# Preliminary Assessment of Limits and Guidelines Available for Classifying Auckland Coastal Waters

April 2017 Technical Report 2017/035

Auckland Council
Te Kaunihera o Tāmaki Makaurau

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Cover photograph of Pine Harbour Marina, Beachlands Auckland, by Jay Farnworth

# Preliminary Assessment of Limits and Guidelines Available for Classifying Auckland Coastal Waters

R B Williamson
Diffuse Sources

C W Hickey
National Institute of Water and Atmospheric Research, NIWA

B M Robertson
Wriggle Coastal Management

# Executive summary

## Report purpose

It is now over 10 years since the Environmental Response Criteria were formulated for the Auckland Regional Plan: Coastal, and these have had very mixed usage. Over the intervening period, there have been many advances in the underlying science, the application of guidelines, as well as in guidelines themselves. Auckland Council continues to evaluate the full range of triggers, criteria, standards, targets, objectives (all given the generic name “guideline”) that are available, and that are sufficiently robust and appropriate to the local environment, so that they could be incorporated into plan considerations as needed.

Up until recently, New Zealand waters were largely managed using an effects-based approach. There were few national standards and guidelines. However the release of national policy statements associated with coastal and freshwater environments are changing this management approach. The New Zealand Coastal Policy Statement 2010 (NZCPS) and National Policy Statement for Freshwater Management 2014 (NPSFM) direct water quality and discharge policy towards a limits based approach, the NPSFM directly and the NZCPS as an option. Enhancement and restoration of water quality is also strongly emphasised in both policy statements. They also imply an expectation for some form of classification framework for water where appropriate limits can be applied that aim to maintain existing uses, aim to maintain or improve water quality, identify where water has “deteriorated” (NZCPS), and improve water quality that has become “degraded” (NPSFM). A similar approach to the limits-based approach for fresh water could be applied to coastal waters. The approach proposes:

1. Numerical objectives for different water bodies, and failing that, tight narrative objectives. National standards would be derived where appropriate as environmental “bottom lines” and regional councils would need to adjust to these according to desired community outcomes and local conditions.
2. Several levels of protection (fair, moderate, excellent) – following the Ministry for the Environment (MfE) / Ministry of Health classification of waters for recreation, and similar to the ANZECC (2000) levels of pristine, slightly modified, and highly modified.
3. Limits on catchment water takes or discharges that are specific to each catchment.

**Approach taken**

In this report, we summarise physico-chemical guidelines that would be useful in managing Auckland's coastal zone. Such guidelines are potential candidates for the numerical objectives envisaged by the limits-based approach. This information will contribute to and assist the broader process and further work required to assess and apply a limits-based approach to coastal waters. A succinct summary of the indicators and their guidelines and where guideline development is needed, is given in Table 1. In the table we identify:

- Issues
- Indicators
- Whether general guidelines are available for that indicator(s)
- The type of guideline and/or its derivation
- Where in the marine receiving waters there needs to be further development [All (marine waters), marine (= open coastal water), estuaries, upper estuaries)]
- Development needs or strategies
- Whether numerical objectives can be derived
- Where numerical objectives could be translated into a catchment and discharge limits approach
- The relative importance of the indicator in managing Auckland's coastal zone.

**Conclusions following guidelines and indicator assessment**

Overall, we **recommend the use of the ANZECC Guidelines approach where possible**. However, we point out that the uncritical use of the numerical trigger values that are listed in the Guidelines is actually in contradiction to the ANZECC Guidelines approach. **Some numerical values are not appropriate to Auckland, and, consistent with the ANZECC Guideline approach, local numerical values have to be derived for some parameters**. In some cases and water bodies, this could be achieved with existing data (e.g., nutrients in the water column of open coastal and open estuary waters); for others, data will need to be collected (e.g., clarity in estuaries).

The ANZECC guidelines are undergoing revision, and where possible we have indicated any likely changes and tried to factor these into the review as far as we could.

Some other physico-chemical guidelines relevant for managing Auckland estuaries are not found in the ANZECC guidelines. Many of these have been developed in New Zealand including Auckland, and can be used directly or need development and ratification for Auckland (e.g., sediment deposition rates, nutrient loads).

We have recommended a different approach for guidelines for priority pollutants in sediments. We recommend a Weight of Evidence (WOE) approach promulgated by the (yet unpublished) ANZECC guidelines revisions. Underpinning the WOE are two approaches to developing guidelines: one approach would use international SQG, but using more conservative TEL/PEL for heavy metals rather than the ERL/ERM used in the ANZECC guidelines, and also inclusion of Auckland data into the database that derives TEL/PEL values. The other approach is using the Benthic Health Model. Both approaches require development of guidelines for differing levels of disturbance/protection.

Guidelines have not yet been recommended from most bioassay methods because they are still under development (biomarkers, emerging contaminants) or unsuitable for the majority of Auckland conditions (sediment toxicity testing). Exceptions are whole effluent toxicity testing of point sources and bioaccumulation of some primary pollutants.

Table 1 – Summary of issues, indicators, guidelines in managing the coastal zone, whether numerical objectives and limits can be derived and importance to Auckland

| **Issue** | **Indicator** | **Guidelines available** | **Based on** | **Guidelines needed for water bodies?** | **Development need or strategy** | **Numerical objectives** | **Limits approach** | **Importance** | **Need to consider upstream freshwater objectives** |
|---|---|---|---|---|---|---|---|---|---|
| Dissolved oxygen | DO | Yes | Reference conditions | Upper estuaries | Measure reference conditions | Yes | Challenging | Moderate | Yes |
| Salinity | Salinity | n/a | Matrix measurement | | | | | High | N/A |
| pH | pH | Yes | Reference conditions | | | Yes | Yes | Low | No |
| Clarity | Clarity | Open coastal only | MfE guideline | Estuaries | Measure reference conditions | Site specific | Challenging | High | Yes |
| | Turbidity | None | Reference conditions | Estuaries | Measure reference conditions | Yes | Challenging | Clarity preferred | Yes |
| Fine sediment deposition | SDR | Interim | Experimental observations | Estuaries | Workshop to choose values | Yes | Yes | High | Yes |
| | Muddiness | Interim | Expert condition rating | Estuaries | Summarise Auckland data, surveys, develop condition rating | Yes | Yes | High | Yes |

| Issue | Indicator | Guidelines available | Based on | Guidelines needed for water bodies? | Development need or strategy | Numerical objectives | Limits approach | Importance | Need to consider upstream freshwater objectives |
|---|---|---|---|---|---|---|---|---|---|
| Eutrophication | Macrophytes[1] | Interim, approximate | Expert condition rating | Estuaries | Summarise Auckland data, surveys, develop condition rating | Yes | No | High | Yes |
| | Chlorophyll | Interim | Overseas reference conditions | All | Measure reference conditions | Yes | Challenging | Moderate | No |
| | N and P in water | None | Reference conditions | All | Measure reference conditions | Yes | Challenging | Moderate | Yes |
| | $SiO_2$ | n/a | Matrix measurement | | | | | Moderate | N/A |
| | N and P in sediments | Interim | Reference conditions | All | Measure reference conditions | Yes | Yes | Moderate | Yes |
| | N and P loads | Interim | Experimental observations | All | Summarise Auckland data, surveys, develop condition rating | Yes | Yes | High | Yes |
| | Trophic indicators | Overseas - open water | Overseas studies | All | Research in Auckland | No | No | Watch | No |

[1] While this is not a physico-chemical measurement it is included here because it is probably the most important indicator for nutrient enrichment

| **Issue** | **Indicator** | **Guidelines available** | **Based on** | **Guidelines needed for water bodies?** | **Development need or strategy** | **Numerical objectives** | **Limits approach** | **Importance** | **Need to consider upstream freshwater objectives** |
|---|---|---|---|---|---|---|---|---|---|
| Toxicity | Zn, Cu in water | Yes | Experimental observations | | Background | Yes | Challenging | Moderate-High | Yes |
| | Ammonia | Yes | Experimental observations | | | Yes | Yes | Low | Yes |
| Human health | Enterococci | Yes | Epidemological surveys | | | Yes | Challenging | High | Yes |
| | *E. coli*, f. coli | Yes | | | | Yes | Challenging | Low | Yes |
| | Faecal Coliforms (shellfish) | Yes | | | | Yes | Challenging | High | No |
| | QMRA | Yes | | | Norovirus | Yes | Yes | High WWTP | No |
| | Genotype | No | Under international development | | | | | Watch | No |

| Issue | Indicator | Guidelines available | Based on | Guidelines needed for water bodies? | Development need or strategy | Numerical objectives | Limits approach | Importance | Need to consider upstream freshwater objectives |
|---|---|---|---|---|---|---|---|---|---|
| Sediment toxicity priority pollutants | Cu, Pb, Zn, DDT, PAH, TPH, As, Hg, Cd in sediments | Yes | International SQG | All | Criteria for differing levels of protection, background | Yes | Yes | High | Not investigated |
| | BHM and Cu, Pb, Zn in sediments | Yes | Experimental observations | All | Criteria for differing levels of protection, background | Yes | Yes | High | Not investigated |
| | Toxicity responses | Yes | Experimental observations | Not recommended | | Yes | No | Medium WOE[2] (cause/effect linkages) | Not investigated |
| | Biomarkers | No | Under international development | | | | | Watch | Not investigated |
| | Pore water (including ammoniacal-N) | Yes | WQG | May need development of pore water ammoniacal-N guidelines | Robust sampling methods | No | No | Medium WOE[2] (cause/effect linkages) | Not investigated |
| Sediment toxicity emerging contaminants | Biomarker responses | No | Under international development | | | | | Watch | Not investigated |

[2] WOE = weight of evidence (ANZECC revisions)

| Issue | Indicator | Guidelines available | Based on | Guidelines needed for water bodies? | Development need or strategy | Numerical objectives | Limits approach | Importance | Need to consider upstream freshwater objectives |
|---|---|---|---|---|---|---|---|---|---|
| Sediment characteristics | TOC | Interim | Expert condition rating | All | Measure reference conditions | Yes | Challenging | Low guideline<br>High matrix measurement | Not investigated |
| | RPD | Interim | Expert condition rating | All | Measure reference conditions | Yes, possibly | No | Low | Not investigated |
| | AVS/SEM | Yes | Experimental observations, sediment modeling | Not recommended | Summarise Auckland data, surveys, develop condition rating | No | No | Low guideline<br>High WoE[2] | Not investigated |
| Bioaccumulation of priority pollutants | As, Cd, Pb, PCB, DDT, dioxin | Yes - human health | Epidemological surveys, risk models | | Food basket approach | Yes | No | Moderate | Not investigated |
| | Concentrations in indicator organisms | No - ecological health | Experimental observations | Under international development | | | | Watch | Not investigated |

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# 1.0 Background

## 1.1 General introduction

It is now over 10 years since the Environmental Response Criteria were formulated for the Auckland Regional Plan: Coastal, and these criteria have had very mixed usage. Since their development, there have been many advances in the underlying science, the application of guidelines, as well as the guidelines themselves. Auckland Council (AC) have been reviewing their plans and require an improved understanding of the full range of triggers, criteria, standards, targets, objectives (all given the generic name “guideline”) that are now available to environmental managers, and that are sufficiently robust and appropriate to the local environment, so that they could be incorporated into their plan considerations as needed.

This report summarises guidelines suitable for managing the coastal zone in Auckland. It summarises and describes guidelines for individual parameters under the following general Parameter Grouping headings:

- Water Quality
  - general water quality
  - nutrient enrichment
  - toxicity
  - human health
- Sediment Quality (including the bioassays biomarkers and sediment toxicity)
  - toxic substances
  - sediment characteristics
- Biomagnification and bioaccumulation

It also briefly summarises the state of development of promising indicators (generally termed biomarkers), for which there are yet no guidelines that can be recommended to the council.

Finally, it briefly summarises some integrated approaches to assessing coastal water conditions from the USA, the European Union and elsewhere in New Zealand for two reasons. Firstly these include recommended methods and guidelines for specific parameters, and secondly, they consider other parameters for assessing coastal condition, which may be of interest to Auckland Council in the future.

Individual or groups of similar parameters e.g., dissolved oxygen, are assessed by considering:

- A brief overview of environmental issues associated with that parameter.
- Existing guidelines are summarised from Schedule 3 of the RMA, ANZECC (2000) guidelines and likely revisions, and the ARC Environmental Response Criteria (ARC 2002).
- How the parameter is applied or what it is used to represent and manage.
- A critique and review of existing guideline approaches, and any promising emerging alternatives, indices or combinations. This can include other approaches used in NZ, some overseas approaches (e.g., the USA National; Coastal Condition Assessment), and new developments from Auckland Council reviews and the international literature. The latter source is largely confined to literature of known relevance to Auckland only, because it would have been an enormous task to do justice to the entire international literature now available.
- The relevance and suitability of the parameter to Auckland, including different areas, such as the open coast versus estuary etc. Where existing criteria are not suitable, then the practicability of achieving something better is explored (e.g., developing Auckland specific criteria). If improvement is not practical, then the best available (off the shelf) or next best alternative is described.
- Conclusions include a summary of recommendations, and where possible, an assessment of robustness or risks associated with the use of the guidelines is made. Where possible, a summary table of guidelines by water classification type is given.

## 1.2 Report revisions

This current report, *Preliminary assessment of limits and guidelines available for classifying Auckland coastal waters* (hereafter called the 'Coastal Guideline Report') was prepared in 2011 and substantially revised in 2012. Since that time, a companion document *Technical aspects of integrating water quality science in freshwater and coastal environment*s has been prepared (Hickey et al. 2016) (hereafter called the ‘Freshwater Guideline Report’). That report summarises guidelines suitable for freshwaters, and additionally assesses the potential for conflicts to develop between fresh and coastal water management. With its completion, it became necessary to link the ‘Coastal Guideline Report’ to the “Freshwater Guideline Report”. The links are provided through incorporating a number of amendments and improvements, as follows:

1. Providing cross-links to the Freshwater Guideline Report and comment and crosslinks to conflicts in setting guidelines in fresh and coastal waters.
2. Improving and standardizing terminology (targets, objectives etc). Since the draft report was prepared in June 2012, terminology has moved on in the National Policy Statement for Freshwater Management 2014 arena. Historical and present-day guidelines use similar terms with different meanings.

For the process of integrating freshwater and coastal management, the reader is referred to the later “Freshwater Guideline Report”.

In addition, the report was updated, in particular:

1. Updating sediment toxicity and providing more definitive recommendations in the Sediment Toxicity section.
2. Updating numeric guidelines and references from publications that have appeared since the 2012 draft document.

# 2.0 Introduction to water classification/limits based approach

## 2.1 The classification/limits based approach

Up until recently, New Zealand waters were largely managed using a 'best practicable option (BPO)' approach within the RMA's 'effects based' framework. There were few national standards and guidelines. However, the release of new national policy statements associated with coastal and freshwater environments are changing this management approach. The New Zealand Coastal Policy Statement 2010 (NZCPS) and National Policy Statement for Freshwater Management 2014 (NPSFM) direct water quality and discharge policy towards a limits based approach, the NPSFM directly and the NZCPS as an option. Enhancement and restoration of water quality is also strongly emphasised in both policy statements. They also imply an expectation for some form of classification framework for water where appropriate limits can be applied that aim to maintain existing uses, aim to enhance or restore/maintain or improve water quality, identify where water has "deteriorated" (NZCPS) or improve water that has become "degraded" (NPSFM). A similar approach to the limits based approach for fresh water could be applied to coastal waters. The approach proposes:

1. Numerical objectives for different water bodies, and failing that, tight narrative objectives. National standards have been derived for some water quality attributes with environmental "bottom lines" and regional councils need to adjust to these according to desired community outcomes and local conditions.
2. Several levels of protection (fair, moderate, excellent – following the MfE / Ministry of Health classification of waters for recreation, and similar to the ANZECC (2000) levels of pristine, slightly modified, and highly modified[3].
3. Limits on catchment water takes (environmental flows and/or levels) or discharges, which are specific to each catchment[4].

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[3] Note that the NPSFM 2014 has adopted several levels of protection (A, B, and C) for each freshwater quality attribute.

[4] Note that limits now apply to freshwater management units (FMUs) under the NPSFM 2014.

## 2.2 Water classes

This report also describes the employment of water classes relevant to coastal water quality management, similar to those described in Schedule 3 of the RMA: Water quality classes, viz.,

1. Class AE Water (being water managed for aquatic ecosystem purposes)
2. Class F Water (being water managed for fishery purposes)
3. Class FS Water (being water managed for fish spawning purposes)
4. Class SG Water (being water managed for the gathering or cultivating of shellfish for human consumption)
5. Class CR Water (being water managed for contact recreation purposes)
6. Class WS Water (being water managed for water supply purposes)
7. Class I Water (being water managed for irrigation purposes)
8. Class IA Water (being water managed for industrial abstraction)
9. Class NS Water (being water managed in its natural state)
10. Class A Water (being water managed for aesthetic purposes)
11. Class C Water (being water managed for cultural purposes)

Classes 6 and 7 are not relevant to the coastal zone.

## 2.3 New Zealand Coastal Policy Statement (2010)

The New Zealand Coastal Policy statement recognizes that the coastal environment is facing a number of key issues, including the following that are of direct relevance to this report:

- the ability to manage activities in the coastal environment is hindered by a lack of understanding about some coastal processes and the effects of activities, such as rural residential development, on them;
- continuing decline in species, habitats and ecosystems in the coastal environment under pressures from subdivision and use, vegetation clearance, loss of intertidal areas, plant and animal pests, poor water quality, and sedimentation in estuaries and the coastal marine area;
- demand for coastal sites for infrastructure uses (including energy generation) and for aquaculture to meet the economic, social and cultural needs of people and communities;

- poor and declining coastal water quality in many areas as a consequence of point and diffuse sources of contamination, including stormwater and wastewater discharges;
- adverse effects of poor water quality on aquatic life and opportunities for aquaculture, mahinga kai gathering and recreational uses such as swimming and kayaking.

Additional policies to address some of these issues include Enhancement of water quality, Sedimentation, and Discharges of contaminants (see Appendix A).

## 2.4 General approach to choosing guidelines

We recommend the use of the ANZECC Guidelines approach for the Auckland marine environment where possible. It is consistent with the approach required by the NPSFM for numerical objectives for different levels of protection/different levels of disturbance. The ANZECC Guidelines approach utilises ecosystem classification, specifies levels of protection and specifies numerical guidelines as trigger values, as follows.

**Classification**. The primary (first and foremost) principle in the ANZECC guidelines is to decide how specific sites are to be managed by defining primary management aims. This leads to the guideline procedures for classifying ecosystems as:

1. pristine and undisturbed (Condition 1)
2. slightly to moderately disturbed (Condition 2)
3. highly disturbed (Condition 3)

**Levels of protection**. For conditions 2 and 3, the guidelines ensure that these are adequately protected, and it is necessary to decide the levels of protection for these conditions (e.g., 80%, 95%, 99%) in the application of the water quality guidelines. For ecosystems requiring the highest protection, Condition 1, the objective of water quality management is to ensure that there is no detectable change (beyond natural variability) in the levels of stressors. As described earlier, L&WF (2012) proposes fair, moderate, excellent levels of protection.

**Trigger Values**. The ANZECC (2000) guidelines are termed ‘trigger values’ (TVs) below which there is a low risk that adverse biological effects will occur. The physical and chemical trigger values are not designed to be used as “magic numbers" or threshold values at which an environmental problem is inferred if they are exceeded. Rather they are designed to be used in conjunction with professional judgement, to provide an initial assessment of the state of the water body regarding the issue in question. They are the values that trigger two possible responses. The first response, to continue monitoring, occurs if the test site value is less than the trigger value, showing there is a low risk that a problem exists. The alternative response, managerial/remedial action or further site-specific investigations, occurs if the trigger value is exceeded – i.e., a potential risk exists. The aim with further site-specific investigations is to determine whether there is an actual problem.

Four sources of information are available for use when deriving low-risk trigger values: biological and ecological effects data, reference system data, predictive modelling and professional judgement. Ideally, the methods used to develop management bands for lower protection levels require the analysis of environmental gradients in order to establish thresholds linked to ecological or other effects (e.g., aesthetic, recreational, mahinga kai). This information is not available for most cases in Auckland coastal areas.

**Cautionary Statements in the use of ANZECC Guidelines**. While we recommend the ANZECC approach, it is critical the user is aware of a number of issues and limitations.

1. The guidelines are trigger values for further work, not standards or attributes states, unless chosen to be so (see above).
2. Default reference values for physico-chemical parameters are from SE Australia and are probably not appropriate to Auckland (section 3.1) because they do not represent thresholds for adverse effects. **Locally derived reference values are needed.**
3. The ANZECC TVs for sediment Cu and Zn are high compared with concentrations associated with benthic health effects in Auckland marine sediments (Chapter 7).

## 2.5 Differences between ANZECC (2000) guidelines and the NPSFM

The NPSFM requires councils to set freshwater objectives and limits in their regional plans. To assist councils in setting freshwater objectives, the NPSFM has developed a framework and process to guide freshwater objectives called the National Objectives Framework (NOF). The NOF provides councils with water quality attribute tables containing a range of numeric and narrative states and a process for setting freshwater objectives. Leading scientists from across New Zealand have been involved in developing and testing national bottom lines for water quality attributes in the NOF, so that it is provided once in the NPSFM to prevent unnecessary cost and duplication. The NOF provides a framework for choosing values and uses that protect the freshwater environment while allowing allocation of water and its ability to absorb what is discharged into it.

There are some important differences between the guidelines and methods recommended by ANZECC (2000) and the National Objectives Framework (MfE 2014). These are summarised below.

**Table 2.1** Comparing ANZECC (2000) and NOF (MfE 2014)

| | **National Objectives Framework** | **ANZECC (2000)** |
|---|---|---|
| What sets focus for management | Values, uses identified by community<br>(e.g. ecosystem health, human health for (secondary contact) recreation) | Ecosystem and human health in marine waters (plus stock and plant health in freshwaters). |
| Intended environmental outcomes | Objectives: describes the intended environmental outcomes(s) / attribute state of A, B, or C (D indicates a state below the National Bottom Line) in narrative and numeric terms to achieve a freshwater value/s. | (For coastal waters) defining primary management aims, which leads to classifying ecosystems as:<br>1. pristine and undisturbed (Condition 1)<br>2. slightly to moderately disturbed (Condition 2)<br>3. highly disturbed (Condition 3) |
| What is monitored, measured | Attributes (measurable parameters) | Measured parameters, performance indicators, stressors |
| How is measurement | Against attribute state - narrative and/or numeric | Trigger Values (TV) for various levels pf protection (e.g., 80%, 90%, 95%) assigned to Conditions |

| | National Objectives Framework | ANZECC (2000) |
|---|---|---|
| assessed | grade (A to D)<br><br>Standard = bottom line = C/D boundary.<br><br>Indicate levels of protection for each attribute state. | 1, 2, 3 |
| Operation | Desired attribute state is pre-determined by water classification, scientific evaluation and community consultation. Out-of-state triggers management and remediation and/or extension of timeframes. | Potential problems identified if TV exceeded. Exceedance triggers management/remediation or further investigation to see if an actual problem exists. This may also trigger further refinement of TV. |
| Targets | Specifically ‘ is a limit which must be met at a defined time in the future (in context of over-allocation only)’ | General meaning – usually referring to a time frame or an aspiration. |
| Limits | Specifically means the maximum resource limits available, which allows a freshwater objective to be met. Can be expressed as m3/sec, m3/yr or tonnes/yr (for example). | General meaning, such as a maximum value. |
| Outcomes | Are the objectives of the NPSFM being met? Are freshwater objectives in freshwater management units being met? | Has the biophysical or microbiological state improved? Are objectives being met? Are guidelines/standards being met? |

In addition to the above, the NZCPS specifies the need to manage effects by way of numerical objectives, limits, targets in slightly different language, viz., “Identify in regional policy statements, and plans, coastal processes, resources or values that are under threat or at significant risk from adverse cumulative effects.

Include provisions in plans to manage these effects. Where practicable, in plans, set thresholds (including zones, standards or targets), or specify acceptable limits to change, to assist in determining when activities causing adverse cumulative effects are to be avoided".

In addition, a standard in water quality generally means 'an objective that is recognised in enforceable environmental control laws of a level of government'. In the NPSFM, **standard** has an additional meaning in that it is the "bottom line" and is equivalent to boundary between the C and D attribute states (MFE 2014a, b).

## 2.6 The Implications from the NPSFM to choosing coastal water guidelines

The National Policy Statement for Freshwater Management (NPSFM) establishes a legal and policy framework for building a national limits-based approach to freshwater management. There is no requirement to manage estuaries using the NPSFM process at present. Nonetheless, estuaries must be considered when deciding how to manage freshwater under the NPSFM. In particular, the process for setting contaminant load limits to achieve freshwater objectives must take into account aspirations for the use of estuaries and vice versa. There is also an argument that estuarine management will benefit from a limits-based management approach in the same way that freshwater is expected to benefit.

In a related report "Technical aspects of integrating water quality science in the freshwater and coastal environments" (Hickey et al. 2015) we identify limitations or opportunities for each water quality parameter to influence the ability to set effective and efficient management objectives (narrative), appropriate water quality objectives (numeric), or limits (loads) across both freshwater and coastal environments.

Providing general guidance recommendations on how to address and resolve areas of potential conflict in setting objectives between freshwater and estuarine environments is difficult. Downstream sensitivity will in some circumstances dictate the upstream management requirements (and vice versa), and is very dependent on site-specific circumstances in respect to sensitivity and hydrogeomorphology.

In the following chapters, specific reference is made where there are potential issues in setting objectives or limits, thus linking this report to the related report "Technical aspects of integrating water quality science in the freshwater and coastal environments".

Further work is needed to assess the proposition that a limits-based approach could be applied to coastal waters.

# 3.0 General water quality

## 3.1 ANZECC guideline approach

For physical and chemical stressors in this chapter, low-risk guideline values for key performance indicators have been determined by comparison with reference ecosystems. This is in contrast with other guidelines, which are based on cause/effect (e.g., toxicity). Ideally, thresholds for trigger values should be developed from actual studies of ecological effects, however, in the absence of such information, the 80th (or 20th) percentile for “slightly to moderately disturbed” ecosystems is suggested as a guide. This general approach used in the ANZECC (2000) guidelines to derive physico-chemical water quality benchmark concentrations would be applicable to the derivation of Auckland region-specific data and potential triggers for management bands.

*We point out that the uncritical use of the numerical trigger values that are listed in the Guidelines is actually in contradiction to the ANZECC Guidelines approach.* Some numerical values are not appropriate to Auckland, because the reference conditions are based on Australian ecosystems. So, consistent with the ANZECC Guideline approach, local numerical values need to be derived for some parameters, using site-specific reference conditions.

## 3.2 Dissolved oxygen

### 3.2.1 Overview/Issues

Dissolved oxygen (DO) is a key requirement for healthy aquatic systems. Appropriate guidelines are important to maintain DO at sufficiently high concentrations to prevent adverse effects on marine organisms and on marine sediments.

Estuaries can experience lower DO due to discharges of oxygen demanding substances (Biochemical Oxygen Demand or BOD) or due to algal blooms; these would be managed by controlling discharges and eutrophication. Estuaries may also experience lower DO due to benthic sediment oxygen demand. DO depletion can be exacerbated by sheltered conditions (and hence reduced wind mixing and re-aeration), and poor mixing (e.g., caused by density gradient/salt wedges).

### 3.2.2 Existing guidelines

Schedule 3 specifies dissolved oxygen conditions for Class AE, F, FS, and SG as “The concentration of dissolved oxygen shall exceed 80% of saturation concentration.” Dissolved oxygen criteria are implicit in Class IA, NS and C (Table 3.1).

**Table 3.1** - Standards for Water Quality Classes for Dissolved Oxygen from Schedule 3 RMA

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic ecosystem | The concentration of dissolved oxygen shall exceed 80% of saturation concentration |
| SG | Gathering or cultivating of shellfish | The concentration of dissolved oxygen shall exceed 80% of saturation concentration |
| CR | Contact recreation | - |
| IA | Industrial abstraction | - |
| NS | Natural state | The natural quality of the water shall not be altered |
| A | Aesthetic | - |
| C | Cultural purposes | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified cultural or spiritual values |

The ANZECC (2000) guidelines describe default trigger values (Table 3.2). The guidelines recommend that trigger values are developed from local monitoring data using ANZECC protocols (see Section 3.1 above).

**Table 3.2 -** ANZECC trigger guidelines for DO per cent saturation (their Table 3.3.2, page 3.3-10)

| Water bodies | Lower limit | Upper limit |
|---|---|---|
| Estuaries | 80 | 110 |
| Marine | 90 | 110 |

The Environmental Response Criteria (ERC; ARC 2002) specify that the local receiving environment average dissolved oxygen concentration for all samples except the bottom samples, is to remain above 80% saturation at all times. At any single sampling station, the depth averaged DO saturation will remain above 65% at all times, and the average bottom sample DO saturation in sediment deposition areas is to be above 65% at all times.

### 3.2.3 Use/applicability/importance

DO is a key water quality parameter for measuring the life supporting capacity of coastal waters and can be affected by many processes.

It is generally regarded as unlikely to be a major issue in most open coastal and estuarine waters (ARC 2002), but being easy and relatively cheap to measure, should be measured in most monitoring programmes. When monitoring results, observations or concerns raise water quality issues, it is very useful for managers to know the DO concentration, because it is affected by many processes, and will reflect those processes. In addition, it may be an issue in the headwaters of estuaries (see below) and bottom waters.

### 3.2.4 Critique/review of existing approaches

The ERC (ARC 2002) proposed a value of 65% saturation for bottom waters to allow for the commonly observed situation of lower bottom water DO concentrations in the headwaters of estuaries. However, hypoxia may still occur at this level with sensitive animals and life stages (Vaquer-Sunyer & Duarte 2008). The onset of hypoxia symptoms occurs over a large range of DO levels and does not only occur at low concentrations, with fish being the most sensitive. Levels of 4.6 mg/L DO (64% saturation at 20$^{o}$C) will protect all but the most sensitive 10% of fish. Consequently, the 65% saturation in bottom waters may not be protective to the most sensitive 10% of all species. The 80% saturation guideline would offer a high level of protection to most estuarine animals; however this must be maintained through the whole water column.

80% saturation may not be met in the headwaters of sheltered estuaries. For example, the upper reaches of the brackish water sections of the Helensville, Otamatea River, and Arapaoa River in the Kaipara Harbour frequently have low DO. (Diffuse Sources 2007, Northland Regional Council data reviewed by Haggitt et al. 2008). The implications for local biota are not understood.

ANZECC (2000, p 3-10) alludes to this problem in their Table 3.3.2 which lists the DO trigger values described above, “dissolved oxygen (trigger) values were derived from daytime measurements. Dissolved Oxygen may vary diurnally and with depth. Monitoring programmes should assess this potential variability” (and then refers readers to their monitoring methodology).

It is interesting that the ANZECC (2000) default trigger values for marine waters, (meaning open coast), are 90–110% saturation, which are more rigorous than the New Zealand RMA standards.

### 3.2.5 Relevance/suitability for Auckland

Many New Zealand surface coastal waters easily meet the Schedule 3 Standard of 80% saturation for Class AE and SG waters, and are often better than 90%. This is observed in state of the environment monitoring for Auckland (Scarsbrook 2008). However, state of the environment monitoring only samples in the main parts of estuaries (along with some parts of the open coast) near high tide.

It is known that 80% saturation is not being met in the upper reaches of some sheltered estuaries, and it is highly likely that this situation is found in the headwaters of many other Auckland estuaries that have not been monitored, especially in bottom waters (see Section 3.2.4). This may also be true for bottom waters in parts of the main body of estuaries; this is poorly understood at present with insufficient monitoring data available to draw definitive conclusions. While the implications for local biota are not well understood, many of the animals commonly found in upper estuaries are probably fairly robust and naturally adapted to cope with the variable DO that naturally occurs in these environments. Under the RMA Schedule 3, these waters should be managed to >80% saturation, which may not naturally occur at all times. At present, it is not known whether management could achieve this.

If the 80% standard was found not to be met (and it is argued above that this is likely the situation in the headwater of some estuaries), then management would need to respond. If the response is to improve DO conditions, then the role of mixing, hydrodynamics, sediment oxygen demand, as well as biochemical oxygen demand (BOD) and sediment oxygen demand (SOD) inputs in both fresh and saline waters would need to be understood. These processes are not well understood in Auckland estuaries. This will make a limits-based approach using 80% DO very challenging.

The NPSFM requires councils to set freshwater objectives and limits in their regional plans. A companion report (Hickey et al. 2015) assesses the potential for conflicts or cross-over issues arising from setting objectives in freshwaters and downstream coastal waters. For DO, setting objectives in the freshwater environment may affect the ability of estuaries to meet 80% saturation. The B and C states in NOF5 are close to or below 80% saturation. If this is taken into account, plus the possibility that DO in freshwater discharged to an estuary may be further reduced by SOD and BOD, it could potentially result in DO falling below 80% saturation, depending on mixing and temperature.

Understanding DO dynamics and cause-effect relationships at individual locations would be a complex and costly exercise, unless there is an obvious reason for the decrease such as a point source discharge. Development of trigger values for DO concentrations and saturations using ANZECC protocols to set trigger values for reference estuary headwaters associated with specific types or groups of estuaries (i.e., estuaries of a similar type) may be a better approach until sufficient data is acquired to support a more robust numeric limit.

The Class NS and C waters are special cases. It is highly likely that these classes would need to have relatively pristine quality, including life-supporting capacity. For Class C waters, probable requirements are healthy conditions for mahinga kai and taonga, and, possibly more important, the Mauri of the water. This should be met with the same conditions that support aquatic ecosystems. These are expected to be similar to DO levels that occur in natural waters.

**Levels of protection:** There is some information that would allow defining different DO levels of protection at different levels of disturbance, based on overseas studies described earlier.

### 3.2.6 Conclusions

Standards have been set for Class AE and SG Waters of 80% saturation in Schedule 3. This is widely met in Auckland coastal waters, in the open coast and main bodies of harbours near high tide during daylight hours. This standard is very robust, universally applied across the planet and will protect most, if not all, aquatic species. While strictly speaking, the ANZECC (2000) protocols recommend developing regional trigger levels, these are highly unlikely to differ from the default values in offshore and in the main bodies of estuaries.

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[5] In the NPSFM, measurable characteristics of fresh water, such as DO, which supports particular values, are called attributes. Four different attribute ‘states’ are specified for DO (A, B, C or D) in the National Objectives Framework (NOF).

**Table 3.3** - Recommended Dissolved Oxygen Criteria for Auckland Coastal Waters

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | Marine off-shore waters 90-110%<br>Estuaries and nearshore waters 80-110%<br>Estuary headwaters - develop trigger values under ANZECC protocols using monitoring results |
| SG | Gathering/cultivating shellfish | Same as Class AE |
| CR | Contact Recreation | - |
| IA | Industrial Abstraction | - |
| NS | Natural State | Same as Class AE |
| A | Aesthetic | - |
| C | Cultural | Same as class AE |

It is probable that 80% saturation may not be being met in the upper reaches of sheltered estuaries, especially in bottom waters and at low tide. In these instances, where water quality criteria are required, we recommend establishing trigger values for use in upper estuaries, using New Zealand data to establish reference conditions consistent with ANZECC (2000) protocols.

It will be challenging to develop a limits type approach for dissolved oxygen. Unlike rivers where highly successful BOD models have been developed, as described above oxygen concentrations in estuaries are much harder to predict. This may, of course, not become an issue that needs to be addressed. However, we can only be sure of that once DO concentrations throughout estuaries have been monitored at sufficient sites.

## 3.3 Salinity

Salinity is naturally highly variable among and within marine ecosystem types, and natural biological communities are adapted to the site-specific conditions. In the ARC Environmental Targets report (2002), salinity was not included because it was considered unlikely to be an issue in Auckland's coastal waters, even though salinity regimes have changed with land development (Hayward et al. 2006).

Salinity can have indirect effects on toxicity by affecting the speciation of other toxicants such as ammonia and heavy metals. For coastal regions, salinity varies from freshwater values through to oceanic values. It is important to measure salinity in the coastal region, because it provides data associated with the coastal water matrix and coastal processes such as freshwater inflows and wedges, saltwater wedges and the degree of freshwater and seawater mixing, as well as its effects on toxicity of other contaminants.

There are no existing guidelines for salinity in NZ, as far as we are aware. While salinity is important to measure as part of coastal water quality monitoring, we do not recommend that guideline values be developed.

## 3.4 pH

### 3.4.1 Overview

pH can be naturally highly variable for estuaries and near large freshwater discharges in the coastal area, and can vary from freshwater through to oceanic values. Biological communities are generally well adapted to these natural site-specific conditions which are reflected in their distribution patterns. The pH in estuaries and within estuary sediments can also vary due to estuarine processes such as algal blooms and sediment diagenetic changes and the amount of freshwater input. The pH of oceanic waters varies over a narrow range (typically 8.0 to 8.4, ANZECC 2000).

Global warming is also causing ocean acidification with the ongoing decrease in the pH of the Earth's oceans, caused by the uptake of $CO_2$ from the atmosphere. Seawater is slightly basic (meaning pH > 7), and the process in question is a shift towards pH-neutral conditions rather than a transition to acidic conditions (pH < 7). Between 1751 and 1996 surface ocean pH is estimated to have decreased from approximately 8.25 to 8.14. It is expected to drop by a further 0.3 to 0.5 pH units by 2100 as the oceans absorb more anthropogenic $CO_2$.

Changes in pH can affect aquatic life because too low or too high pH is toxic. pH can also have indirect effects on toxicity by affecting the speciation and bioavailability of other toxicants such as ammonia and heavy metals[6].

### 3.4.2 Existing guidelines

RMA Schedule 3 specifies pH Standards for Class AE waters, however pH guidelines are also implicit in the narrative of standards for Class CR, NS and C waters (Table 3.4).

**Table 3.4** - pH Standards for Water Quality Classes from Schedule 3 RMA

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | The following shall not be allowed if they have an adverse effect on aquatic life: any pH change |
| F | Fishery | |
| FS | Fish Spawning | |
| SG | Gathering/Cultivating Shellfish | |
| CR | Contact Recreation | The water shall not be rendered unsuitable for bathing by the presence of contaminants |
| IA | Industrial Abstraction | - |
| NS | Natural State | The natural quality of the water shall not be altered. |
| A | Aesthetic | - |
| C | Cultural | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified cultural or spiritual values |

[6] pH is explicitly used in calculating ammonia guidelines, while pH may be used to help interpret heavy metal toxicity.

It is debatable as to what constitutes a pH change. Oceanic waters are relatively highly buffered, so changes outside the normal range (8 to 8.4) represent a strong chemical signal. Where freshwater discharges to marine waters in estuaries, there can be large natural variability (e.g., from ~ pH 7 to about 8.2)

The ANZECC (2000) default trigger values are typically found across the freshwater – oceanic continuum. These values are regarded as robust, current and unlikely to change (Table 3.5) in current revisions.

**Table 3.5** - ANZECC default trigger guidelines for pH

| Water bodies | Lower limit | Upper limit |
|---|---|---|
| Estuaries | 7.0 | 8.5 |
| Marine | 8.0 | 8.4 |

pH criteria were not included in the ARC Environmental Response Criteria (2002), because pH was considered unlikely to be an issue in Auckland's coastal waters.

### 3.4.3 Use/applicability/importance

Monitoring results have suggested that pH toxicity per se is unlikely to be an issue in most situations in coastal waters. However, it may be still important to measure pH (and if necessary manage it) in the coastal regions, because it is an important measurement when assessing ammonia and heavy metal toxicity in coastal waters and the sediment pore water matrix.

### 3.4.4 Critique/review of existing approaches

The ANZECC criteria described in 3.4.2 are currently referred to for Auckland waters.

### 3.4.5 Relevance/suitability for Auckland

The narrative Standards in Schedule 3 require no changes in pH for Class AE if there is an adverse effect on aquatic life. This would also be expected to be the condition for Class C waters. However, Schedule 3 requires no changes in pH for Class NS waters (irrespective of adverse effects on aquatic life). For these Water Classes, pH should fall within the default ANZECC trigger values (Table 3.6), because these will offer general water quality protection (the values in Table 3.5

are typical of many guidelines and standards throughout the world). If ammonia or heavy metal toxicity needs to be assessed, the pH results would be used there. Strictly, ANZECC (2000) requires that trigger values be derived from local monitoring data of reference conditions, but this seems unnecessary, because local trigger values for pH are likely to be very similar to the ANZECC (2000) default trigger values.

Setting pH objectives in freshwaters that discharge to estuaries may be required under the NFS-FM. This probably will not influence the ability of estuarine waters to meet estuarine objectives (Hickey et al. 2015). Desirable pH values are similar in each water types and estuary waters are well buffered.

### 3.4.6 Conclusions

Schedule 3 requires monitoring of pH for Class AE, C and NS Waters. For Class AE, NS and C Waters, the results would be assessed using ANZECC (2000) trigger values and also be used for assessing any ammonia or heavy metal issues. The ANZECC (2000) default trigger values are typically found across the freshwater – oceanic continuum. These values are regarded as robust, current and unlikely to change.

Limits-based approaches can be developed with pH using mass balance of the cations, anions and organic acids that determine pH. However, this is only likely to occur in a few site-specific situations, if at all.

**Table 3.6 -** Recommendations for pH Criteria in Auckland Coastal Waters

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | ANZECC (2000) trigger values (Table 3.5) |
| SG | Gathering/Cultivating Shellfish | - |
| CR | Contact Recreation | - |
| IA | Industrial Abstraction | - |
| NS | Natural State | ANZECC (2000) trigger values (Table 3.5) |
| A | Aesthetic | - |
| C | Cultural | ANZECC (2000) trigger values (Table 3.5) |

**Levels of protection:** There is no information that we are aware of that would allow defining pH for different levels of protection at different levels of disturbance. However, this could probably be defined in a site-specific situation through monitoring and reference conditions, if needed.

## 3.5 Water clarity

### 3.5.1 Overview

Water clarity and underwater visibility is important for recreation such as swimming and boating. To allow good visibility for swimming, you should be able to see at least 1.6 metres underwater (MFE 1994). It is also important from an aesthetic point of view – most people prefer to see clear water in our estuaries and beaches. Clarity also impacts ecology in a number of ways. It directly affects visual acuity for both predators and prey. It directly impacts on food quality for filter and particle feeders via the concentration, physical, chemical and biological characteristics of the particles that determine clarity. At present there are no clear guidelines for clarity in terms of recreation or aquatic ecosystem protection in New Zealand freshwater and marine systems. Similarly there are some turbidity guidelines for freshwaters, but not for marine waters.

Clarity and turbidity are interchangeable; they can be calculated from one another for a specific water body. Turbidity is measured in Auckland Council monitoring programmes and can be indirectly compared with clarity criteria, once a turbidity/clarity relationship has been established for individual sites.

### 3.5.2 Existing guidelines

RMA Schedule 3 only specifies clarity for CR waters, but clarity criteria are implicit in Class NS, A and C Waters.

**Table 3.7** - Standards for Clarity (turbidity) for Water Quality Classes from Schedule 3 RMA

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | - |
| SG | Gathering/Cultivating Shellfish | - |
| CR | Contact Recreation | The visual clarity of the water shall not be so low as to be unsuitable for bathing |
| IA | Industrial Abstraction | - |
| NS | Natural State | The natural quality of the water shall not be altered. |
| A | Aesthetic | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified aesthetic values |
| C | Cultural | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified cultural or spiritual values |

MfE Guidelines are 1.6 m for clarity in freshwater (MfE 1994), which is appropriate for open coastal waters but not suitable for many estuaries that are naturally more turbid (MfE 1999).

ANZECC (2000) list default trigger values for turbidity in slightly disturbed marine waters and estuaries of 0.5 to 10 NTU, but acknowledge that these values are of little practical use. Values of 0.5 NTU are expected for off-shore waters, while values of up to 10 NTU can naturally occur in estuaries (ANZECC 2000; Table 3.3.3, page 3.3-11).

The Environmental Response Criteria (ERC) report (ARC 2002) was unable to define clarity guidelines. It concluded that observed clarity and turbidity is highly location and weather dependent, and the present clarity/turbidity criteria (MfE or ANZECC) is of doubtful relevance for shallow and muddy Auckland estuaries, which can become naturally quite turbid on windy days, irrespective of catchment discharges.

### 3.5.3 Use/applicability/importance

Clarity guidelines are currently not applied in the Auckland coastal environment. But clarity and turbidity are very important indicators of water quality in terms of recreation and ecology, and should be addressed and managed as best as possible.

Note that they are also useful indicators to help understand the fate and transport of sediments and any attached contaminants, and for interpreting other water quality measurements (e.g., Total Nitrogen and Phosphorus, bacterial pollution).

### 3.5.4 Critique/review of existing approaches

At present, there are no existing coastal clarity or turbidity criteria applied in Auckland. The MfE 1.6 m criteria for visual clarity is an appropriate clarity criterion for open coastal waters.

For recreation in freshwaters, clarity is generally measured in rivers during baseflow[7], because most people do not swim during high or flood flows, when waters almost invariably contain higher levels of suspended sediment (and hence low clarity). While fewer people recreate during rainstorms in estuaries or coastal areas as well, some recreation will occur (e.g., surfers, wind surfers and kite boarders who may be looking for conditions sometimes associated with rainstorms and higher winds).

In a river system, clarity is determined by erosion/runoff processes on the land, and erosion/resuspension/deposition processes in the water body (including the flood plain). These processes also determine the clarity in the immediate vicinity of coastal areas where freshwaters discharge. There will be some dilution and flocculation across the salinity gradient, so clarity should increase (turbidity decrease) with distance away from any freshwater discharge. However, in coastal regions, the impact of flood flows on water clarity may last much longer than the storm passage, because flushing may be slower (as the tide moves water back and forth compared with the linear flushing in a river), and particles may stay in suspension longer because of wave resuspension. The processes involved are complex because of the complexity of the freshwater/salt water interactions; e.g., variable mixing and the formation of freshwater lenses and saltwater wedges, and the effect of salt water-induced flocculation on site-

[7] The Auckland Regional Council stopped measuring clarity in freshwater around 2007/08 – Auckland Council only measures TSS and turbidity in the present time.

specific particle size distributions. For example, see Davies-Colley & Nagels (1995) study on Mahurangi Harbour, which describes these complexities in an Auckland estuary.

The situation is more complex in estuaries and sheltered embayments that have accumulated muddy sediments. In such coastal areas, wind-induced resuspension is an important process that determines clarity during fine weather. There may be sufficient stored mud to cause low clarity during wind events, depending on the wind fetch and direction. This contrasts to the open coast with small freshwater inflows, where the water clarity is generally good under most conditions because there is a low level of very fine sediments (mud) available for resuspension. This is because in the near shore of the open coast, energetic processes discourage mud from settling. Where deposition does occur, energetic processes resuspend and disperse any mud particles to depositional areas often associated with deeper waters further away from the coastal margin.

It will be very challenging to develop a limits-based approach for clarity by itself, based on catchment models, because of the complexity of the processes that describe particle behaviours in marine receiving environments described above.

Issues associated with setting clarity limits include:

1. some recreation in marine areas may take place during rain-runoff and storm events;
1. low clarity may persist longer in some coastal areas receiving freshwater runoff, and this is very site specific; and
2. particularly because recreation is likely to take place in coastal areas during fine windy days, when there may be wind induced resuspension of fine particles and hence turbid waters;

Clarity/turbidity criteria in coastal regions would need to be developed for different water bodies or on a more site-specific basis (e.g., open coast, open coast with significant freshwater discharges, sheltered harbours, estuaries etc.).

The complexity described above precludes a process-orientated approach for this development, because of the complexity and associated costs. The ANZECC (2000) protocols offer a way forward to define criteria. Site-specific information can be collected on the clarity of estuaries and sheltered embayments under different weather conditions. This data can then be used to define reference conditions. This approach has been successfully applied in the USA National

Coastal Assessment (see Chapter 11.1). Within estuaries in the USA, water clarity varies between the different regions within an estuary, as well as at a single location in an estuary due to tides, storm events, wind mixing, and changes in incident light. The probabilistic nature of their monitoring design accounts for this variability so that the results can be assessed on larger regional scales, such as for classifying waters and grading waters.

### 3.5.5 Relevance/suitability for Auckland

Recreational use of coastal regions around Auckland is huge in New Zealand terms because of the relatively large population, and the attractive, variable, accessible, user-friendly coastal areas. Managing the coastal environment and adjoining catchments for clarity would seem to be priority for these reasons.

While there is some existing turbidity data for the SoE sites, there is very little clarity data available. Consequently, there is no possibility of developing criteria in the immediate short term. A comprehensive monitoring programme would need to be undertaken, possibly through community participation. Many years' data would need to be accumulated over a wide range of sites (hence the attractiveness of site-specific community group participation).

Limits-based approaches based on numerical values will be very challenging, given the complexity of particle dynamics in estuaries, as described above. It may be possible and more expedient to couple a limits-based approach for clarity with one for total suspended solids (described in the next section).

Setting clarity objectives in freshwaters as required under the NPSFM will not directly affect the ability to set future objectives for clarity in any downstream coastal environment (Hickey et al. 2015). Management generally focuses on clarity for rivers during low flow. This will have little influence on estuaries where clarity is more often related to wave resuspension and flood flows. However, TSS loads and limits may be set for upstream rivers to manage riverine clarity, which will also affect delivery and deposition of muds in the estuary, and hence estuarine clarity.

**Levels of protection**: There is no information that we are aware of that would allow for defining clarity for different levels of protection at different levels of disturbance. However, this could probably be derived from monitoring programmes.

### 3.5.6 Conclusions

We recommend that a comprehensive monitoring programme would need to be carried out over many years and a wide range of sites in Auckland (and other New Zealand) coastal regions to support a limits-based approach using clarity measurement. The accumulating clarity and turbidity data would be assessed using ANZECC protocols to establish reference conditions to establish guidelines, such as trigger values, for use in Auckland's coastal waters. With appropriate design and monitoring, site-specific reference conditions should provide robust and defensible data for management. For open coast marine waters, the MfE guideline of 1.6 m should be used.

Any limits-based approach may be better explored for total suspended solids, which then could be used to develop associations (qualitatively, semi-quantitatively) with clarity.

**Table 3.8 -** Recommendations for clarity and turbidity in Auckland Coastal Waters

| Class | Purpose | Criteria for Marine (Open Coast) and Estuaries/sheltered embayments. |
|---|---|---|
| AE | Aquatic Ecosystem | Marine (open coastal waters): 1.6 m (MfE guideline)<br>Estuaries and sheltered embayments: ANZECC protocols to establish trigger values based on measurement of reference conditions. The monitoring design would be based on protocols established in the USA National Coastal Assessment. |
| SG | Gathering/Cultivating Shellfish | Same as Class AE |
| CR | Contact Recreation | Marine (open coastal waters): 1.6 m (MfE guideline)<br>For estuaries and on open coats with significant freshwater discharges, use procedures described in Class AE, and develop criteria with the aim that "visual clarity of the water shall not be so low as to be unsuitable for bathing" |
| IA | Industrial Abstraction | - |
| NS | Natural State | Marine (open coastal waters): 1.6 m (MfE guideline)<br>ANZECC protocols would be used as described in Class AE Waters, to develop trigger values for the natural state condition, so that the natural quality of the water is not altered. |
| A | Aesthetic | Marine (open coastal waters): 1.6 m (MfE guideline)<br>ANZECC protocols would be used as described in Class AE Waters, to develop the reference condition, so unacceptable aesthetic changes can be defined. |
| C | Cultural | Marine (open coastal waters): 1.6 m (MfE guideline)<br>ANZECC protocols would be used as described in Class AE Waters, to develop trigger values for the reference condition that meets cultural aspirations. |

## 3.6 Total suspended sediment

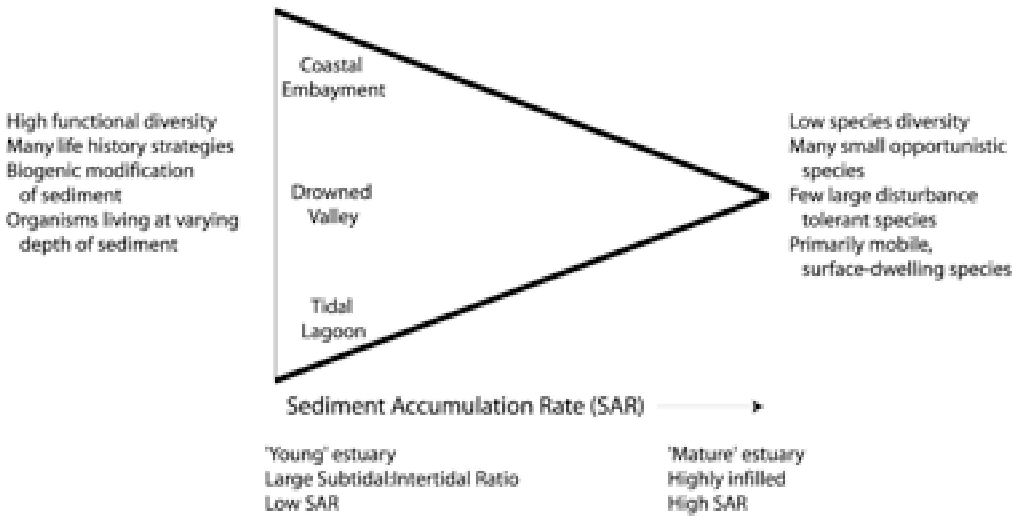
### 3.6.1 Overview

Suspended sediment discharge to the coast has four important impacts:

- Firstly, it can decrease clarity and thus affect ecological integrity and recreational use (see above).
- Secondly, it may smother estuarine sediments with a layer of terrigenous sediments. If the layer is thick enough (e.g., >20 mm) and persists long enough (e.g., > 5 days) it can kill most of the benthic animals due to oxygen starvation (Gibbs & Hewitt 2004). Frequent, thinner and/or less persistent layers can also impact on benthic animals. A mud thickness of around 10 mm, persisting for longer than 10 days, will reduce the number of animals and the number of species present, thereby changing the structure of the animal community. Frequent deposition of mud less than 10 mm, has been observed to have long-term impacts that can change the animal communities.
- Thirdly, it may increase sediment muddiness, i.e., change sediment texture from relatively coarse (e.g., sandy) to relatively fine (e.g., muddy) consistency. The latter effects can be single event based (punctuated) as described above, or through the gradual build-up of muds (increasing depth and spatial extent). This change in texture (either punctuated or gradual) can also alter the types of animal communities present in soft sediment environments.
- Fourthly, it will accelerate estuarine infilling and shallowing.

The above processes can change animal communities and habitats. In estuaries, multiple habitat types, such as salt marsh, sea grass and unvegetated intertidal flats promote diversity by enhancing recruitment and maintaining species with requirements for multiple resources. The modification of available habitats due to elevated sedimentation has been shown to reduce the overall ecological heterogeneity as described in Figure 3.1 (Gibbs & Hewitt 2004).

**Figure 3.1** - Conceptual model of changes in macrofaunal community with increasing sedimentation.




Managing total suspended sediment (TSS) impacts requires a two-pronged approach, involving two sets of guidelines:

- Clarity (see above Section 3.5)
- Sedimentation. This can be managed through placing limits on the sediment deposition rate (SDR) for annual or storm-related loads. The rest of this section describes this type of sedimentation criterion.

### 3.6.2 Existing Guidelines

Schedule 3 specifies narrative deposition criteria only for AE waters, but they are implicit in NS, A and C Waters (Table 3.9).

**Table 3.9 -** Standards for TSS for Water Quality Classes from Schedule 3 RMA

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | The following shall not be allowed if they have an adverse effect on aquatic life: any increase in the deposition of matter on the bed of the water body or coastal water |
| SG | Gathering/Cultivating Shellfish | - |
| CR | Contact Recreation | The visual clarity of the water shall not be so low as to be unsuitable for bathing |
| IA | Industrial Abstraction | - |
| NS | Natural State | The natural quality of the water shall not be altered |
| A | Aesthetic | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified aesthetic values |
| C | Cultural | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified cultural or spiritual values |

Policy 22 of the New Zealand Coastal Policy Statement (2010) specifically relates to sedimentation in the coastal environment, with the following requirements:

1. Assess and monitor sedimentation levels and impacts on the coastal environment
2. Require that subdivision, use, or development will not result in a significant increase in sedimentation in the coastal marine area, or other coastal water.
3. Control the impacts of vegetation removal on sedimentation including the impacts of harvesting plantation forestry.
4. Reduce sediment loadings in runoff and in storm water systems through controls on land use activities.

TSS criteria are not covered in ANZECC (2000) or in the Environmental Response Criteria (ARC 2002).

### 3.6.3 Use/applicability/importance

TSS Guidelines would be used to manage the impacts of land use activities and the deposition of terrigenous sediment from erosion on marine flora and fauna, sediment texture, shallowing (infilling) of harbours and estuaries and recreation use of the coastal environment. They would be highly relevant to protect Auckland's marine receiving environment, and deposition limits are highly appropriate and have already been applied in the site-specific situations described below.

Such criteria would assist with defining limits for the magnitude of erosion from developing and developed catchments. With advanced catchment runoff models that include quantitative benefits of management intervention (such as enhanced CLUES), this would assist with determining catchment management strategies and erosion control techniques that would need to be implemented to meet the limits and the requirements of the NZCPS. There is also a great deal of overlap with freshwater criteria and limits here.

### 3.6.4 Critique/review of existing approaches

At present there are no operational general TSS criteria for Auckland. Two aspects of terrigenous mud deposition are currently managed at some sites in Auckland:

1. thick (>10 mm) dumps of fine, mostly muddy, terrigenous sediment from deposition of sediment carried to coastal areas in large storms; and
2. sediment deposition rates.

One addresses short-term while the other addresses long-term impacts.

Management criteria are still being refined, but interim management objectives of <10 mm and <2 mm per year have been proposed in the Auckland region for acute sediment dumps and chronic accumulation, respectively (Green & Oldman 1999, Norkko et al. 2001, 2002, M. Green, NIWA, pers. comm.).

Limits can be derived from the sediment deposition rate (SDR). Such limits can only be set on a site-specific basis and when converted to a catchment load, can be termed the "sustainable load". This approach is comparable to the USEPA Total Maximum Daily Load (TMDL) approach to limit impacts in receiving waters. In NZ, it can be developed robustly for a few estuaries where comprehensive catchment loads and sediment deposition models are available (e.g., Okura, Mahurangi, Whitford, Central Waitemata, South East Manukau). These studies are relatively costly, so a new approach is currently being developed by NIWA and applied in Porirua Harbour, near Wellington (Mal Green, NIWA, pers. comm).

Similar SDR criteria are suggested for long-term management of Tasman (Nelson) estuaries (Robertson & Stevens 2009a) – probably based on the Auckland studies. Robertson & Stevens (2009a) use a 5-point condition rating and give an early warning trigger value (Table 3.10). This rating is used in the Estuarine Vulnerability Assessment (Chapter 11.3).

**Table 3.10** - Condition rating for sediment deposition rate (SDR) (Robertson & Steven 2009a)

| Condition rating | Sediment deposition rate (mm/yr) |
|---|---|
| Very low | <1 |
| Low | 1 - 5 |
| Moderate | 5 - 10 |
| High | 10 - 20 |
| Very high | >20 |
| Early warning trigger | Increasing |

The distribution of mud is used as a management criterion in Wellington, Tasman and Southland estuaries. These are measured as the per cent cover of the estuary and increase in area of soft muds. The relative area and increase in area are used as condition ratings and to trigger evaluation. Robertson & Stevens (2009a) use a 4 point condition rating based on the proportion (%) and total cover (ha) in the estuary and give a trigger value for initiating an evaluation and response plan. Management objectives to arrest or reverse the spread of muds could be expressed as reducing sedimentation rates as above. This rating is used in the Estuarine Vulnerability Assessment (Chapter 11.3).

**Table 3.11 -** Sediment mud cover condition rating (Robertson & Steven 2009a)

| Condition rating | Soft Mud Per cent Cover | Soft Mud Area |
|---|---|---|
| Very good | <2% of estuary substrate is soft mud | Area of cover is not increasing |
| Good | 2-5% of estuary substrate is soft mud | Increase of <5% above baseline |

| Condition rating | Soft Mud Per cent Cover | Soft Mud Area |
|---|---|---|
| Fair | 5-15% of estuary substrate is soft mud | Increase in area 5-15% above baseline |
| Poor | >15% of estuary substrate is soft mud | Increase in area 15% above baseline |
| Early warning trigger | 5% of estuary substrate is soft mud | Trend of increase in area of cover |

Another measure of muddiness involves measuring benthic macroinvertebrate health, which is measured as diversity and number of benthic macroinvertebrates, and as the occurrence of indicator species, which may include the number and size of rare macrofauna. The Benthic Health Model (Anderson et al. 2006) was created to assign ecological health values to intertidal areas of the region based on storm water contaminant levels. The model has recently been extended to assigning benthic health based on mud content of the sediment (Benthic Health Model mud – BHM-mud) (Hewitt & Ellis 2010). It assigns pollution or mudiness to 4 categories A, B, C, D.

### 3.6.5 Relevance/suitability for Auckland

TSS guidelines as sedimentation rates are highly relevant to the protection of aquatic ecosystems, shellfish gathering and farming areas, and recreational, natural, aesthetic and cultural values, in estuaries and sheltered embayments and harbours around Auckland and the rest of New Zealand.

The soft sediment cover guideline of Robertson & Stevens (2009a) does not appear at first glance to be generally appropriate for Auckland sub-estuaries. The inner estuaries in Waitematā and Manukau are probably naturally muddy. Mangroves and sediment accumulation dynamics within mangrove habitat will complicate the application of this sort of measure. The approach would probably be unsuitable for large complex harbours like the Manukau or Waitematā that have areas of high hydraulic energy, which determine the deposition and retention of mud. However, the guideline may have some use in smaller, sandy, sheltered estuaries, such as the Orewa Estuary. It should be relatively easy to measure and may be a useful management tool in addition to measures of clarity (or turbidity) and sedimentation rate. We therefore recommend that this condition rating be evaluated for application in Auckland estuaries.

Catchment load limits will probably be set to control SDR, clarity and fine sediments in rivers, so this will also affect SDR or muddiness in coastal areas receiving the freshwater discharges. Therefore an integrated approach is needed when setting objectives in fresh and coastal waters.

Levels of protection are plausible for SDR, and there are Auckland experimental data to derive these.

### 3.6.6 Conclusions

For TSS impacts on deposition, we recommend that site-specific sedimentation rates are decided for Auckland coastal waters from existing experience, data and expertise. For TSS impacts on clarity and turbidity, see back to section 3.5.

Any guidelines will be fairly robust because they are developed from comprehensive studies of sedimentation impacts in Auckland, and from NIWA research in other parts of the country. A TSS limits-based approach has already been widely applied in Auckland sheltered waters, as described for Okura, Whitford, Central Waitemata, and South East Manukau.

We also recommend that that Robertson & Stevens (2009a) system for rating the soft mud cover is evaluated in Auckland estuaries.

**Table 3.12 -** Recommendations for TSS in Auckland Coastal Waters

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | Open Coast - N/A<br>Estuaries and sheltered embayments<br>- chronic accumulation 2 mm/year (may be site specific)<br>- acute storm dumps 10 mm/event<br>- investigate and ratify % muddiness approach for Auckland (Table 3.11) |
| SG | Gathering/Cultivating Shellfish | As in Class AE |
| CR | Contact Recreation | As in Class AE |
| IA | Industrial Abstraction | - |
| NS | Natural State | |
| A | Aesthetic | As in Class AE |
| C | Cultural | As in Class AE |

# 4.0 Nutrient enrichment

## 4.1 Overview of eutrophication in New Zealand

Excessive nutrients in the coastal zone can cause eutrophication. Primary responses to eutrophication can be increased summer blooms of phytoplankton, and/or a shift in phytoplankton species that may include an increase of harmful blooms such as toxic algae. Another more common response in New Zealand is an increase in ephemeral benthic algae (epiphytes and bloom-forming macroalgae). The reason for macroalgae blooms rather than phytoplankton blooms is because most of our estuaries are shallow, have large tidal ranges, are well flushed (usually by much of the water draining from the estuary at low tide) and are turbid and light limited (Williamson et al. 2003). These conditions prevent the build-up of phytoplankton in the water column.

Phytoplankton blooms have been observed in New Zealand estuaries with excessive anthropogenic nutrient inputs and long residence times, such as has historically occurred in the Manukau Harbour because of treated sewage inputs. In the nutrient-rich north-east part of Manukau Harbour, concentrations of algal chlorophyll a in excess of 50 mg/m3 have been recorded during several summers between the late 1980s and 1990s (Vant & Budd 1993). Elsewhere in New Zealand, phytoplankton blooms may also occur naturally due to upwelling and inputs of oceanic nutrients due to weather patterns created by the Southern Oscillation (Rees 2009).

Macroalgae are commonly found in many estuaries throughout New Zealand including Auckland – usually the green algae Ulva, or Enteromorpha or red alga Gracilaria. Macroalgal blooms can cause substantial changes in estuarine ecosystems. They can be associated with loss of perennial plants such as seagrass, and consequential loss of perennial habitat. Excessive blooms can cause sediment surface hypoxia and an increase in sediment anoxia. This decreases denitrification, which although requiring low oxygen conditions, is slowed with anoxia. This has important implications for estuaries, because denitrification is one of the major mechanisms that limits nitrogen concentrations in estuaries. Anoxia can also increase nutrient recycling and release from sediments, and increase the amount of toxic sulphide and ammonia in sediments. All these changes can lead to subsequent changes such as loss of seagrass, habitat, clarity, benthic health and benthic microalgae (Rees 2009). Beach-cast macroalgal debris can also be visually unappealing and cause odour issues while they decompose which can restrict recreational use of beaches.

For phytoplankton, nitrogen is the limiting nutrient in summer, and is generally regarded as the most important nutrient to control in the coastal environment. This limitation comes about mainly through loss of nitrogen through denitrification in surface marine sediments. However, some overseas studies have found phosphorus is limiting in spring, while light limitation is important in New Zealand during winter (Rees 2009).

## 4.2 Existing guidelines

Schedule 3 has narrative standards for undesirable biological growths for Class AE, CR and IA waters, but because eutrophication can have a profound effect on water quality, these standards are implicit for Class NS, A and C Waters.

**Table 4.1 -** Standards for Eutrophication for Water Quality Classes from Schedule 3 RMA

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | There shall be no undesirable biological growths as a result of any discharge of a contaminant into the water. |
| SG | Gathering/Cultivating Shellfish | - |
| CR | Contact Recreation | There shall be no undesirable biological growths as a result of any discharge of a contaminant into the water |
| IA | Industrial Abstraction | There shall be no undesirable biological growths as a result of any discharge of a contaminant into the water |
| NS | Natural State | The natural quality of the water shall not be altered |
| A | Aesthetic | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified aesthetic values |
| C | Cultural | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified cultural or spiritual values |

The NZCPS Policies 21 and 23 are applicable here (see Appendix A). In particular from Policy 23:

1. In managing discharges to water in the coastal environment, have particular regard to:
   a. the sensitivity of the receiving environment;
   b. the nature of the contaminants to be discharged, the particular concentration of contaminants needed to achieve the required water quality in the receiving environment, and the risks if that concentration of contaminants is exceeded;
   c. and the capacity of the receiving environment to assimilate the contaminants.

The Environmental Response Criteria (ARC 2002) refer to ANZECC (2000) guidelines for chlorophyll concentration.

*The ANZECC (2000) guidelines have default trigger values for nutrients and chlorophyll concentrations in the water column based on SE Australian estuaries and coastal waters (Table 4.2). They also explicitly state the need to develop nutrient guidelines using local reference data, and this has yet to occur*. ANZECC (2000) default trigger values are still widely (and mostly inappropriately – see later) used for reference when assessing nutrient and chlorophyll concentrations in New Zealand.

**Table 4.2 -** ANZECC (2000) default trigger values for nutrients and chlorophyll (µg/L) in coastal waters. Note that these have limited applicability to Auckland waters (see later).

| Parameter | Chlorophyll | Nitrate | Ammonia | TN | DRP | TP |
|---|---|---|---|---|---|---|
| Marine | 1 | 5 | 15 | 120 | 10 | 25 |
| Estuaries | 4 | 15 | 15 | 300 | 5 | 30 |

## 4.3 Use/applicability/importance

Enrichment of marine waters has not been defined as a widespread problem in the Auckland region to date, although monitoring results for nutrients and chlorophyll around Auckland often marginally exceed the ANZECC (2000) guidelines. Enrichment and algal blooms have been a problem in specific areas because of excessive discharges of nutrients e.g., Manukau Harbour because of past discharges from the Manukau Wastewater Treatment Plant. Algal blooms affect the colour and clarity of water, and if toxic species are involved, the health and well-being of aquatic life, as well as humans and other terrestrial animals which come into contact with the bloom (see Section 4.5 for more detail).

Undesirable biological growths have been noted in Auckland estuaries, usually in the form of macroalgae, and usually in response to nutrient enrichment from wastewater discharges, e.g., Gracilaria in Manukau Harbour. Excessive growth of certain types of macroalgae can cause sediment deterioration, oxygen depletion, bad odours and adverse impacts to biota (Robertson & Stevens 2010), as well as affecting access, recreation and aesthetic experience.

## 4.4 Critique/review of existing approaches

### 4.4.1 Chlorophyll and water column nutrients

ANZECC (2000) Guidelines for nutrients and chlorophyll are commonly used to assess nutrient enrichment of coastal waters by wastewater discharges and from land runoff. We have not been aware of any further action or investigation being triggered by such exceedances. The appropriateness of ANZECC default trigger values has been questioned many times in New Zealand. There is currently an initiative by MfE and regional councils to develop New Zealand trigger values from a compilation of New Zealand data.

The current ANZECC trigger values are too low for Auckland, because they refer to SE Australian marine waters and data source is unknown, they are not related to threshold effects, and Auckland waters have naturally high productivity. However, the ANZECC guideline protocols are appropriate to develop guidelines. Until these trigger values are available, B. Vant (Environment Waikato, pers. comm.) considers it more appropriate to use the overseas trigger values of 10 µg/L (Cloern 2001, Rees 2009) to “trigger” management response. The median concentrations of chlorophyll a are typically below 5 µg/L in New Zealand marine waters including Auckland, but rarely above 10 µg/L. Policy 21 and 23 of NZCPS point more towards maintaining chlorophyll below 5 µg/L rather than allowing it to reach 10 µg/L.

We are unable to make any recommendations for nutrient concentration guidelines in the water column in this report. However, it seems pointless to continue to use ANZECC (2000) default trigger values which when triggered seem to only engender the response that the default values are not relevant.

There are a number of other approaches that are being developed in New Zealand or other countries, and these are summarised below. Some of these are highly relevant to Auckland.

### 4.4.2 Nitrogen and phosphorus loads

Nutrient loading rates are often more important than seawater nutrient concentrations because they can be rapidly cycled through ecosystems and stored in macroalgae. Loads of <50 g N $m^{-2}$ $yr^{-1}$ and 5 g P $m^{-2}$ $yr^{-1}$ can usually be assimilated by primary producers in “healthy” shallow coastal embayments and tidal lagoon estuaries (Rees 2009, Robertson & Stevens 2009a). New Zealand typically has loads <100 g N $m^{-2}$ $yr^{-1}$ (Rees 2009) for tidal lagoon estuaries and coastal embayments. In the more poorly flushed estuaries (e.g., Intermittently Closed and Open Lakes and Lagoons ICOLLs), the loads need to be much lower, < 4 g N $m^{-2}$ $yr^{-1}$ and 0.7 g P $m^{-2}$ $yr^{-1}$ (Scanes et al. 2007, Robertson and Stevens, 2009b) to achieve a similar mesotrophic status. In well-flushed tidal river estuaries, the loads can be much higher (<200 g N $m^{-2}$ $yr^{-1}$) (Robertson and Stevens, 2010).

Loads have also been linked to management of certain habitat and community types in overseas studies, particularly for protection of perennial coastal seagrass communities. In coastal waterbodies overseas, loads of 30 – 100 g N $m^{-2}$ $yr^{-1}$ have been linked to ~50% reduction in seagrass, while loads >100 g N $m^{-2}$ $yr^{-1}$ are associated with 100% loss of seagrass (Rees 2009).

Robertson & Stevens (2009a) have collated some criteria, which could be used as interim values for Auckland.

**Table 4.3 -** Nutrient load limits Robertson & Stevens (2009a)

| Receiving water | Nitrogen load (g N $m^{-2}$ $yr^{-1}$) | Phosphorus load (g P $m^{-2}$ $yr^{-1}$) |
|---|---|---|
| Tidal lagoons and coastal embayments | <50 | 5 |
| ICOLLs | <4 | <0.7 |
| Tidal river estuaries | <200 | unknown |
| Open, well-flushed coast | unknown | unknown |

Nutrient loads limits (expressed as mass per unit area of estuary) are easily translated into a catchment-specific limits-based management.

### 4.4.3 Trophic state indices

The current international direction in nutrient and eutrophication assessments appear to be trending towards trophic state multiparameter indices similar to those applied to freshwater systems (S. Speed, AC, pers. comm.). There are many of these; a good appraisal is the scientific supporting document for the "Canadian Environmental Quality Guidelines for Nutrients: Canadian Guidance

Framework for the Management of Nearshore Marine Systems" - http://www.ccme.ca/assets/pdf/ntrnts_gdnc_frmwrk_ssd_1387.pdf.

One of the better known and earliest is the Trophic Index of Marine Systems (TRIX) that measures TN, TP, Chlorophyll a, Secchi Disk (a measure of clarity). Unlike freshwaters, enrichment effects would be confined to a few of the lower trophic levels (the equivalent eu-, hyper- and super-trophic levels frequently found, and of most concern, in freshwaters are unlikely to occur in Auckland's marine waters). As described later, we do not recommend a trophic state index yet, until further development is carried out.

A similar approach described in Chapter 11 is the OSPAR Eutrophication Objectives, which uses multiple indicators of nutrient enrichment besides water column concentrations (dissolved N and P, chlorophyll a, and DO); incorporating biotic monitoring of trophic state such as indicator species, blooms and fish kills. However, these are not combined into one index.

The above shortcomings have been recently addressed with the New Zealand Estuary Trophic Index (ETI) Tool (Robertson et al. 2015), which is a stand-alone, hard-copy methodology that includes two sets of tools that provide screening guidance for assessing where an estuary sits in the eutrophication gradient, and what is required to shift it to a different location in the gradient.

- Screening Tool 1. Physical and Nutrient Susceptibility Tool.

This method is designed to provide a relatively robust and cost effective approach to enable the prioritisation of estuaries for more rigorous monitoring and management. It applies a desktop susceptibility approach that is based on estuary physical characteristics, and nutrient input load/estuary response relationships for key New Zealand estuary types. The tool produces a single physical susceptibility score that can be used to classify either the physical susceptibility (i.e. very high, high, moderate, low susceptibility), and/or be combined with nutrient load data to produce a combined physical and nutrient load susceptibility rating. Nutrient areal load/trophic state bands for each estuary eutrophication type will be developed as a long term goal, with data currently available for some estuary types, but not all as yet. This section also provides guidance on the use of a simple load/response model tool provided in the ETI toolbox, and recommendations for the use of more robust approaches for setting load limits.

• Screening Tool 2. Trophic Condition Assessment Tool.

This tool is a monitoring approach that characterises the ecological gradient of estuary trophic condition for relevant ecological response indicators (e.g. macroalgal biomass, dissolved oxygen), and provides a means of translating these ratings into an overall estuary trophic condition rating (the ETI). It provides guidance on which condition indicators to use for monitoring the various estuary types (and why they have been chosen), and on assessing the trophic state based on the indicator monitoring results and their comparison to numeric impairment bands (e.g. very high, high, moderate, low). The latter involves measurement of the expression of both primary (direct) eutrophication symptoms (e.g. macroalgae phytoplankton) and supporting indicators for secondary (indirect) symptoms of trophic state.

The Estuary Vulnerability Assessment (EVA - See Section 11.3) developed in New Zealand incorporates many of the indicators that might be used in a trophic index. While it does use measures of water column nutrients, chlorophyll a, and clarity, as described above (Section 4.1) enrichment in New Zealand is more likely to cause an increase in ephemeral benthic algae (epiphytes and bloom-forming macroalga). This may be associated with sediment enrichment and anoxia and benthic films. Consequently EVA more commonly utilises a range of indicators (and associated guidelines) including N and P in sediments, total organic matter, redox discontinuity, macroalgae and epiphyte coverage and extent, surface films, loss of benthic algae and macrophytes and benthic community health. These are not combined into one “trophic” index, however. The physico-chemical indicators N and P in sediments, total organic matter, redox discontinuity are described below or in later sections.

### 4.4.4 Nutrients (N, P) in sediments

Nutrient enrichment may increase available N and P in sediments (as well as potentially toxic ammonia). At the present time, there are no criteria for N and P in sediments in Auckland, and SoE monitoring focuses on concentrations in the water or catchment loads. Inclusion of their measurement in management programmes may lead to development of useful criteria.

Robertson & Stevens use a 4-point condition rating and give a trigger value (Robertson & Steven 2009a) in their Estuary Vulnerability Assessment. Their values would need to be ratified for Auckland through measuring reference conditions as per ANZECC guidelines. The implications for elevated nutrients in sediments are not well understood, and guidelines for sediment nutrients might be very useful if a link was established with an undesirable effect such as growth of nuisance macrophytes.

**Table 4.4 -** Sediment nutrient condition rating (Robertson & Steven 2009a)

| Condition rating | Total Nitrogen (mg/kg) | Total Phosphorus (mg/kg) |
|---|---|---|
| Very good | < 500 | < 200 |
| Good | 500 – 2000 | 200 - 500 |
| Fair | 2000 - 4000 | 500 – 1000 |
| Poor | > 4000 | > 1000 |
| Early warning trigger | 1.3 x mean of highest baseline year | 1.3 x mean of highest baseline year |

Nutrient concentrations in sediments are reasonably amenable to a catchment-specific limits-based approach. Although processes are quite complex (e.g., denitrification in estuarine sediments), they have been extensively modelled overseas.

### 4.4.5 Silica (as $SiO_2$) concentrations

SiO2 is the other major nutrient implicated in eutrophication and phytoplankton biomass and type. It is sourced mainly from rocks and soils and would not be managed in terms of criteria etc. However, its measurement may be important to help understand nutrient enrichment in Auckland waters, especially if a shift from diatom-dominated systems was observed. Therefore, as part of developing trigger values for coastal nutrient concentrations from New Zealand data (see below), SiO2 should also be measured to enable comparison and calculation of N (and P) ratios with SiO2. In the future, the relationship between these ratios and any known phytoplankton responses, such as phytoplankton blooms, harmful algal blooms or changes from diatom to non-diatom predominance, can be assessed and developed.

## 4.5 Relevance/suitability for Auckland

Nutrient enrichment is of widespread concern in Auckland (e.g., Kaipara Harbour - Haggitt et al. 2008, IKHMG 2010) but has not been shown to be a widespread problem, although there are relatively few definitive studies. The concerns relate to many reported problems overseas, relatively little information for New Zealand, increasing nutrient inputs from land runoff because of time lag effects, land use intensification (especially dairy conversions) and increasing human populations.

Harmful algal blooms (HAB) occur from time to time in NZ; they have been documented 3 times in Auckland in the past 10 years. Most algal blooms throughout the world's aquatic systems usually pose no direct health risk, but occasionally certain species produce endo- or exotoxins that may accumulate in edible shellfish and also can have direct health effects on human/animal consumers, or may be harmful to aquatic life, such as causing mass fish mortality. All algal blooms require adequate nutrients, flushing times greater than algal growth rates (i.e., greater than population doubling times), adequate light conditions, and limited grazing of algae by higher animals. In New Zealand HAB seem to be related to natural conditions (e.g., climatic factors). Overseas, HABs have been additionally associated with excessive nutrients, changes in nutrient speciation and poor flushing. Harmful algae are regularly monitored throughout New Zealand to protect shellfish farmers and consumers, and warnings are issued, when appropriate, to shellfish farmers, health authorities and their agents. Regional trigger values to identify general algal bloom conditions will presumably help manage and minimise the risk of blooms (including HABs) occurring.

Eutrophication effects are more likely to manifest in the form of undesirable macroalgae growths rather than phytoplankton blooms, and sediment enrichment. Therefore guidelines are needed as indicators of these conditions, one of which is described above (nutrient loads), while the other biological ones are beyond the scope of this report. In this respect, the EVA approach (Chapter 11.3) may be highly relevant to managing eutrophication and other forms of enrichment in Auckland, and could be used to select indicators and their regional trigger values developed under ANZECC protocols for Auckland.

Multiparameter Trophic Indices may be developed in the future if experience shows that it is warranted and useful, but current information suggests that this would not be the best way forward. There is a need to develop a lot more understanding of nutrient enrichment around Auckland before choosing the appropriate metrics for a multi-parameter indicator.

Undesirable biological growths affect all water quality classes, and this is summarised in summary Table 4.5.

Setting objectives in freshwaters for nutrients may need to take into account any objectives set in coastal waters in the future. The nutrient concentrations in estuaries and their influence on plant growth (algae, macroalgae) and productivity will depend on the estuary nutrient budget, which will be partly dependent on freshwater discharges from the catchment. At the present day, The NOF only specifies TN and TP concentrations for lakes, while objectives for rivers are set for periphyton growths. To manage these will require understanding of nutrient sources, dynamics and budgets, and limits may be developed around nutrient loads. While it is not yet clear what nutrient criteria will be applied to estuaries, any criteria management may also require understanding of the contributing catchment nutrient sources, dynamics and budgets.

Potential conflicts between objective setting in freshwater and coastal water arise if dissolved nutrients are specified for rivers (e.g., nitrate toxicity) and for estuaries, e.g., reference conditions (Hickey et al. 2015). This could be easily checked by comparing concentrations after taking dilution into account; dilution may be estimated using salinity. The primary nutrients of concern are inorganic nitrogen – which are expected to be the limiting nutrient in most marine environments. A lack of marine guidelines for eutrophication nitrogen currently limits assessment of potential effects.

## 4.6 Recommendations

As described in General Water Quality above (Section 3.1), regional reference conditions need to be developed for nutrient and chlorophyll concentrations in the different types of marine receiving waters. To achieve this, both management objectives and levels of protection for managing nutrient enrichment and eutrophication first need to be defined in order to specify the trigger values. This process could be helped from the reference data itself, because it will include both nutrient and chlorophyll concentration, and so thresholds for the onset of impacts may be able to be derived from environmental gradients. It is anticipated that more robust trigger values for nutrients and chlorophyll in the water column will come out of a current review of New Zealand data[8]. $SiO_2$ monitoring should also be implemented as described above.

Nutrient in sediments and nutrient loads appear to be promising approaches for nutrient management, especially for the dominant problem of macroalgae proliferation. Nutrient loads would be easily adapted to a limits-based approach.

**Table 4.5** - Guidelines for Eutrophication for Water Quality Classes

| Class | Purpose (all Classes) | Criteria |
|---|---|---|
| All | Aquatic Ecosystem, Gathering/Cultivating Shellfish, Contact Recreation, Industrial Abstraction, Natural State, Aesthetic, Cultural | In the interim, default trigger values of 5 µg/L chlorophyll for management investigations for potential blooms (including harmful algal blooms)<br>Need to develop criteria using ANZECC protocols for:<br>• nutrient concentrations in water (N, P, $SiO_2$);<br>• chlorophyll;<br>• nutrient loads (load per area of estuary);<br>• macroalgae and epiphytes;<br>• sediment enrichment (N and P);<br>based on local data |

Harmful algae blooms (HAB) are regularly monitored throughout NZ, and it would be timely that this information is evaluated to see if any management implications can be teased out. In the meantime, regional trigger values to identify general algal bloom conditions, will presumably help management minimise the risk of toxic blooms occurring, at least in respect to any role of nutrient enrichment in these blooms.

We do not recommend developing a Trophic State Indicator at this stage, but recommend instead that a New Zealand database is established for all ecological quality objectives relevant to eutrophication and enrichment in New Zealand (listed in Robertson & Stevens 2009a and Rees 2009 and summarised in Table

[8] At the time of writing it was not clear when this will be completed.

4.5). This existing and new information would then be assessed to develop guidelines, including some that relate nuisance macroalgae, epiphytes and phytoplankton to areal nutrient loads.

# 5.0 Toxicants in surface waters

## 5.1 Introduction

There are a large number of commonly used anthropogenic chemicals that are potentially discharged in stormwater, farm runoff and wastewater. The toxic effects are exerted in the water column through a multitude of mechanisms.

The only water column toxicants of known interest to Auckland are probably ammonia and the metals copper (Cu), zinc (Zn), and lead (Pb). Criteria for these contaminants are described in this chapter. Toxic algae are discussed in the previous chapter.

There are many other potentially toxic contaminants listed in criteria tables such as the ANZECC guidelines, but none of these have yet been shown to be an issue for Auckland. These include some of the Chemicals of Potential Environmental Concern (CPEC). Their concentrations are low and often challenging to measure in water, let alone assess effects, but they do tend to accumulate in sediments and so are discussed further in Chapter 7. If any other dissolved toxic contaminant becomes an issue in Auckland, say, associated with point source discharges, then reference can be made to the comprehensive list of toxicants in the ANZECC (2000) guidelines under a “catch-all” guideline category.

Schedule 3 of the RMA has explicit narrative standards for toxic contaminants for Class AE Waters (Table 5.1), but similar standards are implicit in Class NS and C Waters.

**Table 5.1** - Standards for Toxic Contaminants for Water Quality Classes from Schedule 3 RMA

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | The following shall not be allowed if they have an adverse effect on aquatic life:<br>(c) any discharge of a contaminant into water |
| SG | Gathering/Cultivating Shellfish | - |
| CR | Contact Recreation | - |
| IA | Industrial Abstraction | - |
| NS | Natural State | The natural quality of the water shall not be altered |
| A | Aesthetic | - |
| C | Cultural | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified cultural or spiritual values |

Policy 23 of the New Zealand Coastal Policy Statement (2010) specifically relates to discharge of contaminants to the coastal environment, with the following requirements:

In managing discharges to water in the coastal environment, have particular regard to:

a. the sensitivity of the receiving environment;
b. the nature of the contaminants to be discharged, the particular concentration of contaminants needed to achieve the required water quality in the receiving environment, and the risks if that concentration of contaminants is exceeded;
c. and the capacity of the receiving environment to assimilate the contaminants;
d. and:
e. avoid significant adverse effects on ecosystems and habitats after reasonable mixing;
f. use the smallest mixing zone necessary to achieve the required water quality in the receiving environment; and

g. minimise adverse effects on the life-supporting capacity of water within a mixing zone.

The ANZECC (2000) guidelines use a decision tree approach for assessing toxicity. As with all their guidelines, they list trigger values or propose developing site-specific trigger values. Two responses are possible if “triggers” are exceeded:

1. Incorporation of additional information or further site-specific investigation to determine whether the chemical is posing a real risk to the environment.
2. Accept the trigger value without change as a guideline (e.g., numerical objective) applying to the site and initiate an investigation.

Investigations as to whether the chemical is posing a real risk or not usually follow decision trees. This approach is relatively complex, and in some cases it is challenging to develop a simple numerical objective. If the trigger value is accepted as a guideline, then that decision needs to be justified or, at least, ratified.

While a decision can be made to accept TVs as numerical, as outlined in Section 3.1, the first step is choosing the level of protection, and that depends on the ecosystem condition. ANZECC (2000) recognised three types of ecosystem conditions;

1. high conservation/ecological value,
2. slightly to moderately disturbed ecosystems,
3. highly disturbed ecosystems.

Appropriate levels of protection must be decided, often in consultation with stakeholders, and are typically either: 1) 99%, 2) 95%, 3) 90% or 80% across these conditions (see ANZECC (2000) page 3.4-11). These levels of protection can be directly related to summaries of toxicity information.

## 5.2 Ammonia

### 5.2.1 Overview

Ammonia is a basic industrial chemical, a product of organic matter decomposition, a soil nutrient, and a common product of human and animal wastes. It commonly occurs in point sources, catchment runoff and receiving waters in the Auckland region, but usually at low, non-toxic concentrations. The main concerns are associated with elevated levels in human and animal wastes and in enriched sediments.

Ammonia occurs as two forms that are in equilibrium in water: unionised ammonia ($NH_3$) and ionised ammonia ($NH_4^+$); and its toxicity is mainly due to the unionised form. Its toxicity is well characterised and depends on the equilibrium between the two forms, which in turn depends on pH, temperature and salinity.

### 5.2.2 Existing guidelines

Schedule 3 Narrative Standards are described above.

The ANZECC (2000) guidelines for waters are current, robust and unlikely to change with the current revision of the guidelines. However, whereas in the ANZECC (2000) guidelines, pore water toxicity was assessed using water column values, the revised guidelines will have unique guidelines for ammonia in sediments (i.e., in the sediment pore water).

Table 5.2 lists values for the ANZECC (2000) trigger values for seawater at pH 8. The guidelines can be consulted for ammonia levels at other pH and salinity situations.

**Table 5.2 -** ANZECC (2000) trigger values for Total Ammonia as $NH_3$-N in seawater at pH 8.

| **Level of protection** | | | |
|---|---|---|---|
| **99%** | **95%** | **90%** | **80%** |
| 500 μg/L | 910 μg/L | 1200 μg/L | 1700 μg/L |

The Environmental Response Criteria (ARC, 2002) recommended using ANZECC (2000) guidelines.

### 5.2.3 Use/applicability/importance

Ammonia is relevant in marine waters receiving wastewater discharges, which can contain high concentrations of ammonia.

It is also a critical parameter in sediment toxicity (see Chapter 7).

### 5.2.4 Critique/review of existing approaches

Current and any revisions in ANZECC are appropriate.

### 5.2.5 Relevance/suitability for Auckland

Ammonia toxicity is an important consideration with wastewater discharges and industrial discharges.

Setting objectives in freshwaters for ammonia may need to take into account any objectives set in coastal waters in the future. Ammonia limits may be more stringent in marine than fresh waters, because of higher seawater pH, counteracted to some degree by higher salinity. In the extreme case, managers may need to need consider dilution/pH/salinity effects in special cases involving larger ammonia mass loads; these circumstances are expected infrequently, if at all.

There is also the potential that the TN load to estuaries may need to be considered because sediment pore water concentrations of ammonia will increase with high TN loads (Hickey et al. 2015). High pore water ammonia concentrations may adversely affect infaunal species and the survival of desirable macrophyte species (e.g., eel grass). As described above in section 3.5, management of an estuary for macrophytes will require consideration of sediment characteristics (e.g., muddiness, nutrients, ammonia). Thus an integrated assessment of multiple factors may be required for macrophyte management, which may be influenced by upstream freshwater objectives and limits for nutrients and TSS.

### 5.2.6 Recommendations

We recommend that the ANZECC (2000) guideline trigger values for different levels of protection are used (Table 5.2), which will require measurement of salinity, temperature and pH. While freshwater ANZECC guidelines have become more conservative in recent (as yet unpublished) ANZECC revisions, there are no plans known by the authors to revise the marine ammonia guidelines. Some caution will need to be applied when using these guidelines in estuaries with significant proportions of freshwater.

## 5.3 Heavy metals copper (Cu), lead (Pb) and zinc (Zn)

### 5.3.1 Overview

Cu, Pb and Zn are well-known contaminants in urban runoff that can be toxic in water at relatively low concentrations. They have been extensively studied, so their toxicity is well understood. However, for any given concentration, toxicity can vary widely across different receiving waters depending on types of animals present and the chemical characteristics of the water.

### 5.3.2 Existing guidelines

Schedule 3 Narrative Standards are described above.

The current ANZECC (2000) guideline trigger values are under review and there is likely to be a small change in values. The ANZECC (2000) approach is described in Section 5.1.

**Table 5.3** - ANZECC (2000) trigger values for Cu, Pb and Zn in marine waters.

| | Level of protection | | | |
|---|---|---|---|---|
| | **99%** | **95%** | **90%** | **80%** |
| Cu | 0.3 μg/L | 1.3 μg/L | 3 μg/L | 8 μg/L |
| Pb | 2.2 μg/L | 4.4 μg/L | 6.6 μg/L | 12 μg/L |
| Zn | 7 μg/L | 15 μg/L | 23 μg/L | 43 μg/L |

For management of short-term pulses of toxicants, such as might occur in urban stormwater, it may be necessary to specify maximum concentrations. These are not available in the ANZECC (2000) guidelines, but it has been proposed that the updated ANZECC guidelines will provide maximum (i.e., acute values). Until these are available, we recommend the USEPA Ambient Water Criteria - Criteria Maximum Concentration (CMC) as shown in Table 5.4.

**Table 5.4**- USEPA acute and chronic trigger values for Cu, Pb and Zn in marine waters.

| | CMC | CCC |
|---|---|---|
| Cu | 4.8 μg/L | 3.1 μg/L |
| Pb | 210 μg/L | 8.1 μg/L |
| Zn | 90 μg/L | 81 μg/L |

The CMC is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed briefly without resulting in an unacceptable effect (=acute concentration). [Note that the Criterion Continuous Concentration (CCC) is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect (=chronic concentration). The CMC and CCC are just two of the six parts of an aquatic life criterion in the USA; the other four parts are the acute averaging period, chronic averaging period, acute frequency of allowed exceedance, and chronic frequency of allowed exceedance.]
http://water.epa.gov/scitech/swguidance/standards/current/index.cfm

### 5.3.3 Use/applicability/importance

Cu is a common contaminant in Auckland in stormwater and may also be associated with ports and antifouling paints, and use in existing and historical horticultural areas.

Pb is a common contaminant in Auckland associated with stormwater chiefly from its use as an additive in petrol. It is still being discharged from urban areas despite its removal from petrol. Present day sources may be from buildings (historical use of lead-based paints, lead flashing/plumbing) and the accumulation of Pb in soils near buildings and roads from past use in paints and petrol.

Zn is a common contaminant in Auckland associated with stormwater and rural runoff, and may also be associated with port activities, structures (galvanised iron), paints and paint removal (e.g., on steel structures), antifouling paints, point sources and animal remedies (e.g., facial eczema in pasture areas).

### 5.3.4 Critique/review of existing approaches

The ANZECC (2000) trigger values (and eventually the revised values) are highly appropriate for filterable Zn, Cu and Pb concentrations in Auckland waters. It is highly unlikely that Pb concentrations will exceed trigger values because the fraction of Pb in the soluble phase (and hence potentially bioavailable) is well below trigger values.

Background concentrations for Cu and Zn are not well understood, but are needed in order to manage these metals robustly (ANZECC (2000) page 3.4-16), because trigger levels are close to background. There is a little, and insufficient, information available from a few past and present studies in Auckland (reviewed in Mills and Williamson 2008, Williamson and Mills 2009b).

Another approach, particularly for point source discharges, is to use toxicity testing. This is a well-developed approach and is routinely available in New Zealand at a number of specialist laboratories. A promising new approach, which has been successfully applied for Cu in freshwater, is the Biotic Ligand Model. These toxicity testings are described in the following sections.

**General aquatic toxicity**

Toxicity testing includes WETT (Water Effluent Toxicity Testing) and DTA (direct toxicity assessment – ANZECC (2000)). There are a range of well-developed standard techniques, which test aquatic animals in laboratories that could be used to develop toxicity guidelines. The most obvious application is for point sources that discharge multiple toxicants (e.g., the tests would need to show that the effluent is not toxic to appropriate test organisms after reasonable mixing). Expert opinion from qualified ecotoxicologists would be required for site-specific quality of point discharges.

**Developing a Robust Copper Criterion – the Biotic Ligand Model (BLM)**

The BLM is a computer model that uses 10 water chemistry parameters to calculate a freshwater copper criterion in the USA. The BLM is the basis of USEPA's 2007 national recommended 304(a) freshwater criterion for copper and this could be developed in Auckland. There is the possibility that it will be extended to Zn and other metals (e.g., Ni) and also to marine waters; however it is probably some time off before robust criterion can be developed (if at all) for Cu and Zn in marine waters.

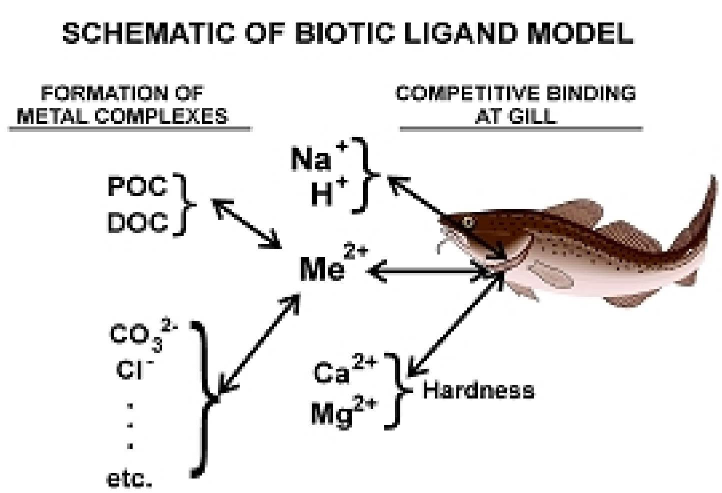
Water quality can affect metal toxicity (in particular, Natural Organic Matter [NOM], and pH have a strong effect on copper, but hardness cations, alkalinity, and sodium also play a role). Failure to consider these effects may make a WQC overprotective or underprotective for a large number of sites where permits for metals discharges are needed. The BLM can be used to consider these effects when developing copper criteria. The BLM predicts acute freshwater WQC using an approach similar to that of predicting organism toxicity; chronic WQC is derived from acute using the Acute/Chronic Ratio (ACR). A recent review by Van Genderren et al. (2008) has indicated that major reductions in bioavailability may occur in many waters as a result of elevated DOM concentrations.

Using the BLM allows regulators and dischargers to account for the effect of water chemistry parameters (e.g., DOC, pH, major ions, and alkalinity) on metal toxicity to aquatic organisms on a more site-specific basis. Using the BLM provides more accurate WQC without the expense or time required for deriving a water effect ratio (WER). The ideas behind the BLM are not new. Similar ideas were proposed nearly 30 years ago (such as Pagenkopf's Gill Surface Interaction Model, and the Free ion activity model).

The BLM requires a description of water chemical parameters that can influence metal toxicity. These parameters include: pH; DOC (a convenient measure of NOM); and major ions. Major ions also have specific effects on copper toxicity including:

- To calculate ionic strength, which affects speciation
- Calcium, Magnesium, and Sodium (which can all reduce copper toxicity)
- Either alkalinity or dissolved inorganic carbon (used by the BLM to estimate copper-bicarbonate complexation)

**Figure 5.1** - The Biotic Ligand Model



Copper toxicity in freshwater fish occurs due to disruptions of ion regulation in gill membranes. Similar mechanisms have been demonstrated for other aquatic organisms. Anything that might affect how copper interacts with gill membranes (such as the presence of calcium in the water) may also influence copper toxicity.

- http://water.epa.gov/scitech/swguidance/standards/criteria/aqlife/pollutants/copper/2007_index.cfm
- http://water.epa.gov/scitech/swguidance/standards/criteria/aqlife/pollutants/copper/
- http://www.hydroqual.com/wr_blm.html

The application of the BLM to marine waters was reviewed by Gadd & Cameron (2012) and the key points from this review are summarised here. “Arnold et al. (2006) proposed equations for calculating site-specific criteria for copper based on the concentration of dissolved organic carbon (DOC) in the water body. These equations were derived from toxicity of copper to *Mytilus galloprovincialis*, the blue or Mediterranean mussel, and incorporated toxicity testing in laboratory water and water collected from estuarine environments. These equations were:

- *Chronic criterion (µg/L) = 3.59 * DOC 0.60*
- *Acute criterion (µg/L) = 5.61 * DOC 0.60*

The European Copper Institute (2008) has recently undertaken a voluntary risk assessment for copper, which included deriving a Proposed No Effect Concentration (PNEC) for copper in marine waters. This PNEC incorporated the influence of DOC by normalising toxicity data to a standard DOC concentration prior to PNEC calculation. The calculated PNEC for dissolved copper was 5.2 µg/L, for a water body with DOC of 2 mg/L. This is very similar to Arnold’s chronic criterion of 5.4 µg/L for DOC of 2 mg/L. The European Copper Institute’s risk assessment has been adopted by the European Commission (European Commission 2009)”.

Overall the marine BLM has not yet progressed to the point that it provides sufficient robustness in application. However, given that studies are underway on Cu and DOM binding in the Whau estuary, it is likely that AC will further its investigations into its application in the future. Its robust application in Auckland coastal waters would benefit from studies on the toxicity reduction for Cu to local, key representative species (e.g., mussel larvae, algae).

### 5.3.5 Relevance/suitability for Auckland

Heavy metals are elevated in urban streams during base and storm flows. A recent study (Elwood et al. 2008) found concentrations of dissolved Cu and Zn that exceed ANZECC trigger values in the Whau River estuary. The high levels appeared to be due to high concentrations in the freshwater flow and because of release of Cu and Zn from sediments. Another study in the contaminated Mangere Inlet with small freshwater flows has found Cu and Zn levels well below trigger values (Williamson et al. 1996).

Recent studies commissioned by the New Zealand Environmental Protection Authority (NZ EPA) have modelled antifouling-derived dissolved copper concentrations in 11 ports and 13 marinas throughout New Zealand (Gadd et al. 2011). Both Auckland ports (Auckland, Manukau) and five marinas (Gulf Harbour, Westpark, Westhaven, Bayswater, Half Moon Bay, Pine Harbour) were modelled based on the average number and size of boats present. All of the Auckland marinas were predicted to exceed the 95% protection guideline for dissolved copper, with the highest in Half Moon Bay where the total copper was predicted to exceed the 80% protection level.

A follow-up survey was undertaken of eight marinas in Auckland (Gulf Harbour, Westhaven, Westpark, Bayswater, Half Moon Bay, Pine Harbour, Milford and Orakei) to measure dissolved and total copper concentrations for comparison with model MAMPEC predictions (Gadd & Cameron 2012). In most of the marinas, the concentrations exceeded ANZECC (2000) water quality guidelines for aquatic protection based on either 95% or 90% levels of protection. In four of the eight marinas, the concentrations also exceeded site-specific chronic water quality guidelines for aquatic protection derived from the concentration of Dissolved Organic Carbon (DOC) measured at each site. In two of the eight marinas (Westpark and Milford), the concentrations also exceeded site-specific acute water quality guidelines derived from the concentration of DOC at each site. Overall the predictions of total copper from the MAMPEC model were considered to be close enough to the measured concentrations to support the further use of this model in risk assessment.

Gadd & Cameron (2012) used measured copper load results for Westpark Marina, together with predictions from the MAMPEC model for other port and marina loads, to estimate of the total export of copper to Auckland harbours. This equated to approximately 3100 kg/year, which is roughly double that predicted from stormwater for the entire Waitematā Harbour catchment (Gadd & Cameron 2012).

Auckland's urban streams have elevated levels of Zn and Cu (Mills & Williamson 2009a), which results in a wide spatial and temporal input of elevated dissolved metal concentrations to coastal waters. These discharges, coupled with the Elwood et al. (2008) and Gadd & Cameron (2012) studies, signal the need to seriously consider Cu and Zn toxicity in the water column of estuaries, and adopt guidelines for their management. At some point, the AC will need to make a decision whether to accept ANZECC trigger values as guidelines, or promote the approach which investigates whether any exceedances represent a real risk to aquatic ecosystems (Section 5.1).

This might involve the application of the 'more robust' Biotic Ligand Model (BLM) (Gadd & Cameron 2012). As described above, a version of the BLM model is currently used in the USA in freshwater to determine Cu guidelines. Research on its application to marine situations is currently under way and currently provides acute and chronic guideline values in relation to dissolved organic carbon (DOC) (Arnold et al. 2005, 2006). However, this BLM for marine waters is based on toxicity to a single species and is yet to be developed to a stage that provides predictions for multispecies effects and bands for differing levels of protection. The BLM approach may also extend to Zn guidelines in the future. The AC will need to keep abreast with developments of this model.

Potential conflicts between objective setting in freshwater (under the NPSFM) and in coastal water in the future are not immediately apparent. Criteria commonly used (based on ANZECC 2000) are similar in fresh and marine waters. However, changes in speciation and toxicity with changes in salinity, pH, DOC and accompanying desorption/adsorption processes are not being taken into account in the present day, but may be in the future. If future criteria include speciation assessments as described earlier, or the development of different levels of protection (and hence attribute bands), or involve loads of heavy metals, then potential conflicts will need to be reassessed (Hickey et al. 2015).

### 5.3.6 Recommendations

We recommend that trigger levels for different levels of protection for dissolved (i.e., filterable) Cu and Zn are adopted from the revised ANZECC guidelines

(Table 5.3). The current ANZECC (2000) guideline trigger values are under review and there is likely to be a small change in values.

At some point a decision will need to be made whether to accept trigger values as guidelines, or promote the approach which investigates whether any exceedances represent a real risk to aquatic ecosystems. If the former is used, then the numerical objectives would be trigger values listed in Table 5.3. However, we recommend the latter approach using the ANZECC protocols: a well-designed and executed bioavailability study should provide numerical objectives that are applicable across the region.

Background (baseline) concentrations of Cu and Zn are not well understood, but are needed in order to demonstrate that trigger levels are significantly higher than background. Some background concentration information is gradually accumulating from a few past and current studies in Auckland, but more is needed.

Pb is unlikely to be toxic in the water column under most conditions because the dissolved fraction is usually very low and well below trigger values. On the basis of existing information of low concentrations, we recommend that any Plans do not specifically include trigger levels for Pb, but that it is included in a “catch-all” clause.

## 5.4 Other metal, metalloid and organic toxicants in water

Other metal, metalloid and organic toxicants have not been identified at potentially toxic concentrations in surface waters around Auckland or are still poorly understood. Requirements for their management will probably be infrequent and usually associated with location specific issues such as contaminated sites, spillages or industrial point sources. Elevated concentrations of other contaminants found in sediments (e.g., metals and metalloids such as mercury and arsenic, and PAHs, (McHugh and Reed 2006; pharmaceuticals and emerging contaminants, Stewart 2013, Stewart et al. 2014) are generally expected to be transported to the estuary as sediment-associated contaminants rather than as elevated dissolved concentrations – which could exceed their respective water quality guidelines. As such, controls to manage sediment-associated contaminants in stormwaters and sewage overflows will be expected to result in reduced concentrations of these contaminants entering the marine environment. We recommend that any plans refer to the possibility of using ANZECC trigger levels in a “catch-all” clause for any metals, metalloids and organics that are found in the future to represent a threat to aquatic ecosystems and need managing. This approach is limited to those metal, metalloid and organic toxicants listed in ANZECC, and may not cover many of the emerging contaminants.

# 6.0 Human health

## 6.1 Human health risks in the aquatic environment

Primary recreation risk is largely determined by health risks, which in turn are determined by the risk of disease and by physical hazards. The importance of water clarity and its effects with physical hazards has been briefly discussed elsewhere (see Clarity). The water quality in which contact recreation activities occur (swimming, skiing, paddling, kayaking, shellfish gathering and consumption) needs to be such that accidental ingestion of small quantities of the water or consumption of seafood does not result in illness. In these activities there is a reasonable risk that water-borne contaminants will be swallowed, inhaled, or come in contact with ears, nasal passages, mucous membranes or cuts in the skin, allowing pathogens to enter the body (MfE 2003).

The risk of catching a water-borne disease is determined by many factors, but of primary interest is the concentration of pathogenic microorganisms in the water column. Water contaminated by human or animal excreta may contain a range of pathogenic (disease-causing) micro-organisms, such as viruses, bacteria and protozoa.

## 6.2 Existing guidelines

Schedule 3 specifies a narrative microbiological standard for SG and CR waters, but they are implicit in C Waters.

**Table 6.1** - Standards for Microbiological Quality for Water Quality Classes from Schedule 3 RMA

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | - |
| SG | Gathering/Cultivating Shellfish | Aquatic organisms shall not be rendered unsuitable for human consumption by the presence of contaminants |
| CR | Contact Recreation | The water shall not be rendered unsuitable for bathing by the presence of contaminants |
| IA | Industrial Abstraction | - |

| Class | Purpose | Criteria |
|---|---|---|
| NS | Natural State | - |
| A | Aesthetic | - |
| C | Cultural | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified cultural or spiritual values |

Policy 23 of the New Zealand Coastal Policy Statement (2010) specifically relates to discharge of contaminants to the coastal environment (Section 2.3), with the following specific requirements:

2. In managing discharge of human sewage, do not allow:
   a. discharge of human sewage directly to water in the coastal environment without treatment; and
   b. the discharge of treated human sewage to water in the coastal environment, unless:
      i. there has been adequate consideration of alternative methods, sites and routes for undertaking the discharge; and
      ii. informed by an understanding of tangata whenua values and the effects on them.
3. Objectives, policies and rules in plans which provide for the discharge of treated human sewage into waters of the coastal environment must have been subject to early and meaningful consultation with tangata whenua.

### 6.2.1 Quantitative criteria

**Shellfish tissue: FSANZ (2002)**

- No more than 1 of 5 replicates >230 E. coli per 100 g
- No replicates >700 E. coli per 100 g
- [The previous standard was no more than 2 of 5 reps >230 FC per 100g, none >330 per 100 g]

**Water Quality for Shellfish Gathering: MFE (2003)**

- Median Faecal Coliform over shellfish gathering season <14 MPN per 100 mL
- No more than 10% of samples >43 MPN per 100 mL

**Marine Water Quality for Contact Recreation: MFE (2003)**

- Indicator organism is Enterococci. Limits based on 95%iles (Hazen method).
- Microbiological Assessment Category (Hazen 95%iles, per 100 mL) (MAC):
  - A: <40
  - B: 41–200
  - C: 201–500
  - D: >500
- Green/acceptable surveillance mode: No single sample >140. Continue weekly sampling.
- Amber/Alert mode: single sample >140, but <280. Increase to daily sampling. Sanitary survey.
- Red/Action mode: 2 consecutive samples >280. Daily sampling. Sanitary survey. Warning signs.

Note also for beach grading need to do sanitary assessment (Sanitary inspection Category – SIC).

**Freshwater (and Marine in special circumstances) Water Quality for Contact Recreation: MFE (2003)**

- Indicator organism is E. coli. Limits based on 95%iles (Hazen method).
- Microbiological Assessment Category (Hazen 95%iles, per 100 mL):
  - A: <130
  - B: 131–260
  - C: 261–550
  - D: >550
- Green/acceptable surveillance mode: No single sample >260. Continue weekly sampling.
- Amber/Alert mode: single sample >260. Increase to daily sampling. Sanitary survey.

- Red/Action mode: single sample >550. Daily sampling. Sanitary survey. Warning signs.

## 6.3 Use/applicability/importance

Enterococci, (and faecal coliforms and E. coli) are used to assess the risk to human health in contact recreation, and these are key indicators of marine water quality. Special regular surveys are now de rigor at the more popular bathing beaches, and results are published on-line.

Stormwater runoff invariably contains high levels of indicator bacteria irrespective of land use, although developed land will have higher concentrations than undeveloped land. In urban stormwater, sources could include birds, dogs, cats, rats, either directly or surviving indicator bacteria in soils and sediments; non-reticulated sewage systems; leaky sewerage; and pump overflows.

Studies in New Zealand have shown that farmed livestock, along with birds, are a significant source of microbial indicator organism contamination of water bodies (McBride et al. 2002). Pathways of water contamination are varied and complex, including direct deposition (e.g., from stock in waterways), surface runoff, and movement through the soil profile. The relative importance of these pathways is influenced by a range of factors: e.g., soil type, slope, soil moisture content, rainfall, and farm management practices. While stock spend proportionately little time in water bodies while grazing, they tend to defecate proportionately more in those water bodies, with significant impact on microbial loadings. Soils that are naturally poorly drained are often associated with high by-pass flow of microbes via soil cracks or worm channels, whereas naturally freer draining soils often have a high level attenuation of microbes as they pass through the soil profile. Artificially drained soils can also often result in significant microbial flows through the drainage channel.

## 6.4 Critique/review of existing approaches

This review/critique summarises the rationale for the enterococci criteria and points to other indicators such as E. coli and Quantitative Microbial Risk Assessment (QMRA) when monitoring gives ambiguous results or where treated wastewater discharges are involved. It also points to some common methods used in research, which offer future possibilities for monitoring and risk assessment, such as Microbial Source Tracking. Another recent development is to predict indicator loads using catchment runoff models that could potentially include sources (number and types of animals) and attenuation of microbes (e.g., from best management practices). This is in its infancy, but could develop in the future as the basis for a limits-type approach (e.g., limiting the total load of indicator bacteria discharged to an estuary). These are described later in this section.

### 6.4.1 MFE guidelines

It is difficult and impractical to measure the level of pathogens in the water directly. Instead, levels of "indicator bacteria" are measured that provide an indication of the likely pathogenicity of the water, providing a feasible monitoring approach, as comprehensively reconfirmed by a recent study (Till et al. 2008). Microbiological guidelines for recreation and shellfish gathering in marine and fresh waters are provided in MFE 2003. These guidelines present a protocol for determining the suitability of waters for recreational use, including a grading system based on catchment characteristics and receiving water quality. The following paragraphs are summarised from MFE (2003).

For marine waters the preferred indicator is enterococci. The New Zealand Marine Bathing Study showed that enterococci are the indicator most closely correlated with health effects in New Zealand marine waters, confirming a pattern seen in a number of overseas studies. Faecal coliforms and E. coli were not as well correlated with health risks, although they may be used as an indicator in addition to enterococci in environmental conditions where enterococci levels alone may be misleading. In addition, E. coli rather than enterococci should be used as an indicator wherever the primary source of faecal contamination is a waste stabilisation pond. Faecal coliform concentrations are specified as the indicator for waters where there is shellfish collection for consumption.

Wastewater treatment processes often effectively reduce microbial indicators such as enterococci but are less effective at removing pathogens such as viruses. The result may be an altered pathogen-to-indicator ratio compared to that of untreated waste. This means that if there is a wastewater treatment plant present, pathogens may still be present even when indicator levels are very low (MoH/MfE 2003). Consequently, it is better to conduct a Quantitative Microbial Risk Assessment (QMRA), which is described below.

Another approach to circumvent the difficulties of using indicator bacteria is to measure specific pathogens directly. This is used in research studies, and the prognosis for their application to routine monitoring is summarised below.

### 6.4.2 Quantitative microbiological risk assessment

QMRA uses the best spatial and temporal measurements of pathogenic microbial concentrations to estimate the risk (including the uncertainty in the risk) that they pose to human health[7]. QMRA has been used in New Zealand to characterise the risk of treated sewage discharges to human health at a number of freshwater and marine locations including the following Auckland sites: Beachlands, Waiwera, Warkworth (Mahurangi Harbour) and Helensville (Kaipara Harbour) (Stott & McBride 2004, Stott & McBride 2008, McBride 2007/2008, McBride 2008, Palliser & McBride 2009a).

QMRA has four stages[9] as follows.

**Hazard Identification:** Identify a pathogen and the disease it causes, including symptoms, severity, and death rates. Identify sensitive populations that are particularly prone to infection. In New Zealand, rotavirus has been used because there is good clinical trial data available and associated dose-response relationships. Rotavirus is also one of the most UV-resistant viruses implicated in water-borne disease outbreaks, it is a hazard for humans, and is a good representative for other pathogens (McBride 2007/2008). It can infect through shellfish ingestion, as well as accidental water ingestion. However, there is some uncertainty whether it is a good representative for norovirus, which has a very low infectious dose (Palliser & McBride 2009b).

**Dose-Response:** Data sets from human studies allow the relationship between the dose (number of microbes) received and the resulting health effects to be quantified, and allows the construction of mathematical models to predict dose-response. In the application of QMRA in New Zealand, the more conservative risk of infection is characterised rather than the risk of illness.

**Exposure Assessment:** Describes the pathways that allow a microbe to reach people and cause infection (e.g., by ingestion). The size and duration of

[9] Centre for Advancing Microbial Risk Assessment http://camra.msu.edu/qmra.html

exposure by each pathway needs to be determined. In the application of QMRA in New Zealand, model variables and inputs are;

1. the virus concentration in the sewage plant influent;
2. efficacy of the treatment plant;
3. mixing factors;
4. duration of swim event;
5. ingestion rate of water during swimming;
6. shellfish consumed; and
7. bioaccumulation factor for the microbe in shellfish.

Transport of microbes has been carried out using particulate transport/hydrodynamic models, which also allow for die-off. Exposure is usually characterised at known accessible recreational areas (e.g., at beaches, shellfish gathering sites). In addition the risks associated with extreme microbial concentrations have been calculated, i.e., when a community-wide viral illness has occurred (Palliser & McBride 2009b).

**Risk Characterisation**: The information from the steps above is integrated into a mathematical model to calculate risk – the probability of an outcome of infection. Monte Carlo Analysis is used to generate a risk profile including average and worst-case scenarios.

QMRA could be specified as a requirement to assess health risks associated with wastewater discharges.

### 6.4.3 Direct measure of pathogens

A three-year study was initiated in 1997 to investigate microbial contamination in New Zealand water bodies (McBride et al. 2002). Although this study focussed on surface freshwater sites throughout New Zealand, they are still relevant because freshwater inflows are the main source of contamination of marine water. Four microbial indicators and six highly relevant pathogens (enteroviruses, adenoviruses, Cryptosporidium oocysts, Giardia cysts, Salmonellae and Campylobacter) were monitored. The Campylobacter detection rate was 60%, virus pathogens were detected in about one-third of all samples, the Salmonellae detection rate was low (10% of samples), while Giardia and Cryptosporidium cysts were detected infrequently (8% and 5% respectively).

The main outcomes of the risk assessment from this study were that, of the pathogens assessed in this study, Campylobacter and adenoviruses are the most likely to cause human waterborne illness to recreational freshwater users. An estimated 5% of total notified Campylobacteriosis in New Zealand could be attributed to water contact recreation.

Land use within the study catchments showed that “bird” catchments were the most contaminated across nearly all micro-organisms. Dairying catchments were often the second most contaminated, but not for Campylobacter or adenoviruses. The municipal and forestry/undeveloped catchments were generally the least contaminated.

**Campylobacter**

Campylobacter is a human pathogen, a specific zoonose, largely derived from dairy cows. It is therefore an indicator of human pathogen pollution and contamination of waterways by dairy cows. It is, however, difficult to measure. It could be developed as an indicator in the future when stable, routine analytical methodology becomes available.

**Viruses**

Direct measures of human adenoviruses and retroviruses are indicators for human viral pollution, septic tanks and poor WWTP treatment. However, they are very difficult and costly to monitor. Major issues at present are associated with analytical methodology, which keeps changing (and improving) and the results from different methods are not directly comparable. It is probable that this indicator (including norovirus in QMRA) will become more widely applied in the future.

### 6.4.4 Catchment sources and modelling loads

Indicator organism loads and concentrations can be calculated by using the Catchment Land Use for Environmental Sustainability (CLUES) package (Semadeni-Davies et al. 2009). At present CLUES predicts average annual E. coli loads and not enterococci loads or concentrations. The CLUES model is being developed to include management practices. This development is centred on the Kaipara Harbour, and thus will be useful in coastal management in the future.

### 6.4.5 Faecal source tracking

Faecal Source Tracking is commonly practised overseas to investigate indicator exceedances and has been applied in New Zealand (Devane et al. 2010, Nobel 2014). Overseas the issues are related to management of sources, the difficulty of managing diffuse sources - especially in undeveloped lands, multiplication of indicators organisms within the aquatic environment (e.g., in the tidal wrack) and the expectation that most of these non-human sources do not constitute a direct health risk. However, in NZ, there is strong evidence that there are health risks with animal sources (birds and ruminants). Faecal Source Tracking is likely to continue to be used as an investigative tool for greater understanding of sources of faecal contamination and their management in areas where further investigation of issues have been identified from E. coli and enterococci monitoring results.

Specific faecal source monitoring studies have been undertaken in Auckland coastal freshwater lagoons – which are considered regionally important because of their high recreational use. The West Coast lagoons at Karekare, Piha, North Piha and Te Henga have formed where their freshwater and marine environments interact. These lagoons have been monitored under the council bathing beach ‘Safeswim’ programme, with data indicating that they would be graded “very poor”. A pilot study was undertaken in 2012-2013 to identify the biological sources of the faecal pollution of the West Coast lagoons (Noble 2014). The results of that study showed that a range of animal faecal sources were polluting the lagoons which were originating from human (septic systems), dogs, wildfowl, livestock and unidentified sources. The findings of the study indicated that a range of site-specific measures will be required to provide suitable water quality for primary contact recreation.

## 6.5 Relevance/suitability for Auckland

The MfE Guidelines are highly relevant and applicable in Auckland. Marine waters involve a great deal of contact recreation, especially in near shore waters at the popular bathing beaches. Microbiological water quality is generally good at most Auckland beaches over the summer bathing season, with relatively few alert or action levels recorded. However, results vary from year-to-year, depending largely on the number of times sampling coincides with rain events, which increase stormwater inflows and hence indicator levels (summarised in Mills & Williamson 2008). For example, Auckland Council's Safeswim programme recorded only 1% of 361 samples above the alert level in 2004-5 compared with 4% in the previous summers. As described previously, problem sites are generally downstream of catchments with significant wastewater overflow problems. For example, overflows triggered most beach "closures" in North Shore City over summer 2003-4 and short periods of rainfall were reported to be responsible for most of the elevated levels in Manukau.

Other contact recreation in open water (boating, kayaking, wind surfing) is less well served than bathing beaches, but there is useful data from state of the environment monitoring sites. Monitoring is also carried out near wastewater outfalls, to ensure treatment is sufficient and risks to contact recreation in these areas are very small.

The bacteriological quality in the headwaters of estuaries is relatively unknown, but limited data suggests relatively high levels of indicator bacteria (e.g., Diffuse Sources 2007), probably mostly from catchment runoff, complicated by survival of indicator bacteria in sediments, tidal wrack etc, and their release during resuspension in waves etc. The characteristics and implications of this remain very much unquantified and not well understood.

Primary contact recreation is a major activity in estuaries and coastal lagoons. Therefore, any future objectives for coastal waters are expected to be fairly rigorous and to be influenced by objectives in any freshwaters discharging to the coastal environment. The NPSFM requires councils to set freshwater objectives and limits in their regional plans. A companion report (Hickey et al. 2015) assesses the potential for conflicts or cross-over issues arising from setting objectives in freshwaters and downstream coastal waters. If lower water quality states are specified for freshwater which discharges near a coastal beach, then potential conflicts will need to be assessed (Hickey et al. 2015). This is somewhat complicated in the general case when different indicators (enterococci for marine, E. coli for freshwaters) are used for the different media, and probably require the measurement of both indicators in freshwater when it discharges to coastal waters. Objectives and limits for E. coli would also need to assess the implications for enterococci in any coastal discharge. Such conflicts may be better first addressed using the MfE (2003) catchment sanitary assessments, which calculate beach grades.

## 6.6 Recommendations

The current MfE criteria (see Section 5.2) are appropriate for Auckland and are already widely used in Auckland at many bathing beaches and in the main bodies of estuaries, harbours and their mouths. Levels of protection are implicit in MfE Guidelines for bathing and explicit in QMRA. Recommended guidelines are listed in Table 6.2. The criteria also include a catchment assessment, which although not a true limits-based approach, is probably a very good approach for assessing catchment sources, and dealing with a number of very complex sources and processes. It is also a good starting point for resolving potential conflicts when setting limits and numerical objectives in freshwater and downstream receiving waters. Nevertheless catchment modelling, which is in its infancy, should be continued to be developed, to see if this will yield a robust catchment-specific limits-based approach to managing microbiological pollution in both freshwaters and downstream coastal waters.

QMRA should be continued to be used to assess health risks associated with sewage discharges.

The faecal source tracking approaches look to be very promising in some situations to help understand any non-compliance and its management and Auckland Council should keep a watching brief on developments overseas.

**Table 6.2** - Recommendations for microbiological guidelines in Auckland Coastal Waters

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | |
| SG | Gathering/Cultivating Shellfish | MfE Guidelines for faecal coliforms and shellfish gathering (Section 6.2)<br>QMRA for WWTP discharges |
| CR | Contact Recreation | MfE Guidelines for enteroccoci and catchment Sanitary Inspection Category (Section 6.2)<br>QMRA for WWTP discharges |
| IA | Industrial Abstraction | - |
| NS | Natural State | |
| A | Aesthetic | As in Class SE |
| C | Cultural | MfE Guidelines for faecal coliforms and shellfish gathering<br>MfE Guidelines for enteroccoci and catchment Sanitary Inspection Category<br>QMRA for WWTP discharges |

# 7.0 Sediment toxicity

## 7.1 Overview

Some sediment contaminant concentrations can reach levels that are toxic to the aquatic life that lives in the sediments. This has the potential to affect the ecology of immediate areas, and beyond, because sediment-dwelling (benthic) organisms serve as key food sources for animals further up the food chain (e.g., fish, birds).

Sediment quality in Auckland's coastal region has been the subject of many studies, which have been comprehensively and succinctly reviewed (e.g., Mills & Williamson 2007, Williamson & Mills 2008, Kelly 2009) and will be only summarised briefly here.

Contaminants of most concern in Auckland are Cu, Pb, Zn, PAH, and DDT. Contaminants of lesser concern, but which may still be having an impact, are Hg, Cd, As, TPH, and PCBs. All these contaminants are classified as "priority pollutants", which have consistently high environmental persistence, high bioaccumulation and high acute toxicity.

A new group of chemicals are emerging throughout the world as being of potential environmental concern, based on their toxicity, persistence, and widespread or on-going use. These have been termed Chemicals of Potential Environmental Concern (CPEC) or Emerging Chemicals of Concern (ECC). In contrast to the "priority pollutants", many CPECs have a lower environmental hazard profile. Notably, many CPECs have lower acute toxicity than Priority Pollutants (PP). Nevertheless, some CPECs have the potential to exert chronic effects by being neuroactive or acting as hormone mimics (endocrine disrupting chemicals). Some are associated with high production volumes, so there is a potential for accumulation of these chemicals in Auckland's receiving environment, with unknown consequences. The differences between PP and CPEC are summarised in Table 7.1.

**Table 7.1** - Comparison of risk profile of priority pollutants and emerging chemicals of potential environmental concern (adapted from Ahrens 2008)

| Property | Priority Pollutants | CPEC |
|---|---|---|
| Toxic effects and mode of action | Acute and chronic | Most not likely to be acutely toxic at environmental doses, but potentially bioactive (e.g., estrogenic, neuro-active), sometimes at very low concentrations |
| Environmental concentrations | Frequently monitored; stable or decreasing (except Zn, Cu, PAH in urban stormwater) | Not frequently monitored, assumed to be increasing |
| Persistence | High | Variable: unknown, low, medium, high |
| Bioaccumulation potential | High | Variable: unknown, low, medium, high |
| Sources | Mainly industrial and agricultural; building materials and vehicles; few domestic (i.e., sewage) | Some industry and agriculture runoff; mostly domestic (via sewer overflows, wastewater discharges) |
| Existing water quality guideline | Yes | No |
| Discharge regulated | Often (but not in diffuse runoff in NZ) | Rarely |
| Detection and quantification | Relatively easy; methods are well established | Often difficult and expensive to measure; focus now on use of biomarker techniques |
| Examples | As, Cd, Hg, Pb, DDT, PCB, PAH, dioxin, Cu, Zn | Surfactants, plasticizers, disinfectants, modern pesticides, flame retardants, hormones, cosmetics, new antifouling paints, medicines, veterinary medicines |

## 7.2 Existing guidelines

### 7.2.1 RMA Schedule 3

RMA Schedule 3 has explicit narrative standards for toxic contaminants for Class AE Waters (Table 7.2), but similar standards are implicit in Class NS and C Waters.

**Table 7.2** - RMA Standards for Toxic Contaminants for Water Quality Classes from Schedule 3 RMA

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | The following shall not be allowed if they have an adverse effect on aquatic life:<br>(c) any discharge of a contaminant into water |
| SG | Gathering/Cultivating Shellfish | - |
| CR | Contact Recreation | - |
| IA | Industrial Abstraction | - |
| NS | Natural State | The natural quality of the water shall not be altered |
| A | Aesthetic | - |
| C | Cultural | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified cultural or spiritual values |

### 7.2.2 ANZECC (2000) Sediment Quality Guidelines (SQG)

The ANZECC (2000) guideline utilises international SQG values, termed ‘trigger values’ (TVs). The recommended application of the SQGs involves a tiered, decision-tree approach, in keeping with the risk-based approach in the water quality guidelines described in the previous sections. Firstly, the total concentrations of contaminants are compared to the TVs and if the contaminant concentrations exceed one or a number of the TVs, further investigations should be initiated to determine whether there is indeed an environmental risk associated with the exceedance. The TVs are not intended to be used on a pass/fail basis.

### 7.2.3 Environmental Response Criteria (ERC)

In 2002, Environmental Response Criteria (ERC) were recommended for Auckland from the currently available international sediment quality criteria (Williamson & Mills 2009a, ARC 2002, 2004). The fundamental reason for the development of the ERC was that:

1. concentrations of Cu, Pb, Zn and DDT exceeded ANZECC TVs in older urbanised estuaries;
2. preliminary evaluation of bioavailability suggested that metals could be bioavailable;
3. the concentrations of major stormwater contaminants, Cu and Zn would continue to increase, and more areas would become triggered in the future from the continual discharge of urban stormwater;
4. benthic ecology provided strong evidence that the exceedance of TVs had caused a decrease in benthic ecology health;
5. the ANZECC (2000) TV values for Cu and Pb were considered unjustifiably too high.

Therefore guidelines were needed to guide management responses. These guidelines were called thresholds or targets, but here ‘target’ does not have the same meaning as it is used in the NPSFM (Table 2.1). Exceedance of orange ‘traffic light’ ERC targets for Cu, Pb, Zn, PAH and some organochlorines (e.g., DDT) “triggered” further assessment (e.g., benthic health evaluations) and exceedance of red ‘traffic lights’ “triggered” assessing contaminant sources and magnitude and their management options. These ERC were never fully ratified beyond the draft Coastal Plan and were subject to considerable debate and a number of proposed revisions. The current wisdom is that the red ERC are not useful, and that the amber ERCs (the “TELs”) are used as “triggers” for full evaluations (benthic ecology impacts; source and discharge management).

**Table 7.3** - Environmental Response Criteria (from ARC 2004).

**A.** Primary contaminants (mg/kg)

| Parameter | Red[1,2] | Amber[1,2] | Green[1,2] | Source of Red-Amber Threshold | Source of Amber-Green Threshold[3] |
|---|---|---|---|---|---|
| Zinc | >150 | 124-150 | <124 | ERL | ISQG (CCME) |
| Copper | >34 | 19-34 | <19 | ERL | ISQG (CCME) |
| Lead | >50 | 30-50 | <30 | ISQG ANZECC | ISQG (CCME) |
| HMW PAH[4] | >1.7 | 0.66-1.7 | <0.66 | ISQG ANZECC | TEL |

**B.** Secondary toxic organics ($\mu$g/kg)

| Parameter | Red[1,2,4] | Amber[5] | Green[1,2,4] | Source of Green-Red Threshold[3] |
|---|---|---|---|---|
| Chlordane | >2.3 | | <2.3 | ISQG (CCME) |
| p,p'-DDD | >1.2 | | <1.2 | ISQG (CCME) |
| p,p'-DDE | >2.1 | | <2.1 | ISQG (CCME) |
| p,p'-DDT | >3.2 | | <3.2 | ISQG (CCME) |
| DDT, total | >3.9 | | <3.9 | TEL |
| Dieldrin | >0.72 | | <0.72 | ISQG (CCME) |
| Lindane | >0.3 | | <0.3 | ISQG (CCME) |
| Total PCB | >22 | | <22 | ISQG (CCME) |

[1] Values rounded to two significant figures.

[2] Values are for the total sediment in the settling zone and for the mud fraction within the outer zone.

[3] Sources (Summarised in Appendix B):

ERL = Effects Range Low (Long et al. 1995, Long 1992, NOAA 1999)

TEL = Threshold Effects Level for Florida Department of Environmental Protection (MacDonald 1996)

ISQG-Low = Interim Sediment Quality Guideline-Low (ANZECC 2000)

ISQG CCME = Interim Sediment Quality Guideline for Canadian Council of Ministers for the Environment (CCME 1999).

[4] After normalization to an organic carbon content of 1% (normalization range 0.5 to 10% organic carbon).

[5]Amber values for organochlorines are not specified because of uncertainties about the sources of these contaminants and trends in their concentrations. Any exceedance of the green values will therefore require an investigation into trends and effects. In contrast, the major sources of heavy metal and PAH contaminations are relatively well known. Concentrations are expected to increase into the foreseeable future because their sources will persist in urban catchments.

Policy 23 of the New Zealand Coastal Policy Statement (2010) specifically relates to discharge of contaminants to the coastal environment. See Section 2.3 for general policies, including the following:

In managing discharges to water in the coastal environment, have particular regard to:

a. the sensitivity of the receiving environment;
b. the nature of the contaminants to be discharged, the particular concentration of contaminants needed to achieve the required water quality in the receiving environment, and the risks if that concentration of contaminants is exceeded;
c. and the capacity of the receiving environment to assimilate the contaminants.

### 7.2.4 ANZECC revisions

The following is a summary of what the authors believe to be the current status of the ANZECC revisions. There may be further changes before they are published. As with the original ANZECC (2000) guidelines, the first-level screening compares the TV with the measured value for the total contaminant concentration in the sediment. The use of the term sediment quality guideline values (SQGVs) has now been adopted to replace the originally termed 'trigger values' (Simpson et al. 2013), and the interim sediment quality guidelines low and high (ISQG_L; ISQG_H) from the ANZECC (2000) guidelines. Simpson et al. (2013) have also proposed retaining the SQG-High value as a higher likelihood of effects threshold. If the SQGV is exceeded, then the next levels of screening considers the background concentrations and fractions of the contaminant that is likely to be bioavailable or can be transformed and mobilised in a bioavailable form (based on chemical measurements). Toxicity testing of sediments is included at a lower tier of the decision framework. The contaminants whose concentrations exceed SQGVs following consideration of contaminant bioavailability are termed contaminants of potential concern (COPCs).

The decision-tree now proceeds to the evaluation of additional lines of evidence (LOEs) to determine whether the COPCs are likely to affect ecosystem health. Chemistry (including bioavailability measures), ecotoxicology, bioaccumulation and benthic ecology are general LOEs, but other LOEs may be added on a case-specific basis. A weight of evidence (WOE) approach evaluates the combination of the individual LOEs.

The current revisions of the ANZECC (2000) guidelines are likely to bring about further changes to organochlorines and PAH SQGVs, but possibly not Cu, Pb and Zn SQGVs. The lack of revision of the Cu and Zn SQGVs would be unsatisfactory for Auckland Council, who have made several representations for them to be revised. However, there will be significant changes to the organochlorine and PAH guidelines due to a shift in methodology used to derive these values. This will result in more defensible dieldrin values (which were almost always exceeded in sediment under the old guidelines), while DDT values will be lowered significantly (and theoretically trigger more exceedances), but changes to PAH values will not be significant for Auckland. Other important changes include the introduction of a total petroleum hydrocarbon (TPH) SQGV, which although based on limited information, is an important addition to the common suite of urban-related contaminants. Additionally, pore water ammoniacal-N guidelines for sediments are also being proposed, although these are currently not being recommended for generic application.

As before, the SQGV will not be pass/fail numbers, but triggers for further investigation.

## 7.3 Use/applicability/importance

### 7.3.1 Cu, Zn, Pb

Sediment contamination (and the consequential effects on aquatic life) is probably the major impact of stormwater on marine receiving environments in Auckland. It has therefore been comprehensively studied and has become one of the most well defined measures of impacts of urban stormwater in the Auckland region. Sediments and associated contaminants settle in the sheltered areas of estuaries and harbours. Muddy sediments build up over time, and so do contaminant concentrations. At some point, the contaminant concentrations reach levels that are toxic to the aquatic life that lives in the sediments.

Cu, Pb, and/or Zn concentrations exceed ERL sediment quality guideline values at 22% of the ARC's 72 stormwater contaminant monitoring sites, but concentrations do not exceed the ANZECC (2000) ISQG-Hi (=ERM; =SQG-HIGH) guideline values at any of these sites.

Existing contaminant concentrations are highest in sheltered inlets with long histories of urbanisation i.e., sub-estuaries of the central Waitematā Harbour, the upper Tamaki Estuary and Mangere Inlet (Kelly 2008). Concentrations of all three metals exceed the amber ERC guideline value (=TEL), and frequently the red ERC guideline value (=ERL). Metal concentrations are slightly elevated in areas with mixed rural urban land use. Marine receiving environments adjacent to other rural catchments and exposed coast have metal concentrations close to background levels (Kelly 2008).

### 7.3.2 As, Cd, DDT, Hg, PAH, PCB, TPH

These contaminants have been a source of concern in Auckland (Mills & Williamson 2008, McHugh & Reed 2006).

Arsenic is close to and occasionally exceeds trigger levels (e.g., Hewitt et al. 2006), although most experts regard this as reflecting natural levels. In line with ANZECC (2000), these exceedances are undergoing investigation.

Cd levels have been building up in New Zealand pasture soils from superphosphate additions (Kim 2005), but this has not appeared to have as yet resulted in exceedances of ANZECC trigger levels in rural estuaries. The potential problem may resolve itself by the use of other fertiliser types and sources (but may still be a problem in freshwater sediments and farm soils).

DDT has been found to exceed ERC in some estuarine urban sites around Auckland (Williamson & Mills 2009a). DDT is largely a legacy contaminant and investigations have described its occurrence in estuary sediments and catchment soils (Diffuse Sources, 2003a, 2003b, 2004a). Limited information suggests that it is not increasing in estuarine sediments (Diffuse Sources 2004b, Tonkin & Taylor 2005a, b).

Mercury often exceeds ERC-Amber threshold levels slightly (e.g., Hewitt et al. 2006, Mills 2014a). Six of the 27 monitoring sites in 2005 exceeded the ANZECC (2000) ISQG-Low threshold, with all falling below the ISQG-High threshold (McHugh and Reed 2006). It is possible that the Hg contamination is associated with past and ongoing stormwater contamination (Mills 2014a) because they correlate with Zn concentrations. However, the sources, fate and effects have not been studied.

PAH has been a major cause of concern, because high concentrations were found in a few sub-estuaries associated with the legacy production and use of coal tars in roading. Generally PAH is unlikely to exceed guidelines in sediments, or if it does, is likely to be classified as non-toxic after employing the ANZECC Weight of Evidence approach because of limited bioavailability (Ahrens and Hickey 2003, Dupree & Ahrens 2007). UV-activation of body-burden PAH is possibly plausible for surface-dwelling, transparent organisms, including juvenile Macomona bivalves (Ahrens et al. 2002), but this requires more research. A very preliminary appraisal considered it unlikely to be a significant issue (Williamson & Mills 2009a, ARC 2002).

Elevated PCBs can occur at older urbanised sites, but do not exceed the ERC-Red threshold (Mills 2014b). (There are no ERC-Amber thresholds). It has been recommended that PCBs at the more contaminated sites should be continued to be monitored on a “watching brief” to determine trends (Mills 2014b).

TPH levels have not been measured much in Auckland (although many earlier environmental impact assessments probably included this contaminant). Their distribution and concentrations in Auckland needs further research (there may be sufficient data in Environmental Impact Assessments to conduct a preliminary regional assessment and compare with the new ANZECC guidelines).

### 7.3.3 Other “priority pollutant” metals, metalloids and organics

Other metal, metalloid and organic toxicants have not been identified at potentially toxic concentrations in intertidal sediments around Auckland, and so requirements for their management will probably be infrequent and usually associated with contaminated sites or industrial point sources. Dieldrin concentrations in Auckland sediments often exceeded ANZECC (2000) TVs, because these TVs were artificially low being based on limited data. The currently proposed revisions address this problem, and dieldrin concentrations are likely to be much lower than the currently revised TVs (see Section 7.2.4).

### 7.3.4 Emerging contaminants

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

In addition to urban stormwater as a potential source for such CPEC as pesticides, plasticizers, and petroleum products, Ahrens (2008) identified many other potential sources in the urban landscape including marinas, sewage outfalls, combined sewage overflows, landfill leachate, and agricultural runoff.

CPEC have been surveyed in Auckland on two occasions, and these surveys characterise typical concentrations and distributions (Stewart 2013, Stewart et al. 2009, 2014). In addition, EDC measurement and assessment have been reviewed in relation to Auckland (Singhal et al. 2009).

## 7.4 Critique/review of existing approaches

This section is relatively comprehensive; SQG are arguably the most important criteria for managing the marine environment in Auckland, so this section carefully examines their relevance, robustness and use. It then describes other indicators besides chemical concentrations (the Benthic Health Model), measuring sediment toxicity directly and new work developing site-specific SQGs for Auckland.

### 7.4.1 International sediment quality guidelines

There are trigger values for a large number of potential toxicants in the ANZECC (2000) guidelines (and even a greater number in other recent sediment quality guidelines (e.g., see SQuiRTs – Screening Quick Reference Tables – Buchman 2008)). Most of these are based on international sediment quality guidelines, so we need to describe these first.

In the following we refer to specific contaminants when they are only likely to be an issue for management of the coastal region in Auckland.

The use of large effects databases is now the most widely-accepted approach to sediment quality guideline development, and a number of guidelines have been developed using this approach. These guidelines were derived from the relationship between adverse biological effects (e.g., toxicity) on benthic animals and contaminant concentrations in sediments. The chemical concentrations observed and predicted by the different methods to be associated with biological effects were sorted from lowest to highest concentrations, and the lower 10 percentile and median concentrations are usually identified.

The effects database provides no insight into reasons or mechanisms for toxicity, and cannot be used to determine cause-effects relationships. They include results where mixtures of chemicals have resulted in the observed effect. One or more of these chemicals (or unmeasured chemicals) may have produced the effect, but it is ascribed to all chemicals in the mixture. Effects levels entered for some chemicals may therefore be well below actual effects thresholds. This limitation is overcome to some extent by a concordance analysis so that only those chemicals exhibiting a concentration-dependent relationship with observed toxicity were included. Also some guidelines have other procedures to filter out so-called co-occurrence data involving matching sediment chemistry and biological effects, prior to inclusion in the database.

These SQGs have other limitations. They are based upon the associations between contaminant concentrations and biological responses in (mainly) real-world, soft-bottom sediments. They largely rely on a definition of toxicity for a standard test protocol, which in many cases is a single-species laboratory-based toxicity measure (e.g., 10-day amphipod survival test) rather than a multiple-species field measure (which would have greater ecological relevance). It is possible that factors other than the concentration of the contaminant under investigation may have influenced some of the test results, reducing their reliability. Factors which control contaminant bioavailability (e.g., sulphides, organic carbon, particle size), and hence toxicity, are not included in SQG derivation, limiting the reliability of the SQGs over wide ranges of sediment types. They are most useful in the types of sediments used in their derivation and testing (fine grained sediments) and may not work well in other types of sediments (e.g., coarser sandy sediments, sediments contaminated with particulate contaminants e.g., coal fragments). Relatively few chronic toxicity results have been used in deriving the SQGs, limiting their application when assessing long-term, low level effects.

These SQGs may predict non-toxicity (relative to a given test) reliably, but by themselves, they should never be used to conclude that a type of sediment will have a toxic effect on the environment, or that particular chemicals are going to have toxic effects. Rather, they signal the need for further investigations to determine whether significant impacts are likely to occur in the environment, and which contaminants (if any) are likely to responsible for adverse effects. This is reflected in the ANZECC (2000) decision tree approach to determining impacts of contaminated sediments.

Thus there is uncertainty about their ability to predict actual ecological impacts in receiving waters and causative linkages to a specific, individual contaminant. Their use is very much restricted to their designated role as “triggers” for further work when exceedances occur. Furthermore, even when specific guideline thresholds are not exceeded, there is still a low probability of a related ecological effect. Such effects may be caused by unmeasured chemical contaminants or complex interactions occurring in the specific sediment.

### 7.4.2 Robustness of international SQG

It is instructive to consider the efficacy for prediction of “toxic” effects for various guidelines. For ERL and ERM (USEPA 2009):

*“O’Connor et al. (1998), using a 1,508-sample EPA and NOAA database, found that 38% of ERM exceedances coincided with amphipod toxicity (i.e., were toxic), 13% of the ERL exceedances (no ERM exceedance) were toxic; and only 5% of the samples that did not exceed ERL values were toxic. O’Connor and Paul (2000) expanded the 1,508-sample data set to 2,475 samples, and the results remained relatively unchanged (41% of the ERM exceedances were toxic, and only 5% of the nonexceedances were toxic). In a database generated in the EPA National Sediment Quality Survey (U.S. EPA, 2001), 2,761 samples were evaluated with matching sediment chemistry and 10-day amphipod toxicity. Of the 762 samples with at least one ERM exceedance, 48% were toxic, and of the 919 samples without any ERL exceedances, only 8% were toxic (Ingersoll et al., 2005). These data also showed a consistent pattern of increasing incidence of toxicity as the numbers of ERMs that were exceeded increased. These analyses are consistent with the narrative intent of ERMs to indicate an incidence of toxicity of about 50% and ERLs to indicate an incidence of toxicity of about 10%”.*

Ingersoll et al. (2005) compared the effectiveness of ERL, ERM, TEL, PEL and no effect concentrations. They considered the ability of these sediment effects criteria to correctly classify toxicity or no toxicity and the respective abilities to classify non-toxic samples as toxic (Type I error, false positive) or toxic samples as non-toxic (Type II error or false negative). They concluded that ERMs and

ERLs were generally as reliable as PELs and TELs in respectively classifying samples as toxic or non-toxic, but stressed the need to use field generated data, noting the problems with other contaminants in contributing to the observed effect.

In conclusion, the reviews imply that SQGs used in the ANZECC guidelines and in the ERC are useful as triggers for potential toxicity as defined by laboratory toxicity tests to amphipods, but are not sufficiently robust to predict toxicity and non-toxicity in all cases.

### 7.4.3 ERC

Section 7.4.1 and 7.4.2 describe international sediment quality guidelines and their robustness, and these form the basis of the Auckland Council's ERC. The ERC are triggers for further assessment, and can be used to classify Auckland sediment contamination. They have been criticised because the difference between amber and red is arbitrarily based on the relative magnitude of TEL and ERL values, rather than on varying levels of protection (although the outcome – the numerical values - would be approximately the same). Another criticism is that the red ERC are not useful ("too late") and exceedance of amber ERC are sufficient to trigger full assessments. However, whatever the criticism, what is more important is that a considerable body of work has been conducted since they were derived and it is timely that sediment quality guidelines for use in Auckland is reassessed, as they are in this chapter.

### 7.4.4 Benthic Health Model

The Benthic Health Model (BHM) is an Auckland regional model of benthic ecosystem health (Anderson et al. 2006, Hewitt & Ellis 2010). Benthic community "health", as measured by observed ecological assemblages, generally relates to contamination gradients very well, and this forms the basis of the model. In the model, the degree of contamination is a single metric that includes concentrations of Cu, Pb and Zn. Clusters of 5 groups were identified along each gradient (in rank order from 1 = healthy to 5 = polluted). Groups 4 and 5 along the gradients coincide with existing "amber" and "red" sediment quality guidelines of the Environmental Response Criteria ("ERC") (ARC 2004). However, the axes also gave additional resolution and discrimination among healthier sites (groups 1-3). This indicates that there are effects on benthic community when concentrations of Cu, Pb and Zn are below the ERC trigger levels. Note however, that while the benthic community "health" correlates to the contamination gradient, the metals that define this gradient may not necessarily cause the effects on community structure. Benthic community also correlates with other factors, but where possible, these have been "taken out" so that only the relationship between benthic health and contamination gradient are predicted and illustrated.

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

Thrush et al. (2008) analysed density distributions apparent in the natural populations of 46 macrofaunal taxa in Auckland harbours. The distributions were correlated with habitat requirements (sediment particle size), food (sediment organic content) and heavy metal contamination (copper, zinc and lead). The results can be interpreted as a combination of multiple stressors, multiplicative effects, differential responses across habitat gradients, suggesting that interactions may be driven by indirect effects and feedbacks defined by species biology and habitat requirements. These results indicate that the use of field-based species sensitivity distributions (f-SSDs) need to take into account various habitat-related factors in deriving contaminant-specific effects threshold. The multivariate analysis approach used by Hewitt et al. (2009) incorporated methods to incorporate habitat factors into the metal effects calculation for the Auckland intertidal sediments.

The BHM can be used instead of SQG (in whatever format) to determine "effects" which may trigger further actions, such as stormwater treatment etc.

### 7.4.5 Auckland field-based species sensitivity distributions - Cu, Pb and Zn

Most forms of SQGs have been derived from effects data generated from laboratory ecotoxicity bioassays, supplemented with ecology data (Long et al., 1995; MacDonald et al., 2000). As discussed above, there are limitations to these approaches. More recently, there have been attempts to derive SQGs from field-based species sensitivity distributions (f-SSD), utilizing field data on benthic communities and contaminant loadings concurrently measured in sediment samples. Hewitt et al. (2009) used the ARC Benthic Health Model dataset to derive sediment quality guidelines for total Cu, Zn, and Pb, on the basis of field-based species sensitivity distributions for intertidal sites. This work focussed on rare (and hence sensitive) species. Hewitt et al. (2009) highlighted that rare species collectively dominate community structure and can make a substantive contribution to traditional measures of biodiversity. Rare species are also important in maintaining the stability and resilience of ecosystems. Large sediment dwelling animals can make a disproportionately large contribution to ecosystem functions such as deep-bioturbation of sediments, the modification of boundary layer flows, or through providing large food items for fish and other consumers. In addition, the high mobility of many large taxa increases their potential to affect bioturbation and mediate oxygen and nutrient exchanges between the water column and the seabed (Kelly 2009).

Sediment quality guidelines were derived from the threshold concentrations of Cu, Pb, and Zn where statistical modelling predicted a 50% decrease in abundance of 5% of the taxa occurred (i.e., a hazardous concentration for 5% of the species, HC5). The derivation was carried out for each metal using the whole dataset so co-occurrence effects were not filtered out (see Thrush et al. 2008), with statistical attribution of each species having the highest correlative link to a specific metal being a used for that f-SSD derivation (see Hewitt et al. 2009, supplementary information).

The sediment quality guidelines obtained from the analysis of species-sensitivity distributions ranged from 6.5 - 9.3 mg/kg for Cu, 18.8 - 19.4 mg/kg for Pb, and 114 - 118 mg/kg for Zn. The values for Cu and Pb are notable because they were between 35 - 50%, and 61 - 64% of their respective reported TELs (Table 7.2) (Kelly 2009). Cu concentrations are very close to background. Therefore these very low f-SSD pose a conundrum to managers, if taken at face value. This is particularly true when further changes in community composition were apparent below the derived f-SSD values. These included other reductions in the occurrence and/or abundance of rare and large species.

The f-SSD derived by Hewitt et al. (2009) examined the site-specific distributions with only total Cu, Pb and Zn. The effect of particle size was “factored out” by the statistical computations. The authors were limited by the data available, so other potential stressors could not be examined. The very low f-SSD for Cu and Pb (and low value for Zn) may be due to additive metals effects together with multiple stressors (Thrush et al. 2008), and may more accurately reflect the level of disturbance in Auckland urban sediments.

Known disturbance includes substantial increases in sedimentation rates by muds, such that pre-European sediments are buried by as much as 1 m of muds; this has also been accompanied by reduction in particle sizes (a greater areal distribution of mud) and changes in the physical texture of the sediment in and on which the organisms live. Freshwater inflows have increased, in the form of the frequency and size of storms, and have been proposed for the loss of calcareous foraminifera (Hayward et al. 2006, summarised in Williamson & Mills 2009b). Potential impacts (i.e., those not measured so not demonstrated at a field or community scale) include the build-up of ammonia and free sulphide to toxic levels accompanying the increase in muds and organic matter; UV activated PAH toxicity on surface sediments (demonstrated in laboratory experiments)[10], DDT (and DDE, DDD) toxicity because concentrations can exceed trigger values – especially the revised ANZECC trigger values, or other unmeasured but commonly-found toxicants in overseas urban estuaries.

There is anecdotal evidence from field observations for dynamic changes in sediment texture, as mud moves around sheltered estuaries and embayments depending on wind strength and direction, thus changing sediment types from sandy to muddy and back again. Also large areas of Auckland estuaries, notably the Central Waitematā and Tamaki Estuary are sediment starved because their catchments are small and oceanic inputs of sediment are small. This can be observed as thin layers of fine sediment on bedrock (and poses a problem in Auckland Council's sediment quality monitoring programme) which offer very limited habitat for benthic animals – especially large, rare animals.

The f-SSD values have been criticised in the ANZECC revision as not meeting the concordance criteria used in developing other SQG. Furthermore, from a

[10] An earlier assessment (Diffuse Sources 2002) concluded that this is unlikely to be a problem, but this assessment needs updating with more recent data.

chemical point of view, the very low thresholds for Cu and Pb challenge our understanding of bioavailability constraints of heavy metals in sediments, notably adaptation to background levels, adsorption or co-precipitation with FeS, complexation with inorganic anions and dissolved organic matter, and adsorption on particulate organic matter.

The f-SSD has identified clearly and succinctly the impoverished nature of Auckland's urban sediments in respect to rare, large and important species and shown that this is not necessarily associated with relatively high concentrations of individual metal contaminants. However it remains highly uncertain that low concentrations of the total Cu, Pb or Zn have caused this, so we do not recommend their use at the present time as trigger values in the sense of the intent of the ANZECC guidelines.

#### 7.4.6 Toxicity testing

The toxicity of environmental pollutants can be determined by a variety of approaches including:

- laboratory bioassays, in which test organisms are exposed to polluted water, sediments or pore water;
- in-situ tests, where test organisms are exposed to the contaminants in the environment.

Laboratory tests are the most commonly used methods for assessing toxicity to marine test organisms.

Toxicity tests are described in two ways.

1. Acute toxicity, which occurs when the concentration of a pollutant is high enough to cause death or some other adverse effect in a relatively short time (typically, 2 to 10 days for most test organisms) compared to the organism's life span.
2. Chronic toxicity occurs when the pollutant(s) are present in concentrations too low to cause an immediate effect but high enough to have detrimental effects on aquatic living organisms in the long term. Examples of detrimental effects include abnormal development of juveniles, retarded growth rates, reduced reproductive capacity, impaired behaviour (e.g., reduced burrowing rates by shellfish), avoidance behaviour, physical

deformities (e.g., tumours, abnormal shell structure) and physiological stress (which can underlie retarded growth and a reduced capability to reproduce).

An acute toxic response to test organisms indicates a serious problem associated with high levels of one or more chemical contaminants or non-chemical stressors. Chronic toxicity is less immediate, but no less serious, as it indicates stressful conditions that may upset longer-term ecological health.

Chemical analysis may reveal contaminants at high enough levels to be responsible for the observed toxicity, but often this is not the case, and the cause of the toxicity remains unexplained. Toxicity tests are therefore useful "integrative" indicators of the impacts of contaminants or stressors introduced by stormwater contamination, providing "biological effect" information that chemical analysis of suspected causative agents may not be able to provide.

Sediment toxicity testing is widely used in many programmes around the world notably in North America (e.g., see Chapter 11), and is recommended as part of the decision tree when investigating exceedances of ANZECC (2000) guidelines to determine actual risk to ecosystems. Criteria have been developed in terms of laboratory test responses (e.g., the USA National Coastal Condition Assessment uses a 10-day amphipod test – see Chapter 11).

### 7.4.7 Pore water concentrations

Pore water concentrations can be used as a measure of sediment toxicity (Carr & Nipper 2002). The basis for this approach is that toxicity of contaminants dissolved in pore water is similar to their toxicity in overlying water. Pore water concentrations are compared with Water Quality Guidelines (Section 5).

### 7.4.8 Biomarkers

Biomarkers show biochemical and/or physiological changes in an organism following exposure to contaminants. They therefore indicate that organisms have been exposed to a toxicant/stressor, but the response is not necessarily related directly to a toxicity-specific mechanism. They can be very sensitive indicators of sub-lethal ecological effects and an early warning tool, expressing a stress response prior to population health becoming compromised to the point of mortality effects that result in changes to ecological assemblages. They provide both quantitative and qualitative estimates of exposure and physiological effects. They can replace expensive chemical analysis for emerging contaminants and may be a cost effective primary screening tool (Allan et al. 2006) for chemicals that are bioactive at very low concentrations, often below the limits of laboratory detection. Some specific COPCs includes the endocrine disrupting chemicals (see Tremblay et al. 2011 for a recent review), and specific bioassays have been developed for biotic responses. One of the primary standardised bioassays used in Europe uses the New Zealand mud snail (*Potamopyrgus antipodarum*, OECD 2010). This bioassay technique could be used to test the EDC activity of the complex suite of chemicals found in marine sediments around Auckland (Stewart et al. 2014). A brief overview of the status of biomarkers is given in Chapter 10. The techniques look very promising, especially for emerging contaminants (CPEC), but there needs to be considerably more research before they can be applied as criteria in Auckland.

## 7.5 Relevance/Suitability for Auckland

From the above, it can be seen that there are a wide range of techniques for assessing sediment toxicity, and methods have been developed for all of these and trialled in Auckland. So how do we develop criteria (which might include numerical objectives and limits as being recommended for the NPSFM) for Auckland sediments?

To address this, it first must be recognised that Auckland urban sediments have been “well and truly triggered” for Cu, Pb and Zn irrespective of which guidelines are used to make this call. The spatial distribution and rate of change of contamination is very well understood for the major priority pollutants Cu, Pb and Zn (see above) and reasonably well understood for other priority pollutants such as As, PAH, Cd, DDT. The benthic ecology in Auckland’s estuaries is clearly impacted, and impacts are strongly related to levels of Cu + Pb + Zn contamination.

The current approach in Auckland is that management actions have been triggered by the BHM, from monitoring and from model predictions, and so now options are being explored for effective management. Therefore, what is important now are criteria and objectives that address these issues where they have already occurred and apply lessons learned to new development areas, rather than develop trigger values as described in ANZECC (2000). An environmental classification approach and associated limits framework that has regard for historical development may be required to address the different complexities associated with existing environmental issues and ensure most appropriate long-term management decisions are made. Nevertheless, we are still some way from developing such a robust framework.

Setting limits is also complicated by the fact that predicted contaminant build-up in estuary sediments (by urban storm water contaminant models USC1, USC2, and USC3) shows that even with improved stormwater treatment, contaminants will continue to build-up, albeit at a slower rate.

It also must be remembered the primary (first and foremost) principle in the ANZECC guidelines (and reiterated by the Land and Water Forum, 2012) is to decide the level of protection for specific water bodies. This is very important for Auckland sediments because a range of disturbance has been already identified (e.g., highly–, slightly–, or un–disturbed[11]). Consequently, a key consideration for Auckland Council is to make that management decision; this will guide the choice of contaminant criteria approach, and the level of protection required to meet explicit values.

### 7.5.1 Using international guidelines

The first approach involves the use of international sediment quality guidelines as numerical objectives. As argued above, Auckland sediments have been well and truly triggered and investigated, so Auckland Council can choose to use the SQG as "standards" (enabled by the NZCPS) rather than "triggers". In the short term the PEL and TEL can be used. In the longer term, we recommend that the BEDS database (which was used to develop ERL/ERM and TEL/PEL) is revised by inclusion of the Auckland results to derive new TELs and PELs.

Examples of criteria options for different types of receiving environments could include:

- Numerical objectives for older urbanised estuaries in Auckland, which aim to prevent further loss of the resilient animals that currently exist in these

[11] Not as highly disturbed as in some harbours of some industrialised countries nor as pristine as in unpopulated areas of NZ

estuaries. This could be PEL, or PEL quotients. [Mean PEL quotients = $\Sigma$PELs/n; where n= number of contaminants].

- Numerical objectives for newer urbanised estuaries that have/should have incorporated BMPs with the aim of protection of existing animals (likely to be more sensitive and diverse than in older areas). For example, adopting a fraction of mean PEL quotients.
- Numerical objectives for the urbanised Outer Zones, with the aim of preventing these from becoming more contaminated and affecting more sensitive animals in these areas (for example, TELs, adopting a lesser fraction of mean PEL quotient).
- Trigger values for un-urbanised areas – especially rural areas and subtidal urban areas (which have not yet been "triggered") using the TEL values in the short term.

Management, including limits-based modelling would be based on managing inputs of all the individual contaminants defined in the TELs, PELs or PEL quotients.

### 7.5.2 Benthic Health Model

In this case, numerical objectives are set along the Benthic community "health"/pollution axis, and these objectives are set to prevent further loss of the resilient animals in contaminated estuaries, and loss of sensitive animals in uncontaminated estuaries. In this case, the contamination is measured as Cu, Pb and Zn concentrations, but is interpreted as "level" of disturbance.

Management would be based on limiting all contaminants (mud, heavy metals, toxic organics) through source control, treatment, and flow reduction, and Cu, Pb and Zn would be used as surrogates for management performance.

With time and greater understanding, the guidelines would be refined to include other contaminants, or other ways to measure "disturbance".

### 7.5.3 Field-based Site-Specific Species Sensitivity Distributions (f-SSD)

The results of the BHM have been used to develop numerical objectives through Field-based Species Specific Distributions (f-SSD). However, the f-SSD developed for Auckland are so low for Cu (and to some extent Pb), that they prompt more questions. They do not make toxicological sense (see back Section 7.4.5) and may barely exceed background levels in some situations. Current wisdom (authors' unofficial survey) is that benthic health reflects a cumulative response to multi-layered levels of disturbance (see Section 7.4.5), which may include;

- other contaminants (e.g., As, PAH, TPH, DDT, $NH_4$-N),
- past events,
- historical pollution,
- increased freshwater pulses,
- sediment textural effects, and
- sediment dynamics (including pore water deoxygenation);

acting in additive, synergistic or antagonistic combinations in situ. So no clear, "single numerical objective" results.

### 7.5.4 Toxicity testing approaches

Toxicity measurements form the basis of the international sediment quality guidelines, as described above (Section 7.4.6). These international SQG rely on a definition of toxicity, which in many cases is a single-species laboratory-based toxicity measure (e.g., 10-day amphipod survival test) rather than a multi-species field measure that would have greater ecological relevance. Here, if a toxic response is encountered, it is highly likely that the sediments would have had an ecological effect. However, if a non-toxic response is encountered, there is no surety that ecological effects would not have occurred. Therefore the level of ecosystem protection from sediment toxicity testing is not necessarily high.

Pore water toxicity has only been occasionally used in Auckland. Nipper et al. (1998) investigated the effects of contamination in mudflats using the "triad" approach. The sediment triad approach compares the relationship between sediment toxicity, sediment and pore water chemistry, and benthic community structure. Six of the eight study sites were in Auckland (others were at Raglan and Kawhia) − three stations in Tamaki Estuary, two in Manukau Harbour, and one at Okura. Some of the sites were amongst the most contaminated in Auckland. Pore-water toxicity tests were conducted with embryos of the sand dollar (*Fellaster zelandiae*). None of the toxicity tests responded more strongly to sediments or pore waters from contaminated sites than from uncontaminated reference sites. Levels of contamination were still relatively low compared to internationally based sediment quality guidelines, indicating that the tests were not sufficiently sensitive to detect effects.

Direct sediment toxicity testing has been used in Auckland and elsewhere in New Zealand. Toxicity testing has shown that stormwater itself is mildly toxic to marine organisms. However, dilution in the marine receiving environment should rapidly reduce this to non-toxic levels (Hickey et al. 1997). As stormwater sediments are deposited and mixed in estuaries, contaminant concentrations are reduced to only moderate levels, which are possibly too low for consistent responses from toxicity tests. At these levels, the interaction of physical, biological, and chemical processes in marine sediments affects contaminant bioavailability and hence toxicity, and complicates interpreting relationships between toxicity and contaminant concentrations. Toxicity therefore becomes ambiguous (Nipper et al. 1998, Hickey & Martin 2008, and see summaries in Mills & Williamson 2008, Williamson & Mills 2009b, Kelly 2009). Toxicities, as defined by laboratory tests, are unsuitable as criteria for the contaminated estuarine environment around Auckland.

Because of the ambiguous results from toxicity testing in Auckland, we recommend that neither pore water nor whole sediment toxicity be routinely used to develop region-wide sediment quality criteria expressed as standard toxicity results (e.g., sediments not producing a >20% mortality with a test animal). Longer-term chronic tests (>30 day) were able to measure toxic effects at some sites (Hickey and Martin 2008). However, they may be essential for special investigations. For example, managers need to know thresholds for when combinations of muddiness, organic enrichment and eutrophication result in sediments becoming toxic due to build-up of ammonia and sulphide, and/or decrease in pH, in which case use of pore water toxicity testing may be very useful.

### 7.5.5 Pore water concentrations

Pore water concentrations have been measured in a few Auckland sediments and summarized by Williamson & Mills (2009a) and Elwood et al. (2008). The ANZECC trigger value (for 80% protection) is occasionally exceeded for Cu and Zn at the more contaminated sites, but probably never for Pb. In contrast, even the 99% values for minor contaminants Cd and Ni were not exceeded. This suggests that the Cu and Zn contamination of Auckland estuaries is potentially toxic through this mechanism at the more contaminated sites and further work has been recommended (Williamson & Mills 2009a, Williamson & Mills 2009b) to understand the fate and transport of these contaminants, as well as to elucidate toxic mechanisms.

While heavy metal concentrations in Auckland marine sediments are low on the world stage, the evidence that the metals are at least partly responsible for benthic health is supported by these measurements of pore water, as well as by overlying water concentrations and by relatively low acid volatile sulphide (AVS) concentrations (Williamson & Mills 2009a, Elwood et al, 2008, Williamson & Mills 2009b).

Pore water concentrations of ammoniacal-N were measured as part of the sediment monitoring at 27 sites in 2005 (McHugh and Reed 2006) and for the 10 sites for sediment toxicity testing and chemical assessment in the regional discharge monitoring project (Hickey and Martin 2008). The ammoniacal-N concentrations for the 27 sites averaged 0.46 NH4-N/L and ranged 64-fold, with 26% of the sites exceeding the ANZECC (2000) chronic water quality guideline for 95% protection (Appendix B). The pore water ammoniacal-N for 10 regional discharge monitoring sites were higher (average 3.1, range 0.86-5.8 mg NH4-N/L; Hickey and Martin 2008). Increasing ammonia concentrations were correlated with chronic toxicity (30 d tests) to amphipods (survival and reproduction) and juvenile shellfish (morbidity). A weight of evidence (WOE) analysis indicated that multiple chemical contaminants and physical factors contributed to the community biotic responses and toxicity measures, with sediment ammonium levels being a potential contributing factor. Furthermore, they concluded that cause-effect linkages were poorly identified with the routine suite of chemical contaminants (i.e., Cu, Zn, Pb) and that chemical stressors not routinely measured and other factors (e.g., intermittent events) may be adversely affecting community health.

## 7.6 Recommendations

Existing guidelines currently employed by Auckland Council and other regional councils are summarised in the first two lines of Table 7.4.

Two new approaches stand out as possibilities for deriving improved guidelines for the primary pollutants Cu, Zn, Pb, As, PAH, TPH, DDT and Hg (and possibly pore water ammoniacal-N). They are:

1. using international guidelines amended with Auckland data (termed BEDS modified in Table 7.4); or
2. using the Benthic Health Model.

We highly recommend that these two new approaches (Table 7.4) are used to develop guidelines, either individually or together under the revised ANZECC Guidelines 'weight of evidence' (WOE) approach.

As described in Section 7.2, the WOE approach considers multiple lines of evidence (LOEs) to determine whether the contaminants of potential concern (COPCs) are likely to affect ecosystem health. Sediment chemistry (including bioavailability measures), ecotoxicology, bioaccumulation and benthic ecology are general LOEs, but other LOEs may be added on a case-specific basis, in Auckland such as resuspension, mobilisation, solubilisation and dispersal of contaminants (see below). Given the uncertainties described in the forgoing sections, measures of bioavailability and cause/effect linkages would provide robustness to any guidelines.

In proceeding with this approach, it must be remembered the primary (first and foremost) principle in the ANZECC guidelines (and reiterated by the Land and Water Forum, 2012) is to decide the level of protection for specific water bodies. This is very important for Auckland sediments because a range of disturbance has been already identified (e.g., highly–, slightly–, or un–disturbed[12]). Consequently, a key consideration for Auckland Council is to make that management decision; this will guide the contaminant criteria development, and the level of protection required to meet explicit values.

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[12] Not as highly disturbed as in some harbours of some industrialised countries nor as pristine as in unpopulated areas of NZ

The NPSFM uses longer term 'targets'[13] where water quality limits cannot be met at the time a limit is set. This takes on an additional meaning in the application to Auckland urban sediments, where models predict that primary contaminants will continue to build up. Hence targets will need to take this into account. One of the implications of this relates to how the understanding of fate, transport and effects of contaminants has fallen behind the understanding on benthic ecology. Guidelines robustness (numerical objectives, limits) for sediment toxicity would benefit from the following additional Lines of Evidence (LOE).

1. Dispersion processes in Auckland harbours and estuaries, and their inclusion in models. The fact that present models on the fate of catchment derived Cu and Zn (urban storm water contaminant models USC1, USC2, and USC3) always predict increases[14] in concentrations needs to be examined very carefully; it has profound implications for management and for numerical objectives and limits.
2. Sources of contaminants in rural areas such as the Upper Waitematā Harbour, which may be biasing interpretation of the BHM.
3. A key information gap in guideline development is the uncertainty surrounding background concentrations. In a recent review of core information, Williamson & Mills (2009) found an ambiguous picture for background concentrations. Although there was no compelling evidence for elevated background concentrations, the large amount of disparate data from over ~60 cores, different analytical methods, uncertainty over contamination of rural estuaries and uncertainty over whether some cores penetrated and inadvertently sampled underlying bedrock, prevented the construction of a reliable picture of background concentrations. A more systematic analysis of the data is recommended, where the data is subdivided on the basis of methodology, individual study uncertainties are more rigorously assessed, and background data is extracted.

[13] The NPSFM defines target as 'a limit which must be met at a defined time in the future. This meaning only applies in the context of over-allocation.' The NZCPS uses the terms 'zone', 'standard' and 'target' in Policy 7(2) as types of threshold or acceptable limit to change.
[14] Unless there are relatively large dilutions by uncontaminated sediments

4. Developing cause-effect linkages for contaminants by investigating potential toxicity mechanisms and bioavailability though measuring pore water concentrations, AVS/SEM, toxicity testing and up-to-date chemical modelling of marine sediments.

**Table 7.4** - Currently used and recommended guidelines for Auckland Council

| Guideline | Protection levels | Applies to: | Contaminants | Advantages | Disadvantages / Limitations |
|---|---|---|---|---|---|
| Existing numeric hybrid ARC Environmental Targets | Low/Med/High probability of effects | Everywhere | Cu, Pb, Zn | Simplicity<br>In existing planning documents<br>Applied to all sediment types | Controversial<br>Only international database<br>Somewhat arbitrary in derivation<br>Some of weight-of-evidence |
| ANZECC | Low/High effects probability | Everywhere | Inorganics and organics | Simplicity<br>Has credibility through use<br>Applied to all sediment types | Controversial<br>Only international database<br>Somewhat arbitrary in derivation<br>Lack of weight-of-evidence<br>Challenged by Auckland Council |
| Benthic Health Model | Range of percentile protections based on Auckland data | Intertidal | Cu, Pb, Zn | Derived from local species data<br>Species responses relate to local contaminant mixtures<br>Highly relevant<br>Facilitates analysis of sensitivity of various species/community groups | Limited to measured contaminants (Cu, Pb, Zn)<br>Only intertidal sediments<br>No cause-effect relationship (could be caused by contaminants other than Cu, Pb or Zn)<br>Lack of weight-of-evidence |

| Guideline | Protection levels | Applies to: | Contaminants | Advantages | Disadvantages / Limitations |
|---|---|---|---|---|---|
| Use Biological Effects for Sediments (BEDS) database[1] (modified) | As above integrated with international database | Everywhere | Inorganics and organics | Integrates local and international data<br>Is updatable<br>Facilitates analysis of Sensitivity of various species/community groups | Single contaminant approach<br>Lack of weight-of-evidence |
| WOE | As above with thresholds for different receiving water types | Various site-specific environments | Inorganics and organics | Combines multiple lines of evidence<br>Expert consensus approach<br>define different levels of Protection for different land use and estuary types<br>Uses any or all of the above | Increased complexity for application |

[1] MacDonald et al. (1996)

For any other metals, metalloids and organics that are found in the future that need managing, we recommend that plans refer to the possibility of using ANZECC SQGVs in a “catchall” clause. This approach is limited to the metal, metalloid and organic toxicants listed in ANZECC, which does not cover many emerging contaminants. Therefore it may be necessary to include provisions in the catchall clause to adopt interim SQGVs from other guidelines, such as those summarised in NOAA SQUiRTs or based on increases in concentrations from reference site conditions, as in Norway’s Sediment Quality Guidelines (Bakke et al. 2010). As with Zn, Cu and Pb, this may require obtaining robust background concentrations for some metals from a critical review of all core data (see section 7.5 above).

The current Auckland Council programme of surveying emerging contaminants (CPEC) at regular intervals should continue, along with expanding/contracting the list of analyses according to the findings, the development of understanding from overseas studies, monitoring programmes and review, and the improvements in methodology.

Table 7.5 summarises the recommended guideline approaches for Auckland for differing water uses. **Levels of protection** are explicitly included in the numerical objectives or limits.

**Table 7.5** - Recommended Guidelines for Priority Pollutant Toxic Contaminants in sediments

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | Numerical objectives and limits using a Weight of Evidence approach for differing levels of protections/degrees of disturbance and the range of receiving water types, based on either (or both) 1) international sediment quality criteria, 2) Benthic Health Model |
| SG | Gathering/Cultivating Shellfish | See Table 9.3 |
| CR | Contact Recreation | - |
| IA | Industrial Abstraction | - |
| NS | Natural State | No change |
| A | Aesthetic | - |
| C | Cultural | As in AE and Table 9.3 |

# 8.0 Sediment characteristics

Sediment characteristics are very important in terms of benthic habitat, chemical environment for toxicants and nutrients, aesthetics, water clarity, and recreation.

Sediment characteristics criteria are not explicitly covered in RMA Schedule 3. There are no criteria in the Environmental Response Criteria (ARC, 2004) or ANZECC Guidelines.

We do not recommend that guidelines be developed for the following indicators (texture, TOC, RPD and AVS), but highlight their importance in describing the sediment matrix and interpreting other impacts such as sedimentation, nutrient enrichment and sediment toxicity. However, we also do not rule out the possibility that guidelines will be developed for these indicators at some time in the future.

## 8.1 Sediment texture

Sediment texture is a very important characteristic that is almost always monitored in sediment, sediment chemistry and benthic ecology studies.

Sediment texture is a critical parameter determining the health of ecosystems. Changes in sediment texture through anthropogenic disturbance can have a major impact on chemistry and ecology of sediment. It can also have a significant impact on human uses of the coastal margin such as social and amenity value. Changes are largely dealt with under TSS Criteria (Section 3.6), and in terms of guidelines and management, we recommend guidelines or guideline development described for TSS (Section 3.6).

## 8.2 Total organic carbon

### 8.2.1 Overview and importance

Total Organic Carbon (TOC) is a measure of the organic matter that drives sediment diagenetic processes in sediments and consequently the benthic functional groups that inhabit the sediments and the sediment environment. It is not currently managed in the Auckland region, but is used as an estuary condition rating index in other regions.

### 8.2.2 Existing guidelines

There are no standards in RMA Schedule 3, or Environmental Response Criteria (ARC, 2004) or ANZECC (2000). TOC is measured under ANZECC protocols in order to normalise organic contaminants to 1% TOC.

### 8.2.3 Use/applicability/importance

Excessive inputs of particulate organic matter will cause increased anoxia in sediments, and have a profound effect on animals that live there. Highly visible and smelly surface anoxia does not seem to be a problem in Auckland, except under very unusual conditions (Williamson & Wilcock 1994). There may be less obvious impacts such as increased AVS concentrations, less toxic sediment, difficulties in oxygenating burrows etc. It can be an ecological barrier to many sediment-dwelling animals and the deeper sediments may be toxic because of free sulphide, ammonia and low pH. It may develop into an issue with macroalgae and their die-off. It is a useful metric in South Island estuaries along with redox potential discontinuity (RPD; next section). From the point of view of providing additional information to help interpret sediment texture, both heavy metal and toxic organic results, and for normalisation of toxic organic results, it is an important metric for monitoring sediment quality.

The effect of variation in TOC levels on any application of the ANZECC guidelines to Auckland estuaries is that guidelines for sandy sediments will be effectively lower than for muddy sediments (and to those TVs listed in the ANZECC (2000) guideline document, Table 3.5.1) and guidelines for muddy sites may be higher. Sandy sites with TOC levels of (say) 0.5% will have guidelines of half of those listed, while muds (typically 2–3%, but could be up to several % values) will have guideline values of ca. 2–3 times higher than those listed. Under the ANZECC regime, sandy sites are therefore more sensitive to the effects of organic contaminants than muddy sites.

### 8.2.4 Critique/review of existing approaches

TOC is included in the sediment toxicity index used to describe the Coastal Condition in the USA (Chapter 11). There, sediments with TOC < 2% are generally considered as healthy; 2 – 5% as fair and > 5% as poor and unhealthy (USEPA 2008).

Robertson & Steven (2009a) use a 4-point condition rating and give a trigger value for use in Tasman, Wellington and Southland estuaries.

**Table 8.1** - Sediment TOC condition rating (Robertson & Steven 2009a)

| Condition rating | TOC (%) |
|---|---|
| Very good | <1 |
| Good | 1 - 2 |
| Fair | 2 - 5 |
| Poor | >5 |
| Early warning trigger | 1.3 x mean of highest baseline year |

### 8.2.5 Relevance/suitability for Auckland

TOC should be measured when measuring toxic organics in order to normalise concentrations to 1% TOC. Like sediment texture, it is a fundamental descriptor of sediment characteristics, and should be measured in all sediment monitoring programmes. It is not costly to analyse.

### 8.2.6 Recommendations

We recommend TOC concentrations in sediments are measured in all sediment monitoring programmes and assessed using Robertson & Steven (2009), but not managed directly as a guideline trigger value or as a numerical objective. This is because it is difficult to manage TOC by specifying practices or measures in the catchment or coastal region to limit TOC. Instead, effectively managing fine sediment inputs (Section 3.6) and nutrient inputs (Chapter 4) should maintain TOC concentrations at healthy levels.

## 8.3 Redox Potential Discontinuity (RPD) depth

### 8.3.1 Overview

The RPD is the depth of transition from oxygenated yellow-brown sediments near the surface to grey – black sediments at depth. It is a measure of the transition from oxygenating to reducing conditions, and this gradient of colour change, though continuous, is known as the apparent RPD depth when reduced to an average transition point. It can be an ecological barrier to many sediment-dwelling animals and the deeper sediments may be toxic because of free sulphide, ammonia and low pH. The vertical position of these boundaries can vary seasonally and locally by as much as 1 cm $d^{-1}$ in response to organic detrital supply and mixing (due to bioturbation or physically mediated mixing).

### 8.3.2 Existing guidelines

There are no standards in the RMA Schedule 3, or criteria in the Environmental Response Criteria (ARC, 2004) or ANZECC (2000).

### 8.3.3 Use/applicability/importance

A rising RPD will force some animals towards the surface and indicate greater nutrient availability from the sediments. It is related to nutrient and organic enrichment of sediments, and to weed and macroalgal blooms on sediments.

### 8.3.4 Critique/review of existing approaches

Robertson & Stevens (2009a) use a 4 point condition rating and an early warning trigger.

**Table 8.2** - Redox Potential Discontinuity condition rating (Robertson & Steven 2009a)

| Condition rating | Depth of RPD |
|---|---|
| Very good | >10 cm |
| Good | 3 – 10 cm |
| Fair | 1 – 3 cm |
| Poor | <1 cm |
| Early warning trigger | 1.3 x mean of highest baseline year |

### 8.3.5 Relevance/suitability for Auckland

The RPD depth has not been monitored in Auckland. It appears to be a useful metric in South Island estuaries (Tasman, Southland) when assessing estuary condition.

### 8.3.6 Recommendations

We recommend that the Robertson & Stevens' (2009a) system for rating RDP is evaluated for its application and usefulness as an environmental tool in Auckland estuaries. The metric is easy to collect during sediment sampling.

## 8.4 Acid Volatile Sulphide (AVS)

### 8.4.1 Overview

Marine sediments contain relatively high concentrations of ferrous sulphide (FeS), which give the sediment its characteristic black colours. Estuarine sediments contain relatively high concentrations of sulphide from anaerobic metabolism of organic matter.

Other heavy metals react with FeS to form insoluble sulphides (e.g., ZnS, CuS, HgS) because Zn, Cd, Cu and Pb are more insoluble than FeS. The metal sulphides are termed Acid Volatile Sulphides because of the way they are analysed. In uncontaminated sediments, AVS is dominantly FeS.

### 8.4.2 Existing criteria

There are no standards in the RMA Schedule 3, or criteria in the Environmental Response Criteria (ARC, 2004) or ANZECC (2000). AVS is measured under ANZECC protocols as part of the decision tree to determine the real risk of toxic metals in sediments.

### 8.4.3 Use/applicability/importance

The Acid Volatile Sulphide (AVS) model recognises that a number of heavy metals form insoluble precipitates with sulphides in sediments, rendering the heavy metals non-bioavailable. When the concentration of heavy metals ("Simultaneously Extracted Metals" – SEM) exceeds the concentration of sulphides (the "sulphide buffering capacity"), then heavy metals are released into the interstitial water where they are able to exert toxicity (note that the model does not include effects due to ingestion of particles). The model has been well tested and validated in North America (see references in Williamson & Mills 2009a) and in New Zealand for Cd (DeWitt et al. 1999).

### 8.4.4 Critique/review of existing approaches

The approach has not been used in Auckland as far as we are aware. Many overseas studies have found high AVS concentrations and have shown that sediments were not acutely toxic when the heavy metal concentrations were lower than the AVS concentrations.

### 8.4.5 Relevance/suitability for Auckland

Diffuse Sources (2002) reviewed the situation in Auckland sediments. High AVS concentrations can be found in Auckland surface sediments, especially in samples taken over relatively wide depth ranges (e.g., 0-10 cm). Concentrations of heavy metals necessary to exceed such AVS concentrations would have to be very high, and much higher than presently observed in Auckland. However, most studies in Auckland have found substantially lower concentrations in surface intertidal sediments (the top few centimetres). This is because of strong bioturbation, where burrowing and burrow irrigation bring oxygen into the sediments, which in turn oxidises AVS as well as bringing AVS to the surface. Low AVS concentrations are found in sandy sediments.

In the latter cases, the concentrations of AVS are a similar order of magnitude to the sum of the concentrations of Cu, Pb and Zn commonly observed in more polluted estuaries in Auckland. This suggests the possibility of [SEM]>[AVS] and that these sediments are potentially toxic.

Williamson & Mills (2009a) additionally concluded that the AVS/SEM model in the Auckland situation would predict that:

- There is potential toxicity in contaminated Auckland sediments.
- Of the three major metals Cu, Pb and Zn, possibly only Zn is toxic.

- AVS concentrations are low enough to suggest Zn is a concern at some Auckland sites in the present day but will become more of a concern in the future, as Zn levels increase further.

### 8.4.6 Recommendations

AVS has not been measured in routine monitoring, and we see this as a useful tool in understanding toxic mechanisms in Auckland, and in the Weight of Evidence approach promulgated in the revised ANZECC guidelines. Measurement (or estimation) of SEM and measurement of the temporal variability of AVS, in Auckland sediments, would be required to develop this guideline. There is a rapid AVS method developed by NIWA that could be applied, at relatively low costs.

# 9.0 Bioaccumulation

## 9.1 Overview

(Adapted from Mills & Williamson 2008).

Aquatic organisms such as shellfish and fish can accumulate substantial levels of chemical and microbial contaminants when exposed to polluted water and sediment. In the case of microbial contamination, this can lead to these organisms being unfit for human consumption (Chapter 6). While chemical contamination is not generally high enough around Auckland to be a significant, general concern for human consumers of fish or shellfish (Stewart et al. 2016) (exceptions are natural marine biotoxins and some localised contaminated areas), chronic health effects on the aquatic organisms themselves, or on other animals that feed on them, are possible ecological consequences. Some chemicals, such as organochlorine pesticides and PCBs, can bioaccumulate in the tissues of some aquatic organisms and may cause chronic, long-term ecological problems. Species that are likely to accumulate highest levels of contaminants are those that live in contaminated environments, particularly when exposed to polluted sediments e.g., shellfish, snails, bottom-feeding fish, and worms.

Some chemicals are transferred through the food chain, so higher trophic level organisms, in particular birds that feed on contaminated worms, fish, and shellfish, can accumulate high concentrations, and this can cause serious ecological problems (e.g., the infamous egg-shell thinning problems for American birds of prey, caused by exposure to organochlorine pesticides such as DDT). It is worth noting that even modest levels of some contaminants in sediments can lead to biological problems. San Francisco Bay is an example, where a clean-up target of 2 ppb total PCBs in sediments has been put in place to protect bird life from PCBs accumulated through the food chain (Hetzel 2004). Human health risks from bioaccumulation are highly significant in other countries - see Chapter 11 below for the USA National Coastal Condition Report, which has found that 77% of sites throughout the coastal USA have unsatisfactory fish tissue concentrations, mainly due to PCBs, Hg and DDT.

In New Zealand, the potential for tissue bioaccumulation and food-chain transfer of contaminants were major components of the Devonport seabed remediation assessment study (Hickey et al. 2007). This study particularly concentrated on Hg and PCBs which had been identified as major contaminant issues in the marine sediments around the dockyard.

## 9.2 Existing criteria

The RMA Schedule 3 specifies a narrative bioaccumulation standard for SG waters, and they are probably implicit in Class AE and C Waters.

**Table 9.1** - Standards for Bioaccumulation for Water Quality Classes from Schedule 3 RMA

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | The following shall not be allowed if they have an adverse effect on aquatic life:<br>(c) any discharge of a contaminant into water |
| SG | Gathering/Cultivating Shellfish | Aquatic organisms shall not be rendered unsuitable for human consumption by the presence of contaminants |
| CR | Contact Recreation | - |
| IA | Industrial Abstraction | - |
| NS | Natural State | The natural quality of the water shall not be altered. |
| A | Aesthetic | |
| C | Cultural | The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified cultural or spiritual values |

### 9.2.1 Food

The Australia New Zealand Food Standards Code (ANZFS, 2011) prescribes maximum levels for Arsenic (As), Cadmium (Cd), Lead (Pb), Mercury (Hg), polychlorinated biphenyls (PCB), histamine and marine biotoxins in seafoods. Standard 1.4.1, Contaminants and natural toxicants, sets out the maximum levels (MLs) of specified metal and non-metal contaminants and natural toxicants in nominated foods (http://www.foodstandards.gov.au/foodstandards/foodstandardscode.cfm).

The maximum levels for the toxic metals are summarised in Table 9.2. The measurement and management of natural marine biotoxins are beyond the scope of this report (but see Chapter 5 for a summary on Harmful Algal Blooms).

**Table 9.2** - Food standards (mg/kg) for Australia and New Zealand (ANZFS 2011).

| Contaminant | Crustaceans | Fish | Molluscs | Seaweed |
|---|---|---|---|---|
| As | 2 | 2 | 1 | 1 |
| Cd | | | 2 | |
| Pb | | 0.5 | 2 | |
| Hg* | 0.5 | 0.5-1.0 | 0.5 | |

*Two separate maximum levels are imposed for fish — a level of 1.0 mg mercury/kg (as a mean) for the fish that are known to contain high levels of mercury (such as long-lived or large marine species) and a mean level of 0.5 mg/kg for all other species of fish. A mean limit of 0.5 mg/kg is also imposed for crustacea and molluscs. The Australia New Zealand Food Standards Code also specifies a standard based on the number of serves (meals) of different fish that can be safely consumed (FSANZ 2008).

For the other common heavy metals Cr, Cu, Ni and Zn, Turner et al (2005) state “These four heavy metals are environmentally ubiquitous in New Zealand, and their levels are often higher in areas associated with human activity. For this reason, they are commonly included for analysis during heavy metal studies. While toxic to humans at high concentrations, Cu, Zn and probably Cr are essential elements and all are well regulated by the body. For this reason their concentrations in foods are not regulated by the NZFSA and there are no food safety limits in New Zealand.”

### 9.2.2 Ecological effects

There are no criteria to protect aquatic animals from bioaccumulation and biomagnification effects.

## 9.3 Use/applicability/importance

The management of discharges to protect human health is primarily the concern of the Ministry of Health, but clearly of interest to Auckland Council in terms of interpreting environmental information. This can fall under the desirable understanding of knowing more about bioaccumulation of the heavy metals, and also the need to understand reports of human or animal poisoning by marine biotoxins (e.g., the cause is biotoxins and not contaminants that might fall under the Auckland Council’s orbit). In addition, bioaccumulation will probably be part of the decision tree in ANZECC (revised) to determine the real risks of contaminants triggered by exceedances of ANZECC guidelines, as part of the Weight of Evidence (WOE) approach.

## 9.4 Critique/review of existing approaches

Internationally, concentration levels that protect human consumers have decreased in food for some organochlorines in recent years and there has been an increased awareness that some members of the community consume or wish to consume larger and more frequent meals of seafood, which may include parts of fish which bioaccumulate more contaminants (the standard fish advisory assumes consumption of fillets). There is a need to address these issues, and although the primary responsible agency is not Auckland Council, it has a role in advocacy and the need to ensure that any monitoring “fits” with their own monitoring programmes, so that it is able to yield robust insights into bioavailability, bioaccumulation and biomagnification.

The Australia New Zealand Food Standards Code prescribes maximum levels for As, Cd, Pb, Hg, PCB, histamine and marine biotoxins in seafoods. Some oysters are allowed to breach the Cd standard by large amounts, because levels are regarded as “natural”. The standards do not address DDT or dioxins and furans, which have been found to trigger advisory notices around fish consumption in the USA (Chapter 11).

We recommend that maximum consumption recommendations are developed by the Food Safety Council for members of the population that aspire to consume a wider variety of fish and shellfish collected from water bodies within the Auckland Region. To do this, a full Health Risk Assessment (HRA) for food consumption relevant to Maori and other ethnic groups would need to be undertaken. This would involve measuring Hg, Pb, Cd, As, PCB, dioxins and DDT levels in targeted species and assessing the health risk associated with a “food basket” of the same widely utilised species. Concentrations of PCBs and DDTs, while not especially high in terms of toxicity effects (Section 7.3.2) could probably trigger bioavailability studies or even fish advisories in the USA (Section 11.1). HRA could also consider differing risk categories (general population, women of child-bearing age, children) and for realistic levels of consumption (moderate and high consumers), or utilise guidance from the most sensitive for establishing the “guidelines”. Outcomes may be no risk from "normal" consumption levels or the need for guidance to limit consumption. The application and methodology have been developed in the Bay of Plenty region and are proposed for the Waikato River clean up (WRISS 2010). This type of monitoring could be applied to areas identified and classified/zoned for gathering/cultivating shellfish.

Biomarkers show biochemical and/or physiological changes in an organism following exposure to contaminants. They therefore indicate that organisms have been exposed to a toxicant/stressor, but the response is not necessarily related directly to a toxicity-specific mechanism. They can be very sensitive indicators of sub-lethal ecological effects and provide both quantitative and qualitative estimates of exposure (van der Oost et al. 2003). A brief overview of the status of biomarkers is given in Chapter 10. The techniques look very promising, but there needs to be considerably more research before they can be applied as criteria in Auckland.

## 9.5 Relevance/suitability for Auckland

Auckland Council monitoring of resident and deployed shellfish show that chemical contaminants, Zn, Cu and Pb and organic compounds including PAH, OCPs, and PCBs, are accumulated by these biota from the water column, enabling spatial patterns and temporal trends in contamination to be measured. This programme has previously been an important part of Auckland Council’s state of the environment monitoring. By international standards, organic contaminant concentrations in mussel and oyster tissues are low and are unlikely to cause ecological or health effects.

There have been a number of other research studies in Auckland that have measured bioaccumulation, and these have been reviewed by Kelly (2009). However there has been little or no assessment of effects on animals, ecology or human consumers. Of these, the only identifiable effects of bioaccumulation are Pb levels in oyster catchers from Mangere Inlet which might induce chronic toxicity (Thompson and Dowding 1999).

What can be concluded from all these studies is that bioaccumulation can occur with priority contaminants and, as expected, this is consistent with overseas studies, although concentrations in Auckland are generally much lower. There are some indications of potential toxicity to higher animals (oyster catchers, flounder), and while the evidence is not strong, because they are preliminary studies only, ecological effects from bioaccumulation cannot be ruled out.

## 9.6 Conclusions

We are unable to recommend criteria for bioaccumulation/ biomagnification to protect aquatic life. However, we recommend a review/study of the situation for Hg, PCB and DDT accumulation in the local food chain, in order to assess the risk of these contaminants to higher animals, especially human consumers and New Zealand threatened and endangered birds (Table 9.2).

In general, measurement of contaminants in biota may yield some useful information as to whether or not a contaminant is bioavailable. However, such studies need to be conducted skilfully because some contaminants may be bioavailable and toxic, but not bioaccumulate, while some animals may regulate and minimise the bioaccumulation of a toxic chemical. Bioaccumulation is an important component of special investigations into the fate and effects of bioaccumulative toxic contaminants, such as in toxicity studies or in Weight of Evidence approaches.

In terms of human consumers, there are few reports of high risks to human health from accumulation of priority pollutants in aquatic organisms in New Zealand, except for mercury, as noted above. Cadmium levels exceed food safety limits in oysters, but this seems to be a natural phenomenon. This situation could be worsened by the build-up, and subsequent runoff of Cd in pasture soils from superphosphate application (Butler & Timperley 1996). Auckland Council should keep a watch on this situation; perhaps through its liaison with others who are actively monitoring Cd contamination of pasture soils (e.g., Kim 2005).

We recommend that maximum consumption recommendations are developed by the Food Safety Council for members of the population that aspire to consume a wider variety of fish and shellfish collected from water bodies within the Auckland Region. Although not the primary responsible agency, Auckland Council has a role in advocacy and the need to ensure that any monitoring “fits” with their own monitoring programmes, so that results are able to yield robust insights into bioavailability, bioaccumulation and biomagnification.

**Levels of protection** are explicitly included in identifying sensitive segments of the population.

**Table 9.3** - Recommended Guidelines for Bioaccumulation of Priority Pollutant Toxic Contaminants

| Class | Purpose | Criteria |
|---|---|---|
| AE | Aquatic Ecosystem | - |
| SG | Gathering/Cultivating Shellfish | ANZFA (2011) for As, Cd, Pb, Hg, PCB<br>ANZFA (2011) limits of consumption of types of fish and sensitive members of the population |
| CR | Contact Recreation | - |
| IA | Industrial Abstraction | - |
| NS | Natural State | - |
| A | Aesthetic | - |
| C | Cultural | Develop food basket approach to assessing and managing risk from Hg, PCB, DDT, As, Cd, Pb and dioxins in seafoods |

# 10.0 Biomarkers

Biomarker techniques are likely to become more robust in the future and their development will likely be very rapid. Auckland Council should maintain a watch on development of biomarkers by commissioning a progress report from experts approximately every 2 years.

“A biomarker, or biological marker, is in general a substance used as an indicator of a biological state. It is a characteristic that is objectively measured and evaluated as an indicator of e.g., normal biological processes, pathogenic processes, or pharmacologic responses to a therapeutic intervention. It is used in many scientific fields. A biomarker can also be used to indicate exposure to various environmental substances in epidemiology and toxicology. In these cases, the biomarker may be the external substance itself (e.g., heavy metal), or a variant of the external substance processed by the body (a metabolite).” (from Wikipedia).

Biomarkers are proposed as part of more holistic, integrated environmental risk assessment. Biomarkers show biochemical and/or physiological changes in an organism following exposure to contaminants. They therefore indicate that organisms have been exposed to a toxicant/stressor at a concentration sufficient to cause a change in homeostasis and illicit a response, but the response is not necessarily related directly to a toxicity-specific mechanism. They can be very sensitive indicators of sub-lethal ecological effects. They provide both quantitative and qualitative estimates of exposure. They can replace expensive chemical analysis and are a cost effective primary screening tool (Allan et al. 2006), particularly for contaminants whose biological effect concentrations may be very small and below standard laboratory detection limits, for example hormone mimics.

Some biomarkers are specifically associated with the stressor/toxicant’s mode of action and so the magnitude of the biomarker response can be related to magnitude of adverse effect. In this case, they can provide insight into ecological consequences. One of the most well-known examples is endocrine disruption, where biomarkers show androgenic (masculinising) or oestrogenic (feminising) effects. Examples are imposex in gastropods (due to TBT) and vitellogenin induction in male fish and reduced fecundity (due to oestrogens and oestrogen mimics such as nonylphenol in municipal effluent (Hecker & Hollert, 2011).

Biomarkers have been developed and used for decades. However, their inclusion in environmental management is not universally accepted due to issues and uncertainties around sensitivity, practicality, reproducibility, standardisation of methodology and comprehensive quality assurance programmes to ensure compatibility of data and validation that observed biomarker responses are indeed due to stressors of interest (Sanchez & Porchera, 2009; Allan et al., 2006). A fairly recent paper highlights the role of one of the more well developed biomarkers for Endocrine Disrupting Chemicals as “signposts rather than traffic lights” for environmental risk assessment (Hutchinson et al. 2005). Furthermore, notwithstanding their usefulness described above, currently there are no methods to translate biomarker response to management actions (e.g., Hecker & Hollert 2011), so they have limited use on their own, but are promoted in Weight of Evidence approaches (ANZECC 2000; Haggar et al. 2006).

The current status of biomarkers as a monitoring technique has been demonstrated in Auckland estuaries (Diggles et al 2000, Evans et al 2001, Reed et al. 2010). A number of different tests showed biochemical responses that could be related to metal and PAH stressors, but the techniques need further investigations across environments and seasons before becoming useful monitoring tools or forming the basis of guidelines.

Internationally, the use of biomarkers as primary screening tools has emphasised the need for a ‘battery’ of tests on different phyla. However, at this stage, their use may be more appropriately limited to applications to demonstrate exposure in a Weight of Evidence approach, which includes the usual chemical and biological biometrics. Bioaccumulation studies also demonstrate exposure more simply (e.g., Hg, PCBs, DDT) but biomarkers will be necessary where there are a variety of stressors producing an effect e.g., the variety of endocrine disruptors in sewage which are difficult and expensive to measure in biota (Singhal et al. 2009).

Biomarkers have been proposed and are being investigated for various aquatic ecosystem monitoring programmes around the world, notably in Europe and North America. Biomarkers are currently being developed and used in the following monitoring programmes:

- OSPAR[15];
- Joint Assessment and Monitoring Program (JAMP) (see Chapter 11.2);
- International Council for the Exploration of the Seas (ICES);
- European Environment Agency;
- UN Environment Program Mediterranean Plan (UNEP-MED).

[15] OSPAR is the mechanism by which fifteen Governments of the western coasts and catchments of Europe, together with the European Union, cooperate to protect the marine environment of the North-East Atlantic. It started in 1972 with the Oslo Convention against dumping. It was broadened to cover land-based sources and the offshore industry by the Paris Convention of 1974. OSPAR is so named because of the original Oslo and Paris Conventions ("OS" for Oslo and "PAR" for Paris)

# 11.0 Alternative approaches

In addition to the guideline recommendations above, we include a short section on some of the integrated approaches to coastal management. The approaches integrate multiple indicators to construct a comprehensive picture of coastal condition. We have chosen the following approaches because:

1. Some of the methodology developed in these integrated approaches has been used in the development of recommendations for individual parameters in the fore-going chapters.
2. Many of the parameters described in the fore-going are included in these integrated approaches.
3. They illustrate the use of other parameters or indicators of coastal water condition used overseas. These help provide a more comprehensive picture, and include:
   - Megafauna
   - Fishery stocks, bycatch
   - Coastal wetlands
   - Trends in seabird populations
   - Discharge of oils
4. One of these integrated approaches has been developed and applied in New Zealand (Estuarine Vulnerability Assessment – Section 11.3).

## 11.1 USA National Coastal Condition Report

This brief summary of the approach used in the USA National Coastal Condition Report (NCCR) assessment shows an alternative to assess coastal water condition. Some of the methodology is of direct relevance to recommendations in the main text (TOC, water clarity, DO, dissolved nutrients, chlorophyll), but other methodology used in the NCCR may be of future interest.

**Overview of Programme**

The U.S. Environmental Protection Agency (USEPA) reports periodically on the condition of USA coastal waters using nationally consistent monitoring surveys to minimise the problems created by compiling data collected using multiple approaches. The results of these assessments are compiled periodically into a National Coastal Condition Report. This series of reports contains one of the most comprehensive ecological assessments of the condition of USA coastal bays and estuaries. The assessment presented in each report is based on data from more than 2,000 sites.

The NCCR presents three main types of data:

1. coastal monitoring data,
2. offshore fisheries data (the overfishing and overfished status of 688 marine fish and shellfish stocks), and
3. fish consumption advisories, and beach advisories and closures.

**Indices**

The NCCR reports the coastal monitoring data using a system for integrating different metrics in an overall index. A rating is based on five indices of ecological condition:

1. **Water Quality Index**

   The water quality index includes the components, dissolved inorganic nitrogen (DIN), dissolved inorganic phosphorus (DIP), chlorophyll a, water clarity, and dissolved oxygen.

   The approach used to measure and assess water clarity forms the basis for the approach recommended to the AC (see Section 3.5) to establish reference conditions and management trigger guidelines for water clarity in Auckland estuaries, where turbidity is likely to be highly variable. In the USA, water clarity was assessed against regional guidelines for these components. Water clarity varies between the different regions within an estuary, as well as at a single location in an estuary due to tides, storm events, wind mixing, and changes in incident light. The monitoring approach uses a probabilistic design, which can account for this local variability when the results are assessed on larger regional or national scales. In the USA, water clarity also varies naturally among various parts of the nation; therefore, the water clarity indicator is based on a ratio of observed clarity compared to regional reference conditions. The regional reference conditions were determined by examining available data for each of the U.S. regions. Reference conditions are then set for a site.

2. **Sediment toxicity index**

   The NCCR is an example of the use of toxicity in classification and management of coastal waters. The sediment quality index includes sediment toxicity (using 10 day test on amphipods), sediment contaminants (using ERL and ERM) and total organic carbon (TOC) (<2% good, 2-5% fair, >5% poor).

3. **Fish Tissue index**

   The frequency at which mercury, DDT, and PCB exceed fish flesh limits in the USA and the importance of this in assessing the state of their coastal waters, provides a compelling argument to ensure that this is adequately addressed in Auckland Council’s management, especially where such resources are used as a food source. This has been recommended in Chapter 9 above.

## 4. Coastal habitat index

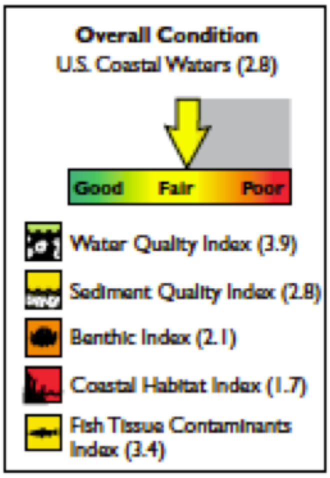
The Coastal Habitat Index is simply the loss of coastal wetlands.

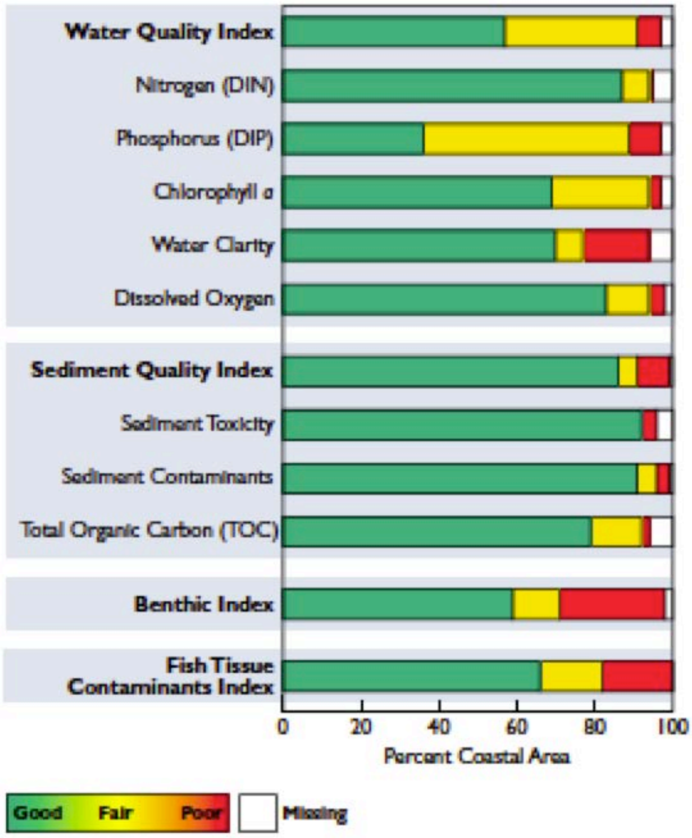
## 5. Benthic index

Regional (Southeast, Northeast, and Gulf coasts) benthic indices of environmental condition reflect changes in benthic community diversity and the abundance of pollution-tolerant and pollution-sensitive species. A high benthic index rating for benthos means that sediment samples taken from a waterbody contain a wide variety of benthic species, as well as a low proportion of pollution-tolerant species and a high proportion of pollution-sensitive species. A low benthic index rating indicates that the benthic communities are less diverse than expected, are populated by more pollution-tolerant species than expected, and contain fewer pollution-sensitive species than expected.

Figure 11.1 shows the overall condition of USA coastal waters in 2001-2 as rated fair; the water quality index is rated fair-good; sediment quality and fish tissue contaminants are rated fair; the benthic index and coastal habitat index are rated poor. A more up-to-date report is available (USEPA 2016).

**Figure 11.1 -** The overall condition of U.S. coastal waters in 2001-02 is rated fair.




## Offshore fisheries data, fish consumption advisories, and beach advisories and closures

Offshore fisheries data, fish consumption advisories, and beach advisories and closures are used in addition to the indices described above to provide a broader perspective of the coastal ecosystem.

For example, in 2002-03 (USEPA 2008), 77% of the coastal waters are under fish consumption advisories; mostly due to four primary contaminants; PCBs; mercury; DDT (and DDD, DDE); and dioxins and furans. 20% of beaches had had an advisory or closing in effect once during the 2003 season, mostly due to microbiological quality.

## 11.2 The European Union Water Framework Directive and OSPAR

The OSPAR[15] programme (e.g., OSPAR 2005) is briefly summarised here for a number of reasons. It shows a number of differences from the programmes currently undertaken by Auckland Council and the classical monitoring programmes such as the USA NCCR (Section 11.1). These are:

- Focus on a different set of hazardous substances, including CPEC. It is likely that significant advances will be made on biomarkers in order to meet the objectives of the programme;
- Ecological objectives highlight known specific problems (e.g., Hg in bird eggs), megafauna (e.g., seals) and big-picture ecosystem objectives (e.g., restore large fish). This gives the programme a higher, more visible and understandable public profile. We recommend that Auckland Council investigates appropriate high-visibility ecosystems approaches in coastal planning.

**Outline of Programme**

The European Union Water Framework Directive is a directive that commits European Union member states to achieve good qualitative and quantitative status of all water bodies (including marine waters up to 1 kilometre from shore) by 2015. It is a framework in the sense that it prescribes steps to reach the common goal rather than adopting the more traditional limit value approach. The OSPAR convention falls under this directive.

OSPAR is the mechanism by which fifteen Governments of the western coasts and catchments of Europe, together with the European Community, cooperate to protect the marine environment of the North-East Atlantic using an ecosystem approach. A suite of five thematic strategies addresses the main threats that it has identified (the Biodiversity and Ecosystem Strategy, the Eutrophication Strategy, the Hazardous Substances Strategy, the Offshore Industry Strategy and the Radioactive Substances Strategy), together with a Strategy for the Joint Assessment and Monitoring Programme, which assesses the status of the marine environment and follows up on implementation of the strategies and the resulting benefits to the marine environment. These six strategies fit together to underpin the ecosystem approach.

**Nutrient Management Objectives**

Of most relevance to the AC are the specific Ecosystem Quality Objectives (EcoQOs) for eutrophication, which are:

a. Winter DIN and/or DIP should remain below a justified salinity-related and/or area-specific % deviation from background not exceeding 50%;

b. Maximum and mean chlorophyll *a* concentrations during the growing season should remain below a justified area-specific % deviation from background not exceeding 50%;

c. Region/area-specific phytoplankton eutrophication indicator species should remain below respective nuisance and/or toxic elevated levels (and there should be no increase in the duration of blooms);

d. Oxygen concentration, decreased as an indirect effect of nutrient enrichment, should remain above area specific oxygen assessment levels, ranging from 4-6 mg oxygen per litre;

e. There should be no kills in benthic animal species as a result of oxygen deficiency and/or toxic phytoplankton species.

**Contaminant Assessment**

The contaminants monitored include Contaminants of Potential Environmental Concern and specific subsets of priority pollutants. These are known to be causing problems (OSPAR focuses on actual rather than potential problems). Monitoring, in the form of repeated measurements of key aspects of the state of the marine environment at key locations, provides the basis for assessing progress towards good environmental status and the evaluation of the effectiveness of actions being taken to protect the sea. The core marine environmental monitoring activity is the OSPAR Co-ordinated Environmental Monitoring Programme (CEMP). The CEMP is currently focussed on monitoring of the concentrations and effects of selected contaminants and nutrients in the marine environment as follows:

- metals (Cd, Hg and Pb) in sediment and biota;
- PAHs in biota and sediment;
- PCBs in biota and sediment;
- brominated flame retardants in biota and sediment;
- the effects of tributyltin in gastropods and concentrations in sediment and/or biota;
- nutrients in sea water;
- eutrophication effects.

Specific components to be monitored after the development of monitoring guidance, quality assurance procedures and/or assessment tools will include:

- planar PCBs in biota;
- alkylated PAHs in biota and sediment;
- TBT in biota;

- Perfluorooctanesulfonic acid (PFOS) in sediment, biota and water;
- dioxins and furans in biota and sediment;
- PAH- and metal-specific biological effects;
- general biological effects.

**Ecological objectives**

Ecological objectives also focus on known problems rather than overall ecosystem health (as is done in Auckland). This approach gives it a high public profile by focusing on megafauna and actual problems. General measures of ecosystem health are being developed. Ecological objectives (EcoQOs) are:

- Safe fish stocks;
- Healthy seal populations;
- Minimise bycatch of harbour porpoise;
- Limiting the input of oil into the sea - low number of guillemots killed by oil;
- Decreasing the impact of TBT containing antifouling paints - imposex in dog whelks and other sea snails;
- Limiting the input of mercury into the marine environment - level of mercury in seabird eggs;
- Limiting the input of organochlorines into the marine environment - level of organochlorines in seabird eggs;
- Diminishing litter in the marine environment: - plastic particles in fulmar stomachs;
- Restore large fish;
- Eutrophication.

Additional issues have been identified for which EcoQOs are currently under development:

- Adequate management of threatened and declining species;
- Adequate sand-eel as prey species for predators – breeding success of black-legged kittiwakes;
- Seabird population trends;
- Healthy benthic communities;
- Restore and/or maintain habitat quality.

## 11.3 Estuarine Vulnerability Assessment (EVA)

The Estuary Vulnerability Assessment (EVA) is a comprehensive evaluation of New Zealand estuarine condition and likely response to stressors. It is packaged in a highly systematic, ecosystem approach and the results presented in an easy to understand way. The approach has been applied in Southland, Tasman and Wellington estuaries (Robertson & Stevens 2006, 2007a, 2007b, 2009a, 2009b, 2010, Stevens & Robertson 2010). The methodology grew out of New Zealand protocols for estuary monitoring (Robertson et al. 2002).

Some of the methodology from EVA has been included in the main text. The overall approach may be of interest to AC in the future.

**Outline of approach**

Ecological Vulnerability Assessment of an estuary involves the application of a tool (adapted from a UNESCO (2000) methodology) used by experts to represent how an estuary ecosystem is likely to react to the effects of potential “stressors” (the causes of estuary issues). The ecological vulnerability assessment reviews current uses and values, physical susceptibility, and existing condition (based on existing data, local knowledge, field observations and expert judgement) before considering how stressors may affect uses and values in relation to the five main problems affecting most New Zealand estuaries (Table 11.1); excessive sedimentation, excessive nutrients, disease risk, toxic contamination, and habitat loss.

**Table 11.1** - Summary of the major issues that may potentially affect most New Zealand estuaries (from Robertson & Stevens 2009a).

| Issue | Impact |
|---|---|
| Sedimentation | If sediment inputs are excessive, an estuary infills quickly with muds, reducing biodiversity and human values and uses. |
| Eutrophication | Eutrophication is an increase in the rate of supply of organic matter to an ecosystem. If nutrient inputs are excessive, the ecosystem experiences macroalgal and/ or phytoplankton blooms, anoxic sediments, lowered biodiversity and nuisance effects for local residents. |
| Disease Risk | If pathogen inputs are excessive, the disease risk from bathing, wading or eating shellfish increases to unacceptable levels. |
| Toxins | If potentially toxic contaminant inputs (e.g., heavy metals, pesticides) are excessive, estuary biodiversity is threatened and shellfish and fish may be unsuitable for eating. |
| Habitat Loss | If habitats (such as saltmarsh) are lost or damaged through drainage, reclamation, building of structures, stock grazing or vehicle access, biodiversity and estuary productivity declines.<br>If the natural terrestrial margin around the estuary is modified by forest clearance or degraded through such actions as roading, stormwater outfalls, property development and weed growth, the natural character is diminished and biodiversity reduced. |

The methodology involves:

1. Assessment of the human and ecological uses and values of an estuary;
2. Assessment of the physical susceptibility of the estuary;
3. Assessment of existing condition;
4. Identification of key “stressors” (the causes of estuary issues - often farming and other land use activities) potentially affecting the estuary;
5. Integration of the above to identify vulnerability to key issues, and the indicators best suited to monitor change in specific stressors.

The output is a transparent assessment of estuary vulnerability, from which management and monitoring priorities can be set.

Stressors include catchment runoff for sediment, nutrients, pathogens and toxicants; point source discharges; sea level rise, climate change effects on rainfall and temperature; spills; grazing; fire; aquaculture; freshwater abstraction; reclamation and drainage; causeways and foodbanks; seafood collection; structures, especially seawalls; invasive pests and weeds; vehicle damage; margin encroachment; floodgates.

Assessing the condition of estuaries to make recommendations for their future monitoring and management is part of the vulnerability assessment and can be undertaken for its own sake. The approach taken is to apply established broad and fine scale estuary “condition ratings” for those major issues facing many New Zealand estuaries (Table 11.1). Details on the methodology and condition rating can be found in many of the reports prepared by Wriggles Coastal Management (e.g., Robertson & Stevens 2009a, Stevens & Robertson 2010). Figure 11.2 and Table 11.2 summarises some of the main metrics and how these are measured.

**Figure 11.2** - Assessing the condition of estuaries

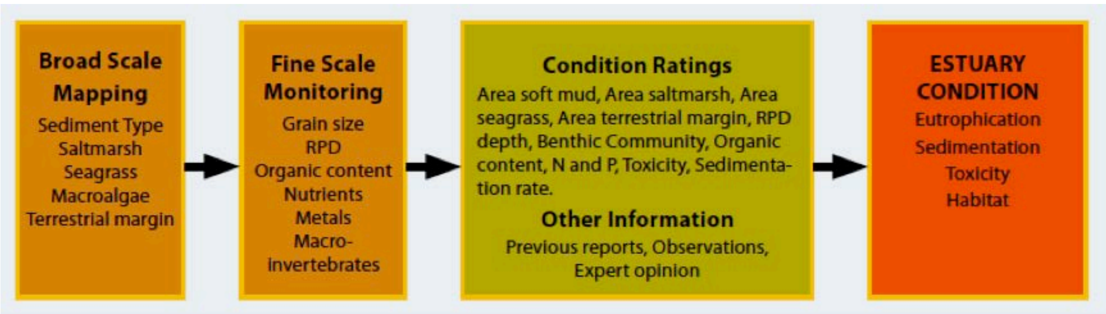


**Table 11.2** - Summary of the major broad and fine scale indicators used to assess estuary condition (adapted from Robertson & Stevens 2009a).

| Issue | Indicator | Method |
|---|---|---|
| Sedimentation | Soft Mud Area | Broad scale mapping - estimates the area and change in soft mud habitat over time. |
| Sedimentation | Sedimentation Rate | Fine scale measurement of sediment deposition. |
| Eutrophication | Nuisance Macroalgal Cover | Broad scale mapping - estimates the change in the area of nuisance macroalgal growth (e.g., sea lettuce (*Ulva*), *Gracilaria* and *Enteromorpha*) over time. |
| Eutrophication | Organic and Nutrient Enrichment | Chemical analysis of total nitrogen, total phosphorus, and total organic carbon (calculated from ash free dry weight) in replicate samples from the upper 2cm of sediment. |
| Eutrophication | Redox Profile | Measurement of depth of redox potential discontinuity profile (RPD) in sediment estimates likely presence of deoxygenated, reducing conditions. |
| Toxins | Sediment Contaminants | Chemical analysis of indicator metals (total recoverable cadmium, chromium, copper, nickel, lead and zinc) in replicate samples from the upper 2 cm of sediment. |
| Toxins, Eutrophication, Sedimentation | Biodiversity of Bottom Dwelling Animals | Type and number of animals living in the upper 15 cm of sediments (infauna in $0.0133m^2$ replicate cores), and on the sediment surface (epifauna in $0.25m^2$ replicate quadrats). |
| Habitat Loss | Saltmarsh Area | Broad scale mapping - estimates the area and change in saltmarsh habitat over time. |
| Habitat Loss | Seagrass Area | Broad scale mapping - estimates the area and change in seagrass habitat over time. |
| Habitat Loss | Vegetated Terrestrial Buffer | Broad scale mapping - estimates the area and change in buffer habitat over time. |
| Physical Susceptibility | Dilution and flushing potential | Based on freshwater inputs, tidal range and estuary area |
| Human Health | Enterococci and faecal coliforms | MoH/MfE Indicators |

Apart from Habitat Loss above, additional Habitat Assessment involves broad-scale assessments on a number of other estuarine and coastal edge habitat measures including:

- Sediment texture (see Chapter 8);
- Litter;
- Macroalgae cover and change;

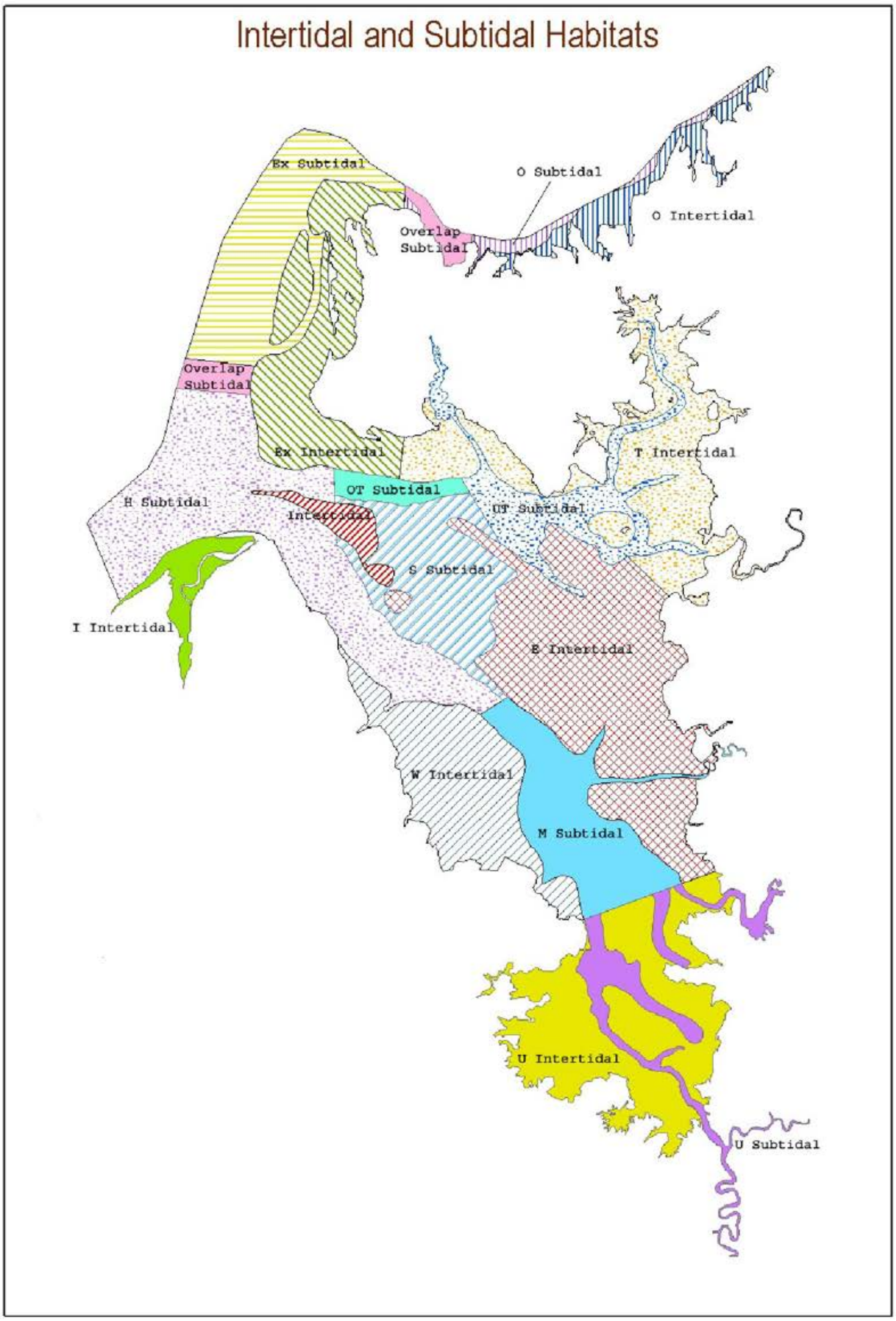
- Mangrove cover and change;
- Toxic algae and slugs;
- Coastal edge land use;
- Structures;
- Fisheries;
- Shellfish resources;
- Birds.

Robertson & Stevens (2009a) have developed conditional rating assessment and trigger values (for further evaluation and management) for the different vegetation covers. For fisheries, shellfish resources and birds, information is obtained from surveys: guidelines could be developed reflecting community aspirations for these. For other stressors (such as structures, drainage, reclamation), they assess the impact of these stressors on the coastal edge habitat, rather than specify numerical/narrative guidelines.

## 11.4 Habitat mapping

EVA addresses habitat loss, however, habitat mapping provides a more comprehensive picture of the coastal resources the council is managing. Habitat maps indicate spatial extent of estuarine sediment types and locations of extensive shellfish populations (e.g., cockles) and other filter-feeding and deposit-feeding fauna, together with key ecosystem components such as seagrass beds and mangroves. Such information is essential to the design of monitoring programmes and for understanding both direct contaminant effects and food-chain transfer of contaminants to higher trophic levels, including humans. Such mapping is undertaken around Auckland, for example biota habitats of the southern Kaipara Harbour (Hewitt & Furnell 2005, Figure 11.3).

**Figure 11.3 -** Example of habitat map from the Kaipara Harbour. On the basis of the distinct differences between the types of fauna and communities found in the Southern Kaipara Harbour, the harbour was divided into 7 intertidal and 8 subtidal habitat areas (Hewitt & Furnell (2005).



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# 13.0 Glossary

| Acronym | Definition |
|---|---|
| A | Aesthetic (Water class RMA) |
| AC | Auckland Council |
| AE | Aquatic Ecosystem (Water class RMA) |
| ANZECC | Australian and New Zealand Environment and Conservation Council |
| ANZFS | Australia New Zealand Food Standards |
| AVS | Acid Volatile Sulphur |
| BEDS | Biological Effects for Sediments (BEDS) database (MacDonald et al. (1996) |
| BHM | Benthic Health Model |
| BOD | Biochemical oxygen demand |
| BLM | Biotic Ligand Model |
| C | Cultural (Water class RMA) |
| CEPC | Chemicals of Emerging Potential Concern (also ECC) |
| CCC | Criterion Continuous Concentration (US EPA chronic criterion) |
| CCME | Canadian Council of Ministers for the Environment |
| CMC | Criterion Maximum Concentration (US EPA acute criterion) |
| COPC | Contaminant of potential concern (ANZECC 2000) |
| CR | Contact Recreation (Water class RMA) |
| DDT | dichlorodiphenyltrichloroethane (widely-used organochlorine pesticide, now banned) |
| DO | Dissolved oxygen |
| DOC | Dissolved organic carbon |
| ECC | Emerging Chemicals of Concern (also CEPC) |
| ERC | Environmental Response Criteria (ARC 2002) |
| ERL | Effects Range Low (Long et al. 1995) |
| ERM | Effects Range Medium (Long et al. 1995) |
| EVA | Estuary Vulnerability Assessment |
| FC | Faecal coliform |
| f-SSD | field-based species sensitivity distributions |
| HMW PAH | High Molecular Weight PAH |
| HRA | Health Risk Assessment |
| ISQG-Low | Interim Sediment Quality Guideline-Low (ANZECC 2000) |
| ISQG-High | Interim Sediment Quality Guideline-High (ANZECC 2000) |
| ISQG | Interim Sediment Quality Guideline for Canadian Council of |

| | |
|---|---|
| CCME | Ministers for the Environment (CCME 1999). |
| IA | Industrial Abstraction (Water class RMA) |
| LOE | Lines of Evidence (ANZECC 2000) |
| MfE | Ministry for Environment (NZ) |
| MoH | Ministry of Health (NZ) |
| MPN | Most Probable Number (microbiological measure) |
| NOF | National Objectives Framework |
| NOM | Natural Organic Matter |
| NS | Natural State (Water class RMA) |
| MSP | Marine Spatial Planning |
| NCCR | National Coastal Condition Report (USA) |
| NPS | National Policy Statement |
| NPSFM | National Policy Statement for Freshwater Management |
| NRWQN | National River Water Quality Network |
| NTU | Nephelometric Turbidity Units |
| NZCPS | New Zealand Coastal Policy Statement |
| PAH | Polycyclic aromatic hydrocarbons |
| PAUP | Proposed Auckland Unitary Plan |
| PCB | Poly-chlorinated biphenyl (widely-used organochlorine chemicals, now banned) |
| PEL | Probable Effects Levels (MacDonald 1996) |
| PNEC | Proposed No Effect Concentration |
| PP | Priority Pollutants |
| QMRA | Quantitative Microbiological Risk Assessment |
| RDP | Redox Potential Discontinuity |
| RMA | Resource Management Act |
| SDR | Sediment Deposition Rate (rivers) |
| SEM | Simultaneous Extractable Metals (associated with AVS) |
| SG | Gathering/cultivating shellfish (Water class RMA) |
| SOD | Sediment Oxygen Demand |
| SoE | State of the Environment |
| SQG | Sediment Quality Guideline |
| SQGV | Sediment Quality Guideline Value (revised ANZECC) |
| TBT | Tributyl Tin (antifoulant) |
| TEL | Threshold Effects Level (MacDonald 1996) |
| TLI | Trophic level index |
| TOC | Total organic carbon |
| TPH | Total petroleum hydrocarbons |

| | |
|---|---|
| TSS | Total suspended solids |
| TV | Trigger value |
| US EPA | United States Environmental Protection Agency |
| WER | Water effect ratio |
| WETT | Whole Effluent Toxicity Testing |
| WOE | Weight of Evidence (ANZECC 2000) |
| WQC | Water Quality Criteria |
| WWTP | Waste Water Treatment Plant |

| Term | Definition (reference source) |
|---|---|
| Attribute | Defined under the NPSFM as a measurable characteristic of fresh water, including physical, chemical and biological properties, which supports particular values (MfE 2014a) |
| Attribute state | Defined under the NPSFM as the level to which an attribute is to be managed for those attributes specified in Appendix 2 of MfE (2014a) |
| Condition rating | Guidelines developed by Robertson & Steven (2007 - 2010) to describe an estuary's condition (e.g., sediment type, nutrient enrichment, macrophytes) |
| Contaminant | Biological (e.g., bacterial and viral pathogens) or chemical (e.g., toxicants) introductions capable of producing an adverse effect in a water body (ANZECC 2000). |
| Crossover issue | Relates to a parameter's numeric objective or load, which is established in freshwater and its applicability to an estuarine environment. A crossover issue may relate to various physico-chemical factors (e.g., pH, depth, lower marine guideline) or multiple parameters required for management (e.g., muddiness, light, sediment nutrients for estuarine macrophytes). |
| Ecological health | Indicates the preferred state of sites that have been modified by human activity, ensuring that their ongoing use does not degrade them for future use (Karr 1999) |
| Ecological integrity | The degree to which the physical, chemical and biological components (including composition, structure and process) of an ecosystem and their relationships are present, functioning and maintained close to a reference condition reflecting negligible of minimal anthropogenic impacts." This means full integrity is attained when human actions have little or no influence on sites. This definition distinguishes ecological integrity from "ecosystem health", which assesses the state of an ecosystem in terms of the stresses put on it, and its ability to keep providing products and processes for both economic and ecological means. (Schallenberg et al. 2011). |
| Guideline (water or sediment quality) | Numerical concentration limit or narrative statement recommended to support and maintain a designated water use (ANZECC 2000). Includes triggers, criteria, standards, targets, numerical objectives. |
| Hardness | The concentration of all metallic cations, except those of the alkali metals, present in water. In general, hardness is a |

| Term | Definition (reference source) |
|---|---|
| | measure of the concentration of calcium and magnesium ions in water and is frequently expressed as mg/L calcium carbonate equivalent. (ANZECC 2000) |
| Limit | Limit has both a general meaning (maximum or minimum value) as well as a specific meaning under the NPSFM. The term limit is defined in the NPSFM as “the maximum amount of resource use available, which allows a freshwater objective to be met”. (MfE 2014a). The NPSFM Implementation Guide expands on the above definition by stating that a limit is a specific quantifiable amount. The NPSFM Implementation Guide gives an example of a maximum contaminant load for a water quality limit. The Implementation Guide says this would be a “common type of limit”, but does not suggest that this is the only type of limit. However, it does not give examples of what other types of limits might be. |
| (Management) Objective | Describes the intended environmental outcomes(s) (definition from National Policy Statement for Freshwater Management). Freshwater objectives are set in regional planning documents and describe the desired state of the water body, having taken into account all desired values. |
| Pollution | The introduction of unwanted components into waters, air or soil, usually as result of human activity; e.g., hot water in rivers, sewage in the sea, oil on land. (ANZECC 2000) |
| Potential conflict | Relates to a parameter’s numeric objective or load, which is established in freshwater and its applicability to an estuarine environment. A potential conflict may occur for that parameter if it is relevant to the values (uses) within the estuary. Some potential conflicts may not occur because of low marine sensitivity (e.g., water nitrate toxicity) or the use cannot occur (e.g., drinking water). The assessment is made without consideration of dilution/dispersion/flushing which may occur in a specific estuary. |
| Reference condition | An environmental quality or condition that is defined from as many similar systems as possible and used as a benchmark for determining the environmental quality or condition to be achieved and/or maintained in a particular system of equivalent type. (ANZECC 2000) |
| Secondary | Means people’s contact with fresh water that involves only |

| Term | Definition (reference source) |
|---|---|
| contact | occasional immersion and includes wading or boating (except boating where there is high likelihood of immersion). (MfE 2014a). |
| Standard (water quality) | An objective that is recognised in enforceable environmental control laws of a level of government. |
| State | State has both a general meaning (form, physical stage) as well as a specific meaning under the NPSFM. In the latter case it means a range in the level of an attribute that may be described as a narrative or numerically. Four different states are specified for attributes (A, B, C or D). The term ‘band’, instead of the term ‘state’, was previously used in the development of the National Objectives Framework. |
| Target | Under NPSFM, is a limit that must be met at a defined time in the future. This meaning only applies in the context of over-allocation. (MfE 2014a) |
| Threshold | A term used in denote guidelines in the ARC Environmental Targets (ARC 2002)<br>Also used in the NZCPS as an overall term which includes ‘standards’ or ‘targets’, viz., “Where practicable, in plans, set thresholds (including zones, standards or targets), or specify acceptable limits to change, to assist in determining when activities causing adverse cumulative effects are to be avoided”. |
| Toxicant | A chemical capable of producing an adverse response (effect) in a biological system, seriously injuring structure or function or producing death. Examples include pesticides, heavy metals and biotoxins (i.e., domoic acid, ciguatoxin and saxitoxins). (ANZECC 2000) |
| Toxicity | The inherent potential or capacity of a material to cause adverse effects in a living organism. |
| Trigger value (TV) | These are the concentrations (or loads) of the key performance indicators measured for the ecosystem, below which there exists a low risk that adverse biological (ecological) effects will occur. They indicate a risk of impact if exceeded and should ‘trigger’ some action, either further ecosystem specific investigations or implementation of management/remedial actions. (ANZECC 2000) |
| Uses | The uses for a water body – equivalent to values. |
| Values | Under the NPSFM, is any national value, and any value in |

| Term | Definition (reference source) |
|---|---|
| | relation to fresh water that a regional council identifies as appropriate for regional or local circumstances (including any use value). Values are equivalent to uses. |
| Water quality criteria | Scientific data evaluated to derive the recommended quality of water for various uses (ANZECC 2000) |
| Water quality objective | A numerical concentration limit or narrative statement that has been established to support and protect the designated uses of water at a specified site. It is based on scientific criteria or water quality guidelines but may be modified by other inputs such as social or political constraints. (ANZECC 2000) |

# Appendix A New Zealand Coastal Policy Statement policies

(Source: Department of Conservation, 2010)

**Policy 21: Enhancement of water quality**

Policy 21 relates to the overall management of issues with the following requirements.

Where the quality of water in the coastal environment has deteriorated so that it is having a significant adverse effect on ecosystems, natural habitats, or water based recreational activities, or is restricting existing uses, such as aquaculture, shellfish gathering, and cultural activities, give priority to improving that quality by:

a. identifying such areas of coastal water and water bodies and including them in plans;
b. including provisions in plans to address improving water quality in the areas identified above;
c. where practicable, restoring water quality to at least a state that can support such activities and ecosystems and natural habitats;
d. requiring that stock are excluded from the coastal marine area, adjoining intertidal areas and other water bodies and riparian margins in the coastal environment, within a prescribed time frame; and
e. engaging with tangata whenua to identify areas of coastal waters where they have particular interest, for example cultural sites, wāhi tapu, other taonga, and values such as mauri, and remedying, or, where remediation is not practicable, mitigating adverse effects on these areas and values.

**Policy 22: Sedimentation**

Policy 22 specifically relates to sedimentation in the coastal environment, with the following requirements:

1. Assess and monitor sedimentation levels and impacts on the coastal environment.
2. Require that subdivision, use, or development will not result in a significant increase in sedimentation in the coastal marine area, or other coastal water.
3. Control the impacts of vegetation removal on sedimentation including the impacts of harvesting plantation forestry.
4. Reduce sediment loadings in runoff and in storm water systems through controls on land use activities.

**Policy 23: Discharge of contaminants**

Policy 23 of the New Zealand Coastal Policy Statement (2010) specifically relates to discharge of contaminants to the coastal environment, with the following requirements:

3. In managing discharges to water in the coastal environment, have particular regard to:
   a. the sensitivity of the receiving environment;
   b. the nature of the contaminants to be discharged, the particular concentration of contaminants needed to achieve the required water quality in the receiving environment, and the risks if that concentration of contaminants is exceeded;
   c. and the capacity of the receiving environment to assimilate the contaminants; and:
   d. avoid significant adverse effects on ecosystems and habitats after reasonable mixing;
   e. use the smallest mixing zone necessary to achieve the required water quality in the receiving environment; and
   f. minimise adverse effects on the life-supporting capacity of water within a mixing zone.
4. In managing discharge of human sewage, do not allow:
   a. discharge of human sewage directly to water in the coastal environment without treatment; and
   b. the discharge of treated human sewage to water in the coastal environment, unless:
      I. there has been adequate consideration of alternative methods, sites and routes for undertaking the discharge; and
      II. informed by an understanding of tangata whenua values and the effects on them.
5. Objectives, policies and rules in plans which provide for the discharge of treated human sewage into waters of the coastal environment must have been subject to early and meaningful consultation with tangata whenua.
6. In managing discharges of stormwater take steps to avoid adverse effects of stormwater discharge to water in the coastal environment, on a catchment by catchment basis, by:
   c. avoiding where practicable and otherwise remedying cross contamination of sewage and stormwater systems;
   d. reducing contaminant and sediment loadings in stormwater at source, through contaminant treatment and by controls on land use activities;
   e. promoting integrated management of catchments and stormwater networks; and
   f. promoting design options that reduce flows to stormwater reticulation systems at source.
7. In managing discharges from ports and other marine facilities:

a. require operators of ports and other marine facilities to take all practicable steps to avoid contamination of coastal waters, substrate, ecosystems and habitats that is more than minor;
b. require that the disturbance or relocation of contaminated seabed material, other than by the movement of vessels, and the dumping or storage of dredged material does not result in significant adverse effects on water quality or the seabed, substrate, ecosystems or habitats;
c. require operators of ports, marinas and other relevant marine facilities to provide for the collection of sewage and waste from vessels, and for residues from vessel maintenance to be safely contained and disposed of; and
d. consider the need for facilities for the collection of sewage and other wastes for recreational and commercial boating.

# Appendix B Pore water ammonia analysis undertaken for ARC 2005 sediment monitoring programme

Source: (McHugh and Reed 2006)

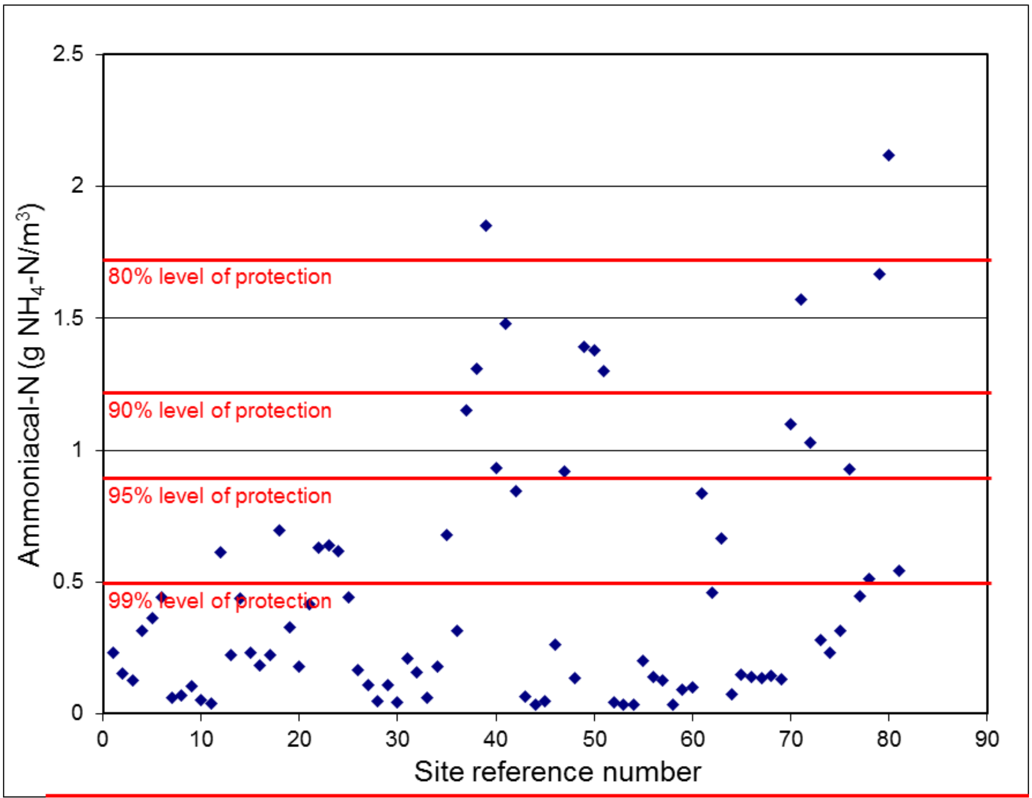
**Table B1 -** Sediment pore water analysed for ammoniacal-N concentrations following method described in Hickey and Martin (2008). Shading indicates exceedance of ANZECC (2000) water quality guideline for 95% protection

| Ref No. | Site | Rep | $NH_4$-N ($mg/m^3$) | $NH_4$-N ($g/m^3$) |
|---|---|---|---|---|
| 1 | Whau WA | 1 | 230 | 0.23 |
| 2 | Whau WA | 2 | 154 | 0.154 |
| 3 | Whau WA | 3 | 127 | 0.127 |
| 4 | Whau Upper | 1 | 314 | 0.314 |
| 5 | Whau Upper | 2 | 362 | 0.362 |
| 6 | Whau Upper | 3 | 443 | 0.443 |
| 7 | Whau Lower | 1 | 59 | 0.059 |
| 8 | Whau Lower | 2 | 71 | 0.071 |
| 9 | Whau Lower | 3 | 105 | 0.105 |
| 10 | Henderson | 1 | 52 | 0.052 |
| 11 | Henderson | 2 | 41 | 0.041 |
| 12 | Henderson | 3 | 613 | 0.613 |
| 13 | Big Muddy | 1 | 223 | 0.223 |
| 14 | Big Muddy | 2 | 439 | 0.439 |
| 15 | Big Muddy | 3 | 230 | 0.23 |
| 16 | Te Matuku | 1 | 182 | 0.182 |
| 17 | Te Matuku | 2 | 222 | 0.222 |
| 18 | Te Matuku | 3 | 697 | 0.697 |
| 19 | Tamaki | 1 | 328 | 0.328 |
| 20 | Tamaki | 2 | 180 | 0.18 |
| 21 | Tamaki | 3 | 416 | 0.416 |
| 22 | Pahurehure | 1 | 629 | 0.629 |
| 23 | Pahurehure | 2 | 638 | 0.638 |
| 24 | Pahurehure | 3 | 616 | 0.616 |
| 25 | Vaughan | 1 | 442 | 0.442 |
| 26 | Vaughan | 2 | 165 | 0.165 |
| 27 | Vaughan | 3 | 110 | 0.11 |
| 28 | Lucas | 1 | 48 | 0.048 |
| 29 | Lucas | 2 | 107 | 0.107 |
| 30 | Lucas | 3 | 42 | 0.042 |
| 31 | Motions | 1 | 209 | 0.209 |
| 32 | Motions | 2 | 158 | 0.158 |
| 33 | Motions | 3 | 61 | 0.061 |
| 34 | Weiti | 1 | 179 | 0.179 |
| 35 | Weiti | 2 | 677 | 0.677 |
| 36 | Weiti | 3 | 315 | 0.315 |
| 37 | Mangere Inlet | 1 | 1150 | **1.15** |

| Ref No. | Site | Rep | $NH_4$-N ($mg/m^3$) | $NH_4$-N ($g/m^3$) |
|---|---|---|---|---|
| 38 | Mangere Inlet | 2 | 1310 | **1.31** |
| 39 | Mangere Inlet | 3 | 1850 | **1.85** |
| 40 | Medla | 1 | 932 | **0.932** |
| 41 | Medla | 2 | 1480 | **1.48** |
| 42 | Medla | 3 | 844 | 0.844 |
| 43 | Browns Bay | 1 | 65 | 0.065 |
| 44 | Browns Bay | 2 | 36 | 0.036 |
| 45 | Browns Bay | 3 | 49 | 0.049 |
| 46 | Oakley | 1 | 264 | 0.264 |
| 47 | Oakley | 2 | 921 | **0.921** |
| 48 | Oakley | 3 | 136 | 0.136 |
| 49 | Pukaki | 1 | 1390 | **1.39** |
| 50 | Pukaki | 2 | 1380 | **1.38** |
| 51 | Pukaki | 3 | 1300 | **1.3** |
| 52 | Cheltenham | 1 | 43 | 0.043 |
| 53 | Cheltenham | 2 | 33 | 0.033 |
| 54 | Cheltenham | 3 | 36 | 0.036 |
| 55 | Kaipatiki | 1 | 199 | 0.199 |
| 56 | Kaipatiki | 2 | 138 | 0.138 |
| 57 | Kaipatiki | 3 | 128 | 0.128 |
| 58 | Hobson | 1 | 34 | 0.034 |
| 59 | Hobson | 2 | 90 | 0.09 |
| 60 | Hobson | 3 | 99 | 0.099 |
| 61 | Puhinui | 1 | 837 | 0.837 |
| 62 | Puhinui | 2 | 460 | 0.46 |
| 63 | Puhinui | 3 | 666 | 0.666 |
| 64 | Te Tokaroa | 1 | 73 | 0.073 |
| 65 | Te Tokaroa | 2 | 147 | 0.147 |
| 66 | Te Tokaroa | 3 | 140 | 0.14 |
| 67 | Awaruku | 1 | 135 | 0.135 |
| 68 | Awaruku | 2 | 144 | 0.144 |
| 69 | Awaruku | 3 | 131 | 0.131 |
| 70 | Ann's Creek | 1 | 1100 | **1.1** |
| 71 | Ann's Creek | 2 | 1570 | **1.57** |
| 72 | Ann's Creek | 3 | 1030 | **1.03** |
| 73 | Paremoremo | 1 | 279 | 0.279 |
| 74 | Paremoremo | 2 | 231 | 0.231 |
| 75 | Paremoremo | 3 | 314 | 0.314 |
| 76 | Pakuranga Lower | 1 | 929 | **0.929** |
| 77 | Pakuranga Lower | 2 | 447 | 0.447 |
| 78 | Pakuranga Lower | 3 | 511 | 0.511 |
| 79 | Pakuranga Upper | 1 | 1670 | **1.67** |
| 80 | Pakuranga | 2 | 2120 | **2.12** |

| Ref No. | Site | Rep | $NH_4$-N (mg/m$^3$) | $NH_4$-N (g/m$^3$) |
|---|---|---|---|---|
| | Upper | | | |
| 81 | Pakuranga Upper | 3 | 541 | 0.541 |

**Figure B1**: Sediment pore water ammoniacal-N concentrations compared with ANZECC (2000) protection thresholds for pH 8.0.



# Appendix C Comparison of sediment quality guidelines.

| Contaminant | NOAA Guidelines[a] | | EC Guidelines (marine)[b] | | ANZECC 2000 Guidelines[c] | |
|---|---|---|---|---|---|---|
| | ERL | ERM | TEL | PEL | ISQG-Low | ISQG-High |
| **METALS** (mg/kg dry wt) | | | | | | |
| Antimony | | | | | 2 | 25 |
| Cadmium | 1.2 | 9.6 | 0.68 | 4.21 | 1.5 | 10 |
| Chromium | 81 | 370 | 52.3 | 160 | 80 | 370 |
| Copper | 34 | 270 | 18.7 | 108 | 65 | 270 |
| Lead | 46.7 | 218 | 30.2 | 112 | 50 | 220 |
| Mercury | 0.15 | 0.71 | 0.13 | 0.7 | 0.15 | 1 |
| Nickel | 20.9 | 51.6 | 15.9 | 42.8 | 21 | 52 |
| Silver | 1 | 3.7 | 0.73 | 1.77 | 1 | 3.7 |
| Zinc | 150 | 410 | 124 | 271 | 200 | 410 |
| **METALLOIDS** (mg/kg dry wt) | | | | | | |
| Arsenic | 8.2 | 70 | 7.24 | 41.6 | 20 | 70 |
| Tributyltin (μg Sn/kg dry wt.) | | | | | 5 | 70 |
| **ORGANICS** (μg/kg dry wt) | | | | | | |
| Acenapthene | 16 | 500 | 6.71 | 88.9 | 16 | 500 |
| Acenaphthalene | 44 | 640 | 5.87 | 128 | 44 | 640 |
| Anthracene | 85 | 1100 | 46.9 | 245 | 85 | 1100 |
| Fluorene | 19 | 540 | 21.2 | 144 | 19 | 540 |
| Naphthalene | 160 | 2100 | 34.6 | 391 | 160 | 2100 |
| Phenanthrene | 240 | 1500 | 86.7 | 544 | 240 | 1500 |
| Low Molecular Weight PAHs | 552 | 3160 | 312 | 1442 | 552 | 3160 |
| Benzo(a) anthracene | 261 | 1600 | 74.8 | 693 | 261 | 1600 |
| Benzo(a)pyrene | 430 | 1600 | 88.8 | 763 | 430 | 1600 |
| Dibenzo(a,h) anthracene | 63.4 | 260 | 6.22 | 135 | 63 | 260 |
| Chrysene | 384 | 2800 | 108 | 846 | 384 | 2800 |
| Fluoranthene | 600 | 5100 | 113 | 1494 | 600 | 5100 |
| Pyrene | 665 | 2600 | 153 | 1398 | 665 | 2600 |
| High Molecular Weight PAHs | 1700 | 9600 | 655 | 6676 | 1700 | 9600 |
| Total PAHs | 4022 | 44792 | 1684 | 16770 | 4000 | 45000 |
| Total DDT | 1.58 | 46.1 | 3.89 | 51.7 | 1.6 | 46 |
| p.p'-DDE | 2.2 | 27 | 2.02 | 374 | 2.2 | 27 |
| o,p'- + p,p'-DDD | 2 | 20 | 1.22 | 7.81 | 2 | 20 |
| Chlordane | 0.5 | 6 | 2.26 | 4.79 | 0.5 | 6 |
| Dieldrin | 0.02 | 8 | 0.72 | 4.3 | 0.02 | 8 |
| Endrin | | | 2.67 | 62.4 | 0.02 | 8 |
| Lindane | | | 0.32 | 0.99 | 0.32 | 1 |
| Total PCBs | 22.7 | 180 | 21.6 | 189 | 23 | – |

[a] Long (1992), Long et al. 1996, NOAA (1999)., [b] CCME (2002), [c] ANZECC (2000.), [d]Simpson, et al. (2013).

### TEL and PEL definition

The effects data were sorted and the lower 15th percentile (ERL) and median or 50th percentile (ERM) calculated. From the no-effects data, the 50th percentile (No Effect Range Median, NERM) and the 85th percentile (No Effect Range High, NER-H) were determined. The threshold effects level (TEL) defines the upper limit of sediment contaminant concentrations of no-effects data (i.e. >75% no-effects data) and was calculated as the geometric mean of the ERL and NERM. A safety factor of 2 was applied to the TEL values to define a no-observed-effects level (NOEL).

TEL = $(\text{ERL X NERM})^{1/2}$

The probable effects concentrations (PEL) defining the lower limit of the range of contaminant concentrations that are usually associated with adverse biological effects (i.e. >75% effects data), was defined as the geometric mean of the ERM and NER-H values:

- PEL = $(\text{ERM X NER-H})^{1/2}$

**Find out more:** phone 09 301 0101, email
rimu@aucklandcouncil.govt.nz or visit
aucklandcouncil.govt.nz and knowledgeauckland.org.nz

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