Prediction of contaminant accumulation in Auckland estuaries

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Prediction of contaminant accumulation in Auckland estuaries

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Prepared for

Auckland Regional Council

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Executive Summary

Managers ask a range of questions concerning contaminant accumulation in estuaries. Some are general, for example:

- What parts of an estuary are most at risk from contaminants?
- Are all estuaries equally vulnerable?

and some are very specific, for example:

- How much contaminant is deposited in an estuary after a big storm?
- How many years will it take for contaminants to accumulate to a prescribed biologically-damaging level, given a certain pattern of landuse in the catchment?
- If the pattern of landuse is changed or managed in some particular way, how will this reduce/enhance patterns of contaminants in the estuary over the long term?

The aim of this project was to develop methodology that can be used to answer these kinds of questions.

Some questions are qualitative (the answer will be presented in words) but others are purely quantitative (a number is needed). Although the questions are often simply stated, answers are not always simply found. Some tools were available at the beginning of this project for providing answers in certain types of estuary (principally the USC model), but we found at the start that there did not exist a coherent conceptual understanding of how Auckland estuaries “physically process” sediments and contaminants, let alone quantitative methods for predicting contaminant accumulation in a wider range of estuary types.

We therefore began the project by constructing that coherent understanding, which we present in the first part of this report as a series of “rules”, which sum up our understanding of how Auckland estuaries work. We decided on a rules framework instead of, say, a more traditional narrative, because it is a succinct method of presentation that can easily be built upon in the future (by adding new rules or relationships between
rules). Expressing ideas as “rules” has also proven to be curiously confrontational: they invite rebuttal and testing, which adds a certain robustness to the whole affair.

We then developed a model for predicting contamination accumulation in Auckland estuaries. We describe it in the report as a “semi-quantitative process-based model”, meaning that it gives actual numbers for answers, and that it is based on our understanding of how Auckland estuaries work, which of course is described in the rules. After digesting the rules, the reader should have a pretty good feeling for why we constructed the model in the way we did. For instance, at the most basic level, the reader should understand the physical basis for our division of the estuary into a “PDA” and an “SRA” in the model.

The model provides a way of making “core” calculations (in the sense of “central”, not anything to do with a sediment core). These are things like contaminant deposition in a particular part of the estuary following a particular-sized flood event. The model is simply a set of mass-balance equations which distribute sediment and contaminants throughout the estuary according to the rules. We give worked examples of core calculations to help clarify and explain the methodology. The way these core predictions are extrapolated or integrated (in the mathematical sense) to answer management’s questions (which typically concern what happens over longer periods of time) depends very much on the specific question. What this means is that the core methodology might well be applied in the future in as many different ways as there are questions. This really is our central achievement: a way of finding answers, not the answers themselves.

Finally, we present a classification of Auckland’s estuaries. In keeping with our overall approach, the classification is based on the way we understand estuaries to work. By providing a process-based grouping of like estuaries, the classification will be useful in broadly transferring results of case studies to other estuaries.

We suggest that the best way to read this report is to recreate our experience with the work. Ask yourself a question concerning contaminant accumulation – take one of the general and one of the specific questions from the examples above. Think about how you would answer them – chances are, like us at the beginning of this project, your ideas will be pretty vague. Now read the report, then return to the same questions. We will have achieved our goal if you now, firstly, better understand the question and, secondly, can see how an answer might be obtained.
Executive Summary

(2) TECHNICAL

The ARC engaged NIWA to develop a way to predict contaminant accumulation over the long term (decades and longer) in sandy estuaries in the Auckland region. The intent was to build on and extend the Urban Stormwater Contaminant (USC) model, which has been used to date to predict long-term accumulation of contaminants in muddy estuaries.

Examples of contaminant accumulation in Auckland are given in Chapter 2. We briefly describe the USC model and the state of understanding shared by NIWA and the ARC prior to the commencement of this project concerning the fate of fine sediments and contaminants in estuaries.

The first step in our approach to developing a new model was to develop an understanding of estuarine processes that is more comprehensive than the one underpinning the USC model. The new understanding is captured in a series of “rules” (Chapters 3–6). Key rules describe the input of terrestrial fine sediments (“muds”) from the land and the accumulation of muds and sands in different parts of an estuary. A fundamental proposition that we develop is that contaminants are principally adsorbed to muds and therefore movement and accumulation of contaminants is governed by the same rules that govern mud behaviour.

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<td>Rule 4. As an estuary infills, a higher proportion of river-borne sediment gets exported from the estuary.</td>
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<td>Rule 8. Channels act as extensions of streams at low tide.</td>
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<td>Rule 9. Marine sands mostly deposit down around the mouth of the estuary.</td>
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<td>Rule 10. Mud does not necessarily remain where it settled during a flood event.</td>
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<td>Rule 11. Mud deposits expand gradually over the borders of sandy beds.</td>
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<td>Rule 12. Mud discharged out through the mouth of the estuary into the adjacent coastal ocean does not come back into the estuary in any large way.</td>
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<td>Rule 13. In a muddy embayment, most of the mud comes from the land and not from the sea.</td>
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Intertidal-flat sedimentation (rules governing “cross-estuary” distribution of sediments)

Rule 14. Muds tend to accumulate along the margins of channels.
Rule 15. Muds tend to accumulate on the upper margins of intertidal flats, unless these are exposed to waves.
Rule 16. Upper-intertidal mud deposits are usually colonised, and therefore stabilised and promoted, by vegetation (mangroves, marshes).
Rule 17. In exposed areas, beaches of sand or shell fragments accumulate on the upper margins of intertidal flats.

Chemical rules

Contaminant behaviour in estuaries

Rule 1. In the estuary, contaminants tend to behave as muds and physical movement is governed by mud rules.
Rule 2. Contaminants stick primarily to muds rather than to sands.
Rule 3. POP and heavy metal concentrations are generally well correlated.
Rule 4. POP concentrations within a particular estuary are correlated to organic matter (OM).

Dilution/dispersion

Rule 5. Contaminated muds are diluted by sands and by uncontaminated muds.

Recovery of estuarine sediments from contamination

Rule 6. Naturally-occurring recovery processes are generally very slow.
Rule 7. The most effective recovery mechanism is by burial by uncontaminated sediments.
Rule 8. Contaminants are not significantly moved to other areas by solubilisation-dispersal processes.
Rule 9. Decomposition of biodegradable POPs such as organophosphates is rapid, so they do not accumulate in estuary sediments.
Rule 10. Decomposition of PAH and organochlorines is insignificant.

Contaminant inputs

Rule 11. Contaminants that are known to accumulate in urban estuaries to levels of concern are Zn, Cu, and PAH.
Rule 12. The concentration of organochlorine pesticides and PCBs will not increase in urban estuaries in any large way.
Rule 13. Lead concentrations are expected to slowly decrease in urban estuaries.
Rule 14. Contaminants are delivered to estuaries during flood events.
Rule 15. Contaminants are delivered to the estuary attached to particulate matter or else they soon become attached to particulate matter.
Rule 16. Incoming contaminants tend to behave like terrestrial-sourced sediment.

Prediction of contaminant accumulation

Rule 17. Contaminant accumulation is an ongoing process.
Rule 18. The magnitude of contaminant inputs is directly proportional to the catchment area urbanised and type of landuse.
Rule 19. Contaminant accumulation in estuarine arms (Primary Deposition Areas) can be predicted from catchment history, catchment land use, tidal excursion and the 4% rule.

To establish a basis for prediction of contaminant accumulation, four key processes are recognised:

- contaminant and sediment input to the estuary (Chapter 7);
• dispersal and deposition following input, which applies to the “Primary Deposition Area” (PDA) of the estuary (Chapter 8);

• resuspension, redispersal and redeposition in the “Secondary Redistribuition Area” (SRA) of the estuary (Chapter 9);

• vertical mixing of the bed sediment (Chapter 10).

To predict contaminant accumulation, we need to know the magnitude and rate of contaminant and sediment input to the estuary. Methods to predict these have already been described in other ARC and NIWA publications, and these are reviewed, updated and re-presented here in Chapter 7. New estimates for PAH are provided.

The “Primary Deposition Area” (PDA) is the area in which muds deposited by primary processes (i.e., flocculation and sudden loss of transporting capacity of freshwater inflow) are not readily redistributed by secondary processes between floods. Hence, muds steadily accumulate in each PDA (there is a PDA associated with each tributary to the estuary). Typically, PDAs cover the upper reaches of the estuary, but can extend into the middle and lower reaches of an estuary along channel margins. Sedimentation in a PDA primarily occurs during storm events, and we are able to construct a prediction scheme based on summing the effects of many such events (Chapter 8). A key parameter in the scheme is the term $R$, where $1/R$ is the fraction of sediment discharged from the source that is deposited at the site of interest. We show how $R$ can be estimated from a sediment-transport model using Okura estuary as the example, and extract some general principles from that analysis that can be applied to other estuaries in the region.

Beyond PDAs, in the middle and lower reaches of the estuary, mud is still deposited during floods by primary processes. However, in these areas – “Secondary Redistribuition Areas” (SRAs) – secondary processes significantly redistribute deposited muds to preferred locations between floods. Waves are primarily responsible for entraining settled, alien muds and thus initiating redistribution of muds in the SRA. While there is also event deposition in the SRA, the long-term picture depends on what happens between events (when secondary processes operate). The problem is that there is no simple way to predict the sequence of secondary processes that might occur between events, which will have a strong bearing on sediment accumulation at any particular site. Sediment accumulation in the SRA is not essentially event-based and so an event-based prediction scheme is not appropriate. Furthermore, we have no easy way of estimating the input of marine sand, which might be a significant portion of the local sediment budget at any particular site in the SRA. To solve this problem and thereby produce a scheme for predicting contaminant accumulation in the SRA, we “deconstruct” the annual average sedimentation rate (rather than “construct” it, which we did for PDAs), using the existing bed-sediment composition as a guideline (Chapter 9).
Contaminated sediments deposited during floods or by secondary processes may be incorporated (“mixed”) into pre-existing bed sediments, which is primarily achieved by bioturbation. We reviewed and summarised the international literature in the search for a realistic model(s) to describe sediment mixing. Critical evaluation of local data showed that the homogeneous mixing sub-model used in the USC model gave a good approximation of contaminant profiles, at least to order of magnitude (Chapter 10). We show how mixing can be incorporated into the PDA and SRA prediction schemes, but improvements in the actual mixing model will have to await better local data.

Finally, an estuary classification scheme that groups like estuaries was devised in order to provide a context for applying the prediction schemes and for extending case-study results to other estuaries in the region (Chapter 11). The scheme combines our understanding of estuarine geomorphology and of the processes that disperse sediments/contaminants. An Auckland estuary archetype is recognised that displays several elements, which are defined in terms of “connections”. The arm connects directly to the land, and the throat connects directly to the coastal ocean. Between the two, and therefore with indirect connections to both the land and to the ocean, is the main body of the estuary. Arms correspond to PDAs and main bodies and throats are generally in the SRA. The classification scheme is thus based on how sediments flow between the archetypal elements. The classification scheme is explained by reference to example estuaries.

To conclude, the prediction methodology developed herein is based on our understanding of sediment processes in estuaries and the way contaminants interact with sediments. The methodology is a foundation, upon which a range of specific approaches will be built over time as regional and case studies are conducted.
1. **THE PROBLEM**

1.1 **The study brief**

The aim of this project is to build a model that will predict long-term contaminant accumulation in a range of estuaries and that will provide a tool for answering such questions as:

- Where in estuaries do contaminants accumulate?
- How rapidly do they accumulate?
- Is the rate different for different estuaries?

This study extends an existing tool – the Urban Stormwater Contaminant (USC) model (Williamson et al., 1998a and 1999; Williamson and Morrisey, 2000; Morrisey et al., 2000) – to estuaries with greater hydrodynamic energy, i.e., those estuaries that are exposed to greater wave and current action.

The timescale of interest is years, decades and centuries.

1.2 **Possible approaches to the problem**

There are several possible approaches:

(1) Numerical. Simulations can be run using a numerical model to develop an understanding of processes. Properly implemented, numerical models may provide the most accurate predictions, but there are two significant problems. Firstly, over the timescales that are of interest here, the computational resources required make numerical models prohibitively expensive. Secondly, the use of numerical models for long-term prediction presents formidable technical problems that are not always solvable.

(2) Observational/experimental. Observations are the foundation of any understanding. However, a purely observational/experimental approach will not provide a good platform for extrapolation (prediction) unless there is a corresponding understanding of underlying processes.

(3) Conceptual. Good conceptual models incorporate an understanding of processes and are consistent with observational/experimental knowledge. They are generally based on broad statements and ideas, and are semi-quantitative at the
most. They are not as expensive to develop and run as numerical models, but they probably will not be as accurate, either. On the other hand, because conceptual models are often holistic and expressed in non-mathematical language, they may be readily understood and applied. Therefore, they are good for comparative studies and management applications.

A semi-quantitative, process-based conceptual model is the best choice here.

- Being process-based, extrapolation will be possible.

- The model needs to be at least semi-quantitative to predict rates/amounts of contaminant accumulation (but that is not necessarily a requirement for predicting patterns of contaminant accumulation). This will be a major challenge.

- Comparisons between estuaries and between sub-environments within estuaries can readily be developed and applied regionally, which will aid the environmental manager.

- This approach is consistent with that adopted for the USC model.

1.3 Nature of the problem

There are four processes in our conceptual model that need to be described and quantified:

1. contaminant and sediment input to the estuary;

2. dispersal and deposition following input;

3. resuspension, redispersal and redeposition;

4. sediment mixing.

Underlying all these are the processes controlling the way contaminants are bound to sediments (contaminant–sediment interactions).
1.4 **Components of our approach**

*First step*

Synthesize/assemble our existing knowledge into sets of “rules” that describe how Auckland estuaries process sediments and contaminants (Chapters 3–6).

*Second step*

From these rules, develop simple quantitative models that can be used to predict contaminant accumulation (Chapters 7–10).

*Third step*

Provide an estuarine classification scheme for Auckland within which models can be applied and methods extrapolated (Chapter 11).

1.5 **Methodology**

The Okura hydrodynamic and sediment model (Green and Oldman, 1999) was used to test ideas and generate values for coefficients in the contaminant-accumulation model. In addition to relying on our experience from other studies (e.g., Mahurangi, Okura, Pakuranga, Helyers, Orewa, Paremoremo), extensive visual surveys were undertaken in Puhoi, Waitera, Shoal, Ngataringa, Hobson, Coxes, and Motion/Meola, where ideas were appraised against the physical reality encountered.
2. BACKGROUND TO THE STUDY

In this chapter we describe briefly the reasons for concern for contaminant accumulation in estuaries. In addition, we describe our pre-existing (i.e., prior to this project) understanding and abilities to predict contaminant accumulation in Auckland estuaries.

2.1 The reasons for managing contaminant accumulation in estuaries

Urban stormwater is New Zealand’s largest and least easily managed source of toxic contaminants to the coastal environment. Typically, urban stormwater carries heavy metals (zinc [Zn], lead [Pb], copper [Cu]), polyaromatic hydrocarbons (PAH) and organochlorines (lindane, PCB, DDT, DDE, dieldrin) (Williamson, 1993; Mills, 1997), which is also the case overseas (Makepeace et al., 1995).

There is ample evidence that stormwater-derived contaminants are accumulating in Auckland estuaries. Fig. 2.1 summarises the spatial distribution of contaminants in Pakuranga Estuary, with higher concentrations found near the major inputs at the head of the estuary, and decreasing concentrations towards the mouth. Fig. 2.2 shows the contaminant profiles in a core taken from the head of the estuary close to the inputs. Pakuranga estuary received no industrial pollution and its catchment was predominantly improved pasture until urbanisation commenced in 1960, so the increase in Cu, Pb and Zn concentrations is solely due to urban-stormwater inputs.

![Fig. 2.1 Distribution of Zn in Pakuranga Estuary.](image-url)
2.2 Urban Stormwater Contaminant (USC) model

The Urban Stormwater Contaminant model (a variation of the Auckland Strategic Plan [ASP] model) (Williamson et al., 1998a and 1999; Williamson and Morrisey, 2000; Morrisey et al., 2000) is presently being used in Auckland to predict average concentrations of Cu, Pb and Zn in the “settling zone” of estuaries. The model was developed to address contaminant accumulation in upper reaches of estuaries where most stormwater is discharged (see Section 2.3). Figs. 2.3 and 2.4 show schematics of the model. The model has been used to predict accumulation of contaminants in estuarine sediments as the result of different urban development scenarios. It calculates that, at the present time, concentrations of Zn, Pb and Cu in many Auckland estuaries exceed the sediment quality guideline that identifies the level at which biological effects are expected to begin to occur (i.e., the ER-L; Long et al., 1995), and this is indeed what is observed. Of more importance is the prediction that concentrations will continue to increase, and so increase the risk of adverse impact on the animals and plants that live in or on the sediments.

Fig. 2.5 shows predicted concentrations for the muddy Pakuranga sub-estuary, using the USC model. Zn concentrations are predicted to increase and exceed sediment quality guidelines (SQG) that indicate that biological effects are expected to occur frequently within 20–50 years (i.e., the ER-M; Long et al., 1995). Cu concentrations
will increase to levels that are 2–3 times the ER-L. Pb concentrations are predicted to gradually decline now that Pb has been removed from petrol.

Fig. 2.3 Flow chart of the computational steps within the USC model.
The USC model has been successfully tested in the settling zones of estuaries, which is where fine sediments accumulate because of low tidal currents and small waves (Williamson et al., 1998a; Williamson and Morrisey, 2000; Morrisey et al., 2000). The model worked reasonably well in predicting spatial-average concentrations, but does not predict vertical profiles well, nor does it predict the overall distribution of contaminants down the estuary.
2.3 The conceptual underpinning of the USC model

This section describes the state of understanding of estuarine processes at the commencement of this study (summarised from Williamson et al., 1999) and the conceptual understanding that underpins the USC model. We revisit this understanding in Chapter 3 of this report.

Dissolved contaminants are substantially diluted when discharged into estuaries, and thus are not expected to exert toxic effects in the water column. However, contaminants primarily adsorb to fine particulate matter in stormwater discharges (Williamson, 1993; Leersnyder, 1993). Upon arrival at the estuary, the fine particles are flocculated and the resulting larger particles settle to the bed. Therefore, the immediate fate of a large proportion of the contaminants after entering the estuary is deposition by settling in the upper reaches of the estuary.

Once settled, particulate-associated contaminants can be resuspended from the seabed into the water column by waves and/or currents. Resuspended sediment may then be dispersed by tidal currents to more quiescent areas, where it settles and is ultimately mixed into the underlying sediments by bioturbation. The ultimate fate of sediments and adsorbed contaminants depends on the amount and spatial distribution of hydrodynamic energy in the estuary.

In estuaries with low hydrodynamic energy, contaminants tend to be trapped. In areas of high hydrodynamic energy, contaminants are “moved on”. Therefore, it is useful to distinguish within an estuary three zones based on three types of processes (Hakanson, 1982):

1. Areas of accumulation, where fine materials are being continuously deposited. Wave and current energies are very low here.

2. Areas of transportation, where periods of accumulation are interrupted by periods of remobilisation (generally of short duration and associated with storms).

3. Areas of erosion, where there is little deposition of fine materials. Wave and/or current energies are high here.

Fine materials tend to be shunted from erosional and transportation zones to sheltered arms and embayments. Thus, according to Hakanson’s view, the ultimate repositories of contaminants associated with fine particles are depositional zones.
Many Auckland catchments discharge into small estuaries that are infilled stream valleys. A good example of this is the Pakuranga Estuary (Fig. 2.1), which is an arm of the larger Tamaki Estuary. The USC model was developed to address accumulation in these small estuaries (later on in this project we classify these as “arms” to the main estuary body). Key considerations in understanding the probable fate of sediment and contaminants discharged by stormwater runoff into sheltered estuaries are as follows:

- Auckland watersheds are relatively steep, and flow velocities are decreased substantially after discharge into estuaries.

- A large proportion of stormwater particulates are silt-sized or greater (Leersnyder, 1993; ARC, 1992), which will settle quickly.

- Stormwater discharges occur at many points around estuaries.

- Estuaries are shallow, with extensive intertidal areas.

Small catchments in Auckland generally discharge small stormwater flows into relatively large estuarine areas. At high tide, the relatively low-volume, low-energy freshwater discharge spreads out over the submerged intertidal area. Within a short distance of the outfall, currents will be low enough to provide ideal settling environments (Hume and McGlone, 1986). As the tide retreats, stormwater will flow in channels incised within intertidal flats. We expect scouring of existing estuarine sediments and deposition of coarse, urban-derived sediments in these channels. On reaching the tide water line, stormwater will be mixed and spread out over lower intertidal and subtidal areas, with the mixing/settling field therefore tending to spread down-estuary. A similar picture holds for the rising tide, except the mixing/settling field moves up-estuary.

In larger catchments, greater discharges will push stormwater further down main channels. At high tide, the discharge field will tend to spill out over the top of the adjacent intertidal areas, which provide ideal settling areas. As the tide ebbs, this intertidal area will become smaller, until most of the flow is concentrated in the channel. Settling will occur only where channels widen significantly.

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

Some of the sediment resuspended by waves will escape from the estuary on the ebb tide. However, since many estuarine arms discharge into effectively enclosed basins (e.g., Pakuranga Estuary, which discharges into Tamaki Estuary), much of the “escaped” material will return on the next flood tide. Because the flood tide is more energetic than the ebb tide, fine sediments tend to march up-estuary, which lead us to conclude that the ultimate fate of the bulk of contaminants discharged into small estuarine arms will be accumulation in the upper reaches.

The USC model assumes that 75% of contaminants discharged to an estuarine arm are retained in that arm and settle in an area that is nominally 4% of the catchment area. The value of 4% was adopted on the basis of settling-pond theory (Vant et al., 1993) and because 4% approximately equates to the intertidal area in estuarine arms. In practice, the 4% value is adjusted to site-specific circumstances.

Other types of deposition environment are found in Auckland. Using Hakanson’s (1982) reasoning, contaminant accumulation should not be a concern in erosional environments. Contaminant accumulation might be a concern in transportational environments, where periods of resuspension and redispersal are more intermittent. Contaminants are still retained, but the accumulation is slowed by intermittent dispersal and by dilution by uncontaminated sandy sediments. Erosional and transportational environments are found downstream from estuarine arms. These are quite extensive in Auckland and are characterised by muddy sands or sandy muds. While Hakanson’s (1982) classification is a useful, broad generalisation, it is not useful in describing contaminant and fine-sediment fate in more complex, energetic estuaries found in Auckland. In the next chapter we examine the processes and likely fate of contaminants occurring in such estuarine areas. We do this within the framework of the whole estuary, from the estuarine arms and out to the open coast.
3. PATTERNS OF SEDIMENT ACCUMULATION IN ESTUARIES

In this Chapter we revisit, update and present our conceptual understanding of how Auckland estuaries function. This information underpins the rules presented in Chapter 4.

Because contaminants primarily adsorb to fine sediments (“muds”, in the colloquial, as opposed to “ sands”), we will focus on accumulation of muds, which is the contaminant “carrier”. This is a useful device, because we can then draw directly on our knowledge of mud sedimentation in estuaries, which is substantial.

(In this chapter, italicised text presented in an indented paragraph synthesises the preceding text and shows application to understanding contaminant behaviour.)

To understand how mud accumulates in estuaries is to understand patterns of contaminant accumulation. This is the aim of this chapter.

Auckland’s estuaries were formed by sea level rising and drowning ancestral river valleys, which began about 15,000 years ago and ceased about 6,000 years ago. Since then, with sea level more or less stable, Auckland’s estuaries have been filling with sediment. There are two principal sources of sediment: (1) terrigenous sediments – usually muds – are delivered to the estuary headwaters by streams and rivers, mainly during floods; and (2) marine sediments – usually sands – are pushed in through the estuary mouths by tides, waves and coastal littoral drift. In addition, and secondarily, sediment may derive from erosion of the margins of the estuary and from in-situ biological production.

The balance between deposition of terrigenous muds and deposition of marine sands has changed over time. Six thousand years ago – early in the life of the estuary – the typical estuary was a deep basin filling primarily with terrestrial mud. At that time, marine sands were only very slowly being combed shorewards from the adjacent continental shelf by waves and tides. Gradually, as sands moved onshore, water depths were reduced along the coastline, which accelerated the processes combing sands shorewards. Eventually, sands started to build beaches and to penetrate estuary mouths and deposit on top of the basal terrigenous muds therein. Today, with Auckland’s estuaries nearly filled with sediments, we see a characteristic axial sequence of sediments that reflects this basic infilling process: terrestrial muds at the head of the estuary, grading to marine sands at the mouth.

Because contaminants attach to terrestrial muds, we expect there to be a corresponding general gradient in contaminant accumulation from
We need to look a little more closely at estuarine sediment transport to understand further implications for mud (and contaminant) accumulation. Waves and associated coastal littoral drift that comb marine sands shoreward are quickly dissipated once they penetrate an estuary mouth. Hence, marine sands carried by these agents do not extend very far inward from the mouth. Tidal currents conveyed in relatively deep channels may carry sands farther up an estuary, building intertidal sandbanks along the channel flanks. Under conditions of low freshwater flow, suspended terrestrial muds are deposited rapidly at the head of the estuary. There are two main reasons. Firstly, suspended terrestrial muds are flocculated by mixing with brackish (estuarine) water. Secondly, there is a sudden and large drop in the transporting capacity of the freshwater as it expands in the estuary basin. Looking ahead, we call these “primary deposition processes”.

Contaminants delivered to an estuary by streams during times of low flow steadily accumulate in the upper reaches of the estuary.

Estuarine channels act as extensions of streams at low tide, thus allowing freshwater and accompanying suspended load to penetrate farther into the middle reaches of the estuary at those times. When the tide turns, the muddy freshwater is lifted out of the channels and onto the adjacent intertidal flats where it experiences a sudden drop in transporting capacity. This results in deposition of suspended mud along the margins of the channel.

Contaminants delivered to an estuary by streams during times of low flow steadily accumulate along channel margins in the middle reaches of the estuary.

During floods, the freshwater/brackish interface is pushed out towards the mouth of the estuary under the higher volume of freshwater runoff, which allows more suspended terrestrial fines to penetrate the middle and lower reaches of the estuary (and, indeed, escape to the coastal ocean). Terrestrial muds may deposit, at least temporarily, in the middle and lower reaches of the estuary, including over the top of sands.

During floods, contaminants may be discharged into the adjacent coastal ocean, and may be deposited widely in the middle and lower reaches of the estuary, including over the top of sands.
Mud does not necessarily remain where it settled during a flood. Between floods, a number of (sometimes very subtle) “secondary processes” can act to redistribute “alien” muds in preferred directions. In general, biological processes (bioturbation) mix alien muds downwards into pre-existing sediments, and physical processes (waves, currents) resuspend and transport alien muds back up to the upper reaches of the estuary. As far as physical processes go, waves typically resuspend deposited alien muds, and currents then transport the resuspended muds in the preferred directions.

The long-term fate of contaminants deposited in the middle and lower reaches of an estuary during floods is complicated. A fraction may be incorporated into pre-existing sediments by bioturbation, another fraction may be transported to and permanently re-deposited back in the upper reaches of the estuary, and another fraction may be buried in situ by continuing deposition of sand or by mud deposited during the next flood.

In addition, there are certain areas in the middle and lower reaches of an estuary that “attract” (promote the deposition of) mud. These include: channel margins (for reasons previously described); vegetated settling zones (e.g., mangroves and marshes, which reduce currents thereby promoting mud deposition); areas sheltered from waves (since it is waves that principally entrain alien muds); and along outer fringes of the estuary and shorelines (except shorelines that are exposed to waves). The latter areas (i.e., along outer fringes and shorelines) attract mud for several reasons: the same subtle processes that push alien muds back to the upper reaches of the estuary also act to push muds outwards in middle and lower reaches; muds that become stranded on outer fringes and shorelines on ebbing tides have time to compact and increase their resistance to erosion while the tide is out; stranded muds also compact during neap tides.

Contaminants may accumulate in the middle and lower reaches of an estuary in preferred locations.

If soil erosion in the catchment is increased (by, for instance, a change in landuse) then both low-flow and flood sediment yields will increase. The former will translate into increased mud deposition in the upper reaches of the estuary. The latter may result in more mud being deposited in middle and lower reaches after floods than can be removed between floods, which will result in increased mud accumulation throughout middle and lower reaches. Bioturbation will mix increasing amounts of mud throughout the pre-existing sediment column. At least on the east coast of the Auckland region, there appears to be little more sand stored on the continental shelf that can be combed shorewards onto beaches and into estuaries. Hence there may be
decreasing amounts of sand moving into the lower reaches of estuaries that can dilute mud. In short, as the estuary “ages”, the middle and lower reaches will get muddier.

*We can expect contaminant accumulation to accelerate in middle and lower reaches of estuaries, particularly in the face of landuse change that increases soil erosion.*

As estuary infilling advances, a higher proportion of river-borne sediment will get exported directly to the coastal ocean. Ultimately, should the estuary completely fill with sediment (which might happen if sea level remains stable for long enough and sediment supply is maintained), streams will discharge their suspended-mud loads directly into the coastal ocean over a growing submarine delta.

*Contaminants in the coastal ocean is a growing concern for the future.*
4. PHYSICAL RULES

In this chapter we present the rules describing sediment behaviour in Auckland estuaries.

4.1 General characteristics of Auckland estuaries

Auckland estuaries have the following general characteristics:

- They range from partially filled to nearly completely filled with sediment.

- Those estuaries that open to the coast have had a marine sand source, but those that open to embayments have not had a marine sand source.

- They are usually sheltered from ocean waves.

- They have low volume and episodic freshwater input.

- They are well mixed, with insignificant thermohaline circulation, although they become stratified during periods of high freshwater runoff.

- Auckland’s east-coast estuaries open onto a sediment-starved, low-littoral-drift coast (Hauraki Gulf)

- Auckland’s west-coast estuaries open onto a sediment-rich, high-littoral-drift coast (Tasman Sea).

- Sea level right now is about as high as it has ever been and has been that way for the last 6,000 years.

4.2 Physical rules

Nature of estuaries

Rule 1. Streams and rivers dump terrestrial sediments into estuaries.

Terrestrial sediments delivered to Auckland estuaries tend to be very fine-grained because catchments are small and there is little or no erosion of rock (except for the Waitakere Ranges).
Rule 2. Waves and tides push marine sediments into estuaries through the estuary mouth.

When estuaries were first formed, it was under a rapidly rising sea level which left behind marine sand out on the continental shelf. Estuaries initially began filling with mud only, but as sands slowly moved onshore, water depths over the inner continental shelf decreased, which accelerated the onshore sand movement.

Rule 3. Estuaries are sediment sinks.

Sediment discharged into the estuary by streams or pushed in through the estuary mouth by waves and tides tends to stay there. In the grand scheme of things, estuaries are ephemeral features. Their ultimate fate, under stable sea level and steady sediment supply, may be to become land.

Rule 4. As an estuary infills, a higher proportion of river-borne sediment gets exported from the estuary.

When the estuary is 100% infilled (the estuary no longer exists as such; there is land where the estuary used to be), all of the river-borne sediment gets exported (discharged into the coastal ocean directly over a submarine delta).

Rules of accumulation

Rule 5. Mud is most likely to accumulate where the bed is already muddy.

The corollary is that mud is least likely to accumulate where the bed is sandy.

Rule 6. Waves and mud do not co-exist.

Muds are hindered from accumulating in exposed areas, but they accumulate in sheltered areas.

Rules governing longitudinal distribution of sediments

Rule 7. Terrestrial fine sediment mostly deposits at the head of the estuary where streams discharge.

Two processes are happening here. Firstly, there is a large current-speed drop where freshwater meets the estuary basin, which rapidly reduces transport capacity of the stream. Secondly, fines suspended in the freshwater flocculate rapidly in the presence of very low salinities. This occurs at the head of the estuary where freshwater sources
discharge and first contact seawater and results in the formation of larger particles with increased settling speeds. As a consequence of these two processes, terrestrial sediment is deposited mainly at the head of the estuary.

**Rule 8. Channels act as extensions of streams at low tide.**

The fluvial/estuary transition with accompanying high-deposition zone moves down-estuary at low tide and up-estuary at high tide.

**Rule 9. Marine sands mostly deposit down around the mouth of the estuary.**

There is a transition zone between marine sands and terrestrial fines where the two mix and “interleave”.

Estuaries that discharge into sheltered embayments may have no marine sand source and/or no way of moving marine sands into the estuary. The sand wedge will be absent in that case, and the estuary will be filled with mud.

**Rule 10. Mud does not necessarily remain where it settled during a flood event.**

“Secondary processes” redistribute muds in all three dimensions in preferred directions over time following a deposition event:

- muds are pushed up-estuary, to join other muds;
- muds are pushed outwards (from the centre of the estuary) and upwards (in the vertical sense), to join other muds on the estuary fringes;
- muds are mixed and bioturbated downwards into the pre-existing bed sediments.

**Rule 11: Mud deposits expand gradually over the borders of sandy beds.**

New mud deposits usually do not suddenly appear in the middle of previously sandy areas (but it can and does happen in sediment-impacted estuaries). An increase in terrestrial mud load will accelerate the movement of the mud/sand transition zone down-estuary, resulting in sandy sediments getting progressively buried by mud. It follows that if you are looking for ecological impact associated with extra mud, then the place to look is at the mud/sand transition zone because that is where fundamental changes in sediment composition generally take place.

**Rule 12. Mud discharged out through the mouth of the estuary into the adjacent coastal ocean does not come back into the estuary in any large way.**
Coastal currents and waves rapidly disperse muds in the coastal ocean. Mud that does escape out through the mouth of the estuary settles in deep water seaward of the surfzone and beyond the zone of shoaling waves (the “shoreface”).

Rule 13. In a muddy embayment, most of the mud comes from the land and not from the sea.

*Intertidal-flat sedimentation* (rules governing “cross-estuary” distribution of sediments)

Rule 14. Muds tend to accumulate along the margins of channels.

The reasons for this are uncertain but some possibilities are:

- Channel margins could be a zone of low wave energy. When the water is shallow enough for wave-orbital motions to penetrate to the bed, the fetch is too small for there to be any significant wave activity.

- Overtopping. Muddy water spilling out of a confined channel on a rising tide across the banks may promote sedimentation in the same way that levee banks are built along rivers.

- Internal drainage. We have noted that the channel margins drain at low tide (into the channel). Perhaps this helps to consolidate and therefore stabilise the mud bank in some way.

Rule 15. Muds tend to accumulate on the upper margins of intertidal flats, unless these are exposed to waves.

A corollary is that in the mud/sand transition zone of the estuary, central areas of intertidal flats tend to be sandy. Also, an increase in terrestrial mud load will push (a) the upper-intertidal mud fringe downwards and (b) the channel mud margin upwards, thus tending to obliterate the central sandy area.

Rule 16. Upper-intertidal mud deposits are usually colonised, and therefore stabilised and promoted, by vegetation (mangroves, marshes).

Rule 17. In exposed areas, beaches of sand or shell fragments accumulate on the upper margins of intertidal flats.

Intertidal flats adjacent to natural rock outcrops, walls or revetments tend to be devoid of mud because reflection of even small waves from those hard structures scours the
bed and prevents mud from accumulating. If a sand or shell source is available, a
beach may develop. Sandy and shelly beach deposits may “ride on top of” or impound
upper-intertidal mud deposits. These deposits are beaches, ridges, spits.
5. CHEMISTRY PROCESSES IN ESTUARIES

In this chapter, we present an overview of the chemical processes that determine the behaviour of contaminants in estuaries. A lot of this information has been given in other ARC publications (ARC, 1992; ARC, 1994; ARC, 1995; ARC, 1998a), so we only briefly describe these processes here. This information underpins the chemical rules in Chapter 6.

In the USC model we assumed that most of the contaminants arrived at the estuary in particulate form. There is increasing evidence that some contaminants at source may be predominantly dissolved (e.g., Zn$^{2+}$) or occur as oily films (e.g., PAH). However, we still assume that by the time these are delivered to the estuary they have become attached to particulate matter through a variety of adsorption processes. Consequently, we expect contaminants in estuaries to behave like terrestrial-sourced sediment, and thus undergo flocculation, settling and deposition.

Contaminant accumulation is an ongoing process and continues in the same place, unless there is a major change in estuary hydraulics (e.g., due to infilling of the estuary). To reiterate our understanding from Chapters 2 and 3, flood-delivered deposition of contaminants may be widespread and it mimics the dispersal and settling of flood-borne muds. In sheltered muddy arms, contaminants settle and accumulate in the primary deposition zone. If a flood pushes contaminants into higher energy zones, then the accumulation rate, location and extent depend on redistribution processes, again mimicking mud behaviour. Contaminant accumulation does not occur in open coastal situations with significant hydraulic energy.

As pointed out in Chapter 3, contaminants stick predominantly to muds rather than to sands. This is generally true because contaminant binding phases, organic matter (OM), hydrous iron oxides (FeOOH), acid volatile sulphides (AVS), iron pyrites (FeS$_2$) and general surface area are higher in muds than in sands. This generalisation is not always true for Persistent Organic Pollutants (POP) such as PAH, because these stick to OM, irrespective of size, and the sand-sized fraction can occasionally form a significant fraction of the OM pool.

Contaminants stick directly to sands as well to a limited extent, so sands are not completely contaminant free. In this case, adsorption may be occurring through contaminant binding phases coating the sand particles (e.g., amorphous AVS can be observed to coat sand particles in anoxic sandy sediment). Sometimes sand-sized particles are observed to contain equally high concentrations as muds, e.g., heavy metals in Meola Creek, PAH in Motions Creek, part of the Middle Waitemata Harbour (ARC, unpublished data). However, it is not certain that these are truly sand-sized particles and are not just coagulated mud.
POP and heavy metal concentrations are generally well correlated in estuarine sediments. This is partly due to a common source and partly due to the estuarine processes. Correlation is often good over wide concentration ranges, but tends to be weak or moderate over small concentration ranges (i.e., the confidence bands are fairly wide).

POP concentrations within a particular estuary are correlated to organic matter. This is because POP equilibrates with organic matter. Equilibrium partitioning theory describes this process. POPs may achieve equilibrium with organic matter after deposition through the dynamic processes of solubilisation, absorption, diffusion, irrigation, biological uptake, depuration, excretion and death.

As has been frequently pointed out in this report, estuaries are usually regarded as excellent traps and accumulators of contaminants. However, there are some competing processes that lead to reduction in contaminant concentrations. These are:

- dilution or burial by uncontaminated sediments;
- solubilisation and dispersal;
- sediment resuspension and redispersal;
- decomposition of POPs.

Burial by large inputs of uncontaminated muds can be a major recovery process. This may be occurring naturally in Shoal Bay, where shores are eroding, and has appeared to occur in many estuaries (e.g., Pakuranga, Hellyers) from subsoils eroded during urban development. In Mangere Inlet, high mud sedimentation rates, perhaps sourced from elsewhere in the harbour, have resulted in burial of highly contaminated sediments.

Contaminants are not significantly mobilised to other areas by solubilisation-dispersal processes. Processes such as surface desorption, oxidation–mobilisation (e.g., of organic matter or metal sulphides), diffusion and irrigation can release contaminants to the overlying water column, where they can be dispersed by currents. While concentrations are low, significant amounts can be mobilised because the tidal prism (the volume of water moving over the contaminant repository) is large. However, readsorption onto particulate matter and mud redistribution mechanisms tend to return the contaminants to the original repository, so there is little net loss. Similarly, sediment resuspension and dispersal is a major process, but as described in Chapter 4, mud redistribution mechanisms tend to return the contaminants to the repository areas.
POPs and other undesirable organic material undergo a variety of transformations such as oxidation, UV photolysis, reduction, hydrolysis and microbial breakdown. Individually, these reactions lead to the formation of less or more toxic forms, but collectively and ultimately, they lead to recovery. This will be an important recovery route for the unstable toxic organics such as organophosphates, which is one of the reasons why we have ignored their accumulation in estuaries. These reactions are very slow for the recalcitrant POPs such as PAH and organochlorines (DDT, DDE, PCB, Lindane).

Naturally-occurring recovery processes are generally very slow. This can be seen from the above mechanisms. Furthermore, the rates of the dominant recovery process – burial and resuspension/dispersal – do not depend on the concentration of the contaminant, so they do not speed up as the estuary becomes more contaminated. The slowness of recovery processes can also be seen from the fact that estuaries with small amounts of urbanisation have accumulated contaminants, i.e., the natural recovery processes have not kept pace with inputs.

The dominant recovery mechanism in Auckland is by burial by less contaminated sediments, and self-cleansing processes are dominated by mud behaviour.
6. CHEMISTRY RULES

In this chapter we present the rules on chemical processes that govern contaminant behaviour in estuaries.

Contaminant behaviour in estuaries

Rule 1. In the estuary, contaminants tend to behave as muds and physical movement is governed by mud rules.

Rule 2. Contaminants stick primarily to muds rather than to sands.

Contaminant binding is higher in muds, but that is not always true for POPs, since these stick to organic matter, irrespective of size, and the sand-sized fraction can occasionally form a significant fraction of organic matter.

Corollary a: Contaminant concentrations are highest where muds are dominant.

Corollary b: Contaminant concentrations are lowest where sands are dominant.

Corollary c: Contaminant concentrations correlate with sediment-particle surface area.

This is a global rule that breaks down regionally (inter-estuarine) because of source factors or near-field effects of discharges. It may also not hold in some other situations near terrestrial inputs where significant amounts of POM may be in the form of discrete organic material rather than adsorbed surface films.

Rule 3. POP and heavy metal concentrations are generally well correlated.

Rule 4. POP concentrations within a particular estuary are correlated to organic matter (OM).

Corollary a: Concentrations of POP in OM tend to be constant (but see Rule 2 Corollary c).

Corollary b: The bioavailable concentration of POP is relatively constant throughout an estuary.

Corollary c: POP concentrations can be predicted on the basis of a mean POP concentration and organic matter distribution throughout an estuary, and the equilibrium partitioning coefficient.
**Dilution/dispersion**

Rule 5. Contaminated muds are diluted by sands and by uncontaminated muds.

**Recovery of estuarine surface sediments from contamination**

Rule 6. Naturally-occurring recovery processes are generally very slow.

Rule 7. The most effective recovery mechanism is by burial by uncontaminated sediments.

Rule 8. Contaminants are not significantly moved to other areas by solubilisation-dispersal processes.

Rule 9. Decomposition of biodegradable POPs such as organophosphates is rapid, so they do not accumulate in estuary sediments.

Rule 10. Decomposition of PAH and organochlorines is insignificant.

**Contaminant inputs**

Rule 11. Contaminants that are known to accumulate in urban estuaries to levels of concern are Zn, Pb, Cu, and PAH.

Urban impervious areas are continually supplied with fresh Zn, Cu and PAH (and in the past, Pb). There are other contaminants that require further investigation, such as dioxin.

Rule 12. The concentration of organochlorine pesticides and PCBs will not increase in urban estuaries in any large way.

While DDT, PCB and dieldrin are found in urbanised estuaries at levels of concern, their use and input into the urban environment has declined.

Rule 13. Lead concentrations are expected to slowly decrease in urban estuaries.

While Pb has built-up in the past it will now decrease with the removal of Pb from petrol.

Rule 14. Contaminants are delivered to estuaries during flood events.
Most contamination comes from small urban catchments and low flow inputs are very small.

**Rule 15.** Contaminants are delivered to the estuary attached to particulate matter or else they soon become attached to particulate matter.

Natural processes tend to adsorb contaminants to particles.

**Rule 16.** Incoming contaminants tend to behave like terrestrial-sourced sediment.

Corollary: They undergo flocculation, settling and deposition.

**Prediction of contaminant accumulation**

**Rule 17.** Contaminant accumulation is an ongoing process.

Contaminant accumulation continues in the same place, and shifts from this place require major changes in estuary hydrodynamics (e.g., such as associated with infilling of the estuary).

**Rule 18.** The magnitude of contaminant inputs is directly proportional to the catchment area urbanised and the type of landuse.

**Rule 19.** Contaminant accumulation in estuarine arms (Primary Deposition Areas) can be predicted from catchment history, catchment land use, tidal excursion and the 4% rule.
7 SEDIMENT AND CONTAMINANT INPUTS FROM ESTUARY WATERSHEDS

This chapter describes the resources available for estimating contaminant and sediment inputs. It builds on earlier descriptions (Vant et al., 1993; Williamson et al., 1998a).

7.1 Calculating sediment inputs

Sediment inputs to estuaries can be calculated from observational data and/or from models. We present some of the more accessible approaches here, although the list is not exhaustive. The following are ways in which we have estimated suspended-sediment inputs from rural land, urban construction, streambank widening, and mature urban areas in a number of projects around the Auckland area.

7.1.1 Rural sediment loads

A value of 0.5 t ha$^{-1}$ yr$^{-1}$ has been recommended based upon a study of storm sediment yields from basins with various landuses in the Auckland area (Hicks, 1994). This is consistent with erosion rates for rural land use measured during 1981–82 for the Upper Waitemata Harbour Watersheds (van Roon, 1983). It is also consistent with that used in the USC Model to date (Vant et al., 1993; Williamson et al., 1998a).

Event or storm loads and concentration data are available from a few catchments around Auckland, including the Manukau Experimental Basin (Hicks, 1994) and two Mahurangi catchments (Swales et al., 1997).

7.1.2 Urban construction loads

Measured yields

Sediment yields from erosion of subsoils during urban construction were measured by Hicks (1994), who developed long-term predictions for sediment specific yield for Auckland watersheds. His predictions for urban subdivisions were based on suspended sediment exported over a two-year period of urban development from the 220-ha Alexandra development. For developments with 100% bare soil and no detention ponds (ponds were not widely used until the late 1980s), the average specific yield over a 20-year period is 168 t ha$^{-1}$ yr$^{-1}$. Hicks (1994) also gives event loads from the Alexandra catchment. Sediment detention ponds in Auckland are commonly assumed to only trap 50% of eroded sediments. Thus, if ponds are installed and properly maintained, long-term average exports drop to 84 t ha$^{-1}$ yr$^{-1}$. Not all soil eroded from catchments reaches receiving waters; much is deposited on floodplains. Delivery
ratios (the proportion of eroded soil that is delivered to the receiving waters) commonly range from 0.33 to 0.66 (Novotny and Chesters, 1981), with larger values associated with small steep catchments, and smaller values associated with larger catchments with gentle slopes.

*Universal Soil Loss Equation Computations*

The Universal Soil Loss Equation (USLE) has recently been adopted by the ARC to estimate soil erosion. It is a predictive model of soil loss that relies on detailed rainfall records and the standardisation of soil characteristics. The USLE model was derived in the USA originally to estimate sheet and rill erosion from cropland. The USLE is formulated as follows:

$$A = R \times K \times LS \times C \times P$$

where: $A =$ soil loss (tonnes/year), $R =$ rainfall erosion index, $K =$ soil erodibility factor, $LS =$ slope length, $C =$ vegetation cover (crop) factor, $P =$ erosion control practice (protection) factor. The USLE approach has the following limitations in its application to Auckland earthworks (ARC, 1996):

1. It is an empirical formula only and not a mathematical representation of the actual erosion process.

2. It predicts average annual sediment generation, and unusually light or heavy rainfall is not represented. It does not predict event loads.

3. Sediment deposition in the catchment (e.g., on floodplains) is not calculated.

4. Sheet and rill erosion is calculated only. Channel erosion is not calculated and any rill erosion greater than 100 mm in depth will be underestimated.

5. USLE has not been calibrated for New Zealand conditions.

To estimate the final volume that is delivered to the estuary environment, the yields computed from the USLE need to be modified by:

1. Sediment control efficiencies, which are incorporated into the coefficients $C$ and $P$.

2. Duration of the earthworks season.
3. Delivery ratio (i.e., the amount of sediment deposited near the source or on floodplains on the way to the estuary).

4. Channel erosion (this is addressed later).

**Use of BNZ and WAM outputs**

Sediment inputs can be computed from physically based catchment models which use mathematical expressions to predict the various physical, biological and chemical process occurring in the soils and streams. Two of these models (BNZ, WAM) have generated information on soil erosion for many different landuse, slope, slope-aspect and soil types in the Auckland region. Such models are spatially distributed, that is, the catchment is broken down into a number of subcatchments or grid cells to account for spatial variability in landuse, soils, climate and stream conditions. These models work by calculating the load from each cell, then combining cell loads to derive the load at each subcatchment outlet. Cell loads are routed to the subcatchment outlet and then down the stream network to the estuary. Point sources are added during routing where appropriate, and attenuation is taken into account. The routing process does not incorporate stream erosion and deposition.

Cell outputs have been used to generate daily, seasonal and annual loads for catchments in parts of Auckland by employing output data appropriate to soil type, ground cover, slope and slope-aspect under consideration. This approach was used to estimate sediment inputs into Orewa Estuary (Williamson et al., 1998b) and Pakuranga (Williamson et al., 1998a).

7.1.3 **Streambank erosion**

Urbanisation results in an increase in flood frequency and size. Urban streams widen in response to such changes in hydrology, as the morphology re-adjusts to the new flow regime (Hammer, 1972; Morisawa and Laflure, 1982; Herald, 1989). Herald (1989) showed that cross-sections of stream channels draining to Helyers Creek increased threefold for an area that was 85% urbanised, which is consistent with overseas studies (Hammer, 1972; Morisawa and Laflure, 1982). Channel widening is an additional source of sediment to that derived from surface erosion. Only annual or other long-term estimates of sediment yields from streambank erosion are plausible at present.

Vant et al (1993) derived an equation for estimating the sediment loss from streambank erosion in urbanising catchments in the Upper Waitemata Harbour. The following assumptions were made:
1. Drainage density (m of channel per ha of watershed area) for watersheds are as given by Smith (1983) for Auckland watersheds.

2. Proportion of stream channel that is in 1st, 2nd, 3rd and 4th order is 0.6, 0.2, 0.15 and 0.05, respectively (expressed as water course lengths). (Here, stream order follows the classification method of Strahler, 1963).

3. Channel cross-sections are as measured in Albany basin (Herald, 1989). For 1st, 2nd, 3rd and 4th order streams, these are 0.44, 0.71, 1.29 and 2.92 m², respectively.

4. Stream widening is independent of stream order and is on average three times the rural cross-section area (Herald, 1989).

Then, streambank erosion in units of t yr⁻¹ is given by the total length of stream channel multiplied by the change in cross-sectional area per year:

\[ A \times L \times 3 \times 1.5 \times [(XSN_1 \times P_1) + (XSN_2 \times P_2) + (XSN_3 \times P_3) + (XSN_4 \times P_4)] \]

where \( A \) = area developed in one year (ha/yr); \( L \) = drainage density (m/ha); 3 is the widening factor (see above); 1.5 is the bulk density of the soil (t/m³); \( XSN_n \) (m²) is the cross-sectional area of order-\( n \) streams and \( P_n \) is the proportion of order-\( n \) stream channel.

7.1.4 Mature urban

Sediment specific yields from mature urban areas are similar to those found in average urban runoff (Williamson, 1993), i.e., ~0.4 t ha⁻¹ yr⁻¹. Urban stormwater treatment will reduce these by about 75% (ARC 1992).

7.2 Calculating heavy metals inputs

Urban impervious areas are continually supplied with fresh zinc (Zn), copper (Cu), polynuclear aromatic hydrocarbons (PAH) and, to a lesser extent, lead (Pb). While Pb has built up in the past it is now expected to decrease following the removal of Pb from petrol.

Specific yields (mass of potential contaminants exported from a unit area over a period of one year and expressed as kg ha⁻¹ yr⁻¹) for Pb, Zn and Cu have been compiled from data collected in New Zealand and overseas (Williamson, 1993;
Williamson et al., 1998a). Flow-weighted means are also available (Williamson, 1993) and these can be used to estimate storm event inputs to estuaries.

Lead exports have varied over the years. From 1986, Pb concentrations in petrol were approximately halved. Then, from 1995, Pb was phased out of petrol. For the period 1986–1995, Pb loads for any one year can be calculated by adjusting pre-1986 loads by the ratio of use of tetra-ethyl lead in that year divided by the use of tetra-ethyl lead in 1985.

The information in Williamson (1993) is somewhat dated, considering many of the studies cited in that work were conducted in the 70s and 80s. This may be a shortcoming for estimating estuary inputs because new urban areas have higher degrees of imperviousness, there have been changes in building materials, and traffic densities have increased. It is unlikely to be a major problem with inputs from older residential areas, but may underestimate loads from recent urban landuse, or where traffic densities have increased considerably. At the time of writing of this report, concentration and load data were being collected for 10 Auckland catchments, 4 Hamilton catchments and 3 Rotorua catchments. This new information should become available in 2001–2002, and should considerably enhance our ability to predict urban watershed exports.

For inputs of heavy metals from rural watersheds, specific yields can be estimated using suspended-sediment yields or loads and measured background concentrations.

7.3 Calculating PolyAromatic Hydrocarbons (PAH) inputs

Export coefficients for PAHs were estimated from measured concentrations in Hellyers, Pakuranga and Motions estuaries by performing a back-calculation with the USC model. Values from Hellyers, Pakuranga and Motions are listed in Table 7.1. On this basis we recommend using specific yields of 6 g ha$^{-1}$ yr$^{-1}$ for residential and general urban areas, and 20 g ha$^{-1}$ yr$^{-1}$ for areas with a high proportion of roads (e.g., commercial, industrial areas). The latter value is consistent with recommended values in the Urban Runoff Data Book (Williamson, 1993) that was derived from overseas studies.

As described above for heavy metals, new information, especially from Auckland catchments, should provide better specific yield data, as well as event data such as flow-weighted means.
Table 7.1  PAH loads in urban watersheds

<table>
<thead>
<tr>
<th>Estuary</th>
<th>Specific yield (g ha(^{-1}) yr(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pakuranga</td>
<td>4</td>
</tr>
<tr>
<td>Hellyers</td>
<td>8</td>
</tr>
<tr>
<td>Motions</td>
<td>20</td>
</tr>
</tbody>
</table>

7.4  Organochlorines

Organochlorine compounds such as DDT, PCB and Dieldrin are also found in urbanised estuaries and are sometimes of concern (Mills, 1997). However, these are mainly due to past use or to minor point sources and their use is not expected to continue or to grow. Erosion (and deposition) of topsoils is the major source of agricultural chemicals such as DDT. Topsoil DDT concentrations will not be increased by urbanisation. So, while they continue to be delivered to estuaries through topsoil erosion, their concentrations are not expected to increase to any significant extent.
8 PREDICTING CONTAMINANT ACCUMULATION IN PRIMARY DEPOSITION AREAS

Summary: to calculate $\Omega_N$ and $\Lambda_N$ for a PDA

- define site
- define events
  - for each event define $V_s$, $C_s$, and $R$ for all sources
- calculate $\Omega$ for each event
- calculate $\Lambda$ for each event
- sum event results over $N$ years

(Symbols are defined in the text that follows.)

Terrestrial muds are delivered to the estuary by one or more source streams. We conceive of each source as having an associated Primary Deposition Area (PDA), which is the area in which muds deposited by primary processes (i.e., flocculation and sudden loss of transporting capacity of freshwater inflow) are not readily redistributed by secondary processes between floods. Hence, muds steadily accumulate in each PDA.

(The PDA corresponds quite closely to the “tidal creek” of Hume et al. [2001], which is normally muddy. In the middle and lower reaches of the tidal creek, mud accumulates in the main reach and side arms, and particularly on the intertidal flats and amongst the mangroves. Once deposited in the tidal creek, sediment tends to remain trapped because mud is cohesive, tidal currents are weak there, and it is also sheltered from waves.)

The PDA associated with each source tends to radiate out from the source mouth, in the upper reaches of the estuary. There are two reasons for this. Firstly, since flocculation and loss of transporting capacity occur quickly, mud deposition rates are high close to where the source disgorges. Secondly, the upper reaches of estuaries tend to be sheltered from wind, and so wave entrainment of deposited muds also tends to be negligible. In addition, PDAs usually have significant intertidal areas, and settling in these areas is “stretched” over the estuary by tidal movement. A PDA may thus typically extend into the middle and lower reaches of an estuary along channel margins. PDAs from different sources may overlap.
Fig. 8.1  Okura estuary – bed sediments.

Using Okura estuary as an example (Fig. 8.1, Fig. 8.2), we would conclude that the North Arm PDA and the Redvale Arm PDA do not extend below, approximately, the Okura township. In this upper region of the estuary, the bed sediment is principally mud and waves are negligible (implying that secondary processes are negligible), mainly because the sand ridge affords shelter from NE winds. Since the channels draining the respective arms merge within this area, it is reasonable to assume that the two PDAs overlap. That is, mud deposited west of Okura township originates from both the North and Redvale Arms. Mud does drape the channel margins east of the sand ridge, so the PDAs could be taken as extending into these margins, but if so, they are relatively small in area.

Fig. 8.2  Okura estuary, showing the SRA and the Redvale and North Arm PDAs.
Contaminant accumulation within a PDA can be estimated as follows. To develop the method, we first address the case of deposition at a site that is located where PDAs associated with neighbouring sources overlap. Only one source is contaminated. The following equations relate to one flood event.

- The sediment discharged from source 1 has total mass of contaminant \( m_{s1} \) at a dimensionless (mass per mass) concentration \( C_{s1} = m_{s1} / V_{s1} \), where \( V_{s1} \) is total mass of sediment discharged from source 1.

- A flood event results in deposition of a total mass of sediment \( V_1 + V_2 \), where \( V_1 \) is mass of (contaminated) sediment from source 1 deposited at the site and \( V_2 \) is mass of (uncontaminated) sediment from neighbouring source 2 also deposited at the site.

- Concentration of contaminant associated with sediment from source 1 does not change during transport, and so \( C_1 = m_1 / V_1 \), where \( m_1 \) is mass of contaminant from source 1 deposited at the site.

- The deposited layer has mass \( \Lambda \) of contaminant, which in this single flood event equals \( m_1 \).

- The dimensionless (mass/mass) concentration of contaminant in the deposited layer is \( \Omega \), which in this case is \( \Omega = \Lambda / (V_1 + V_2) = m_1 / (V_1 + V_2) \). [Note this is the concentration in the deposited layer, not the final concentration after mixing with the underlying sediment through bioturbation].

- Now, assume \( R = V_s / V \), where \( V_s \) is mass of sediment discharged from a source and \( V \) is mass of sediment from that source deposited at the site. That is, \( 1 / R \) is the fraction of sediment discharged from the source that is deposited at the site.

- By substitution and using a little algebra, we can now find expressions for both \( \Omega \) and \( \Lambda \) in terms of variables that can be estimated: 
  \[ \Omega = R_2 V_{s1} C_{s1} / (R_2 V_{s1} + R_1 V_{s2}) \]  and 
  \[ \Lambda = \Omega ([V_{s1} / R_1] + [V_{s2} / R_2]). \]

- Both \( \Omega \) and \( \Lambda \) relate to contaminant (and sediment) deposition per event, and so multiplication by the number of those events in a given time period will give contaminant accumulation in that time period.

Similar equations can be derived for different scenarios (Table 8.1), and a pre-existing background contaminant concentration in the estuary can be quite easily accounted for.
Table 8.1  PDA equations.

<table>
<thead>
<tr>
<th># of sources / # of those sources that are contaminated</th>
<th>$\Omega$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>dimensionless (mass/mass) concentration of contaminant in deposited layer</td>
</tr>
<tr>
<td>1/1</td>
<td>$C_{s1}$</td>
</tr>
<tr>
<td>2/1</td>
<td>$R_{2}V_{s1}C_{s1}/(R_{2}V_{s1}+R_{1}V_{s2})$</td>
</tr>
<tr>
<td>2/2</td>
<td>$(R_{2}V_{s1}C_{s1}+R_{1}V_{s2}C_{s2})/(R_{2}V_{s1}+R_{1}V_{s2})$</td>
</tr>
<tr>
<td>3/1</td>
<td>$R_{2}R_{3}V_{s1}C_{s1}/(R_{2}R_{3}V_{s1}+R_{1}R_{3}V_{s2}+R_{1}R_{2}V_{s3})$</td>
</tr>
<tr>
<td>3/2</td>
<td>$(R_{2}R_{3}V_{s1}C_{s1}+R_{1}R_{2}V_{s2}C_{s2})/(R_{2}R_{3}V_{s1}+R_{1}R_{3}V_{s2}+R_{1}R_{2}V_{s3})$</td>
</tr>
<tr>
<td>3/3</td>
<td>$(R_{2}R_{3}V_{s1}C_{s1}+R_{1}R_{2}V_{s2}C_{s2}+R_{1}R_{3}V_{s3}C_{s3})/(R_{2}R_{3}V_{s1}+R_{1}R_{3}V_{s2}+R_{1}R_{2}V_{s3})$</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th># of sources / # of those sources that are contaminated</th>
<th>$\Lambda$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(mass of contaminant in deposited layer)</td>
</tr>
<tr>
<td>1/1</td>
<td>$\Omega(V_{s1}/R_{1})$</td>
</tr>
<tr>
<td>2/1</td>
<td>$\Omega([V_{s1}/R_{1}]+[V_{s2}/R_{2}])$</td>
</tr>
<tr>
<td>2/2</td>
<td>$\Omega([V_{s1}/R_{1}]+[V_{s2}/R_{2}])$</td>
</tr>
<tr>
<td>3/1</td>
<td>$\Omega([V_{s1}/R_{1}]+[V_{s2}/R_{2}]+[V_{s3}/R_{3}])$</td>
</tr>
<tr>
<td>3/2</td>
<td>$\Omega([V_{s1}/R_{1}]+[V_{s2}/R_{2}]+[V_{s3}/R_{3}])$</td>
</tr>
<tr>
<td>3/3</td>
<td>$\Omega([V_{s1}/R_{1}]+[V_{s2}/R_{2}]+[V_{s3}/R_{3}])$</td>
</tr>
</tbody>
</table>

To use the equations in Table 8.1 for predicting contaminant accumulation at any one location in a PDA: (1) $V_s$ for each source must be specified; (2) $C_s$ for each source must be specified; (3) $R$ for each source must be specified; (4) results must be extrapolated from a single event to the timescale of interest. To aid the discussion of how these are achieved, we present an example calculation in Table 8.2 for the case of two sources, one of which is contaminated. The aim is to evaluate the total mass of contaminant deposited at a site over $N$ years, $\Lambda_N$, and the average contaminant concentration in the sediment deposited over the $N$-year period, $\Omega_N$. For the example calculation, $N = 10$ years.
Table 8.2  Example calculation for a PDA (2 sources, one of which is contaminated). Notes follow Table.

<table>
<thead>
<tr>
<th>site area = 100 m²</th>
<th>5 floods / year</th>
<th>2 floods / year</th>
<th>1 flood / 5 years</th>
<th>1 flood / 10 years</th>
</tr>
</thead>
<tbody>
<tr>
<td>$C_{s1}$ [ ]</td>
<td>0.1</td>
<td>0.1</td>
<td>0.1</td>
<td>0.1</td>
</tr>
<tr>
<td>$V_{s1}$ [kg]</td>
<td>10,000</td>
<td>50,000</td>
<td>400,000</td>
<td>500,000</td>
</tr>
<tr>
<td>$R_1$ [ ]</td>
<td>2000</td>
<td>1000</td>
<td>500</td>
<td>100</td>
</tr>
<tr>
<td>$V_{s2}$ [kg]</td>
<td>20,000</td>
<td>80,000</td>
<td>1,200,000</td>
<td>2,000,000</td>
</tr>
<tr>
<td>$R_2$ [ ]</td>
<td>20000</td>
<td>10000</td>
<td>5000</td>
<td>200</td>
</tr>
</tbody>
</table>

Note 1

Note 2

| $\Omega$ [ ] | contaminant conc [kg] | 0.5 | 5 | 80 | 500 |
| $\Lambda$ [kg] | contaminant mass [kg]  | 6   | 60 | 1000 | 15,000 |
| $\Lambda / \Omega$ [kg] | sediment mass (kg) | 0.000036 | 0.00036 | 0.006 | 0.09 |
| $(\Lambda / \Omega)(\rho_{sed} * area)$ [m] | deposition thickness | 0.000036 | 0.00036 | 0.006 | 0.09 |
| # events / 10 years | 50 | 20 | 2 | 1 |
| mass of sediment deposited in 10 years [kg] | 300 | 1200 | 2000 | 15,000 |
| mass of contaminant deposited in 10 years [kg] | 25 | 100 | 160 | 500 |
| deposition thickness in 10 years [m] | 0.002 | 0.007 | 0.01 | 0.09 |
| $\Lambda_{10}$ [kg] | total mass of contaminant deposited in 10 years | 780 |
| total mass of sediment deposited in 10 years [kg] | 18,500 |
| $\Omega_{10}$ [ ] | average contaminant concentration in sediment deposited in 10 years | 0.04 |
| average annual deposition rate over 10 years [m/y] | 0.01 |

Notes accompanying Table 8.2:
1. All of these numbers are specified.

2. Calculated using equations in Table 8.1

From Table 8.2, \( \Lambda_{10} = 780 \text{ kg} \) and \( \Omega_{10} = 0.04 \). By way of discussion, the following points are noted.

- The site being considered here has an area of 100 \( \text{m}^2 \). Clearly, the value of \( R \) will vary with site area. If the site area is set equal to the entire area of the PDA then \( 1/R \) is the fraction of the flood sediment load that is deposited in the PDA. This is the easiest case to interpret: contaminant and sediment accumulations that are calculated are averages (or totals) over the PDA. As the site area decreases (a particularly sensitive habitat somewhere in the PDA might be of interest), then \( R \) will need to be increased (indicating that a smaller fraction of the total sediment load is deposited on the sensitive habitat). As the site area decreases, \( R \) will need to be specified with increasing precision, which will become increasingly difficult. Hence, it will be better to deal with larger area-averages if at all possible.

- We have chosen in this example to evaluate deposition associated with 4 different sized flood events. There are many possible choices here; we comment further, below, when discussing the 10-year extrapolation.

- \( R \) might also vary with size of event, which is another way of saying that the processes that deliver sediment to a site might vary with size of event. Again: the larger the area of the site, the easier it will be to account for any such variation when selecting values for \( R \). We comment further on selection of \( R \), below.

- Estimating contaminant concentration at source (\( C_s \)) is discussed in Chapter 7. Note that concentration \( C_s \) is assumed (in this example) to be independent of event size, but because mass of sediment delivered to the estuary increases with event size, so too does mass of contaminant.

- Estimating sediment discharge at source (\( V_s \)) is discussed in Chapter 7.

- \( \Omega \) is the mass/mass concentration of contaminant in the sediment layer deposited at the site during a single flood event, and \( \Lambda \) is the mass of deposited contaminant. Therefore, \( \Lambda / \Omega \) is the mass of sediment deposited at the site during the event. Dividing \( \Lambda / \Omega \) by the site area and then dividing by the density of the settled bed (assumed in the example to be 1650 kg/m\(^3\)) yields the event deposition thickness. Here we have our first “reality check”: this should be a reasonable number! (The numbers in Table 8.2 are purely fictitious!)
• The extrapolation presented here is fairly simpleminded. Certain flood sizes and associated frequencies of occurrence have been assumed, and deposition has simply been summed over all events. Clearly, a more rigorous statistical approach could be taken, including a full Monte Carlo simulation for estimating event occurrences. On the other hand, a simpler approach might be to examine a worst-case scenario or perhaps an “average” event. (The approach taken in the USC model was to use average annual loads.) The point here is that the core “per-event” predictions can be manipulated in any number of ways to estimate accumulation.

• The final row in Table 8.2 – average annual deposition rate over the 10-year period – is the final reality check. As previously, this should be a reasonable number!

At this point it is worth discussing briefly the limitations of the method, before returning to the discussion of \( R \). The “core” (or “central” – nothing to do with a sediment core) calculations are for single events. The extrapolations presented above did not allow for interactions between events. For example, there was no allowance for one event closely following another, which might have had an effect on the sediment yield of the second event. There was also no allowance for processes that occur between events, for example, deposition of sediment returned from middle and lower reaches of the estuary by the secondary processes described previously, or bioturbation, which mixes freshly deposited sediment down through the pre-existing sediment column.

The development this far centred on developing predictive equations for single events, and then combining or extrapolating those event-based equations to estimate accumulation over the long term. Although sound conceptually, there are practical difficulties associated with that approach. For instance, in developing catchments, \( V_s \) and \( C_s \) will change from year to year. It can, therefore, be simpler to apply the previous equations on an annual-average basis, in which case \( V_s \) and \( C_s \) for all sources will be annual-average values. \( 1 / R \) for each source now becomes the fraction of the annual average sediment load that deposits in the site of interest. Taking this approach also simplifies the incorporation of bioturbation/physical mixing, which could be done on an annual basis as has been done to date in the USC model (see Chapter 10).

The accuracy of predictions hinges primarily on the accuracy of \( R \) (either event or annual-average). At least initially, simulations with a numerical estuarine sediment transport model are the best way to estimate \( R \). For example, simulations with the Okura sediment transport model have yielded the following patterns in \( R \). Calculations are for the Redvale Arm PDA.
- $R$ varies with event size (Fig. 8.3). For the 2-floods-per-year event, $R = 1.09$ ($1/R = 0.92$), where $R$ corresponds to the entire Redvale Arm PDA. This means that $(1/R) \times 100 = 92\%$ of the sediment load for this event deposits inside the PDA. For the 1:5-year-flood, $R = 1.30$ ($1/R = 0.77$), which means that $77\%$ of the sediment load for this event deposits inside the PDA. For the 1:10-year-flood, $R = 2.13$ ($1/R = 0.47$), which means that $47\%$ of the sediment load for this event deposits inside the PDA. Hence, the fraction of the sediment load that deposits in the PDA decreases with increasing flood size. Higher freshwater runoff is more effective at delivering suspended sediment to the middle and lower reaches of the estuary because the flocculation zone (where freshwater meets seawater) is pushed down-estuary and because transporting capacity is maintained further down-estuary under the higher volume of freshwater. (Note: event size is not strictly related to probability of occurrence of event, which could be construed from the above, but probability of occurrence is a useful, informal indicator of event size.)

- Fig. 8.3 also shows how $1/R$ varies with area being considered. More sediment is seen to deposit on the intertidal flats above MLW (Mean Low Water) compared to below MLW (i.e., $1/R$ is larger for the intertidal flats above MLW). The two main reasons for this are: sediment does not have so far to settle to the bed above MLW; and currents are generally weaker higher on the intertidal flats. In addition, there is more area in the PDA occupied by intertidal flats above MLW than by flats below MLW.

- Fig. 8.4 shows how $1/R$ varies with distance from source. (In Fig. 8.4, calculations are extended beyond what we have previously identified as the Redvale PDA in order to emphasise patterns.) $1/R$ is shown for 13 equal-area segments of the estuary. Most sediment deposits in the upper reaches ($1/R$ is bigger in upper reaches), as expected, but the main locus of deposition varies with flood size. For the smallest flood, most sediment is deposited close to the mouth, but for the largest flood, maximum deposition occurs towards the down-estuary end of the PDA. For each event, the sum of all values of $1/R$ is the fraction of the sediment load deposited inside the estuary. For the smallest flood, approximately $5\%$ of the sediment input is discharged to the coastal ocean, and for the largest flood approximately $25\%$ escapes the estuary.

A concerted investigation in a few different types of estuary will yield patterns in $R$ that can be generalised by reference to the estuarine classification scheme that is presented below. The USC model adopted a value of 0.75 for $1/R$ (Williamson et al., 1998a) based on the ARC Stormwater Management Guidelines (ARC, 1992). The USC model addresses annual loads and thus “averages out” any variation in $R$. Furthermore, it assumes that $R$ values are similar for different sources. The discussion of $R$ presented here is intended to broaden understanding.
At this point we step back and comment on the approach that we are taking to predict contaminant accumulation. At the outset we claimed to be taking a semi-quantitative conceptual approach, yet now we are using a (quite sophisticated) numerical model to advance the method. In effect, we have used a numerical model to evaluate one parameter in our semi-quantitative conceptual model. Other options of course are available to us for putting a number to \( R \). For example, we could have referred to previous case studies, or we could have estimated on the basis of a range of best available information. This will be a common theme: there will usually be more than one way to put numbers to parameters, including running numerical models. In this sense, our conceptual model can be thought of as a way of extrapolating numerical model results, case studies and, indeed, our prior general knowledge of how estuaries work. All of these may provide us with the means for specifying values for the various needed parameters.

![Graph showing variation of \( R \) with event size](image)

**Fig. 8.3.** Variation of \( R \) with event size where the area corresponding to \( R \) is variously defined.
Fig. 8.4. Variation of $R$ with distance from source. The inset shows the area of the estuary corresponding to each $R$. The shading in the inset figure shows water depth.
9. PREDICTING CONTAMINANT ACCUMULATION IN SECONDARY REDISTRIBUTION AREAS

Summary: to calculate $\Omega_N$ and $\Lambda_N$ for the SRA

- define $F_m(t)$ for the site
- for the “annual average event”
  - for each source define $V_s$, $V_{co}$, $C_s$ and $R$, where $R$ is defined for the entire PDA associated with that source
  - calculate $\omega_h$
- calculate $\Omega_N$
- define $S_{\lambda \lambda}(t)$

(Symbols are defined in the text that follows.)

Beyond PDAs, in the middle and lower reaches of the estuary, mud is still deposited during floods by primary processes. However, in these areas, secondary processes significantly redistribute deposited muds to preferred locations between floods. These areas are called Secondary Redistribution Areas (SRAs). Waves are primarily responsible for entraining settled, alien muds and thus initiating redistribution of muds in the SRA. The Okura estuary SRA is shown in Fig. 9.1.

Fig. 9.1 Okura estuary SRA.
(The “drowned river valley”, “tidal lagoon” and “coastal embayment” of Hume et al. [2001] are all types of estuary distinguished by origin, geological setting, shape and/or geomorphology. Each type has an SRA and one or more PDAs. This is recognised explicitly in the case of the “compound estuary”, which is a series of tidal creeks [cf. PDA] draining into a tidal lagoon [cf. SRA]).

There is a fundamental problem to be solved if we are to predict contaminant accumulation in an estuary’s SRA. In the PDA associated with each source, we surmised that sedimentation primarily occurs in events (i.e., single rainstorm or flood). From there, we were able to construct a prediction scheme based on summing the effects of many events. There is also event deposition in the SRA, of course, but the long-term picture depends as much on what happens between events – when secondary processes redistribute sediment that is initially deposited during floods – as what happens during an event. The problem is that there is no simple way to predict the sequence of secondary processes that might occur between events, which will have a strong bearing on sedimentation at any particular site. For instance, mud that is deposited at a site during one flood may be completely eroded from the site during a wind-storm that occurs before the next flood delivers a new layer of mud. Or, the entire time between floods might be windless, in which case the layer deposited during the first flood will be buried by the layer deposited during the second flood. Obviously, any number of permutations is possible, but the fundamental difficulty is clear: sediment accumulation in the SRA is not essentially event-based and so an event-based prediction scheme is not appropriate. Another basic difficulty relates to defining inputs. For PDAs, we were able to quantitatively define (and then algebraically combine) all of the sediment inputs, which all came from the land. For the SRA, we have no easy way of estimating the input of marine sand, which might be a significant portion of the local sediment budget at any particular site in the SRA. To sum up, it is not appropriate to develop an event-based prediction scheme for the SRA, and we do not have enough information to develop an approach based on summing of inputs.

An alternative approach for the SRA is to “deconstruct” the annual average sedimentation rate (rather than “construct” it, which we did for PDAs), which, for the time being, we will take as being known. Furthermore, we will perform the deconstruction using the existing bed-sediment composition as a guideline.

This is a key step forward in quantifying our conceptual model of contaminant accumulation in the SRA. Our aim here is to predict the average concentration of contaminant in the sediment deposited at some given site in the SRA over \( N \) years, \( \Omega_N \), and the mass of contaminant deposited, \( \Lambda_N \). These same quantities were previously predicted for a PDA.
• Assume an annual average sedimentation rate at the site of interest $S_{AA}$ (units: cm/yr). At the present time, the fraction $F_m$ of the bed sediment is mud, where $F_m$ is dimensionless and $0 < F_m < 1$. Mud therefore accumulates at the site at an annual average rate of $S_{AA} F_m$. Note that $F_m$ is specified for the “mixed” layer, which will be beneath any recently deposited mud layers. Hence, $F_m$ reflects any physical and/or bioturbation processes that mix sediment at the site over the year.

• Think now of a single event (flood). We assume that mud from all terrestrial sources entering the SRA is mixed homogenously before it deposits throughout the SRA. By “homogenous”, we mean that the mixture everywhere throughout the SRA is composed of source sediments in the same proportion. (Note: we are not assuming that mud deposits uniformly throughout the SRA, only that muds from the various sources mix thoroughly before deposition.) Furthermore, as secondary processes redistribute the mud deposited during a flood, the concentration of contaminants in the mud being redistributed does not change. We can estimate the contaminant concentration in that homogenous mix, $\omega_H$, as follows.

  o The total mass of sediment escaping the PDA associated with source 1 is $V_{s1}(1 - 1/R_1)$. Likewise, $V_{s2}(1 - 1/R_2)$ escapes the PDA associated with source 2. The term $1/R$ has the same general meaning as previously: it is the fraction of the sediment load discharged from the source that is deposited over some area, but, in this case, we require $R$ to relate to the entire area of the PDA. That is, $1/R$ is the fraction of sediment discharged from the source that is deposited in the source’s PDA, and so $V_s(1 - 1/R) = V_s - V_s/R$ is the mass of sediment from that source that escapes from the PDA.

  o The mass of sediment from source 1 that escapes into the coastal ocean is $V_{col}$. Therefore, the mass of sediment that remains in the SRA is $V_{s1}(1 - 1/R_1) - V_{col}$. Likewise, $V_{s2}(1 - 1/R_2) - V_{col}$ from source 2 remains in the SRA. Adding the two yields the total mass of sediment that remains in the SRA, $V_{s1}(1 - 1/R_1) - V_{col} + V_{s2}(1 - 1/R_2) - V_{col}$.

  o Now, sediment discharged from source 1 has total mass of contaminant $m_{s1}$ at a dimensionless (mass/mass) concentration $C_{s1} = m_{s1}/V_{s1}$. The contaminant concentration associated with sediment solely from source 1 does not change during transport, and so mass of contaminant in the SRA is $C_{s1}[V_{s1}(1 - 1/R) - V_{col}]$. 
The contaminant concentration in the homogenous mixture in the SRA, $\omega_{H}$, is then the mass of contaminant per total mass of sediment, $C_{H}\frac{V_{s1}(1 - 1/R_{1}) - V_{col}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2}}$.

Similar equations for $\omega_{H}$ for different combinations of sources can be derived, as shown in Table 9.1.

**Table 9.1** SRA equations (for clarity, assuming no pre-existing background contamination).

<table>
<thead>
<tr>
<th># of sources</th>
<th># of those sources that are contaminated</th>
<th>$\omega_{H}$ (dimensionless [mass/mass])</th>
</tr>
</thead>
<tbody>
<tr>
<td>1/1</td>
<td></td>
<td>$C_{s1}$</td>
</tr>
<tr>
<td>2/1</td>
<td></td>
<td>$C_{s1}\frac{V_{s1}(1 - 1/R_{1}) - V_{col}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2}}$</td>
</tr>
<tr>
<td>2/2</td>
<td></td>
<td>$C_{s1}\frac{V_{s1}(1 - 1/R_{1}) - V_{col}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2}} + C_{s2}\frac{V_{s2}(1 - 1/R_{2}) - V_{co2}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2}}$</td>
</tr>
<tr>
<td>3/1</td>
<td></td>
<td>$C_{s1}\frac{V_{s1}(1 - 1/R_{1}) - V_{col}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2}} + C_{s2}\frac{V_{s2}(1 - 1/R_{2}) - V_{co2}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2}} + C_{s3}\frac{V_{s3}(1 - 1/R_{3}) - V_{co3}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2} + V_{s3}(1 - 1/R_{3}) - V_{co3}}$</td>
</tr>
<tr>
<td>3/2</td>
<td></td>
<td>$C_{s1}\frac{V_{s1}(1 - 1/R_{1}) - V_{col}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2}} + C_{s2}\frac{V_{s2}(1 - 1/R_{2}) - V_{co2}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2}} + C_{s3}\frac{V_{s3}(1 - 1/R_{3}) - V_{co3}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2} + V_{s3}(1 - 1/R_{3}) - V_{co3}}$</td>
</tr>
<tr>
<td>3/3</td>
<td></td>
<td>$C_{s1}\frac{V_{s1}(1 - 1/R_{1}) - V_{col}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2}} + C_{s2}\frac{V_{s2}(1 - 1/R_{2}) - V_{co2}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2}} + C_{s3}\frac{V_{s3}(1 - 1/R_{3}) - V_{co3}}{V_{s1}(1 - 1/R_{1}) - V_{col} + V_{s2}(1 - 1/R_{2}) - V_{co2} + V_{s3}(1 - 1/R_{3}) - V_{co3}}$</td>
</tr>
</tbody>
</table>

- But how should we interpret this single flood event? Since we are dealing with an annual average sedimentation rate, it makes sense to think of an “annual average (flood) event”. What this means in practice is that $R$ for each source must be an average value (such that $1/R$ is the average fraction of the sediment load that deposits in the PDA). As it turns out, the actual numbers we choose to use for $V_{s1}$ and $V_{s2}$ are quite conveniently irrelevant, so long as the ratio $V_{s1}/V_{s2}$ reflects an average. Similarly, the ratios $V_{col}/V_{s1}$ and $V_{co2}/V_{s2}$ need to reflect an average. (Once those ratios and actual values for $V_{s1}$ and $V_{s2}$ are specified, actual values for $V_{col}$ and $V_{co2}$ will of course become known.)

- We now assume that, on average, mud deposits from the homogenous mix with contaminant concentration $\omega_{H}$. Therefore, the annual average contaminant deposition rate at our site of interest is given by $\omega_{H}S_{AA}F_{m}$. This is the “core” piece of information that we now extrapolate. (Here, we have assumed that the contaminant concentration in the sand is zero, and also that no mud comes from...
marine sources. The latter could be accounted for in calculations as any other source.)

- For the first example of an extrapolation, we assume that $F_m$ does not change over time (i.e., the bed gets neither sandier nor muddier over time). The average concentration of contaminant in the sediment deposited over $N$ years, $\Omega_N$, is the ratio of mass of contaminant deposited to mass of sediment deposited, which in this case is simply:

$$\Omega_N = \omega_h S_{AA} F_m N / S_{AA} N = \omega_h F_m$$

Note that we do not need to know the annual average sedimentation rate, $S_{AA}$, to estimate contaminant concentration.

- For example: assume we calculate $\omega_h = 0.1$ mg/kg or 0.000001 (dimensionless) using the previous equations and $F_m = 0.5$ (i.e., the mixed layer in the bed at the site is half mud and half sand). After 10 years (after any number of years, in fact), $\Omega_{10} = \omega_h F_m = 0.1 \times 0.5 = 0.05$ mg/kg (0.0000005 [dimensionless]).

- To prepare for more complicated extrapolations, we can look a little more deeply at how we arrive at this number. Assume $S_{AA} = 10$ cm/yr. With $F_m = 0.5$, mud therefore accumulates at the site at a rate $S_{AA} F_m = 10 \times 0.5 = 5$ cm/yr. This is equivalent to a mud accumulation rate of 5 cm/yr $\times$ 1.3 g/cm$^3$ = 6.5 g/(cm$^2$/yr) (assuming mud settles at density 1.3 g/cm$^3$). Over 10 years, 65 g/cm$^2$ of mud accumulates. Now, the mass/mass concentration of contaminant in the mud fraction is 0.0000005 (this is half the total concentration, because half the sediment is mud). Thus, each year, 0.0000005$\times$6.5 g/(cm$^2$/yr) of contaminant accumulates, and over 10 years 0.0000005$\times$65 g/cm$^3$ of contaminant accumulates. The average contaminant concentration in the accumulated sediment, $\Omega_{10}$, is simply 0.0000005$\times$65 g/cm$^2$ / [65 (g/cm$^3$)] = 0.0000005, which does indeed equal $\omega_h F_m$.

- For the second example of an extrapolation, we assume that $F_m$ does change over time (i.e., the bed gets either sandier or muddier over time). $\Omega_N$ is still the ratio of mass of contaminant deposited to mass of sediment deposited, which in this case is given by:

$$\Omega_N = \frac{\omega_h}{N} \int_{t=1}^{N} F_m(t) \, dt$$
Here, $F_m(t)$ describes the change in $F_m$ over time. As a simple example, assume $F_m$ changes as shown in Fig. 9.2 (the bed gets muddier over time).

![Graph showing the change in $F_m$ over time]

$F_m = Xt$, where $X = 0.1\, \text{yr}^{-1}$

**Fig. 9.2** Example of change in $F_m$ over time.

We now show in Table 9.2 how the integration might be performed (numerically). For this example $\omega_h = 0.1$ and $N = 3\, \text{yr}$.

**Table 9.2** Example integration for an SRA.

<table>
<thead>
<tr>
<th>$t$ (yr)</th>
<th>$F_m(t)$</th>
<th>$dt$ (yr)</th>
<th>$F_m(t),dt$ (yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1-2</td>
<td>0.1x1.5</td>
<td>1</td>
<td>0.15</td>
</tr>
<tr>
<td>2-3</td>
<td>0.1x2.5</td>
<td>1</td>
<td>0.25</td>
</tr>
<tr>
<td>3-4</td>
<td>0.1x3.5</td>
<td>1</td>
<td>0.35</td>
</tr>
</tbody>
</table>

$\Omega_3 = \omega_h / 3 \int F_m(t)\,dt = (0.1/3) \times 0.75 = 0.025$

The results is $\Omega_3 = 0.025$ (dimensionless). Note, again, that we do not need to know the annual average sedimentation rate, $S_{\text{AA}}$, to estimate contaminant concentration.

- The annual average sedimentation rate, $S_{\text{AA}}$, and the bed-sediment density, $\rho_{\text{bed}}$, are needed to estimate contaminant mass deposited over $N$ years, $\Lambda_N$. Since $\Omega_N$ is defined as the ratio of $\Lambda_N$ to mass of sediment deposited, and sediment mass (per unit area) is given by $S_{\text{AA}}\,N\rho_{\text{bed}}$, then $\Lambda_N$ is given by:

$$\Lambda_N = \Omega_N S_{\text{AA}} N \rho_{\text{bed}}$$
\( \Lambda_t \) has units of mass per unit area.

Because the method is based on a contaminant concentration associated with an “annual average event” (i.e., \( \omega_{\text{H}} \)), the prediction timeframe should be “many” years (\( \gg 10 \) years). The extrapolation is begun from the present sediment composition, which is the net result of present inputs (including marine sands) and mixing (physical and bioturbation) processes. The muddier the site, the more contaminants are concentrated, which is a good feature of the method. In the limit of pure mud, contaminants accumulate at the same concentration that they are delivered from the combined sources. Contaminants cannot “hyperconcentrate”! On the other hand, contaminants do not accumulate at all in pure sand beds, which is also reasonable (given that we are starting from the proposition that contaminants attach to fines).

Calculations could be performed with a range of scenarios for how bed composition changes over time (i.e., rate at which it gets muddier or sandier) to explore, for instance, worst-case contaminant accumulation. Note, the volume calculations are constrained by an annual average sedimentation rate, which should be reasonable. Finally, this approach quite naturally enables the incorporation of bioturbation/physical mixing, which could be done on an annual basis as has been done to date in the USC model (see Chapter 10).
10 SEDIMENT MIXING AND ESTIMATION OF CONTAMINANT CONCENTRATION IN THE BIOTURBATED LAYER

10.1 Introduction

Deposited sediment mixes downwards, principally by bioturbation, thus effectively diluting the freshly deposited sediments. We need to allow for the mixing process in order to calculate the resulting concentration in the surface sediments. Note that there are two issues in respect to the deposition of contaminated sediments in estuaries. There is the situation when sediment is freshly deposited, before it is mixed into the underlying sediments. We may need to know this concentration in order to evaluate the risk to surface feeders. This situation constitutes an acute exposure for these organisms. There is also the second situation when freshly deposited sediments are mixed downward by bioturbation and wave action. This is the long-term situation, and the environmental risk of most concern in this case is chronic exposure to a wider number of estuarine sediment-dwelling animals (including surface feeders, sediment dwellers and sub-surface feeders).

In the USC model, we only assessed the second situation (i.e., the long-term concentration in the bioturbated layer). In this case, we assumed that mixing was very simple and uniform (homogeneous) down to 15 cm, the approximate thickness of the bioturbated layer. We assumed no mixing below 15 cm. We also assumed that complete mixing occurred within a year, the time interval that the USC model dealt with. In reality, mixing is not uniform and is highly dependent on type and number of bioturbators.

10.2 Bioturbation mechanisms

Mud crabs are probably one of the most important bioturbators in the muddy areas of Auckland estuaries, at least in the intertidal zone, since they are abundant, relatively large, and burrow extensively. Shrimps may be the most important bioturbators in the subtidal zone. Shrimps excavate burrows by moving sediments lower in the sediment profile up to the surface. Burrows become blocked with surface sediment falling down the burrow, especially when the surface is agitated by “wavelets” (tiny waves) associated with the tidal front passing over the burrows. This sediment needs to be expelled to maintain the viability of the burrow. In addition, new burrows are formed frequently. Other small animals (worms, etc.) mix sediment between the burrows.

It is generally agreed that Helice crassa’s burrow depth, burrow complexity, and population density are all significantly greater in well-drained fine sediments that have had several hours’ aerial exposure during low tide (Morton and Miller, 1968; Nye, 1977; Jones and Simons, 1982; Takeda and Kurihara, 1987; Morrisey et al., 1999;
Roper et al., unpubl.). Morrisey et al. (1999) report mean burrow depths of ~25 cm for *Helice crassa* in mud or muddy sand, significantly deeper than 15 cm. However, the choice of 15 cm in the USC model was a compromise for a discontinuous mixing function (i.e., completely mixed above 15 cm and not mixed below 15 cm), representing gradually declining mixing over 10–25 cm depth.

*Helice crassa* distribution is also reported to vary seasonally – downshore in summer, up in winter – and activity will certainly vary with temperature and food supply. Furthermore, populations may also be seriously impacted by frequent flood events, which are rather more likely in winter and spring. Thus it seems likely that bioturbation varies both spatially and temporally, neither of which is reflected in the current model.

In sandy sediments, crabs are absent or present only in low numbers. Physical mixing (due primarily to wave action) may become more important than bioturbation, although worms and stingrays may be locally important (Williamson et al., 1991; ARC, 1994).

Whether considering mud or sand, physical or biological mixing, the intensity of this process probably decreases with depth in the sediment profile. Therefore, the assumption of uniform mixing within the bioturbated layer is probably incorrect.

We reviewed the international literature on sediment bioturbation models to see how the heterogeneity in mixing was handled, and to see if more realistic models could be adopted for the problem of long-term accumulation of contaminants.

### 10.3 Literature review of bioturbation models

Most bioturbation models in the literature assume a (one-dimensional) vertical diffusive process, which allows the bioturbation modeller to borrow some results from diffusion theory. This assumption is empirical rather than mechanical, although it has been demonstrated that many bioturbation processes do generate contaminant concentration profiles that look like the results of diffusion (Wheatcroft et al., 1990). Sometimes an advective term is added, but this only seems necessary in the specific case of bioturbation by “reverse conveyor-belt” deposit feeding. The key parameter in pure diffusion models is the biodiffusivity $D_b$, which is either constant or dependent on some controlling factor. This may be bioturbator population density, the lower limit of the animals’ habitat (cf. Boudreau, 1994), the cross-section of the animals (Wheatcroft et al., 1990; Swift et al., 1996), or even oxygen concentration at depth (Dhakar and Burdige, 1996).
However, because the USC model and the new models presented herein for the PDA and SRA are box models, with discrete, (e.g., annual) inputs of contaminant and sediment, it is rather difficult to apply the continuous functions of diffusion theory. It would be necessary to discretize, that is, iterate by solving:

\[
\frac{\Delta C}{\Delta t} = \frac{\Delta}{\Delta z} \left( D_b(z) \left( \frac{\Delta C}{\Delta z} \right) \right)
\]

(where \(z\) is depth downwards) for each deposited layer at each time step (and at each point, if \(D_b\) is allowed to vary around the estuary). This would be very difficult. Other approaches, involving signal theory or Markov chains (cf. Matisoff, 1982) would be even more difficult to implement. To get around this difficulty, several convenient compromises are available.

1. **Allow partial mixing.** The completely mixed fraction could be fixed at some value less than 1, i.e. some of the material in each layer is not subject to mixing processes in any given year. More sophisticated variants of this partial mixing approach could make the mixed fraction dependent on some measurable variables, such as *Helice crassa* population density or a well-correlated physical parameter (e.g., sedimentation rate, cf. Boudreau, 1994). There is at least one example of a simple partial mixing model in the literature (Abril et al., 1992), although it does not seem mechanically reasonable because the unmixed fraction in each layer can never be involved in mixing.

2. **Allow mixing to decrease with layer burial depth (stepwise heterogeneous mixing model).** The mixed fraction might decrease linearly, exponentially, in a half-Gaussian fashion, etc. (cf. Matisoff, 1982). Again, the parameters could be allowed to depend on measurables. This is perhaps most convenient for those functions (such as exponential decay) that have one parameter describing the extent of mixing, and another that describes the depth of mixing (e.g., see equation (7) in Appendix 1).

We suspect that this stepwise heterogeneous mixing model would be the most appropriate model for Auckland estuaries. The mathematical formulation is given in Appendix 1. A simple explanation of this model is as follows. Imagine that part of the sediment is completely mixed in a particular year – perhaps “all sediment used in burrows” or some such. This material includes all of the top layer, most of the second layer, a lot of the third layer, some of the fourth… down to a tiny bit of the tenth layer. In this way, deeper layers undergo less mixing, perhaps because not so many animals burrow that far down. Mathematically, the concentrations in all those mixed fractions
are averaged. Now the top layer has been altered by material from several other layers with different concentrations, so its concentration could be a lot higher or lower than its unmixed value, depending on circumstances. Only a tiny bit of the tenth layer was allowed to mix with shallower layers, so the mean concentration in that layer will not have changed much.

Unfortunately, there are no reliable values available for parameters that drive such a model, or even for the measurables that ought to drive them. In the same predicament, Niedoroda et al., (1996) simply fitted their diffusion model to the observed concentration profiles. There does not seem to be any equivalent published information for where heterogeneous mixing models have been fitted to observed data. However, it must be remembered that macrofaunal bioturbation is inherently “patchy”. Sediments will inevitably be horizontally heterogeneous at most scales, which was observed in the cores collected from Pakuranga estuary (e.g., see Fig. 2.2). However, all the above models predict only depth-averaged results, so it is difficult to fit models to vertical profiles from single cores in any case.

10.4 Review of contaminant concentration profiles

Here we consider contaminant concentration profiles observed in Auckland estuaries to see what we can glean from them as to an appropriate mixing model. Most of our information comes from a study in Pakuranga Creek (Williamson et al., 1998a). Observed profiles are highly variable (Fig. 10.1), but do show an increase in concentration from “background” concentrations to higher concentrations at the surface. When compared with the USC model predictions (Fig. 10.1), it is obvious that the model does not replicate the profiles. The depth of contamination at P1 is much greater (about 4 times) than predicted, while that at P2 is less. P3 shows contamination “peak” at depth. Overall, given the observed variability, it is difficult to conceive of any general model being able to replicate these profiles.
However, before dismissing simple modelling approaches as being inadequate, let us examine the above information more carefully.

Firstly, we can recognize that core P1 was taken near the head of the estuary where there are very high sedimentation rates – about 4 times higher than predicted from the model. The model only purports to represent the “average” situation in the estuary, so the disagreement is not critical. Let us use the sedimentation rates observed at site P1 and “scale up” the sedimentation rates used in the model by that amount. Fig. 10.2 compares model predictions of contaminants with observations after this adjustment for sedimentation rate is made. From this comparison, one can conclude that there is general agreement between observed and predicted contaminants in this case. This agreement may seem artificially contrived, but recall that it is the USC model with
only the sedimentation rate adjusted. The general agreement has some important implications, as we will show later.

Next consider the profile at site P2. The core taken from this site experienced a lot of compression when it was collected, and in the absence of further information, a linear compression correction was applied over the length of the core (Williamson et al., 1998a). However, there is good reason to suspect that there may have been uneven compression. For example, surface sediments are likely to be the most compressible, because they are riddled with crab burrows. We have observed this large-scale compression numerous times when collecting shallow cores greater than 10 cm deep. If the observed compression correction of nearly 20 cm is applied to the top 25 cm, then the agreement between the observed and predicted contaminant profiles at P2 is much better (Fig. 10.3, c.f. Fig. 10.1).

![Graph comparing observed and predicted contaminant profiles.](image)

**Fig. 10.2** Comparison of contaminant observations with model predictions after observed sedimentation rates are used in the USC model.

In respect to P3, it is probably not worthwhile pursuing this line of argument. It lies towards the end of the so-called “settling zone” assumed in the USC model, so it may be more realistic to expect a lesser overall sedimentation rate than the average assumed in the USC model. The complex profile with two peaks of contamination (Fig. 10.1) may be entirely due to some localized effect. This effect could result from greatly increased localized erosion or deposition. The deeper peak may be due to the infilling of an extensive burrow complex by surface muds. It could even be something as trivial as foot-prints pushing surface mud deeper into the sediment (humans normally sink 20–70 cm into the mud when walking through these intertidal mudflats).
Let us now consider the case with no mixing, where sediment is laid down, but does not mix with the underlying sediment. The predicted profile is given in Fig. 10.4 for Zn. Very high concentrations would occur with no mixing, and the depth of contamination is much less than predicted with the USC model (compare Fig. 10.4 and Fig. 10.1). Therefore, one can see that sediment mixing is very important. It can also be seen when comparing observed and predicted in Fig. 10.2 and with Fig. 10.4, that the “magnitude” (or depth) of mixing, assumed to be 15 cm in the USC model, is about right, even if the assumption of homogeneous mixing is unrealistic. Note, however, the wide fluctuations in Zn concentrations in core P1 reflect to some extent poor mixing from time to time.

Fig. 10.3. Predicted and observed contaminant concentrations at P2 after compression correction has been confined to the top 25 cm of the core.

Fig. 10.4. Predicted (USC model) Zn concentrations in Pakuranga estuary with no sediment mixing.
10.5 Conclusions on sediment mixing models

We can make the following conclusions on mixing.

1. It is currently not possible to choose a sediment mixing model on the basis of the observed contaminant profiles. They are too complex and variable, as well as subject to some uncertainty, for us to discern a useful, estuary-average mixing strategy.

2. The homogeneous mixed bioturbation model used in the USC model is probably a good enough approximation at the present time. However, the review (Section 10.3 and Appendix 1) has identified some more realistic models that could be activated and parameterised when better sediment core information becomes available.

3. Further research on contaminant profiles is necessary. The difficulties experienced with the Pakuranga cores lead us to make some recommendations as to the sampling method that should be used.

We have not used the suggested improvements to model bioturbation in this report (Section 10.3 and Appendix 1) because we do not have the necessary information to select and calibrate the most appropriate model. Choosing, deriving and calibrating a more sophisticated model is not a trivial task, so we have delayed taking this step until the requisite information is available.
11. GENERALISING WITH AN ESTUARY CLASSIFICATION SCHEME

Individual case studies may be conducted to identify PDAs, trends in $R$, values of $R$, and bed-sediment composition in the SRA. An estuary classification scheme that groups like estuaries will then be useful for extending case-study results to other estuaries in the region. We present such a scheme here. The scheme combines our understanding of estuarine geomorphology and the processes that disperse sediments/contaminants. Experience gained with application of the contaminant-accumulation model will be transferable to other estuaries using the estuary classification scheme.

11.1 Auckland estuary archetype

We recognise an Auckland estuary archetype, which is shown in Fig. 11.1.

![Diagram of Auckland estuary archetype](image)

The archetype displays several elements, which are defined in terms of “connections”. The arm connects directly to the land, and the throat connects directly to the coastal ocean. Between the two, and therefore with indirect connections to both the land and to the ocean, is the main body of the estuary. Arms correspond to PDAs and main bodies and throats are generally in the SRA. The classification scheme, based on how sediments flow between archetypal elements, and with some example estuaries, is shown in Table 11.1.
Table 11.1 Auckland estuary classification scheme.

Each category has “exposed” and “sheltered” subtypes. The exposed subtype is exposed to wind that frequently whips up energetic waves. The sheltered subtype is sheltered from wind that would otherwise stir up waves. Subtype does not necessarily have a bearing on geomorphology, but it does have a bearing on contaminant accumulation in main bodies.

11.2 Characteristics of archetypal elements – Arm

Source streams discharge directly into arms and so arms are typically muddy. Marine sands do not readily penetrate arms because the connection between the arm and the ocean is an indirect one, via the main body. The main processes governing sedimentation and contaminant accumulation are (1) settling associated with flocculation of fines delivered in suspension in freshwater, and (2) the sudden loss of transporting capacity as freshwater discharges into the estuary. Arms are sheltered from waves. The main geomorphic features are mudbanks; steep-sided channels, sometimes with armoured bottoms; and mangroves, marshes and backswamps. There is little variation with estuary type, the reason being that the nature of the estuary proper has
little to do with how arms function. Arms are the main sites for deposition of terrestrial sediment deposition. Therefore, the potential for contaminant accumulation in arms is high. Arms may extend down-estuary along channel margins.

11.3 Characteristics of archetypal elements – Main body

The main body is muddy in the upper reaches, where it connects with the arms associated with source streams. The lower reaches, which are connected to the coastal ocean via the throat, tend to be sandy. The main process is secondary redistribution of muds injected from arms during floods and of sands that are pushed through the throat by waves and tides. Secondary redistribution is achieved mainly by tides, internally generated waves, and freshwater intrusions. The main body is characterised by mixed sediments (mud and sand) and, in places, encroachment of muds on sands and vice versa.

In addition to the characteristic longitudinal gradient in sediments, there is also a typical cross-estuary gradient: sandy intertidal flats lie between mud along channel margins and mud or sand/shell on upper intertidal-flat margins (Fig. 11.2).

---

**Fig. 11.2** Schematic cross-sections through main body. Refer to text for explanation of labels.
Referring to Fig. 11.2 (colours match those used in the figure)...

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

- (b) Mud accumulations along subtidal margins of channels.

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

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

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

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

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

Variations by estuary type in main body morphology (see Table 11.1) are shown in Table 11.2.
Table 11.2 Variations by estuary type in main body morphology.

<table>
<thead>
<tr>
<th>A1</th>
<th>A2</th>
<th>A3</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Large main body.</td>
<td>• Large main body.</td>
<td>• Large main body.</td>
</tr>
<tr>
<td>• Extensive sand deposits.</td>
<td></td>
<td>• Sand body near mouth limited, therefore longitudinal mud–sand gradient subdued.</td>
</tr>
<tr>
<td>B1</td>
<td>B2</td>
<td>B3</td>
</tr>
<tr>
<td>• Arm and main body compressed together, mud and sand bodies intermingled.</td>
<td>• Main body compressed.</td>
<td>• Main body compressed.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>• Sand body near mouth limited, therefore longitudinal mud–sand gradient subdued.</td>
</tr>
</tbody>
</table>

The potential for contaminant accumulation in the main body is medium relative to sites of primary deposition in arms. This is because the main body is downstream of primary points of injection. Also, the secondary processes characteristic of the main body act to redistribute, mix and dilute contaminants. Nevertheless, contaminants will accumulate locally where mud accumulates. In general, accumulation will:

- follow the longitudinal mud gradient (higher in the upper reaches, lower in the lower reaches);
- follow the cross-estuary mud distribution (high near channel edge and high on upper margins of intertidal flats);
- be high in local sheltered areas or impoundments behind wave-built barriers.

The distribution of mud in the main body will change as the estuary “ages”. Generally, mud spreads at the expense of sandier substrates, and mud deposits may be further stabilised by the advance of vegetation (e.g., mangroves). Encroachment of contaminated mud over previously contaminant-free sandy habitats may represent a significant ecological risk. Estuary infilling by mud may be exacerbated by changes in catchment landuse.

Table 11.3 shows how the potential for contaminant accumulation in the main body varies by estuary type.
Table 11.3 Variations by estuary type in potential for contaminant accumulation in main body.

<table>
<thead>
<tr>
<th>A1</th>
<th>A2</th>
<th>A3</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Efficient overall trapping.</td>
<td>• Efficient overall trapping.</td>
<td>• Efficient overall trapping.</td>
</tr>
<tr>
<td>• Settling largely on sand.</td>
<td>• Settling on sand and mud.</td>
<td>• Settling on sandy mud.</td>
</tr>
<tr>
<td>• Dilution by sand.</td>
<td>• Dilution by sand and mud.</td>
<td>• Dilution by sandy mud.</td>
</tr>
<tr>
<td>• Concentration proportional to mud content.</td>
<td>Exposed subtype:</td>
<td>Exposed subtype:</td>
</tr>
<tr>
<td></td>
<td>• Rapid redistribution to mud/sand discontinuity.</td>
<td>• Rapid redistribution.</td>
</tr>
<tr>
<td></td>
<td>• Less opportunity for incorporation into sands.</td>
<td>Sheltered subtype:</td>
</tr>
<tr>
<td>Exposed subtype:</td>
<td></td>
<td>• Slow redistribution.</td>
</tr>
<tr>
<td></td>
<td>• Slow redistribution to mud/sand discontinuity.</td>
<td>More opportunity for incorporation into sands.</td>
</tr>
<tr>
<td>Sheltered subtype:</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>• More opportunity for incorporation into sands.</td>
<td></td>
</tr>
<tr>
<td>B1</td>
<td>B2</td>
<td>B3</td>
</tr>
<tr>
<td>• Less efficient trapping; most contaminants escape to coastal ocean during floods.</td>
<td>Less efficient trapping; significant contaminant escape to coastal ocean during floods.</td>
<td>Less efficient trapping; some contaminants escape to coastal ocean during floods.</td>
</tr>
</tbody>
</table>

11.4 Characteristics of archetypal elements – Throat

Throats are “conveyor belts” that link the estuary to the adjacent coastal ocean. Tidal currents tend to be strongest in the throat. Channel bottoms may be extensively lagged. Ocean waves may impinge on the throat and penetrate through to the main body of the estuary. Coastal littoral drift is interrupted by the throat, forming reentrant spits, bars and flood-tide deltas. Auckland estuaries tend to be “anchored” between hard headlands, which means that the throat can be restricted. In this case there will be little room for mud accumulation along the margins of the throat. Where margins do occur, they may generally be exposed to waves and winds, which tend to build sandy or shelly beaches.

The potential for contaminant accumulation in throats is small, relative to arms and main bodies. This is because throats are energetic; exposed to swell, waves and strong tidal currents; and usually remote from sources of contaminants.
12. A FINAL COMMENT

We have presented in this report a methodology for predicting contaminant accumulation over the long term in Auckland estuaries. The methodology is based on a broad understanding of sediment processes in estuaries and the way contaminants interact with sediments. The method is more a coherent assimilation of knowledge than it is a cookbook. At the very least, the way the core calculations, which apply to events or annual-average scenarios, are extrapolated to the long term will vary depending on the specific question being asked. It is probably more accurate to view the new methodology as a foundation, upon which a range of specific approaches will be built over time as regional and case studies are conducted.
REFERENCES


APPENDIX 1. MATHEMATICAL FORMULATION OF THE MIXING FUNCTION

Consider an urban estuary with several known layers of sediment, resulting from known annual depositions of sediment from its watershed. Let a uniform fraction $g$ of this sediment be deposited over the settling zone, area $A$, in an even layer of density $\rho$, and for convenience let:

$$k = \frac{g}{\rho A} \quad (1)$$

Let $s_i$ be the mass of sediment released from the watershed in the $i^{th}$ year, which forms the $i^{th}$ layer of sediment in the estuary; let $l_i$ be the thickness of that layer, and $z_{ij}$ the burial depth of that layer in the $j^{th}$ year. Let a mass of contaminant $m_i$ be entrained homogeneously throughout $s_i$, resulting in a concentration $c_{ij}$ in that $i^{th}$ layer in the $j^{th}$ year. Since $s_i$ and $m_i$ are known, derive:

$$l_i = k s_i \quad (2)$$

$$z_{ij} = k \sum_{a=i}^{j} s_i \quad (3)$$

The $c_{ij}$ terms are easy to evaluate if the layers are not mixed:

$$c_{ij} = \frac{m_i}{s_i} \quad (4)$$

But consider a simple partial mixing model where bioturbation function $B$ homogeneously mixes some proportion $b_{ij}$ of each layer. Now it is preferable to generate $c_{ij}$ iteratively, via

$$c_{ij} = b_{ij} m_j + \sum_{h=i}^{j} b_{ij} c_{h,j-1} s_h + (1 - b_{ij}) c_{i,j-1} \quad (5)$$

The concentration in any layer involved in mixing is a combination of the concentration that it had the previous year, and of a function of the concentrations in the other layers. $B$ can take any reasonable form so long as it is confined to the interval $[0,1]$ (i.e., the mixed fraction can change from layer to layer according to any function, so long as meaningless situations are prevented, e.g., trying to mix >100% of a layer). The homogeneous mixing model is simply:

$$b_{ij} = \begin{cases} 0 & z_{ij} > Z_{mix} \\ 1 & z_{ij} \leq Z_{mix} \end{cases} \quad (6)$$

Note that the USC model has a bioturbation function equivalent to “Let the whole layer mix with other layers – unless the layer is more than 15 cm deep, in which case don’t let it mix with other layers”.

If the extent of mixing is expected to decay exponentially with depth, then:
\[ b_{ij} = \beta_1 e^{-\beta_2 z_{ij}} \]  

(7)

In equation (7) the bioturbation function is equivalent to “Let the fraction of each layer which is allowed to mix with other layers decrease more or less quickly with depth, in an exponential fashion; even the top layer doesn’t have to be completely mixed.” It would be preferable to make the constants \( \beta_{1,2} \) dependent on measurables such as sediment type and tidal exposure.

Equation (7) can be reduced to be a function of a single parameter if it is also assumed that the surface is completely mixed:

\[ b_{ij} = e^{-\beta (z_{ij} - z_{i})} \]  

(8)

Compaction effects may require \( z_{ij} \) to be determined iteratively, too. For example:

\[ l_i = \frac{g}{AP} s_i \]  

(9)

\[ P = \rho_{\infty} - \rho_0 e^{-\alpha z_{ij}} \]  

(10)

where \( \rho_{\infty} \) is the density of sediment at some depth \( Z \gg z_1, \rho_{\infty} - \rho_0 \) is the density of surficial sediment, and \( \alpha \) is a suitable compression coefficient. A linear convergence on \( \rho_{\infty} \) is equally easy to set up, though \( Z \) then has to be defined.

If a contaminant profile for the sediment is available, then the goodness-of-fit of \( B \) can be determined. Let \( X \) be the set of measured contaminant concentrations \( x_{z_1}, x_{z_2}, \ldots, x_{z_n} \) at known depths or over known depth intervals. Let \( P \) be a polynomial of order \( q \) with coefficients \( p_1, p_2, \ldots, p_q \) such that:

\[ \sum (P(z) - x_z)^2 \]  

is minimised. Then, the observed value \( \chi_{ij} \) of \( c_{ij} \) is:

\[ \chi_{ij} = \int_{z_{ij-1}}^{z_{ij}} P \, dz \]  

(12)

Theoretically, \( B \) could be optimised by minimising:

\[ \sum (\chi_{ij} - c_{ij})^2 \]  

(13)

with respect to its parameters \( \beta_1, \beta_2, \ldots \) but this is probably not justifiable for any complex rule \( B \) given the number of approximations involved in generating \( c_{ij} \). It seems quite feasible for an expression such as (8).

**TEST CASE**

We did not have enough information to derive heterogeneous mixing models for Auckland. Nonetheless, just as an illustration, and to set up the procedure (via an Excel spreadsheet), the USC model for zinc in the Pakuranga estuary has been
reformulated, and fitted to the profile from the original “Pakuranga core 1”. The initial condition was three 10 cm layers at background concentration, 35 mg/kg. Results for various settling zone sizes are presented below, together with some observational data, read off the graphs on p. 87 of Williamson et al. (1999). Note that the observed profile was actually considerably more variable than shown. Interestingly, the 2.5% SZ fit has only 36% mixing at the surface, dropping to 11% at 15 cm and 5% at 25 cm. The reformulation generally fits the observed values well, but overpredicts the surface concentrations.
BIOTURBATION MODELS THAT APPEAR IN THE LITERATURE

Dhakar and Burdige (1996):

\[ D_b = \frac{D_b^0}{(1 + e^{1000(4z - v^2)}) (1 + e^{-z})} \]

where \( D_b \) is the bioturbation coefficient in cm²kyr⁻¹, and \( z \) is the (positive downward) depth in cm.

Van Cappellen and Wang (1996):

\( D_b = 20 \text{ cm}²\text{yr}⁻¹ \)

Gradually decreasing models did not yield a better result.

Boudreau (1994):

\( D_b = 15.7 \omega^{0.6} \)

where \( \omega \) is the sedimentation rate in cm yr⁻¹. However, this model only accounted for 22% of the variation in a large dataset. Mixing depth was essentially independent of \( \omega \).

Wheatcroft et al. (1990):

\( D_b \propto L_k^{4.25} N_k \)

where \( L_k \) is the length, and \( N_k \) is the population density, of bioturbating deposit feeders of size class \( k \). A variety of mechanisms, in particular advective bioturbation plus horizontal diffusion, result in apparently diffusive vertical distributions.

Robbins (1982):

\( Z_{mix} = Z_{90} \)

the sediment is well mixed down to the depth above which 90% of the zoobenthos can be found (oligochaetes in lacustrine sediments).

Lee and Swartz (1980):

\( D_b = L \nu \)

where \( \nu \) is the “apparent sedimentation rate” or “reworking rate”, with units of length per time.

Katz (1980):

Fiddler crabs \textit{Uca pugnax} turn over approximately 20% of the top 15 cm of salt marsh sediment, based on a density of 42 m⁻³ working 12 weeks per year.

Niedoroda et al. (1996):
Complete mixing down to $Z_{\text{mix}} = 4 - 6$ cm, superseded by $D_b$ deconvoluted from difference between model and observed profiles.

**Christensen and Bhunia (1986):**

$$D = D_0 e^{-z^2 / 2Z_{\text{mix}}^2}$$

with $D_0 = 2 \text{ cm}^2 \text{yr}^{-1}$; implicitly, $D$ is dominated by $D_b$.

**Swift et al. (1996):**

$$D_b = \sum_i \bar{A}_i N_i v_i$$

where $\bar{A}_i$ is the mean cross-sectional area of individuals of species $i$, and $v_i$ is the estimated velocity of sediment displacement for that species.

**Abril et al. (1992):**

Fraction $0 < g \leq 1$ of incoming material is mixed evenly within $Z_{\text{mix}}$; fraction $1 - g$ is never involved in mixing.

**Lent et al. (1992):**

$Z_{\text{mix}} = 3 - 4$ cm (supratidal estuarine sediment).

**Christensen and Goetz (1987):**

$$S_i = \sum_{j=1}^n E_{ij} F_{n+1-j}$$

where $S_i$ is the actual concentration of the analyte in the $i^{th}$ depth interval, $j$ is a time interval, $E$ is a matrix of elementary contributions to the depth profile for unit influx, and $F$ is the historical input flux. $E$ includes compaction, diffusion, and sedimentation terms:

$$\rho = \rho_0 - \rho_1 e^{-\omega z}$$

$$D = D_0 e^{-z^2 / 2Z_{\text{mix}}^2}$$

and $\omega$ is constant. This paper calculated $F$ from known $S$ and $E$, but of course the equation will work for any one unknown.

**Matisoff (1982):**

Diffusion models

Solutions to the diffusion equation

$$\frac{\partial C}{\partial t} = D_b \frac{\partial^3 C}{\partial z^3} - \omega \frac{\partial C}{\partial z}$$
where $D_b$ is constant for $0 \leq z \leq Z_{mix}$, or takes different values in different mixing zones, or $\propto z^{-1}$, $e^{\frac{z}{z_{200}}}$, $e^{-\frac{z^2}{2z_{200}^2}}$, etc. $D_b$ should be proportional to $N$ but this is rarely given in the literature.

**Box models**

Assume some degree of mixing down to $Z_{mix}$, referred to as heterogeneous if the rate of mixing is slow compared to the rate of sedimentation and homogeneous (complete mixing) otherwise. Heterogeneous models may also allow mixing to decrease with depth.

**Signal theory based model**

“Successful deconvolution… remains to be demonstrated”.

**Markov model**

Requires generation of probabilities which seem realistic, but hard to come by.

**MODEL USED IN THIS REPORT**

More mechanistically, assume two processes at steady state, infill of burrows by settling sediment and excavation of new burrows of the same dimension. Let the depth of crab burrows be independent of burrow diameter, and log-normally distributed with parameters $\mu$, $\sigma$. Now $c_{ij}$ is generated via:

$$c_{ij} = \begin{cases} c_{i,j-1} + \left( \frac{m_j}{s_j} - c_{i,j-1} \right) \sum_{h=1}^{j+1} b_{hj} & i < j \\ \frac{m_i}{s_i} + \sum_{h=1}^{i-1} \left( \frac{m_j}{s_j} - c_{h,j-1} \right) \sum_{h=1}^{i} b_{hj} & i = j \end{cases}$$

$$b_{ij} = \beta f_{\text{norm}}(\ln z_{ij}, \mu, \sigma)$$

where $\beta$ is such that:

$$\sum_{h=1}^{j-1} b_{hj} \ll 1$$

because the model does not allow for exchange between two layers $i,j \leq n$, an assumption which ought to break down if a lot of material from lower layers is brought to the surface.
APPENDIX 2. CONCENTRATIONS OF CONTAMINANTS IN SANDY SEDIMENTS

In this appendix we review the information on the level of contamination found in sandy sediments. There is in fact relatively little information. Some sediment quality surveys were carried out in 1996 and 1997 by the ARC that included sites at the overlapping mud/sand region on either side of Meola (Te Tokaroa) Reef. These sites were shown to be relatively contaminated (Table A2.1), but not as contaminated as further upstream at sites examined in the Long Term Monitoring Programme.

Table A2.1 Metal concentrations in 2 M HCl extractions from total sediment samples.

<table>
<thead>
<tr>
<th>Site</th>
<th>June-97</th>
<th>May-96</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Zn</td>
<td>Cu</td>
</tr>
<tr>
<td>Motion</td>
<td>131</td>
<td>8</td>
</tr>
<tr>
<td>Hobson</td>
<td>93</td>
<td>6</td>
</tr>
<tr>
<td>Meola</td>
<td>258</td>
<td>13</td>
</tr>
<tr>
<td>Coxs</td>
<td>188</td>
<td>19</td>
</tr>
<tr>
<td>St Marys</td>
<td>56</td>
<td>15</td>
</tr>
</tbody>
</table>

The survey conducted in 1997 at Motions, Meola, Coxes, Hobson, St Marys Bay and Herald Island examined the effect of particle size. Quite a marked effect is observed (Table A2.2).

Table A2.2 Metal concentrations in 2 M HCl extraction from different sediment size fractions.

<table>
<thead>
<tr>
<th>ID</th>
<th>Rep</th>
<th>Fraction µm</th>
<th>% of Total Sediment</th>
<th>TOC %</th>
<th>Fe mg/g</th>
<th>Cu µg/g</th>
<th>Pb µg/g</th>
<th>Zn µg/g</th>
</tr>
</thead>
<tbody>
<tr>
<td>Motions</td>
<td>1</td>
<td>&lt;63</td>
<td>22.5</td>
<td>5.59</td>
<td>11.0</td>
<td>27.6</td>
<td>94.5</td>
<td>224.4</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>&lt;63</td>
<td>26.2</td>
<td>5.69</td>
<td>10.5</td>
<td>24.9</td>
<td>82.9</td>
<td>193.5</td>
</tr>
<tr>
<td></td>
<td>1</td>
<td>63-125</td>
<td>36.1</td>
<td>0.46</td>
<td>2.7</td>
<td>2.6</td>
<td>18.6</td>
<td>101.8</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>63-125</td>
<td>42.4</td>
<td>0.39</td>
<td>2.7</td>
<td>2.6</td>
<td>15.7</td>
<td>116.5</td>
</tr>
<tr>
<td></td>
<td>1</td>
<td>125-250</td>
<td>31.5</td>
<td>0.38</td>
<td>2.9</td>
<td>2.0</td>
<td>16.8</td>
<td>118.5</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>125-250</td>
<td>26.2</td>
<td>0.57</td>
<td>2.6</td>
<td>2.0</td>
<td>14.8</td>
<td>117.0</td>
</tr>
<tr>
<td></td>
<td>1</td>
<td>250-500</td>
<td>1.7</td>
<td>0.79</td>
<td>3.6</td>
<td>1.6</td>
<td>13.7</td>
<td>93.7</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>250-500</td>
<td>2.1</td>
<td>0.52</td>
<td>2.5</td>
<td>1.7</td>
<td>12.0</td>
<td>75.4</td>
</tr>
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<td>1</td>
<td>&gt;500</td>
<td>8.2</td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>&gt;500</td>
<td>3.2</td>
<td></td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Hobson</td>
<td>1</td>
<td>&lt;63</td>
<td>32.5</td>
<td>3.98</td>
<td>12.6</td>
<td>14.2</td>
<td>67.2</td>
<td>114.2</td>
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<td>&lt;63</td>
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<td>14.2</td>
<td>63.3</td>
<td>121.1</td>
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<td>63-125</td>
<td>41.0</td>
<td>0.58</td>
<td>5.7</td>
<td>2.3</td>
<td>27.2</td>
<td>89.7</td>
</tr>
<tr>
<td></td>
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<td>63-125</td>
<td>51.8</td>
<td>0.60</td>
<td>5.3</td>
<td>2.3</td>
<td>26.3</td>
<td>87.5</td>
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<td>125-250</td>
<td>26.1</td>
<td>0.58</td>
<td>4.2</td>
<td>1.8</td>
<td>26.6</td>
<td>69.0</td>
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<td>125-250</td>
<td>16.2</td>
<td>0.74</td>
<td>3.8</td>
<td>1.8</td>
<td>21.8</td>
<td>64.6</td>
</tr>
<tr>
<td></td>
<td>1</td>
<td>250-500</td>
<td>0.2</td>
<td>*</td>
<td>*</td>
<td>*</td>
<td>*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>250-500</td>
<td>0.2</td>
<td>*</td>
<td>*</td>
<td>*</td>
<td>*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1</td>
<td>&gt;500</td>
<td>0.1</td>
<td></td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>&gt;500</td>
<td>0.1</td>
<td></td>
<td></td>
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<td></td>
</tr>
</tbody>
</table>

Table 4.2 continued...
In this present study, we wished to test whether the relatively high concentrations observed in the mud fraction extends over a SRA. We did this by examining the level of contaminant Zn in the mud fraction over the intertidal area which would be subject to deposition during storm events in Motions Creek. Samples were collected between the foreshore and the furthest extent of the intertidal area (which stretches to nearly the end of Te Tokaroa Reef). The outermost sites were about 50 m past the ARC Long Term Monitoring site. The survey was also carried out as a prelude to any more comprehensive programme to test the model.
The results are summarized in Fig. A2.1. While total metal concentrations vary throughout the intertidal area, the concentration in the mud fraction is far less variable. For comparison, note that in the Long Term Monitoring Programme, Zn concentrations at relatively pristine sites close to Auckland are about 45 µg/g – so we assume that this is the background for the Waitemata Harbour.

Fig. A2.1 Concentration of Zn in the 2M HCL extract from total and <63µm fraction of surface sediments (0-1 cm deep).
SUMMARY

The data available on contamination levels in sandy estuaries suggest:

- Contamination of the sandy sediments is generally low, because of the low level of contamination in the sand fraction.
- Muddy sediments in the Secondary Redistribution Area (SRA) can be highly contaminated.
- The level of contamination in the mud fraction is similar across an SRA, and seems to be independent of the degree of muddiness of the sediment.