaccumulative-Toxicity (PBT) scores) for all CPEC summarised in the 24 general compound classes described in Ahrens (2008). Relative score is rated 0 – 100, with 100 as most hazardous. See Ahrens (2008) for rating method.



3.9 Conclusions

A detailed review and inventory of sources of Cu, Pb and Zn in the urban environment has been undertaken. The refined inventories have estimated contributions from a wider variety of sources than previous studies, but were unable to account for all Cu and Pb exported from three catchments. It has shown where further measurements need to be made.

Continual improvements in catchment load modelling have led to a much-improved model – the Contaminant Load Model (CLM). The model predicts TSS, Zn, Cu and TPH loads from urban catchments. This model improves on previously used predictive tools by estimating contributions from all the major surfaces (roof, roads, motorways, pervious areas, construction sites, other paved areas) in urban areas. A dynamic version of the CLM, which includes predictions of changes of sources into the future, has been applied to Central Waitemata Harbour and SE Manukau Harbour (USC3-CLM). It takes into account roof replacement, projected changes in roofing materials, traffic density projections, population densities and associated projections (infilling, increased impervious areas). Coupled with this development are improvements in predicting contributions of suspended sediment from adjacent or upstream rural areas.

The CLM model has greatly improved predictions, and allows the modeller to match input variables to specific catchment details. This is a substantial advance in estimating sources of contaminants. Further work could extend modelling to other contaminants (such as Pb) and it would be useful to compare the CLM with the source study of Kennedy & Pennington (2008) described above. The USC3-CLM incorporates predictions of the remaining life of existing galvanised steel roofs. Many of these roofs will be replaced with zinc-aluminium coated steel that produces much less zinc in run-off than galvanised steel. This will result in a rapid decline in Zn loads to receiving waters over the next 20 years, which has major implications for impacts of stormwater.

The detailed review of sources of PAH has distinguished "modern sources" (mostly from bituminous roads) and legacy sources (from coal tar). The latter can be a much larger source of PAH to run-off. The review found that PAH loads from atmospheric fallout and petrol stations are small. This review is coupled with a summary of information on PAH in the receiving environments and an assessment of likely ecological effects (see Section 10 – Toxicity).

While work in the Industrial Pollution Prevention Programme (IP3) is not new, this has been included in this review to fill an information gap in the 2005 Review. The IP3 seeks to prevent contamination of stormwater (and groundwater) from site-specific activities on commercial or industrial sites. This contamination and potential impacts, difficult to predict with urban generic models, is indeed preventable, and is appropriately addressed by inspection and education and not source quantification and study of impacts. However, for understanding stormwater impacts, particularly for Chemicals of Potential Environmental Concern (CPEC), it would be useful to estimate the magnitude of these sources.

The potential for impacts from chemicals of potential environmental concern (CPEC) have begun to be addressed through a comprehensive review of CPEC in the overseas literature and by measuring concentrations of some of the more common CPEC in Auckland's marine receiving environment (Section 6).

Overall, there have been major advances in understanding the magnitude of sources and formulating these into a useful model that can be used by managers to estimate loads from any catchment in Auckland. The model also encapsulates the effect of changes in loads accompanying population and traffic projections, as well as changes in roofing materials. The comprehensive review of PAH is a timely, useful and excellent summary on the "state-of-the-science" that would be advantageously repeated with other major contaminants.

4 Impacts on the Freshwater Environment

Water quality of urban streams appears to be improving, and this may be partly due to improvements in stormwater. The ARC Regional Stream Monitoring Programme has shown decreases in TSS, nitrate and faecal coliform levels over the past 20 years. These, in particular TSS and faecal coliforms, have been identified as important factors limiting the potential use of urban streams and their receiving waters, and hence their improvement is a significant step forward. The programme could benefit from pressure-state-response (PSR) analysis to better link the improvement to changes in catchment conditions and hence direct management towards actions that will continue the trends.

The ecological functions of regional streams (including urban streams) are now able to be described and measured through the Stream Environmental Valuation (SEV) methods. This provides a key tool for effective management of urban streams, which, if implemented, should result in real improvements in stream ecology and address many of the impacts of urbanisation.

Toxicological understanding has been advanced on the effects of short-term, high concentrations of TSS, which can occur during major storms in urban streams draining catchments which are undergoing development. The work reported has found that New Zealand stream biota are quite resilient in the most part to TSS, so infrequent, very high TSS levels are not especially toxic. A new initiative studying long-term toxicity across several generations of stream biota commenced during the 2008.

A number of studies have measured the reduction in low flows brought about by urbanisation. This effect can be very significant in some catchments, but not in others, and further work is needed if a regional perspective is to be gained.

Overall, from 1995 to 2008, there was enormous progress in understanding stormwater effects in urban streams, to the point that many of the tools for measuring stream health and effective management are now available. Most of the advances address the effects of flow, habitat modification and changes in ecological functioning. Of these, the SEV methodology is the latest, and potentially most powerful, development.

4.1 What was known by 2005

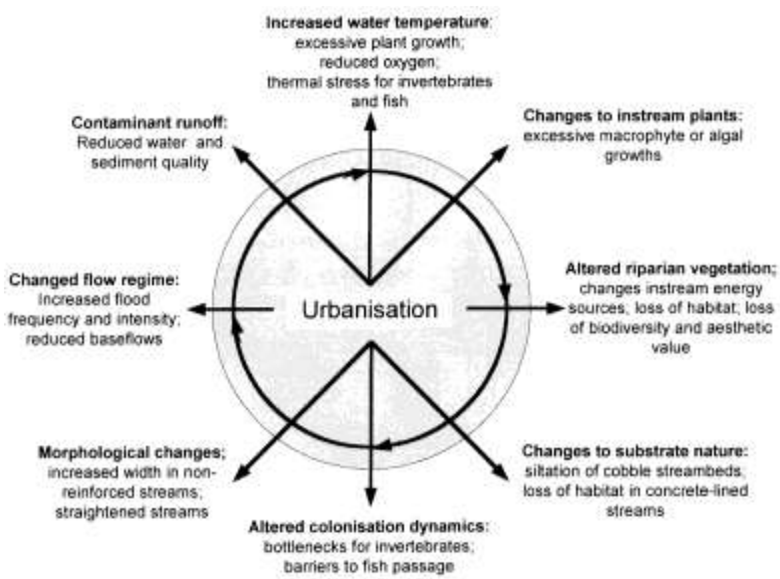
Urbanisation causes a wide range of changes to the stream environment, affecting water quality, stream flows, stream channel shape and stability, riparian vegetation, and streambed characteristics. These changes (or “multiple stressors”) can adversely affect the quality of aquatic plant and animal communities inhabiting urban streams. The diagram below (Figure 10) summarises these potential effects.

By 2005, a very large body of work had been conducted on improving our understanding of the impacts of urbanisation on Auckland's streams. The work had:

- provided an extensive database on water quality, stream ecology, and (to a lesser degree) sediment quality, which have enabled the impacts on streams to be well characterised; and
- 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.

Figure 10

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



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

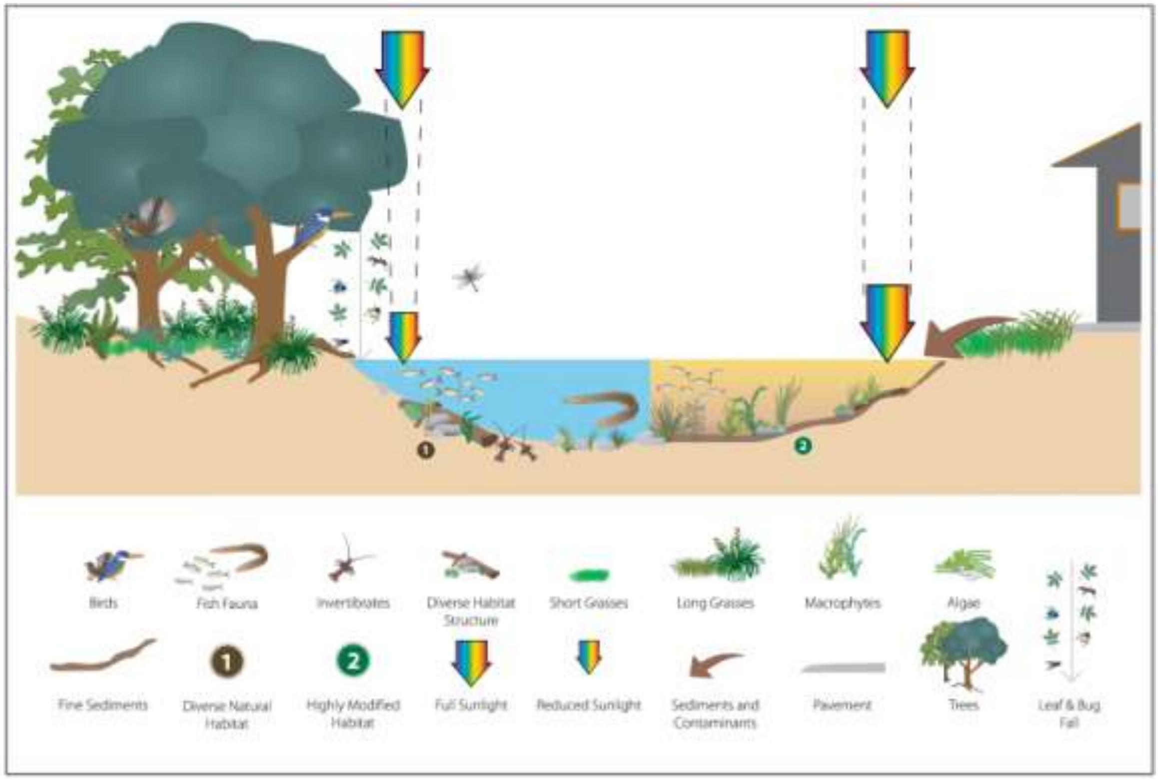
The information shows that urban streams have the following characteristics. Figure 11 illustrates many of these characteristics.

- 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 both 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.

Figure 11

Comparison of typical unmanaged urban reach with natural reach of a small stream.



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.

4.2 Outline of new work 2005-2008

A key gap identified by 2005 was the lack of information on assessing the ecological functions of urban streams and how they might be measured, and therefore managed and restored. While ecological functions had been identified (ARC 2004a) within the ARC's stream categorisation framework, additional methodology was needed to facilitate their quantification. New work was undertaken in the following two areas:

1. The ARC's management framework for urban streams based upon the nature of the stream channel substrate (natural or artificial) and (for natural channels) the percentage of the catchment with impervious cover (%IC) was re-evaluated, updated and ratified (Section 4.3).
2. The ecological functions of Auckland streams were more completely described, and a scoring method developed, and this is now widely used in Auckland and in other parts of New Zealand. This is a key advance (Section 4.4).

It was also timely that a more rigorous assessment of regional monitoring programme was undertaken and this led to the following update:

3. Regional water quality being reassessed using new trend software, which established that urban stream water quality, is generally improving regionally (Section 4.5).

Other work continued elucidating and understanding the response of stream animals to stressors encountered in urban streams or during the process of urbanisation. Three studies expanded or will expand our understanding on stressors in freshwater streams:

4. Stream organisms are exposed to very high suspended solids concentrations for short periods of time in developing areas with inadequate or failed soil erosion management. Laboratory studies examined the response of invertebrates and fish to high levels of turbidity (Section 4.6).
5. A review of the impact of piping on stream ecosystems was conducted on national and international literature and on known Auckland case studies (Section 4.7).
6. Toxicity studies have embarked on assessing toxicity through inter-generational effects, by exposing animals (in this case freshwater clams) to urban streams, then exposing their off-spring at a later date (Section 4.8).

A gap in the 2005 Review (ARC 2008) was information on the impact of urbanisation on the low flow regime of streams. It is commonly assumed that there is a reduction in low flow:

7. The low flow regime of five streams was critically assessed in two studies (Section 4.9).

Since 2005, there have also been a number of other useful reviews published. In their assessment of stormwater quality and impacts for Auckland City Council, NIWA (2005) placed the results in the context of impacts in receiving waters, stream classification, and stream toxicity. Most of this work is covered in the 2005 Review (ARC 2008), but the NIWA (2005) review offers another perspective. Landcare reviewed monitoring data for the Twin Streams region in Waitakere City (Opanuku and Oratia) (Trowsdale et al 2005). There is a great deal of information on stream ecology, as well as some water quality data, in addition to what was summarised in the 2005 Review. Moore & Trowsdale (2005) used images of aquatic life to illustrate the effects of urbanisation (including stormwater discharges) on stream ecosystems. Many of these images are shown on the Landcare

website:
www.landcareresearch.co.nz/research/biosystematics/invertebrates/freshwater_invertsCD/).

4.3 Review of the assessment and management framework of Auckland's urban streams

The usefulness of the pressure indicator "% imperviousness"

The ARC has established a management framework for urban streams based upon the nature of the stream channel substrate (natural or artificial) and (for natural channels) the percentage of the catchment with impervious cover (%IC), with six stream classes being identified (ARC 2004a). Critical thresholds for %IC were determined from surveys of urban streams in the Auckland region (Allibone et al. 2001) and overseas literature. Following the development of the urban stream assessment framework (ARC 2004a), the new biotic index for assessing the health of soft-bottomed streams (MCI-sb) (ARC 2004b) and further research overseas on the relationships between %IC and stream health, the ARC commissioned a reassessment of the usefulness of the framework and the impervious cover metric.

The report (Stark 2006) confirms the use of %IC in the stream management framework. It remains the most-commonly used catchment variable for predicting the potential impacts of urbanisation on stream health. However, the biological responses to catchment urbanisation can be highly variable (especially when %IC is low), so although %IC can provide an initial diagnosis of stream quality it is important to confirm predictions by supplemental field monitoring.

The scientific literature indicates that catchments with <10 %IC may support aquatic communities that are at or near reference condition. Whether they do or not, however, can depend on site-specific in-stream habitat and riparian quality (and probably contamination levels). However, when %IC increases beyond 10 – 15 per cent most studies indicate that stream health is reduced, and beyond 25 per cent streams become highly disturbed. These thresholds, which should be regarded as "fuzzy" boundaries, are embodied in ARC's urban stream management framework.

4.4 Ecological functions of Auckland streams

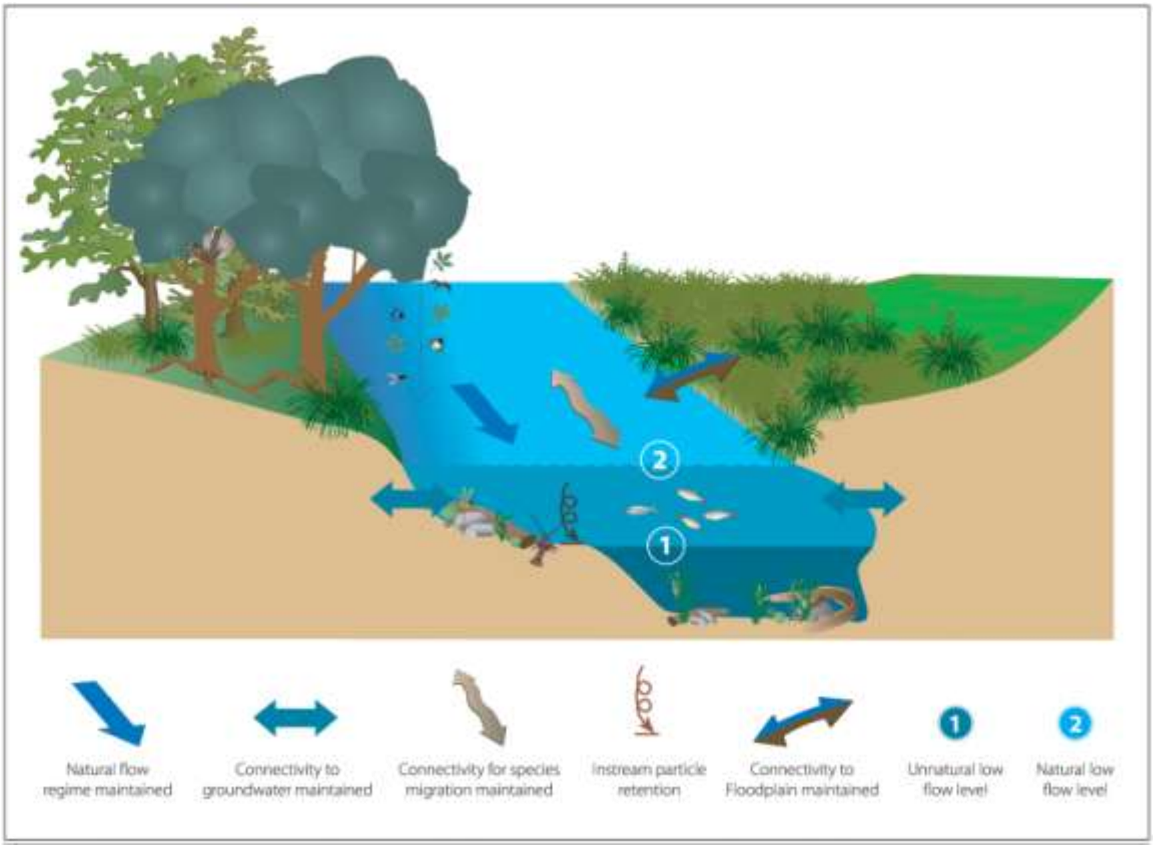
Measuring ecological functions: a new tool for effective management of streams

A critical part of improving river health is accurate assessment of the current ecological state of river ecosystems so that causes of poor health, or the success of rehabilitation efforts, can be measured. River health monitoring, which has traditionally concentrated on the use of structural measurements (such as water quality or the number and type of aquatic organisms), should be complemented in future by functional indicators (Young et al. 2004). Functional indicators are measures of the rate, or relative importance, of a particular process happening in an ecosystem, while structural indicators focus on patterns of abiotic resources or biological community composition. In other words, functional indicators measure the services or functions provided by ecosystems, while structural indicators measure what lives in and the state of an ecosystem.

A major development in stream ecology is the ability to describe and quantify the ecological functions of regional streams. Important ecological functions (Rowe et al 2008) can be categorised into:

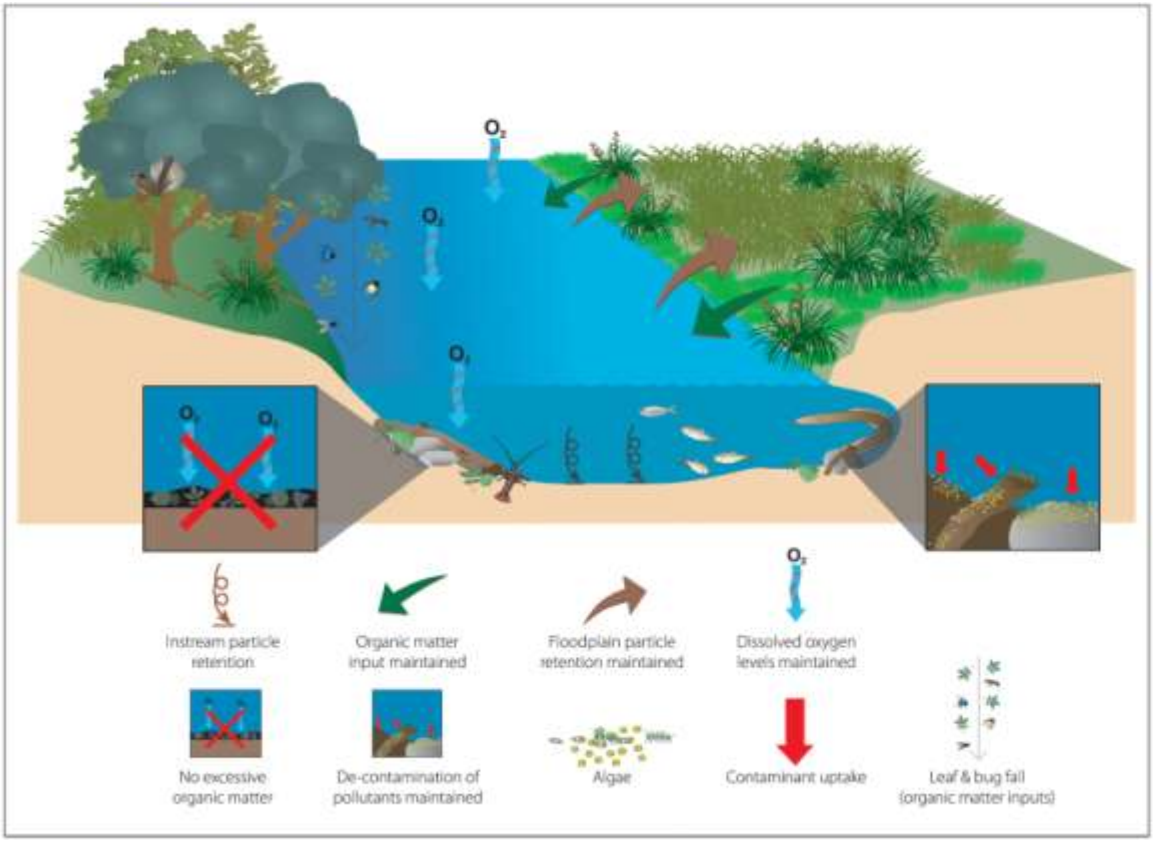
- hydraulic functions (ie, processes associated with water storage, movement and transport) (Figure 12);
- biogeochemical functions (ie, those related to the processing of minerals, particulates, and water chemistry) (Figure 13);
- habitat provision functions (ie, the types, amount, and quality of habitats that the stream reach provides for flora and fauna) (Figure 14); and
- the native biodiversity functions (ie, the occurrence of diverse populations of indigenous native plants and animals that would normally be associated with the stream reach) (Figure 14).

Figure 12
Hydrological functions of streams.



This work resulted from a review of the methods used for quantifying the values of aquatic ecosystems commissioned by the Auckland Regional Council (Rowe 2003). This study recommended applying techniques developed overseas using a panel of expert scientists familiar with Auckland streams. In a subsequent study, this panel identified important ecological functions and a robust method of quantifying these (Rowe et al. 2006, Rowe et al. 2008). This methodology was developed for Albany streams (Rowe et al. 2008), and has been applied to other stream types in Waitakere (Storey 2007) and Papakura (Phillips et al. 2007).

Figure 13
Biogeochemical functions of streams.



Specific ecological functions considered important for the Albany stream reaches within these broad categories are listed in Table 5. The overall relative importance of these functions for Albany stream was also assessed and the results are presented in Table 5.

To assess the performance of the ecological functions listed in Table 5, variables were chosen that reflected these functions and how well they were being performed within a stream reach. Such variables needed to be easily measurable in order for the methodology to be widely adapted and utilised. The variables were also combined into a single comparable measure to obtain an overall score for a particular reach. The methodology and algorithms are provided in the Stream Ecological Valuation (SEV) Guideline (Rowe et al 2006, 2008). Results for each reach are compared against reference values for natural or undisturbed streams.

The method was tested on Albany streams. Reference sites (relatively natural or unmodified) had SEV scores of 0.94. Modified reaches scored from 0.17 to 0.83. Two sites with urban land use scored 0.17 and 0.45, a rural site scored 0.51 and a lifestyle scored 0.83. Thus the methodology has a good deal of discrimination between impacts.

These methods have also been developed for determining Environmental Compensation (EC) (more commonly referred to as "offset mitigation" in the international literature). Environmental Compensation is used to offset the unavoidable residual impacts of an activity once all practical steps have been taken to minimise adverse effects. It is typically applied to a different water body as compensation for the unavoidable effects in the water body being developed. A key part in determining EC requires understanding the similarities and differences between water bodies under development and those considered for restoration as part of Environmental Compensation. The SEV Guidelines provides a tool for making these decisions for stream and river reaches.

Table 5

The main ecological functions and their relative importance for Albany streams as assessed by the team of experts (from Rowe et al. 2008).

Key ecological functions	Overall relative importance in Albany streams
Hydraulic functions	
1. Natural flow regime maintained	High
2. Connectivity to flood-plain retained	Medium
3. Connectivity for species migrations exists	Low
4. Connectivity to groundwater maintained	Low
Biogeochemical functions	
5. Water temperature control maintained	High
6. Dissolved oxygen levels maintained	Medium
7. Organic matter input maintained	Medium
8. In-stream particle retention maintained	Medium
9. De-contamination of pollutants maintained	Medium
10. Flood-plain particle retention maintained	Low
Habitat provision functions	
11. Fish spawning habitat intact	High
12. Habitat for aquatic fauna intact	High
Biotic provision functions	
13. Fish fauna is intact	High
14. Invertebrate fauna is intact	High
15. Aquatic biodiversity intact	High
16 Riparian vegetation intact	High

Papakura streams: The SEV methodology was applied comprehensively to 20 reaches in Papakura streams (Phillips et al. 2006) in addition to measures of biotic intactness and chemistry. While many of the reaches were within urban land use, most had major upstream rural catchments. The reaches scored from SEV = 0.42 to 0.87, and the evaluation provided a wealth of detail on ecological functions. The methodology allowed the authors to comment on the implications of measured impacts on the structural (fish and species diversity) as well as functional components of streams. This allowed identification of potential cause-effect relationships and levels of degradation (Tables 6, 3.3). Consequently, the most appropriate and effective mitigation or restoration options could be recommended for a given reach or the whole stream (Tables 6, 3.3). The study demonstrated that the SEV methodology is a very powerful tool in the classification and management of urban and rural streams.

Waitakere streams: The SEV methodology was also applied to Waitakere City streams. Data had been collected based on the ARC's stream classification system (ARC 2004a) and on the Council's stream asset evaluation (Morphum Environmental 2006a,b, Storey et al 2007, 2008). The SEV methodology could only be partially evaluated because the surveys did not include the full suite of ecological functions (Storey 2007). The analysis showed that generally, urban streams scored more poorly than rural or native bush streams, but there was no statistically significant difference, possibly because of the limited evaluation.

Figure 14

Habitat provision and natural biodiversity functions of small streams.

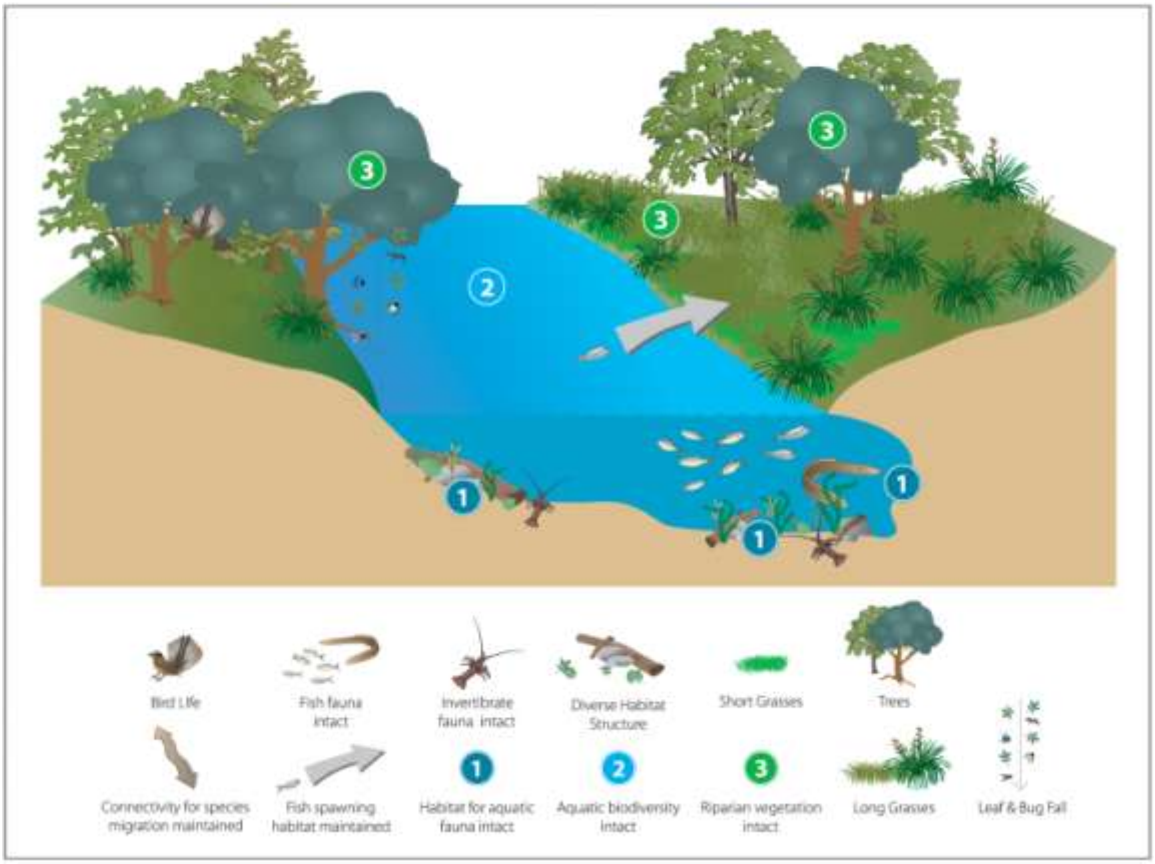


Table 6

Summary of management options suggested for the Papakura stream and the estimated ecological and community benefit (from Phillips et al. 2006).

Action	Ecological benefit	Community benefit
10-15 m wide buffer (each side) of native vegetation	High	High
Stream side planting unit of *Carex* sp. sedges	Med	Med
Row of tall trees for shade and bank stabilisation	Med	Med
Replace willows in channel with cabbage trees and *Carex* sp.	Med	Med
Walkway with stream view points	Nil	High
Repair sewage overflows	High	High
Replace weed spraying with shade planting	High	Med
Reinstate meanders in straightened channels (Site 2)	High	High
Minimise stormwater run-off in new and existing suburbs	High	High
Remove fish barriers	High	Med
Enhance flood plains and create spawning areas	High	Med
Fence stock out of streams in rural areas	High	High
Enhance parks	Med	High

4.5 Trends in water quality in urban streams

An update on the Regional Rivers and Stream Monitoring Programme

Since 2005, the long-term ARC's monitoring datasets for streams and rivers have been rationalised into the Rivers & Streams Water Quality Programme, which now encompasses 27 individual sites spanning a range of land uses and disturbance regimes (ARC 2007) (Figure 15). The ARC regional stream monitoring programmes have, since the mid-1980s, documented the quality of these streams, 17 of which are significantly impacted by urbanisation.

As summarised in the 2005 Review, 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 established and developing urban areas. Apart from one stream (Oakley Creek), all of 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 shows cumulative impacts from a range of potential sources rather than those associated purely with urban run-off.

By 2000, monitoring since around 1987 had not revealed any significant changes in urban stream water quality, apart from a rise in temperature, which was affecting all monitored streams. In 2007, a detailed analysis of 25 sites from the Rivers and Streams Water Quality Programme was carried out (Scarsbrook 2007). Summaries of state (annual medians) and analyses of long-term trends (Seasonal Kendall tests) in water quality were carried out on this dataset, both for the entire period of record (1986-2005) and for more recent ten years (1995–2005). Relationships between land use and water quality state and trends were also assessed.

Regional scale trends of warming water temperatures, and decreasing faecal bacteria, nutrient (NO_3+NO_2, DRP and TP) and suspended sediments (SS & Turbidity) were observed. Decreases in oxidised nitrogen is a positive sign, although it should be noted that concentrations still regularly exceed ANZECC (2000) guidelines for lowland streams (ie, 0.444 mg L^{-1}).

Figure 15

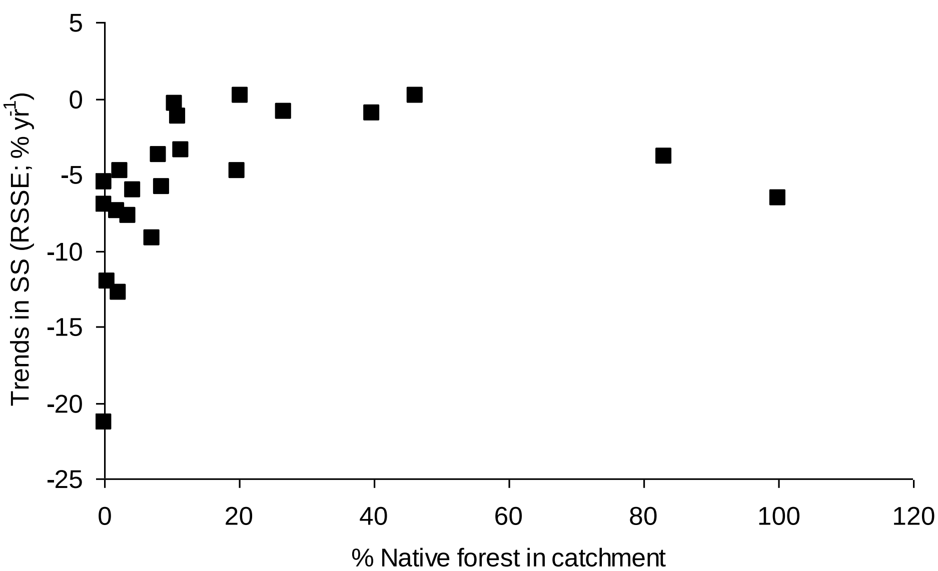
The 27 monitoring sites of the Rivers & Streams Water Quality Programme (ARC 2007).

This decrease is a regional trend and also occurs in most rural streams. A decreasing trend in concentrations of faecal coliforms is apparent at a number of other sites, and should therefore be considered as very positive. However, despite this downward trend, faecal coliforms are still relatively high at many of the urban sites, and are usually above guidelines for safe human contact.

A number of these trends were correlated with land use characteristics. For example, the magnitude of decreasing trends in suspended sediment concentrations were greatest at highly modified urban sites (Figure 16).

Figure 16

Scatterplot of trends in SS (expressed as Relative Sen Slope Estimates; RSSE) on % native forest & scrub for 23 monitoring sites (from Scarsbrook 2007).



Overall, the significant improvements in faecal coliform, suspended sediment and nutrient concentrations are a positive signal for the region, given that these parameters are major contributors to poor water quality in Auckland streams, and urban streams in particular. The report notes that the interpretation of the data (cause and effects) is limited by a lack of information on catchment pressures that relate to water quality. Therefore it was only able to say that changes have occurred, but notes that the precise reasons for these are still largely unknown. A number of reasons can be offered for these changes, including the maturity of the urban area (eg, time since developed from rural land use or when septic tanks were used in the catchment, time since major soil disturbance), streambank protection, exhaustion of stored sediment in streams, increase in imperviousness area, and reduction in sewer leaks and waste water overflows. However, as noted in the report, the quantitative linkage between catchment drivers/pressures and in-stream effect measures has not yet been made.

4.6 Toxicity of high suspended solids on aquatic life

The impact of poor or failed sediment controls during urban development

Very high suspended sediment concentrations can be lethal to fish, but such exposures rarely occur in nature. During urban development, very high suspended solid concentrations can occur in streams if soil erosion prevention fails or is inadequate. After heavy rain, turbidity levels in some Auckland streams can be relatively high (>10,000 NTU[6]) and there is concern that these levels are acutely toxic to NZ fish species (Rowe et al. 2005). Chronic effects have been observed on NZ fish species (eg, avoidance behaviour) at much lower turbidity levels (<100 NTU) (reviewed in Rowe et al. 2005).

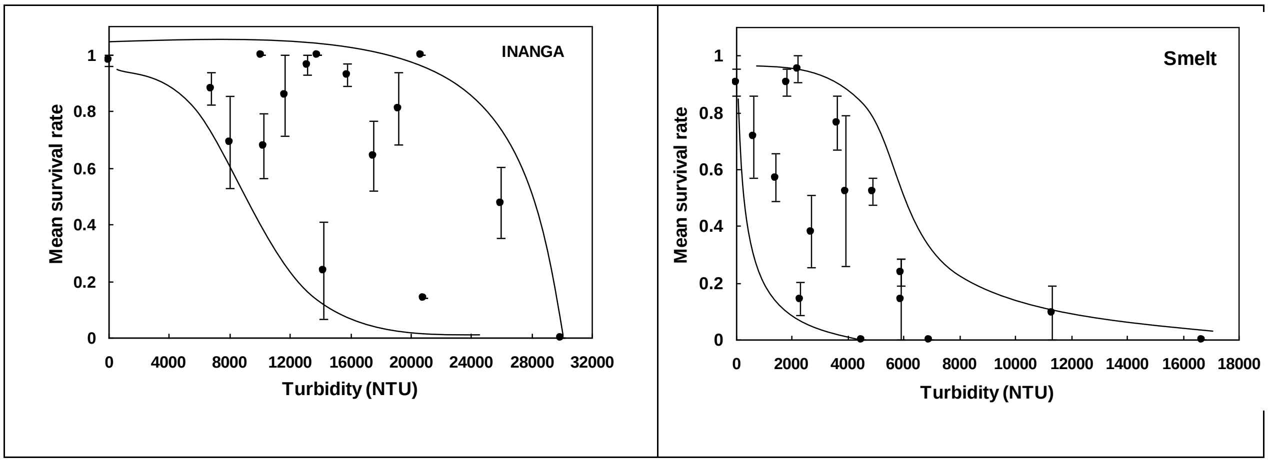
NIWA conducted laboratory studies where stream animals were exposed to high concentrations of clay (< 4 μm particles) for short periods of time (<24 hours) (Rowe et al. 2005). Exposure of the aquatic larval stage of 2 caddisflies, a damselfly and 2 mayfly species to high turbidity levels (up to c. 20,000 NTU) had no effect on their 24-hour survival. Similarly, the freshwater crayfish, adult redfin bullies and juvenile banded kokopu were not affected by turbidities of c. 20,000 NTU. As the highest turbidity levels recorded in Auckland streams are well below 20,000 NTU, it can be concluded that these species will not be directly affected by the high turbidities associated with flood flows.

The survival of inanga was reduced by turbidity levels ranging from 7000 NTU upwards (Figure 17), and 50 per cent of the animals were killed close to 20,000 NTU. This species survival is therefore unlikely to be greatly affected by the turbidity levels occurring naturally in Auckland streams.

[6] NTU = nephelometric turbidity units.

Figure 17

Mean survival rates (± 1 Standard Error) over 24 h for smelt and inanga at turbidity levels ranging up to c.15,000 and 30,000 NTU, respectively. Solid lines enclose the data points and illustrate the variability among the different groups of fish within a species, as well as the greater variability for inanga compared with smelt. (Reproduced from Rowe et al. (2005)).



Smelt were an order of magnitude more sensitive than inanga (Figure 17). Their survival was reduced at turbidity levels ranging from 700 NTU upward. Their LC50[7] was close to 3000 NTU and turbidities over 10,000 NTU resulted in 0 per cent survival over 24 hours. These data indicate that smelt survival could well be affected in streams when sediment holding ponds are breached and unusually high turbidities occur for relatively short periods. Repeat exposure of smelt to turbidities of 1000 NTU for four hours every two-to-three days over a period of 20 days did not affect their survival. These data establish a "safe" baseline for repeat exposures.

Therefore, with the exception of smelt, this study has largely eliminated short-term water column turbidity as a major, acutely-lethal stressor of fish and invertebrates in Auckland streams.

4.7 Effects of piping on the freshwater animals of Auckland streams

This work arose from the ARC's concern for the cumulative loss of Auckland streams through piping. The review (Moore 2004) indicated that piping not only replaces stream habitat but also interrupts the input of organic matter (the food basis of the stream) as well as the recruitment of invertebrates by downstream drift. However, valuable habitats can still remain in short reaches downstream of extensive piping, such as spawning habitat for inanga in the tidally affected reaches of small streams. The review also found that valuable fish and invertebrate communities could exist in short reaches (several 100 m) of bush

[7] LC50 = lethal concentration where 50 per cent of test animals die.

upstream of several hundred meters (or even kilometres) of piping. Banded kokopu can migrate though such lengths of pipe. The review provides many useful case examples as well as guidance for the management of urban streams.

Figure 18
An example of stormwater outfall piping.

4.8 Intergeneration responses to low level contamination in urban streams

New method of toxicity testing over several generations of stream animals

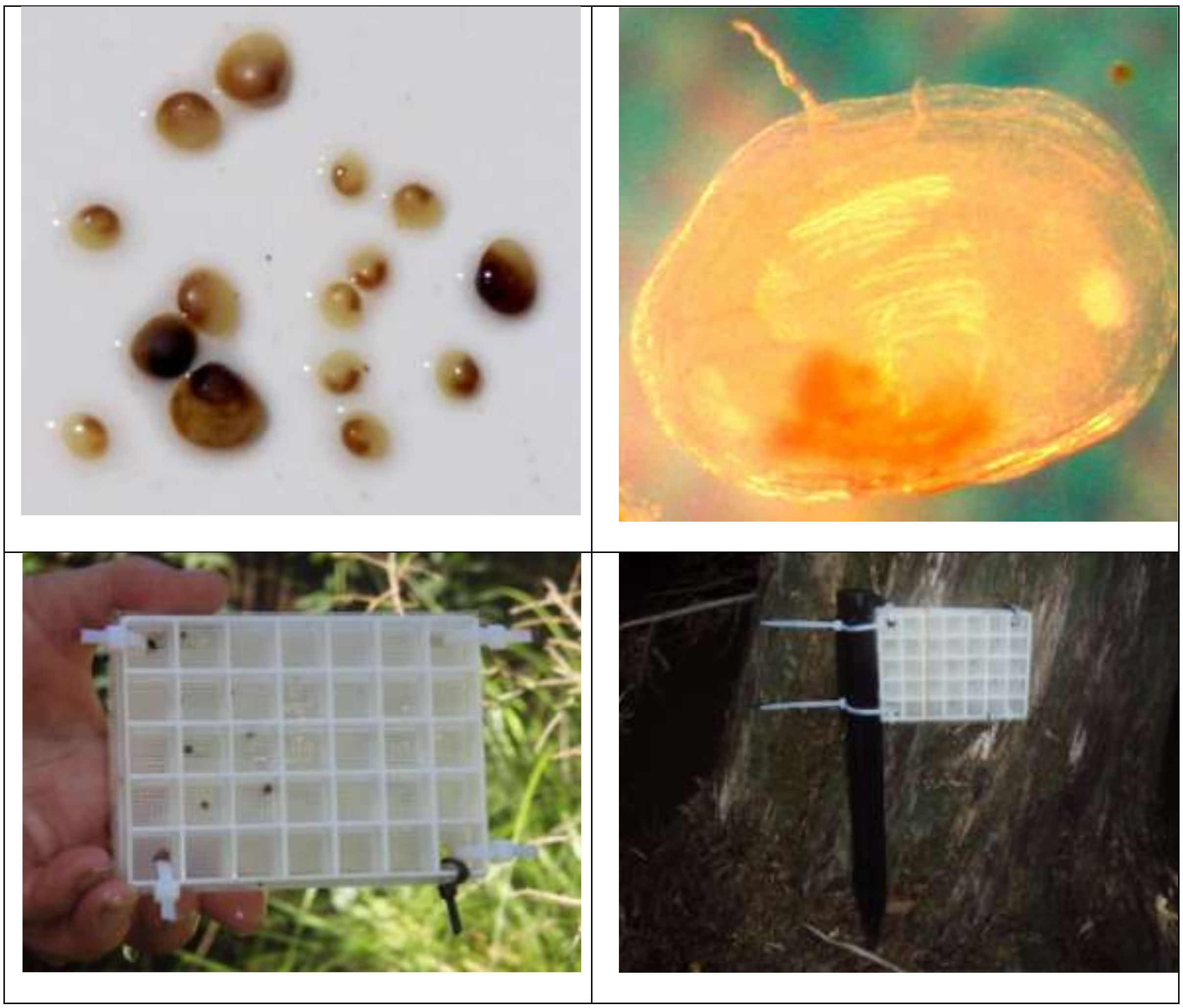
This work, commissioned by ARC from NIWA, investigates the multi-generational effects of stormwater contaminants on freshwater organisms (Phillips & Croker 2008). This new research direction addresses the complexity of the effects of low level stressors in urban streams. The technique employs changes in genetic structure within the freshwater clam, which can reflect both acute and chronic effects of contaminants over long exposure periods, involving multiple generations. In contrast, most toxicity tests usually employed in testing contamination involve only short-term exposure, and thus can only assess short-term effects. Effects examined in this study include selection against genetic variants, loss of genetic diversity (and resilience of populations), adaptation, as well as the usual toxicity endpoints of survival and chronic effects on animal health.

The native New Zealand clam (*Spahaerium novaezelandiae*) was chosen because its filtering and movement activities expose it to overlying water and sediment contamination. It also has a relatively rapid life cycle and can produce young at three months old (Figure 19). Freshwater clams were deployed in cages at a number of uncontaminated and contaminated sites (Figure 19). After adults were retrieved and assessed, they were maintained in clean or slightly contaminated water, until offspring were produced. Methodology is still being assessed and developed, but the exposure of off-spring occurred during 2008.

This research technique is under development and it is too soon to assess how successful it will be. However, if it is successful, it will provide a test that more realistically reflects ecological impacts of the low levels of multiple stressors found in urban streams, compared with short-term toxicity tests.

Figure 19

Upper: Adult clams and sphaerid juvenile. Lower: Sphaerid field deployment cage and deployment to stakes (shown above water level) (Phillips & Croker 2008).



4.9 Flow impacts

Quantifying the effect of urbanisation on the low flow regime of streams

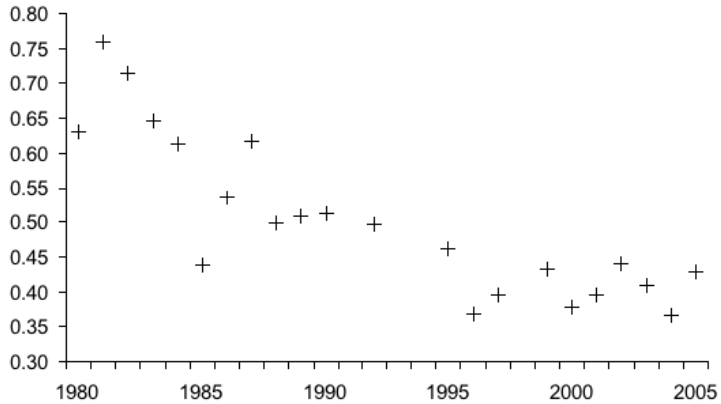
The impacts of stormwater on flows in streams and rivers were not covered in the 1995 – 2005 Review (ARC 2008), except for the effect of high flow on ecology. There had been little research in Auckland, but the impact of urbanisation on storm flows was well understood (ARC 1999). Hydraulic engineering was well advanced with routine and widespread application of stormflow models to estimate flows, pipe and channel size requirements, flooding risk and flood plain differentiation throughout Auckland. What was lacking was the effect of urbanisation on low flows.

An analysis of streamflow for 15 months for three catchments in Waitakere City revealed impacts of urbanisation on flows (Trowsdale 2005). The catchments were similar in size, geology and rainfall, but contained different amount of impermeable surface; 6 per cent (rural), 16 per cent (peri-urban) and 33 per cent (urban). There was a reduction in baseflow with urbanisation. This manifested in the baseflow regime; baseflow in the rural catchment was sustained above 1.9 $L.s^{-1}$ km^{-2}, but was 1 $L.s^{-1}$ km^{-2} in the peri-urban catchment. Flow ceased completely for 32 days in the urban catchment. Flows were greater in the rural catchment for 95 per cent of the time – corresponding to low flows. Flows were greater in the urban catchment 1.5 per cent of the time – corresponding to storm peak flows.

A more comprehensive analysis was carried out on Puhinui Catchment (South Auckland) and Oteha Catchment (North Shore City) (Davie et al. 2008). The proportion of urban land area increased 1 – 22 per cent in Puhinui from 1959 to 2001, and 1 – 49 per cent in Oteha from 1963 to 2006. There was little impact on the hydrology of the Puhinui stream, except that quickflow increased slightly. For Oteha there was a noticeable decrease in the volume of baseflow, while total and stormflow volumes increased (Figure 20). The percentage of annual rainfall that becomes stream flow between storm events decreased from 17 to 12 per cent. The reasons for differences were not understood but might be attributed to the differences in impermeable area, land use, geography, geology and the flow measurement methods. The study concluded it would not be possible to develop a regional-wide picture of the effect of urbanisation on low flows from only a few studies. The widely accepted notion of reduced low flows accompanying urbanisation is not necessarily proven in all situations.

Figure 20

The decrease of Base Flow Index (BFI) in the Oteha stream. BFI (total baseflow in a year divided by the total flow) is a measure of what proportion of the total flow is baseflow. There is a large decline in BFI between 1980 and 1996 from ~ 70 per cent to ~ 40 per cent. (Reproduced from Davie et al. 2006).



4.10 Conclusions

The ecological functions of streams have been clearly defined and methodology developed for their measurement. Along with measurements for biotic integrity, streams can be described in terms of structural (species diversity) and functional characteristics. Cause-effect relationships can be identified and extent of degradation quantified. This provides an objective basis for prioritisation, mitigation, remediation or environmental compensation of streams.

The review of the water quality data has confirmed earlier findings that urban streams have poorer water quality, but that this is improving over time for TSS, nitrate and faecal coliforms. While a number of explanations can be offered to explain these changes, the mechanisms are likely to be complex. This points to a need for better linkage between catchment drivers/pressures and in-stream effects, and this should be a focus for the future.

Freshwater toxicity studies have demonstrated that fish are unlikely to be adversely affected by SS and turbidity during normal stormwater discharges into Auckland streams. Smelt were found to be the most sensitive of the fish tested, and a “safe” threshold has been defined for their protection. Studies that have begun on intergenerational toxicity may, in future, provide a better understanding of the long-term ecological impacts of contaminated urban stream environments.

New studies on the effects of urbanisation on base or low flows showed variable results. Baseflows had been reduced in three catchments but not in one other. Not all the factors that control this phenomenon are understood, and it is not yet possible to derive a region-wide model for low flow impacts of urbanisation.

5 Impacts on Marine Water Quality

The ARC's marine water quality monitoring programme gives some indication of the effect of urban stormwater, but it is not specifically targeted at stormwater impacts, being undertaken during periods of low freshwater flows. General marine water quality is poorest at inner harbour or estuary sites subject to freshwater inflows from urban and rural areas. The improvements in stream quality described above (reductions in TSS, nitrate and faecal coliforms) for freshwater have been linked to slow and gradual improvements in marine quality at some sites. Some of the improvement may be due to conversion of rural land (high indicator bacteria, nutrients) to urban land (lower indicator bacteria, nutrients), and some to the reduction in wastewater overflow. Most importantly, despite the large increase in urbanisation since monitoring programs begun in 1987, the general water quality in the marine environment has not deteriorated and may even have improved slightly. As with general water quality of freshwaters, the cause and effect linkages are not well quantified and would benefit with PSR modelling.

New work has challenged some paradigms on stormwater in Auckland's marine environment. Zinc concentrations in water in the Whau Estuary (one of Auckland's most impacted estuaries) can be relatively high during low flow conditions and exceed water quality guidelines in saline waters. This is at odds with the previous views, which proposed that Zn (and Cu) concentrations would be low and below water quality guidelines in the water column in the wider marine receiving environment (see the 2005 Review, ARC 2008). This proposition was based on dilution calculations and actual measurement of Zn and Cu levels in Mangere Inlet. The new work proposes that Zn desorbs from particulate matter. High Zn concentrations were also measured in pore waters. The implications of these findings and propositions could be important, and therefore worth listing:

- They point to the significance of the dissolved phase for Zn fate, effects and transportation in the marine environment, while previous work has emphasized the particulate phase.
- As described above, Zn concentrations are high enough in the saline water column to exceed WQG, and hence potentially cause adverse effects on estuarine biota.
- Zn concentrations can also exceed WQG in pore water; which provides a mechanism for Zn toxicity in sediments.
- Mobilisation of Zn from sediments into the water column along with dispersal in tidal flushing could be a significant redistribution mechanism to other parts of the marine environment, including losses to the Hauraki Gulf and the Tasman Sea.

However, the Whau Estuary work is only one study, so further investigations are warranted, especially because the implications of the initial work are so far reaching.

Overall, advances in our understanding on trends in general marine water have been made, with general water quality improving despite the continued expansion of urban land use. There is new evidence for the greater importance of dissolved phase for the fate, effects and transport of Zn in marine receiving waters.

5.1 What was known in 2005

The 2005 Review explained that relatively few direct measurements have been made of the impact of stormwater on marine water quality. The exception is microbiological water quality, for which there is a large amount of data. This reflects the importance and public interest in human health risk, and the perceived less acute nature of other water quality issues. Much of what is assumed about water quality impacts relied on predicting concentrations in estuaries on the basis of average concentrations in stormwater, and assuming complete mixing in the marine receiving water.

Part of the reason for relatively little data is because the episodic and highly variable storm-derived inputs discharge to a wide range of receiving waters, from upper sheltered tidal creeks to open coastal water. Dilution and mixing with seawater is highly variable, depending on the nature of the receiving water, while turbulent mixing by winds, waves, and currents also resuspends bottom sediments, leading to lower water quality in shallow, near-shore areas of beaches and estuaries during storm events.

Because of these difficulties, the focus has been on more "integrative" assessment methods –sediment chemistry and biological monitoring using sediment-dwelling animals. Shellfish, both resident 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).

The state of knowledge by 2005 was:

- The ARC's SoE regional water quality monitoring program indicates that urban run-off, including sewer overflow, contributes to a broader-scale reduction in marine water quality in Auckland's harbours and estuaries, in particular, increased indicator bacterial levels. It shows that these effects decline markedly in the wider harbour and coastal zones due to dilution by cleaner oceanic waters and bacterial die-off.
- The ARC's shellfish contaminant monitoring programme showed that chemical contaminants, in particular organic compounds (such as PAH, OCPs, and PCBs) 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. Some long-term trends over time have been recorded, with most dramatic changes relating to changes in discharges associated with major point-sources (eg, chlordane discharges to the Manukau Harbour) and, in a more complex

way, to large scale long-term environmental cycles (eg, possibly the Southern Oscillation Index and El Nino-La Nina patterns).

- Data from short-term studies of wet- versus dry-weather water quality at several near-shore sites and routine bathing beach monitoring conducted by TLAs show that stormwater contamination restricts contact recreation at a significant number of sites around Auckland. Wastewater overflows are a major source of high-level contamination. 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). Severe effects are generally short-term (<24 hours) following rainfall and localised around stormwater outlets (ca.100 m).
- 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.
- There was little actual information on the influence of stormwater on the concentrations of heavy metals or other toxic contaminants in Auckland's marine waters. Predictions based on stormwater concentrations indicate that potential adverse effects on marine water quality are likely to be localised and short-lived due to dilution in the receiving waters.

5.2 New work 2005-2008

Apart from the ongoing regional water quality monitoring programs, there is little new information. New and ongoing work includes:

- Detailed analysis of the regional State of the Environment (SoE) programs have provided some interesting and encouraging information on trends in water quality (Section 5.3).
- Updating the shellfish monitoring programme (Section 5.4).
- A research study on heavy metals in the Whau Estuary has provided some interesting, but yet uncertain, insights on Zn fate in marine waters (Section 5.5).

5.3 Updating the Regional Saline Water Quality Monitoring program

A picture of marine general water quality across the region through time

Auckland Regional Council (ARC) operates a long-term water quality monitoring network at 27 sites located among the Region's harbours, estuaries and wider coastal zone. This monitoring network has produced New Zealand's most comprehensive long-term water quality dataset for coastal waters. Objectives of this network include SoE reporting, identification of major environmental issues, and assessment of the efficacy and efficiency of Council policy initiatives and strategies. While not a direct measure of urban stormwater impacts, this program provides some broad regional perspectives. The results of the program to 2005 were summarised in the 2005 Review (ARC 2008). However, recent trend analysis (Scarsbrook 2008) has uncovered some important improvements in regional water quality.

As described in the 2005 Review (ARC 2008), inner harbour sites tended to have poor water quality, whereas water quality in coastal or outer harbour sites was relatively good. This is hardly surprising as many of the parameters measured have much higher concentrations in the freshwater inputs than is found in seawater. Data analyses showed a strong overlap in temporal trends observed in streams and estuaries in the region, indicating that stream water quality is a major driver of water quality at inner harbour sites. The regional-scale trends observed in the present study mirror those observed in freshwater (see Section 4) (Scarsbrook 2007). Furthermore, temporal patterns of nitrate concentrations in Tamaki Estuary were strongly correlated with nitrate concentrations in two major tributaries (Otara Creek and Pakuranga Creek). These results, along with the observed relationship between salinity and water quality, indicates that stream water quality is a major driver of water quality at inner harbour sites.

The highest levels of faecal indicator bacteria were observed in the upper Waitemata and Mahurangi harbours, but these may be related to rural inputs. These elevated levels should be of concern for resource managers, because uses such as shellfish gathering and recreation are impacted by these high levels of faecal indicator bacteria. Relationships between salinity and both average water quality rank and the number of shellfish gathering guideline exceedences at a site supported the hypothesis that sites close to the influence of streams (ie, upper harbour) tended to have lower water quality. As described in the 2005 Review (ARC 2008), exceedances of bathing beach guidelines are much higher at inner harbour sheltered sites, than in more open, outer sites.

Figure 21

The 27 coastal water quality sites (Scarsbrook 2008).

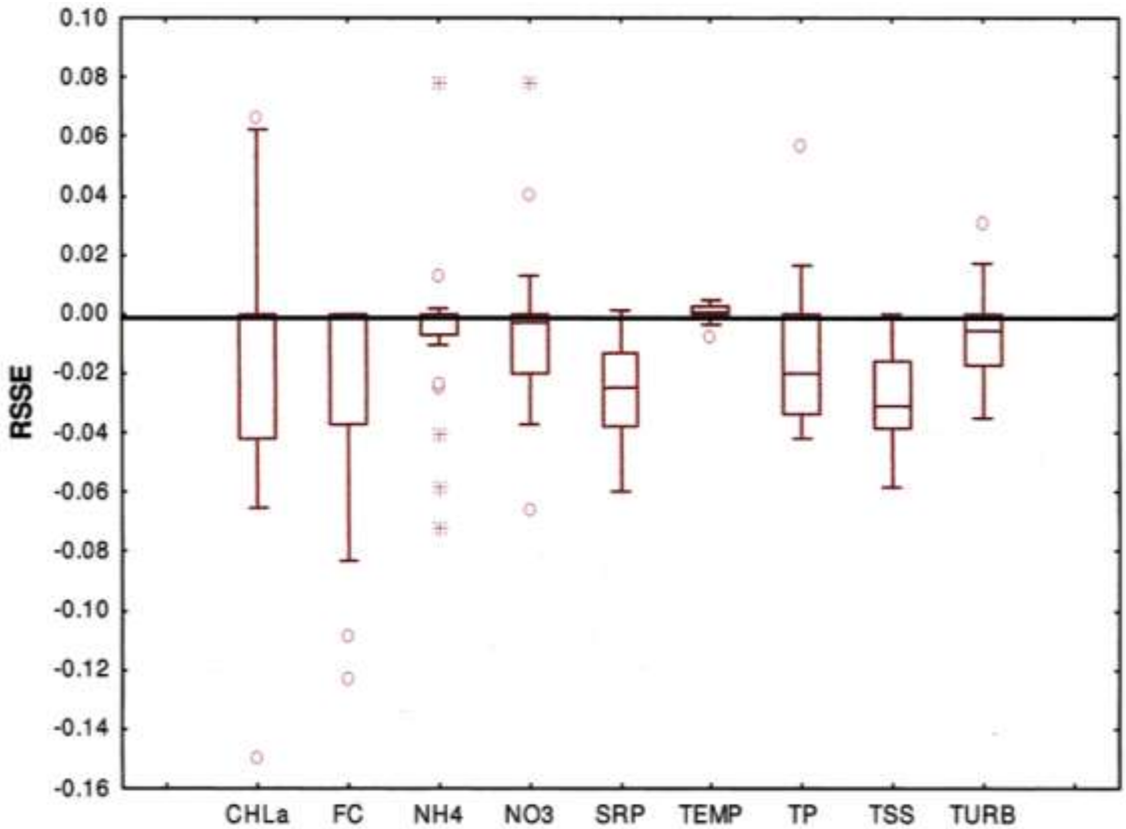
Climate variability, as measured by the Southern Oscillation Index (SOI) was also found to be strongly associated with temporal patterns of water quality, especially relating to temperature and nitrogen concentrations.

Across the region as a whole there were significant improving trends in the levels of faecal indicator bacteria, total suspended sediments, total phosphorus, soluble reactive phosphorus and nitrate. Figure 22 summarises trends at all 27 monitoring sites for nine water quality variables. With the exception of temperature, all variables show a distribution of trends with the centre of the distribution (ie, median) below zero. This implies regional-scale reducing trends. Note that the changes are very small in most places because monitoring sites are not near sources and monitoring takes place near high tide when dilution with marine waters is near maximum. Most of these improvements are consistent with decreased anthropogenic pressures, such as the changes in the Mangere Waste Water Treatment Plant (MWWTP) and sediment controls. Nevertheless, the actual causes of these changes are not well understood (except those related to the MWWTP), and point to the need for stronger linkages between pressures and the state of the receiving environments, as is the case with the Rivers monitoring program (Section 4, Scarsbrook 2007).

Overall, water quality is similar in urban and rural areas, except indicator bacteria may be higher in the rural situation.

Figure 22

Style Trends in water quality expressed as the relative Sen slope estimate (RSSE), shown as the proportional change relative to the median concentration of all the data. A positive RSSE value indicates that the parameter is increasing overall through the monitoring period, a negative value indicates a decrease. (Reproduced from Scarsbrook 2007). (see Glossary for explanation of parameters on X axis).



5.4 Updated results from the ARC Shellfish Contaminant Monitoring Programme

Monitoring water quality through sentinel shellfish

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).

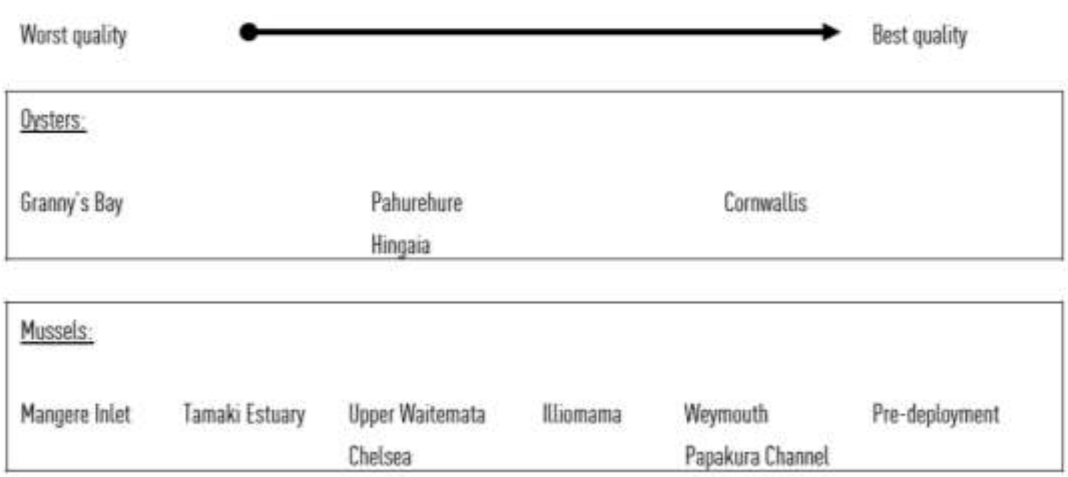
Data collected in the programme to 2005 was reviewed in 2007 (Kelly 2007a). Results were similar to those reported for previous years. In general, the levels of organic contaminants present in shellfish tissues were low by international standards, but clear differences were apparent between monitoring sites. Highest levels were detected in Mangere Inlet and Tamaki Estuary. DDT, chlordane, dieldrin and PCB levels were elevated in mussels and oysters from Mangere Inlet, and dieldrin and PCBs levels were elevated in mussels recovered from the Tamaki Estuary.

Marked changes in the concentrations of organic contaminants in shellfish have been observed through time. Significant declines have been recorded in the levels of lindane, chlordane and dieldrin in Manukau oyster tissues since these pesticides were deregistered in 1989-1990. However, recent pulses in DDT, chlordane, and PCB concentrations were observed in oysters and mussels from sites in Mangere Inlet. These pulses coincided with the decommissioning, and return to the sea, of treatment ponds at the Mangere Waste Water Treatment Plant.

Copper concentrations in Manukau Harbour oyster tissues in 2005 were relatively high by international standards, and are at levels considered to be indicative of contamination by human activity. Zinc concentrations in oyster tissues were within the “typical” range reported from international databases. Cyclical fluctuations in the concentrations of these two contaminants appear to be partly driven by natural variation in weather patterns, and in particular, those associated with the Southern Oscillation Index. However, further investigation is required to confirm this link and determine the causative factors.

Overall, relative site quality had not changed substantially since pre-2005, and can be ranked according to contaminant concentrations in oyster and mussel tissues as follows:

Figure 23
Rank of contaminant concentrations found in oyster and mussel tissues.



5.5 Trace metal cycling in the Whau River Estuary

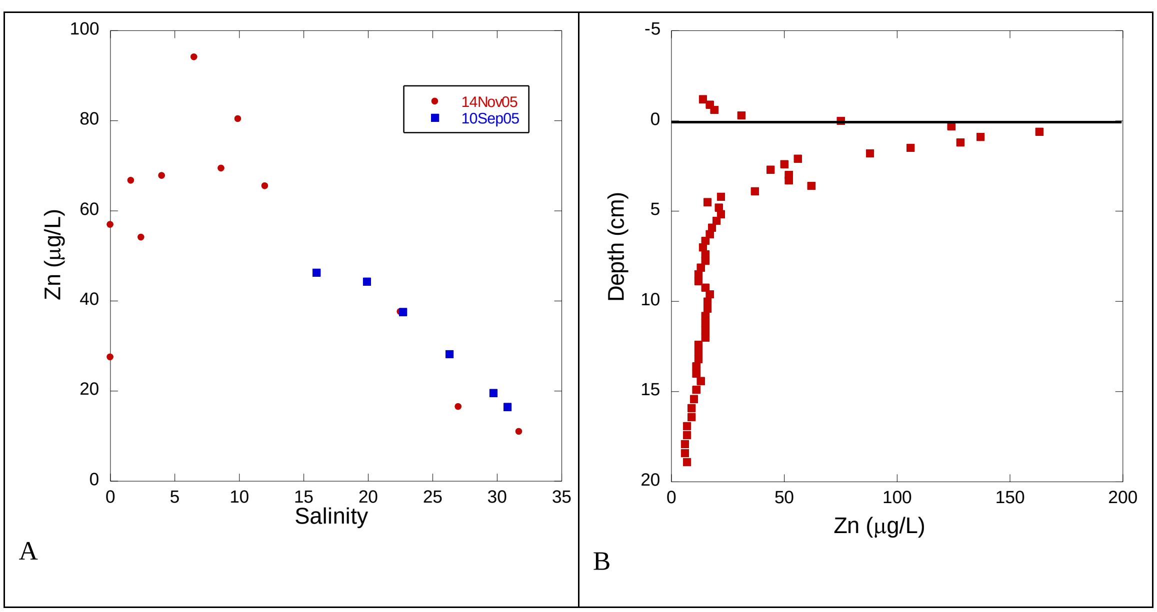
Evidence for relatively high concentrations of dissolved zinc in the water column and in pore water of a relatively contaminated estuary

Ellwood et al (2008) measured trace metal concentrations in the water, pore water and sediments in the Whau Estuary. This research study utilises state-of-the-art measurement of water and pore water. The surveys were carried out during low flow, and provide very interesting information on dissolved concentrations of Zn, Cu, Cd and Pb in the water column and in the pore water of the sediments.

Salinity increased as seawater mixed with the stream water down the estuary. Salinity was very low, ~0$^{\circ}/_{oo}$ at the stream outlet, but was similar to seawater near the mouth of the estuary (~32 $^{\circ}/_{oo}$). Dissolved Zn concentrations (ie filterable Zn) showed a maximum concentration at a salinity of about 10$^{\circ}/_{oo}$ and it was proposed that dissolved Zn was being released from particulate matter. The maximum concentration observed on one sampling occasion was 95 µg/L at 6 $^{\circ}/_{oo}$, and was about 10 µg/L at the mouth of the estuary (at 32 $^{\circ}/_{oo}$) (Figure 24a). Pore water concentrations in the surface layer of the sediments (4 cm) were relatively high, and ranged from ~10 µg/L up to 150 µg/L (Figure 24b).

Figure 24

Zinc concentrations in the (A) water column and (B) pore water in the Whau Estuary (adapted from Ellwood et al. 2008).



Note that Zn concentrations will probably exceed ANZECC guidelines at lower salinities. For example, the concentrations of dissolved Zn in urban streams is typically ~50 µg/L and 1-5 µg/L in oceanic waters. Therefore ANZECC guidelines may be exceeded in the less saline areas of estuaries (M. Timperley pers. comm., Table 7).

These results are very relevant to stormwater effects. They demonstrate that it is possible that dissolved Zn concentrations can exceed water quality criteria in saline waters and pore water, and offer a plausible mechanism for at least part of the observed effects of stormwater discharges in estuaries. The trigger values for protection of marine water species are listed in Table 7, for dissolved Zn in seawater (or pore water).

Table 7

ANZECC (2000) trigger guidelines for zinc.

Level of protection (number of species)	Trigger value (µg/L)
99%	7
95%	15
90%	23
80%	43

Note these are only trigger values (trigger = need for further investigations), but comparison with concentrations measured in the Whau Estuary suggest that further investigations are indeed warranted, at least in highly contaminated estuaries such as the Whau.

Concentrations of dissolved copper also exceeded Water Quality Guidelines (WQG) (ANZECC 2000), but that is not unusual; the WQG are close to background levels. Dissolved Cd and Pb were much lower than WQG.

The study utilised these results to estimate how much Zn was being lost to the water column from sediments ("cycled") and estimated that 71 per cent of total Zn could be lost from suspended sediments and deposited sediments. However, these estimates were based on preliminary estimates of Zn inputs to the estuary, which have since been substantially refined and updated (Timperley & Reed 2008b). They are also based on measurements made during dry weather, and utilised theory developed in large river estuaries. The measurements may not reflect the situation during stormwater flows, when there are much higher freshwater flows and suspended sediment concentrations. Also theories developed in large estuaries with large freshwater flows (such as the Amazon river) need to be validated in small estuaries such as the Whau. Irrespective of arguments about how applicable the mechanisms proposed in this study are to understanding the fate of all Zn discharged to the estuary, it provided key measurements and represents one of the most important advances in understanding. Current models used to predict the fate of Zn and Cu (Section 8 – Predictions) do not explicitly include cycling of Zn between the sediments and water (although the models do include a general mechanism for loss which could include this process). Future modelling should explicitly include these processes. The 2005 Review (ARC 2008) proposed that concentrations of dissolved Zn and Cu would be rapidly diluted and would fall below water quality guidelines that protect aquatic life. This study on the Whau Estuary, and the calculations above, call into question that assumption, and further work is warranted.

5.6 Conclusions

Analysis of the status and trends of the regional SoE programs have provided some interesting and encouraging information on trends in regional water quality. Note that the changes are very small in most places because monitoring sites are not near sources and monitoring is conducted during low freshwater flows. Further work is need to develop clear links between pressures, the state of the environment and management (pressure-state-response modelling) in order to establish causal links between water quality and what is happening on the land. A better understanding of the effects of broad-scale natural factors, for example the Southern Oscillation Index (SOI), on water (and shellfish) quality would also be useful to aid data interpretation.

While not a direct measure of urban stormwater impacts, this program provides some broad regional perspectives. Water quality in inner harbour areas is considerably poorer than in open coastal sites. More importantly, inner harbour water quality is similar in urban and rural areas, except indicator bacteria may be higher in the rural situation. The ARC's shellfish contaminant monitoring programme showed that chemical contaminants in shellfish, in particular organic compounds (such as PAH, OCPs, and PCBs) have not changed substantially since pre-2005.

A new research study has established that Zn concentrations in overlying water and pore water can exceed water quality trigger values to protect aquatic life. It also showed the possibility of significant Zn release from suspended sediment and estuarine bed sediments. Verifying and extending this work should be a high priority for further study to help understand potential toxicological implications, and to improve modelling in order to understanding the ultimate fate and build-up of contamination in Auckland estuaries.

6 Chemical Contamination of Marine Sediments

Sediment contamination (and the consequential 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 measures of impacts of urban stormwater in the Auckland region. Sediments and associated contaminants settle out of the water in the sheltered areas of estuaries and harbours. Muddy sediments build up over time, and so do contaminant concentrations. At some point, the contaminant concentrations could reach levels that are toxic to the aquatic life that lives in the sediments. This has the potential to affect the ecology of immediate areas, and beyond, because sediment-dwelling (benthic) organisms serve as key food sources for animals further up the food chain (eg, fish, birds).

A large number of new measurements have been made of heavy metal concentrations in marine sediments in existing programmes (SoE, RDP) and in new studies. The understanding of spatial patterns has not changed significantly from those described in the 2005 Review (ARC 2008), except that there is much better detail in Outer Zones and in areas which are undergoing urbanisation (Upper Waitemata Harbour, South East Manukau Harbour).

The continuing assessment of trends in heavy metal concentrations has uncovered some significant changes from previous findings summarised in the 2005 Review. The previously described linear trends have not been maintained and trend profiles have become more complex. Nevertheless, substantial increases in Zn (and to a lesser degree Cu) have occurred in many urbanised estuaries over the past decade. New developments in contaminant load and receiving environment response modelling (eg, CLM and USC3) may be useful in gaining a better understanding of the monitoring data, and will provide insight into future trends (See Section 8).

A very recent survey of chemicals of potential environmental concern (CPEC) in marine sediments is an important step in developing an understanding of the importance of these chemicals in Auckland's marine environment. It is too early to assess the significance of these initial data.

Overall, the picture of the distribution of contaminants in the marine environment that emerged from studies described in the 2005 Review has been "filled in" with more detail. Advances have been made on understanding other factors that control levels of contamination (besides proximity to sources), such as sediment chemistry and texture (particle size) and hydrodynamic forces (tides, currents, waves – see Section 8) and sediment supply (also summarised later in Section 8). The simple picture of linear increases in Zn and Cu and decreases in Pb obtained from early trend monitoring (as described in the 2005 Review, ARC 2008) has not prevailed, and the recent data show that changes over time are more variable than this. Reasons are, as yet, unknown and are currently being investigated.

6.1 What was known in 2005

By 2005 there had been a great deal of work had been carried out on sediment quality. The development of systematic monitoring protocols and their application across the region led to the acquisition of a very large chemical contaminant database, covering some 80+ sites. These data show that:

- Urban stormwater is contaminating Auckland's urban estuaries with heavy metals and, at lower levels, a range of persistent organic pollutants (POPs).
- Over 50 per cent of regional monitoring sites are contaminated to the point where aquatic life may be beginning to be adversely affected (ranked amber or red under Environmental Response Criteria (ERC)[8]). Worst affected sites are in muddy estuaries receiving run-off from older, fully urbanised catchments, particularly in the Tamaki and Whau Estuaries.
- Zinc is the contaminant of most concern at present. It has the greatest proportion of ERC exceedances.
- Persistent organic pollutants, such as PAH, OCPs, and PCBs are widely found, but concentrations are low (ie, 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.
- Zn is increasing over time at most urban sites, with smaller increases in Cu, and uncertain trends for Pb and PAH.

Another key advance in the period to 2005 was the development of models to predict contaminant accumulation and distribution as a result of urban development. These models were applied to sheltered depositional tidal creek environments (eg, Pakuranga, Hellyers and Lucas Creeks) and, at a larger more complex scale, to the Upper Waitemata Harbour.

[8] These are explained in the "Abbreviations" section..

6.2 New work 2005-2008

Since 2005, several lines of enquiry have broadened our understanding of sediment contamination. These fall under four main categories:

1. Update on sediment contamination trends from temporal monitoring at State of Environment sites (Section 6.3).
2. Detailed sampling of bed sediments in receiving waters not well represented in the regional database, especially in the Outer Zones (Section 6.4 and 6.5).
3. A greater understanding of factors controlling heavy metal concentrations in sediments (Section 6.6).
4. Emerging potential contaminants of concern (Section 6.7).

6.3 Regional distribution and trends in major contaminants

Updating the State of Environment and Regional Discharges Monitoring programs

The results of the State of Environment and Regional Discharges Project monitoring programmes are currently being undertaken at the time of preparing this report (M. Stewart, ARC, pers. comm). The major findings of these updated evaluations are described briefly below. The spatial distribution for sediment contaminants has been well described in the 2005 Review (ARC 2008) and only a brief description is given here for convenience. The Auckland Regional Council has been monitoring the regional effects of urban stormwater discharges at 78 sites (27 from 1998, the rest from 2002) (Figure 5.1, Kelly 2007b). The contaminants monitored are copper, lead, zinc, and polycyclic aromatic hydrocarbon (PAH) in estuary sediments. Individual sites are monitored at 2- or 5-year intervals, depending on the concentrations of these key contaminants. Highest concentrations of copper, lead and zinc continue to be observed from estuary sites adjoining the older urban catchments of Waitakere, Auckland, and Manukau Cities, ie Henderson Creek to Coxs Bay along the southern shores of the Waitemata Harbour, upper reaches and side-branches of Tamaki Estuary, and Mangere Inlet.

With the exception of Mangere Inlet, the concentrations of copper, lead and zinc were below threshold effect levels (ie TEL sediment quality guideline values) in the Manukau Harbour, and the Orewa and Weiti estuaries (Kelly 2007b).

Copper concentrations were slightly above TEL values at a number of Upper Waitemata Harbour sites. However, with the exception of Hellyers Creek, lead and zinc concentrations were below TEL values.

A strong relationship was apparent between copper, lead and zinc concentrations and benthic community structure, indicating that current levels of contamination (or covariates of copper, lead and zinc) are affecting the ecological function of urban estuaries (See

Section 9 – Ecology). In general, the spatial pattern of ecological condition reflected levels of contamination, ie proximity to urban development (Kelly 2007b).

Time trends at the State of Environment (SoE) monitoring sites are undergoing a major update, as described earlier (ARC unpublished results). Data is now available for the period 1998 to 2007. Trend profiles for concentrations in silt vary considerably between sites, and seldom approximate simple linear changes over time. Extractable Cu and Zn concentrations have increased over time at most sites, and Pb has decreased slightly. PAH concentrations in <500 μm fraction have changed very little. The greatest change has been for Zn, which on average has increased by approximately 30 mg/kg over the 9-year monitoring period. These findings are consistent with expectations, given the key sources of these contaminants in urban catchments.

The most common trend profile showed a "plateau" in metals' concentrations after approximately 2003 (Figure 26, ARC unpublished results). This "levelling off" of metals' concentrations may have important implications for future trends in sediment contaminants (and how they are measured), and the causes need to be investigated.

As a consequence of the plateau trend profiles, trends for the first few years of the monitoring programme are quite different from those measured in the last few years. Trends over the past few years are substantially smaller than those recorded earlier in the programme, and some have reversed direction. These results are difficult to explain from our current knowledge of urban run-off and estuarine sediment processes

It is possible that the changing trend profiles are a consequence of oversampling causing site disturbance, rather than a reflection of real environmental changes relating to land use. This must be clarified before continuing with the monitoring programme. The recent decline in the rate of Zn and Cu increases may also reflect short-to-medium-term variations around a longer-term generally increasing trend. Reasons for the differences in trend profiles and rates at different locations around the region have yet to be determined, but have been identified as a priority for future study.

Figure 25

The SoE and RDP sites in the Auckland region. (Source: ARC.)

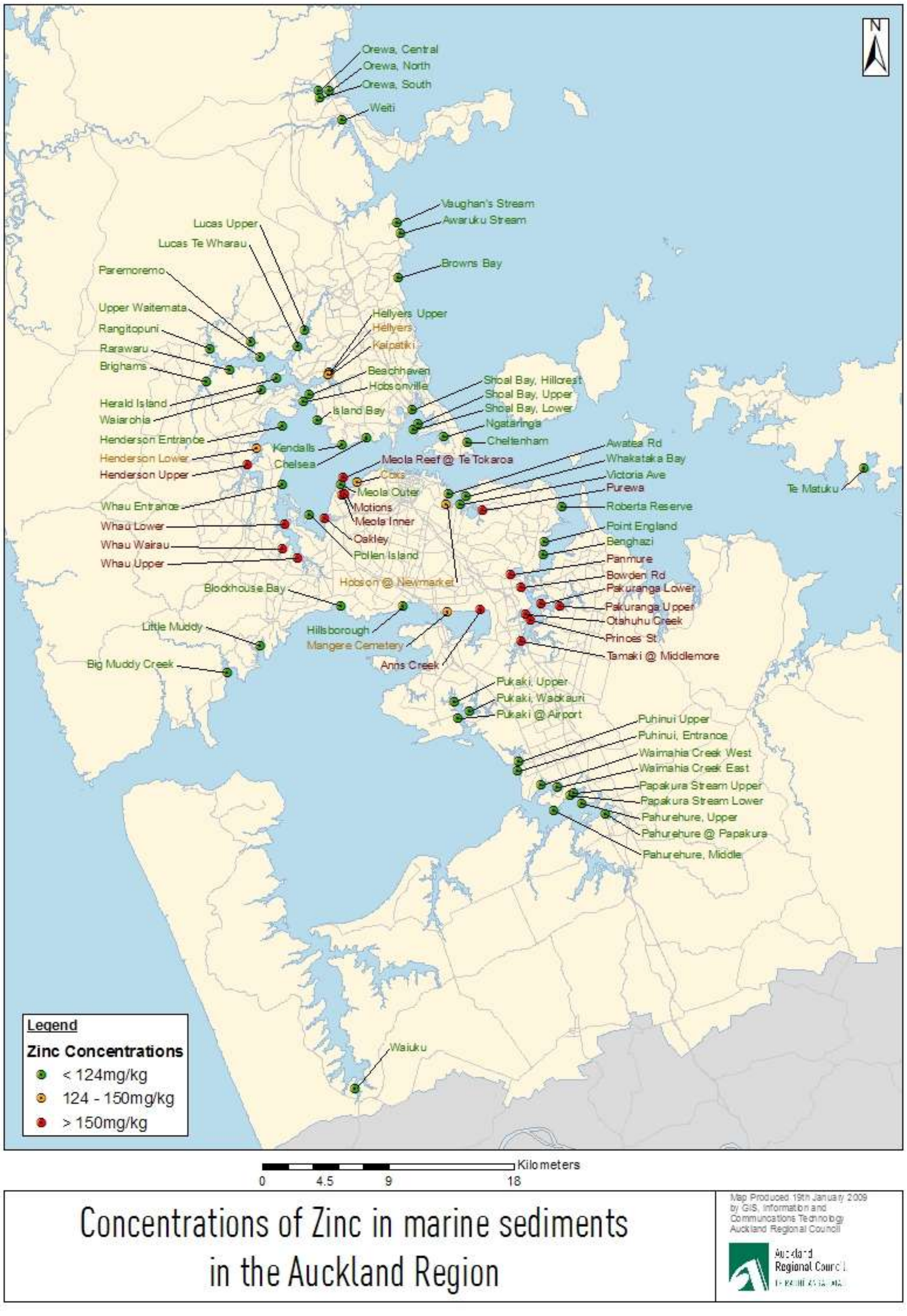
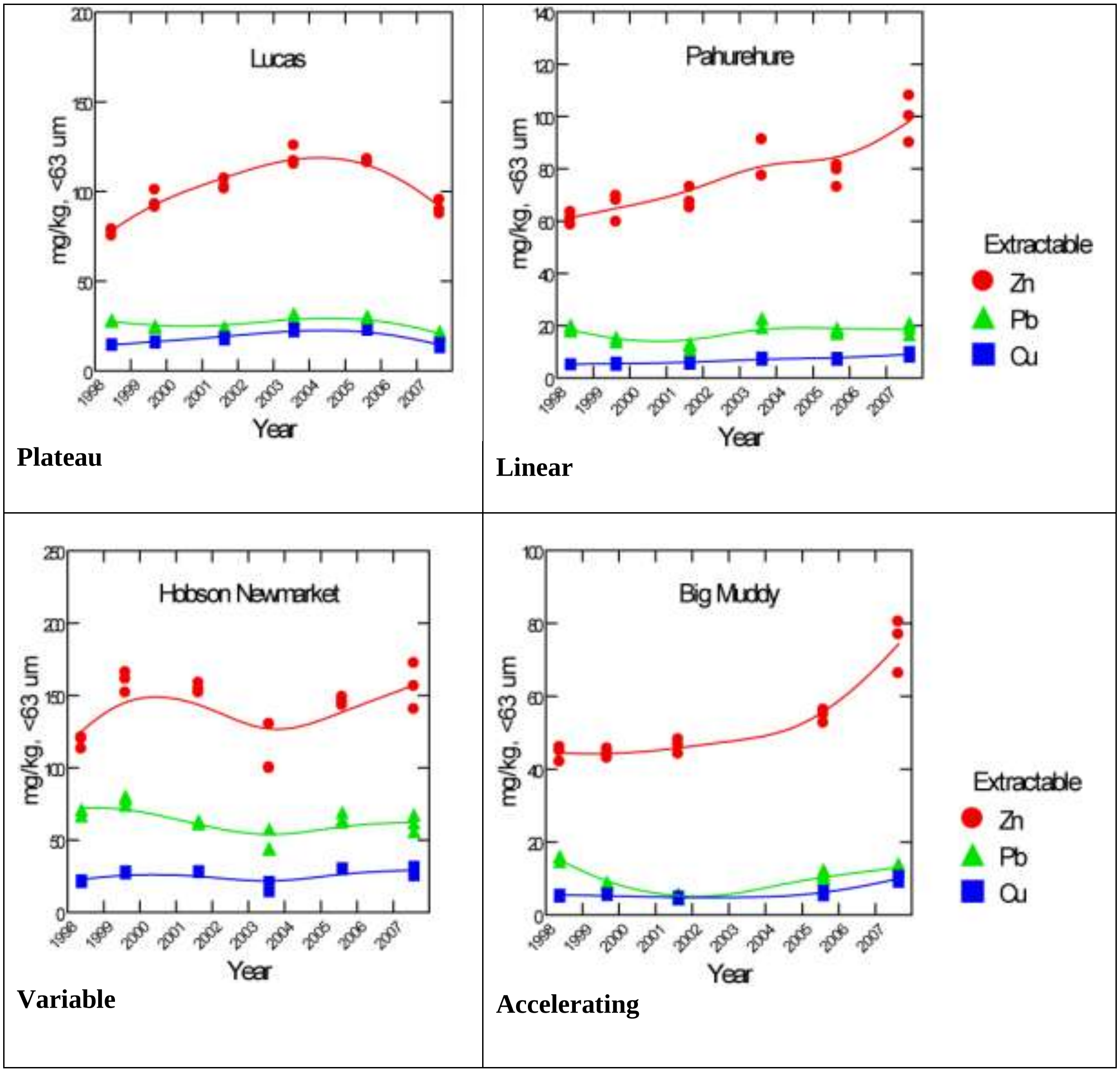


Figure 26

Examples of trend profile shapes for extractable zinc (<63 µm fraction) for the 1998–2007 period. Trend lines are smoothed by Distance Weighted Least Squares (DWLS). (Source ARC.)



6.4 Trace metal concentrations in Central Waitemata Harbour

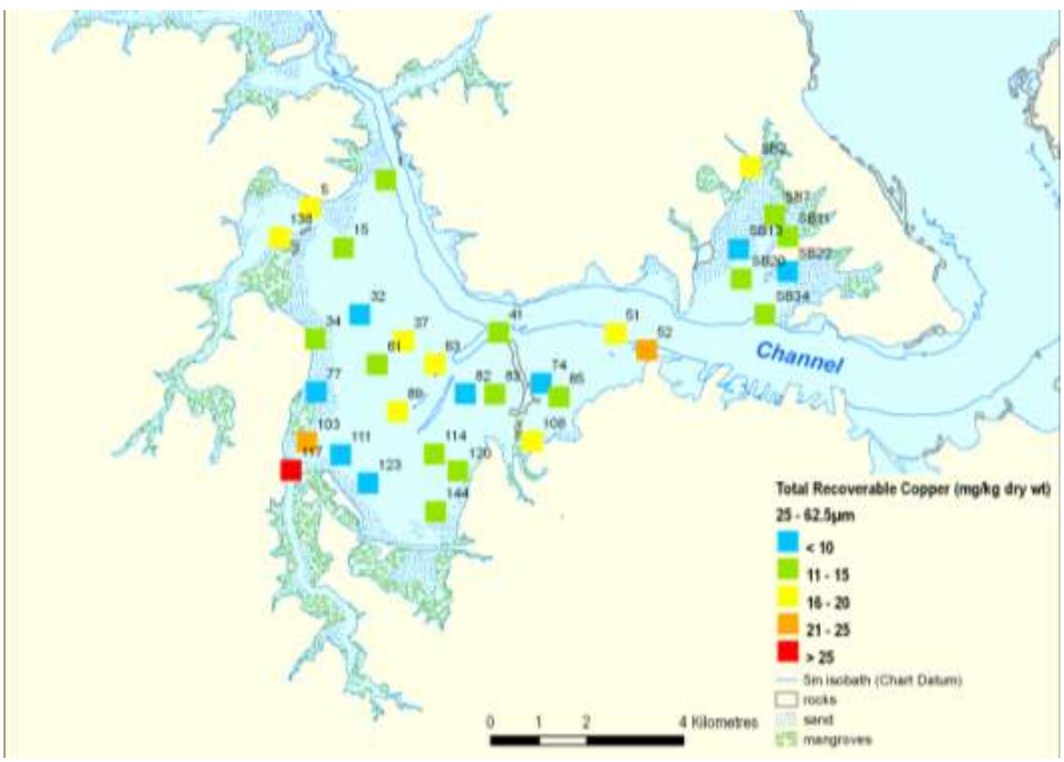
Understanding concentration distributions in the harbour environment

In 2005, while there is good regional coverage by monitoring sites, many of the sites were near sources (major urban stormwater discharges) and in sheltered muddy areas where contaminants can accumulate. Some receiving waters, such as high-energy environments, were under-represented. The Central Waitemata Harbour Study (Ahrens et al. 2008) collected many sediment samples to provide calibration data for predictive modelling (see Section 8 – Predictions) and to obtain more detailed information on the variability of heavy metals in this major harbour area. The study also provided some insights into the factors (other than proximity to sources) that influence heavy metal concentrations.

Trace metal concentrations were measured in different grain size fractions: clay, silt and sand, where clay was defined as < 25 μm, silt 25 – 63 μm and sand 63 – 250 μm. Measurements of total iron and total organic carbon (TOC) are good estimates for geochemical phases that bind these metals in the sediments, ie, hydrous iron oxides, total iron (Fe), and organic matter. Surface sediment samples (33) and sediment core profiles were collected. The particle size distribution and water content of the sediment was also measured, tidal height noted, thus producing a very comprehensive dataset to interpret and understand the fate and transport of the Zn and Cu. An example of the comprehensive dataset is shown in Figure 27.

Figure 27

Copper concentrations in CWH surface sediments, in "silt" fraction (25-62.5 μm) (Ahrens et al. 2008).



Overall there was an approximately 2-3 fold enrichment of Cu and Zn in Central Harbour surface sediments compared with reference sites and deeper sediment layers. In the cores, concentrations increased fastest after 1950, coincident with the onset of rapid urbanisation – this is consistent with other studies (see Section 7 – History). Not surprisingly, concentrations were higher nearer sources such as stormwater outfalls and tidal creeks. Normalising Zn or Cu concentrations to the amount of TOC or Fe in the sample generally reduced variability within grain size fractions, and clarified information on spatial gradients. Zn/Cu ratios were similar for the clay and silt grain size fractions throughout the central harbour and down the core profiles. Ratios were two-fold higher in sands, which the authors ascribed to the fact that sands have lower proportions of surface-absorbed meals relative to primary mineral metals, than have silts and clays. Therefore, ratios for sands reflect more of the primary mineral contribution. The study showed that in the Central Harbour, there were concentrations gradients with distance from sources, but sediment characteristics such as grain size and sediment geochemical phases also controlled trace metal concentrations.

6.5 Sediment contamination in other estuaries

Expanding and filling in the detail on sediment contamination

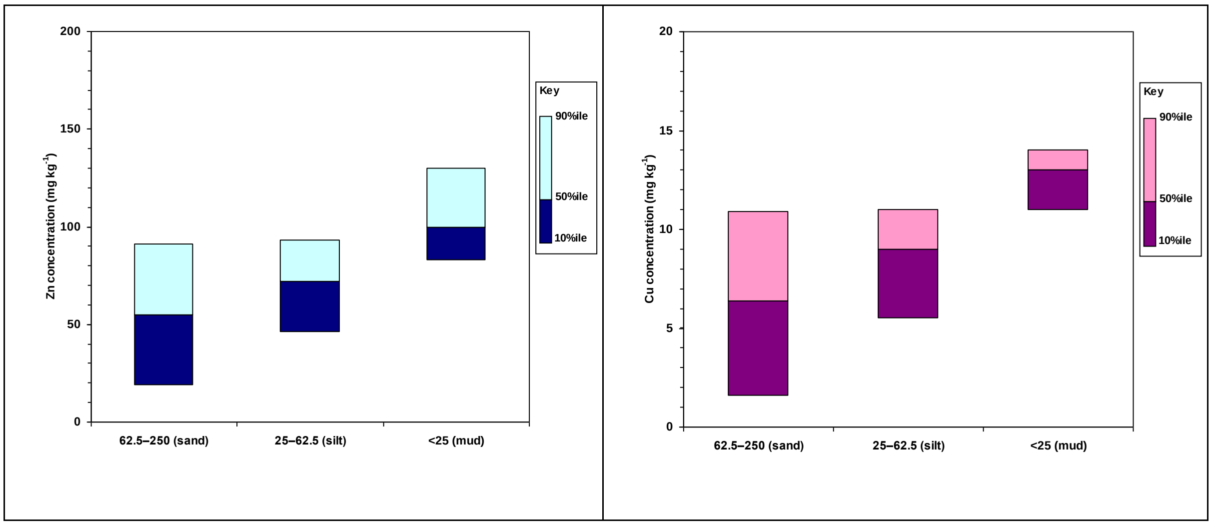
A number of other studies examined the variation in contaminant concentrations and sediment characteristics. While not revealing anything new about stormwater impacts, they do provide further data on the nature of Auckland estuarine sediments, contaminant distribution and conceptual understanding of the factors controlling contaminant distribution in specific estuarine areas.

South East Manukau and Pahurehure Inlet

Surface samples from 33 sites (see Figure 29) were analysed for particle size, bulk density, and copper and zinc concentrations (Reed et al 2008). This study provided calibration information for predictive modelling in this area (See Section 8 – Predictions). The surface sediments had moderate levels of copper and zinc contamination (2–3 times higher than the estimated background levels), with zinc contamination being more widespread. Concentrations were generally higher in the northern sites (closer to urban areas) than the southern sites (closer to rural areas). The muddiest sediments, close to the State Highway 1 bridge and developed urban centres, had the highest copper and zinc concentrations. The ranges of concentrations of Zn and Cu in different particle sizes found in the SE Manukau are shown in Figure 28.

Figure 28

Concentration ranges for Zn and Cu in different particle sizes at 63 sites in the South East Manukau (from Reed et al. 2008).



Tamaki Estuary

Abrahim et al. (2007) examined the spatial distribution of Zn, Cu Cd and Pb throughout the Tamaki Estuary at 36 sites (Figure 30). They examined acid extractable metals in the bulk (total) and fine fraction (< 63 µm). The study concluded that concentrations reflected catchment history, the estuaries geomorphic shape with a mid estuary constriction, sediment texture and mineralogy.

Rangitopuni Estuary (Upper Waitemata Harbour)

Hayward et al. (2004) examined the spatial distribution of Zn, Cu, Cd and Pb throughout the Upper Waitemata Harbour from Rangitopuni river to Hobsonville covering 22 sites (Figure 31). This was effectively a longitudinal survey of the Rangitopuni river and estuary. They determined metal concentrations in the fine fraction (< 63 µm) using XRF[9] analysis. The longitudinal survey found elevated concentrations right throughout the estuary, but concentrations were lower in the river sediment upstream, suggesting that the source was not the river but the Central Waitemata Harbour. The study concluded that increases in Zn, Cu and Pb (see Section 7 – History) found in deep cores in the estuary which seemed to occur around an inferred age of 1955, was due to inputs from the Central Waitemata Harbour.

[9] XRF = XRay Fluorescence is an element analysis method which is different to those used to produce most of the results in this report.

Figure 29

Location of sampling sites in South East Manukau (adapted from Reed et al. 2008).

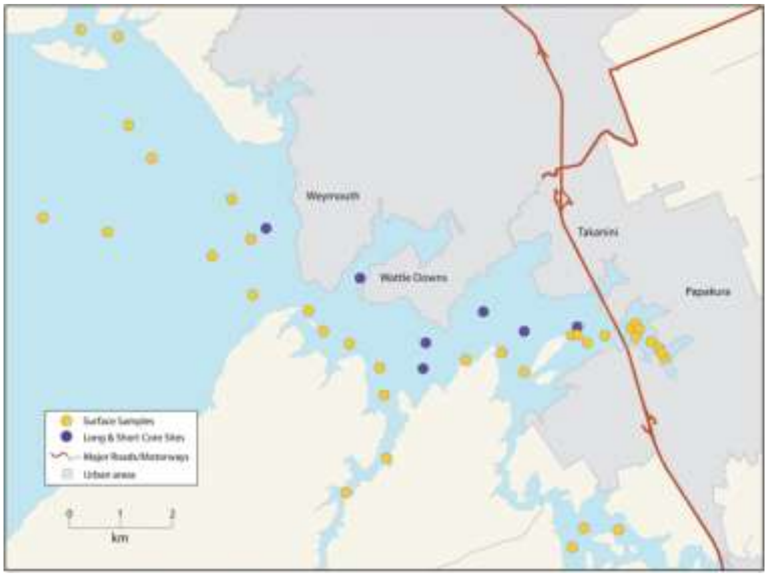


Figure 30

Surface sediment and core sites in the Tamaki Estuary analysed by Abrahim et al (2007).

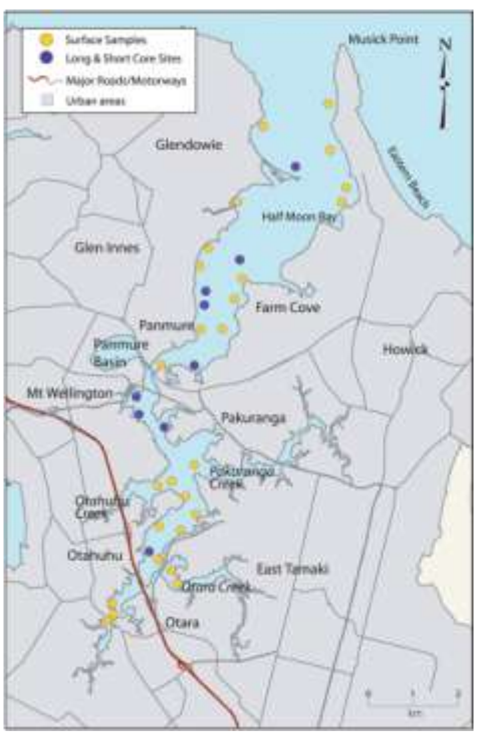
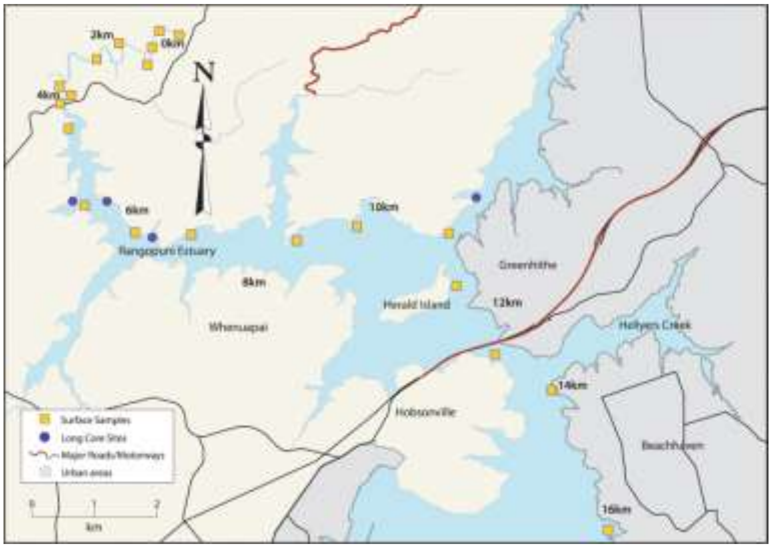


Figure 31

Longitudinal survey of Zn, Cu and Pb concentrations in the Rangitopuni river and estuary in the Upper Waitemata Harbour (Hayward et al. 2004).



6.6 Chemicals of potential environmental concern

Concentrations of CPEC found in Auckland's estuaries and harbours

ARC has monitored many of the traditional priority pollutants (eg, metals, PAH, organochlorine pesticides, PCBs) in recent times. A new group of chemicals are now emerging as being of potential environmental concern, based on their toxicity, persistence, and widespread or on-going use (see Section 3.9). These have been termed Chemicals of Potential Environmental Concern (CPEC). The difference between these two groups is summarised in Section 3.9.

Stewart et al. (2009) measured selected CPEC in sediments from 13 estuarine sites receiving run-off from urban, farming and horticultural areas, within marinas, and near treated sewage outfalls. Of these sites, seven are substantially impacted by diffuse urban stormwater, two were in the immediate vicinity of sewage discharges (Mahurangi and Puketutu), three were from marinas (Milford, Half Moon Bay, Westhaven), and one was primarily agricultural (Taihiki – receiving run-off from the Pukekohe market gardening area).

The survey measured the concentrations of 34 CPECs including surfactants, flame retardants, plasticizers, oestrogen hormones, antifoulants and pesticides, covering the major groups identified in a review of potentially important compounds (Ahrens 2008, Section 3.9). A major issue was the difficulty in analyzing many of these CPEC, and the fact that there are currently few commercial laboratories that do this. This resulted in only 34 compounds out of the 42 originally selected for study being analysed to date.

The levels of most CPEC were below analytical detection limits. This was partly due to the detection limits of the analytical methodologies available for the study and therefore does not completely rule out the possibility that CPECs are present at environmentally significant concentrations. The major classes of CPEC analysed, and the concentrations found are summarised in Table 8 below.

Table 8

Summary of CPEC assessed in the initial survey of Auckland estuarine sediments (data taken from Stewart et al. 2009).

CPEC	No. sites found (of total 13)	Concentration range	Site with highest concentration
Polybrominated diphenyl ethers (PBDEs)	13	0.7–570 ng/g (ppb)	Puketutu (sewage)
Total dithiocarbamates[1]	9	<0.02–0.11 mg/kg (as CS_2)	Shoal Bay (urban)
Antifouling biocides	0	<0.01 mg/kg	N/A
Organonitrogen and phosphorus pesticides (ONP)	0	<0.01–0.3 mg/kg	N/A
Acid herbicides	0	<DL. (0.01–0.03 mg/kg)	N/A
Glyphosate	9	<0.04–1 mg/kg	Meola (urban)
Phthalate plasticisers[2]	4	<0.6–12 mg/kg	Milford (marina)
Alkyl phenols	12	0.1–36 mg/kg	Puketutu (sewage)
Bisphenol-A	3	0.05–0.16 mg/kg	Milford (marina)
Triclosan	0	<0.1mg/kg	N/A
Steroid hormones[3]	9	0.47–2.8 ng/g (ppb)	Meola (urban)

1. Dithiocarbamates measured as the "total" present, as detected by released carbon disulphide (CS_2).
2. Concentration range given is for BEHP (bis(2-ethylhexyl)phthalate), the major phthalate plasticiser detected.
3. Oestrogenic hormones included estrone (E1), 17b-estradiol (E2), 17a-ethynylestradiol (EE2), and estriol (E3). The dominant compounds found were E1 and E2.

This study is the first step to determine whether significant concentrations of CPECs occur in the Auckland area. The report study does not provide any interpretation of the data collected in terms of possible environmental effects of the contaminants found, or the relative importance of the various sources covered by the monitoring sites. Such assessments would be valuable for future studies, to provide some indication of the relative risks associated with the various sources, both present-day and historical.

6.7 Conclusions

The large regional database on sediment contamination clearly confirms that urban stormwater is contaminating Auckland's urban estuaries with heavy metals and, at lower levels, a range of persistent organic pollutants (POPs). The 2005 Review (ARC 2008) indicated that over 50 per cent of regional monitoring sites are contaminated to the point where aquatic life may be beginning to be adversely affected (in terms of exceeding ARC's ERC).

While worst affected sites are in muddy estuaries receiving run-off from older, fully urbanised catchments, contamination is also found in harbour areas. Concentrations are lower, however. Source factors are important, such as proximity to source and catchment land use. Geochemical factors, such as organic matter, iron oxides, and sediment texture, also come into play as factors that determine concentrations. The most important directions for the future must address the question "what does this mean?" This is being addressed through the links between sources and concentrations and predicting future concentrations (see Section 8 – Predictions), benthic community health (see Section 9 – Ecology) and toxicity (see Section 10 – Toxicity).

Investigation into the concentrations and distribution of key "Emerging Contaminants" (CEPC) has begun. A review (Section 3.9) assessed which CPEC are most likely of relevance to Auckland, and initial data for key contaminant classes have been collected. These projects represent a major step forward for contaminant research in Auckland. Future work will need to assess the significance of the data collected, with respect to potential environmental impacts of the contaminants and also in relation to key sources (historical, present, and likely on-going).

Temporal trend monitoring conducted since 1998 continues to show increasing concentrations of Zn at most urban sites, smaller increases in Cu, and uncertain trends for Pb and PAH. Trend profiles are not simple linear increases and decreases as observed in the first six years of monitoring and described in the 2005 Review (ARC 2008), but more complex. The reasons for this are not clear and are being investigated.

7 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 (ie, sediments that have built up over recent time) and by deciphering these profiles, we are able reconstruct the impact of urban stormwater on sedimentation, sediment texture and sediment quality. This reconstruction can also give indications of future trends.

Various studies have produced a great number of results on contaminant profiles in sediments and/or sedimentation and hence the history on sediment contamination and deposition. Cores have been examined from the Rangitopuni (2 cores), Lucas (1), Central Waitemata (21), Tamaki (12), Mangere Inlet (6), SE Manukau (6), and region-wide (12). Many of these cores have been dated or have other relevant information on stormwater impacts. There is a wealth of information embodied within these studies that, if integrated and rationalised with other sediment profile findings described in the 2005 Review (ARC 2008), would greatly enhance our understanding of land use impacts.

The classical picture of heavy metal build up close to the sediment surface is generally supported, but there is a great deal of variability from place to place. Deposition rates are available for a much greater area than before and in greater detail in some areas.

Studies of fossil foraminifera have resulted in the postulation of impacts on benthic communities from increasing freshwater run-off accompanying development.

Overall, the information on history of stormwater impacts has been greatly enriched by many studies on sediment profiles, and it would be timely to integrate, synthesize, and hence clarify a regional picture from this information.

7.1 What was known in 2005

By 2005, a number of detailed sediment coring studies had: 1) reconstructed the history of contamination by urban stormwater-derived contaminants, 2) measured sedimentation rates in different estuarine environments, 3) measured the mixing of sediment by the burrowing activities of benthic animals (bioturbation):

- Sediment cores from Pakuranga, Lucas, Henderson, and the Tamaki Estuaries clearly showed the accumulation of urban-derived contaminants and sediment within urbanised estuaries. Sedimentation effects were largest in the upper reaches of muddy urban estuaries where sedimentation rates were very high. For example, about 1 m had been deposited in the uppermost reaches of Pakuranga Creek since 1960. There was a decrease in deposition rates down the estuary. While accumulation was less severe beyond the confined estuarine arms, it was still measurable in the sub-tidal areas of the Waitemata Harbour and Tamaki Estuary.

- Conversely, and probably less commonly, coring had also shown decreasing contamination over time as a result of reducing contaminant loads, and/or increasing loads of less contaminated sediment (eg, rural soils or subsoils). Mangere Inlet (metals, PAH) and Henderson Creek (DDT) were examples.
- Detailed cores, such as those taken in Pakuranga Creek Estuary, had 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, showed sediment mixing by animals living within the sediment (bioturbation) is represented by a homogeneous mixing depth of approximately 11-15 cm for intertidal mudflats in tidal creeks. This was in good agreement with earlier estimates based on ecological studies and modeling.
- There was evidence, that organic chlorinated pesticides (OCP) concentrations in estuarine sediments had increased as a consequence of urban development. Increases were relatively small (as in Lucas Creek, where the proportion of horticultural land was small), or large (eg, Henderson Creek, where a relatively large amount of horticultural land had been urbanised).
- Region-wide coring confirmed that Auckland's estuaries were infilling more rapidly than in the past, and that this will bring inevitable changes to ecology, hydrodynamics, sediment transport (including to the wider coastal receiving environment), and contaminant accumulation and dispersion. This infilling was due to sediments from all development and is not only an urban-related phenomenon. However, urban development had greatly accelerated sedimentation, especially in the headwaters of tidal creeks.

7.2 New work 2005-2008

New work undertaken since 2005 has been directed towards:

1. Filling in gaps in our knowledge on natural or background concentrations (ie, concentrations of metals in estuarine sediments in pre-European times). This work also attempted to determine whether estuarine sediment profiles showed evidence of volcanic soils in some areas and hence higher background levels of heavy metals (Section 7.3).
2. A number of University studies have measured heavy metal profiles in Auckland estuaries (Section 7.4, 7.6).

3. Sediment deposition rates and mixing (by bioturbation and wave disturbance) have been studied in the higher energy environments of the Central Waitemata Harbour and Pahurehure Inlet (Manukau Harbour) (Section 7.5).
4. Studies on fossil foraminifera in sediment profiles have led to new hypotheses about the causes of loss of benthic organisms in developed estuaries (Section 7.6).

7.3 Pre-development heavy metal levels in estuarine sediments

What are the natural levels of Zn, Cu and Pb in estuarine sediments before urbanisation?

During the development of Integrated Catchment Management Plans (ICMP), concerns were raised that concentrations of some metals in the marine environment are elevated due to underlying volcanic rocks causing high background concentrations of these metals. High metal concentrations typical of volcanic rocks had not previously been observed in pre-urban sediments at the base of marine sediment cores, nor in surface sediments at sites remote from urban stormwater discharges, as described in the 2005 Review (ARC 2008). Heavy metals, including those associated with volcanism (Vanadium, Cobalt, Nickel) were measured in cores taken at 12 sites widely spread over the Auckland region (Figure 32) (Reed 2008).

None of these 12 sites shows high natural nickel, vanadium or cobalt concentrations in the core that would imply the presence of sediments derived from volcanic lithology. For 6 cores, typical pre-urban zinc concentrations range between about 20 mg/kg and 40 mg/kg irrespective of the catchment geology, in line with earlier descriptions of background concentrations from cores taken in Lucas and Pakuranga estuaries. Three sites were not sampled deeply enough to reach background (ie, concentrations were still decreasing with depth at the bottom of the cores). A core taken in Rangitopuni Estuary (Upper Waitemata Harbour) showed consistently high Zn concentrations of 94-111 mg/kg, but may not have reached background concentrations. [Note that another study (Hayward et al 2004) indicated that lower Zn concentrations would be found at deeper levels]. At Kauri Point (Central Waitemata Harbour), Zn concentrations were consistently elevated at about 50 mg/kg. At Takanini (Manukau Harbour), Zn concentrations reached a minimum within the typical range described above (20 – 40 mg/kg), then increased, which may reflect penetration of underlying rock, or textural differences. In these later three cores, there were no elevated levels of other metals (V, Co, Ni) associated with volcanic rocks or soils. Overall, there was no clear evidence for volcanic soils and rocks increasing background concentrations.

7.4 Heavy metal profiles

Heavy metal profiles have been measured in eight cores from Tamaki Estuary (Abrahim & Parker 2008), three cores from the Upper Waitemata harbour (Hayward et al 2004, Hayward et al. 2006), and six cores from Mangere Inlet (Matthews et al. 2005). These studies confirmed the general findings that sediments have suffered significant systematic heavy metal contamination following catchment urbanisation.

Baseline concentrations and the degree of enrichment of surface sediment were assessed in sediment cores from Tamaki Estuary (Figures 32, 27) (Abrahim & Parker 2008). Average baseline concentrations were 13.2, 22.4, 72.5 and 0.08 mg/kg for Cu, Pb, Zn, and Cd respectively. These Cu, Pb and Zn concentrations are much greater than background described elsewhere (eg, Zn concentrations described above from Reed 2008), but at least part of this difference will be due to different analytical methods. Baseline concentrations varied greatly (eg, Cu 1 – 27 mg/kg, Zn 25 – 129 mg/kg). This large difference is not explained, but higher concentrations could be due to cores penetrating underlying soils or rocks. Abrahim (2005) and Abrahim et al (2008) also describe the sediment types, depositional setting, mineral maturity and sequence stratigraphy of the Tamaki Estuary. Hence with all this information, it may be possible to investigate these variable and high baseline concentrations further.

Sediment profiles in Mangere Inlet were taken to study the effect of sewage and industrial outfall impacts on foraminiferal records (Matthews et al. 2005) (see Section 7.6 for more findings from this study). Cores were taken to 60 – 100 cm depth at six sites (Figure 32). Heavy metal (As, Cu, Ni, Pb, Zn) concentrations were measured on the mud (<63 μm) fraction down the profile, and show general enrichment of Cu, Pb and Zn toward the surface.

Other studies in Tamaki Estuary and Upper Waitemata Harbour investigated the chronology of the fossil foraminifera record as a means to identify anthropogenic impacts in these estuaries (see Section 7.6). Heavy metal (As, Cu, Ni, Pb, Zn) concentrations were measured on the mud (<63 μm) fraction in cores taken from Panmure Basin, Tamaki Estuary, Rangitopuni and Lucas (Upper Waitemata Harbour) (Figure 32).

Together, these studies provide a comprehensive dataset for heavy metal profiles, as well as other information (eg, texture, detailed chronology). They could be further interrogated to investigate heavy metal contamination through time.

Figure 32

Sites of cores taken in Auckland estuaries by Abrahim & Parker (2008), Reed (2008), Hayward et al. (2004), (2006), Matthews et al. (2005).

7.5 Sedimentation patterns, rates and mixing

Where fine sediments settle and infill estuaries, and how deep settled contaminants are mixed into surface sediments

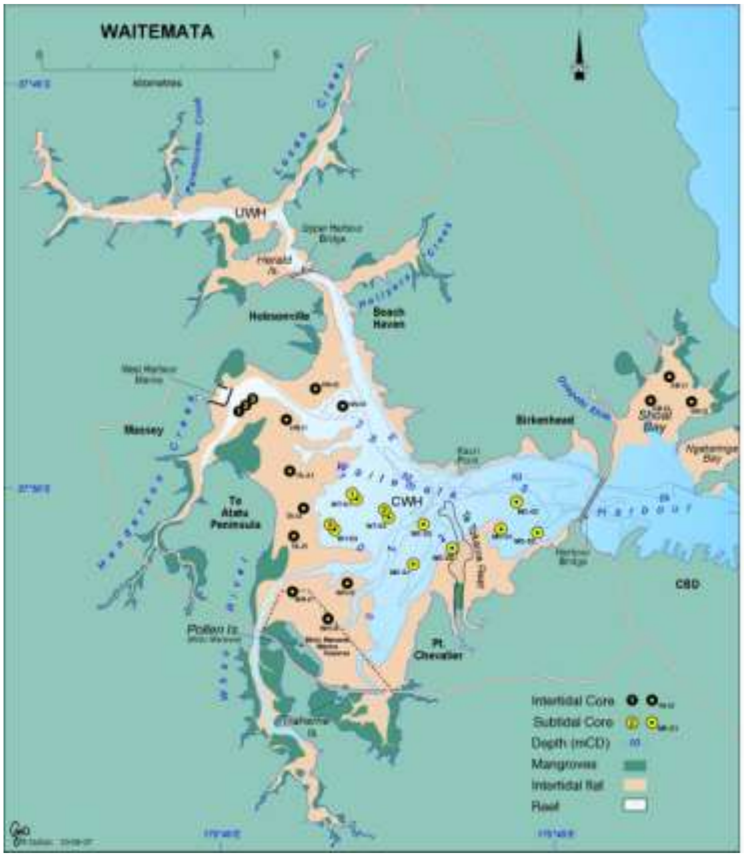
In the 1995–2005 period, major advances were made in understanding the sedimentation and mixing in Settling Zones and sedimentation in a regional context, but there remained an important gap in our understanding of sedimentation patterns and mixing of surface sediments in the higher energy Outer Zones. This information is necessary to understand how these more energetic water bodies process sediments and associated contaminants, the sources and sinks of sediments, and how far contamination is mixed into the underlying sediments. This gap has been addressed through studies on the Central Waitemata Harbour, the South East Manukau Harbour and the Tamaki Estuary.

Central Waitemata Harbour

A detailed study was carried out on sedimentation in the Central Waitemata Harbour (Swales et al. 2008). Cores (21) were taken throughout the harbour and sedimentation rates and mixing were assessed from radioisotopes using Berillium-7 (^{7}Be), Lead-210 (^{210}Pb) and Caesium-137 (^{137}Cs) (Figure 33).

Figure 33

Location of sediment cores collected for radioisotope dating in the Central Waitemata Harbour (Swales et al. 2008).

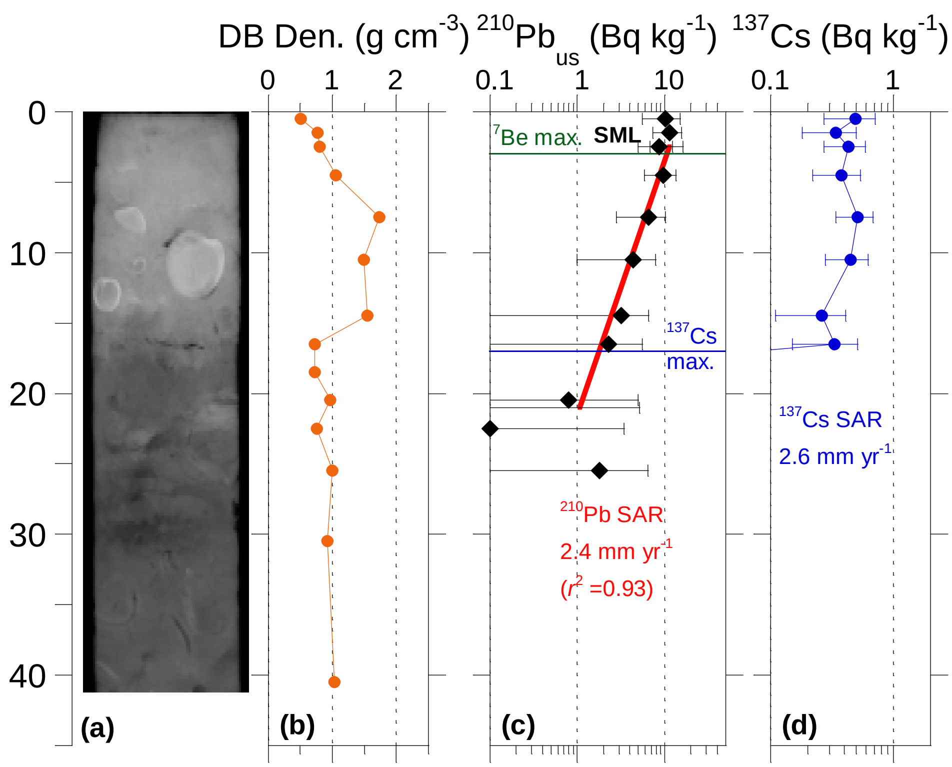


Sediment mixing was also assessed from X-ray photographs, and the burrowing and feeding behaviour of sediment-dwelling organisms. An example of results is shown in Figure 34. Deposited sediments were mostly muddy sands often containing buried cockle-shell layers between 0 and 15 cm below the surface. The mud content (<63 μm) of surface sediments was found to be less than 16 per cent over much of the central harbour. The mud content has not changed substantially in 50 years. Intertidal and subtidal flats are accumulating sediments at similar rates of ~ 3 mm/year.

^{7}Be was measured down to 5 cm, and profiles suggested that surface sediment were completely mixed in < 100 days to 1-5 cm depth. On this basis, the mean residence time of sediment is estimated to be 4–14 years in the surface mixed layer. The authors concluded that the ^{210}Pb profiles generally supported the concept of shallow mixing depths, although some profiles did show greater mixing depths. X-rays revealed few structures within the sediment profile that would suggest bioturbation structures at greater depths than 5 cm, as is observed in the muddy sediments of tidal creeks. However, an assessment of biota present and their known modes of feeding or motility suggest that bioturbation is >10 cm at most sites, but can vary markedly from year-to-year. Overall, the authors concluded that mixing depths were shallower than observed in tidal creeks and that the most intense mixing typically occurred to 3-5 cm, instead of to 10-15 cm measured in tidal creeks. While the evidence supports this in many cores, data from some cores and the biotic modes of feeding suggest deeper mixing depths over longer times.

Figure 34

Core profile for Te Atatu Intertidal: (a) x-radiograph; (b) dry bulk sediment density; (c) unsupported ^{210}Pb concentration profile with 95 per cent confidence intervals, time-averaged sediment accumulation rate (SAR) and coefficient of determination (r^2) derived from fit to data (red line), maximum ^{7}Be depth and maximum ^{137}Cs depth; (d) ^{137}Cs concentration profile with 95 per cent confidence intervals and time-averaged SAR. Note: the ^{137}Cs SAR is calculated after subtraction of ^{7}Be mixing depth. The vertical axis is in cm (Swales et al. 2008).



A conceptual model was developed in the study for the Waitemata Harbour. Fine sediment is delivered to the tidal creeks fringing the harbour, and most is trapped in the tidal creeks. Whau and Henderson estuaries have been substantially infilled and trapping/storage may now be limited from the infilling, which may mean that more fine sediment may escape these creeks in the future. The sediment that does escape these tidal creeks is widely dispersed. Intertidal flats and shallow subtidal flats provide temporary storage for fine sediments. Shoal Bay and the central basin of the Central Waitemata Harbour and the Hauraki Gulf are long-term sinks for fine sediments. The general pattern of sediment fate and transport was generally supported by subsequent modelling studies (see Section 8 – Predictions).

South East Manukau and Pahurehure Inlet

Core samples from 6 sites (Figures 32 and 29) were analysed for particle size, bulk density, and copper and zinc concentrations (Reed et al. 2008). Radioisotope profiles (^{7}Be, ^{210}Pb, ^{137}Cs) were also analysed, from which sediment accumulation rates were subsequently calculated.

Calculated sediment accumulation rates were between 2.2 mm yr^{-1} and 12.6 mm yr^{-1}. One core, collected from the sandy intertidal flats in the South Eastern Manukau Harbour, showed zero sediment accumulation. These calculated sediment accumulation rates are typical of other estuaries studied in the Auckland region, such as the Central Waitemata Harbour.

Vertical profiles of the copper and zinc concentrations for each of the 6 sediment cores show that current concentrations in the surface sediments are elevated approximately 1–2 times over background levels. This seems to be a relatively recent increase, and is line with the relatively recent age and degree of urbanisation in this area.

7.6 Fossil foraminifera reveal ecological changes with human impacts

Evidence for the impact of increased freshwater run-off accompanying development

In assessing impacts of urbanisation on aquatic ecosystems, it would be useful to know the pre-development ecology. Some information on the nature of pre-impact ecosystems comes from the fossilised hard-parts of the original biota that are preserved in sediments. Foraminifera have a particularly good fossil record and their taxonomic composition and general ecological distribution in these environments is the best known of all the microfossil groups.

Fossil foraminiferal faunas were studied in short cores from Upper Waitemata Harbour, Tamaki and Mangere estuaries (Hayward et al 2004, Matthews et al. 2005, Hayward et al. 2006) (Figure 6.1). The cores record similar major changes in their fossil content since the arrival of humans (ca. 1300 AD), with faunal changes continuing through to the modern times. Cores were dated using palynology (pollen identification and counts), and ^{137}Cs in the earliest study.

For the Rangitopuni Creek, Lucas Creek estuaries (in the Upper Waitemata Harbour), and in Tamaki Estuary there is observed at roughly the same depth in the sediment profile: an abrupt decline in calcareous foraminifera; a rapid increase in abundance of agglutinated foraminifera, large diatoms, and freshwater species; an abrupt decline and disappearance of marine molluscs; a major decline and virtual disappearance of ostracods; an increase in sedimentation rate.

The foraminiferal faunal changes with depth are similar to the present day changes in fauna in surface sediment collected in a longitudinal transect from Hobsonville (near the mouth of the Upper Waitemata) to upper reaches of the Rangitopuni Estuary. In the authors view, these similarities supported a common explanation. The changes in fauna in the longitudinal transect were attributed to decreasing salinity, and additionally decreasing pH (causing carbonate dissolution and loss of calcareous foraminifera) from Hobsonville to Rangtopuni. Therefore, the studies assessed whether a similar cause (decreasing salinity or increased freshwater run-off) could have brought about the observed changes historically in the sediment profile.

The authors postulated that the decrease in foraminifera in historical times was due to changes in salinity from increased freshwater run-off associated with forest clearance from Polynesian (ca. 1300–1840) and European (1840–1990s) times. They argue that it is unlikely to be due to heavy metal contamination. In the mouth of Lucas Creek Estuary, three phases of foraminiferal faunal change occurred: minor changes during initial Polynesian forest clearance (1500–1800 AD), a major change in early European times (1840–1870 AD) with clearance of most of the remaining native forest, and another small change in very recent times (1990s) with urbanisation in the Lucas Creek catchment. In Tamaki Estuary, which has a smaller catchment (and hence lower freshwater inflows), no faunal changes occurred in association with complete forest clearance and establishment of pastoral farming in Polynesian and early European times (before 1950s). Major foraminiferal and other faunal changes occurred in the late European period (1960s–1970s) coincident with the onset of major urbanisation spreading throughout the Tamaki catchment. The authors hypothesize that increased freshwater run-off is the major culprit for many of the observed biotic changes in the urbanised estuaries of Auckland. While this may be a plausible explanation for foraminiferal changes, these microscopic animals are likely to respond differently to environmental disturbance than higher organisms such as macroinvertebrates (Section 9).

The fossil foraminiferal faunas preserved in intertidal sediment in sediment cores from Mangere Inlet reconstruct a contaminant-related ecological history of the inlet (Matthews et al. 2005). Here Polynesian forest clearance and horticulture had negligible impact on the harbour foraminiferal biota. Major changes occurred in the early European to 1960-period, but seemed to be due to organic-rich discharges from four large meat works around the head of Mangere Inlet. After 1960, when the Mangere Sewage Treatment Plant (MSTP) was opened and the meat works and other outfalls progressively closed, patterns of *in situ* faunal composition move back toward those obtained from the pre-human period.

7.7 Conclusions

Recent coring results provide no evidence for background zinc and copper concentrations being elevated in Auckland estuaries because of volcanic rocks (Reed 2008). Six out of 12 core concentrations were similar to the background concentrations found at uncontaminated sites throughout Auckland. Some sites did have slightly-moderately elevated Zn and Cu concentrations above background at depth. These cores lacked the confirmatory evidence of elevated levels of other elements associated with volcanic rocks (V, Co and Ni), or concentrations were still decreasing at the base of the cores (ie, the cores were not deep enough to reach background), or cores may have sampled underlying rock.

In another study of 12 cores from Tamaki Estuary, higher concentrations were found in at the base of some cores, and while analytical methodology is different, this warrants further investigation.

Additional work on contamination and sedimentation in Auckland estuaries have shown typical contamination profiles for heavy metals with background concentrations constant at the base of the sediment profile and increasing near the surface. For the Central Waitemata Harbour these increases are consistent with contamination accompanying the urban changes in the middle of the last century of rapid expansion of urban area, traffic volumes, stormwater reticulation and its connectivity to receiving waters. For the South Eastern Manukau, these increases are relatively modest and close to the surface and thus reflect the recent age, degree of urbanisation and the larger rural catchments. These studies provide detailed information on sedimentation rates and sediment mixing for these harbour areas.

In studies on foraminiferal changes in sediment profiles, Hayward and co-workers measured heavy metal profiles (Hayward et al. 2004, 2006). Methodology is different from other studies but these cores show a classical picture of metal enrichment, which has been dated and related to biotic changes. These studies would provide additional data on contaminant accumulation if concentrations and methodology could be benchmarked to other studies.

There is now a wealth of information on sediment texture, sedimentation rates, and sediment mixing rates and depths from sediment cores taken in the Central Waitemeta and SE Manukau Harbours. The mixing depth and intensity are important part of the understanding of fate, transport and cycling of trace metals in the harbour.

Studies on foraminifera remains preserved in sediment profiles has led to the hypothesis that benthic ecosystems have been greatly affected by changing catchment run-off quantity. This warrants further examination because it does not fit with current evidence that impacts are due to contamination, sedimentation, and/or changes in sediment texture. Possible ways forward from here are to see if the hypothesis is supported by an historical construction of freshwater run-off, or to compare results to the Benthic Health Model or other regional monitoring programmes.

Overall, new work has greatly increased the body of information on the build up of contaminant concentrations in sediments over time, sedimentation rates and on sediment mixing. An hypothesis of increased freshwater inflow as an additional stressor in urban estuaries has been proposed and warrants further investigation.

8 Predicting Contaminant Fate in the Marine Environment

Two new suites of models have been developed to predict the fate of contaminants (heavy metals, TSS and bacteria) in the wider marine environment: (1) CREA and (2) USC-3. The models take information on stormwater inputs and through hydrodynamic and particle tracking models, determine the fate of stormwater particulate solids, and hence suspended sediment, Zn, Cu and/or indicator bacteria. These, and earlier models described in the 2005 Review (ARC 2008), are compared and summarised.

The CREA model predicts Zn concentrations in the sediments Whau Estuary, Tamaki Estuary, Hobson Bay, the western bays of the Waitemata harbour and north-east Manukau. Predictions are based on recent (1999) loads from upstream catchments and are likely to overestimate contaminant concentrations in the future. Further work is needed for the CREA model using the more accurate CLM-USC3 model for stormwater inputs. Prediction of bacterial contamination of bathing beaches provides an additional tool to investigate stormwater impacts and management, because it provides the Pressure (stormflows) – State (Bathing beach quality) linkages to explain and investigate exceedance of recreational water quality (eg, MfE) guidelines.

In Auckland harbours, including the Central Waitemata and South East Manukau Harbours, the USC-3 models predict that contaminants that escape from tidal creeks and embayments or are directly discharged to the harbours only increase in concentration slowly in the sediments and do not reach the high concentrations predicted (and found) in tidal creeks and embayments. Sediment supply has a major influence on the fate and concentrations of Zn and Cu in the marine sediments through dilution (SE Manukau) and destabilising sediment beds. Overall, contaminants are widely distributed in harbours and can be concentrated within parts of the harbour by hydrodynamic conditions (eg, Shoal Bay receives much of its contamination from catchments west of the Auckland Harbour Bridge).

The USC-3 model is able to take into account inputs from all catchments, the exchange of contaminants between parts of the harbour, and resuspension and redistribution processes. It shows the links between sources and fate, so that contamination can be traced back to its discharge (and catchment) and the fate of all contaminants from a specific catchment discharge can be tracked. Both models can assess the effect of management intervention, such as land use change, source control and stormwater treatment. Overall, the USC-3 and CREA models are good PSR models, linking Pressure (catchment loads) – State (concentrations in sediments) – Response (management intervention).

8.1 Introduction

Contaminant build-up in marine sediments is a major concern for the long-term health of the Auckland's marine environment. In some parts of Auckland, the present day concentrations exceed sediment quality guidelines to protect aquatic animals, and in many areas the concentrations are predicted to 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!).

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? Extrapolating existing conditions into the future with conceptual and mathematical modeling enables us to predict the effects of different land use management and stormwater treatment scenarios.

8.2 What was known in 2005

TModelling, both relatively simple (eg, the USC1 model) and sophisticated (USC2 model), enabled the consequences of urban development on contaminant concentrations in sheltered estuary and harbour sediments to be predicted. The relatively simple USC1 model was found to be useful for predicting impacts in estuarine settling zones such as Motions Creek or Hellyers Creek, whereas the USC2 model can be applied to predict effects in more complex settling and transportational environments, such as the Upper Waitemata Harbour. The models were validated by field measurements of contaminant and sediment accumulation, and were therefore the most robust methods available to assess the potential impacts of different land development scenarios and management options.

Modelling up to 2005 highlighted the increases in Zn, and to a lesser degree Cu, concentrations in estuarine sediments that can occur as a result of urban stormwater discharges.

8.3 New work 2005-2008

Models have continued to be improved and applied to even more challenging receiving water conditions:

1. The new model (CREA) was developed by Auckland University and applied to the marine waters receiving run-off from Auckland City. In many ways, this model is similar to the USC2 model described in the 2005 Review (ARC 2008), in that contaminants and sediment are delivered by representative storms, and particles are allowed to disperse and settle in the marine environment depending on tidal currents and wind-induced waves. In addition, the CREA model tracks bacterial pollution and predicts the length of time bathing beaches will breach safe swimming guidelines (Section 8.5).
2. Predicting the transport and fate of contaminants in the higher-energy harbour environments is more challenging because the harbour sediments are subject to greater wave energies. Consequently, sediments can become eroded during wind events and be dispersed and deposited elsewhere. This is a particularly important mechanism in Auckland's harbours, which are shallow with large intertidal areas. A major advance in predicting contaminant concentrations in sediments was to include mechanisms in the predictive models that would allow deposited sediments to be resuspended and dispersed. This led to the development of "USC-3" model that was applied to the Central Waitemata Harbour and the South East Manukau Harbour (Section 8.6, 8.7).
3. The implications from the model predictions are described more fully in Section 8.8.
4. The USC3-CLM model incorporates the projected future changes in population, infilling, traffic density and building materials (the model is described in Section 3). This advance provides a more realistic prediction of future loads. These changes are included in the USC3 model predictions, which has changed our view of the fate and transport of contaminants, and consequently the accumulation of sediment, zinc and copper in estuary sediments.

Many assumptions and simplifications have to be built into these models to make them work. Nevertheless, the models are based on extensive knowledge and include the major transport and fate processes. They are also calibrated with known data (eg, heavy metal concentrations in sediments and measured sediment deposition rates).

8.4 Comparison of models

There are now a number of models predicting the fate of urban-derived contaminants. These include the Urban Stormwater Contaminant models (USC-1, USC-2 and USC-3) as well as the CREA models. In addition, there are several catchment models used to predict metal and sediment loads. These include the GLEAMS (for rural sediments) and Contaminant Load Models (CLM) described in Section 3. Table 9 gives a brief outline of the models used to predict contaminant fate in Auckland's marine area. There is a brief description of each model and its application, along with its capabilities and limitations.

8.5 Predicting zinc accumulation in bed sediments and bacterial concentrations in the water column at Auckland City's coastline

Coastal Receiving Environment Assessment (CREA)

As an aid to making future investment decisions in stormwater infrastructure, Auckland City Council/Metrowater required estimates of contaminant concentrations in the Waitemata and Manukau harbours, under current and future stormwater (and wastewater and combined overflow) loading conditions and also under various contaminant reduction scenarios. The CREA set of models were developed and used to predict the expected sediment deposition and benthic zinc levels to the year 2150 and the expected bacteria levels at bathing beaches, for a "typical" year.

The areas modelled are shown in Figure 35 and include Whau River Estuary, Tamaki Estuary, Hobson Bay, West Harbour Bays and major Auckland City beaches (Croucher et al. 2005a-f, Croucher et al. 2007a,b).

Table 9a

Summary of the various models used to predict contaminant fate in Auckland marine environments.

Model & developer	Initial purpose/application	Attributes	Reference
USC1 (NIWA)	Predicting contaminant (metal) accumulation in single settling zone that receives run-off from single, adjacent catchment source. 1992-2004.	• Uses a simple mass balance approach with simple assumptions on magnitude of major processes. • Inputs are annual loads based on specific yields for different land uses. These can vary through time with land use development scenarios. • Simple to run. Small number of input variables. Validated by core profiles and hind-casting to present day. • Ignores particle size. • Single source & sink only (sink receives run-off directly from source). • Average concentrations predicted in wide area (the SZ; typically km to 10 km scale). • Limited to simple sheltered deposition zones. Can't account for contaminant or sediment remobilisation & secondary transport. • Does not deal with uncertainty.	Vant et al. 1993, Williamson et al 1998, Seyb & Williamson 2004
USC2 (NIWA)	Predicting sedimentation and metal accumulation throughout "low-energy" estuary consisting of network of SZ and OZ connected to network of sources on the land. Developed for Upper Waitemata Harbour Contaminant Study (2004).	• Uses calibrated hydrodynamic and particle transport model to predict the fate of sediment and contaminants. • Metal inputs are annual loads; sediment inputs are "event" loads, both are based on specific yields for different land uses. These can vary through time with land use development scenarios Metal loads are distributed across representative storm sizes to match event sediment loads. • Expert operation. Validated by core concentration profiles and sedimentation rates. • Includes particle size – contaminants are distributed across 3 particle sizes. • Accommodates multiple sources & sinks. • Performs source tracking. • Average concentrations predicted in subestuaries (typically at km to 10 km scale). • Appropriate for SZ and OZ (sediment & contaminant redistribution) in low energy environments, in that the model allows for erosion, resuspension and redistribution between representative storm inputs, but only what has been deposited in most recent event. • Estimates prediction uncertainty only.	Green et al. 2004
CREA (UoA)[a]	Predicting contaminant accumulation in Auckland estuaries receiving stormwater run-off from Auckland City. Applied to parts of Waitemata Harbour, Hobson Bay,	• Initially used fixed loads from pollutographs (1999 values). • Inputs are annual loads based on specific yields for different land uses. These can vary through time with land use development scenarios Loads are distributed across representative storm sizes. • Expert operation. Validated by hind-casting to present day. • Includes particle size – contaminants are distributed across 8 particle sizes.	Croucher et al. 2005a-g, Croucher et al. 2007a, b

Model & developer	Initial purpose/application	Attributes	Reference
	Tamaki Estuary, and NE Manukau Harbour 2005.	• Accommodates multiple sources & sinks. • Does not perform source tracking. • Concentrations predicted in relatively small areas (typically 100m to km scale). • Appropriate for SZ and OZ in low energy environments. Does not handle resuspension and redistribution except very simply. • Estimates prediction uncertainty only.	
USC3 (NIWA)	Predicting sedimentation and metal accumulation throughout "high-energy" estuary consisting of network of SZ and OZ connected to network of sources on the land. Developed for SE Manukau & Central Waitemata Harbours 2008	• Uses calibrated hydrodynamic and particle transport model to predict fate of sediment and contaminants. Includes wave models and resuspension/dispersion/resettling. These models (all part of the DHI estuary model suite) provides inputs to the USC3 model. • Metal inputs are annual loads; sediment inputs are daily loads, both are based on specific yields for different land uses. These can vary through time with land use development scenarios. Metal loads are distributed across days to match daily sediment loads. • Expert operation. Validated by core concentration profiles, sedimentation rates, hind-casting to present day, and dispersal and deposition patterns in the harbours. • Includes particle size – contaminants are distributed across four particle sizes. • Accommodates multiple sources and sinks. • Performs source tracking. • Average concentrations predicted in subestuaries (typically km to 10 km scale). • Appropriate for SZ and OZ (sediment and contaminant redistribution) in high-energy estuarine environments. Allows for erosion, resuspension and redistribution of any deposited sediment. • Can be run in Monte Carlo mode to estimate prediction uncertainty.	Green 2008a-e

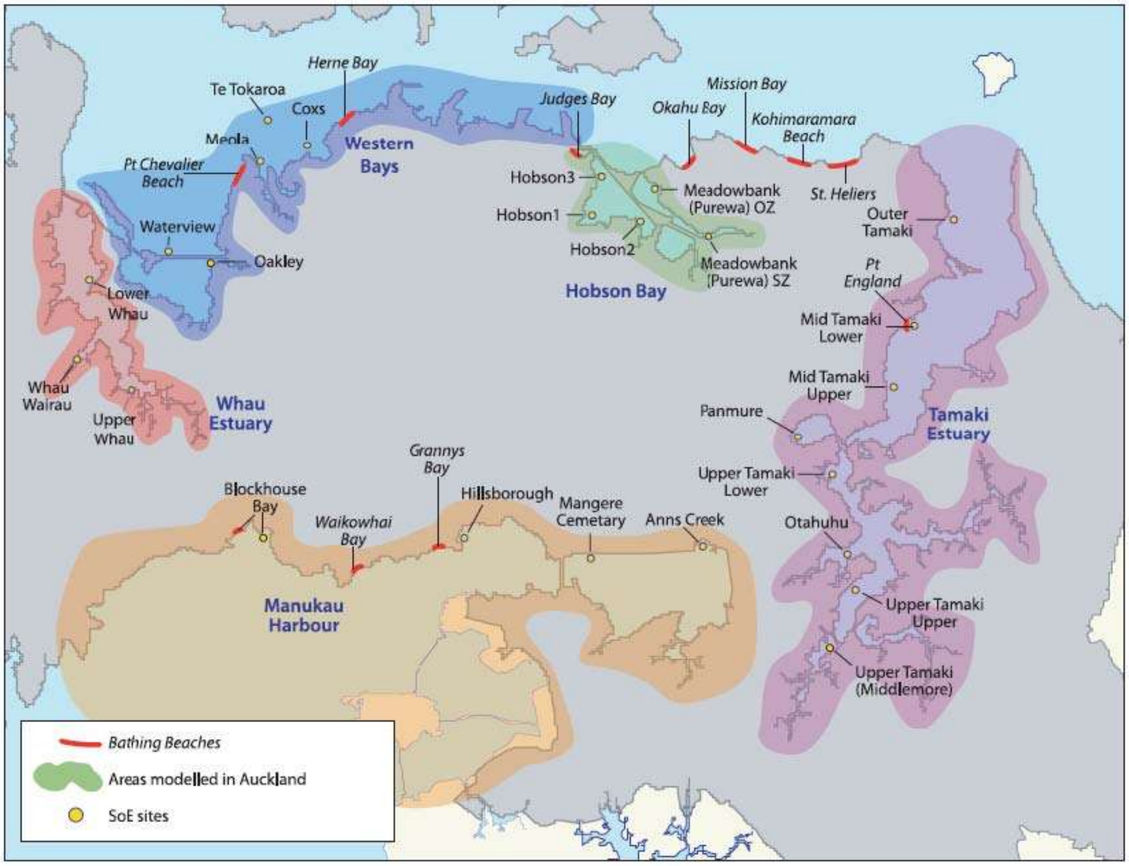
[1]University of Auckland.

Table 9b

Summary of the various models used to predict contaminant loads to Auckland's aquatic environments.

Model & developer	Initial purpose/application	Attributes	Reference
CLM (ARC)	Predict loads of Cu, Zn, TSS, TPH from urban areas. Developed as general load model for Auckland.	• Simple model predicting catchment load by adding loads from each land use area. • Uses specific yields for contaminants for specific land use areas. • No variation of specific yields with time. • Suitable for present day and past years only. • Annual loads. • Publically available spreadsheet operation (ARC website). • Validated from catchment and source measurements. • Multiple sources and catchments.	Timperley & Skeen 2008a, b
CLM-USC3 (Timperley + NIWA)	Predict loads of Cu, Zn, TSS from urban areas Applied to SE Manukau & Central Waitemata Harbours 2008.	• Simple model predicting catchment load by adding loads from each land use area. • Uses specific yields for contaminants for specific land use areas. • Variation of specific yields with time due to population growth, traffic density changes, infilling and intensification. • Suitable for hindcasting, and forecasting. • Annual loads. • Expert operation. • Validated from catchment and source measurements. • Multiple sources and catchments.	Timperley & Reed 2008a, b
GLEAMS (NIWA)	Predict loads of Cu, Zn, TSS from rural areas Applied to Upper Waitemata Harbour 2004, Orewa and Weiti 2004, SE Manukau & Central Waitemata Harbours 2008.	• More complex deterministic model based on the processes associated with rainfall/run-off/infiltration/storage/erosion. • Generates TSS loads. Zn and Cu loads calculated from TSS based on soil concentrations. • Variation through time possible. • Suitable for hindcasting, and forecasting. • Daily loads, and storm loads can be estimated from daily loads. • Expert operation. • Validated from catchment load measurements. • Multiple sources and catchments.	Green et al. 2004, Parshotam 2008, Parshotam & Wadhwa 2008a-c, Parshotam et al. 2008a-c

Figure 35
Areas modelled in Auckland with the CREA model (Adapted from Croucher et al. 2005g).



There are over 2000 stormwater and wastewater outfalls discharging to the Auckland City coastline (Croucher et al. 2005a). Outfalls that are close together and discharge to approximately the same location were grouped together into one "consolidated" outfall. A simplified set of 78 "consolidated outfalls" were used to represent the full set of physical outfalls.

A hydrodynamic model was used first to simulate currents and tides in the model areas. Estimates of stormwater loads of sediment, zinc and enterococci were provided for each consolidated outfall as pollutographs from Metrowater. These were then used as input to the particle tracking model (PTM), which simulated the movement of sediment and bacteria by those currents. Other factors such as settling and bacterial die-off were also included. For the sediment modelling, an additional "contaminant accumulation" modelling step was included to simulate the mixing and long-term accumulation of zinc in benthic sediments.

For modelling sediment deposition, each model area was divided into "deposition zones". For each consolidated outfall, the sediment model calculated the proportions of discharged sediment and zinc mass that would be deposited in each zone. Particles were allowed to sink and eventually settle on the bottom, and they could also be resuspended by the current or by wind-induced wave action in shallower waters. The pattern of sediment deposition over a "typical" year was estimated by simulating three representative storms of varying intensity from the typical year. The calculated deposition patterns for these three storms were then combined, according to the total amounts of sediment and zinc mass annually discharged at each storm intensity level, to give an estimate of the complete deposition pattern for the whole year.

The major assumption, that the sediment and Zn inputs for 1999 can be extrapolated over 150 year, presently limits the use of the model outputs. The Central Waitemata Study (Section 8.6) has predicted large changes in Zn and sediment loads from its catchments over the next 100 years (Timperley & Reed 2008b). This shortcoming in CREA will be addressed in the near future, with more realistic future loads being input to the model. Therefore results are not reported here, and the original estimates can be found summarised in Croucher et al. 2005.

Concentrations of bacteria at bathing beaches are also modelled using the particle tracking model. However, the physical processes simulated are slightly different. Bacteria, unlike sediment particles, are assumed to not settle. Also, exposure to sunlight causes the bacteria to die off. Although the currents calculated by the hydrodynamic model are two-dimensional, the bacterial model uses a quasi-3D approach to approximate the effects of stratification on bacterial transport.

Because bacterial contamination events are short-lived and the effects strongly dependent on prevailing conditions (eg, tide), all storm events in the "typical (1999) year" were simulated, rather than the three typical storm sizes described earlier. Actual tidal conditions are used for each event (rather than the mean tide used for long-term modelling of sediment and zinc deposition). Original estimates summarised in Croucher et al. 2005g, were updated in Croucher et al. 2007a,b, and Table 10 shows the updated results of modelling.

Table 10

Modelled beach grades for bathing and non-bathing seasons, calculated from predicted bacterial concentrations on every Tuesday (this is one of MfE Protocols: weekly Tuesday sampling). Predictions are for stormwater only and stormwater+wastewater (in brackets) (Adapted from Croucher et al. 2007a,b).

Beach	Bathing season	Non-bathing season
Pt Chevalier	A (A)	B (B)
Herne Bay	A (B)	A (C)
Judges Bay	A (B)	B (C)
Okahu Bay	A (B)	B (C)
Mission Bay	A (A)	B (B)
Kohimaramara Bay	A (A)	B (C)
St Heliers Bay	B (B)	C (C)
Pt England Beach	A (A)	B (C)

Beach	Bathing season	Non-bathing season
Grannys Bay	A (A)	A (B)
Waikowai Bay	A (A)	B (B)
Blockhouse Bay	A (A)	B (B)

The grading was also reported for daily monitoring and continuous modelling, and for the frequency and proportion of time that the MfE (2003) alert and action levels were breached.

The modelling to date provides the Pressure (stormflows) – State (Bathing beach quality) explanation for observed bathing beach quality. Although stormwater and combined sewer overflows (CSO) affect water quality in terms of bathing beach standards, it is only in the inner harbour sites that the frequency of exceedance results in significant deteriorations (more breaches of MfE guidelines). The model has be used to explore human health risk assessed by differing sampling strategies, and could explore the benefits of management intervention scenarios.

8.6 The fate and transport of Zn, Cu and sediment in the Central Waitemata Harbour

Will sediments in the wider marine environment become contaminated?

The tidal creeks[10] on the Central Waitemata harbour are amongst the most contaminated of Auckland estuaries. Contaminants such as sediment and heavy metals discharged into tidal creeks are mostly trapped there because of the favourable conditions for settling and accumulation. However, some sediment and heavy metals escape or "spill-over" into the harbours. The fate of urban stormwater-related contaminants that "escape" the tidal creeks, as well as those directly discharged to the harbour, was not understood. It was known from monitoring and sediment profiles that there has been some accumulation of Zn and Cu in places such as the Central Waitemata Harbour (see 2005 Review, Mills & Williamson 2008. Continued accumulation was expected to be far less than the tidal creeks, because of the size (ie, dilution effects) and energy (ie, greater potential for loss from the harbours). This is still of concern because many of the animals that live in harbour sediments are more sensitive than those typically found in tidal creeks. Therefore, it was important to look into the future to assess the potential future problem and the possible benefits of management intervention.

The Central Waitemata Harbour Contaminant Study aimed to predict the fate and transport of fine sediment, Cu and Zn discharged to the Central Waitemata Harbour from surrounding catchments, under a variety of land use and contaminant treatment scenarios, over the next 100 years (to 2100).

Models were used to predict sediment run-off loads (and natural Zn and Cu loads) from rural catchments (using the GLEAMS model) and sediment, Zn and Cu exports from the urban catchments (Contaminant Load Model; USC3-CLM). These models and their

[10] Tidal creeks characteristics and other marine receiving environments are described fully in the 2005 Review.

predictions have been described in Section 3 and summarised in Table 9b. The output from these models includes loads of sediment, Cu and Zn in different particle size fractions from 1940 until 2100. These outputs form the inputs to the USC-3 model, which assesses the fate and transport of fine sediment, Cu and Zn discharged to the Central Waitemata Harbour (Green 2008a-c).

The USC-3 model uses the output from the "DHI estuary model suite", which comprises the DHI Water and Environment (DHI) MIKE3 FM hydrodynamic model, the DHI MIKE3 MT sediment transport model, and the SWAN wave model. Together, these simulate tidal propagation within the harbour, tide- and wind-driven currents, freshwater mixing, waves, and sediment transport, resuspension and deposition (Green 2007). The USC-3 model is run in two ways:

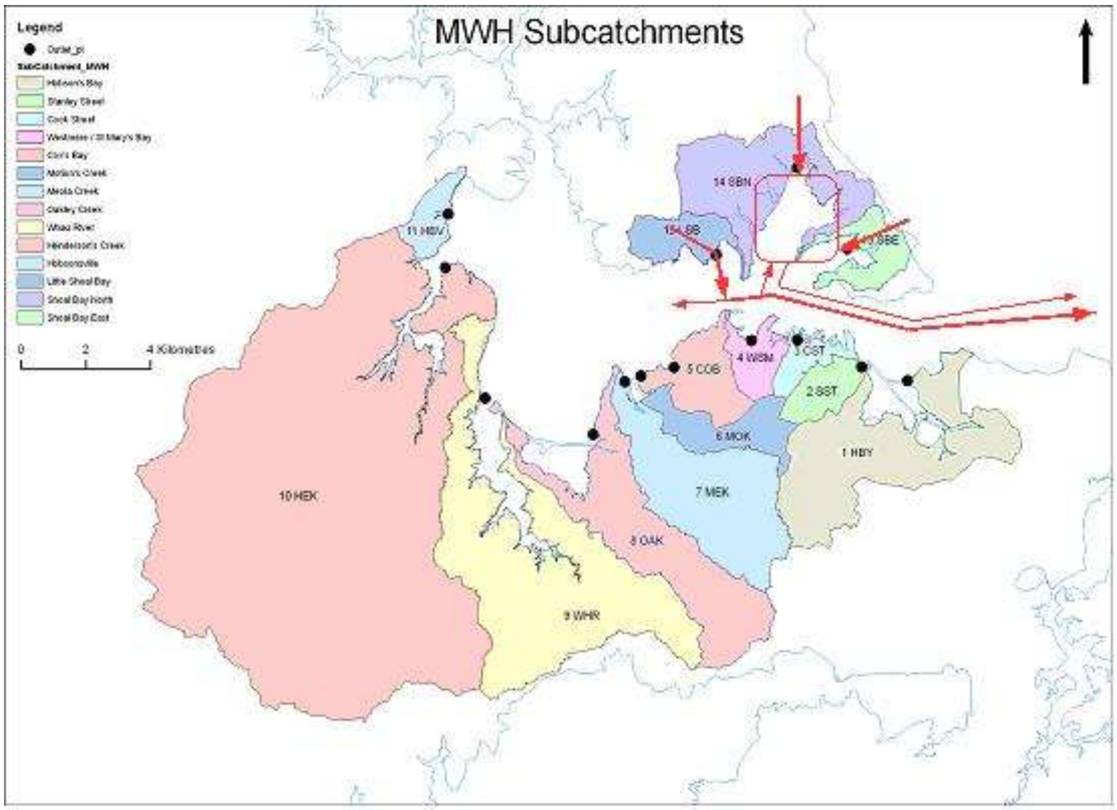
- to make "hindcast" predictions (from 1940 – 2001) for model calibration using present day measurements of sedimentation rates and predicted historic catchment contaminant loads (Green 2008a); and
- to make future predictions out to 100 years under likely urban development scenarios and under various management scenarios of source control and stormwater treatment (Green 2008b,c).

The model allows scientists to determine the **fate** and the **sources** of sediment and metals in various sub-estuaries within the harbour and to predict the change in concentrations of Zn and Cu around the harbour into the future.

An example of the **fate** predictions is shown in Figure 36 (Green 2008b), for sediment discharged from Shoal Bay and the surrounding catchments. About two-thirds of the sediment from Little Shoal Bay sub-catchment is predicted to be lost to the Hauraki Gulf. A quarter turns the corner to the east and gets trapped in Shoal Bay. That pattern is reversed for Shoal Bay sub-catchments, which drain directly into Shoal Bay: a little more than 50 per cent of the sediment from each sub-catchment is deposited in Shoal Bay, and about 40 per cent is lost to the Hauraki Gulf.

Figure 36

Schematic summarising fate of sediment and metals originating from the Little Shoal Bay, Shoal Bay North and Shoal Bay East sub-catchments. (Reproduced from Green 2008b.)

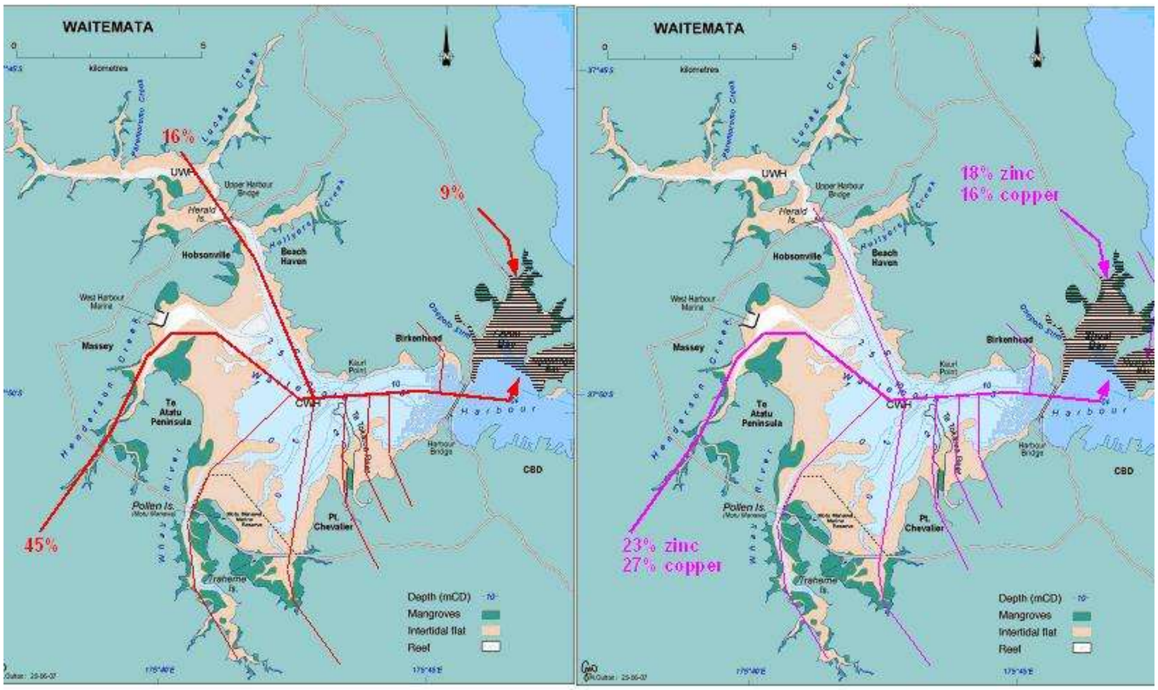


The fate is different for other catchments. Contaminants discharged from the CBD are mostly lost to the Hauraki Gulf. The majority of contaminants from catchments with tidal creeks or embayments end up depositing in these sheltered settling zones at the bottom of the catchment. All catchments to the west of the Auckland Harbour Bridge contribute contaminants to the wider harbour and to Shoal Bay (see Figure 37 and further discussion below).

The **sources** of deposited sediment and metals can also be identified. Most of the sediments and metals that deposit in tidal creeks (Henderson Creek, Whau River) and sheltered embayments (Limeburners Bay, Waterview embayment, Hobson Bay) come from the stormwater discharges that directly drain into them. Shoal Bay is an exception, because it receives sediments and metals from the whole watershed, except for discharges from the CBD, Hobson Bay and St Mary's Bay. This surprising result occurs because the harbour bridge embankments mix and steer contaminants into Shoal Bay on the ebb tide. This means that Henderson Creek, the largest source of sediments and metals to the Central Harbour, is predicted to be a significant source of sediments and metals for Shoal Bay. In the wider harbour, sediments and contaminants tend to be derived from the whole watershed and are thoroughly mixed together. An example of identified source areas of sediments and metals is given in Figure 37.

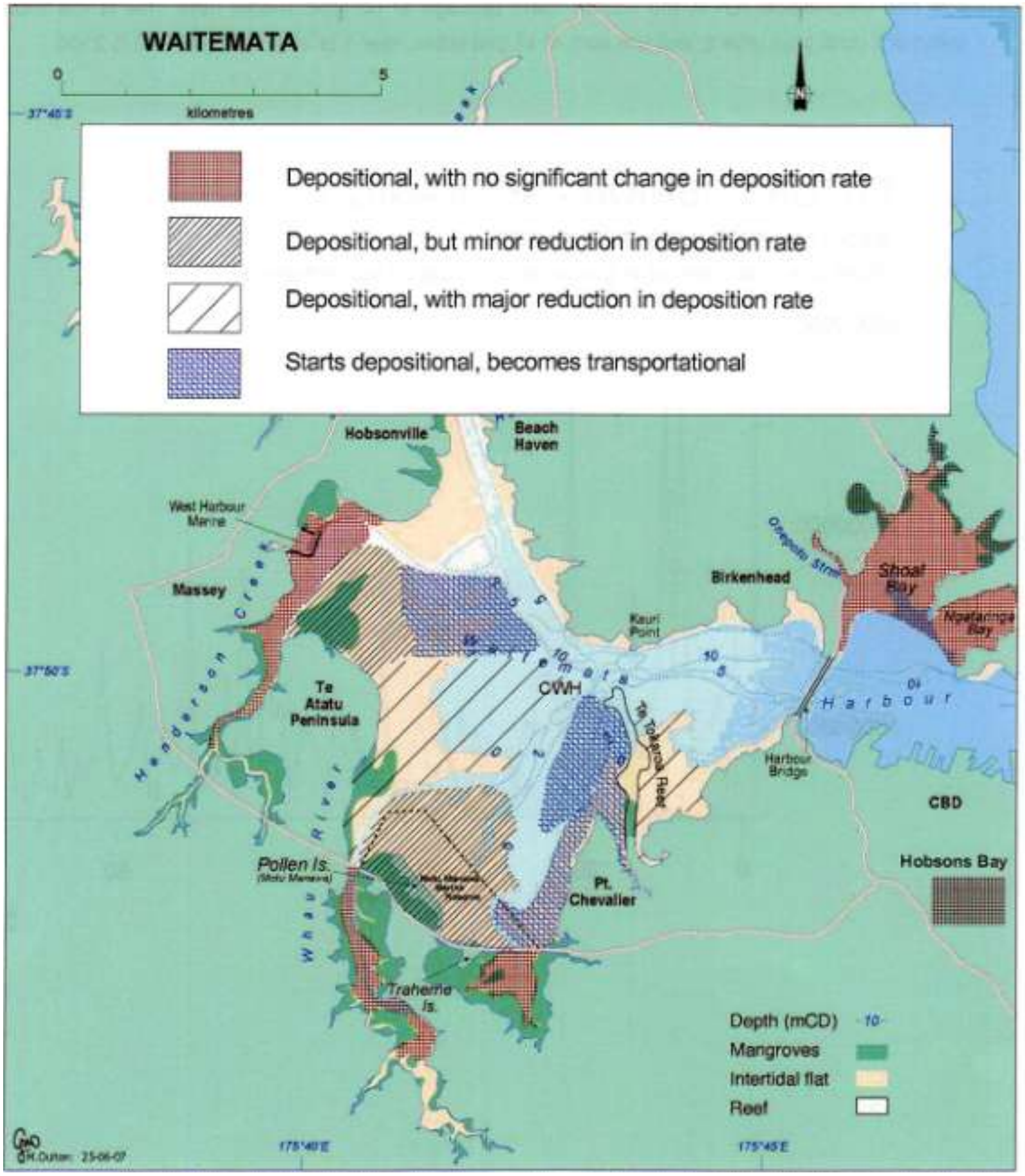
Figure 37

Schematic summarising origin of sediment and metals that deposit in Shoal Bay subestuary (reproduced from Green 2008b).



The model also predicts how the harbour "processes" the discharged sediment. Sediment loads were much higher in the past because most of the Waitemata catchment was rural land, and excessive erosion occurred during urban development. However, sediment loads decreased over time as more and more land changed from rural to urban land use. In the future, the USC-3 model predicts that the decrease is sufficient to result in some areas of the wider harbour having greatly reduced sediment deposition rates and some areas effectively turning into transportational zones or source areas for sediment, rather than depositional zones (Figure 38). Not unexpectedly, this has profound implications for the build up of contaminant concentrations.

Figure 38
Implications for the harbour from reduction in sediment inputs. Schematic showing the change in sedimentation due to a widespread reduction in sediment run-off from the catchment over the next 15–20 years under additional stormwater treatment. (Reproduced from Green 2008b.)



Prediction of the accumulation of contaminants with time for scenario 1 (no additional stormwater treatment or source control) are summarised in Figure 39. Note that similar results were obtained for other scenarios modelled, mainly because the source and other treatment controls examined in the model have relatively little impact on resulting average metal concentrations in the harbour.

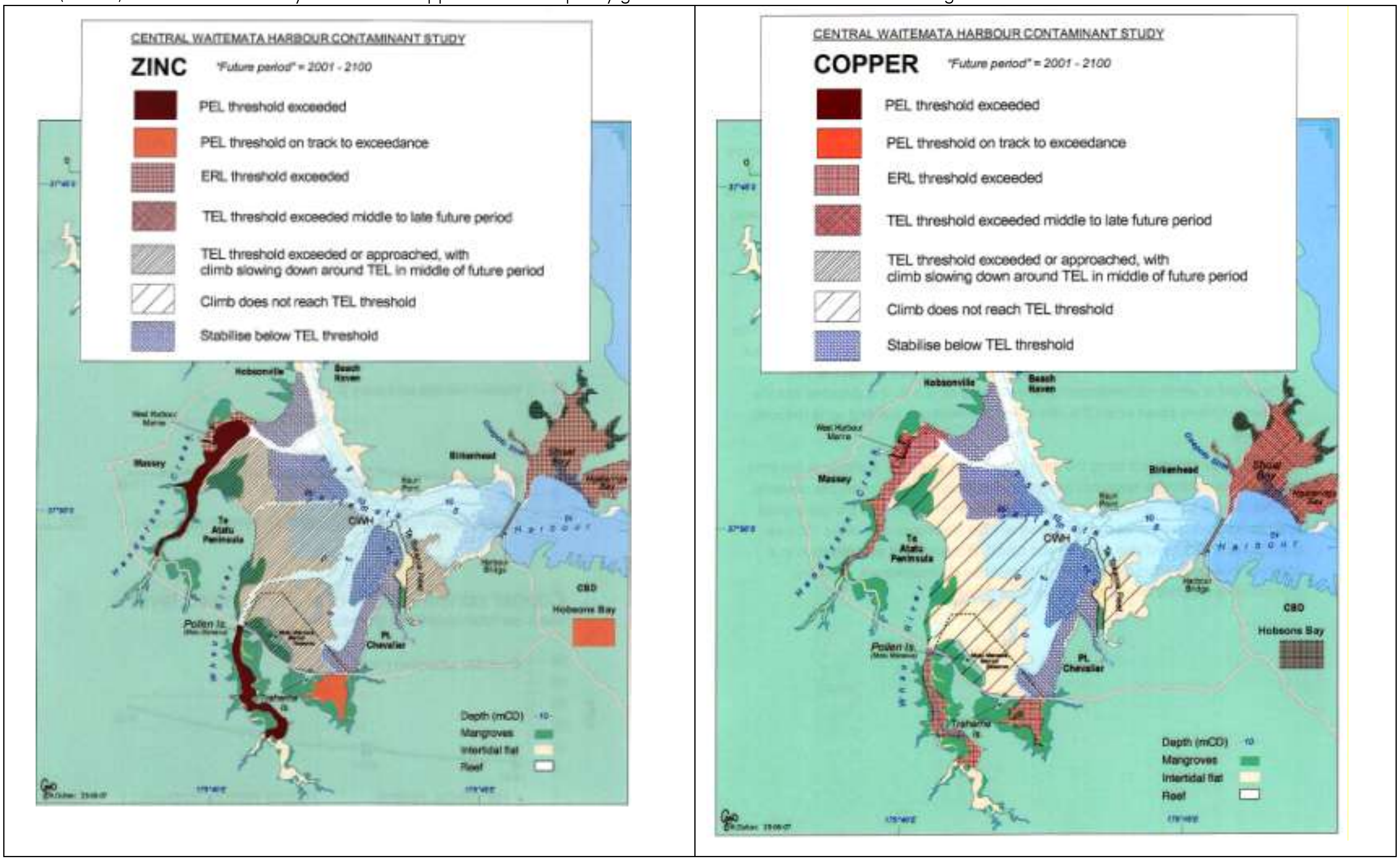
The predicted changes in Zn and Cu concentrations vary depending on the changes in sediment deposition/transportation that occur over time in the different types of estuarine environments around the harbour:

- In tidal creeks and sheltered embayments that will experience virtually constant sedimentation throughout the future period, zinc and copper concentrations are predicted to rise continuously but at decreasing rates. The tidal creeks (Whau, Henderson) will exceed the Probable Effects Level (PEL) for Zn by 2100, Hobson Bay and Waterview embayment shortly after. Shoal Bay will exceed Effects Range Low (ERL) by 2100 (Zn) or soon after (Cu).
- The situation is more complicated in subestuaries that will remain depositional, but with a decrease in sedimentation rate partway through the future period. The rise in zinc and copper concentrations early in the future period slows down when the sedimentation rates drop, but the rise is not fully arrested. The Threshhold Effects Level (TEL) in most of these subestuaries will be exceeded, but late in the modelling period (2050-2100).
- For subestuaries that will become transportational partway through the future period the situation is different again. Zinc and copper concentrations in the subestuaries reach a steady state partway through the future period. The heavy metal concentrations are predicted to stabilize below the TEL.

Although the detailed predictions of contaminant accumulation vary around the harbour, the modelling demonstrates a key feature of the future impacts of stormwater discharges – ie that Zn and Cu will continue to rise in many receiving estuaries, despite predicted changes (reductions) in contaminant loads associated with roof replacement, source control and realistic stormwater treatment scenarios. However, the increases are largely concentrated in the tidal creeks and embayments (Whau, Henderson, Waterview, Hobson Bay), and in Shoal Bay. Most of the wider harbour areas are predicted to reach a steady state condition, or to slowly increase to TEL late in the modelling period (2050–2100).

Figure 39

Green's (2008b) schematic summary of zinc and copper sediment-quality guideline threshold exceedance throughout the harbour.



8.7 The fate and transport of sediment, zinc and copper in the South Eastern Manukau Harbour

Application of the USC-3 model to a complex estuarine system whose catchments are undergoing partial urbanisation

The USC-3 model was applied in the South Eastern Manukau Harbour (Figure 40) to understand future contamination and the impacts of various management actions in source control and stormwater treatment (Green 2008d-e). The SE Manukau is different from the Central Waitemata in that the degree of urbanisation is far less and more recent and mostly confined to the northern and north-eastern catchments. The rural catchments to the south are minor sources of zinc and copper but major sources of sediments (Figure 40). The USC-3 model allows for the dilution of anthropogenic zinc and copper from urban areas with relatively uncontaminated sediment from the rural areas.

Figure 40

Division of the catchment of Southeastern Manukau Harbour/Pahurehure Inlet into sub-catchments for the purposes of application of the USC-3 model. (Reproduced from Green 2008e.)

As in the Central Waitemata Harbour, the fate of sediment and metals discharged to the SE Manukau could be tracked to the various subestuaries and the greater Manukau (where it would be widely dispersed). Of more interest are the sources of these contaminants in each subestuary (tidal creek, embayments and open tidal flats), because this identifies where management effort should be best placed to protect sub-estuaries at risk. Examples of the sources of Zn or Cu are shown in Figure 41 and 42.

Figure 41

Sources of Zn for some specific subestuaries (shown in blue hatching) in the SE Manukau. The proportion of total zinc loads from major source catchments is shown as a % (Green 2008e).

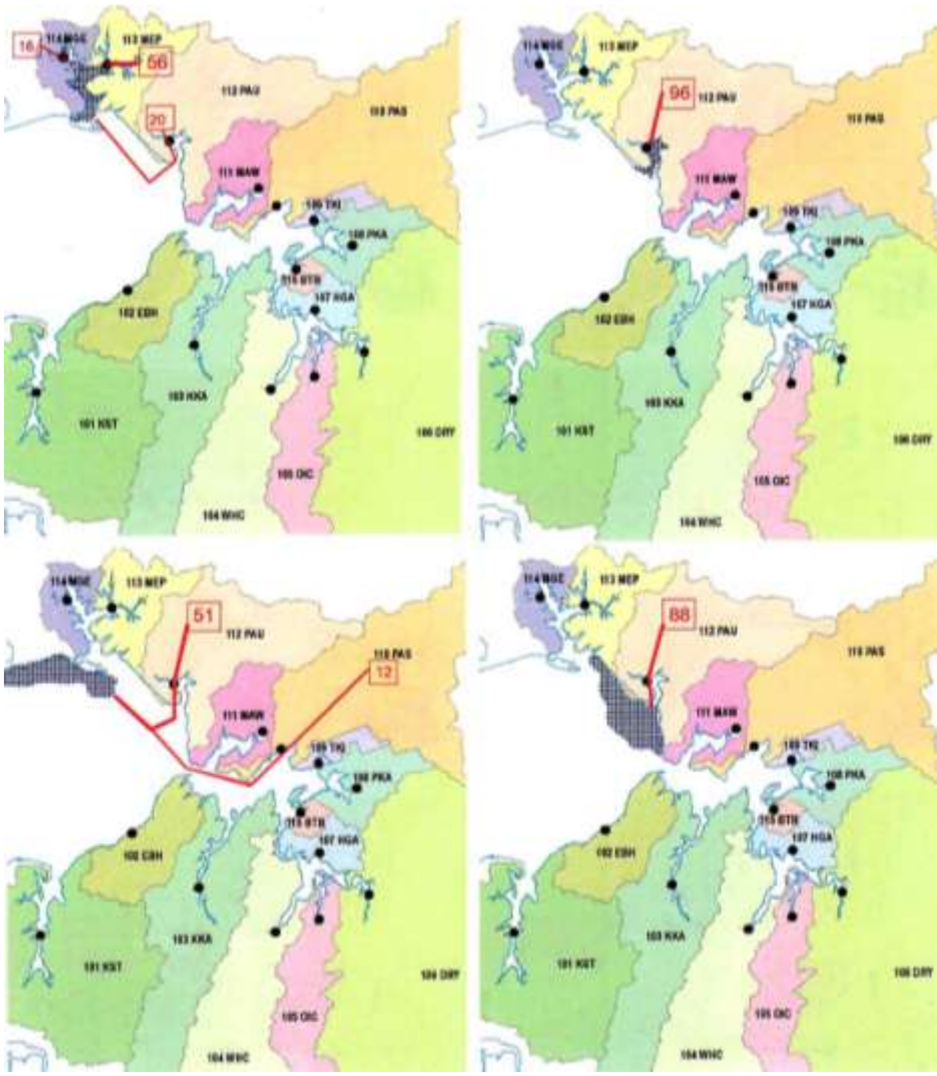


Figure 42

Sources of Cu for some specific subestuaries (shown in blue hatching) in the SE Manukau. The proportion of total Cu loads from major source catchments is shown as a % (Green 2008e).

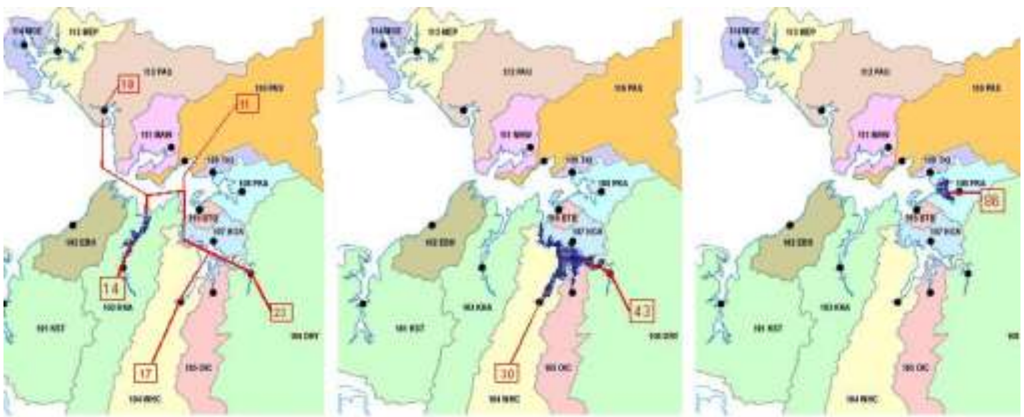
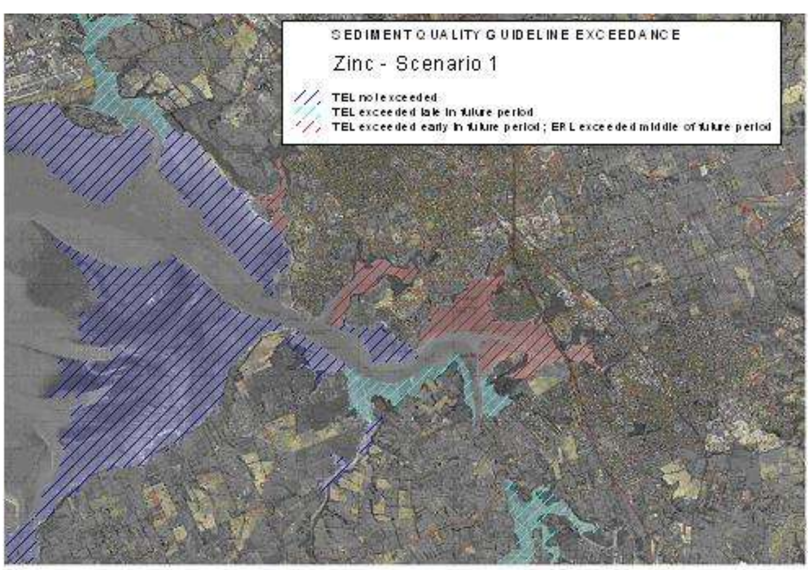


Figure 43 provides a summary of the metal-concentration predictions for Scenario 1 (ie no additional stormwater treatment or source control). There is no threat on the horizon for subestuaries designated "TEL not exceeded" (dark blue hatching). These subestuaries are the intertidal flats of Southeastern Manukau Harbour and tidal creeks that drain predominantly rural sub-catchments.

Figure 43

Summary of the Zinc concentration predictions for Scenario 1 (ie no additional stormwater treatment or source control.) (Reproduced from Green 2008e.)



The threat is low for subestuaries designated “TEL exceeded late in future period” (light blue hatching). These subestuaries include the larger harbour and intertidal flats on the southern side of the Pahurehure Inlet, Drury Creek and Pukaki Creek.

The threat is heightened in the remaining subestuaries, in which the zinc TEL is predicted to be exceeded early in the future period, and the zinc ERL is predicted to be exceeded in the middle of the future period (red hatching). Management may need to act now (or soon) to safeguard ecological values in these areas. These subestuaries are Puhinui Creek tidal creek, and the inner, sheltered parts of Pahurehure Inlet, all of which drain major urban centres with large areas of industrial galvanised roofs. Exceedance of copper sediment-quality guideline thresholds is predicted to be somewhat delayed relative to zinc.

8.8 Implications from modelling on sources and fate of contaminants

The models encapsulate our current understanding on the magnitude and location of sources of sediment, zinc and copper, within each sub-catchment, the way the receiving waters process these contaminants and the susceptibility of different parts of the coastal zone to sedimentation and contamination. Thus the model outputs tell us a great deal about where contaminants end up and where they have come from. This provides management with the tool to identify where it is best to implement practices to reduce contaminant inputs.

The modelling predictions are a good summary of our understanding of sources, fate and impacts of contaminants associated with urban stormwater in the wider marine environment. Their outputs are “state-of-the-art science” and thus encapsulate the physical and chemical aspects of stormwater impacts.

The USC-3 and CREA models and their related contaminant inputs models (eg, USC3-CLM, GLEAMS) are also effective Pressure-State-Response models because they are able to relate pressures (sources) to the state of the receiving environment and investigate the effect of management response (eg, source control, stormwater treatment). They are very effective PSR models because they are able to deal with the complexity of the sources (multiple catchments and land use) and receiving waters (interconnected tidal creeks, tidal and subtidal flats, embayments and harbours).

There are a number of implications about the effects of stormwater from the modelling process. In discussing these implications it is important to point out that the model “is a model” and uses mathematical equations to simulate processes; ie, it is important not to exactly attribute reality to model processes, even though the modeller tries to mimic the most dominant processes as accurately as possible. On the other hand, is it not appropriate to ignore implications of model predictions, but to acknowledge that the model may be pointing to some hitherto unknown process or that known processes are greater or smaller than commonly assumed.

One of the most important implications arises from the calibration of the USC-3 model, which involved allowing the "metal retention factor" (the proportion of Zn or Cu retained in the harbour) to vary to match predictions and observations. This factor is about 40 per cent, ie, the model implies that 60 per cent of the metal that reaches the Central Harbour is lost to the Hauraki Gulf. If this reflects the reality, then this is a major paradigm shift about heavy metal retention, because previously it has been assumed that metals are mostly bound to particulates and not dissolved, and most are retained within the estuary/harbour. This factor may reflect loss of metals from the Central Harbour such as desorption/dissolution of metals from sediments and flushing from the harbour, and losses to deeper sediment from mixing. It also fine-tunes other loss processes (such as resuspension/flushing of fine particulate matter and adsorbed contaminants) already included in the model. Because it is an important factor in the model, and it has major implications for our understanding of metal fate and effects, further work on factors contributing to the "retention factor" (eg, sediment mixing depths) is warranted.

Another key implication from the modelling process is the predicted decrease of Zn loads in the next 20 years because of the need to replace existing galvanised roofs as they reach the end of their useful lives with other sheet materials. Despite substantial drops in Zn loads, concentrations in the harbour sediments are still predicted to increase, but at much slower rates than in the recent past.

The technical review committee examining the model outputs pointed out that the model does not include the cycling and fate of dissolved metals. The USC3-CLM produces dissolved metal loads but for simplicity these were assumed to fully attach to TSS before dispersion by the USC-3 model. Processes that model the fate of fine particulates will compensate for processes involving dissolved metals, as will the metal retention factor. However, in the light of the relatively low metal retention factor, as well as other information (See Section 5 – Marine Water Quality) it would be instructive and more realistic to include dissolved metals in future models.

8.9 Conclusions

The CREA and USC3 models provide detailed information on the fate of contaminants and their build up in sediments for Waitemata and Manukau harbours. In the CREA modelling, catchment loads from 1999 are assumed to remain constant for the next 150 years. This is a major limitation because more recent work (see Section 3.3) has suggested that loads of Zn and TSS will decrease substantially, while Cu loads increase. This is to be addressed in the future with the more recent methodology developed in the CLM modelling suite. The CREA modelling is also applied to specific water bodies and does not allow for inputs from other catchments, or the interaction of these inputs. Therefore, although the CREA modelling is a substantial improvement on the USC-1 model, because it is able to disperse and settle sediments and contaminants more realistically, it does not include the more holistic approaches incorporated in the USC-2 and USC-3 models (which include inputs from other catchments). Nevertheless, it should perform reasonably well for estuaries whose contaminant inputs are dominated by their immediate catchment (eg, Whau, Hobson Bay, Tamaki Estuary). It is likely that it would be less successful at predicting outcomes for more complex estuaries with strong connections to other catchments and other sub-estuaries (eg, Western Bays, NE Manukau).

The CREA model also includes predicting indicator bacteria at beaches during and after storms, which provides useful additional predictive ability.

The USC-3 model predicts that Zn and Cu concentrations in harbour sediments (Outer Zones) increase slowly and exceed TEL levels (and in Shoal Bay ERL levels) in the distant future. It confirms that contaminant build-up will be much more rapid in urbanised tidal creeks (than in outer harbour areas) and that the levels of these metals will become quite high (eg, Whau, Henderson, Puhinui Creek). Concentrations in the SE Manukau are predicted to increase more slowly and to a lesser extent than the Central Waitemata Harbour because of newer and smaller urban areas and greater inputs of rural sediments. These outcomes are summarised simply and usefully in maps (eg, Figures 36 – 43).

The USC-3 model predicts that sediment inputs will decrease to the Central Harbour, and that this will result in some areas that were depositional (ie, accumulating sediment) becoming transportational (ie, temporarily store deposited sediment). The model predicts that this will decrease the rate of build up of contaminants, which will lead to a "plateau" in Zn and Cu concentrations in the future. Sediment loads in the SE Manukau will continue to be relatively high in the future because of the large rural catchments to the south. In the USC-3 model, metals and sediment loads from the sub-catchments are "decoupled" so that future trends in the fate of sediment and metals are not identical (although they are both transported as particles) and so differential transport processes (identified in the hydrodynamic model) can result in greater or lesser dilution of metals by incoming sediments.

The USC-3 model links the deposited metals and sediments with their sources, so provides the means to decide where to focus management efforts for greatest effectiveness, eg, to protect sensitive receiving waters. For example, Henderson Creek is the largest contributor of metals and sediment to the Central Harbour (it is by far the largest catchment) but it is also a significant source of sediments and metals for Shoal Bay, which is some distance away – and this is a surprising result. The ability of models such as the USC-3 and USC-2 to describe both the fate and sources of contaminants is their most powerful feature. Knowing predicted sources and fate will also help focus monitoring efforts for sediment chemistry and benthic ecology.

Predictions of contaminant fate have been made for many of Auckland's estuaries and harbours and most of the important stormwater receiving areas have now been modelled. This predictive work is not complete because some of the modelling will need to be revisited (eg, CREA, USC-1) with improved stormwater input loads. Additionally, it is recommended above that further refinements to models are made through:

- further checks on some of the model assumptions (through studies or sensitivity analysis);
- including other fate and transport mechanisms to make the models more realistic (and extend their application); and
- extend to other contaminants.

While further refinements to some of the predictions may be necessary, these models have already provided a great deal of confidence in understanding what the future holds for stormwater contaminant fate and effects.

9 Impacts on Marine Benthic Ecology

The concept of benthic macroinvertebrate community health

Healthy benthic macroinvertebrate communities are usually found at natural sites that are not impacted by anthropogenic activities. The structure of these communities (ie, types and number of animals) may be highly variable, but this is due to natural, local environmental conditions (eg, changes in sediment texture because of changes in exposure to waves). Communities at sites that have been impacted by anthropogenic activities are regarded as "unhealthy" when the benthic community structure has been changed by these activities. This defines the concept of benthic health.

A large number of animal species inhabit the soft sediment intertidal areas of the Auckland region. For example, the 84 sites included in the Benthic Health Model (BHM) had 102 different types (taxa) of animals. Within the urbanised estuaries and harbours of Auckland, these animals are exposed to stormwater discharges and any of the changes brought about by these discharges. Measuring the abundance and diversity of animals in sediments is an appropriate measure of the effect of stormwater on ecological communities because:

1. A diverse range of organisms live in the estuarine sediments.
2. Organisms are sedentary so animals at any one site are exposed to the stormwater discharge regime at that site (ie, the animals don't move around and experience different exposures).
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. Contaminants in stormwater run-off accumulate in sediments within marine receiving environments.
5. Sediment contaminants can reach concentrations that have been shown to be toxic to sediment dwelling organisms.
6. Some contaminants can bioaccumulate to concentrations in sediment dwelling organisms that are potentially toxic to animals higher up in the food chain even at relatively low concentrations in water and sediment. While this has not been shown for Auckland, this mechanism is of major concern overseas, eg, for DDT, PCB and Hg.

9.1 What was known by 2005

Contaminants derived from urban run-off (Zn, Cu and Pb) have accumulated in estuarine sediments, reaching concentrations that are potentially capable of causing adverse biological effects. 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, relatively low levels of contamination (in most places) compared with known biological effect thresholds, other chemicals that have not been measured and natural variation in biological populations. Nevertheless, up to 2005, very significant advances were made to overcome these difficulties, through:

1. Methods of analysis.
2. Specific detailed studies of benthic ecology in urban estuaries.
3. The accumulation of regional long-term contaminant and ecological monitoring data in both urban and rural areas.

It required the development of sophisticated multivariate statistical methods, coupled with consistent contaminant sampling and analysis protocols, to tease out statistical relationships between a pollution gradient and benthic animal community composition and present them in an easily understandable form. The Benthic Health Model was a breakthrough in showing that the composition of benthic animal communities in urban estuaries was correlated with a "contaminant gradient" which in turn reflected the degree of urbanisation in the estuaries' catchments. Hence the link between urban run-off and impaired ecological health in estuaries was more clearly demonstrated, even though the actual causes of the impaired ecological health remained unknown.

9.2 New work 2005-2008

Major advances 2005–2008 were improvements in the Benthic Health Model (BHM) and the commencement of scientific studies to understand the implications of the BHM and the underlying factors that might be driving the model. These and other advances were:

1. The key advance has been in the improvement of the BHM and the accompanying assessment of its implications (Section 9.3).
2. Research programmes have commenced to understand the reasons for the statistical relationships identified by the BHM, including methods for assessing multiple stressors on multiple animals (Section 9.4).
3. The extensive, long-term regional monitoring programmes have continued and have been expanded. These improve the understanding of benthic communities in a regional context as well as improving scientific understanding (Section 9.5).

4. A number of research studies have been undertaken on the pivotal species (species which have a major role in estuarine functioning or are important in the food chain because of size and/or abundance). These include the cockle (*Austrovenus stutchburyi*) (see Section 10 – Toxicity) and the mud crab *Helice crassa* (Section 9.6).

9.3 The Benthic Health Model

A regional model of benthic ecosystem health

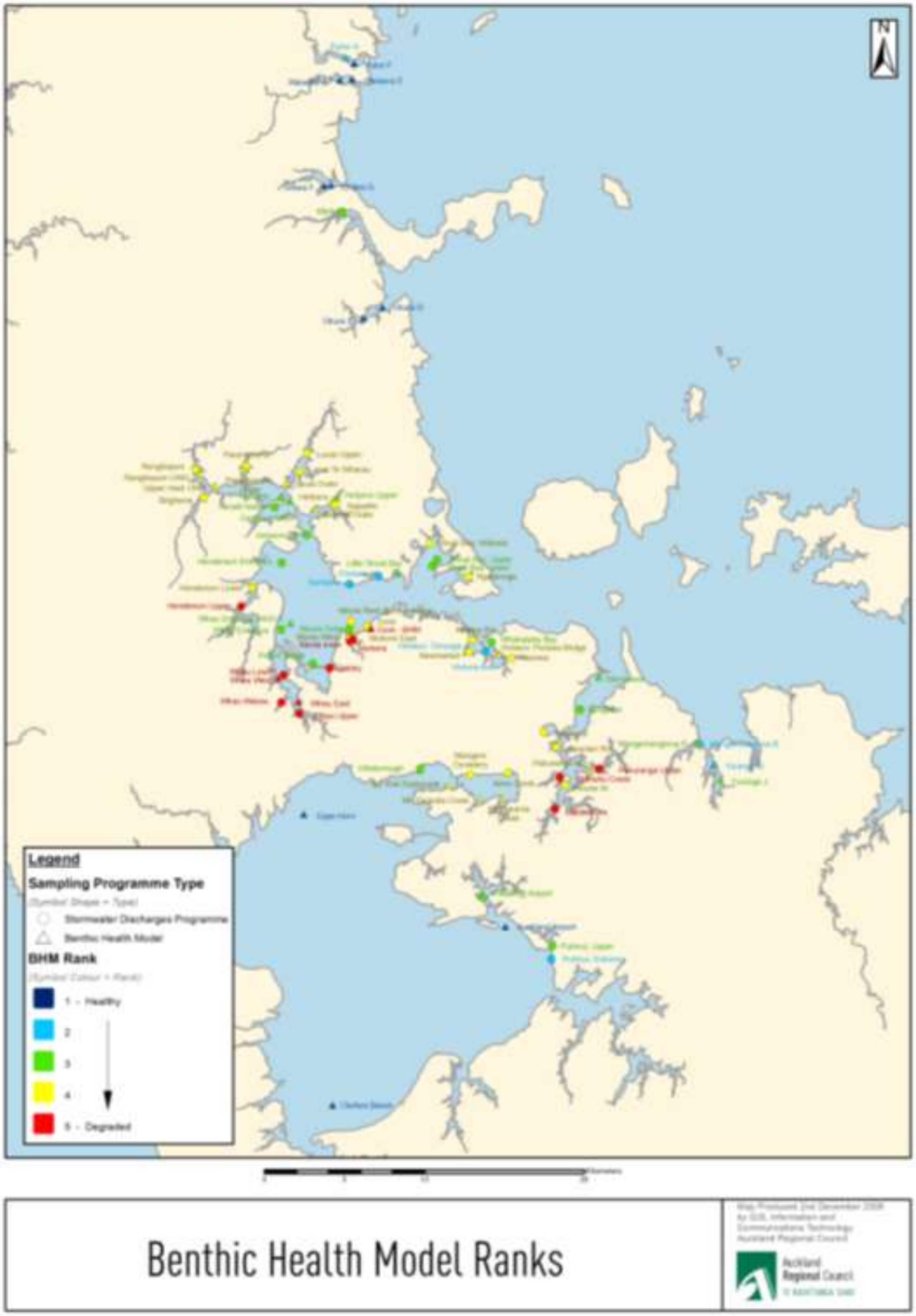
A major development described in the 2005 Review is the Benthic Health Model (BHM). Its subsequent refinements (Anderson et al. 2006) relate benthic community to a contamination gradient instead of a defined pollution gradient[11]. The most recent model was developed for 81 sites and tested (validated) on a further 14 sites (Figure 44).

In terms of benthic community "health", the observed ecological assemblages generally relate to the contamination gradients very well; and this, of course, forms the basis of the model. Note that this does not mean that the changes in benthic assemblages are soley related to the contamination gradient. Benthic community also correlates with other factors, but these have been "taken out" so that only the relationship between benthic health and contamination gradient are predicted and illustrated.

[11] The pollution gradient was defined by best professional judgement, while the contamination gradient is defined by concentrations of Cu, Zn and Pb.

Figure 44

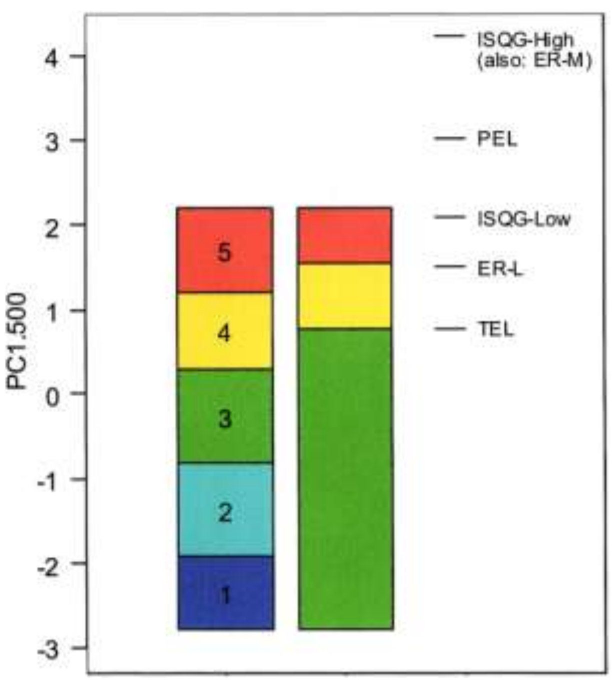
Sites showing grades for the Benthic Health Model (BHM) (from Anderson et al. 2006).



The degree of contamination is a single metric that includes concentrations of Cu, Pb and Zn. Clusters of 5 groups were identified along each gradient (in rank order from 1 = healthy to 5 = polluted). Groups 4 and 5 along the gradients coincide with existing "amber" and "red" sediment quality guidelines of the Environmental Response Criteria ("ERC") (ARC 2004b) (Figure 45). However, the axes also gave additional resolution and discrimination among healthier sites (groups 1-3). This indicates that there are effects on benthic community when concentrations of Cu, Pb and Zn are below the ERC trigger levels. Note however, that the benthic community "health" **correlates** to the contamination gradient, but the implied effects on community structure are not necessarily caused by these metals.

Figure 45

Pollution gradient grades versus sediment quality guidelines. The first bar shows the pollution grades using total Zn, Cu and Pb concentrations (expressed as Principal Component Analysis Axis 1.500). The second bar shows the ARC's Environmental Response Criteria and other Sediment Quality Guidelines (ER-L = Effects Range Low, ER-M = Effects Range Median (Long et al. 1995), TEL = threshold effects level, PEL = Probable effects level (McDonald et al. 1996, ISQG-low and –high are ANZECC (2000) Interim Sediment Quality Guidelines. Reproduced from ARC TP317 (Anderson et al. 2006).

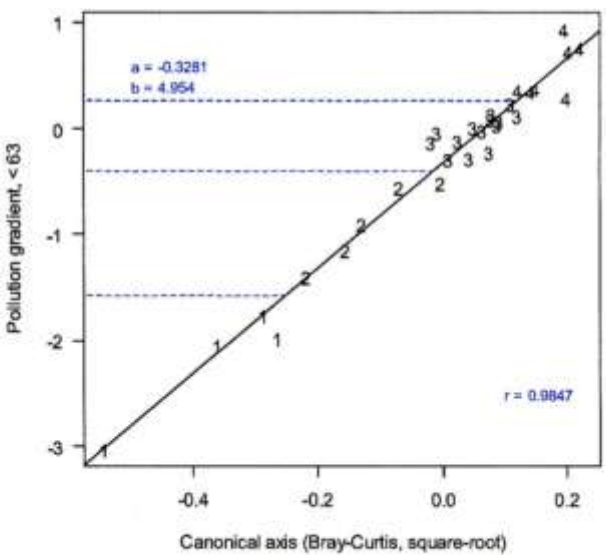


Methods have been developed for modeling the contamination gradient "axis" using ecological data using canonical analysis of principal co-ordinates (CAP). A multivariate computer software package, PRIMER v6, (MJ Anderson, Massey University, pers. comm.) will be commercially available and will be able to be used to carry out this analysis.

In the development of the model two physical groups of sites were identified: those having coarser sediments and greater exposure to wind and waves and those having finer sediments and lesser exposure. The best overall regional models were obtained using all sites together, regardless of their physical characteristics. An excellent model was also obtained by considering sites in coarse sediments alone and relating these to metal concentrations in the mud fraction (Figure 46).

Figure 46

Relationship between metal concentrations in the silt (<63 µm) fraction (Y-axis) and benthic community (X-axis) for sites with coarse sediments (approximately corresponding to Outer Zones). The blue dotted lines demark the pollution groupings (Reproduced from Anderson et al. 2006).



What do the results of this analysis and the modelling imply? In a related paper, Hewitt et al. (2009) describe the dissimilarity between groups along the contamination axis. These groups differed from one another in community composition by more than 70 per cent. A number of taxa showed a ~50 per cent or greater decrease in relative abundance between groups 1 and 2. These taxa are deposit feeders, suspension feeders and large and mobile organisms, including cockles, wedgeshells and pipis. Taxa that increased in numbers were mostly polychaetes.

The information from the model has been used to look more closely at specific receiving waters. Kelly (2008) assessed all ecological information for Whau River Estuary, Tamaki estuary and Mangere Inlet. These estuaries contain areas that are the most contaminated in Auckland in terms of their ecological scores (Grade 4-5 = polluted). Despite this, Kelly (2008) recognizes that these areas still contain functioning benthic communities that continue to provide a range of functions and services. The estuaries are utilised by large numbers of wading and coastal birds, including a number of endangered species, and all contain a wide variety of fish species. These latter observations provide an additional perspective on benthic community health; the differences noted in the benthic communities through the BHM have not yet resulted in obvious, visible changes further up the food chain.

9.4 Multiple stressor effects on benthic invertebrates

The animals that make up the benthic community are likely to be affected by stressors in many complex and different ways

One of the most puzzling aspects of the benthic health model is what is driving the relationship between benthic animal community composition and the contaminant concentrations in marine sediments adjacent to urban development. Thrush et al. (2008) propose that effects are probably due to multiple stressors, which can interact between themselves in many complex ways. They conducted a scientific study to develop a methodology that would provide insights into multiple stressors and their relationship with one another.

They used the benthic community site data from which the BHM was developed and validated to examine the relationship between the numbers of animals (abundance) and potential stressors. Potential stressors chosen in this study represented habitat requirements (sediment particle size), food (sediment organic content) and heavy metal contamination (copper, zinc and lead). The study used 46 different macrofaunal taxa at 84 sites across Auckland. For each animal, a model or relationship was developed using multiple regression between abundance and each of 6 potential stressors. The final model for each animal included only those stressors having a significant correlation to numbers of animals.

The potential response observed between abundance and any environmental stressors can range between linear and logarithmic decreases, threshold decreases, unimodal (increases in abundance followed by decreases), threshold increases, linear or logarithmic increases, or no response. When abundance correlates to the sum of stressor "values" the effect is called additive. Alternatively, the stressors can interact and affect animal response. These multiplicative effects can be antagonistic (a stressor will "cancel" the effect of another stressor) or synergistic (a stressor will enhance the effect of other stressor). Clearly, the possible responses are very complex when considering whole benthic communities and many stressors.

Simple responses to either a single stressor or stressors acting additively were less apparent in the models, than were multiplicative effects. Overall, the study found additive responses for 21 animals, but in only six cases were the models composed of only additive effects. Multiplicative effects (ie, where two or more stressors are interacting) dominated, occurring in models for 27 taxa; of these 12 synergistic and 12 antagonistic interactions were identified. Most multiplicative effects occurred between habitat and contaminant variables.

This study is of primary interest to researchers studying multiple stressors rather than to the management of stormwater effects. However, it does have some important implications for understanding stormwater effects. It illustrates the important point that the causative stressors associated with stormwater effects may be multiple, that these multiple stressors may be interacting between themselves in complex ways, and the stressors and interactions may be different for different animals.

9.5 Regional monitoring and research programmes

Multiple studies provide regional perspectives on benthic communities

The ARC has developed an extensive series of programmes to monitor the "health" of the benthic communities in the region and hence the overall ecological health of Auckland's estuaries, harbours and embayments (Table 11, Figure 47). The BHM programme is one of these regional monitoring programmes. The regional perspective is important here because it provides context within which to assess the effects of urban stormwater. Different programmes are designed for different purposes, and only some explicitly monitor the impacts of urban development or mature urban areas.

Most of the regional programmes include measures of habitat quality such as sediment texture, sediment organic matter, benthic chlorophyll content and/or sediment deposition in traps. Only the Upper Waitemata Harbour and the BHM programmes measure contaminants associated with urban stormwater.

While the regional programmes are designed for specific purposes, in some cases they may provide additional information of stormwater impacts. This specific and potential information includes:

- State of the Environment (Manukau, Waitemata, Mahurangi, Meola Reef).
- Long-term cycles that affect habitat quality, such as those generated by El Nino/El Nina effects (especially SoE programmes).
- The importance of the sediment textural gradients within estuaries (especially Mahurangi, Okura, Orewa, Whitford).
- Effects caused by excessive inputs of fine sediments (from soil erosion) and excessive sedimentation (also strongly linked to the process of urbanisation and rural land use)(eg, Okura, Orewa, Whitford, Upper Waitemata Harbour, Long Bay).
- Changes accompanying future urban development (eg, Okura, Orewa, Upper Waitemata Harbour, Long Bay).
- Whether urban stormwater is causing far-field impacts (eg, Manukau Harbour, Waitemata Harbour, Long Bay).

The programmes cover a large part of Auckland's marine near-shore environment (Figure 47). A brief "snapshot" is provided below of these programmes.

Table 11

Benthic ecology monitoring programmes in the Auckland region.

Location	Reference
Whitford Bay (Mangemangeroa, Turanga, Waikopua)	Anderson et al 2007
Meola Reef	Ford et al. 2006
Manukau Harbour	Hewitt & Hailes 2007
Mahurangi Harbour	Cummings 2007
Central Waitemata Harbour	Halliday et al. 2006
Puhoi, Waiwera, Orewa, Okura	Anderson et al. 2007
East Coast Bay beaches and rocky reefs	Anderson et al. 2007
Upper Waitemata Harbour	Hewitt et al. 2007, Miller et al. 2008
Tamaki Estuary	Hayward & Morley 2005
Weiti Estuary and Keripiro Bay	Hewitt 2008

The **Manukau Harbour Ecological Monitoring Programme**, established in October 1987, provides a stocktake of resources under ARC stewardship; feedback on harbour management activities; and a baseline against which future cause-effect or impact studies can be conducted. There is no evidence to suggest detrimental effects on ecosystem health within the extensive intertidal flats that make up the main body of the Manukau Harbour (Hewitt & Hailes 2007). This clearly demonstrates that urban land use in the Manukau Harbour catchment has had little impact on the wider harbour to date.

The **Mahurangi Ecological Monitoring Programme** is a State of the Environment programme and is one of the key programmes assessing the effect of deposition of fine sediments from rural activities in Auckland estuaries; which can also occur under the urbanisation process. Since its inception, the programme has raised concerns about declines of species sensitive to increased suspended sediment concentrations and deposition of fine sediments including two ecologically important bivalve species, *Macomona liliana* (wedge shell) and *Austrovenus stutchburyi* (cockle). This has resulted in an investigation of the sources of the sediment and to implement sediment controls.

A monitoring programme encompassing seven **rural** estuaries (**Puhoi, Waiwera, Orewa, Okura**, and three arms of the Whitford embayment, **Mangemangeroa, Turanga and Waikopua** (Anderson et al. 2007) has been underway for seven years. The Whitford embayment and Okura will receive potentially greater inputs of fine sediment from rural intensification, while the four northern estuaries have/will receive run-off from Motorway development. In addition, Orewa has a significant proportion of its catchment in urban land use, while more is under urban development.

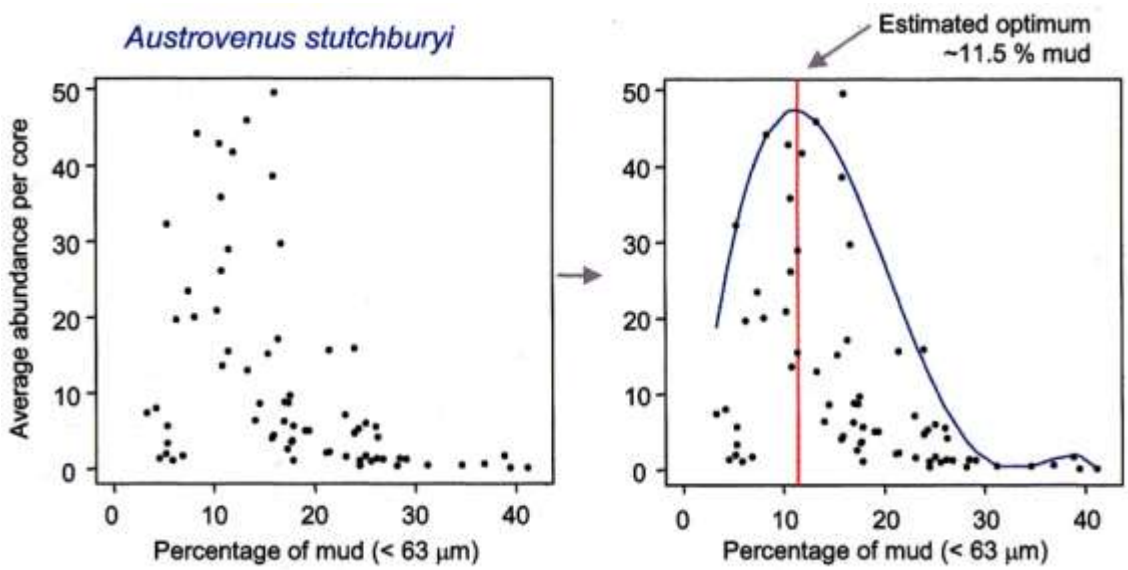
Figure 47

Locations of regional benthic ecology programmes.

The study shows that ambient sediment texture and the amount of sediment collected in traps poised above the seabed explain most of the important gradients in biotic variation, and a robust model was developed of benthic community structure in response to the proportion of mud in sediments (Anderson 2008). An example of the study findings is shown in Figure 48. Over the 7 years, changes of the benthic community and sediment texture at each site have been small compared with spatial differences between sites and estuaries. Populations of important shellfish, such as cockles, wedge shells, and pipis have remained stable. There has been no evidence of increased muddiness. The data provides an excellent baseline with which future changes can be detected, as well as providing regional perspectives (eg, there have been no regional increase in muddiness, no major changes in community structure) with which to help assess the overall state of ecological health of the Auckland region. These results indicate that the extent and the management of motorway, urban and rural development have not (yet) resulted in any significant deterioration (muddiness, community structural changes) in these estuaries.

Figure 48

relationship between taxa and percentage mud in surface sediments. Each point is an average of all samples taken at each site (6 cores x 12 times = 72 samples). The model for the 95th percentile is shown as the blue line, with the maximum of the model (red line) interpreted as the optimum condition for that specific animal (from Anderson et al. 2007, Anderson 2008).

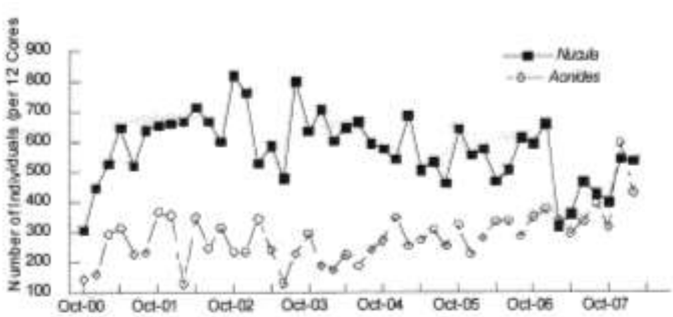


A long-term program was established in 2005 in the Upper Waitemata Harbour to monitor the status and trends in marine macroinvertebrate species, and monitor habitats that have the potential to be affected by sedimentation, pollution and other impacts associated with the development of the surrounding catchments (Hewitt et al. 2007, Miller et al. 2008). The programme has been running too short a time to yet detect trends. Organic content, Zn, sediment accumulation rate and sediment texture explained 69 per cent of the variation between site community compositions.

The focus of the **Central Waitemata Harbour** Ecological Monitoring Programme was to monitor the ecological status and trends of change in macrobenthic communities in the harbour (Halliday et al. 2006, Townsend et al. 2008). Five sub-regions were distinguished based on harbour hydrodynamics and drainage areas. The five sites integrate the effect of many anthropogenic inputs, while being distant from any specific source. Many changes in abundance of the monitored taxa have been observed, but, similar to the Manukau, these are primarily seasonal and multiyear cycles. All sites have shown minimal change in sediment texture over time. Much larger changes have been observed in community composition, suggesting that grain-size is not a predominantly controlling factor. A general trend observed across the Central Waitemata was a decrease in the abundance of the deposit feeding bivalve *Nucula hartvigiana*. There have been notable increases in other species. At every site annual and greater than annual patterns in abundance for most species were evident and an example is given in Figure 49. The trends in abundance that have been observed are not consistent with either increased sedimentation or contamination (Halliday et al. 2006) and may turn out to be long-term cycles in abundance.

Figure 49

Trends in abundance of the two dominant species Nucula hartvigiana decreasing and Aonides trifida increasing at the Hobsonville Site (from Townsend et al. 2008).



State of the Environment Monitoring of **Meola Reef** (Ford et al. 2006) began in 2001, with the objective of tracking long-term trends in community composition. These trends were then to be placed into a regional context by comparing them with those of the Long Bay Marine Monitoring Programme (see below). No temporal trends in community structure were detected on Meola Reef that suggested ecological change. Spatial patterns of communities were partly related to the sediment cover on the reef as well as gradients in currents. Diversity at Meola Reef was low compared to the diversity on the open coast from Campbells Bay to Waiwera (Anderson et al. 2005). This is not surprising, given the small amount of reef present at Meola, and the large amount of fine sediment present on the bed and in the water column, compared with the open coast.

A habitat survey of **Tamaki Estuary** have been carried out by Hayward and Morley (2005). Animals generally reflected changes in "quality" up the estuary (as measured by sediment and water quality monitoring programmes), changes from sandy to muddy substrate, and changes in habitat diversity, particularly the availability of hard, rocky substrate. Thus species diversity was highest in the outer estuary where water quality is strongly influenced by coastal processes, and there is a relatively high degree of habitat diversity. Diversity declined up the estuary and the authors note that this is consistent with reef habitat becoming scarce, water more turbid and sediments muddier. However, these changes are also consistent with the degree of contamination.

The **Long Bay Marine Monitoring Programme** is designed to detect the impact of urban development on the intertidal and subtidal marine environments in the coastal environments around Long Bay. The communities were monitored in two habitats: intertidal beaches and shallow subtidal rocky reefs. Reference sites are also monitored at Mairangi, Torbay and Browns Bay, while additional subtidal bays include Waiwera, Stanmore, Little Manly, Torbay and Campbells Bay. Potential impacts of increased sedimentation with changes in land use through time are of major interest, so sediment traps deployed at subtidal sites provide an indication of changes in water-borne sediment.

Differences among the beaches are likely to be caused by differences in their physical features and beach morphology. There is no evidence to suggest that there have been any specific impacts of urbanisation at Long Bay over the past seven years. Beach monitoring has been stopped.

There are strong geographical gradients in subtidal community structure, which correlated with increases in the average trap rate of sediments and increased variability in trap rate. The more sheltered southern bays showed biological patterns consistent with documented patterns for shallow reef areas having reduced wave-action and greater sediment loads and turbidity. There have been significant changes in community structure through time for all bays which was strongly correlated with decreases over the past seven years in the proportion of fine sediments (< 63μm) obtained in traps across the entire region. This observed trend of a decrease in the proportion of fine sediments is the opposite of what would be expected to occur under the scenario of increased terrestrial run-off from urbanisation and land works, and instead may reflect declining sediment yields from predominantly mature urban catchments.

An additional survey has been recently undertaken for the **Weiti Estuary/Keripiro Bay** system (Hewitt 2008). This will form the basis of resource assessment and the starting point to monitor changes from development in the Okura, Weiti and Whangaparoa areas.

There is a wealth of **scientific research** information on Auckland estuaries, which supports and is supported by the regional monitoring programmes. Most of the research addresses the reasons for the complex distribution and variation in benthic marine communities and uses benthic community data from the monitoring programmes described above. It is beyond the scope of the review to list or discuss all these, because they do not address the impact of urban stormwater specifically. A recent relevant example to illustrate this research is the study by Ellis et al. (2006) who developed a model linking the presence/absence of benthic intertidal fauna to changing environmental variables such as sediment characteristics, depth/elevation, tidal currents and wind-generated wave disturbance. The final model for each species contained between 1 and 6 environmental variables, where the percentage correctly predicted was moderate to high, ranging from 59 to 97 per cent. These models were developed for Whitford embayment and estuaries (Mangemangeroa, Turanga and Waikopua) and tested in Manukau Harbour, Mahurangi Harbour, and Puhinui Estuary, which cover a wide range of environmental conditions. Because one of these estuaries was Puhinui, the study findings will have some relevance for the overall assessment of stormwater impacts (ie, they will need to be integrated with other findings, specifically the BHM). The study also went some way in identifying relationships between higher level variables such as estuary type, basin morphometry and catchment-draining processes in driving macrobenthic community composition; thus indicating the possibility of developing a pressure-state (ie, catchment-benthic community) model.

Summary on regional monitoring programmes

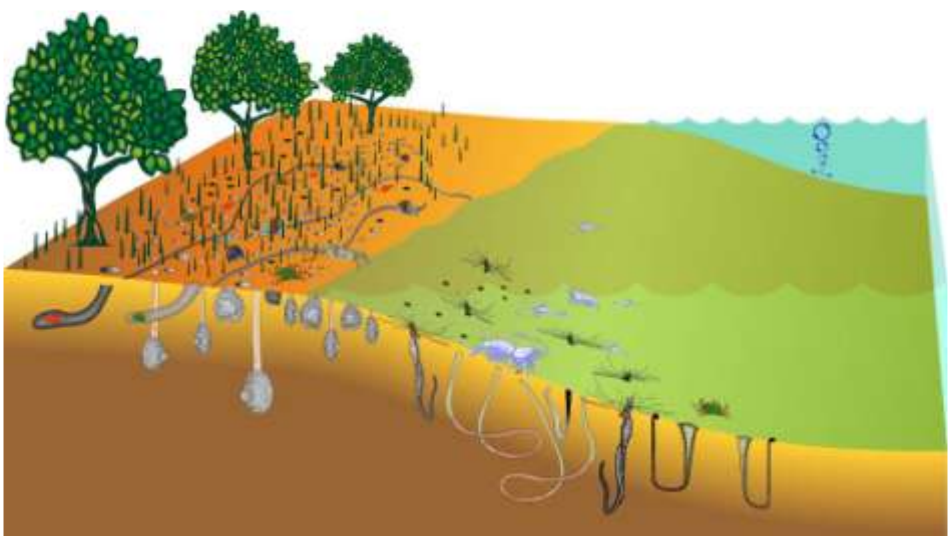
The regional monitoring programmes include some programmes that explicitly examine the impact of urban areas or urban development. Some of the other regional programmes are not designed to determine effects from stormwater, but because of their proximity to urban areas or areas undergoing development, provide some information on urban stormwater impacts. All monitoring programmes that explicitly or implicitly examine the effects of stormwater have not yet distinguished changes in benthic communities that can be attributed to urban stormwater discharges. For example, the most robust programme in the Manukau Harbour has not attributed any changes to urban stormwater, and this may be because the urban catchment is small relative to harbour size and flushing, as well as some distance from the monitoring sites. The Central Waitemata has a far larger urban catchment, and a clear chemical signal from urbanisation, but the programme has not yet distinguished changes that could be attributed to urban stormwater. The length of monitoring may be too short (7 years) and sites are not close to stormwater inputs. Other programmes with sites near significant urban areas (eg, Orewa, Lucas Creek) do not report urban stormwater effects, but monitoring may be too short, effects may be overwhelmed by rural inputs or urbanisation may be too recent. The strong benthic community gradients in the Tamaki Estuary reflect urban stormwater impacts, but the study authors identify other causative factors such as changes in rocky reef, muddiness and turbidity. There are many overlaps in the monitoring programmes and also research findings described above. It is probably timely that findings from all these programmes are integrated and compared with the BHM.

9.6 Bioturbation by the estuarine mud crab

Monitoring the benthic community health is the key programme assessing ecological health in Auckland. However, it is also important to understand the functions provided by the animals that make up the community, reasons for their presence or absence and their role in the overall ecosystem. To illustrate this we describe a recent PhD study on the mud crab (*Helice crassa*). This is a common inhabitant of Auckland estuaries that also lives in estuaries strongly affected by urban stormwater. It is important because it continues to occupy this niche, and because it influences the fate and bioavailability of contaminants. In particular, its burrowing activity mixes or "bioturbates" the sediment, thereby affecting the depth distribution of contaminants, the chemical environment in the sediment (Morrisey et al. 1999, Williamson et al 1999) and the bioavailability of contaminants (Williamson & Burgess 2003) (Figure 50).

Figure 50

Biological and physical processes, such as the burrowing and feeding activities of animals and sediment resuspension by waves, mix near-surface sediments. The depth of the surface mixed layer (SML) is indicated by the yellow zone. (From Swales et al. 2008).



Sivaguru (2006) studied the feeding and burrowing of the mud crab in relation to bioturbation of the sediment and effects of any contaminants in that sediment. The crab burrows extensively, creating new burrows and repairing existing burrows by pushing or carrying sediment to the surface. It is well adapted to feeding on a wide range of material. It possesses the physiology necessary to feed on muddy or sandy substrata at both low and high water. Sediment turnover rate, and consequently the total amount of new surface area exposed per unit time was very high and was greater than previously published data on the bioturbational activity of other crab species. Part of the reason why the mud crab is a robust species may be that it can regulate a wide range of ambient zinc and copper concentrations to approximately constant concentrations within its body. Another consequence of this is that the mud crab is not likely to be an effective bioindicator for these metals. However, it could be used as an indicator for higher levels of lead pollution in estuarine sediment, because crabs were found to accumulate lead in proportion to the concentration of lead in the sediment.

Overall, the study has comprehensively described the behaviour and physiology of the mud crab. In terms of stormwater impacts, it confirmed that the mud crab is an efficient bioturbator in urbanised estuaries, and has measured mixing rates of sediment across different physical environments. The crab is a potentially useful indicator for bioaccumulation of lead. It can be considered as a pivotal species in the natural functioning of urbanised estuaries in Auckland and its effect on stormwater contaminants. This type of fundamental research is important in providing robust information that can be used to understand fate and effects of contaminants.

9.7 Conclusions

The Benthic Health Model has provided very clear evidence that the health of the benthic community, as measured by the diversity and numbers of animals compared with natural sites, is strongly correlated with the urban stormwater-related contaminants Cu, Zn and Pb. The causative agents are yet to be confirmed through scientific study, but if they do not include Cu, Pb and/or Zn, then they must be strongly correlated to the concentrations of these contaminants. The BHM provides a means to classify sites in terms of the benthic community composition. Since 2005, the model has been refined and improved, and its applicability has been broadened to include Outer Zones. The BHM provides support for the monitoring of Cu, Pb and Zn as indicators of urban influence in the mud fraction of Outer Zones, which was originally based on the scientific arguments that this fraction more accurately reflected contaminant bioavailability than the total sediment.

The extensive regional monitoring programmes have continued and have been expanded. These programmes have provided a regional perspective on the effect of sediment texture gradients and the effect of excessive deposition of fine sediments. Variations in animals have been able to be explained in terms of short-term and long-term cycles, such as those caused by the El Niño Southern Oscillation.

Monitoring in estuaries and harbours have not yet been able to ascribe any observed changes in benthic communities through time to impacts of stormwater, although the programmes specifically designed for that purpose have only begun fairly recently. However, in the long-term, they will provide a robust benchmark for any assessing changes in urbanised estuaries and harbours. The BHM has found spatial differences in benthic communities that are correlated to urban stormwater contaminants. Given the wealth of information collected it is probably timely that the ecological monitoring and research programmes are compared to see if more information can be teased out on impacts of stormwater on estuarine ecology and related stressors.

10 Toxicity in the Marine Environment

Toxicity to animals in the marine environment can occur through exposure to toxic substances in the water column or by exposure to toxicants in the marine sediments and/or accumulation of toxic substances from contaminated biota in the food chain. Toxicity from contaminants in sediments is more likely, and this can occur through exposure to pore water (interstitial water) or through ingestion of sediment, or ingestion of biota that have bioaccumulated the toxicant(s).

Studies on cockles have demonstrated that their health, abundance and size distribution is adversely affected in urban estuaries, confirming their presumed susceptibility to urban stormwater. These comprehensive studies provide considerable detail on the nature of the effects on a key species.

A state-of-the-science review of PAH sources, distribution and effects in Auckland's marine and freshwaters has been prepared. PAHs in the receiving environment are tightly bound to particulate organic matter and only a small fraction (typically less than 5 per cent) can be extracted under conditions that mimic bioavailability to marine biota. On the basis of this and other evidence, the authors concluded that PAH concentrations in marine sediments are unlikely to be having adverse effects on marine biota at present and are unlikely to in the near future.

Laboratory-based toxicity tests were used to improve the understanding of the potential ecological effects of stormwater-contaminated sediments. Testing of sediments from ten estuarine RDP sites, spanning a wide range of contamination levels, showed only a weak toxicological response. Thus contaminants in Auckland sediments are, at most, only weakly toxic under the conditions of the toxicity test procedures. In contrast, the benthic ecology surveys described in Section 9 have shown a wide gradient of ecological changes associated with these sites, presumably reflecting a long-term, integrative response to all environmental perturbations. Combining toxicity test results with other measures of contamination–bioaccumulation of metals in oysters collected at the sites, and a simple measure of biotic diversity/abundance – in a weight of evidence (WoE) approach – gave a clearer picture of effects, which was approximately similar with the range shown by the BHM. As found in previous toxicological studies of Auckland's marine sediments, these results indicate that current laboratory toxicity tests alone are unlikely to provide a clear measure of potential ecological effects of stormwater. However, the advantages and limitations of the latest methodology were clearly demonstrated, thus providing better information on the most appropriate methods to be selected in future to address specific questions.

Overall, these studies have added to our understanding of stormwater toxicity in marine environments. The review of PAH brought together most of the information currently available to obtain a comprehensive assessment of risks to marine ecology. It would be timely to repeat this type of exercise for other major contaminants (eg, Zn, Pb, Cu, DDT, PCB). The cockle studies provide much greater depth of understanding of the effects of urbanisation on this important species. Laboratory toxicity testing differentiated a gradient of effects across a range of contaminated sites but only if combined with other measures in a Weight of Evidence approach. The BHM (see Benthic Ecology) remains the most powerful tool measuring the effect of stormwater but cannot identify the direct causal links between stressors and benthic animals, as described earlier. These links may be best provided by toxicity testing *in situ* under controlled conditions, where cause and effect can be identified. However, given the complexity of toxicity mechanisms and responses, this gap in our knowledge would be best addressed though more and extensive fundamental research.

10.1 What was known in 2005

The 1995 and 2005 Reviews of stormwater impacts gave a comprehensive review of observations on toxicity tests in Auckland. 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 decreasing sediment contamination gradient.
- Laboratory studies showed that the relatively low levels of contaminants (primarily heavy metals) associated with urban run-off pollution 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.
- Toxicity testing has continued to show that stormwater and stormwater pond sediments are mildly toxic to marine organisms. However, dilution in the marine receiving environment was expected to rapidly reduce these to non-toxic levels.
- PAHs exhibited phototoxicity to marine animals (amphipods). This will occur to some extent on the intertidal sediments surfaces. However, it was concluded that 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 because most animals do not live on the surface.
- A study on sediment chemistry and benthic ecology, coupled with laboratory testing of sediments and pore water gave ambiguous results. It was concluded that 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. The interaction of physical, biological, and chemical processes in marine sediments affects contaminant bioavailability and hence

toxicity, and complicates interpreting relationships between experimentally-determined toxicity and contaminant concentrations.

Ecological studies (see Section 9 – Ecology) have documented ecological effects that are correlated to the major stormwater contaminants Cu, Pb and Zn, and linked to sedimentation. While toxicity studies have provided some circumstantial evidence for the probability of toxic effects in the Auckland marine environment, they have not been able to confirm that the pronounced ecological differences observed between different areas are due to toxicity from contaminants. Understanding the effects of even individual contaminants on benthic ecology requires 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.

10.2 New work 2005-2008

Since 2005, advances have come from three areas of study:

1. Ecological effects of urban stormwater was studied on cockles (a major, "keystone" species in Auckland's intertidal marine environment) (Section 10.3).
2. A critical evaluation was made of the potential environmental effects of PAH (Section 10.4).
3. Toxicity testing was re-evaluated in the Auckland marine receiving environment (Section 10.5).

Additional advances are described in Section 9 on the Benthic Health Model and in correlative evidence of the effects of multiple stressors. Despite considerable efforts to clearly establish whether or not stormwater contaminants are toxic in aquatic receiving environments, a simple clear answer remains elusive.

10.3 Ecological effects associated with urban development on populations of the New Zealand cockle

Understanding reasons for the disappearance of cockles

Cockles are an important species in many Auckland estuaries. It is common to encounter beds of cockle shells buried in Auckland estuary sediments, which might reflect major catastrophic historical events. It is therefore appropriate that ecological effects of urban stormwater effects on cockles be examined more closely. A Ph D thesis by Stewart (2006) examines the ecological effects of urbanisation on the estuarine bivalve *Austrovenus stutchburyi* (New Zealand cockle). The effects of changes in habitat quality on cockle health were investigated using physiological (condition indices and glycogen content), individual (reproductive capacity) and population (abundance and size) indicators.

Figure 51

Example of the New Zealand cockle.

Cockles were studied in four estuaries (Whangateau, Okura, Tamaki and Mangemangeroa) along the east coast of Auckland, with different degrees of anthropogenic impact. Sediments in more highly urbanised estuaries contained higher percentages of silts (<63 µm) and higher concentrations of contaminants than the less urbanised estuaries. Numbers of cockles were lower in the more urbanised estuaries. Recruitment of cockles in estuaries was also lower. Cockles from the most urbanised estuary, Tamaki, did not spawn completely, with ripe individuals found throughout the year. Glycogen content and condition of cockles were generally lowest in this estuary, except during the period when cockles were not spawning, when higher condition and glycogen coincided with the retention of gametes.

The effects of changes in contamination were experimentally investigated by transplanting cockles from a clean environment along a pollution gradient in Tamaki Estuary. The gradient had both increases in fine sediment and contaminants. Survival of cockles was greatly reduced at the most polluted site. Cockles could survive in polluted environments, but only for a limited time (<1 year). Patterns in condition, glycogen content and reproduction in cockles along the pollution gradient reflected those patterns found in the four Auckland estuaries with their varying levels of contamination.

The changes observed along the pollution gradient provide evidence that existing changes in habitat quality (fine sediment and contaminants) associated with urban development, negatively affect the ecology of cockles.

A related study (De Luca 2007) examined sublethal stress responses in cockles. Several biochemical and physiological biomarkers were used to assess effects of contaminants on energetics, fecundity and growth of cockles. As part of this study, cockles were transplanted from an uncontaminated site to a series of uncontaminated and contaminated sites in one of Auckland's major harbours (Manukau Harbour). There were significant differences among sites, which indicated that the chosen suite of biomarkers could potentially be used in environmental quality assessment.

Overall, the cockle studies showed that, in relation to stormwater contamination, cockle survival, health, growth, and reproduction were markedly lower in urbanised estuaries with elevated levels of key urban contaminants. In addition, some biomarkers may be good indicators of potential problems.

10.4 Potential environmental risks of Polycyclic Aromatic Hydrocarbons in Auckland's aquatic environment

The effect of PAH are limited by their low bioavailability in Auckland's sediments

Polycyclic aromatic hydrocarbons (PAHs) are a well-documented class of persistent organic pollutants that are found above background levels in most urbanised and industrial areas. Monitoring has found that sediments in the urbanised parts of the Waitemata and Manukau Harbours and the Tamaki Estuary contain moderately elevated levels of PAHs. In 2005, the ARC commissioned a major review of sources, concentrations and environmental risk posed by PAH (Depree & Ahrens 2007). The aspects of this review related to sources of PAH identified by these authors is summarised in Section 3.

The majority of sediment PAH levels in Auckland's estuaries are well below ARC's environmental response criteria (Red ERC) of 1.7 mg/kg (for high molecular weight PAHs) and are currently not increasing discernibly, which suggests a low level of environmental risk to benthic estuarine biota.

Figure 52

Nevertheless, in a small number of estuarine locations and freshwater creeks (such as Meola, Motions and Oakley Creeks, the Whau River and the upper Tamaki Estuary), sediment PAH concentrations are markedly higher, and are close to, or above, ARC's ERC red criterion of 1.7 mg/kg. In these restricted locations, first-tier risk assessment suggests that observed PAH levels, combined with other contaminants, could pose an environmental risk to resident biota. Consideration of PAH bioavailability and toxicity was therefore warranted.

A targeted study involving seven estuarine and four stream sediments from the Auckland area was carried out to determine sources of PAHs, contribution from modern road run-off, bioavailability and acute toxicity of sediments. The results supported previous findings relating to the low PAH bioavailability of Auckland's estuarine sediments (see 2005 Review, ARC 2008). This was confirmed by complementary "selective" extraction methods employing semi-permeable membrane devices (SPMD's) and a synthetic gut fluid "cocktail" to mimic uptake of PAHs by sediment ingesting organisms. Results from SPMD extractions revealed that only 3-7 per cent of sediment PAHs are bioavailable via pore water exposure; while gut fluid extractions showed that only 0.1-3.4 per cent of the sediment PAHs were bioavailable via sediment ingestion route.

Short-term bioassays with amphipods did not reveal acute PAH-related mortality, although amphipod survivability in Auckland estuarine sediments was generally lower than the control sediment (Raglan Harbour). This observed toxicity was poorly correlated to sediment PAH concentrations, which combined with an absence of enhanced UV-induced toxicity, indicated that the reduced amphipod survival was attributable to some other sediment parameter. While the experiments performed cannot provide reliable information whether PAHs might exert chronic toxicity, the low 14-day extraction efficiencies by SPMD membranes suggests that only a very limited pool of PAHs is available for uptake by organisms from porewaters.

It is likely that a sizable fraction of PAHs is locked up in sediment organic carbon phases and therefore not readily available for uptake by benthic organisms. PAHs are not bound to the entire TOC pool, but rather to certain TOC sub-fractions. This hypothesis has been confirmed by field data showing more than 75 per cent of the PAHs are sorbed to low-density, organic carbon-rich fractions, rather than to TOC-coatings on heavier sediment particles.

The available data on PAH bioavailability in Auckland's estuaries indicate that there currently is only minor PAH accumulation in sentinel benthic organisms (eg, shellfish) and little evidence for PAH-related toxicity in estuarine benthic macrofauna, even in sediments with elevated PAH levels. The authors asserted that given the low bioavailability of sediment-bound PAHs and their lack of biomagnification, PAH-related effects in Auckland waters appear to be unlikely for pelagic organisms (eg, plankton and nekton) and higher trophic levels, such as fish, birds and humans. However, an earlier study showed that

PAHs were entering the food chain in Auckland and inducing biochemical "signals" typical of PAH contamination in flounder (Diggles et al. 2000, described in the 2005 Review).

The studies reviewed in Depree & Ahrens (2007) provide a comprehensive, integrated, assessment of the probably risks posed by PAH in Auckland's marine receiving waters. The authors concluded that their studies indicate a low risk from PAHs, both currently and in the near-to-medium future.

10.5 Toxicity of marine sediments

Can toxicity testing demonstrate effects in Auckland sediments?

Toxic responses to stormwater constituents have been proposed based on sediment contamination of sediments and the BHM. In 2005, a number of toxicity studies were undertaken in Auckland estuaries, but results were ambiguous and the relatively low level of contaminations (c.f. contaminated sites in more industrialised countries) was perceived to limit the applicability of laboratory toxicity tests. However, since then, continued improvements in toxicology methodology prompted the ARC to assess the ability of toxicity testing to further the understanding of the potential ecological effects of contaminated sediments.

A multi-faceted toxicity study was undertaken using sediments of varying levels of contamination, taken from ten sites from the Waitemata Harbour, Hobson Bay, and Tamaki Estuary (Hickey and Martin 2008). These sites, which are part of the ARC's RDP programme, have both sediment chemistry and benthic ecology information, which combined with tissue bioaccumulation in shellfish and sediment toxicity, provided a four-pronged ("quadrad") "weight of evidence" (WOE) effects' assessment.

The underlying basis of the WOE approach is that observations of elevated concentrations of contaminants (ie exceeding guideline values) in sediments alone are not in themselves indications of ecological degradation. The WOE approach is consistent with the ANZECC (2000) decision-making framework for contaminated sediments, and offers greater and more specific guidance on the nature of the environmental stressors that may be responsible for adverse effects.

A core group of toxicity bioassays (shaded in Table 12 below) were used to measure whole-sediment toxicity for acute (10 d) and chronic (30 d) exposure effects using amphipods and bivalves and pore water toxicity using algae and mussel embryo development. An additional four species/bioassays were included to provide indicative testing for non-standardised methods.

The species were selected based on a number of criteria related to method standardisation, sensitivity and relevance to the environment, and provided a range of trophic levels (eg, algae, invertebrates). These species were exposed to sediments, pore water and overlying water. Measurements included a range of sensitive endpoints (survival, growth and reproduction) together with sub-lethal behavioural measures such as reburial rate.

Table 12

Toxicity tests used to assess sediment and water toxicity (Hickey & Martin 2008).

Test type	Species	Test duration	Endpoints measured
Whole sediment – acute	Amphipod	10d	Survival
Whole sediment – acute	Shellfish – wedge shell	10d	Survival, morbidity (reburial)
Whole sediment – acute	Polychaete	10d	Survival
Whole sediment – chronic	Amphipod	30d	Survival, growth, reproduction
Whole sediment – chronic	Shellfish – wedge shell	30d	Survival, morbidity (reburial), growth
Whole sediment – chronic	Polychaete	30d	Survival
Whole sediment – surface	Alga	1d	Enzymatic activity
Sediment pore water – chronic	Alga	2d	Growth
Sediment pore water – chronic	Blue mussel embryo	3d	Development
Overlying water – acute	Mysid shrimp	7d	Survival

The results for the core toxicity tests for whole sediments and pore waters are summarised in Table 13 below.

The acute exposures indicated toxicity at two out of 10 sites (20 per cent), with chronic whole sediment tests toxic to either amphipods or bivalves in five out of 10 sites (50 per cent). Pore water was not significantly toxic to algae (but simulated growth in six out of 10 sites) and was not toxic to mussel embryos.

Organisms in the overlying water are exposed to contaminants that leach from the sediment pore waters and through direct contact and possible ingestion of surficial sediment. There was no significant overlying water toxicity.

The toxicity results were then combined with sediment contaminant chemistry, a simple measure of benthic community "health" (eg, taxa numbers and abundance), and contaminant bioaccumulation to produce an overall WOE assessment for each site. Note that the toxicity thresholds (Table 14) are quite low and relatively sensitive. The overall WOE assessment is summarised in Table 14.

Table 13

Summary of toxicity test results for whole sediments and pore waters (modified from Hickey & Martin 2008).

Site	Whole sediments				Pore waters	
	Acute toxicity		Chronic		Chronic	
	Toxic to amphipods?	Toxic to bivalves?	Toxic to amphipods ?	Toxic to bivalves?	Toxic to algae?	Toxic to mussel embryos?
Tamaki-Benghazi Rd	No	No	No	No	No*	No
Tamaki-Bowden Rd	No	Yes	Yes	No	No*	No
Tamaki-Princess St	No	Yes	Yes	No	No*	No
Tamaki-Middlemore	No	No	No	Yes	No*	No
Hobson-Awatea Rd	No	No	No	No	No	No
Hobson-Purewa	No	No	No	No	No*	No
Waitemata-Meola Inner	No	No	No	No	No*	No
Waitemata-Whau Wairau	No	No	no	Yes	No	No
Waitemata-Whau Upper	No	No	No	No	No	No
Waitemata-Henderson Upper	No	No	No	Yes	No*	No

* = stimulates growth.

The overall integrated effects assessment also included consideration of a statistical analysis of the relationships between biotic and physico-chemical measures. This analysis indicated that both physical and chemical components contributed to the biotic responses, with sediment ammonium concentrations being a potential contributing factor to the observed toxicity.

The WOE analysis involves consideration of the various lines of evidence (LOE), specifically: sediment chemistry, toxicity, benthic community and bioaccumulation. The overall WOE classification is based on integration of the various individual LOE classifications and defines the likelihood of chemical effects as: "No significant adverse effects"; "Potential adverse effects"; "Significant adverse effects"; and a decision assessment relative to the multiple lines of evidence.

The WOE analysis highlights that a "one-size-fits-all" approach is not appropriate for these sediment effects relationships, despite the site selection criteria applied in order to reduce factors such as habitat variability.

Cause-effect linkages were poorly identified with the routine suite of chemical contaminants. Chemical stressors not routinely measured (eg, pore water ammonia) and other factors may be adversely affecting community health (eg, via intermittent events).

Table 14

Weight of evidence (WOE) summary for contaminant chemistry, benthic community, toxicity, and bioaccumulation measures. The toxicity response levels for each test were categorised relative to the control site as: Low, L = reduction of 20 per cent or less in all toxicological endpoints; Moderate, M = statistically significant reduction of more that 20 per cent in one or more toxicological endpoints; High, H = statistical reduction of more than 40 per cent in one or more toxicological endpoints. The overall WOE descriptive assessment is based on integration of the results matrix.

Site	Chemistry	Benthic community	Toxicity	Bioaccumulation	Overall assessment
Tamaki-Benghazi	Low	Low	Low	Low	No significant adverse effects.
Tamaki-Bowden	Medium	Medium	Medium	Low	Significant adverse effects. Determine reasons for sediment toxicity and benthos alteration.
Tamaki-Princes	Medium	Medium	High	Low	Significant adverse effects.
Tamaki-Middlemore	Medium	High	High	Not assessed	Significant adverse effects.
Hobson-Awatea	Medium	Low	Low	High	Assess risk of biomagnification.
Hobson-Purewa	Medium	Medium	Low	Medium	Potential adverse effects. Determine reasons for benthos alteration.
Waitemata-Whau Wairau	Medium	High	High	Not assessed	Significant adverse effects.
Waitemata-Whau Upper	Medium	High	Low	High	Potential adverse effects. Determine reasons for benthos alteration.
Waitemata-Meola Inner	Medium	Medium	Low	Medium	Potential adverse effects. Determine reasons for benthos

					alteration.
Waitemata-Henderson Upper	Medium	High	High	Not assessed	Significant adverse effects.

The conclusions of the project were:

- Toxicity was generally low for a range of sites incorporating some of the most contaminated sites in the RDP programme. Chronic (long-term, 30 day) bioassays were generally required to detect effects. [However, holding sediment for 30 days may significantly alter their chemistry and hence toxicological effects].
- Patterns of toxicity indicated that the predominant causes of toxicity differed between sites in this contamination gradient based on total sediment metal levels.
- Comparison with sediment quality guidelines indicated that contaminants other than the routinely measured heavy metals (Cu, Zn, Pb) were potentially contributing to the contaminant risk. These included As, Hg and ammonia.
- Multivariate analysis identified both physical (particle size, TOC) and chemical factors (including Cu, Zn, ammoniacal-N) as correlated with biological responses.

This study reinforced the findings in Section 9 that the impacts of stormwater contamination on estuarine ecosystems are complex, and are likely to vary between locations. Determining the causative factors, and the toxic responses, remains a significant challenge.

10.6 Conclusions

Some progress has been made towards obtaining a better understanding of toxicity in urban-impacted estuaries. The studies summarised above showed that:

- Cockle survival, health, growth, reproduction was markedly lower in urbanised estuaries with elevated levels of key urban contaminants including fine sediment. In addition, some biomarkers may be good indicators of potential problems.
- PAH toxicity has been very well examined and a comprehensive assessment of the likely toxicity of PAHs presented. The authors concluded that PAHs are unlikely to be a widespread contaminant of significant concern in Auckland in the near-to-medium (eg, decadal) term.
- Estuary sediments are only weakly toxic during laboratory tests. A "weight of evidence" (WOE) approach provides a framework that integrates multiple possible contributing factors in assessing the potential toxicity (and overall effects) of marine sediments.

The cockle studies provide much greater depth of understanding of impacts of urbanisation on this important species. Given the complexity of toxicity mechanisms and responses, more fundamental research, such as these studies, is warranted. Sediment toxicity testing showed the advantages and limitations of the latest toxicity methodology. However, Auckland sediments are only weakly toxic under the conditions of the test, and so toxicity testing by itself does not help understand effects in the sediments, and a full Weight of Evidence approach is needed. The PAH review (Depree & Ahrens 2007) provided a comprehensive, integrated, assessment of the probably risks posed by PAH in Auckland's marine receiving waters. It would be useful to produce similar reviews for other major contaminants (such as Cu, Pb and Zn).

11 Acknowledgments

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. Thanks to all the agencies and individuals that have made the research findings available to us for this review. We hope this report does justice to their work.

12 Abbreviations

Acute toxicity	Rapid adverse effect (eg, death) caused by a substance in a living organism.
Additivity	Refers to toxicity where the environmental concentrations are below the levels where a specific chemical is known to effect organisms, but where groups of chemicals may act in concert, producing an additive adverse effect.
ANZECC	Australian and New Zealand Environment and Conservation Council.
Algae	Comparatively simple chlorophyll-bearing plants, most of which are aquatic and microscopic in size.
Base Flow Index (BFI)	Measure of what proportion of the total flow is baseflow (total baseflow in a year divided by the total flow).
Benthic health	see "Ecological integrity".
BHM	Benthic Health Model.
Bioaccumulation	General term describing a process by which chemical substances are accumulated by aquatic organisms from water, either directly or through consumption of food containing the chemicals.
Bioavailability	The fraction of the total of a chemical in the surrounding environment that can be taken up by organisms. The environment may include water, sediment, soil, suspended particles, and food items.
Biofilms	Thin films of microscopic organisms that occur on surfaces in films.
Biological community	An assemblage of organisms characterised by a distinctive combination of species occupying a common environment and interacting with one another.
Bioturbation	Mixing of surface sediment by the activities (eg, burrowing, feeding) of animals.
Bivalve	A mollusc with a hinged double shell.
BMP Best Management Practices	Activities, prohibitions of practices, maintenance procedures, and other management practices to prevent or reduce the discharge of pollutants to waters. BMPs also include

	treatment facilities (eg, stormwater wetlands), operating procedures, and source controls.
Cd	Cadmium.
Co	Cobalt.
Chla	Chlorophyll a.
Chronic	A stimulus that lingers or continues for a relatively long period of time, often one-tenth of the life span or more. Chronic should be considered a relative term depending on the life span of an organism. The measurement of a chronic effect can be reduced growth, reduced reproduction, etc., in addition to lethality.
CLM	Contaminant Load Model (Section 3.4).
	Model developed for prediction present day loads of Zn, Cu , SS and total petroleum hydrocarbons. USC3-CLM is a refined model developed for Central Waitemata and SE Manukau for load forecasting of Zn, Cu and SS.
CPEC	Chemicals of Potential Environmental Concern. A new group of chemicals of potential environmental concern, based on their toxicity, persistence, and widespread or on-going use. In contrast to the "priority pollutants" many CEPCs have a lower environmental hazard profile, notably a lower acute toxicity.
Community	An assemblage of organisms characterised by a distinctive combination of species occupying a common environment and interacting with one another.
Community composition	All the types of taxa present in a community.
Community metabolism	The biological movement of carbon in an ecosystem, involving two processes, production (via photosynthesis) and respiration.
Community structure	All the types of taxa present in a community and their relative abundances.
Conventional pollutants	Pollutants typical of municipal sewage, and for which municipal secondary treatment plants are typically designed; BOD, TSS, fecal coliform bacteria, oil and grease, and pH.
CREA	Coastal Receiving Environment Assessment. Modelling studies undertaken by Auckland City to assess impact of stormwater and wastewater discharges.

Criteria	The numeric values and the narrative standards that represent contaminant concentrations that are not to be exceeded in the receiving environmental media (surface water, ground water, sediment) to protect beneficial uses.
CSO	Combined Sewer Overflows. A discharge of untreated wastewater from a combined sewer system at a point prior to treatment plant. CSOs generally occur during wet weather. During periods of wet weather, these systems become overloaded, bypass treatment works, and discharge directly to receiving waters.
Cu	Copper.
DDT	dichloro-diphenyl-trichloroethane. Organochlorine pesticide (OCP) in common use from 1947 to early 1970s.
DRP	Dissolved Reactive Phosphorus (=SRP).
EC (Environmental Compensation)	Environmental Compensation (more commonly referred to as "offset mitigation" in the international literature) is something that is done when all practical steps have been taken to minimise adverse effects and relates to the residual effects that cannot be mitigated.
Ecosystem metabolism	The combination of gross primary production – GPP, and ecosystem respiration - ER).
Endocrine disruptors	Chemicals that mimic hormones and can potentially disrupt endocrin systems in animals.
Ecological integrity (health)	The "health" or "condition" of an ecosystem. The ability of an ecosystem to support and maintain key ecological processes and organisms so that their species compositions, diversity and functional organisations are as comparable as possible to those occurring in natural habitats within a region.
Enterococci	Any streptococcal bacteria normally found in the human intestinal tract; usually non-pathogenic.
ER	Ecosystem respiration.
ERC	Environmental Response Criteria. Trigger concentrations developed by ARC for waters receiving stormwater and wastewater discharges in the Regional Discharges Project.
ERL	Effects Range-Low. Long et al (1995) took a large database of sediment toxicity tests and screened it for samples labelled as toxic by the original investigators. The lower 10th

	percentile of these concentrations is the ER-L.
ERM	Effects Range-Medium. See ER-L. The ER-M is the median concentration of the database screened for samples labelled as toxic by the original investigators.
Fate	Disposition of a material in various environmental compartments (eg, soil or sediment, water, air, biota) as a result of transport, transformation and degradation.
FC	Faecal coliform.
Foraminifera	A large group of microscopic animals, usually producing a shell and mostly found in the marine environment.
GLEAMS	Groundwater Loading Effects of Agricultural Management Systems is a continuous simulation, field scale model for predicting run-off and soil loss from rural lands (Section 3.5).
GPP	Gross Primary Production. Production of organic compounds from atmospheric or aquatic carbon dioxide, principally through photosynthesis, with chemosynthesis being much less important.
Hg	Mercury.
hindcasting	Predicting past and present-day conditions or concentrations using historical information.
IC	Impervious Cover. %IC is the percentage of the catchment with impervious cover.
IP3	Industrial Pollution Prevention Programme conducted by ARC.
Macroinvertebrate	A small animal generally visible to the unaided eye, usually larger than 0.5 mm. These animals do not have a backbone. Common examples include insects, worms, shellfish, snails, crayfish.
Macrophyte	A member of the macroscopic plant life of an area, especially of a body of water; large aquatic plant.
Neurotoxin	Toxic substances which adversely affect the nervous system.
MfE	Ministry for the Environment.
NH_4, NH_4-N,	Ammonium, ammonium nitrogen.
Ni	Nickel.

NOx-N	Nitrogen oxide - nitrogen including nitrate (NO_3^-) and nitrite (NO_2^-).
NTU	Nephelometric Turbidity Unit – a measure of turbidity.
OCP	Organochlorine pesticides (eg, DDT).
OZ	Outer Zones (see Seeing Zones). Outer Zones are wider estuarine areas downstream of the settling zone or are located in higher energy environments were contaminants are less likely to settle permanently.
PAH	Polynuclear aromatic hydrocarbons.
Pathogen	An organism capable of eliciting disease symptoms in another organism.
Pb	Lead.
PBT	Persistence-bioaccumulation-toxicity. A measure of environmental hazard.
PCB	Polychlorinated biphenyls.
Percentile	Division of a frequency distribution into one hundredths.
PEL	Probable Effects Levels (McDonald et al. 1996). These sediment Sediment Quality Guidelines, like ER-L and ER-Ms, are derived databases of sediment toxicity bioassays or benthic community metrics. The database for TEL/PELs, however, is much larger than for ER-L and ER-Ms. TEL and PEL calculations also make use of the non-toxic results in calculating values.
Photodegradation	Breakdown of a substance by exposure to light; the process whereby ultra-violet radiation in sunlight attacks a chemical bond or link in a chemical structure.
Photosynthesis	The conversion of carbon dioxide to carbohydrates in the presence of chlorophyll using light energy.
Phthalate esters	Chemicals commonly used as plasticizers in plastics, paints and tyre rubber. They are part of CPEC.
Phytoplankton	Small (often microscopic) aquatic plants suspended in water.
Pollutant, Conservative	Pollutants that do not readily degrade in the environment, and which are mitigated primarily by natural stream dilution after entering receiving bodies of waters. Included are pollutants such as metals.

Pollutant, Non-Conservative	Pollutants that are mitigated by natural biodegradation or other environmental decay or removal processes in the receiving stream after in-stream mixing and dilution have occurred.
POP	Persistent organic pollutants (eg, DDT, PAH).
Pore waters	Interstitial waters. Water that occupies the space between particles in a sediment, as distinct from overlying water which is the water above the sediment layer.
PSR	**P**ressure (eg, catchment inputs) – **S**tate (eg, concentrations in receiving waters) – **R**esponse (eg, management intervention).
Petrogenic	Hydrocarbons of petroleum origin (eg, road/tyre abrasion and engine oil).
pyrogenic	Hydrocarbons of combustion origin (eg, combustion soot).
RDP	ARC's Regional Discharges Project.
RSSE	Relative Sen Slope Estimate. In trend analysis using Seasonal Kendall tests on raw data, the Sen Slope Estimator (SSE) can be used to represent the magnitude and direction of trends in data. Values of the SSE can be "relativised" by dividing through by the raw data median (RSSE), allowing for direct comparison between sites.
SEV	Stream Environmental Valuation (SEV).
SMU	Stormwater Management Units (stormwater catchments).
SoE	State of Environment.
SRP	Soluble reactive phosphorus (also called DRP - dissolved reactive phosphorus.
Stressors	Factors which stress ecosystems eg, contaminants.
Synergism	A phenomenon in which the effect or toxicity of a mixture of chemicals is greater than that to be expected from a simple summation of the effects or toxicities of the individual chemicals present in the mixture.
SZ	Settling Zones are areas were most (~75%) catchment-derived contaminants and sediments settle out of suspension and become incorporated into benthic sediments

	(see Outer Zones).
TEL	Threshold Effects Level (see PEL).
TEMP	Temperature.
Toxic pollutant	Toxicant. Pollutants or combinations of pollutants, including disease-causing agents, which after discharge and upon exposure, ingestion, inhalation or assimilation into any organism, either directly from the environment or indirectly by ingestion through food chains, will, cause death, disease, behavioral abnormalities, cancer, genetic mutations, physiological malfunctions, (including malfunctions in reproduction) or physical deformations, in such organisms or their offspring.
Toxicity Test	A procedure to determine the toxicity of a chemical or an effluent using living organisms. A toxicity test measures the degree of effect on exposed test organisms of a specific chemical or effluent. Toxicity tests can be carried out in the field (in situ) or in the laboratory. They can use uncontaminated media spiked with the chemicals(s) of concern or real contaminated sediment or water.
TP	Total phosphorus.
TPH	Total Petroleum Hydrocarbons.
TSS	Total Suspended Solids. Also often referred to as Suspended Solids (SS).
TURB	Turbidity – a measure of the clarity or opaqueness of water.
USC	Urban Stormwater Contaminant model. There are three models of increasing complexity USC1, USC2, USC3 (see Section 8).
V	Vanadium.
WOE	Weight of evidence.
Zn	Zinc.
XRF	XRay Fluorescence (an elemental analytical method).

13 References

The following convention has been used for citing references. Reports have been referred to where possible by the original authors, rather than the publishing organisation. This was done because most publications are ARC reports and using the convention "ARC year" produces similar-looking citations (eg, ARC 2008a, ARC 2008m etc.), which do not allow easy identification of individual studies while reading the text, and makes checking and ordering references difficult. One of the exceptions is the 2005 Review, because it is frequently cited in the text, and is cited as "ARC 2008" for simplicity and convenience.

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