NATIONAL WATER QUALITY MANAGEMENT STRATEGY

PAPER No. 4

Australian and New Zealand Guidelines for Fresh and Marine Water Quality

Volume 1

The Guidelines

(Chapters 1–7)

October 2000

Australian and New Zealand Environment and Conservation Council

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Preamble

The Australian National Water Quality Management Strategy (NWQMS) aims to achieve the sustainable use of Australia's and New Zealand's water resources by protecting and enhancing their quality while maintaining economic and social development. The NWQMS is a joint strategy developed by two Ministerial Councils — the Agriculture and Resources Management Council of Australia and New Zealand (ARMCANZ) and the Australian and New Zealand Environment and Conservation Council (ANZECC). The Australian National Health and Medical Research Council (NHMRC) is involved in aspects of the strategy that affect public health. The NWQMS aims to meet future needs by providing policies, a process and national guidelines for water quality management.

Further information on the National Water Quality Management Strategy is provided in Appendix 2.

The Australian Water Quality Guidelines for Fresh and Marine Waters (ANZECC 1992) was one of a suite of 21 documents forming the NWQMS and was released in 1992 as one of the first guideline documents. In 1993 the ANZECC Standing Committee on Environmental Protection (SCEP) agreed to review the water quality guidelines to incorporate current scientific, international and national information in a clear and understandable document.

Since the ANZECC Guidelines were published in 1992 there have been a number of important advances. First, there have been some major policy initiatives at federal and state level that, combined with the National Water Quality Management Strategy, have increased the focus of attention on ecologically sustainable management of water resources in Australia and New Zealand (e.g. Council of Australian Governments (COAG) reform framework, State of the Environment reporting, and modification and implementation of the NZ Resource Management Act). Second, there is a pleasing trend towards a more holistic approach to the management of aquatic systems. Third, as initially recommended in the 1992 ANZECC Guidelines, there has been an increased use of biological indicators to assess and monitor the 'health' of aquatic ecosystems. Finally, a number of major environmental studies (e.g. the Port Phillip Bay Study in Victoria, the Southern Metropolitan Coastal Waters Study in Western Australia) have led to significant advances in knowledge about estuarine and coastal ecosystems.

The scope of this revised version, the Australian and New Zealand Guidelines for Fresh and Marine Water Quality, has also been extended to include a consideration of both Australia's and New Zealand's water resources. The review program is outlined in Appendix 4.

The Guidelines have been revised using data, relevant literature, and other information available to at least 1996, specifically:

- Databases used to derive guideline values for toxicants and sediments (Chapter 3) and aquaculture and human consumers of aquatic foods (Chapter 4) have been updated to include information available to late 1996, while default guidelines for physical and chemical stressors (Chapter 3) have been derived from databases current to early 2000.
- The guidelines for biological indicators (Chapter 3), advice for monitoring and assessment (Chapter 7) and support text for physical and chemical stressors

(Chapter 3) have been revised to include information available to late 1998. However, all support text for aquatic ecosystems (Chapters 3 and 8) and aquaculture and human consumers of aquatic foods (Chapters 4 and 9) capture important developments and key references available to early 2000.

- The guidelines for agricultural water uses (irrigation and general water use and livestock drinking water, Chapters 4 and 9) have been revised to include information available to early 2000.
- The guidelines for recreational water quality and aesthetics (Chapter 5) are still in revision in Australia, while New Zealand readers are referred to the relevant 1999 guidelines. For guidelines for drinking water (Chapter 6), Australian and New Zealand readers are referred to the relevant 1996 and 1995 guidelines respectively.

To be continuously relevant to its users, the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality*, like other NWQMS benchmark documents, will require ongoing review and revision. The present version was current up to October 2000. Users are invited to comment on the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* by contacting the offices listed on the next page. These addresses can also receive comments on the *Australian Guidelines for Water Quality Monitoring and Reporting*, so users should name the document to which their comments apply.

Contacts for comments and information

For information and advice about the Water Quality Guidelines and to advise of possible errors, omissions and changes required for future revisions, please contact the designated agency for your state or territory in Australia or for New Zealand. The agency contacts are listed below.

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The revision of the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* with its extended scope was a challenging and difficult task given the complexity of the issues and the many advances made recently in water resource management worldwide. The task could not have been possible without the substantial assistance received from a large number of people and organisations across Australia and New Zealand over the last four years.

Both the Project Committee and ANZECC and ARMCANZ Contact Group and its working parties (Appendix 4) co-ordinated input from all the relevant government jurisdictions, water quality experts, industry and conservation groups, and kept the revision on track.

ANZECC and ARMCANZ would like to acknowledge the efforts of the following organisations which coordinated the drafting of key chapters or sections of the Guidelines:

- ***eriss***
  Overall coordination, Chapters 1 & 2, biological assessment for ecosystem protection and monitoring and assessment;
- CRC for Freshwater Ecology
  Physical and chemical stressors for ecosystem protection;
- NSW EPA
  Toxicants for ecosystem protection;
- NIWA (NZ)
  Metal data for toxicants;
- CSIRO Energy Technology
  Sediment quality for ecosystem protection;
- QLD Dept Natural Resources
  Primary Industries, Agriculture: irrigation, livestock & general water use;
- CSIRO Land and Water
  Primary Industries, Agriculture: irrigation;
- PSM Group P/L and Dosaqua
  Primary Industries, Aquaculture and harvesting of aquatic foods;
- NH&MRC
  Recreation and aesthetics and drinking water.

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During the public consultation phase held over July–October 1999, 96 submissions were received from individuals and agencies from across Australia and New Zealand. The comments received in these submissions were used as the basis for the final drafting conducted in early 2000 and these resulted in substantial improvements to the Guidelines. There are also many other organisations and individuals who contributed to the review and, although they are too numerous to list here, we would like to thank them all.

Finally, a special thanks to ***eriss*** who co-funded the review and whose staff co-ordinated technical aspects of the revision and provided logistical support.

Photography on ring-binder cover of the Guidelines (from top left)

Child drinking, Ministry for the Environment, New Zealand;

Logan River (Qld), Qld EPA;

Maori war boat, Photosource New Zealand Limited Image Library;

Hereford, Qld EPA;

Irrigation channels, Bruce Cooper, NSW DLWC;

Water quality monitoring, Bruce Cooper;

Bondi Beach (NSW), Brian Robson;

Rakaia River, South Island, New Zealand, Clint McCullough, ***eriss***;

Aquaculture, Qld EPA;

Beenleigh Rum distillery (Qld), Qld EPA;

Aboriginal cultural ceremony on upper Katherine River (NT), Diane Lucas:

> '*Because our great grandmothers and grandfathers been here before, their spirits are still here. Now the spirits smell your sweat and it goes down to the deep water and makes it alright for you to be here, without any harm*' — Margaret Oenpelli & Penny Long, Barunga, NT. The washing of people by spraying water on their head is an Arnhem Land ceremony for newcomers to country to keep away bad health for people and water.

1 Introduction

This document updates the Australian water quality guidelines for fresh and marine waters released in 1992 (ANZECC 1992).

Specifically, this document:

- outlines the important principles, objectives and philosophical basis underpinning the development and application of the guidelines;
- outlines the management framework recommended for applying the water quality guidelines to the natural and semi-natural marine and fresh water resources in Australia and New Zealand;
- provides a summary of the water quality guidelines proposed to protect and manage the environmental values supported by the water resources;
- provides advice on designing and implementing water quality monitoring and assessment programs;
- has been revised using data, relevant literature, and other information available to at least 1996.

A note on the structure and other features of the Guidelines

Readers should note the following features of the Guidelines:

- Given the broad scope of the Guidelines, it has been necessary to load much of the detailed rationale and reference information, including software, onto a CD-ROM, which is in the pocket of the ring-bound folder.
- While many users will be satisfied with use of the default guideline values provided in this volume, others will want to tailor guidelines for local conditions, or may simply seek further reading. To assist users to refine the guidelines in this way or to acquire further information, cross-reference to the support information referred to in point 1 above is provided. These cross-references are indicated in the text by way of superscript letters that link the relevant passage to the corresponding italicised notes in the left hand margin of the page.
- The loose-leaf format of the Guidelines is a feature that will enable discrete subject areas to be revised in future independently of other sections. To assist this, the page numbering is independent for each of the short chapters (e.g. 2–1 to 2–xx) and to the first subsection level of the longer, more complex chapters, i.e. chapters 3, 4 and 7 (this volume), 8 (Volume 2) and 9 (Volume 3) (e.g. pages 3.1–1 to 3.1–xx, 3.2–1 to 3.2–xx etc).
- A glossary of the main terms is provided at the end of this volume to assist readers further in understanding the main issues. Users are encouraged to check the glossary for all key terms because the terminology used by the various jurisdictions throughout Australia and New Zealand is not always consistent with the terminology used in these Guidelines.

These Guidelines should not be used as mandatory standards because there is significant uncertainty associated with the derivation and application of water quality guidelines. For example, data on biological effects are not available for all

local species; there is uncertainty over the behaviour of contaminants in the field; there is uncertainty in water quality measurements. The user should be aware of this uncertainty when determining if an environmental value has been supported or not. However, the Guidelines should provide a framework for recognising and protecting water quality for the full range of existing environmental values.[a] The Guidelines also provide risk-based decision frameworks wherever possible, simply to help the user refine guideline trigger values for application at local and/or regional scales.

a Environmental values are defined in Section 2.1.3

Box 1.1 Water quality guidelines may be used to trigger action

The guidelines provided in this document are designed to help users assess whether the water quality of a water resource is good enough to allow it to be used for humans, food production or aquatic ecosystems (these uses are termed *environmental values*). If the water quality does not meet the water quality guidelines, the waters may not be safe for those environmental values and management action could be triggered to either more accurately determine whether the water is safe for that use or to remedy the problem.

For some environmental values the guideline number provided may be an adequate guide to quality (e.g. for recreation or drinking). For other specific environmental values the guideline can be just a starting point to trigger an investigation to develop more appropriate guidelines based on the type of water resource and inherent differences in water quality across regions. For water whose environmental value is aquatic ecosystem protection, for example, the investigation should aim to develop and adapt these guidelines to suit the local area or region. This document incorporates protocols and quite detailed advice to assist users in tailoring the water quality guidelines to local conditions. Invariably, the process of refining these guidelines — 'trigger values' — to local conditions will result in numbers for toxicants at least, that are less conservative and hence less constraining on surrounding activities.

Box 1.2 Application of the guidelines to groundwater

Groundwater is an essential water resource for many aquatic ecosystems, and for substantial periods it can be the sole source of water to some rivers, streams and wetlands. Groundwater is also very important for primary and secondary industry as well as for domestic drinking water, particularly in low rainfall areas with significant underground aquifers.

Generally these Guidelines should apply to the quality both of surface water and of groundwater since the environmental values which they protect relate to above-ground uses (e.g. irrigation, drinking water, farm animal or fish production and maintenance of aquatic ecosystems). Hence groundwater should be managed in such a way that when it comes to the surface, whether from natural seepages or from bores, it will not cause the established water quality objectives for these waters to be exceeded, nor compromise their designated environmental values. An important exception is for the protection of underground aquatic ecosystems and their novel fauna. Little is known of the lifecycles and environmental requirements of these quite recently-discovered communities, and given their high conservation value, the groundwater upon which they depend should be given the highest level of protection.

As a cautionary note the reader should be aware that different conditions and processes operate in groundwater compared with surface waters and these can affect the fate and transport of many organic chemicals. This may have implications for the application of guidelines and management of groundwater quality.

The present Water Quality Guidelines have been prepared under the auspices of Australia's National Water Quality Management Strategy (NWQMS) and relate to New Zealand's National Agenda for Sustainable Water Management (NASWM). More information on the NWQMS is provided below and in Appendix 2. Guidelines for the management of effluent discharges (including stormwater) and other activities affecting water resources are covered in other NWQMS documents (Appendix 2) and in the documents released by the NZ Ministry for the Environment listed in Appendix 3. All of these guidelines are complementary, and users are encouraged to take a holistic approach to water resource management by integrating these documents with other considerations such as catchment management and habitat related issues.

A 24-page introductory brochure that summarises the main features of the guidelines is also available for users who are seeking a general overview.

1.1 Background

The current Guidelines, including this working volume, arise from a revision of the NWQMS Guidelines published in 1992 (ANZECC 1992). The revision was necessary to:

- incorporate current scientific, national and international information in a clear and understandable format;
- ensure that the Guidelines complement major policy initiatives and directions undertaken at the state and federal levels in the areas of ecologically sustainable development and water resource management;
- promote a more holistic approach to aquatic ecosystem management;
- incorporate more detailed guidance on how to refine national or regional guidelines for site-specific application.

Important input to the review process from Australia and New Zealand has included: public submissions on the 1992 Guidelines and on an earlier draft of the revised document; the most recent local and overseas scientific and resource management documents and information; relevant overseas water quality guideline documents and government submissions.

In keeping with the underlying philosophy of the 1992 Guidelines, the chapters in this document describe how to apply state-of-the-art practices of water resource management and assessment, for the protection of the environmental values. The key changes in direction taken in revising the water quality guidelines are summarised below.

Management strategy

- The management strategy adopted in the 1992 guidelines has been refined so that it provides a greater focus on local environmental conditions, which should allow the water quality guidelines to be tailored to specific sites or regions.

Aquatic ecosystems

- Methods for deriving the physical and chemical water quality guidelines for ecosystem management (now termed 'guideline trigger values') have also been

updated in the light of an increased understanding of ecosystems, and improving technologies.

- There is greater focus on issue-based management of water quality rather than on the management of individual parameters. In practice, this means integrating monitoring programs so that managers measure biological parameters and related physical and chemical parameters, in both water and sediment. Therefore guidelines have been developed for these other indicator types (e.g. biological assessment, sediment quality and environmental flows).

Primary industries

- The Guidelines have amalgamated agriculture, aquaculture and human consumption of aquatic foods into one environmental value called 'Primary Industries'.

Recreation and aesthetics

- At the time of publication of these Guidelines, the material for Australian users on *Guidelines for Recreational Water Quality and Aesthetics* was still under review. Until these Guidelines are revised and endorsed, users should apply the guidelines from the *Australian Water Quality Guidelines for Fresh and Marine Waters* (ANZECC 1992). In New Zealand, water managers should refer to the Ministry for the Environment publication *Recreational Water Quality Guidelines* (New Zealand Ministry for the Environment 1999).

Drinking water

- The Guidelines refer to the Australian NHMRC and ARMCANZ (1996) *Australian Drinking Water Guidelines* and the New Zealand Ministry of Health (1995) *Drinking-Water Standards for New Zealand*, to avoid duplication and confusion.

Industrial water

- After extensive consultation with representative industrial groups, the current Guidelines provide no specific guidance for industrial water use, because industrial water requirements are so varied (both within and between industries) and sources of water for industry have other coincidental environmental values that tend to drive management of the resource. Industrial water use continues to be a recognised environmental value that has high economic benefit to the community. It must be given adequate consideration during the planning and management of water resources.

Cultural issues

- The current Guidelines recognise that water resources have important cultural and spiritual values, particularly for indigenous peoples. No specific guidance for protection of these values is provided, but consideration must be given to cultural issues in the planning and management of water resources, and as required by existing legislation, regulations and guidelines.

Monitoring and assessment

- The Guidelines discuss the essential elements of water quality monitoring and assessment programs, but with extensive reference to the recent NWQMS Monitoring and Reporting Guidelines (ANZECC & ARMCANZ 2000).

1.2 Guiding principles

The *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* are primarily based on the philosophy of ecologically sustainable development (ESD). The Australian *National Strategy for Ecologically Sustainable Development* (ESD Steering Committee 1992) defined ESD as:

> [development] using, conserving and enhancing the community's resources so that ecological processes, on which life depends, are maintained, and the total quality of life, now and in the future can be increased. Put more simply, ESD is development which aims to meet the needs of Australians today, while conserving our ecosystems to the benefit of future generations.

The need to comply with ESD principles is being included in statutes throughout Australia, with the commitment to continuous environmental improvement through comprehensive and integrated public policy.

In New Zealand, the Purpose and Principles in the *Resource Management Act (1991)* (RMA) set out the philosophy and approach for water management. The purpose of the RMA is to promote sustainable management, which is broadly equivalent to the ESD philosophy.

The Guidelines are also based on the policies and principles of the Australian National Water Quality Management Strategy which are explained in ANZECC and ARMCANZ (1994). The principles include:

- ecologically sustainable development;
- an integrated approach to water quality management;
- community involvement in water resource management, including establishment of the environmental values and development of management plans;
- government endorsement of the water quality policy objectives.

Four further guiding principles have also been adopted:

- A coordinated and cooperative approach to water quality management is vital and involves all spheres of government, the community, local and indigenous groups and the private sector.
- The high variability and complexity inherent in natural water resources needs to be recognised and taken into account when evaluating water quality or developing management strategies.
- Water resources are special features of the environment and their quality and integrity should be conserved and managed according to the intent of the *Australian National Strategy for Ecologically Sustainable Development*, the *Wetlands Policy of the Commonwealth Government of Australia* and the *National Strategy for the Conservation of Australia's Biological Diversity*.
- Ongoing research into the inter-relationships between ecological processes, water quality and the biota, and the dissemination of these findings in a readily usable form, are essential for effective management of water resources.

1.3 Objectives

The primary objective of the Australian National Water Quality Management Strategy (NWQMS) (ANZECC & ARMCANZ 1994) is based on ecologically sustainable development of water resources. The main objective of the Guidelines for fresh and marine water quality is intended to support this overall objective:

> to provide an authoritative guide for setting water quality objectives required to sustain current or likely future environmental values for natural and semi-natural water resources in Australia and New Zealand.

It is recognised that a nationally consistent approach to water quality management is underpinned by the development of high-status guidelines which can provide guidance when issues arise. The adoption of national guidelines provides a shared national objective while allowing flexibility of response to different circumstances at regional and local levels. Where appropriate, state and/or local jurisdictions can use their own legislative and regulatory tools to refine these national water quality guidelines either into their own regional guidelines or into specific water quality objectives.

The Guidelines are intended to provide government, industry, consultants and community groups with a sound set of tools that will enable the assessment and management of ambient water quality in a wide range of water resource types, and according to designated environmental values. They are the recommended limits to acceptable change in water quality that will continue to protect the associated environmental values. They are not mandatory and have no formal legal status (e.g. they are not National Environmental Standards as provided for in Section 43 of the *New Zealand Resource Management Act 1991*). They also do not signify threshold levels of pollution since there is no certainty that significant impacts will occur above these recommended limits, as might be required for prosecution in a court of law. Instead, the guidelines provide certainty that there will be no significant impact on water resource values if the guidelines are achieved.

The management framework, guidelines, protocols and strategies set out here complement other documents produced under the NWQMS umbrella (Appendix 2).

2 A framework for applying the guidelines

2.1 Water quality management framework

For the long-term management of any water resource, there must be:

- a designated and clearly stated set of environmental values;
- understanding of the links between human activity (including indigenous uses and values) and environmental quality, at an acceptable level of confidence;
- unambiguous goals for management;
- appropriate water quality objectives; and
- effective management frameworks, including cooperative, regulatory, feed-back and auditing mechanisms.

Management strategies that combine prediction, acknowledgment of uncertainty, monitoring and review are sufficiently flexible to adapt as the knowledge base improves. However, before management can decide on strategies that will ensure ecologically sustainable development in the long-term, society must have a collective vision of what it wants for each water resource, and there must be a good scientific understanding of the impact of human activities on the resource.

Until recently, management of Australian and New Zealand water resources was primarily focused on protecting environmental values based on human health, such as quality of drinking water, agricultural water and water from which aquatic foods are harvested. Maintenance of water quality to protect aquatic ecosystems was often included, but based on a very deterministic view of ecosystems that assumed that factors controlling ecosystem function could be identified and managed to prevent problems. However, it is now well recognised that the relationships between key ecological processes and their components are complex and variable (probabilistic) and cannot be determined precisely. The guidelines provided in this document attempt to take these factors into consideration.

2.1.1 The broad strategy

Australia and New Zealand both have a regional or local government framework in place. The political boundaries imposed within Australia place most of the responsibility for the management of natural resources with the states and territories. In New Zealand primary responsibility for water management rests with regional councils.

Water resource management is best implemented by integrating national, state and regional powers and responsibilities, and by using complementary water quality planning and policy tools. After all available and technical information has been collated for a defined water body, the steps listed below (and shown in figure 2.1.1) could be followed to implement a broad national management strategy at a local level.

1. Identify the *environmental values* that are to be protected in a particular water body and the spatial designation of the environmental values (i.e. decide what values will apply where).

2. Identify *management goals* and then select the relevant *water quality guidelines* for measuring performance. Based on these guidelines, set *water quality objectives* that must be met to maintain the environmental values.
3. Develop statistical performance criteria to evaluate the results of the monitoring programs (e.g. statistical decision criteria for determining whether the water quality objectives have been exceeded or not).
4. Develop tactical monitoring programs focusing on the water quality objectives.
5. Initiate appropriate management responses to attain (or maintain if already achieved) the water quality objectives.

   (Note: Several of the key terms from the broad management strategy outlined above, some of them in italics, are explained in the sections below.)

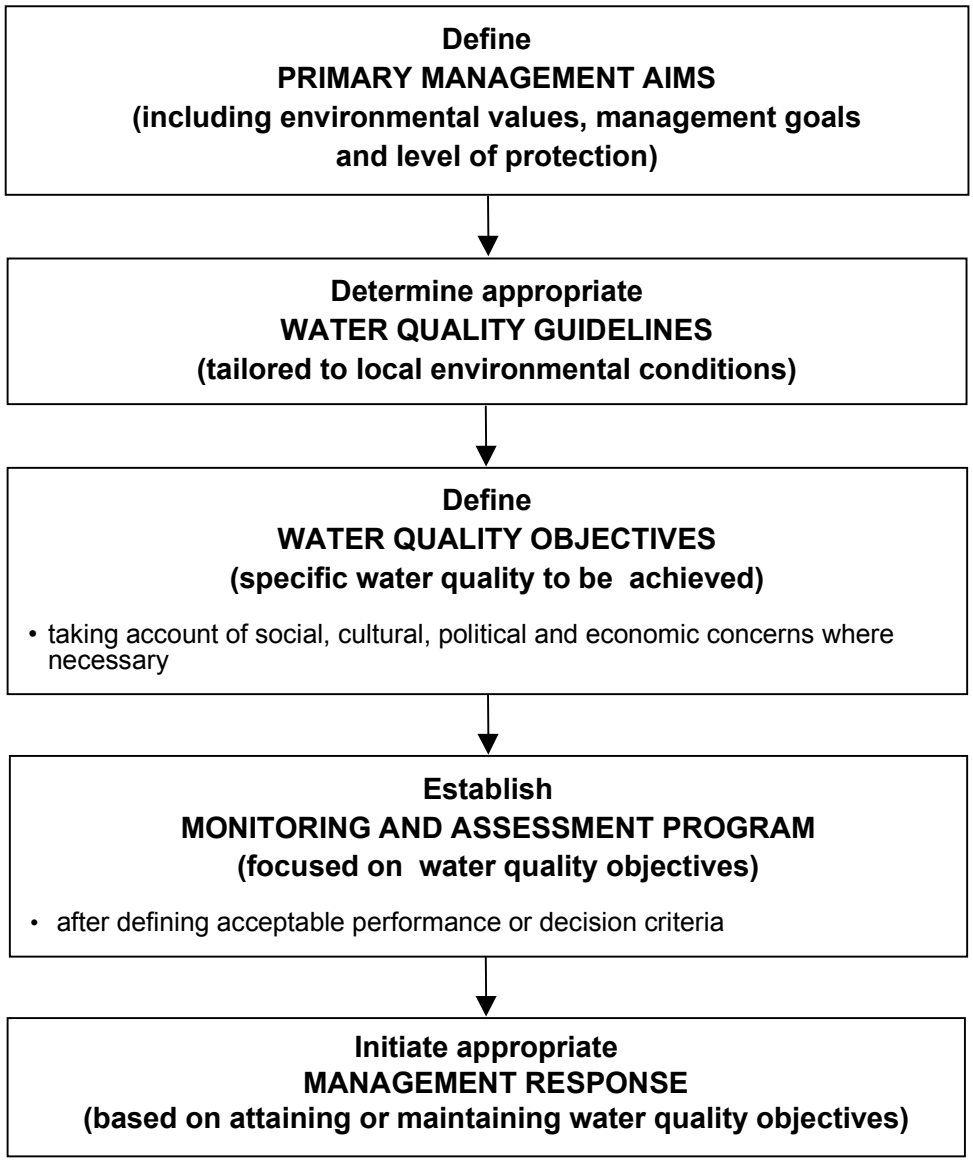


Figure 2.1.1 Management framework for applying the guidelines

The elements of this management strategy can be incorporated into comprehensive planning practices such as integrated (or total) catchment management plans (ICM or TCM) or can remain relatively small-scale plans for local areas. However, there must be consultation with stakeholders and the effective use and integration of a multi-disciplinary array of skills and knowledge to achieve success.

With respect to point 5 above, the management responses will depend on the issue of concern, the cause(s) of the poor water quality and the available tools, and should be negotiated and agreed upon by the local or regional stakeholders. In Australia, strategic management can be in the form of catchment management plans or state or national policies (e.g. statutory Environmental Protection Policies) and in New Zealand, in the form of Regional Policy Statements, regional plans or National Policy Statements, based on the agreed environmental values and their associated water quality objectives. Regulation could be achieved through discharge consents and codes of practice designed to ensure water quality objectives are not exceeded and taking into account cumulative impacts from all sources.

The monitoring programs identified in point 4 above should be maintained during and after implementation of the agreed management response(s), to evaluate their performance in achieving the water quality objectives and hence the management goals. This process should be iterative and on-going to ensure the environmental values continue to be sustained.

2.1.1.1 Responsibilities

The NWQMS outlines a three-tiered approach to water quality management at:

- the national level — a vision of achieving sustainable use of water resources by protecting and enhancing their quality while maintaining economic and social development together with overarching national guidelines for minimum water quality;
- state or territory level — implementation through state water quality planning and environmental policy processes, to provide a planning and management framework with goals and objectives consistent with the agreed national guidelines;
- regional or catchment level — complementary planning, with local or catchment management strategies developed and implemented by the relevant stakeholders. Regional communities are encouraged to participate in identifying the local environmental values and to monitor and report on progress and performance of the plans.

To underpin water resource management at the national, state and territory levels in Australia, a range of legislative and regulatory tools are being used. Examples include state and territory water and land resources management Acts, environment protection Acts, the development of water quality guidelines focused on state and territory water resources, and the development of national environmental protection measures. Each state or territory uses its own water quality planning and environmental policy tools to establish a framework compatible and consistent with the agreed national guidelines.

In New Zealand, these guidelines are designed to assist water managers with the implementation of the *Resource Management Act 1991* (RMA) which gives regional councils primary responsibility for water management. The RMA

empowers councils to develop statutory plans and local laws for water management. The RMA also enables central government to develop national policy and standards on a statutory basis.

Overall responsibility for water resource management rests with the community. The tools, strategies and policies developed to manage and protect environmental values should be applied in this wider context. In effect, there must ultimately be education and change in community behaviour toward a more environmentally sustainable approach.

The responsibilities for monitoring water resource quality should not always rest with government alone and ideally would be shared with the dischargers/users of the environment in question (these shared responsibilities could extend to the waters beyond the mixing zone of outfalls). Many community and catchment groups have already become involved in, or taken responsibility for, water quality monitoring programs and are developing management strategies to maintain or improve their water resources.

2.1.2 Stakeholder involvement

Stakeholders need to be actively involved in many of steps 1–5 outlined above, to help ensure that:

- community needs are accurately reflected;
- impacts on the community are well understood and incorporated into the decision-making (e.g. cultural, social, economic and political);
- the costs (financial, amenity, etc.) associated with decision making will be acceptable to the community;
- management strategies are appropriately targeted; and
- a shared ownership of catchment knowledge and commitment to action are being developed.

Relevant stakeholders include individuals and groups that directly and/or indirectly use, derive benefit from, and/or have an impact on the waterway being considered. These may include indigenous groups, community groups, government agencies and utilities, catchment and water managers, regulators, industry (urban and rural), agricultural groups, pest control groups, environmental groups, recreational users (e.g. fishers, swimmers) and individual residents.

The stakeholders can be involved at a number of different levels, depending on their interest and expertise, and the mechanisms available for their involvement. The latter in particular will vary depending on the approach taken by state, territory and local governments.

Box 2.1 shows examples of stakeholder involvement in Australia and New Zealand.

Box 2.1 Examples of stakeholder involvement in Australia and New Zealand

- New South Wales — A six month public consultation program in 1998 identified interim environmental values and objectives for various catchments in the State. The process involved written submissions, and information and discussion forums located in central and regional locations.
- Victoria — State Environment Protection Policies (SEPP) for water set out the 'beneficial uses' (or environmental values) to be protected in various parts of rivers, lakes, estuaries and bays and related environmental values. The SEPP process includes a legislative requirement for a period of at least three months for submissions to be received.
- Queensland — The Queensland Environmental Protection Policy for Water requires appropriate consultation with the community before environmental values and water quality objectives for a water are decided. Community groups can set an example on working to improve water quality, e.g. the Condamine Balonne Water Committee Inc. took responsibility for establishing and implementing a comprehensive water quality monitoring program in their catchment.
- Western Australia — In 1998, development of the proposed Environment Protection (Marine Waters) Policy involved community consultation to set the environmental values and environmental objectives of Perth's coastal waters. The process included key stakeholders, stakeholder reference groups and a two month consultation period.
- South Australia — Catchment goals, objectives and actions for the Torrens Catchment were formulated using a consultation program involving the community, local government, state and federal agencies and other stakeholders. A series of technical papers was presented as background to the catchment plan.
- Australian Capital Territory — The ACT Environment Protection Policy on Water Pollution requires public consultation with individuals, community groups, industry, government agencies and other stakeholders.
- Tasmania — Local communities and other stakeholders have a key role in identifying the water quality values for regional wetlands and waterways as part of the *State Policy on Water Quality Management 1997.* Information provided on these values assists the Board of Environmental Management and Pollution Control and local councils to finalise Protected Environmental Values (PEVs) for surface waters. The process of setting PEVs takes a minimum of 3 months. These values are reflected in management plans for the regions and in local council planning schemes. The *Water Management Act 1999* provides for enhanced stakeholder and community input into water allocation and management.
- Northern Territory — Environmental value declarations, informed by extensive public participation, have been used to establish the framework for water resource management in the territory since 1994. For example Darwin Harbour waters were declared under the *Water Act* in 1996 to have aquatic ecosystem and recreation and aesthetics values protected. This followed an extensive public consultation phase with public meetings, newspaper and other media promotions. Environmental objectives and water quality targets for harbour waters will be developed over 2000/01 through a further public consultation process steered by a committee with broad government agency and community representation.
- New Zealand — Consultation is an integral part of natural resource management under the Resource Management Act. All statutory plans require a period of consultation and a submission process. Consultation with stakeholders also frequently occurs in non-statutory management processes by government (central and local) and industry.

Possible forms of stakeholder involvement are listed below. Stakeholders could be part of:

- statutory reviews of development proposals;
- community forums or discussions to identify broad community goals and potential areas of conflict;
- specific groups that relate these broad goals to the environmental values that need to be protected in a particular water body, decide where these may apply, and evaluate the potential implications of different options;
- specific groups such as stakeholder advisory committees (as outlined in the NWQMS *Implementation Guidelines*) which would bring together all major interests in the one forum to discuss ideas, issues and proposals and provide a broad-based sounding board;
- community and industry participation in processes for developing management strategies (e.g. through catchment management planning) and assessing progress against water quality objectives and management goals (through monitoring of discharges and ambient conditions);
- public hearings, although this form of community forum is not commonly used in Australia.

(The NWQMS *Implementation Guidelines* (ARMCANZ & ANZECC 1998) provide more detail.)

2.1.3 Environmental values

Environmental values are particular values or uses of the environment that are important for a healthy ecosystem or for public benefit, welfare, safety or health and which require protection from the effects of pollution, waste discharges and deposits. They were often called 'beneficial uses' in the water quality literature but this term has lost favour because of its exploitative connotations. For this reason, the term 'environmental value' has been adopted by the NWQMS.

The following environmental values are recognised in these guidelines:

- aquatic ecosystems,
- primary industries (irrigation and general water uses, stock drinking water, aquaculture and human consumption of aquatic foods),
- recreation and aesthetics,
- drinking water,
- industrial water (no water quality guidelines are provided for this environmental value), and
- cultural and spiritual values (no water quality guidelines are provided for this environmental value — see box 2.2).

Box 2.2 Cultural importance of water

Water resources have important cultural and spiritual values, particularly for indigenous peoples of New Zealand and Australia.

In *New Zealand*, water has enormous cultural importance for Maori. Water acts as a link between the spiritual and physical worlds, and many waterbodies are associated with waahi tapu (sacred sites). All elements of the natural environment (including people) are believed to possess a mauri (life force) which Maori endeavour to protect. The well-being of an iwi (tribe) is linked to the condition of the water in its rohe (territory). In addition, water provides important mahinga kai (food collected from marine and freshwater areas). Supply and exchange of mahinga kai forms part of the social fabric of Maori tribal life. The *New Zealand Resource Management Act (1991)* recognises Maori values, such as through Sections 6(e), *the relationship of Maori and their culture with their ancestral lands, water, sites, waahi tapu, and other taonga* (treasures), and Section 7(a), *Kaitiakitanga* (guardianship).

Giving effect to these values may present a considerable challenge to water managers. For example, in New Zealand, water managers require guidance on how to manage water for values associated with (i) mahinga kai, (ii) waahi tapu, and (iii) mauri. These Guidelines do not provide such guidance. The New Zealand Ministry for the Environment proposes:[1]

- preparing guidelines and case studies which develop practical methods for reflecting the values of mahinga kai, waahi tapu and mauri in the management of water;
- incorporating mahinga kai values into the relevant ecosystem outcomes and actions.

Likewise, in *Australia*, indigenous cultural and spiritual values may relate to a range of uses and issues including spiritual relationships, sacred sites, customary use, the plants and animals associated with water, drinking water or recreational activities. Native title legislation, and Commonwealth and state cultural heritage legislation, provide for recognition and management of indigenous interests in water.

At this stage no water quality guidelines have been developed for the protection of cultural and spiritual values in either New Zealand or Australia. Because of the lack of such guidelines, in the water management framework, cultural values can be taken into account through the process of establishing the specific water quality objectives for a particular water resource (see figure 2.1.1).

Until further work is undertaken to better define cultural and spiritual value for users in both Australia and New Zealand, managers in both countries, in full consultation and co-operation with indigenous peoples, will need to decide how best to account for cultural values within their own management frameworks. They will need to take account of existing legislation, regulations and guidelines.

All water resources should be subject to at least one of the above environmental values, and in most cases more than one could be expected to apply. Where two or more agreed environmental values are defined for a water body the more conservative of the associated guidelines should prevail and become the water quality objectives. It is essential that the needs and wants of the community be identified when environmental values are being defined for a particular water resource.

1 NZ Ministry for the Environment 1999. *Making every drop count — A draft National Agenda for Sustainable Water Management.* New Zealand Ministry for the Environment, Wellington.

It should be recognised that environmental values are often interdependent. For example, all relevant environmental values need to be considered when evaluating the quality of return water for any one user to ensure that all agreed values are maintained; functioning ecosystems and ecosystem processes are essential for supporting wild fish populations and can provide some protection to water quality through chemical degradation or buffering capacity; there may also be situations where the water quality required to support downstream environmental values (e.g. lake, estuary or marine) will influence the establishment of water quality objectives upstream. This will be particularly relevant where downstream ecosystems are more sensitive to a particular contaminant (e.g. nitrogen in marine environments), where there are cumulative effects from persistent discharges (e.g. nutrients), or where persistent contaminants are accumulating in depositional areas downstream (e.g. heavy metals).

Once the environmental values for a water body have been defined by the relevant management authority, the level of environmental quality or water quality necessary to maintain each value must be determined. It may be broadly defined through the establishment of ***management goals*** that describe more precisely and in greater detail what is to be protected. As with environmental values, the management goals should be defined according to community needs and desires and therefore will involve consultation with relevant stakeholder groups. They should be structured so that they can become the key objectives to be achieved through management plans and therefore should relate to particular parts of the environment that can be measured. In particular, management goals should reflect the specific problems and/or threats to the established values, the desired levels of protection for aquatic ecosystems, and the key attributes of the resource that must be protected (e.g. endemic or key species, key agricultural or aquacultural species, primary or secondary recreation). From the management goals it should be obvious which the key water quality indicators are, and therefore which guidelines should be selected for establishing water quality objectives. The specific water quality objectives more tightly define the desired level of water quality, and are compared with the existing water quality to assess performance.

In some cases, the water quality needed to support the desired environmental value may not be attainable immediately. Where restoration is possible, there may be costs associated with restoring the level of quality that the community desires. Once full costs of restoration are known, the community may choose to accept a lower quality based on a full cost–benefit analysis. The environmental values and management goals for a particular area need to be well thought out, with full knowledge of the implications to the broader community. This is a process involving broad consultation with representatives of the whole community, with the aim of reaching a desirable, practical and agreed set of management goals, and hence water quality objectives.

Guidance on how to undertake community consultation processes is provided in the NWQMS *Implementation Guidelines* (ARMCANZ & ANZECC 1998).

In the absence of a clear and agreed set of environmental values for a particular water resource, managers should take a conservative approach and assume that all *appropriate* environmental values apply to the resource, by default. For example in the case of a coastal marine embayment, ‘drinking water’ would not apply by

default, but 'ecosystem protection', 'recreation and aesthetics', and 'primary industries — aquaculture' would apply.

2.1.4 Water quality guidelines

A water quality guideline is a numerical concentration limit or narrative statement recommended to support and maintain a designated water use. This document includes guidelines for chemical and physical parameters in water and sediment, as well as biological indicators. The guidelines are used as a general tool for assessing water quality and are the key to determining water quality objectives that protect and support the designated environmental values of our water resources, and against which performance can be measured.

Water quality parameters can be divided into those that have direct toxic effects on organisms and animals (e.g. insecticides, herbicides, heavy metals and temperature) and those that indirectly affect ecosystems causing a problem for a specified environmental value (e.g. nutrients, turbidity and enrichment with organic matter). Whether the effects are direct or indirect has important implications for management, and perhaps for how a guideline might be derived. Some physical and chemical stressors can also indirectly modify the toxicity of other contaminants. While specific guidelines are not provided for this mode of action, guidance is provided in each relevant section on how it can be taken into account.

The guidelines have been derived with the intention of providing some confidence that there will be no significant impact on the environmental values if they are achieved. Exceedance of the guidelines indicates that there is potential for an impact to occur (or to have occurred), but does not provide any certainty that an impact will occur (or has occurred). In areas where protection of aquatic ecosystems is a designated environmental value, the Guidelines recommend direct assessment of the biological community to assess whether ecosystem integrity is being maintained, threatened or compromised to a level that causes pollution. Biological indicators should therefore be used to complement the use of physical and chemical indicators for this value. These Guidelines describe indicators for biological assessment and give guidance for determining an acceptable level of change so that the relative condition of the ecosystem can be estimated.

For some environmental values it may not be feasible to protect all water resources to the same level, and the community may wish to aim for different levels of protection for different resources. Whatever the level of protection, it should be reflected in the management goals and the water quality objectives determined for a particular resource. In this document three levels of protection, based on ecosystem condition, are recognised for aquatic ecosystems.[a] For aquatic ecosystems the guidelines in this document have mainly been developed for use at the second and third levels of protection: *slightly to moderately disturbed ecosystems* and *highly disturbed ecosystems*. The highest level of protection is for *high conservation/ecological value* systems where management would be expected to ensure there is no change[2] in biological diversity, relative to a suitable reference

a See Section 3.1.3

2 'No change': In practice and in the absence of information that would define the thresholds of ecological change, refers to statistically conservative changes from a baseline mean or median value, e.g. change of 10% or one standard deviation from a baseline mean — see sections 3.2.4.2 and 7.2.3.3 (Stage 1).

condition.[a] For highly disturbed ecosystems that cannot feasibly be returned to a slightly to moderately disturbed condition, these Guidelines provide advice to assist managers to derive alternative guidelines that give lower levels of protection.

a See Section 3.1.4

The earlier guidelines (ANZECC 1992) acknowledge there is such inherent variability within the environment that 'site-specific' environmental information needs to be used to develop appropriate guidelines and indices of environmental quality. For example, light availability is a key factor controlling the growth and survival of benthic plants. In naturally turbid waters the biomass of a particular species may decrease with depth to a limit beyond which there is insufficient light. This limit would be deeper in less turbid waters. Thus the selection of a water clarity guideline value (e.g. light attenuation coefficient) would need to take into account these site-specific considerations.

Guideline numbers and decision frameworks

These Guidelines have adopted an innovative risk-based approach that is intended to improve the application of guidelines to all Australian and New Zealand aquatic environments. It uses decision frameworks (particularly for the protection of aquatic ecosystems) that help users to tailor water quality guidelines to local environmental conditions.[b] In this approach the old 'single number' guidelines (see ANZECC 1992) are regarded as *guideline trigger values* that can be modified into regional, local or site-specific guidelines by taking into account factors such as the variability of the particular ecosystem or environment, soil type, rainfall and level of exposure to contaminants. Trigger values are concentrations that, if exceeded, would indicate a *potential* environmental problem, and so 'trigger' a management response, e.g. further investigation and subsequent refinement of the guidelines according to local conditions. Thus these Guidelines have moved away from promoting single-number guidelines that are applied universally, towards guidelines that can be determined individually according to local environmental conditions.

b Sections 2.2.1.4 & 3.1.5

It is not mandatory to use decision frameworks, but they can reduce the amount of conservatism necessarily incorporated in the guideline trigger values, and so produce values more appropriate to a particular water resource. Decision frameworks or tools also allow more flexibility and scope for water managers. Hence guidelines that are more relevant to a specific water resource and environmental value can be developed where considered appropriate. However, it may take more time, expertise or resources to implement the risk-based decision frameworks, particularly where additional data collection is required to augment the data already collated.

Which stakeholder(s) are responsible for data collection and implementation of the decision frameworks will depend on the issue (e.g. environmental impact assessment process or management strategy development) and the jurisdictions' legislative and regulatory tools, and should therefore be decided on a case-by-case basis. Management agencies with responsibility for a number of water resources may need to prioritise their water resources based on factors such as condition of the system, increasing land use pressures, data availability, public concern, conservation issues and the outcomes of risk and cost-benefit analyses, so that limited resources can be appropriately allocated.

Alternatively, where resources, data and/or time are significant constraints, users can take a more conservative approach and initiate an appropriate management response when either the initial trigger value or a partly modified trigger value

(only part of the decision framework applied) is exceeded. The availability of data, expertise, resources and time will determine which steps in the frameworks are used.

Note: it is emphasised here, and elsewhere throughout the document, that the use of the term 'risk-based' does not imply the need for a full (quantitative) risk assessment. For example, the aquatic ecosystem guideline trigger values for toxicants are risk-based in the sense that they are calculated to protect a pre-determined percentage of species with a specified level of confidence,[a] while the decision frameworks simply provide a site-specific estimate of whether low, possible or high risk exists.[b]

a See Section 3.4.2

b Section 3.1.5

2.1.5 Water quality objectives

A water quality guideline was defined above as a numerical concentration limit or descriptive statement *recommended* for the support and maintenance of a designated water use. Water quality objectives take this a step further. They are the specific water quality targets agreed between stakeholders, or set by local jurisdictions, that become the indicators of management performance. Normally, only those indicators considered relevant to the environmental issues or problems facing the resource are selected for deriving water quality objectives. They serve to protect the designated environmental values of a resource and would normally be based on the information from these Guidelines.

A water quality objective is a numerical concentration limit or descriptive statement to be measured and reported back on. It is based on scientific water quality criteria or water quality guidelines but may be modified by other inputs such as social, cultural, economic or political constraints. Some of these inputs may be intangible and therefore hard to quantify, but nevertheless they are valid inputs to the management process. The relative weighting or importance placed on the water quality guidelines and these other, potentially very important but less tangible, considerations would be area specific, and therefore would be determined on a case by case basis. The process of modifying guidelines to establish water quality objectives would normally be carried out through cost–benefit analysis programs involving input from stakeholders or local jurisdictions.

An additional consideration when setting water quality objectives in rivers and streams is the water quality required to meet management goals and hence protect the environmental values established further downstream, including estuaries and coastal marine environments. The water quality required to support local environmental values may not be sufficient to support downstream environmental values, particularly for chemicals that persist in the environment or where downstream ecosystems are more sensitive to the contaminant (e.g. heavy metals or nutrients).

2.2 Application of the guidelines for water quality management

A primary aim of this document is to help users to develop management frameworks for protecting the environmental values of Australian and New Zealand natural and semi-natural water resources, and to derive appropriate water and sediment quality guidelines for the ambient waters that will protect their designated values.

The guidelines can:

- provide water resource managers with information that helps them identify and prioritise key environmental issues (such as the loss of seagrass beds if light intensity decreases below a critical level) and hence determine the management goals;
- assist managers to establish management goals and water quality objectives (preferably with appropriate baseline data);
- provide information that helps resource managers decide on the types of management actions they need for achieving the desired goals and targets;
- provide a basis upon which to assess whether the management actions are achieving the targets set for the management unit.

The preferred approach is to use the guidelines in a proactive way (where management focuses on preventing change beyond some pre-determined level), although in already degraded systems this may not be an option.

The purpose of a water quality management program should be to ensure that environmental values will be supported through the management goals and by meeting the agreed water quality objectives. It is recommended that this should be done through a process of cooperative best management (involving all stakeholders), and based on sound environmental arguments. Where the environmental values are not being supported because the associated management goals are not being met, remedial management programs, with appropriate performance indicators and associated time frames, should be developed and implemented to ensure the management goals will be met.

2.2.1 Philosophical approach to applying the guidelines

New ways of managing water quality have developed to match growing scientific understanding of ecosystem complexity. Traditional scientific and management approaches are now often inappropriate; instead there must be increasing reliance on holistic best-practice approaches to ensuring sustainable use of water resources. Key issues underpinning the new philosophy espoused in these Guidelines are outlined below. Some of them were also fundamental to the previous (ANZECC 1992) Guidelines.

2.2.1.1 Sustainable use

The fundamental aims of the NWQMS in Australia and the Resource Management Act in New Zealand are the sustainable use and management of each nation's water resources in environmental, economic and social contexts. To achieve these aims, the concept of integrated catchment management[3] (ICM) is promoted today. The concept is consistent with the management framework outlined in these Guidelines, and encompasses all aspects of environmental management within a catchment, including water quality. Within the ICM framework, environmental values are identified by all stakeholders of individual resources, namely landowners and the community, in partnership with relevant government agencies.

[3] Under section 30/1/a of the *New Zealand Resource Management Act (1991)* all regional councils are required 'to achieve integrated management of the natural and physical resources of the region'.

2.2.1.2 Cooperative best management

Formerly, deterioration in water quality was largely controlled by regulation and management. While these command and control approaches successfully deal with the obvious point source problems, they have produced an end-of-pipe and minimum compliance culture. It is now also clear that a regulatory approach is generally not an appropriate tool for resolving the problems of diffuse sources of contamination which have just as much or more of an impact on water quality than point sources.

Environmental regulation and management in Australia and New Zealand are currently undergoing major change, adopting a more holistic and integrated pollution-prevention approach to environment protection. This involves a shift from control to prevention, from end-of-pipe regulation to cleaner production, from a focus on prescriptive regulation to a focus on outcomes and on cooperation rather than direction. This new approach is being increasingly adopted in formulating water resource management policies and strategies. It requires the commitment of industry and government and the involvement of the community to establish cooperative best management and overall responsibilities for maintaining and improving water resources. The NWQMS *Implementation Guidelines* (ARMCANZ & ANZECC 1998) outlines a framework for involving all stakeholders in the management of water resources.

The success of cooperative best management relies on negotiated agreements developed through processes involving the entire community; they set boundaries within which business can maintain the defined environmental values. Cooperative best management provides a framework through which many sources of pollution may be addressed in an equitable and effective manner. Sharing of responsibility, cooperative action, and effective monitoring and reporting arrangements are key aspects. Also important is the emphasis on flexibility and integrated management to achieve the best feasible environmental outcomes. In practice there may be issues over which stakeholders are unable to reach agreement. Local jurisdictions may therefore need to consider establishing conflict resolution mechanisms to facilitate the decision making process.

It is also important that communication networks be developed across whole catchments to address broad-scale issues cooperatively. For example, when setting the water quality objectives for upstream riverine ecosystems, effects on downstream environmental values, including cumulative effects, must also be considered.

Cooperative best management focuses on attaining goals of environmental quality rather than on compliance *per se*. For example, licence conditions or agreed levels of unacceptable environmental change in monitoring programs would be negotiated between all the stakeholders, with the overriding objective of attaining the established management goals for a water resource (and hence protecting its environmental values), rather than simply regulating to meet individual water quality parameters. The process would consider best management practices and the ability of the industry to achieve adequate effluent quality within a reasonable time frame. Where the agreed licence conditions were not met, or a trend toward a significant change in ambient water quality was detected, there would be an attempt to resolve the problem cooperatively before using a regulatory approach as a last resort. To complement their cooperative approach, jurisdictions might need to introduce a

number of tools for controlling diffuse and point source pollution (e.g. economic incentives, emissions trading).

Cooperative best management involves monitoring and impact assessment. Although risk assessment concepts are familiar to many water resource managers, analogous concepts of the potential for errors in statistical inferences based on monitoring data are often poorly understood, or neglected. An alternative approach to statistical decision making (Mapstone 1995, 1996) is suggested (see box 2.3) that should jointly involve all stakeholders. As mentioned above, even where water quality meets the agreed water quality objectives and is 'acceptable' in a statistical sense, stakeholders should work cooperatively to develop a clear understanding of the issues associated with, and consequences of, water quality that is trending towards the established objectives. In this way, intervention, including changes to industry practices, can be set in place at an early stage if deemed necessary.

Box 2.3 An alternative approach to statistical decision making (Mapstone 1995, 1996)

Traditionally, statistical analysis of monitoring data has only considered minimising the probability of concluding that an environmental impact has occurred when, in fact, no impact has occurred — a Type I error. However, to maximise protection of the environment it is perhaps more important to consider Type II errors — the probability of concluding that an impact has not occurred when, in fact, it has. The first step in the suggested alternative approach is to decide on the size of effect that would cause concern (or constitute an early warning). Then, with this *critical effect size* in mind, the stakeholders consider the possibility that such an effect might either be missed (a Type II error) or inferred incorrectly (a Type I error). Monitoring and data collection are then designed to keep the risks of both Type I and Type II errors to the values agreed up-front by the stakeholders, given the stipulated critical effect size. The significance criterion used in statistical tests should be that which ensures that the agreed *ratio* of Type I and Type II errors is maintained.[a]

Consistent with the principle of cooperative best management, this approach should result in benefits to all stakeholders. All parties would be aware of the targets for monitoring, the statistical criteria by which statistical decisions will be made, what will trigger management action, the level of safeguard built into the decision making process, and the risks of expense or environmental impact arising from errors in the assessment and monitoring process. Clear knowledge of these factors should reduce the risk of wrangling or litigation when impacts have been inferred and should allow explicit planning for mitigation or restoration actions that might arise in the future, but with some known minimum probability of a wrong conclusion.

a Type I and II errors are explained more fully in Section 3.1.7

Other tools that might be considered in cooperative best management are memoranda of understanding and catchment management plans. Non-point source pollution problems, in particular, could be addressed through the development and implementation of catchment management plans by landowners and the community, in partnership with relevant government agencies. One of the main objectives of these catchment management plans would be to achieve the management goals set for the aquatic environment. Well designed and appropriately focused monitoring programs could assess the effectiveness of catchment management plans in meeting specified water quality targets.

2.2.1.3 Management focus on issues not guidelines

The philosophical approach for using these revised Guidelines is this: protect environmental values by meeting management goals that focus on concerns or potential problems, e.g. toxicity. This is in contrast to previous approaches which more often focused on simple management of individual water quality parameters, e.g. toxicant concentration, to meet respective water quality guidelines or objectives. First, identify the water quality concern (e.g. toxicity, algal blooms, soil structure degradation, loss of animal vigour, deoxygenation, loss of biodiversity), and establish and understand the environmental processes that most influence or affect the particular concern. Then select the most appropriate water quality indicators to be measured, and identify the relevant guidelines.

Usually a range of environmental problems is responsible for degradation of water resources in Australia and New Zealand and so issues typically involve a range of water quality parameters. An issue-based approach to management would focus on the overall problem, and ensure an integrated approach to addressing relevant biological, chemical and physical aspects of water quality. For example, in situations where sediment contamination is likely, water managers should not focus solely on whether the measured sediment concentrations are above or below a guideline. They should also consider the bioavailability of the contaminant, and analyse trends and consider risk factors to determine whether, under current or proposed management regimes, guideline values are likely to be exceeded in the future.

2.2.1.4 Tailoring guidelines for local conditions

Optimum water quality characteristics differ between regions. There is a wide range of ecosystem types and environments in Australia and New Zealand, and it is not possible to develop a universal set of specific guidelines that apply equally to all. (Some of the default guidelines, however, do now distinguish amongst several different ecosystem types and regions making these values much more focused than they were in the previous Guidelines.) Further, environmental factors can significantly alter the toxicity of physical and chemical stressors at a site and these factors can vary considerably among sites. The present Guidelines move away from single number values that are mostly conservative, and emphasise guidelines that can be determined individually, according to local environmental conditions. This is done through the use of local reference data and 'risk-based decision frameworks'.

Decision frameworks provide guideline trigger values (equivalent to the old guideline default values) that refer to the concentration of the chemical available for uptake by organisms. Guideline trigger values are concentrations that, if exceeded, will indicate a potential environmental problem, and so 'trigger' further investigation. The investigation aims to both assess whether exceedance of a trigger value will result in environmental harm and refine a guideline value, by accounting for environmental factors that can modify the effect of the chemical.[a] Although in some cases this will require more work, it will result in much more realistic goals for management and therefore has the potential to reduce both costs for industry and confrontation.

a See also Section 3.1.5

2.2.1.5 Water or environmental quality

Water (and sediment) quality is only one aspect of maintaining some environmental values. In many cases (e.g. for primary industries and aquatic ecosystems) other factors are also important, e.g. flow, habitat, soil type, animal

diet, groundwater hydrology and barriers to recruitment. In many parts of Australia and New Zealand, water quality is reasonably good but management goals for maintaining aquatic ecosystems are not being met because of loss or degradation of habitat, particularly riparian vegetation. In these situations, enhancement of water quality is unlikely to result in any significant environmental benefit because improvement in *habitat* is needed to achieve management goals and protect the environmental value.

Before investing in water quality management strategies, managers need to be sure that water quality is the key issue to be addressed in the water body under consideration, and that resources would not be better spent on other aspects of the water resource, such as riparian vegetation, habitat or hydrological regime.

2.2.1.6 Integrated water quality assessment

Water quality, environmental values and the surrounding environment are all intimately connected and need integrated assessment. This should also acknowledge that ecosystems and environmental values upstream and downstream are linked and can affect each other.

a Section 3.2

b Section 3.5

These Guidelines include a substantial section on assessment of biological aspects of aquatic ecosystems,[a] to accompany physical and chemical indicators in assessing impacts on ecosystem integrity. Sediment quality guidelines are also given.[b] This is important because pollutants become partitioned between water, sediment and biota and move between them depending on prevailing environmental conditions. These Guidelines also advise on suitable environmental flows in rivers and streams.

Similarly, in assessing water quality for irrigation, the Guidelines include consideration of soil and plant aspects of the production system, as well as the off-farm implications of water use.

2.2.1.7 Continual improvement

An overriding principle that should guide management should be *continual improvement*. This is more obvious where water or sediment quality does not match the water quality objectives. In badly polluted waters it might even be necessary to set intermediate levels of water quality to be achieved in well defined stages, each subsequent target closer to the required water quality objective, until it is finally met. However, in waters that are of better quality than that set by the water quality objectives, some emphasis could still be given to reducing the level of contamination from all sources, particularly for highly modified water resources. Wherever possible, ambient water quality should not be allowed to degrade to the levels prescribed by the water quality objectives.

2.2.1.8 Guidelines not standards

The Guidelines recommend numerical and descriptive water quality guidelines to help managers establish water quality objectives that will maintain the environmental values of water resources. *They are not standards*, and should not be regarded as such. The vast range of environments, ecosystem types and food production systems in Australia and New Zealand require a critically discerning approach to setting water quality objectives.

State, territory or local jurisdictions will need to determine whether water quality objectives should be enshrined in legislation based on the particular local circumstances.

2.2.1.9 Ambient waters

The Guidelines have not been designed for direct application in activities such as discharge consents, recycled water quality or stormwater quality, nor should they be used in this way. (The exception to this may be water quality in stormwater systems that are regarded as having some conservation value.) They have been derived to apply to the ambient waters that receive effluent or stormwater discharges, and protect the environmental values they support. In this respect, the Guidelines have not been designed to deal with *mixing zones*, explicitly defined areas around an effluent discharge where the water quality may still be below that required to protect the designated environmental values. As such, the application and management of mixing zones are independent but very important processes.

2.2.2 Mixing zones

Even when stringent effluent limits are set and strict waste minimisation is practised, effluents may be of poorer quality than the receiving water. It has been accepted practice to apply the concept of the *mixing zone*, an explicitly defined area around an effluent discharge where certain environmental values are not protected (see description in box 2.4).

Box 2.4 Mixing zones adjacent to effluent outfalls

Mixing zones are often defined as explicit areas around effluent discharges where the management goals of the ambient waters do not need to be achieved and hence the designated environmental values may not be protected. In this context mixing zones are sometimes termed *exclusion zones.* Appendix 1 of Volume 2 provides some key references and further information and advice on mixing zones. The following issues are covered there:

- the nature of mixing zones;
- difficulties with mixing zones;
- the management of mixing zones;
- best-practice effluent release and mixing zone management, as a case study; and
- mixing zone models.

Effective discharge controls that consider both the concentration and the total mass of contaminants, combined with *in situ* dilution and waste treatment, should ensure that the area of a mixing zone is limited and the values of the waterbody as a whole are not jeopardised. The environmental conditions within a mixing zone, and its size, are important concerns, particularly because degraded areas around effluent discharges reduce environmental benefits. If mixing zones are to be applied, then management should ensure that impacts are effectively contained within the mixing zone, that the combined size of these zones is small and, most importantly, that the agreed and designated values and uses of the broader ecosystem are not compromised.

2.2.3 Application of water quality prediction models

Development of a water quality management strategy depends on the quality of available information and a capacity to predict the effects of various actions on water quality. This can be done via conceptual models, which are often used to predict effects of discharges on the environment. Conceptual models can be very simple flow diagrams that illustrate the linkages between the components of the system or they can be more complex models built up from information arising from previous studies. They should indicate which key processes are influencing the system and highlight those processes likely to be affecting the water quality indicators that are of concern. These models may be conservative and rely on worst-case conditions of dilution and degradation of effluents, or they may attempt more complex analysis of cause and effect. Because of the unique features of each system, it has generally been found that models developed for a particular waterbody cannot be used for other waterbodies without significant modification. A more detailed discussion of conceptual models is provided in the *Australian Guidelines for Water Quality Monitoring and Reporting*, the Monitoring Guidelines (ANZECC & ARMCANZ 2000).

The preferred approach for managing persistent contaminants that have concentration-related toxicity potential is appropriately based on controlling ambient concentration in the environment. Generic guidelines for the protection of environmental values can be established for these types of contaminants, based on modelled relationships between concentration/exposure of the contaminant and the toxicity to test organisms, and applying safety margins designed to take account of the uncertainty associated with transferring laboratory-derived data to the open environment and the likelihood and pathways of bioaccumulation/ persistence/degradation. This model is suitable for managing toxicants in general, but alternative approaches are needed for managing substances such as nutrients that may stimulate rather than retard growth of particular species.

An example of a model that is raised frequently in the context of water pollution control is that of *assimilative capacity*. The underlying philosophy of this concept is that a natural system has the capacity to receive some level of human-induced nutrient input without unacceptable changes occurring. This concept has been defined using a variety of terms including: *assimilative capacity* or *environmental capacity* (GESAMP 1986, WAEPA 1990, Masini et al. 1992); *receiving capacity* or *absorptive capacity* (UNESCO 1988, WAWA 1994) and *carrying capacity* (French 1991, Jenkins 1991). Regardless of what name is used this ecosystem-based approach is now recognised as central to the principle of ecological sustainability (IUCN, UNEP & WWF 1991, Jenkins 1991, Folke et al. 1993).

This ecosystem-based approach is based on establishing linkages between total nutrient loadings to an ecosystem and the response of the most sensitive or important component of that ecosystem. Once these relationships have been quantified, and the desired management goals defined, regulatory agencies can set ecologically-based maximum nutrient loadings consistent with maintaining the desired environmental quality. An ecosystem-based approach linking nutrient loadings to environmental response has been successfully applied to Perth's coastal waters (WADEP 1996).

In many freshwater and estuarine systems[a] the biological response to nutrient additions can become confounded and less predictable because plant growth and biomass may be significantly limited by factors other than nutrient availability. Light availability is the dominant limiting factor in many naturally–turbid or tannin stained waters for at least part of the year. Under these conditions nutrients can behave more conservatively and concentrations can differ between seasons. For effective management in this case, key pathways of nutrient transformation need to be known, and seasonal and interannual variation in flow regimes need to be understood. Under these circumstances, models of system behaviour employing flow-weighted concentration-based approaches can be very useful and are sometimes essential.

a See Section 3.1.2

No matter which 'predictive' model is used, it is essential that environmental quality and the attainment of management goals are regularly assessed through monitoring to determine whether regulation or other management is necessary. Undesirable trends and the necessity for proactive management can be identified if data are collected at appropriate temporal scales. This relies on regular monitoring of indicators within some or all of the environmental media (water, sediment and biota) and assessment against appropriate guidelines or water quality objectives and reference sites. This information improves our conceptual understanding of the ecosystem being managed and in particular the pathways that underpin predictive models.

2.2.4 Deriving guidelines for compounds where no guidelines currently exist

The Guidelines focus on water quality management in Australia and New Zealand, but situations will arise where there is not enough information to address an issue. There are more than 70 000 chemicals in use around the world. It is not feasible to develop guidelines for all of them, either because there are insufficient toxicological studies available, or because the chemical is currently not available in Australia or New Zealand or not considered a risk there. There could also be situations where effluent contained a range of chemicals and complexes, and the chemical make-up might not be well understood. In this instance the complex chemistry might increase or reduce the toxicity of the overall mixture to an unknown degree and so the guidelines would be irrelevant. A third possible situation relating to the protection of aquatic ecosystems is where there is a well founded suspicion that a particular natural community may have atypical sensitivity to one or more contaminants.

Direct toxicity assessment is a useful tool that can be used in these circumstances, although it is mainly used to assess the toxicity of complex effluents and to derive guidelines for the amount of dilution required to safely discharge an effluent to aquatic environments. It can also be used as a monitoring tool, testing the ambient waters after they have received effluent discharges. The main advantage with using direct toxicity assessment is that it is not necessary to know the exact chemical make-up of the test effluent, and the interactions between the components, to determine potential impacts.

If water quality guidelines do not exist for a specific chemical, or if effluents contain a complex range of chemicals, expert advice should be sought from the relevant authorities on whether a current guideline exists or how a guideline might be derived. These sorts of situations are most likely to arise for the protection of aquatic ecosystems, and later chapters of the Guidelines give extensive guidance for addressing these problems.

3 Aquatic ecosystems

3.1 Issues for all indicator types

This chapter specifies biological, water and sediment quality guidelines for protecting the range of aquatic ecosystems, from freshwater to marine. As already noted the guidelines are not sufficient in themselves to protect ecosystem integrity; they must be used in the context of local environmental conditions and other important environmental factors, for example, habitat, flow and recruitment. For the protection of rare aquatic communities and/or species, guidelines for the highest level of protection should be applied.[a]

a See Section 3.1.3

The chapter is divided into five sections: Section 3.1 is introductory and covers information common to all indicator types; Section 3.2 contains guidelines for the biological assessment of ecosystem condition; Section 3.3, guidelines for physico-chemical stressors; Section 3.4, guidelines for toxicants in water; and Section 3.5, guidelines for toxicants in sediments.

The scientific rationale behind the guidelines, and other useful background information for applying the guidelines, are provided in Volume 2 of the Guidelines. Guidelines for the design and implementation of monitoring and assessment programs involving the types of water quality indicators discussed in this chapter, are contained in Chapter 7.

3.1.1 Philosophy and steps to applying the guidelines

Many benefits of aquatic ecosystems can only be maintained if the ecosystems are protected from degradation. Aquatic ecosystems comprise the animals, plants and micro-organisms that live in water, and the physical and chemical environment and climatic regime with which they interact. It is predominantly the physical components (e.g. light, temperature, mixing, flow, habitat) and chemical components (e.g. organic and inorganic carbon, oxygen, nutrients) of an ecosystem that determine what lives and breeds in it, and therefore the structure of the food web. Biological interactions (e.g. grazing and predation) can also play a part in structuring many aquatic ecosystems.

Humans have caused profound changes in Australian and New Zealand aquatic ecosystems, particularly in the 200 years since European settlement of these countries (ANZECC 1992) and the need to protect and even reverse degradation of important aquatic ecosystems is now recognised. Commercial and recreational harvests of fish and shellfish can only be obtained from waters where ecosystems provide the food and habitat to support the growth and reproduction of the harvestable species. Aquatic ecosystems are worthy of protection for their intrinsic value. Effective conservation of endangered species can only be achieved by conserving the ecosystems that support them (ANZECC 1992).

Box 3.1.1 Human activities affecting aquatic ecosystems

A wide range of human activities can cause variations in abiotic factors, which can lead to biological changes more dramatic than those which occur naturally. The effects of human activities include pollution from industrial, urban, agricultural and mining sources; regulation of rivers through the construction of dams and weirs; salinisation; siltation and sedimentation from land clearance, forestry and road building; clearance of stream bank vegetation; over-exploitation of fisheries resources; introduction of alien plant and animal species; removal and destruction of habitat; polluted discharges from industrial, urban, agricultural and mining activities; over-exploitation of the biological resources of freshwater and marine systems; recreation (e.g. lead shot in wetlands, hydrocarbons from boats and jet skis); cold water from reservoirs and hot water from power plants; ship ballast water containing exotic species; intentional introduction of non-native species for recreation or commercial production; and eutrophication (nutrient enrichment that may stimulate the growth and dominance of toxic cyanobacteria in freshwaters and estuaries, and toxic dinoflagellates in marine waters).

The greatest threat to the maintenance of ecological integrity is habitat destruction (Biodiversity Working Party 1991). The previous ANZECC (1992) guidelines foreshadowed the need for a broader, more holistic approach to aquatic ecosystem management, to consider all changes, not just those affecting water quality. Such changes could include serious pollution of sediments, reduction in stream flow by river regulation, removal of habitat (de-snagging, draining wetlands) or significant changes in catchment land use, any of which could cause significant ecosystem deterioration (ANZECC 1992). The guidelines for water quality management documented here are therefore a necessary but only partially sufficient tool for aquatic ecosystem management or rehabilitation.

The objective adopted in this document for the protection of aquatic ecosystems is:

> to maintain and enhance the 'ecological integrity' of freshwater and marine ecosystems, including biological diversity, relative abundance and ecological processes.

Ecological integrity, as a measure of the 'health' or 'condition' of an ecosystem, has been defined by Schofield and Davies (1996) as:

> the ability of the aquatic ecosystem to support and maintain key ecological processes and a community of organisms with a species composition, diversity and functional organisation as comparable as possible to that of natural habitats within a region.

Depending on whether the ecosystem is non-degraded or has a history of degradation the management focus can vary from simple maintenance of present water quality to improvement in water quality so that the condition of the ecosystem is more natural and ecological integrity is enhanced.

For the assessment of ecosystem integrity, these Guidelines focus on the structural components of aquatic communities (biodiversity) and key ecological processes (e.g. community metabolism) as defined in Section 3.2.1.1.

a See Section 3.1.6

With or without biological assessment,[a] chemical and physical water quality indicators continue to be important surrogates for assessing and/or protecting ecosystem integrity. This document therefore provides guidelines for chemical and physical water quality indicators as well as biological indicators.

Box 3.1.2 Protecting biodiversity

Biological diversity is defined as the variety of life forms, including the various plants, animals and micro-organisms, the genes they contain and the ecosystems of which they are a part (Biodiversity Unit 1994, DEST State of the Environment Advisory Council 1996). Broadly, biodiversity is considered at three levels: genetic diversity, species diversity and ecosystem diversity.

Great difficulty arises in establishing a level of protection for biodiversity so that its maintenance is guaranteed. The Biodiversity Working Party (1991) suggested:

> *Ideally, it should be that level that guarantees the future evolutionary potential of species and ecosystems. All development is likely to cause some loss of the genetic component of biodiversity, to reduce overall populations of some species, and to interfere to a greater or lesser extent with the ecosystem processes. Protecting biodiversity means ensuring that these factors do not threaten the integrity of ecosystems or the conservation of species.*

Figure 3.1.1 shows a framework for applying the guidelines to the protection of aquatic ecosystems.[a] The three parts are described below. Each of the first two steps is common to the application of all the indicator types (biological, physico-chemical, chemical and sediment).

a See also box 3.1.3

Box 3.1.3 How to apply the guidelines

The following steps should be followed when applying the guidelines for the protection of aquatic ecosystems; steps 1–3 are the first parts of the broad framework presented in figure 3.1.1.

1. *Define the primary management aims* (Section 3.1.1.1)

2. *Determine appropriate guideline trigger values for selected indicators* (Section 3.1.1.2). After determining a balance of indicator types, each of the remaining steps is common to the application of physical and chemical stressors and toxicants in water and sediment. For the biological indicators, the principles of the steps 'Select relevant indicators' and 'Select specific indicators …' should be applied to the general framework for biological indicators (figure 3.2.1). At this stage, initial sampling can commence, ideally in support of a pilot program.

3. *Assess test site data and, where possible, refine trigger values to guidelines using (i) the general framework for biological indicators (figure 3.2.1), and (ii) the decision frameworks for other indicators*. Frameworks for (ii) are described in Section 3.1.1.3 ('Risk-based application of the guidelines'). Decision frameworks to apply to specific indicators, and detailed guidance on applying these, may be found in the Guidelines figures and sections as follows:

   (a) physical and chemical stressors — figure 3.3.1, Section 3.3
   (b) toxicants — figure 3.4.2, Section 3.4
   (c) sediments — figure 3.5.1, Section 3.5.

4. *Define water quality objectives* (figure 2.1.1, Section 2.1.5)

5. *Establish a monitoring and assessment program* (figures 2.1.1 & 7.1, Chapter 7).

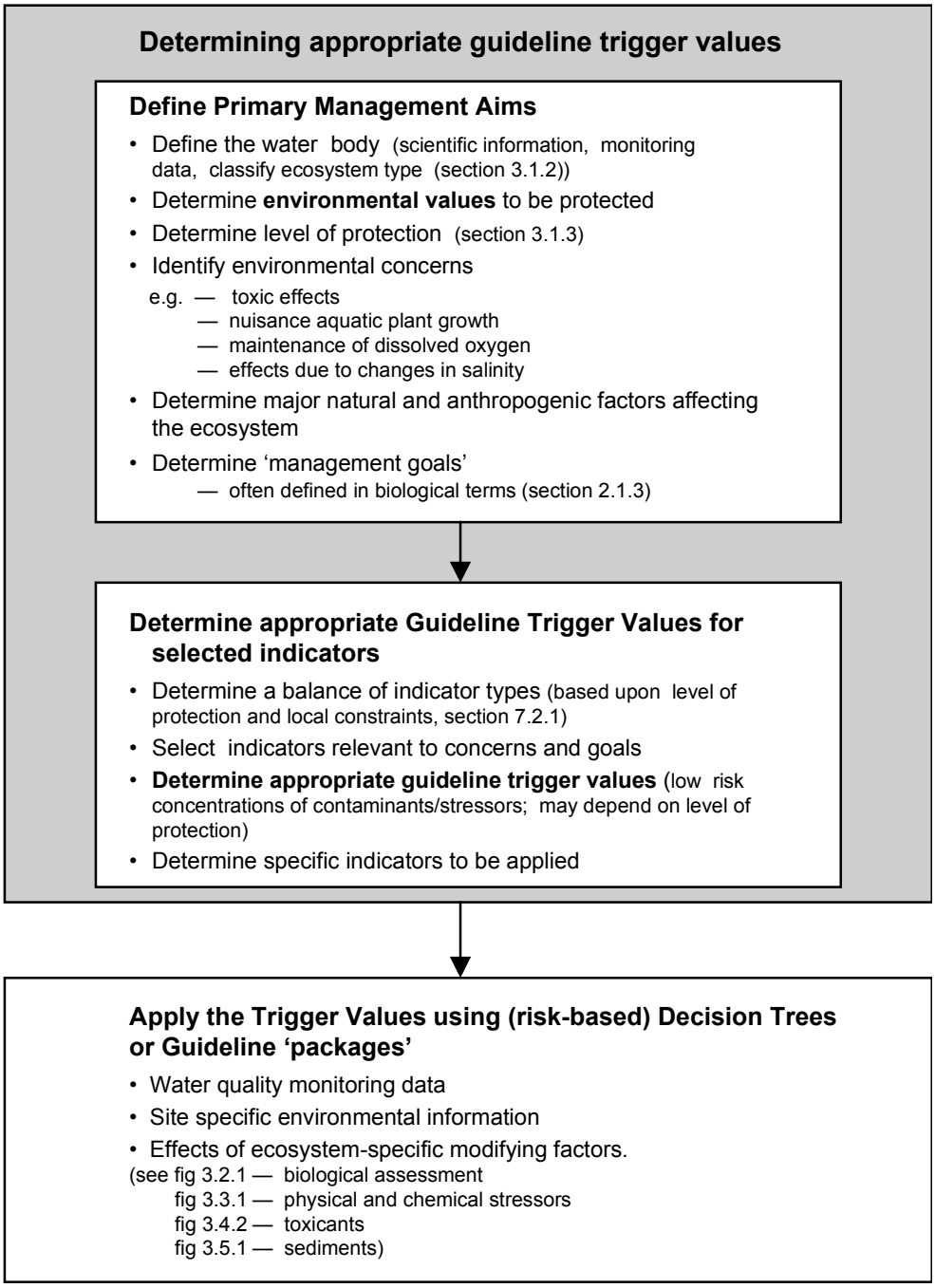


Figure 3.1.1 Flow chart of the steps involved in applying the guidelines for protection of aquatic ecosystems

3.1.1.1 Primary management aims

Define the water body, from scientific information and monitoring data. Good management can only be based on detailed information about the ecosystem being protected. Information can be collected by site-specific studies. The previous Guidelines (ANZECC 1992) also recommended that site-specific studies be undertaken in many cases.

a See Sections 3.1.2 and 3.3

Define the water body by ecosystem classification. Using appropriate scientific information the ecosystem can be classified into its corresponding type (up to six types are recognised for the guidelines for physical and chemical stressors;[a] see figure 3.1.3). The new Guidelines recognise the diverse range of ecosystem types

in Australia and New Zealand, and the need to consider the particular attributes of each ecosystem to achieve effective management.

Determine the environmental values. These have been described in Chapter 2.[a]

Determine the level of protection required. What condition should the ecosystem be in, and what level of change would be regarded as acceptable? Three levels of ecosystem condition are proposed as a basis for applying the guidelines.[b]

Identify environmental concerns. What are the main concerns or problems? For most chemical contaminants the issue is generally toxicity,[c] but eight other problems or issues can result from physical and chemical stressors.[d]

Determine the natural and human-induced factors affecting the ecosystem. It is important to identify and collate information about the most important natural processes and human activities that could influence the system being evaluated. These processes and activities need to be taken into account when conceptual models are being formulated to improve understanding of the system. They will also guide subsequent management strategies developed to improve water quality and designs for water quality monitoring programs.

Determine management goals. Next, define the management goals or targets, in terms of measurable indicators of the condition (or state) of the ecosystem. Indicators are usually biological parameters, but may also be physical and chemical parameters[e] such as toxicant concentrations (in water column and in sediments) and concentrations or loads of physical and chemical stressors.[f]

a See Section 2.1.3

b Section 3.1.3

c Section 3.4 and 3.5

d Section 3.3

e Section 2.1.4 and 3.2

f Section 3.3.2

3.1.1.2 Determine appropriate guideline trigger values for selected indicators

The next exercise is predominately a desk-top study, using existing reference data and other biological, physical and chemical information about the system. Some preliminary analyses may be required to characterise the nature and dispersion behaviour of contaminants. Four steps are involved:

1. *Determine a balance of indicator types.* The extent of the water quality assessment program and the level of detail it must achieve will depend partly upon the level of protection assigned to the water resource and the local information constraints. More detailed investigation (and therefore additional monitoring and assessment effort) would be expected for sites assigned high levels of protection and for sites where serious constraints are identified, such as lack of pre-disturbance data.[g]

2. *Select relevant indicators.* Determine indicators which will be relevant to the environmental concerns and management goals. An indicator is a parameter[4] that can be used as a measure of the quality of water.

3. *Determine appropriate guideline trigger values.* Determine guideline trigger values for all indicators, taking into account level of protection. For physical and chemical stressors and toxicants in water and sediment, the preferred approach to deriving trigger values follows the order: use of biological effects data, then local reference data (mainly physical and chemical stressors), and finally (least preferred) the tables of default values provided in the Guidelines (see figure 3.1.2). (While the default values are the least preferred method of

g Section 7.2.1

[4] Readers who also read the Monitoring Guidelines (ANZECC & ARMCANZ 2000) should note that there the term 'indicator' is only used to refer to parameters that, either severally or singly, can indicate ecosystem condition.

deriving trigger values, it is conceded that these will be most commonly sought and applied until users have acquired local information.)

4. *Select specific indicators for inclusion in the monitoring and assessment program.* The choice of indicators will be based upon the level of protection assigned to the water body, local information constraints, resource constraints, availability of expertise and an initial hazard assessment. The hazard assessment is based upon a comparison of estimated (first-pass) ambient concentrations of indicators against the guideline trigger values determined from the previous step.

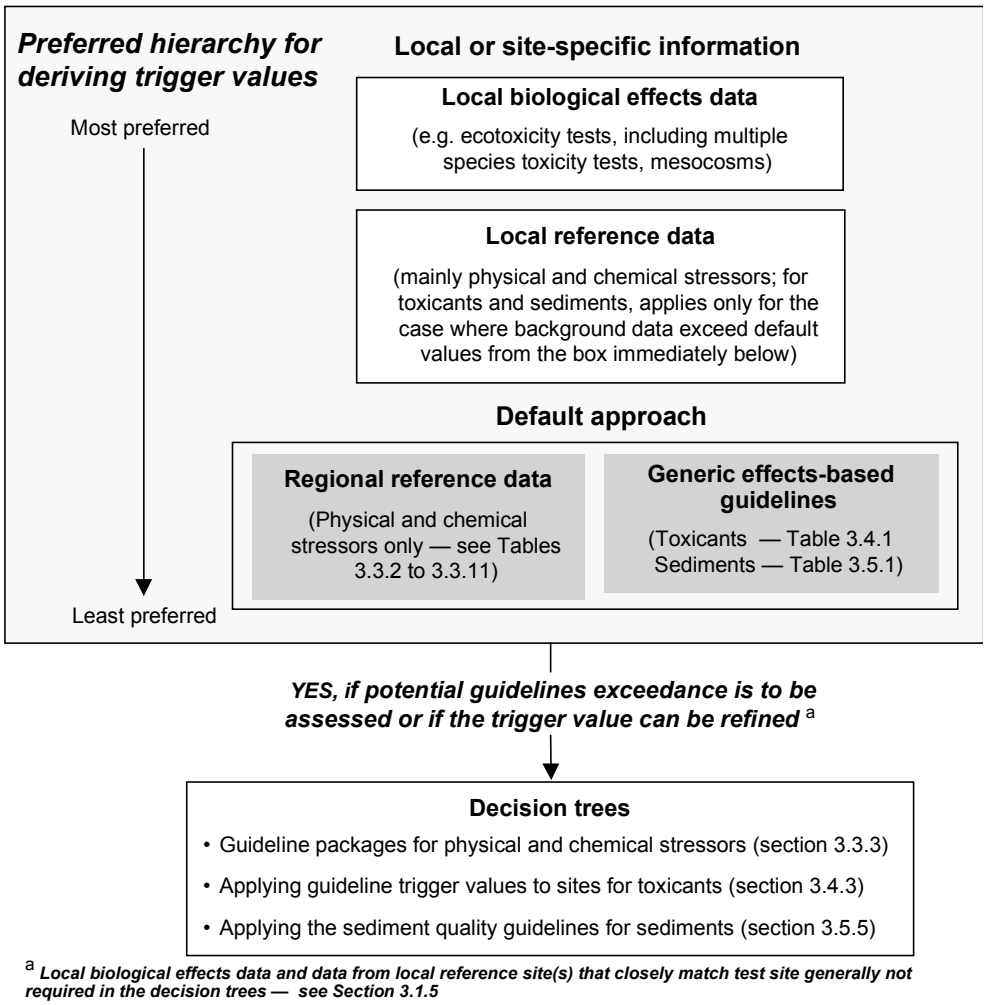


Figure 3.1.2 Procedures for deriving and refining trigger values, and assessing test sites, for physical and chemical stressors and toxicants in water and sediment. Dark grey shading indicates most likely point of entry for users requiring trigger values.

3.1.1.3 Risk-based application of the guidelines

This is the final part of the framework for applying the guidelines. In summary, for each issue (such as toxicity, algal blooms, deoxygenation) or type of water quality indicator (physical/chemical stressor, toxicant and sediment) the Guidelines provide detailed decision frameworks in the form of decision trees or guideline 'packages' for applying the guideline trigger (low risk) values, rather than

simplistic threshold numbers for single indicators. If data from a test site exceed the trigger value, the decision trees are used to determine if the test values are inappropriately (unnecessarily) 'triggering' potential risk and hence management response. For this, ecosystem-specific modifying factors are introduced to assess test data. The decision trees also enable the guideline trigger values to be adjusted and refined. Further introduction to the use of decision trees in this assessment of test site data and refinement of trigger values is provided in section 3.1.5.

While it is not mandatory to use decision frameworks, they are recommended so that the resulting guidelines are relevant to the site. The guideline trigger values are based on bioavailable concentrations, and hence are relatively conservative when compared with total concentrations in the field, so the use of the decision frameworks will increase guideline concentrations in most cases.

For biological indicators a general framework is applied, instead of a decision-tree framework.

3.1.2 Features and classification of aquatic ecosystems in Australia and New Zealand

3.1.2.1 Ecosystem features that may affect water quality assessment and ecosystem protection

There is a diverse range of ecosystem types in Australia and New Zealand, including tropical, temperate, arid, alpine and lowland. Within ecosystem types, waterbodies may be static, flowing or ephemeral, deep or shallow, and fresh, brackish or saline.

Variations in physical and chemical water quality variables can occur naturally through droughts and floods, climatic conditions and erosion events, and can have important consequences for the biota. Variations in climate, and, consequent variations in rainfall, runoff and river flow, are particularly marked in Australia (Finlayson & McMahon 1988, Harris & Baxter 1996, Harris 1996), and are strongly linked to climate variability through mechanisms such as the El Niño–Southern Oscillation or ENSO (Simpson et al. 1993).

Elsewhere in the Guidelines, a comprehensive account of the features of Australian and New Zealand ecosystems is provided, together with some of the consequences of these features that should be taken into account when considering water quality assessment and ecosystem protection.[a] Table 3.1.1 summarises these issues.

a See Appendix 2 (Vol. 2)

3.1.2.2 Classifying the ecosystem

The wide range of geographic, climatic, physical and biological factors that can influence a particular aquatic ecosystem makes it essential that ecosystem management incorporates site-specific information together with more general scientific information relating to ecosystem changes. This is the basis of the new approach to the management of aquatic ecosystems,[b] involving the use of decision frameworks to tailor water quality guidelines to local conditions. A first step in tailoring guidelines to local conditions is to choose an appropriate category of ecosystem; hence the need to classify the ecosystem being monitored.

b See outline in Section 3.1.5

Table 3.1.1 Some features of Australian and New Zealand ecosystems that have possible consequences for water quality assessment and ecosystem protection.

Ecosystem feature	Possible consequence
High degree of endemism amongst the biota of many Australian and New Zealand ecosystems (fresh and marine)	Possible risks to natural heritage and conservation values
Naturally low nutrient status of many of Australia's fresh and marine systems	• Ecosystems are adapted to low nutrient status; (natural) lack of algal grazers for example may mean algal growth/blooms proceed unchecked • Greater accuracy and precision may be required for water sampling programs where early detection of trends in nutrient concentrations is important
Fresh water systems of Australia often dominated by sodium and chloride	Greater 'softness' of these systems places biota at risk from classes of contaminants for which water hardness and acid-buffering capacity may ameliorate toxicity
Water temperatures in Australian aquatic ecosystems are often higher and more varied than those in northern hemisphere ecosystems	More often, toxicity of chemicals increases with increasing temperature — an important consideration given that most toxicity data used in the Guidelines are derived from northern hemisphere studies.
Many of Australia's fresh water systems have only periodic/episodic flow or water availability	• Dilution of contaminants is reduced at low/recessional flow or water levels • After dry periods, oxidative processes can produce degradation products such as acidity that may mobilise deposited contaminants with 'first flush' flows (e.g. oxidation of sulfide deposits) • Classifications based on trophic status, and developed for deep lakes of Northern Hemisphere, unlikely to be applicable to shallow Australian standing waters

Over recent years, there has been considerable activity in classifying ecosystems or parts of them, and this experience has been used to develop the general scheme for these Guidelines. This is a hierarchical classification, with different levels of detail applying to different categories of indicator. For future versions of the Guidelines it is envisaged that this classification will be developed further as knowledge increases, with specific guidelines and protocols being developed for each combination of indicator and ecosystem type. The annex of Appendix 2, Volume 2, describes some of the research in ecosystem classification, with some commentary on recent applications of more detailed schemes in Victoria and New Zealand that may be useful in future revisions of these Guidelines.

The ecosystem classification is given in figure 3.1.3. Note that each of the broad categories of indicators has a different level of detail in terms of the ecosystem classification. Thus for sediments, the guidelines make no distinction between freshwater and marine systems, whereas for chemical and physical stressors there are six categories of ecosystem. This approach has been adopted because different levels of detail are available or applicable to each category of indicator: information about sediment indicators is at a relatively early stage of development whereas chemical and physical stressors have a much longer history of use in water quality monitoring.

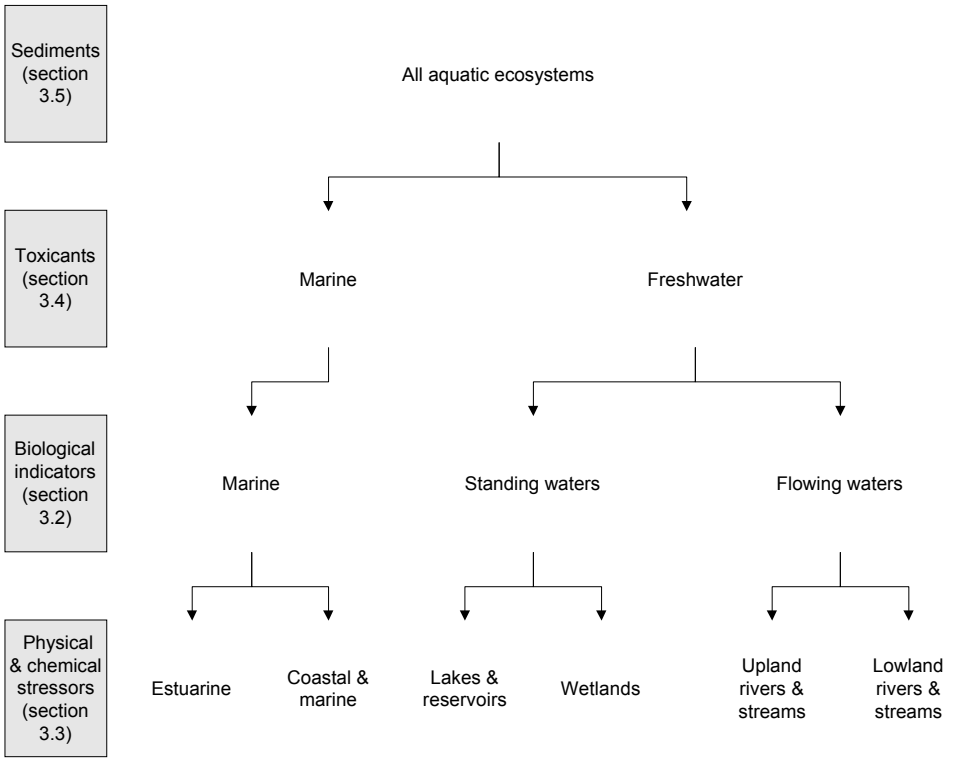


Figure 3.1.3 Classification of ecosystem type for each of the broad categories of indicators (in grey boxes at left of the diagram)

The classification is necessarily coarse. There is no subdivision of estuaries, for example, into those dominated by rivers or by marine influences, or those permanently open to the sea, or temporarily or permanently closed (cf. Hodgkin 1994). Nor is there sufficient information to characterise the water quality requirements of ephemeral rivers or saltwater lakes. Similarly, it should be possible to subdivide these categories on the basis of climate (e.g. tropical vs. temperate), but there is insufficient information available at present about the aquatic ecology of tropical and temperate ecosystems in Australia and New Zealand to make such subdivision meaningful.

Subsequent revisions of the Guidelines should further refine the broad ecosystem classification scheme recommended here. Ideally, within an overall framework of guiding principles and approaches, there should be a separate set of guidelines for each ecosystem type — this should be the long-term aim of the Guidelines.

3.1.3 Assigning a level of protection

To define a level of protection this section describes a hierarchy of ecosystem conditions, and recommends threshold levels of change that are acceptable for each.

The Guidelines also provide data or advice to assist relevant jurisdictions to make their own informed decisions on alternative levels of protection where desired.

3.1.3.1 Ecosystem condition and levels of protection

The previous Guidelines (ANZECC 1992), in describing the concept of *levels of protection*, recognised two categories of aquatic ecosystem condition: (i) pristine or outstanding ecosystems for which maintenance of the existing water quality was

deemed appropriate; and (ii) all remaining ecosystems to which the guidelines would be applied to manage water quality. In this document the concept is extended to acknowledge three categories of ecosystem condition, with a level of protection ascribed to each.

Three ecosystem conditions are recognised.

1. *High conservation/ecological value systems* — effectively unmodified or other highly-valued ecosystems, typically (but not always) occurring in national parks, conservation reserves or in remote and/or inaccessible locations. While there are no aquatic ecosystems in Australia and New Zealand that are entirely without some human influence, the ecological integrity of high conservation/ecological value systems is regarded as intact.
2. *Slightly to moderately disturbed systems* — ecosystems in which aquatic biological diversity may have been adversely affected to a relatively small but measurable degree by human activity. The biological communities remain in a healthy condition and ecosystem integrity is largely retained. Typically, freshwater systems would have slightly to moderately cleared catchments and/or reasonably intact riparian vegetation; marine systems would have largely intact habitats and associated biological communities. Slightly–moderately disturbed systems could include rural streams receiving runoff from land disturbed to varying degrees by grazing or pastoralism, or marine ecosystems lying immediately adjacent to metropolitan areas.
3. *Highly disturbed systems*. These are measurably degraded ecosystems of lower ecological value. Examples of highly disturbed systems would be some shipping ports and sections of harbours serving coastal cities, urban streams receiving road and stormwater runoff, or rural streams receiving runoff from intensive horticulture.

The third ecosystem condition recognises that degraded aquatic ecosystems still retain, or after rehabilitation may have, ecological or conservation values, but for practical reasons it may not be feasible to return them to a slightly–moderately disturbed condition.

A level of protection is a level of quality desired by stakeholders and implied by the selected management goals and water quality objectives for the water resource. The water quality objectives may have been derived from default guideline values recommended for the particular ecosystem condition, or they may represent an acceptable level of change from a defined reference condition; it can be formalised as a critical effect size.[a] Where appropriate, the reference condition is defined from as many reference sites as practicable using pre-impact data where appropriate.[b] The reference condition could correspond to one of the three recognised condition levels described above, depending upon the desired level of protection.

a *See box 2.3 & Section 3.1.7*
b *Section 3.1.4*

Key stakeholders in a region would normally be expected to decide upon an appropriate level of protection through determination of the management goals and based on the community's long-term desires for the ecosystem. The philosophy behind selecting a level of protection should be (1) maintain the existing ecosystem condition, or (2) enhance a modified ecosystem by targeting the most appropriate condition level. (Thus the recommended level of protection for 'condition 1 ecosystems' (above) would be *no change*[c] beyond any natural variability.) This is

c *Footnote 2 on page 2-9*

the starting point from which local jurisdictions might negotiate or select a level of protection for a given ecosystem: in doing so, they might need to draw upon more than the general scientific advice[a] provided in these Guidelines. A number of other factors, such as those of a socio-economic nature, might need to be included in the decision making process.

a *See Section 2.1.3*

3.1.3.2 A framework for assigning a level of protection

When stakeholders are deciding upon an appropriate level of protection for ecosystems, it is suggested that they consider the following framework based on the three ecosystem conditions recognised above.

Some waters (e.g. many of those in national parks or reserves) are highly valued for their unmodified state and outstanding natural values (condition 1 ecosystems).[5] In many countries and in some Australian states these waters are afforded a high degree of protection by ensuring that there is no reduction in the existing water quality, irrespective of the water quality guidelines (ANZECC 1992).

The present Guidelines recommend that for condition 1 ecosystems the values of the indicators of biological diversity should not change markedly. To meet this goal, the decision criteria for detecting a change should be ecologically conservative and based on sound ecological principles.[b] Moreover, a precautionary approach is recommended — management action should be considered for any apparent trend away from a baseline, or once an agreed threshold has been reached. Any decision to relax the physical and chemical guidelines for condition 1 ecosystems should only be made if it is known that such a degradation in water quality will not compromise the objective of maintaining biological diversity in the system. Therefore, considerable biological assessment data would be required for the system in question, including biological effects and an ongoing monitoring program based on sufficient baseline data. The nature of contaminants expected in the receiving waters might also affect decisions on this issue.[c] Where there are few biological assessment data available for the system, the management objective should be to ensure no change in the concentrations of the physical and chemical water quality variables beyond natural variation.

b *Sections 3.2.1.1, 3.1.7 and 7.2.3.3*

c *Section 3.1.3.3*

Where data for a reference/control site have only been collected for a limited period and the reference condition cannot be clearly characterised, the power of detection should be increased by using more indicators, and/or more reference/control sites and/or more monitoring sites placed along any probable disturbance gradients.

For slightly to moderately disturbed ecosystems (‘condition 2 ecosystems’), some relaxation of the stringent management approach used for condition 1 ecosystems may be appropriate. An increased level of change might be acceptable, or there might be reduced inferential strength for detecting any change in biological diversity. Nevertheless, as for condition 1 ecosystems, maintenance of biological diversity relative to a suitable reference condition should be a key management goal. The Guidelines provide specific guidelines for biological indicators for each

5 While waters in many remote and inaccessible locations may retain an unmodified condition, the level of protection assigned to these systems is a jurisdictional decision made in consultation with stakeholders. It does not automatically follow that these waters default to ‘condition 1 ecosystems’.

of the three ecosystem conditions.[a] For the other types of water quality indicator, the default guidelines in Sections 3.3–3.5 provide a suitable level of protection for condition 2 ecosystems.

a See Section 3.2.4

The situation for highly disturbed ecosystems ('condition 3 ecosystems') can be more flexible. The general objective might be to retain a functional, albeit modified, ecosystem that would support the management goals assigned to it. In most cases the ecological values of highly disturbed ecosystems can be maintained by the direct application of the guidelines contained in this chapter. However, there could be situations where these guidelines would be too stringent and a lower level of protection would be sought. Some guidance to assist managers in these situations is provided in the discussion of each indicator type.[b]

b Sections 3.1.8 & 3.2 to 3.5

Table 3.1.2 summarises a general framework for considering levels of protection across each of the indicator types for each of the ecosystem conditions.

The three levels of protection described above form just one practical but arbitrary approach to viewing the continuum of disturbance across ecosystems. Inevitably, stakeholders in different jurisdictions, catchments or regions will make different judgements about ecosystem conditions. For example, an ecosystem that is regarded as highly disturbed in one area could be regarded as only slightly to moderately disturbed in a more populated region. This makes it imperative, as emphasised in these Guidelines, that the setting of levels of protection is carried out in an open and transparent way, involving all key stakeholders, so that a fair and reasonable outcome is achieved.

Note that even though a system is assigned a certain level of protection, it does not have to remain 'locked' at that level in perpetuity. The environmental values and management goals (including level of protection) for a particular system should normally be reviewed after a defined period of time, and stakeholders may agree to assign it a different level of protection at that time. However, the concept of continual improvement should be promoted always, to ensure that future options for a water resource are maximised and that highly disturbed systems are not regarded as 'pollution havens'.

3.1.3.3 Alternative levels of protection

Local jurisdictions may negotiate alternative site-specific levels of protection after considering factors such as:

- whether a policy of 'no release' (total containment) of contaminants applies;
- the nature of contaminants that might reach aquatic ecosystems. (Greater consideration might be given to those ecosystems receiving contaminants or effluents of potentially high toxicity and which are persistent in the environment, e.g. metals. Alternatively, differing levels of protection could apply according to the anticipated capacity of an ecosystem to readily recover from impact if contamination is to be of short duration.)
- perceived conservation/ecological values of the system additional to those recognised in the simple classification of ecosystem condition described in Sections 3.1.2 and 3.1.3.1.

Table 3.1.2 Recommended levels of protection defined for each indicator type

Ecosystem condition	Level of protection			
	Biological indicators	**Physical & chemical stressors**	**Toxicants**	**Sediments**
1 High conservation/ ecological value	• No change in biodiversity beyond natural variability. Recommend ecologically conservative decision criteria for level of detection. • Where reference condition is poorly characterised, actions to increase the power of detecting a change recommended. • Precautionary approach recommended for assessment of post-baseline data through trend analysis or feedback triggers.	• No change beyond natural variability recommended, using ecologically conservative decision criteria for detecting change. Any relaxation of this objective should only occur where comprehensive biological effects and monitoring data clearly show that biodiversity would not be altered. • Where reference condition is poorly characterised, actions to increase the power of detecting a change recommended. • Precautionary approach taken for assessment of post-baseline data through trend analysis or feedback triggers.	• For toxicants generated by human activities, detection at any concentration could be grounds for investigating their source and for management intervention[1]; for naturally-occurring toxicants, background concentrations should not be exceeded. *Where local biological or chemical data have not yet been gathered*, apply the default values provided in sec 3.4.2.4. Any relaxation of these objectives should only occur where comprehensive biological effects and monitoring data clearly show that biodiversity would not be altered. • In the case of effluent discharges, direct toxicity assessment (DTA) should also be required. • Precautionary approach taken for assessment of post-baseline data through trend analysis or feedback triggers.	• No change from background variability characterised by the reference condition. Any relaxation of this objective should only occur where comprehensive biological effects and monitoring data clearly show that biodiversity would not be altered. • Precautionary approach taken for assessment of post-baseline data through trend analysis or feedback triggers.
2 Slightly to moderately disturbed systems	• Negotiated statistical decision criteria for detecting departure from reference condition. Maintenance of biodiversity still a key management goal. • Where reference condition is poorly characterised, actions to increase the inferential strength of the monitoring program suggested. • Precautionary approach may be required for assessment of post-baseline data through trend analysis or feedback triggers.	• Always preferable to use data on local biological effects to derive guidelines. *If local biological effects data unavailable*, local or regional reference site data used to derive guideline values using suggested approach in sec 3.3.2.3. Alternatives to the default decision criteria for detecting departure from reference condition may be negotiated by stakeholders but should be ecologically conservative and not compromise biodiversity. *Where local reference site data not yet gathered*, apply default, regional low-risk trigger values from sec 3.3.2.5. • Precautionary approach may be required for assessment of post-baseline data through trend analysis or feedback triggers.	• Always preferable to use data on local biological effects (including DTA) to derive guidelines. *If local biological effects data unavailable,* apply default, low-risk trigger values from sec 3.4.2.4. • Precautionary approach may be required for assessment of post-baseline data through trend analysis or feedback triggers. • In the case of effluent discharges DTA may be required.	• The sediment quality guidelines provided in sec 3.5 apply. • Precautionary approach taken for assessment of post-baseline data through trend analysis or feedback triggers.
3 Highly disturbed systems	• Selection of reference condition within this category based on community desires. Negotiated statistical decision criteria for detecting departure from reference condition may be more lenient than the previous two condition categories.	• Local or regional reference site data used to derive guideline values using suggested approach in sec 3.3.2.3. Selection of reference condition within this category based on community desires. Negotiated statistical decision criteria may be more lenient than the previous two condition categories. *Where local reference site data not yet gathered,* apply default, regional low-risk trigger values from sec 3.3.2.5; or use biological effects data from the literature to derive guidelines.	• Apply the same guidelines as for 'slightly–moderately' disturbed systems. However, the lower protection levels provided in the Guidelines may be accepted by stakeholders. • DTA could be used as an alternative approach for deriving site-specific guidelines.	• Relaxation of the trigger values where appropriate, taking into account both upper and lower guideline values. • Precautionary approach may be required for assessment of post-baseline data through trend analysis or feedback triggers.

[1] For globally-distributed chemicals such as DDT residues, it may be necessary to apply background concentrations, as for naturally-occurring toxicants.

3.1.4 Defining a reference condition

For some water quality indicators, users will need to define a *reference condition* that provides both a target for management actions to aim for and a meaningful comparison for use in a monitoring or assessment program. The reference condition is particularly appropriate to condition 2 or condition 3 ecosystems, and is a key component of the framework provided in figure 3.1.1[a] for applying the guidelines. For biological indicators, and for physical and chemical stressors where no biological or ecological effects data are available, the preferred approach to deriving guideline trigger values is from local reference data; for toxicants in water or sediment this reference condition, sometimes called *background data*, may in some situations supplant the default guideline values.[b] The next sections summarise the sources of information that can be used for defining a reference condition, and clarify the terminology of 'controls' and what constitutes a 'site', respectively. Chapter 7 describes the design of monitoring programs, but also see the Monitoring Guidelines (ANZECC & ARMCANZ 2000).

a See Section 3.1.1.2

b See also Sections 3.4.3.2, 7.4.4.2, 7.4.4.4

3.1.4.1 Sources of information

The reference condition for sites that may or may not be disturbed at present can be defined in terms of these sources of information: historical data collected from the site being assessed; spatial data collected from sites or areas nearby that are uninfluenced (or not as influenced) by the disturbance being assessed; or data derived from other sources.

1. Historical data collected from the site being assessed will usually represent measurements made before a disturbance or before management actions. For example, measurements of salinity collected from a river before the initiation of an irrigation scheme may be used to set the reference condition for salinity that stakeholders would hope to achieve in a rehabilitation program. For cases where rehabilitation of degraded systems can only be achieved over long time-scales, such benchmarks may be progressively stepped by way of a series of targets intermediate between the existing and pre-disturbance condition.

2. Spatial data can be collected from reference sites or areas nearby that are relatively uninfluenced by the disturbance being assessed. The sites include, but are not restricted to, *control* sites which are identical in all respects to the site being assessed (sometimes called the *test site*) except for the disturbance (the distinction between control and reference sites is explained more fully below). For example, the impact of an ocean outfall on marine benthos may be judged relative to the values of the selected indicators in one or more reference sites that are in the same vicinity but lack any influence of an outfall. For modified ecosystems, 'best-available' reference sites may provide the only choice for the reference condition.[c]

c Section 3.1.8

3. Data can be derived from other sources if there are neither suitable historical data nor comparable reference sites. The reference condition may be identifiable from the published literature, from models, from expert opinion, from detailed consultations with stakeholders, or from some combination of all of these. For example, when setting the reference condition for nutrient concentrations in a series of wetlands, information on desirable and attainable concentrations may come from published studies from similar regions overseas, from nutrient models

with appropriate local adaptations, from scientific advice about what levels of nutrients result in undesirable end-points (e.g. blooms of toxic cyanobacteria) and from input from community groups and landholders about their expectations of what the wetlands should become. The necessary negotiations need considerable technical and social skill. The reference condition should not be defined in terms of ecological targets that are impossible to attain. Conversely, the reference condition should represent a substantial achievement in environmental protection that is agreeable to the majority of stakeholders.

Obviously, the best reference conditions are set by locally appropriate data. If the disturbance to be assessed has not yet occurred, then pre-disturbance data provide a valuable basis from which to define the reference condition. If the disturbance has already occurred then data from reference sites and other appropriate sources can be used to define the reference condition.[a] These issues are treated in more depth in the Monitoring Guidelines (ANZECC & ARMCANZ 2000).

a See Section 3.1.8

In summary, the reference condition must be chosen using information about the physical and biological characteristics of both catchment and aquatic environment to ensure the sites are relevant and represent suitable target conditions. Some of the important factors that should be considered are these:

- data collected prior to the disturbance need to be of sufficient quality and timespan to provide valid comparisons with post-disturbance data;[b]
- where possible, pre-disturbance data should be collected from appropriate control or reference sites as well as from the site(s) subjected to the disturbance;
- the definition of a reference condition must be consistent with the level of protection proposed for the ecosystem in question — unimpacted, or slightly modified or relatively degraded (where the community does not wish to rehabilitate a degraded ecosystem to such a high level);
- sites should be from the same biogeographic and climatic region;
- reference site catchments should have similar geology, soil types and topography;
- reference sites should contain a range of habitats similar to those at the test sites;
- reference and test sites should not be so close to each other that changes in the test site due to the disturbance also result in changes in the reference sites, nor, conversely, should changes in the reference sites mask changes that might be occurring in the test site.

b Section 7.2.3.1 & the Monitoring Guidelines

3.1.4.2 Clarification of the terms ‘control’ and ‘reference’

In the context of monitoring and assessing water quality, a disturbance (or ‘treatment’) is an event or occurrence which may or may not result in an effect on a water body, and the ‘control’ refers to a set of observations taken from conditions identical to the disturbed conditions except for the disturbance.

Controls may be defined in terms of space (‘spatial controls’) or time (‘temporal controls’) or both. For example, if stakeholders had to assess the effect of urbanisation on a wetland, they might be able to find similar wetlands nearby with no urban development in their catchments, to act as spatial controls. If development

had not commenced, the stakeholders could collect data from the wetland at this stage to use as a temporal control, and the inferences that they could make about the effects of urbanisation on the wetland would be strongest if they collected data from the spatial controls before and after urbanisation as well.[a]

a See Section 7.2

In environmental science, as in classical field experiments, 'controls' are unlikely to be completely identical to 'treatments'. If there is important systematic variation between 'controls' and 'treatments', this can be incorporated into the sampling program and statistical analysis via regression-related techniques. Analysis of covariance is one classical technique for handling such differences. Some statistical textbooks refer to these procedures as methods of *statistical control* (which should not be confused with *statistical process control* or *control charting*).

Sometimes controls are impossible to find, but there are still sites or sets of temporal observations that represent a desirable set of conditions that the disturbed site(s) could ultimately match, if rehabilitated. Thus the term *reference condition* or *reference site* denotes something more general than the 'control'. In the wetland example above, there may be no wetlands on similar soil types that are completely free of urbanisation, and even those with little urbanisation may differ in the dominant land-use in their catchments. In this instance, stakeholders would need to negotiate over which wetlands would provide the most appropriate reference conditions.

The use of reference sites to establish targets on a broader regional scale is becoming increasingly popular. For example, this method is the basis of the national rapid biological assessment procedure adopted for the AUSRIVAS program (Schofield & Davies 1996). In this case, reference sites are usually selected in ecosystems that are similar to and in the vicinity of a test ecosystem but unimpacted or little changed.

3.1.4.3 What constitutes 'a site'

For the purposes of these Guidelines, a *site* refers to a location which is being monitored or assessed, and constitutes the smallest spatial unit that will be used in judging whether an impact has occurred. Thus a site may vary in size from a few square metres, as in the case of a stretch of an upland stream, to a few square kilometres, as in the case of a large seagrass bed. In the case of the upland stream, stakeholders may be interested in monitoring the water quality of the site and comparing it with, for example, several other reference sites on other streams nearby. For the large seagrass bed, selected indicators might be measured in that bed and compared with measures from similar seagrass beds elsewhere on the coast.

Only rarely will sites be homogeneous internally. Concentrations of chemicals may vary across a stream, and there may be differences in the sediments and species composition across a seagrass bed. There are a number of strategies for dealing with such *within-site variation*.[b] For large sites, this may involve sampling at more than one spatial scale within the site. For example, in the seagrass bed, several sampling locations of, say, 100 m^2 may be selected, within which smaller 'sub-locations' (e.g. 1 m^2 quadrats) may be selected. Care needs to be taken not to confuse these within-site spatial units with the site itself. Note that in the literature there is little consistency in the use of terms such as 'site', 'location', 'area', etc., so readers should not assume that the term 'site' in other publications automatically equates with the term 'site' as it is used in these Guidelines and in the Monitoring Guidelines (ANZECC & ARMCANZ 2000).

b See Ch 7 and the Monitoring Guidelines

3.1.5 Decision frameworks for assessing test site data and deriving site-specific water quality guidelines

The effect of a particular stressor or toxicant on biological diversity or ecological integrity depends upon three major factors:

- the nature of the ecosystem, its biological communities and processes;
- the type of stressor;
- the influence of environmental factors which may modify the effect of the stressor.

Aquatic ecosystems are variable and complex and difficult to manage. The previous Guidelines recognised the need to address this variability and the influence of environmental factors on stressors. This section introduces the concept of managers using risk-based decision frameworks to assess test site data and to tailor guidelines to suit regional, local or site-specific conditions. It provides a consistent framework that can be used in New Zealand and the states and territories of Australia for applying the guidelines in a meaningful way to the various types of aquatic ecosystems in these regions. The approach addresses the issues of variability and complexity, more realistically and effectively protecting biodiversity or ecological integrity. As emphasised above, the approach does not constitute or require a full risk assessment,[a] but simply assists in providing a site-specific estimate of whether a stressor represents a low, possible or high risk to the aquatic ecosystem of interest.[b]

a See Section 2.1.4

b As indicated in figures 3.3.1, 3.4.1, 3.5.1

As already discussed, for non-biological indicators, these Guidelines recommend *guideline trigger values*, which represent bioavailable concentrations or unacceptable levels of contamination[6] and are equivalent to the old single number guidelines. If exceeded, these values *trigger* the incorporation of additional information or further investigation to determine whether or not a real risk to the ecosystem exists and, where possible, to adjust the trigger values into regional, local or site-specific guidelines. The decision frameworks in Sections 3.3–3.5 demonstrate how this can be done.

Through the decision frameworks the ambient (existing) concentration of a contaminant is compared with the guideline trigger value. The initial measurement may be a relatively simple and therefore low-cost measurement (e.g. total concentration). If the trigger value is not exceeded, the risk of an impact is low and no further action is required. However, if the trigger value is exceeded there is some risk of an impact occurring and successive, more complex steps should be taken to account for environmental factors that modify the bioavailability, biological uptake or toxicity of the stressor; this would also entail considering more complex monitoring designs and negotiating effect sizes explicitly with stakeholders.[c] The final guideline for that parameter should therefore reflect the real hazard to the particular ecosystem.

c Sections 7.2 and 3.1.7

At each step in the process, a decision must be made on whether the adjusted trigger value should be modified further or accepted. In general, the further one travels down the series of steps the more resource-intensive the steps become; the user should consider costs vs. benefits for each step. At any stage the decision tree process can be

6 Formally, the guideline trigger values are held to be a default, conservative statement of the *critical effect size* as explained in section 3.1.7.

terminated and the most recently modified trigger value applied as the guideline for the particular situation. Because the default trigger values for toxicants at least are conservative, a precautionary approach should be applied, using these values where there is no background information on a particular system to which the guidelines are to be applied, and no program for its acquisition. Alternatively the preferred option might be to conduct toxicological studies or direct toxicity assessment relevant to the site and use these data to derive a site-specific guideline.

Where a trigger value is refined using data gathered from a test site on a single or limited sampling occasion(s), this does not automatically mean that this new value applies henceforth in further test site/trigger value comparisons. More extensive information is required before a guideline trigger value can be revised. For this, it is important to distinguish two levels of refinement of guideline trigger values:

1. The first level applies to some indicators where guideline trigger values can be adjusted and refined upfront, relatively simply, with fore-knowledge of the range of values of some key physical and chemical parameters that occur in a waterbody. This is particularly relevant to some toxicants. For example, the toxicity and bioavailability of some metals (e.g. copper, zinc and cadmium) are strongly influenced by water quality conditions such as hardness, dissolved organic matter and pH, and recent literature has increased the understanding of the toxicity of different metal species. The current state of knowledge limits upfront revision of the trigger values for these metals to a hardness correction, using the simple algorithms in table 3.4.3. There is also some scope for modifying the trigger values for a few non-metallic inorganic and organic toxicants, based on associated water quality parameters (e.g. pH, for the ammonia trigger value). [a]

a See Sections 3.4.3, 3.5.5

2. For most indicators and issues, however, trigger values are refined only after continuous and extensive monitoring shows that test site data exceedances are consistently assessed as posing no risk to the ecosystem, using the decision trees. Trigger values can also be refined if longer term monitoring shows that test site data are consistently *below* the trigger values or, for situations such as naturally mineral-rich waters where the natural background total concentrations of some metals exceed the new trigger values. For each of these cases, the methods described in section 7.4.4.2/1 can be used to refine the guideline trigger values for all (non-biological) indicator types.

It is not mandatory to use the decision frameworks, but they are important if meaningful and appropriate guidelines are to be applied. Moreover, simple adjustments and corrections such as those described in 1 above make this a cost-effective exercise where data on key water quality parameters are available.

Generally, local biological effects data and data from local reference site(s) that closely match the test site[7] are not required in the decision trees. If test site data exceed trigger values that have been derived from these local data, this would

[7] This latter situation might be relevant to point-source disturbances in streams, where reference sites are located upstream of test sites; the reference and test sites would be similar in all appearances and there would be no confounding factors, apart from the disturbance and stressor in question, occurring between the sites. Local reference sites even in an adjacent stream/tributary might not necessarily closely match test sites.

normally trigger management action because these locally-derived trigger values already have ecosystem-specific modifying factors built into them. For the same reason, these locally-derived trigger values do not require refinement themselves through the decision trees, though if there was opportunity to derive guideline values based upon sound local biological effects data, these should replace those based upon local reference data.

These decision frameworks have not been developed for all specific indicators and issues but are presented mainly to assist water managers explore some of the ways in which the guidelines can be used in site-specific situations. Water managers and regulators are encouraged to develop their own decision trees to address any additional issues that may be encountered. General guidance on designing monitoring and assessment programs is given in Chapter 7, with additional background in the Monitoring Guidelines (ANZECC & ARMCANZ 2000).

3.1.6 Using management goals to integrate water quality assessment

In general, there is not enough scientific knowledge at present to allow anyone to make confident predictions about the way in which a particular concentration of toxicant or nutrient will affect species, habitats or ecosystems. It is therefore important to measure the characteristics of the biological components of the ecosystem as well as the physical and chemical water quality characteristics, to be able to confidently assess whether an important change has occurred or is likely to occur.

Although there is a considerable body of toxicological knowledge that is very important for use in specific circumstances, the overall effects of mixtures of toxicants on a wide variety of species or habitats are not fully understood. Environments are typically dynamic, as well as being subjected to natural stresses like storms and floods, and little is known about the highly complex internal forces that operate within them. Relatively accurate predictive models can be developed for specific ecosystems,[a] but this generally entails sophisticated, resource-intensive programs which may not be feasible. Use of unproven or overly simplistic causal models to justify avoiding using biological indicators is dangerous.

a See Section 2.2.3

The process of setting management goals,[b] as outlined earlier, is useful for conceptualising the issues surrounding integration in aquatic ecosystem management. The goals should be defined in a quantitative manner, need to be comprehensively related to all valued attributes of the ecosystem, and, typically, should be biologically based. In this sense, the biological variables themselves are the management end-points, and chemical variables such as concentrations of toxicants are the proximal causes in the cause–effect relationship. Management is then directed to these management goals (such as maintaining a certain level of species diversity). All management and assessment activities are integrated by an explicit relationship to the management goals, in this case the maintenance and improvement of species diversity. Hence biological diversity, or some other valued aspect of the ecosystems, becomes the target for management and assessment, and all activities are defined and implemented in terms of management of those ecosystem attributes (Ward & Jacoby 1992).

b Section 2.1.3

Overall, the aim of a monitoring program should be to answer a discrete set of questions (hypotheses)[c] which focus on whether the management goals are being achieved. Conceptual models of the important biological and physical interactions

c Sections 3.1.7, 7.1.2 and 7.2.3.3

within the ecosystem will assist in choosing those indicators that could be potentially useful for the monitoring or assessment program. This is important because monitoring programs must be cost effective and in most circumstances it is not feasible to design and implement a program that intensively monitors all aspects of water quality.

Another important aspect of integrated water quality assessment is the development of communication networks across whole catchments to address broad-scale issues. This is essential at two levels: first, because of the interdependent nature of the environmental values themselves — the water quality of one value can potentially affect others;[a] second, for protection of the whole aquatic ecosystem — while water quality objectives might be met in riverine ecosystems upstream, the cumulative effects of discharges and contaminant build up in depositional areas downstream (e.g. wetlands, estuaries) must also be considered when setting water quality criteria. This applies to a number of environmental values.[b]

a See Section 2.1.3

b Section 7.4.4.3 for related discussion

3.1.7 Decision criteria and trigger values

Indicators used in these Guidelines are likely to respond continuously to the intensity of a disturbance; an example is given in figure 3.1.4. At some point along this continuum, the ecosystem will be deemed to have been adversely affected and the value of the indicator at this point will be used as the criterion to make the decision that 'the ecosystem has been impacted'.

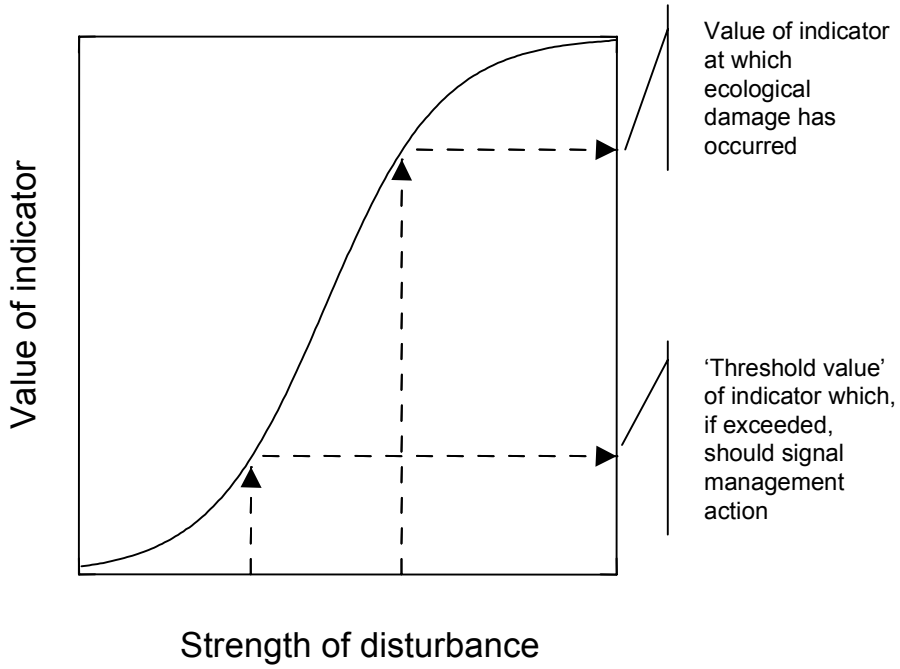


Figure 3.1.4 Graphical depiction of the relationship between indicator response and strength of disturbance, and threshold for management intervention

In most situations, we will need to make a decision *before* the ecosystem becomes adversely affected so that management actions can be implemented in time to prevent the ecosystem becoming damaged. In other words, we will need to select a 'threshold value' of the indicator that is *smaller* than that which indicates that the

ecosystem has been impaired. How much smaller this value needs to be depends on the nature of the impact, the level of our understanding of the relationship between changes in the indicator and ecological impact, and the lead-time necessary to implement management actions.

For example, if the impact is likely to be irreversible or persistent then the threshold value will need to be set at a very small value of the indicator so that irreversible harm is avoided. Also, if there is only a very rudimentary understanding of how a particular contaminant might affect an ecosystem then the threshold value will need to be relatively small in case the ecosystem is more sensitive to the contaminant than expected. Similarly, if there is a long lag between detection that the threshold has been exceeded and implementation of some action or decision, the threshold value will need to be set at a very small value.

Thus, the first task is to choose the threshold value for a given indicator. This is not a trivial exercise, and requires all stakeholders to agree on these values before the program of monitoring or assessment commences.

For the non-biological indicators in Sections 3.3–3.5, the guideline trigger values represent the best currently-available estimates of what are thought to be ecologically low-risk levels of these indicators for *chronic (sustained) exposures*.[a] For these indicators, the guideline trigger values provide the starting point for negotiations about the threshold value and criterion for a management decision (i.e. water quality objectives). Users should also be aware that short-term intermittent (or pulse) exposures to very high contaminant or stressor values may also need to be managed in certain situations. Negotiating the equivalent of a guideline trigger value for the biological indicators in Section 3.2 is more complex, because the use of these indicators has a shorter history in Australia and New Zealand and because these indicators nearly always need to be used in a comparative fashion (e.g. comparing values from the site(s) of interest with those in an appropriate reference condition). This may also be true for the non-biological indicators in situations where a reference condition is being used to establish the water quality objectives.

a See Section 7.4.4

Thus, for all types of indicators, there will be situations in which simple guideline trigger values of the chosen indicator will be inadequate as a threshold value or criterion on which to activate management decisions and actions. In these situations, stakeholders need to negotiate an *effect size*, which describes how much deviation from the reference condition is tolerable before management has to intervene. To understand what an effect size is, stakeholders need to appreciate the following points:

1. the values of all indicators vary naturally, and
2. not all of this variation is ecologically important.

This means that some of the changes that can potentially be detected in an indicator may be ecologically trivial; such small changes should not initiate management action. The situation where we conclude that an important change has happened when, in fact it has not, is technically referred to as a *Type I error*.

Conversely, many indicators are very variable naturally and intensive sampling may be essential to detect ecologically important changes in the indicator. If the sampling intensity is too small and the important change is missed, then a *Type II error* is committed.

In the context of cooperative best management, stakeholders need to balance these two types of 'error' and negotiate these issues before the monitoring or assessment program commences.[a]

a See also box 2.3; these issues are expanded in Section 7.2.3

3.1.8 Guidelines for highly disturbed ecosystems

Apparently common problems in assessing water quality for highly disturbed ecosystems of Australia and New Zealand include:

1. the difficulty in deciding upon suitable water quality guidelines and objectives (and in particular, a level of acceptable ecological change);
2. the lack of suitable reference sites or data;
3. the lack of advice and guidelines for highly disturbed ecosystems in urban regions.

These Guidelines offer the following advice and information on these issues.

3.1.8.1 Determining water quality guidelines and objectives

As discussed in Sections 1.2 and 2.2, the philosophy espoused in the Guidelines is one of 'continual improvement' for places where water or sediment quality is poorer than the agreed water quality objectives. For highly-disturbed ecosystems, the water quality objectives can be seen as progressive and intermediate targets for long-term ecosystem recovery. The Guidelines offer specific advice on assessing the success of remediation programs.[b]

b Sections 3.2.5 & 7.2.3.3

The Guidelines recommend that guideline trigger values for slightly–moderately disturbed systems also be applied to highly disturbed ecosystems wherever possible. If that is not possible, local jurisdictions and relevant stakeholders must negotiate alternative values. For this situation, the Guidelines provide less conservative trigger values for toxicants: the less conservative values suit two lower levels of ecosystem protection (table 3.4.1). The Guidelines also offer the following advice, relevant to all indicators (biological, physical and chemical, toxicants, sediments) when test data are being compared with data from reference sites:[c]

c See also Sections 3.1.4 and 3.1.8.2

1. Where reference sites of high quality are available, lower levels of protection may be negotiated for the site under consideration, through selection of more relaxed statistical decision criteria. This would not necessarily, and should not, result in a water of lesser quality than that already prevailing.
2. Where no high quality reference sites are available, modified water bodies of the best environmental quality in the region serve as reference targets (or intermediate targets for ecosystem recovery). Where these data indicate that certain toxicants occur naturally at levels exceeding the guideline trigger value, the Guidelines make provision for the background level, if clearly established, to become the site-specific guideline level.

Where a reference condition is used to define water or sediment (pore water) quality targets, the bioavailable fraction must be determined and compared for those toxicants that exceed the guideline trigger values.[d] For sediment particulates, the dilute-acid-extractable (1M HCl) fraction is used as a surrogate for bioavailability.[e]

d Sections 3.4 and 3.5

e Section 3.5.5.2

Negotiating the 'acceptable' level of change for disturbed ecosystems, and hence the level of protection of species, is a constant challenge faced by local jurisdictions and relevant stakeholders (including the community).

a See section 8.5.1 in Vol. 2 and Section 7.2.3.3

As is recognised in the Guidelines, more research is needed to develop methods to describe degrees of acceptable ecological change relative to reference conditions.[a] The Guidelines give general advice for determining the size of ecological change that would be considered important. It can be useful to examine data from existing impacts elsewhere, especially if it is possible to compare impacts across a gradient from mild to extreme. These can be used as yard-sticks to decide upon the degree of ecological change or impact.

As a first step towards improvement in water quality, the Guidelines recommend that local jurisdictions assess a range of options for determining site-specific guideline values for highly disturbed ecosystems. One approach is to select different levels of acceptable change (e.g. protection of 90% of species with 50% confidence). Another is to assess the disturbed ecosystem against the best-available reference water body in the region, as a benchmark for water quality.

b Section 3.4.3

Different site-specific guideline values developed using various methods can be examined and weighted according to pre-determined criteria of quality and relevance to the ecosystem. This should be done in a manner consistent with risk assessment principles,[b] to arrive at an appropriate figure.

3.1.8.2 Lack of suitable reference sites or data

Often, water bodies over large continuous tracts of Australia and New Zealand are highly disturbed and none of the adjacent water bodies is necessarily of better quality than the water body(ies) of interest, insofar as serving as useful reference sites. Nevertheless, even if water bodies of only slightly better quality can be found, these provide useful reference data, particularly if these data serve as an intermediate target for ecosystem recovery.

c Sections 3.2, 7.2.1 and 7.3.3

Where the issue is biological assessment of water quality in highly-disturbed inland streams and rivers, rapid assessment using macroinvertebrate communities offers, potentially and in practice, a most useful approach.[c] Recent findings from the Australian Commonwealth-funded National River Health Program from which this rapid assessment approach has been developed, indicate that macroinvertebrate communities are very similar at the family level across vast tracts of inland Australia. This means that relatively intact ecosystems in remote and less developed parts of inland Australia (e.g. channel country of south-western Queensland) may potentially provide useful reference data for highly disturbed ecosystems in, say, north-western NSW, if family-level information about macroinvertebrates serves as a suitable indicator of river health at this spatial scale.

3.1.8.3 Guidelines for highly disturbed ecosystems in urban regions

Most of the populace of Australia and New Zealand lives in large cities where most, but not all, natural aquatic ecosystems are highly disturbed. Approaches from Section 3.1.8.1 above, 'Determining water quality guidelines and objectives', are applicable to the development of guidelines for highly disturbed ecosystems in urban regions. Indeed, a great deal of work has been conducted in urban waterways across Australia and New Zealand and on a variety of chemical and biological monitoring and assessment programs — see box 3.1.4. Utilities in many of the

smaller, and therefore less well-resourced, urban centres will be able to benefit from these larger urban programs by applying the same principles of investigation to their own situations.

Box 3.1.4 Examples of water quality assessment programs conducted in major urban regions of Australia

These are some of the existing monitoring and research programs in streams, estuaries and coastal systems in major urban centres.

For urban streams and wetlands:

- Sydney streams are monitored and studied through the Environmental Indicators program of Sydney Water Corporation, and by NSW DLWC;
- Melbourne streams are monitored and studied by Melbourne Water, VIC EPA and the CRC for Freshwater Ecology;
- a predictive model of the AUSRIVAS type for monitoring and assessing health of streams in the Hobart region has been completed by the University of Tasmania (Zoology Dept);
- wetlands of the Swan Coastal Plain.

For coastal marine areas and estuaries:

- water quality monitoring and assessment are included amongst the research programs of the Centre for Research on Ecological Impacts of Coastal Cities (Sydney University);
- Port Phillip Bay Environmental Study;
- Moreton Bay;
- programs in and around Perth, such as the Perth Coastal Water Study, South Metropolitan Coastal Water Studies, Perth Coastal Waters Management and Consultative Process.

General:

- Thirteen studies on streams and estuaries were commissioned under the Urban sub-program of the National River Health Program, covering physical, chemical and ecological aspects. Reports arising from the sub-program may be found at the LWRRDC website (http://www.lwrrdc.gov.au).

3.2 Biological assessment

3.2.1 Introduction and outline

a See Sections 3.2.1.3 to 3.2.2.2
b Sections 3.2.3 to 3.2.4

In broad terms, this section provides advice about the selection of biological indicators to apply to various water quality problems,[a] and the analytical procedures that should be used to monitor and assess change in these indicators.[b] The material in this section is accompanied by little in the way of rationale or justification; those are provided in other chapters of the guidelines. Generic issues of designing a program for monitoring or assessment are given in Sections 7.1 and 7.2, with much background material provided in the *Australian Guidelines for Water Quality Monitoring and Reporting* (the Monitoring Guidelines, ANZECC & ARMCANZ 2000) (especially Chapters 3, 4 & 6). For substantiation of the recommended approaches and additional guidance, an expanded discussion about the selection of biological indicators is provided in Section 8.1 (Vol. 2), while a detailed account of specific issues for biological monitoring and assessment is provided in Section 7.3. It is important that the material presented in the current Section (3.2) is not read in isolation of these other detailed accounts.

3.2.1.1 Philosophy and approach behind bioindicators of water quality

The following sections discuss the concepts and monitoring frameworks necessary to assess aquatic biological communities. A key concept is that of ecological integrity (health), defined in Section 3.1.1.

Biological assessment (bioassessment) can measure the desired management goals for an ecosystem (e.g. maintenance of a certain diversity of fish species or certain level of nuisance algae) as might be described in the management goals. Bioassessment provides information on biological or ecological outcomes; these may result from changes in water quality but may also result from changes in the physical habitat (e.g. increased fine sediment deposition, or changes in hydrology) or from changes in biological interactions (e.g. the introduction of exotic species or diseases).

Thus, bioassessment should be seen as a vital part of assessing changes in aquatic ecosystems, and as a tool in assessing achievement of environmental values and attainment of the associated water quality objectives. At the same time, the resulting biological *message* provides an insight into a complex system which:

- integrates multiple natural and human changes in physico-chemical conditions;
- integrates disturbances over time;
- absorbs human effects into complex interacting biological communities and processes;
- can give a signal from more than one component (e.g. multiple species or community similarities or ecological processes).

c Section 3.1.4

The guidelines for biological assessment are intended to detect important departures from a relatively natural, unpolluted or undisturbed state — the reference condition.[c] An important departure is deemed to be one in which the ecosystem shows substantial effects, including:

- changes to species richness, community composition and/or structure;
- changes in abundance and distribution of species of high conservation value or species important to the integrity of ecosystems;

- physical, chemical or biological changes to ecosystem processes.

Important in this context does not mean mere statistical significance, which is only a tool in the context of a specific monitoring design. Rather it means a change or departure deemed practically significant, in relation to previously agreed performance criteria, for failing to achieve a water quality objective.

The results of bioassessment may require interpretation using additional supporting information on water quality and physical conditions at site, catchment or regional scales. Bioassessment provides a window onto the condition of the ecosystem being managed.

Bioassessment and biological indicators have come into use because the traditional physical and chemical guidelines are too simple to be meaningful for biological communities or processes. Strong variation in ecosystem processes and biological community composition in time and space is characteristic of many surface water environments, particularly in Australia.

Biological systems are very variable. It is important to understand that because of this variability, sampling designs have a limited capacity to detect and quantify change relative to an undisturbed or reference state. Any given sample size or number of sample units taken during a monitoring or assessment program has quantifiable constraints on its capacity to detect a change of a given magnitude. There is a strong relationship between the power (in statistical terms) of a monitoring program design, the magnitude of the effect that is detectable and the sample sizes involved.

There is also a trade-off between a capacity to detect change, and the sample size, and the chance of not detecting that change (or of detecting a change that has not occurred). This trade-off is often negotiated on the basis of financial resources for monitoring programs, since to increase sample sizes or numbers of sample units is the most common way of increasing the power to detect a change.[a]

a See Sections 3.1.7 & 7.2.3.3

It is vital to recognise the need for high quality, comprehensive designs in bioassessment and biological monitoring. Protocols are being developed for bioassessment, with improved designs and rigour in site selection, sampling approaches and analysis. Several examples of this are given in the following sections on biological assessment.

3.2.1.2 A framework for biological assessment of water quality

Successful employment of a biological monitoring and assessment program for the protection of aquatic ecosystems involves a series of steps:

1. define the primary management aims, including the level of protection desired by the community and other stakeholders; define the management goals for achieving protection of the ecosystem, and the environmental concerns;[b]

b Section 3.1.1.1

2. together with a balance of indicators, identify the biological assessment objectives for protection of the water resource;[c]

c Sections 7.2.1 & 3.2.1.3

3. select appropriate indicators and protocols to apply to the assessment objectives;[d]

d Sections 3.2.2 and 3.2.3

4. select the appropriate experimental design to apply to the indicator;[e]

e Section 3.2.3 and Ch.7

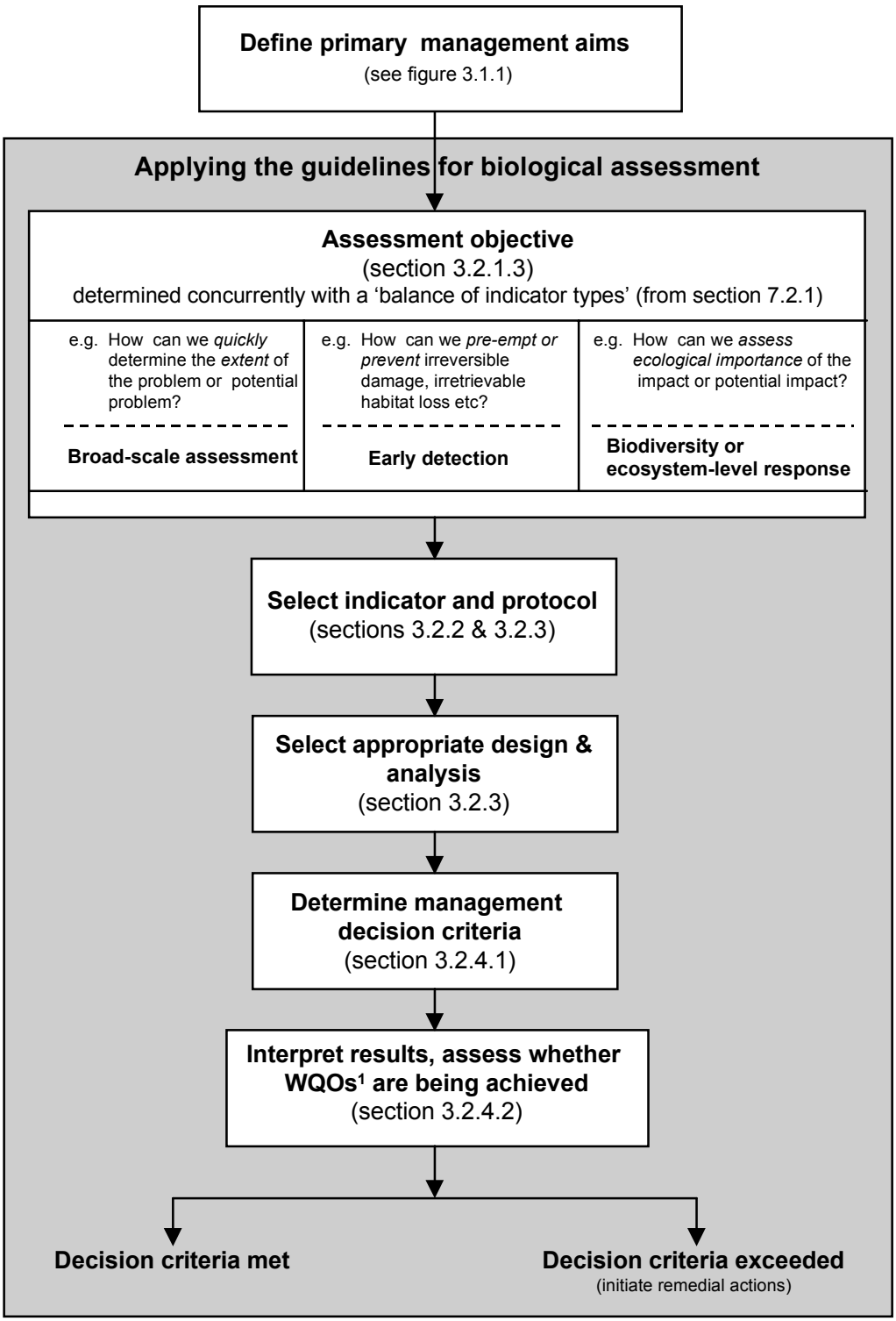
a See Sections 3.1.7, 7.2.3.3 and 3.2.4

5. determine key decision criteria, i.e. acceptable level of change and statistical sensitivity with which to detect such change;[a]

b Section 3.2.4.2

6. assess results from monitoring programs,[b] with feedback to management.

This framework of steps is also shown in figure 3.2.1.



[1] = Water Quality Objectives (section 2.1.5)

Figure 3.2.1 Decision tree for biological assessment of water quality

3.2.1.3 Biological assessment objectives for ecosystem protection

Having determined the level of protection required for an ecosystem, the management goals for achieving that protection, and the environmental concerns (fig 3.1.1), managers should identify *assessment objectives* for protection of the water resource. The objectives will help managers select the most appropriate biological indicators and protocols. Three broad assessment objectives are described as follows:

1. Broad-scale assessment of ecosystem health (at catchment, regional or larger scales)

Resources will never be adequate to provide detailed, quantitative[8] biological monitoring and assessment of water quality over wide geographical areas of Australia and New Zealand. Therefore, tools for rapid biological assessment (RBA) are being developed that, while not providing detailed quantitative information, are cost-effective and quick enough to generate adequate first-pass data over large areas. The data may be adequate for management purposes or they may help managers to decide what type of further information may be required and from where.

Broad-scale assessment can be useful for the following applications:

- rapid, cost-effective and adequate first-pass determination of the extent of a problem or potential problem, e.g. as applied to broad-scale land-use issues, diffuse-source effluent discharges or information for State of Environment Reporting;
- screening of sites to identify locations needing more detailed investigation;
- remediation programs being conducted over broad geographical areas (catchment, regional or larger scales).

The most developed RBA method is AUSRIVAS, a method using macroinvertebrate communities in rivers and stream. Rapid bioassesment protocols are also being developed for riverine benthic algae (diatoms) and fish, as well as for macroinvertebrate communities in wetlands and estuarine sediments.

2. Early detection of short- or longer-term changes

Prediction and early detection of possible effects are useful to any water quality management program so that substantial and ecologically important disturbances can be avoided. Early information enhances the options for management. For example, where an effect is observed from a controlled discharge, it may be possible to adjust the rate of release or of subsequent releases.

Predictive information and early detection in the field can result if specific and sensitive programs are set up, incorporating study of sublethal responses of organisms. If sampling sites for any indicator can be located in mixing zones effectively creating spatial disturbance gradients, they will enhance early detection and predictive capabilities.[9]

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[8] The adjective 'quantitative' from here on, in Section 3.2, refers to an indicator measurement program that permits rigorous and fair tests of the potential disturbances under consideration; typically, conventional statistical tools would be employed to attach formal probability statements to the observations — see Section 3.2.3.

[9] The purpose of sampling in mixing zones in this case is solely for enhancing inference about disturbances in receiving waters, not for determining compliance in this zone.

Also, RBA programs operating over broad geographical regions may, through their extensive coverage, pin-point potential 'hot-spots' that would otherwise be missed. However, these programs do not incorporate very sensitive protocols.

Early detection can be important for:

- sites of special interest (e.g. sites of high conservation value, major developments and/or point-sources of particular potential concern) where the cost of failing to detect a disturbance in a timely manner may be too high;
- timely identification of water quality issues and problems that may exist over a broad geographical region in response to a specific pressure;
- any situation where a management objective has been strongly linked to the Precautionary Principle tenet of the *National Strategy for Ecologically Sustainable Development* (ESD Steering Committee 1992).

3. Assessment of biodiversity

Often it is not sufficient simply to have detected change in an early detection indicator because the information cannot easily be linked (if at all) to adverse effects at population, community and ecosystem levels. To determine effects upon the ecosystem as a whole and as important end-points in themselves, measures of *biodiversity*, including ecosystem processes and the conservation status of sites, should be key responses sought-after in monitoring programs.

Biodiversity and conservation status are best measured using species-level data gathered from quantitative studies. Information gathered at higher levels of taxonomic resolution will serve these needs if the data are correlated with biodiversity or conservation status at species level (e.g. Wright et al. 1998). Even in the best-resourced studies, it is inevitable that biodiversity assessment will usually be limited to the measurement of ecosystem surrogates — communities/assemblages of organisms, or habitat or keystone-species indicators where these have been closely linked to ecosystem-level effects. Information on the ecological importance of effects will best be met in programs that have regional coverage and encompass a full disturbance gradient.

Whether the assessment objective is biodiversity, conservation status or ecosystem-level responses for assessing ecological importance of disturbance (as measured by community structure or ecosystem process attributes), this indicator is hereafter termed biodiversity indicator.

The biodiversity assessment objective may be important for the following applications:

- for sites of special interest where indicators are needed to measure biodiversity, conservation status, and/or ecosystem-level effects for assessing ecological importance of disturbance. Information gathered for such indicators is highly complementary to that gathered for early detection indicators.
- through RBA programs, as a first-pass measure of biodiversity, conservation status and/or ecosystem-level effects for assessing ecological importance of disturbance, at sites and over a broader geographical region.
- in any situation where a management objective has been strongly linked to the Ecologically Sustainable Development tenet of the 'Maintenance of biodiversity and ecological systems' (*National Strategy for Ecologically Sustainable Development*, ESD Steering Committee 1992).

3.2.2 Matching indicators to problems

3.2.2.1 Broad classes of indicators and desired attributes

Desired or essential attributes of the broad indicator types (or methods) required to meet the assessment objectives are listed in table 3.2.1. Each of the three assessment objectives is discussed fully in Section 8.1.1 (Volume 2), but the main points are summarised below.

1. Broad-scale assessment of ecosystem 'health'

The indicator types relevant to a broad-scale assessment objective have these attributes:

i. the measured response adequately reflects the ecological condition or integrity of a site, catchment or region (i.e. ecosystem surrogate);

ii. where community or assemblage data are gathered, these and associated environmental data can be analysed using multivariate procedures;

iii. approaches to sampling and data analysis are highly standardised;

iv. responses are measured rapidly, cheaply and with rapid turnaround of results;

v. results are readily understood by non-specialists;

vi. responses have some diagnostic value.

A range of studies of populations and communities could provide information about the ecological condition or integrity of a site, catchment or region, but only rapid biological assessment (RBA) methods would enable such information to be gathered over wide geographical areas in a standardised fashion and at relatively low cost. Resh and Jackson (1993), Lenat and Barbour (1994) and Resh et al. (1995) elaborate upon features of RBA approaches as applied to stream macroinvertebrate communities. Comment upon some RBA methods currently being applied to freshwater fish communities is provided in Section 8.1.2.1 of Volume 2.

2. Early detection of short- or longer-term changes

To have a predictive or early detection capability, an indicator should ideally have a response that is:

i. sensitive to the type of stressor;

ii. correlated with environmental effects (i.e. linked to higher-levels of biological organisation);

iii. time- and cost-effective to measure;

iv. highly constant over time and space, which confers high power to detect small changes;

v. regionally and socially relevant;

vi. broadly applicable.

These attributes are important because assessments of actual or potential disturbances will only be as effective as the indicators chosen to assess them (Cairns et al. 1993). However, the attributes are *idealised* characteristics only, and in many cases some will conflict or will not be achievable. Therefore the more important and achievable attributes must be decided upon, and appropriate indicators must be chosen accordingly.

Table 3.2.1 Biological assessment objectives for different management situations and the recommended methods and indicators

Assessment objective	Applications	Recommended indicators	Essential or desired attributes of the indicator to be employed
1. *Broad-scale assessment* of *ecosystem 'health' (catchment, regional or larger scale)*	Water quality on a catchment or regional basis (e.g. SoE reporting, catchment management indicators)	Rapid bioassessment (e.g. AUSRIVAS)	• Comparative measures of biological community composition, e.g. multivariate • Measure rapidly and cheaply, rapid turnaround of results • Have a diagnostic value
2. *Early detection of short- or longer-term changes*	Sites of special interest (high conservation value, major developments or point-sources of particular potential concern)	*Laboratory*: Direct toxicity assessment *Field*: Instream/riverside assays, biomarkers, bioaccumulation; spatial disturbance gradients in relevant quantitative biological indicators	• Sensitivity to the type of contaminant expected (and hence diagnostic value) • Respond and measure rapidly (e.g. sublethal) • Demonstrate a high degree of constancy in time and space (i.e. high signal:noise ratio) (field)
	Water quality on a regional basis in response to specific pressure	Rapid bioassessment	• As for 'Broad scale assessment' above
3. *Biodiversity or ecosystem-level response*	Sites of special interest	• Detailed quantitative, preferably regionally-comparative, investigations of communities possibly with species-level taxonomic resolution • Direct and preferably comparative measurement of the ecosystem process of concern	• Direct measures of diversity (using species-level identification for quantitative studies), with regional comparison • Direct measures of ecosystem function (e.g.community metabolism) • Use of surrogate measures for ecosystem biodiversity where relationship between surrogate and biodiversity has been shown (usually community/multivariate) • Have a diagnostic value
	Water quality at sites and on a regional basis	• Direct and preferably comparative measurement of the ecosystem process of concern • Rapid bioassessment (for biodiversity/ conservation status where this has been shown to correlate well with biodiversity)	• As for 'Assessment of biodiversity' above

As mentioned earlier, methods of prediction and early detection fall into two categories: 1) sub-lethal organism responses (e.g. growth, reproduction), and 2) rapid biological assessment (RBA, e.g. AUSRIVAS). The potential of these methods to meet the objective of *early* detection is discussed below.

Sub-lethal organism responses

Sub-lethal organism responses can generally be found to meet, in the same measured response, important attributes (i), (iii), (iv) and (v) above. However, there will inevitably be conflict and difficulty in meeting all six attributes. For example, an indicator with good diagnostic value for a particular stressor may not be particularly applicable to a broad range of stressors. Socially-relevant sub-lethal organism responses are also often difficult to find. A more significant limitation, however, is that in very few situations have indicators of exposure to a pollutant been correlated to environmental effects.

Rapid biological assessment (RBA)

Rapid biological assessment (or RBA) methods are applied and measured in a way that makes them poorly suited to a role of early detection. In particular, they are not designed to detect subtle disturbances so may not have desirable attributes (i) and (iv) above. Nevertheless, unlike other early detection methods, RBA procedures can be carried out at relatively low cost at a large number of sites or over large geographical areas, and will generally have greater ecological, regional and social relevance, i.e. features (ii), (iii), (v) and (vi) above. Indeed, RBA methods such as AUSRIVAS, in which site data are compared with regionally-relevant reference conditions, via a predictive model, and reported using a standard index, are particularly relevant. In their broad coverage they may also be able to locate problems and stressors that would otherwise pass unnoticed.

Sub-lethal organism responses and RBA methods combine different predictive and early detection needs, and in comprehensive monitoring programs may play highly complementary roles. Nevertheless, in a balanced program that measures both early detection and biodiversity indicators, attributes (i), (iii) and (iv) above are regarded as the most important guides to the selection of types of indicator.

3. Biodiversity assessment

The *biodiversity* assessment objective is similar to the broad-scale assessment objective (1) above because both provide information about the ecological condition or integrity of a site. Two important features distinguish the two objectives in practical monitoring programs: the provision of relatively detailed quantitative and accurate assessments of biodiversity indicators — but at limited spatial scales, for reasons of high cost; and the provision of less accurate first-pass assessments of broad-scale indicators — but at greater spatial scales.

Biological indicators used for broad-scale assessment can also be used for biodiversity assessment. Tradeoffs in costs, the level of accuracy and detail of information required will ultimately determine which approach is used.

Desired or essential attributes of biodiversity indicator types include features (i) and (vi) from broad-scale assessment above, as well as either (i) direct measures of diversity (using species-level identification) and/or (ii) surrogate measures for biodiversity where a relationship between surrogate and biodiversity has been shown; and (iii) direct measures of ecosystem function (e.g. community metabolism).

Box 3.2.1 A cautionary note on the use of the AUSRIVAS RBA approach for site-specific assessments

AUSRIVAS, the RBA method using stream macroinvertebrate communities, is at an intermediate stage of development. It may be limited in its ability to detect minor water quality disturbances on biota. This restriction is caused by:

- the low level of taxonomic resolution (family level) used in existing state/territory-level (large-scale) models;
- the use of presence–absence data only;
- the need to factor temporal variability into AUSRIVAS assessments using reference sites as controls.

In general, stronger inference and greater sensitivity to disturbance become more important requirements as the spatial scale of a study narrows. Therefore, for specific assessments conducted at small scales (within a catchment), AUSRIVAS should be conducted using a sampling design that offers sufficient scope (viz site selection, spatial and temporal replication) to meet the study requirements. For more reliable assessments at small scales it may be necessary to combine the data gathered for two seasons (e.g. autumn and spring) and to enter the data into the 'combined-seasons' models developed by many state agencies. However, some of the RBA's 'rapid assessment' aspect would be lost.

These issues are expanded upon in Chapters 7 and 8.

This bioassessment approach is in a phase of ongoing development and refinement. One characteristic of that phase is the need to increase the spatial spread and density of reference sites in various regions in Australia. At present, site numbers and densities may not be sufficient to allow reliable bioassessment in some regions. (It should be noted that existing support software for AUSRIVAS models screens out any data collected from sites outside the geographic region for which the model was derived.)

While the sensitivity of AUSRIVAS for site-specific assessments is being improved, Guidelines' users should seek updates on developments in this area to determine whether the method meets the bioassessment requirements for their particular situation and region. Such updates, including details of the geographic spread of reference sites, may be obtained from the AUSRIVAS homepage, http://ausrivas.canberra.edu.au/ausrivas.

One would expect quantitative biodiversity indicators to be restricted in application to a relatively small region, e.g. a river of interest and sites from rivers in catchments immediately adjacent. This would be less a limitation for broad-scale RBA indicators. In monitoring programs, RBA indicators would not normally be expected to provide direct measures of diversity. Further guidance on whether RBA or quantitative 'biodiversity' indicators (or both) are appropriate for a particular situation is provided in Section 8.1.1.3 of Volume 2.

3.2.2.2 Matching specific indicators to the problem

These Guidelines discuss several stressors, such as metals, suspended solids and/or sedimentation, salinity, herbicides and nutrients, any environmental effects of which can be identified, quantified and assessed by particular biological indicators. Viable protocols (i.e. proven or near-proven) using diatoms and algae, macrophytes, macroinvertebrates and fish populations and/or communities, together with community metabolism, have been developed for use in streams and rivers, wetlands and lakes, and estuarine and marine ecosystems to monitor and assess changes associated with these stressors. The stressors (or water quality issues) and biological indicators recommended to apply to the monitoring and assessment of

water quality are listed in table 3.2.2. Background to the development of the biological indicators, including rationale and justification, is provided in Section 8.1 of the Guidelines.

Development of protocols for the early detection of sediment toxicity using field assessment procedures is at an early stage in Australia and elsewhere. Until suitable indicators are identified and protocols for these are developed, a laboratory assessment approach is recommended (method 2A, table 3.2.2).[a] For this, a potentially contaminated sediment from the field is brought back to the laboratory and standard sediment toxicity tests are conducted to determine its toxicity. A suitable uncontaminated sediment, collected from an adjacent control site or from the same site prior to disturbance, is tested as a reference.

a e.g. Method 2A, Appendix 3, Vol 2

3.2.3 Recommended experimental design and analysis procedures for generic protocols

It is essential that protocols permit rigorous and fair tests of the potential disturbances under consideration. The best protocols are those that have sufficient baseline data collected before as well as after a potential disturbance.[b] There are two advantages of such protocols. Firstly, the logical basis for inferring whether or not a disturbance has occurred is stronger because the natural variation inherent in the indicator(s) is incorporated into the inference; secondly, a properly-designed testing program permits use of conventional statistical tools to attach formal probability statements to the observations.[c] Where such data do not exist or cannot be collected, alternative analytical procedures can be adopted. These two broad groups of procedures are outlined here and described in more detail in Section 7.2 (Table 7.2.1D).

b See Sections 7.2.2 and 7.2.3

c Sections 7.2.2 and 7.2.3

Protocols which rely on conventional statistical procedures (Appendix 3, Volume 2) have two essential features. First, they require that baseline data be collected prior to the supposed disturbance because seasonal and inter-annual variability in the indicators need to be accounted for. Second, pre- and post-disturbance data need to be collected from both the disturbed area and from comparable undisturbed areas. These control areas provide a benchmark against which changes in the indicator in the disturbed areas can be judged. With few exceptions, the more control areas that can be incorporated into the design of the experiment or assessment, the stronger and fairer will be the test of the effect of the disturbance. The conventional statistical procedures that are used to analyse these designs belong to the family of general linear models, which includes univariate and multivariate analysis of variance, analysis of covariance and regression.

Not all situations permit the implementation of inferentially strong designs. Appropriate control areas may be limited in number or not available at all. In this case, statistical methods can be applied to data collected within appropriate designs, but the strength of the inferences that can be drawn is much weaker and there is a correspondingly higher risk of either missing a disturbance or erroneously concluding that a disturbance has occurred. Accordingly these designs should not be implemented merely as a cost-saving measure; they should only be chosen if appropriate control areas cannot be found.

Table 3.2.2 Water quality issues and recommended biological indicators for different ecosystem types: S = streams and rivers, W = wetlands, L = lakes and M = estuarine/marine. Letters or indicator in italics denote that while the indicator is not presently available, it could be developed relatively quickly with additional resourcing.

Code	Issue	Suitable biological indicator or assessment approach	Protocol[1]	Ecosystem type
1A, B	General inorganic (including metals) and organic contaminants: Early detection of short- or longer-term changes from substances in solution/water column	1A Instream/riverside assays measuring sublethal 'whole-body' responses of invertebrate and/or fish species;	1A(i), (ii)	S
		1B Biomarkers (chemical/biochemical changes in an organism)	1B(i), (ii)	S, W, L, M
		Direct toxicity assessment	sec 8.3.6 (Vol 2)	S, W, L, M
2A, B	General inorganic (including metals) and organic contaminants: Early detection of short- or longer-term changes from substances deposited (sediments)	2A 'Whole-sediment' laboratory toxicity assessment (where sediment tests are available)	2A, sec 8.3.6	S, W, L, M
		2B Bioaccumulation/biomarkers (for organisms that feed through ingestion of sediment); other sublethal incl. behavioural responses where protocols developed	2B(i), (ii)	S, W, L, M
3	General inorganic (including metals) and organic contaminants: Changes to biodiversity and/or ecosystem processes	Structure of macroinvertebrate and/or fish populations[2, 3]/communities[3] using rapid, broad-scale (RBA[4]) or quantitative (Q) methods	3A(i)–(v)	S, W
		Stream community metabolism	3B	S
4	Suspended solids in the water column	Structure of macroinvertebrate and/or fish populations[2]/communities using RBA[4] or Q methods	3A(i)–(v)	S
		Seagrass depth distribution	6	M
5	Sedimentation of river bed	As for 4 as well as stream community metabolism	3A(i)–(v), 3B	S
6	Effects of organotins	Imposex in marine gastropods	9	M
7	Salinity: Changes to biodiversity	Structure of macroinvertebrate and/or fish populations[2, 3]/ communities[3] (RBA[4] or Q methods); remote sensing (changes to vegetation structure);	3A(i)–(v), 5	W, S?
8	Herbicide inputs: Changes to biodiversity	Structure of phytoplankton or benthic algal communities; remote sensing (changes to vegetation structure).	4(i), (ii), 5	W, S
9	Nutrient inputs: Early detection of short- or longer-term changes from substances deposited or in solution/water column	Structure and/or biomass of benthic algal or phytoplankton communities	4(i)–(iii)	S, W
		Stream community metabolism	3B	S
10	Nutrient inputs: Changes to biodiversity and/or ecosystem processes	Structure and or biomass of phytoplankton, benthic algal and/or macroinvertebrate populations[2]/communities (Q or RBA[4])	3A(i)–(v), 4(i), (ii)	S, W
		Stream community metabolism	3B	S
11	Nutrient inputs	11a Seagrass depth distribution	6	M
		11b Frequency of algal blooms	7	M
		11c Density of capitellids	8	M
		11d In-water light climate		
		11e Filter feeder densities		
		11f Sediment nutrient status		
		11g Coral reef trophic status		
12	General effluents (non-specific) and effects of hypoxia	Structure of macroinvertebrate communities (Q or RBA[4])	3A(i), (ii)	S, W
13	Broad-scale assessment of ecosystem 'health' (non-specific degradation)	13A Composition of macroinvertebrate communities using RBA methods	3A(i), (ii)	S, W
		13B Habitat distributions		M
		13C Assemblage distributions		M

1. The codes listed in this column refer to protocols that are listed by title in Section 8.1.3 of Volume 2. Summary descriptions of these protocols, with references to important source documents, are provided in Appendix 3, Volume 2. 2. Populations could serve as biodiversity surrogates if a 'keystone' role could be established for a species. 3. For pesticides, study of non-target organisms. 4. Cautionary notes on use of RBA methods for site-specific assessments are provided in various sections of these Guidelines.

With some indicators, such as certain highly specific chemical and biochemical markers, it is possible to use designs that need only limited controls in time or space or no controls at all. However, there must be conclusive evidence that such indicators are unequivocally related to the disturbance before such designs are adopted.

For some situations, a disturbance may have occurred and there are no pre-disturbance data. Alternatively, a development may proceed with insufficient, if any, baseline data. In these circumstances, the rigour of any inferences about the disturbance is severely curtailed; the sometimes novel analytical procedures that have been applied to such data do not compensate for the lack of pre-disturbance data.[a] Where multiple control areas are available, they can be used to describe how atypical the potentially disturbed areas appear.[b] These procedures require the user to assume that the indicator responded similarly in control and disturbance areas before the disturbance. Where multiple control areas are not available, questions are often framed around the extent of the disturbance. As discussed below,[c] under these circumstances it is best that data be collected from a comparatively larger number of disturbance sites than would otherwise be gathered (e.g. along a mixing zone gradient), so that stronger inferences may be drawn about disturbance by way of disturbance gradients. Such additional data may also enhance predictive capabilities of monitoring programs.

a & b See Sections 7.2.2 & 7.2.3

c Section 3.2.4.2/4 & 7.2.2

For all these procedures it is necessary to collect and collate exploratory data. The aim is to define the spatial and temporal extent of sampling and to identify and choose sampling locations within the control and disturbance areas.[d] Such exercises can include use of simulation or other predictive tools to model currents or sediment movements, and/or be new or pre-existing data on the flora or fauna. It is difficult to prescribe protocols for exploratory collections because the amount of pre-existing data or auxiliary models will vary from case to case. In novel or unfamiliar situations such exploratory collections are even more desirable and could lead to substantial savings in time and costs.

d Section 7.2.3.2

Table 3.2.3 summarises the designs that apply to the protocols listed in table 3.2.2. The BACI class of design uses conventional statistical procedures while designs using alternative analytical procedures must be applied if inference is based on temporal change only or spatial pattern alone.

Preferred designs using conventional statistical procedures involve both pre-disturbance baseline data and multiple control areas (MBACI and 'Beyond-BACI' designs of table 3.2.3). Where pre-disturbance baseline data are available or can be collected, but only a single control site can be found, BACIP designs are appropriate. Designs where the length of pre-disturbance baseline and/or the number of control areas are reduced (e.g. BACI) have less inferential rigour because more assumptions need to be made about the similarity of the behaviour of the indicator in control and disturbance areas prior to the onset of the potential disturbance.

It is important to consider using any descriptive and exploratory analytical tools that would enhance interpretation of the analytical procedures employed. These might include graphs and plots accompanying univariate and multivariate approaches, clear tabulations of relevant descriptive statistics in univariate analyses (e.g. means and confidence intervals), and ordination and classification of data in multivariate studies.[e] Some of the specific requirements of biological indicators that need to be considered while designing the monitoring program are detailed in Section 7.3.

e Sections 7.2, 7.3 and the Monitoring Guidelines Ch.6

Table 3.2.3 Experimental design and analysis procedures to apply to generic protocols. The letters used to identify the broad categories of design are those used in figure 7.2.1. Explanations of the possible designs and references are supplied in Section 7.2.3. Letters and numerals in the protocol column correspond to those used in Table 3.2.2 and Section 8.1.3 (Volume 2).

Broad category of design (from Section 7.2.2)	**Possible designs (Described in table 7.2.1)**	**Protocol (from Section 8.1.3, Vol 2)**
A. Inference based on the BACI (Before, After, Control, Impact) family of designs	MBACI	All protocols wherever possible
	Modifications (e.g. MBACIP, inclusion of covariates)	Any protocol if applicable
	'Beyond BACI' designs	Any protocol if applicable.
	BACIP (single control site)	1A, 1B
	Modifications to BACIP	1A, 1B
	Simple BACI	1B
B. Inference based on temporal change alone	Intervention analysis	1B, 2B, 3B, 4, 6, 7, 8. Possibly 3A(ii) but may prove very expensive; behaviour of 3A(i) in face of temporal variations unknown and not recommended for this protocol
	Trend analysis	1B, 2B, 3B, 4, 6, 7, 8. Possibly 3A(ii) but may prove very expensive; behaviour of 3A(i) in face of temporal variations unknown and not recommended for this protocol
	A posteriori sampling	Possibly 1B, 2B, but only if chemical or toxicant is unequivocally related to the effluent
D. Inference based on spatial pattern alone	Conventional statistical designs (e.g. ANOVA, ANCOVA)	Any protocol based on univariate indicator e.g. 1B, 2B, 3B, 4(i)A, 4(ii), 4(iii)A, 6, 8, 9.
	Analysis of 'disturbance gradients'	Any protocol if applicable; may be too cumbersome for 1A
	Predictive models based on spatial controls only	3A(i), 3A(ii)

3.2.4 Guidelines for determining an unacceptable level of change

3.2.4.1 Inferences, assessment of change, setting decision criteria

A priori decisions made between stakeholders (e.g. developer and regulator) about effect size and the probability of making a Type I error (α) and Type II error (β) (generally only 'effect size' needs to be decided upon for RBA) are an essential aspect of the guidelines philosophy.[a] These decision criteria should be pre-established in the following four scenarios: for flexible decision-making; for compliance assessment; when there are multiple lines of evidence; and when data are to be assessed against predictive models.

a See sections 2.2.1.2, 3.1.7, 7.2.3.3

1. Flexible decisions in the spirit of cooperative best practice

Flexible decisions are important where adherence to a precautionary approach has been agreed or stipulated by a regulatory authority or dictated by legislation. Adequate baseline data should be collected according to the design criteria discussed above, given any unavoidable constraints. Integral to design considerations is the principle that monitoring should provide a strong basis for management *response* (through decisions and/or action) to any early indications of adverse disturbances. The decisions about the criteria and about responsive action by management should

be made *a priori*, especially where a superficially positive response might result from the early stages of an abnormal, and therefore undesired, change in environmental conditions; e.g. increased taxonomic richness accompanying a slight increase in eutrophication. Management intervention will depend on the management objective(s) for the receiving waters, but two approaches are possible.

i. Management could make 'super-precautionary' responses, dictated by any statistically significant trend from baseline of a magnitude agreed *a priori* to be important. The probability criteria for statistical significance would be determined under the flexible decision regime proposed by Mapstone (1995, 1996), with the result that α and β would be variable and determined from time to time on the basis of the available data and the critical effect size agreed *a priori*. The emphasis is on setting values for critical effect sizes that would be expected to trigger an early management response to a potential disturbance. It is assumed that it is more important to react quickly to potential problems, even though the response would be to something which had not yet become a major ecological threat. Such a position would be appropriate for activities in particularly sensitive or valuable areas. The precision with which one could specify the location of the baseline reference point would depend on the amount of sampling during the baseline period. Increasing the precision with which the reference point is specified, which would presumably also mean increasing the precision of sampling after the start of a development, would reduce the risk of responding to an erroneous trigger caused by early indications of a shift from baseline conditions. Thus, it becomes to everyone's advantage to seek thorough monitoring.

ii. Management response could be triggered by ongoing feedback or a continuously monitored variable exceeding some threshold value. Control charting techniques such as those used in quality assurance/quality control programs might be employed here. The trigger value for a particular variable might represent a level at which that variable is known to have important biological consequences, or might simply be a statistical parameter used to indicate that an observed event would be considered an outlier under normal circumstances and therefore is worthy of further investigation. As in (i) above, it is important that all parties have agreed *a priori* to intervene when that trigger occurs.

2. Compliance, legal framework: data gathered under strict and rigorous hypothesis-testing framework

In this case, the criteria to which sampling programs are designed are set independently of the particular activity being monitored. Such criteria would not normally be subject to negotiations between regulators and proponents or other interested parties. These external criteria are the reference points that, if exceeded, will trigger action. In these cases, negotiations between regulators, interest groups, and proponents focus on the degree of risk involved in either failing to confidently recognise that the standard has been violated (β) or that apparent violations will be flagged in error (α). As in (i) from Section 3.2.4.1/1 above, the thoroughness of sampling design will directly influence the likelihood of erroneous decisions.

3. Data gathered from multiple lines of evidence, where statistical power for each indicator may be poor (lack of adequate temporal baseline)

For situations where there is a paucity of baseline information and/or adequate spatial controls, it is recommended that users adopt a 'weight-of-evidence' approach (Suter 1996) to inference. The process is based on risk assessment principles and draws on epidemiological precepts in interpreting test results; the concept in various forms has been described by Hodson (1990), Stewart-Oaten (1993) and Suter (1996), amongst others, with examples. There is an onus on those conducting monitoring programs under these situations to enhance the set of monitoring techniques used: it should include chemical monitoring, spatial gradients for a number of biological monitoring protocols,[a] and toxicological and other experimental data in which concordance is sought between field results and controlled experimental findings. In this way, lack of baseline information may be at least partially compensated for, so that conclusions can be confidently drawn and, importantly, agreed upon by all parties.

a See Section 7.2.1.2

4. Data assessed against bands of AUSRIVAS predictive models

Two complementary indices summarise the outputs from the analysis of AUSRIVAS data:

i. *O/E* Family — the ratio of the number of families of macroinvertebrates at a site to the number of families expected (predicted) at that site. (The expected number of families is actually the sum of the probabilities of each taxon occurring at the site as calculated from the model.)

ii. *O/E* SIGNAL which is the ratio of the observed SIGNAL[10] value for a site to the expected SIGNAL value. SIGNAL assigns a grade to each family based on its sensitivity to pollution. The sum of the grades is divided by the number of families involved to give an average grade for the site. A grade of 10 represents high sensitivity to pollution, while a grade of 1 represents high tolerance of pollution.

The values of both indices can range from a minimum of 0 (indicating that none of the families expected at a site were actually found at that site) to a theoretical maximum of 1.0, indicating a perfect match between the families expected and those that were found. In practice, the maximum can exceed 1.0 indicating that more families were found at that site than were predicted by the model. This can indicate an unusually diverse site, but could also indicate mild enrichment by organic pollution where the added nutrients have allowed families not normally found in that site to establish. Conversely, an undisturbed, high-quality site may score an index value less than 1.0 because of chance exclusions of families during sampling.

For reporting, the value of each index is divided into categories or bands. The width of the bands is based on the distribution of index values for the reference sites in a particular model. The width of the reference band, labelled 'A' in table 3.2.4, is centred on the value 1.0 and includes the central 80% of the reference sites. Any site with index within the 10% and 90% bounds around 1.0 is allocated to band A and is described as being of 'reference condition'. A site with an index value exceeding the upper bound of these values (i.e. the index value is greater than the 90th percentile of

[10] SIGNAL is a biotic index, Stream Invertebrate Grade Number — Average Level; see Section 8.1.2.1 and Chessman (1995).

the reference sites) is judged to be richer than the reference condition, and is allocated to 'band X'. A site whose index value falls below the lower bound (i.e. the index value is smaller than the 10th percentile of the reference sites) is judged to have fewer families and/or a lower SIGNAL score than expected and is allocated to one of the lower bands according to its value. The widths of bands B and C are the same as the width of band A, the reference band. The band D may be narrower than these, depending on variability in the index values of the reference sites in the model. In most cases, sites falling in band D on either index are severely depleted in terms of the number of families expected.

In many cases the values of the indices will allocate a site to the same band. In situations where the two indices differ in band allocation, the site will be allocated to lower of the two bands if the index value is below reference condition, or to the above reference band if one of the indices places the site in band X.

These factors should be taken into consideration by stakeholders and management who are setting situation-specific guidelines.

Table 3.2.4 Division of AUSRIVAS O/E indices into bands or categories for reporting. The names of the bands refer to the relationship of the index value to the reference condition (band A). For each index, the verbal interpretation of the band is stated first, followed by likely causes (dot-points).

Band label	Band name	Comments	
		***O/E* Families**	***O/E* SIGNAL**
X	Richer than reference	More families found than expected. • Potential biodiversity 'hot-spot' • Mild organic enrichment	Greater SIGNAL value than expected. • Potential biodiversity 'hot-spot' • Differential loss of pollution-tolerant taxa (potential disturbance unrelated to water quality)
A	Reference	Index value within range of central 80% of reference sites	Index value within range of central 80% of reference sites
B	Below reference	Fewer families than expected • Potential disturbance either to water quality or habitat quality or both resulting in a loss of families	Lower SIGNAL value than expected • Differential loss of pollution-sensitive families • Potential disturbance to water quality
C	Well below reference	Many fewer families than expected • Loss of families due to substantial disturbance to water and/or habitat quality	Much lower SIGNAL value than expected • Most expected families that are sensitive to pollution have been lost • Substantial disturbance to water quality
D	Impoverished	Few of the expected families remain • Severe disturbance	Very low SIGNAL value • Only hardy, pollution-tolerant families remain

It should be noted that the calculation of indices and allocation to a band for a stream site are automatically performed as part of the AUSRIVAS procedure by the AUSRIVAS software package. This software, downloaded over the internet (website address: http://ausrivas.canberra.edu.au/ausrivas) performs all calculations required for performing an RBA AUSRIVAS bioassessment of a site's macroinvertebrate community. Further documentation is provided via the AUSRIVAS homepage, as well as additional aids in diagnosing the disturbance at a site, depending upon the band in which it falls.

3.2.4.2 Situation-dependent guidelines

The following subsections provide guidelines for protection of each of the three ecosystem conditions listed in Section 3.1, i.e. condition 1 ecosystems, of high conservation/ecological value; condition 2, slightly to moderately disturbed systems; and condition 3, highly disturbed systems. For condition 1 and condition 2 ecosystems, management involves tracking the intrinsic attributes of the ecosystems (the key structural and functional components) to ensure they do not deviate outside natural variability as determined from baseline knowledge or accruing knowledge. For any of the ecosystem conditions, local jurisdictions could negotiate site-specific guidelines alternative to those recommended below after considering site-specific factors.[a] (Elsewhere, the Guidelines recommend the type and number of indicators that should be incorporated in an environmental monitoring and assessment program, depending upon the situation.[b])

a See section 3.1.3.3

b Section 7.2.1

1. Sites of high conservation value (condition 1 ecosystems)

For most applications using bioindicators in Australia, there is insufficient information about ecosystems upon which to make informed judgments about an acceptable level of change. All stakeholders (e.g. developer and regulator) are strongly encouraged to adopt the following strategy towards determining appropriate guidelines for indicator responses: first, for collecting baseline data; then, detecting and assessing environmental impacts.

Baseline data collection

Using an appropriate statistical design for the indicator response as prescribed in the protocols,[c] parties should ensure an 'adequate' baseline is gathered for the indicators measured. This may be achieved by setting 'conservative' α, β and effect size, where the effect size is determined on the basis of statistical or other criteria. In the absence of clear information from which to set decision criteria, it is recommended default targets for ecologically conservative decisions be set at $\alpha = 0.1$, $\beta = 0.2$ (power of 0.8) and effect size = 10% of, or 1 SD about, the baseline mean, whichever is smaller. Whether these defaults are applied or not, the importance of sound and numerous baseline data cannot be over-emphasised. It is strongly recommended that baseline data be gathered from at least 3–5 control or reference locations (for biodiversity indicators at least) over a period of at least three years (all indicators) wherever possible. (See case study presented in Appendix 4, Vol 2, and Section 7.2 for rationale, justification and further discussion.) Guidelines are provided below for those situations in which it is not possible to meet these baseline requirements.[d]

c App. 3, Vol. 2 for protocols

d Section 3.2.4.2/4

The default guidelines for α, β, and effect size, from above, should not be simply accepted as a new convention (or dogma), but should be seen as the starting point for considering (and negotiating) what is appropriate or reasonable for each case. The setting of effect size should be an active and explicit decision, usually made on a

case-by-case basis. Mapstone (1995, 1996), for example, provides additional case studies describing the setting of statistical decision criteria. For some situations an effect size as small as 10% is achievable and deemed necessary.[a] For many others of the variables typically encountered in environmental work, it will be very difficult to detect changes of 10% or less about some mean, and perhaps impossible. In some cases, changes of 10% might be inconsequential, even in terms of an early warning system. Seeking to enforce monitoring to arbitrary decision criteria under such circumstances could result in a strong backlash against the principle of setting decision criteria *a priori*. However, relaxation of precautionary values should always be a clearly argued and thoroughly justified step. If insufficient information exists to justify such changes but nominated monitoring variables cannot be sampled rigorously enough to satisfy default criteria, then other candidate variables should be investigated as the mainstays for inferential decisions.

a See App. 4, Vol. 2

It is not always sensible to set an effect size of 10% (or some other value) of the time-averaged baseline mean. In some cases it may be necessary to stipulate an effect size that reflects the dynamics of the control sites and how they are related to the disturbance site during baseline monitoring. For example, say the measurement variable has a seasonal periodicity but the future disturbance site and control sites show different responses to seasonality. Then it would be necessary to model that knowledge into the effect size. At its simplest, this might mean having different effect sizes for tests in summer and winter.

The baseline data referred to above are for use in determining if change has occurred. Much of the information used for environmental impact assessments (EIAs) is required for ecosystem characterisation and impact prediction and whilst not 'baseline' in the statistically rigorous sense described above, should be adequate as pilot data to design monitoring programs used for impact detection. Once an environmental impact statement (EIS) is accepted and a development proposal is approved, either development should be delayed, or there should be a guarantee that no disturbance to aquatic ecosystems would occur, until adequate baseline are gathered. (Humphrey et al. (1999) are critical of aspects of the EIA process in Australia at least, in that too often developments proceed without adequate baseline data gathered to detect and assess potential disturbances.)

Detecting and assessing disturbances

The guidelines for detecting and assessing environmental impacts or disturbances are determined from *a priori* decisions made between all parties.[b] In the case of flexible decision-making in the spirit of cooperative best practice, intervention can be either (i) 'super-precautionary', sought once any apparent trend away from a baseline appears, or (ii) sought once a feedback 'trigger' or threshold has been reached. In the first of these two situations, management action may or may not be required when a 'positive' response is detected. The proponent/discharger may also wish to corroborate the results for an indicator with water chemistry data and data obtained for other biological indicators.

b Section 3.2.4.1

Alternatively, data may be being gathered for compliance assessment within a legal framework, under strict and rigorous hypothesis-testing. Here, using the default settings from (i) above, unless all parties have determined other values *a priori*, an unacceptable disturbance has occurred if $P < 0.1$ in the statistical test applied to the data.

It is strongly recommended that parties adopt a precautionary approach and respond wisely and in a timely manner to data gathered for 'early detection' indicators.

2. Slightly to moderately disturbed systems (condition 2 ecosystems)

Treat condition 2 ecosystems like condition 1 ecosystems[a] acknowledging that there may be negotiated deviations from default values prescribed for condition 1 ecosystems. Nevertheless, any decisions on effect size should be based on sound ecological principles of sustainability rather than arbitrary relaxation of the default values described above, or because of resource constraints.

a See Section 3.2.4.2/1

3. Highly disturbed systems (condition 3 ecosystems)

The philosophy of the Guidelines for these systems is that at worst, water quality is maintained. Ideally, the longer-term aim is towards improved water quality.

Normally, early detection indicators of sublethal toxicity would not be measured at these sites.[b] For these sites, any decisions on effect size can be arbitrary relaxations of the default values described above, although they should still be based on sound ecological principles of sustainability. Guidelines from 3.2.4.2/5 below should be applied for cases in which a rapid, broad-scale biodiversity indicator has been selected. Where rapid assessment methods are applied to small-scale problems (within a catchment), assessment of results must take into account the general inability of the methods to detect all but large water quality problems. Approaches recommended to enhance the general sensitivity of the methods are discussed in box 3.2.1 and in Section 7.3.3.

b Section 7.2.1.1/3

4. Sites where an insufficient baseline sampling period is available to meet key default guideline decision criteria

To compensate for an inability to gather sufficient baseline data, the Guidelines recommend that additional monitoring be carried out, including a greater number of indicators and/or sites for 'early detection' and biodiversity measurement (i.e. the 'multiple lines of evidence' concept[c]). Of course, resource constraints will limit the number of additional indicators and sites that can be monitored, but these resource constraints must be satisfactorily balanced with the need for unambiguous and meaningful results.

c Section 3.2.4.1

For a development that is in the planning stage, if there are inadequate baseline data against which to assess disturbance, it is recommended that data from all monitoring programs be submitted to an independent expert (or panel of experts) on a regular basis for assessment of acceptability. The same ethos of precaution and ecological sustainability, as applied to guidelines in other situations listed here, would influence the decisions made by the experts.

For existing developments for which adequate baseline data were never gathered, the project approval phase probably pre-dated the more stringent discharge licensing conditions that have subsequently been imposed by regulators. Apply the same procedures as for (i) from above.

For *a posteriori* monitoring of accidental discharges, continue monitoring until target indicator goals have been reached, as determined by an independent expert (or panel of experts). [d]

d Section 3.2.5

5. Broad-scale assessment of ecosystem health

Broad-scale assessments of ecosystem health are used to assess water quality for planning purposes, to set goals for remediation and rehabilitation programs, and to monitor and assess broad-scale disturbances such as diffuse pollution.

If a site is found to be below reference condition on the AUSRIVAS banding scheme (band B or lower), then it can be concluded that fewer invertebrate taxa have been found than would be expected on the basis of the particular AUSRIVAS model. A goal of subsequent management should be to improve the water and habitat quality so as to move the site indices closer to reference conditions or into band A.

If a site is found to be above reference condition on the AUSRIVAS banding scheme (band X), then further investigations are needed. The site may be naturally more diverse than surrounding reference sites, and therefore warrants special management to conserve that diversity. Alternatively, a naturally nutrient-poor site has received organic or nutrient enrichment with successful establishment of families of macroinvertebrates that would ordinarily not inhabit this site.[a]

a See Section 3.2.4.1/4

3.2.5 Assessing the success of remedial actions

For aquatic ecosystems long degraded by human disturbances in Australia and New Zealand, biological monitoring will be required to assess the success of remedial works put in place to improve water quality and ecological condition. The goals for remediation might be either restoration or rehabilitation. Restoration refers to attempts to restore an ecosystem to its configuration prior to the disturbance or disturbance. Rehabilitation refers to attempts to improve the ecological status of some attributes of a disturbed ecosystem. The expected management target would be improvement in the ecological condition or integrity of a site (or sites) and specific biodiversity indicators could be selected for the water quality problem identified.[b]

b Section 3.2.2.2

Invariably in these situations, there are no pre-disturbance data available to define a target ecological condition, and because of this the scope for applying formal statistical methods of inference is reduced.[c] The ecological target should then be assumed to resemble that of appropriate control locations, where these are available. The assumption being made in this process is that the indicator responded similarly in the control and disturbance areas before the disturbance. Simple hypotheses may be generated for these cases that test for likely indications of improvement. In all likelihood, there are too few data and too many uncertainties for formal statistical decision criteria[d] to be applied. Rather, monitoring is continued until target indicator goals have been reached. Expert panels can decide upon the goals and, if necessary, decide whether compliance has been achieved. In determining goals for rehabilitation or restoration, stakeholders and their consultants need to take into consideration the desired target ecosystem condition [e] as well as experience elsewhere in achieving biological recovery for the types of contaminants involved.[f]

c Sections 7.2.1.2 and 7.2

d Section 3.2.4

e Section 3.1.3

f Sections 7.2.2 and 7.2.3

3.3 Physical and chemical stressors

3.3.1 Introduction

A number of naturally-occurring physical and chemical stressors can cause serious degradation of aquatic ecosystems when ambient values are too high and/or too low. In this section, the following physical and chemical stressors are considered: nutrients, biodegradable organic matter, dissolved oxygen, turbidity, suspended particulate matter (SPM), temperature, salinity, pH and changes in flow regime. Other chemical stressors, such as ammonia, cyanide, heavy metals, biocides and other toxic organic compounds, are covered in Sections 3.4 and 3.5. Recommendations relating to the development of guidelines for the stressors not covered in these Guidelines (e.g. introduced species and habitat modifications) are contained in Section 8.5.2 of Volume 2.

The purpose of the guidelines provided in this section is to assist those involved in managing water resources to ensure that condition 2 (slightly to moderately disturbed) and condition 3 (highly disturbed) aquatic ecosystems are adequately protected. For ecosystems requiring the highest level of 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 the physical and chemical stressors.[a] For such highly-valued ecosystems, the statistical decision criteria for detecting any change should be ecologically conservative and based on sound ecological principles. This position should only be relaxed where there is considerable biological assessment data showing that such changes will not affect biological diversity in the system.[b]

a Section 3.1.3

b Section 3.1.3.2

Figure 3.3.1 is a flow chart of the steps involved in the detailed application of the guidelines for the physical and chemical stressors using risk-based 'guideline packages'.

The steps consist of selecting key stressors, then guideline trigger values, and then, where appropriate, a protocol for considering the effect of ecosystem-specific modifiers in reducing the biological effects of individual stressors. The steps are discussed in detail in this section.

The new approach for physical and chemical stressors recommended here differs from that in the 1992 ANZECC Water Quality Guidelines (ANZECC 1992) in a number of ways, the most significant being that:

- the guidelines are as specific as possible to each ecosystem. While not all of the required information is available yet, a start has been made by increasing the number of ecosystem types from two in the 1992 ANZECC Guidelines to six in these Guidelines.[c]

c Section 3.1.2

- the focus here is on providing issue-based information, aimed at protecting aquatic ecosystems from eight issues or problems caused by physical and chemical stressors.[d]

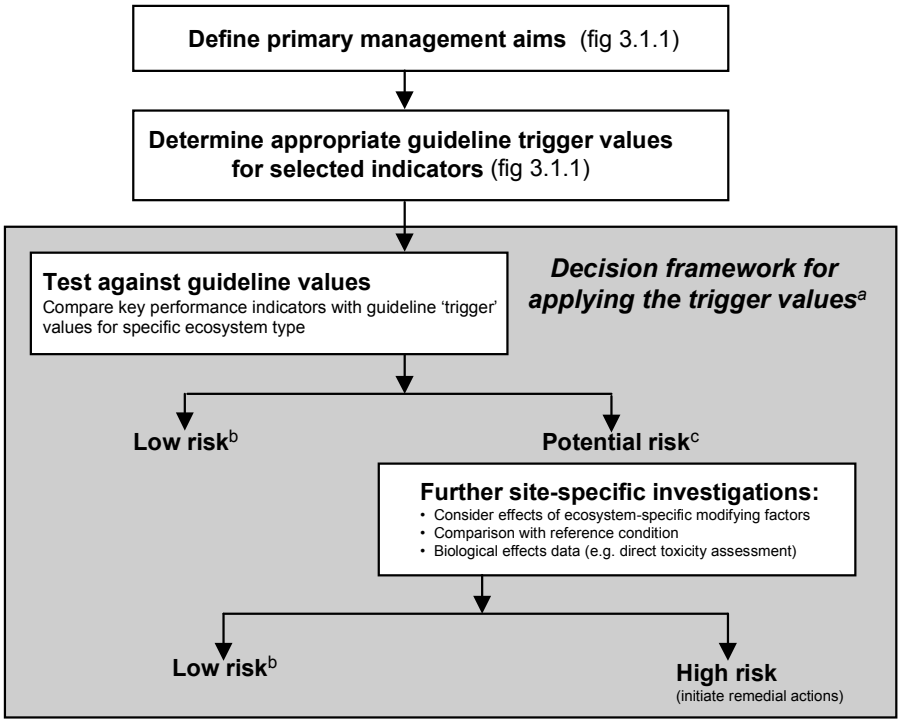
d Section 3.3.2.2

- available biological effects data have been used to determine low-risk guideline trigger values for toxic stressors for each ecosystem-type where sufficient data exist — i.e. a risk-based approach. For non-toxic stressors, low-risk guideline trigger values for key performance indicators have been determined by comparison with suitable reference ecosystems.[e]

e Section 3.3.2.1

- for each issue, the Guidelines give guideline packages (which are also risk-based) rather than simplistic threshold numbers for single indicators. These packages consist of key performance indicators, guideline trigger values and, where appropriate, a protocol for considering the effect of ecosystem-specific modifiers in reducing the biological effects. The packages help managers estimate whether low, possible or high risk exists at their sites as well as providing them with a means of refining guideline trigger values. The steps involved in applying the guideline packages are summarised in figure 3.3.1.
- guidelines for each issue are generally specified as concentrations, although it is recommended that load-based guidelines be developed for nutrients, biodegradable organic matter and suspended particulate matter.

The remainder of this section is divided into two parts: Section 3.3.2 outlines the philosophy adopted in developing guidelines for physical and chemical stressors, while Section 3.3.3 covers the detailed guideline packages for each of the eight issues considered.



[a] *Local biological effects data and some types of reference data (section 3.1.5) generally not required in the decision trees*

[b] *Possible refinement of trigger value after regular monitoring (section 3.1.5)*

[c] *Further investigations are not mandatory; users may opt to proceed to management/remedial action*

Figure 3.3.1 Decision tree framework ('guideline packages') for assessing the physico-chemical stressors in ambient waters

3.3.2 Philosophy used in developing guidelines for physical and chemical stressors

3.3.2.1 Types of physical and chemical stressors

Physical and chemical stressors can be classified broadly into two types (fig 3.3.2) depending on whether they have direct or indirect effects on the ecosystem.

Direct effects

Two types of physical and chemical stressors that directly affect aquatic ecosystems can be distinguished: those that are directly toxic to biota, and those that, while not directly toxic, can result in adverse changes to the ecosystem (e.g. to its biological diversity or its usefulness to humans). Excessive amounts of direct-effect stressors cause problems, but some of the elements and compounds covered here are essential at low concentrations for the effective functioning of the biota — nutrients such as phosphorus and nitrogen, and heavy metals such as copper and zinc, for example.

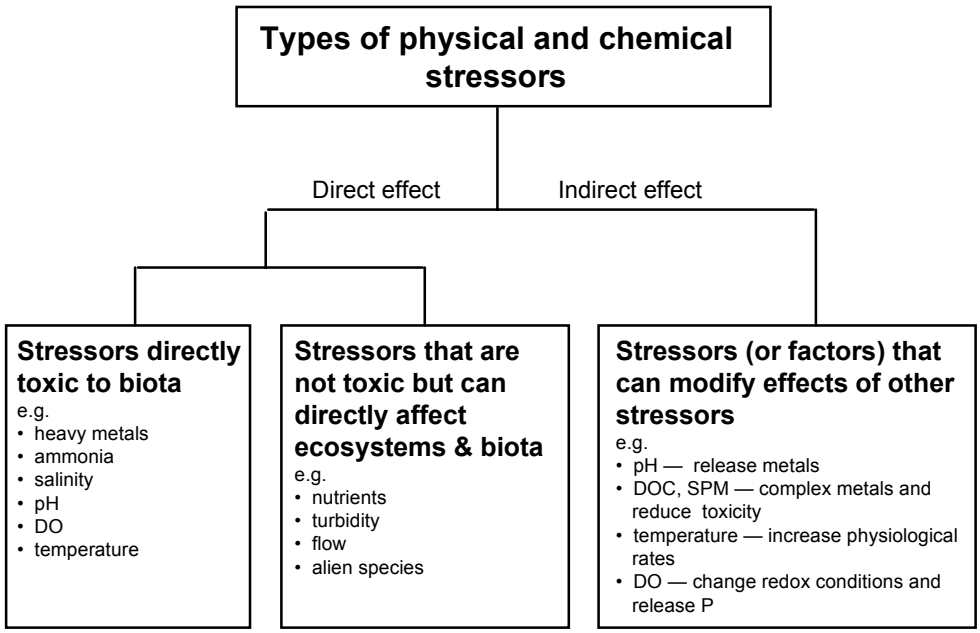


Figure 3.3.2 Types of physical and chemical stressors

a Section 3.4

The trigger values of toxic stressors are generally determined from laboratory ecotoxicity tests conducted on a range of sensitive aquatic plant and animal species.[a] However, salinity, pH and temperature are three toxic direct-effect stressors that are naturally very variable among and within ecosystem types and seasonally, and natural biological communities are adapted to the site-specific conditions. This suggests that trigger values for these three stressors may need to be based on site-specific biological effects data.

Examples of non-toxic direct-effect stressors include:

- nutrients, that can result in excessive algal growth and cyanobacterial blooms;
- suspended particulate matter, that can reduce light penetration into a waterbody and result in reduced primary production, possible deleterious effects on

phytoplankton, macrophytes and seagrasses, or smother benthic organisms and their habitats;

- organic matter decay processes, that can significantly reduce the dissolved oxygen concentration and cause death of aquatic organisms, particularly fish;
- water flow, which can significantly affect the amount and type of habitats present in a river or stream.

Indirect effects

Indirect stressors (or factors) are those that, while not directly affecting the biota, can affect other stressors making them more or less toxic. For example, dissolved oxygen can influence redox conditions and influence the uptake or release of nutrients by sediments. Equally, pH, dissolved organic carbon (DOC) and suspended particulate matter can have a major effect on the bioavailable concentrations of most heavy metals.

a See Section 3.1.5

Through the risk-based decision trees,[a] managers will consider these indirect stressors, with ecosystem-specific modifying factors, during the assessment of each issue. Although many effects of these modifying factors are reasonably well known from a theoretical viewpoint, there are few quantitative relationships (or models) that allow them to be used to develop more ecosystem-specific guidelines (Schnoor 1996). Recommendations made in Section 8.5.2 (Volume 2) cover the type of research and development needed to develop these relationships.[b]

b Section 8.5.2 (Volume 2)

c Section 8.2.1

For both types of physical and chemical stressors (eliciting direct or indirect effects on the ecosystem) background information is provided in Section 8.2.1 by way of Fact Sheets.[c] Key indicators provided in the Fact Sheets are nutrients, dissolved oxygen, turbidity and suspended particulate matter, salinity, temperature, optical properties, environmental flows and hydrodynamics.

3.3.2.2 Issues affecting aquatic ecosystems that are controlled by the physical and chemical stressors

Many aquatic ecosystems experience a range of problems that affect biodiversity or ecological health. These problems mostly result from human activities.

d See Sections 3.3.3, 8.2.3

This section focuses on the development of guideline 'packages' to address the specific issues[d] (summarised in table 3.3.1) likely to result from physical and chemical stressors:

- nuisance growth of aquatic plants (eutrophication);
- lack of dissolved oxygen (DO; asphyxiation of respiring organisms);
- excess suspended particulate matter (SPM; smothering of benthic organisms, inhibition of primary production);
- unnatural change in salinity (change in biological diversity);
- unnatural change in temperature (change in biological diversity);
- unnatural change in pH (change in biological diversity);
- poor optical properties of waterbodies (reduction in photosynthesis; change in predator–prey relationships);
- unnatural flow (inhibition of migration; associated temperature modification of spawning; changes in estuarine productivity).

Table 3.3.1 Summary of the condition indicators, performance indicators, and location of default trigger value tables, for each issue

Issue	Condition indicator/target	Performance indicators	Preferred method for obtaining trigger values [a]	Default trigger value for each ecosystem-type	Consider ecosystem-specific modifiers
1. Nuisance aquatic plants	Species composition Cell numbers Chlorophyll *a* conc	TP conc TN conc Chl *a* conc	Reference data Reference data Reference data	Tables 3.3.2, 3.3.4, 3.3.6, 3.3.8, 3.3.10	Yes — Section 3.3.3.1
2. Lack of DO	Reduced DO conc Species composition/ abundance	DO conc	Reference data	Tables 3.3.2, 3.3.4, 3.3.6, 3.3.8, 3.3.10	Yes — Section 3.3.3.2
3. Excess of SPM	Species composition/ abundance	SPM conc	Reference data	Tables 3.3.3, 3.3.5, 3.3.7, 3.3.9, 3.3.11	Yes — Section 8.2.3.2
4. Unnatural change in salinity	Species composition/ abundance	EC (salinity)	Reference data	Tables 3.3.3, 3.3.5, 3.3.7, 3.3.9, 3.3.11	No
5. Unnatural change in temperature	Species composition/ abundance	Temperature	Reference data	> 80%ile < 20%ile	No
6. Unnatural change in pH	Species composition/ abundance	pH	Reference data	Tables 3.3.2, 3.3.4, 3.3.6, 3.3.8, 3.3.10	No
7. Poor optical properties	Species composition/ abundance	Turbidity Light regime	Reference data Reference data	Tables 3.3.3, 3.3.5, 3.3.7, 3.3.9, 3.3.11	No
8. Unnatural flow regime	Species composition/ abundance Habitat change % wetted area	Flow regime			

[a] Where local biological and ecological effects data are unavailable.

3.3.2.3 Defining low-risk guideline trigger values

The guideline trigger values are the concentrations (or loads) of the key performance indicators, 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 a 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 that there is a 'low risk' that a problem exists. The alternative response, management/remedial action or further site-specific investigations, occurs if the trigger value is exceeded — i.e. a 'potential risk' exists.[a] The aim with further site-specific investigations is to determine whether or not there is an actual problem. Where, after continuous monitoring, with or without site-specific investigations, indicator values at sites are assessed as 'low risk' (no potential impact), guideline trigger values may be refined.[b] The guidelines have attempted as far as possible to make the trigger values specific for each of the different ecosystem types.

a See figure 3.3.1

b Section 3.1.5

Four sources of information are available for use when deriving low-risk trigger values: biological and ecological effects data, reference system data, predictive

modelling, or professional judgment.[a] The guidelines for physical and chemical stressors promote and focus principally on the derivation of low-risk trigger values, from biological and ecological effects data and through the use of reference data.

a See box 3.3.1

Ecosystem condition

As already mentioned, the Guidelines recognise three levels of ecosystem condition (1) high conservation/ecological value (condition 1 ecosystems), (2) slightly or moderately disturbed (condition 2 ecosystems), and (3) highly disturbed (condition 3 ecosystems), each with an associated level of protection (table 3.1.2). For condition 1 ecosystems, the Guidelines advise that there should be no change from ambient conditions, unless it can be demonstrated that such change will not compromise the maintenance of biological diversity in the system. Where comprehensive biological effects data are unavailable, a monitoring program is required to show that values of physical and chemical stressors are not changing, using statistically conservative decision criteria as the basis for evaluation.[b] Values of the criteria as recommended for biological indicators might be used as a starting point in negotiations;[c] further discussion of statistical error rates relevant to detecting change in physical and chemical stressors is provided in Section 7.4.4.1.[d]

b Sections 3.1.3.2, 3.1.7 & 7.2.3.3

c Section 3.2.4.2

d Section 7.4.4.1

Box 3.3.1. Sources of information for use when deriving low-risk trigger values

a) biological and ecological effects data — obtained either from biological effects testing using local biota and local waters (e.g. information derived by ***eriss*** for water release standards in Kakadu National Park), or from the scientific literature (preferably for Australia and New Zealand). This method is most appropriate for stressors directly toxic to biota (e.g. salinity, pH, DO, ammonia), but can also be applied to naturally-occurring stressors such as nutrients (e.g. nutrient addition bioassays). Ecological effects data are obtained through site- or ecosystem-specific laboratory and field experiments (see text below for deriving low-risk trigger values).

b) reference system data — obtained either from the same (undisturbed) ecosystem (i.e. from upstream of possible environmental impacts) or from a local but different system, or from regional reference ecosystems (Section 3.1.4). This is particularly useful for aquatic ecosystems where the management target is to maintain or restore the ecosystem, and where there are sufficient resources to obtain the required information on the reference ecosystem (see the text below for deriving low-risk trigger values).

c) predictive modelling — particularly useful for certain physical and chemical stressors whose disturbance occurs through transformations in the environment (e.g. nutrients, biodegradable organic matter). In these cases, because of the other factors involved, there does not appear to be a direct relationship between the ambient concentration of the stressor (e.g. total P concentration) and the biological response (e.g. algal biomass). However, there is often a plausible relationship between loading (or flux) and biological response.

d) professional judgement — may be used in cases where it will not be possible to obtain appropriate data for a reference ecosystem because insufficient study has been undertaken to provide an adequate data base. Such judgement should be supported by appropriate scientific information (e.g. information from 1992 ANZECC guidelines or other guideline documents, e.g. Hart 1974, Alabaster & Lloyd 1982, USEPA 1986, CCREM 1991), and the scientific literature.

Low-risk trigger values can be developed for condition 2 and condition 3 ecosystems:

- condition 2, slightly–moderately disturbed ecosystems, where the objective is to maintain biological diversity, acknowledging that stakeholders may also decide to allow some small change to biodiversity as well as improve or restore the ecosystem to a substantially unmodified condition, depending upon the situation;
- condition 3, highly disturbed ecosystems, where the management target will be to maintain, and preferably, improve the ecosystem, although in many cases the possibility of restoring the system to a substantially natural ecosystem may not be realistic. Urban aquatic systems (rivers, streams, wetlands, estuaries) are a case in point. For most of these, the hydrology in particular has been so markedly changed that at best a somewhat modified ecosystem can be achieved.

As suggested for high conservation/ecological value sites above, users also need to negotiate statistical decision criteria that can apply to any monitoring program for condition 2 or condition 3 ecosystems designed to detect change in values of physical and chemical stressors. Where maintenance of biological diversity is an important management goal, these criteria need to be set conservatively, but can be relaxed if some change to the system is acceptable.

The following sections outline the preferred hierarchy for deriving low-risk trigger values for aquatic systems (see figure 3.1.2). Where the preferred approach cannot be immediately implemented, a default or interim approach has been outlined.

3.3.2.4 Preferred approaches to deriving low-risk guideline trigger values

Using ecological effects data

For low-risk trigger values, measure the statistical distribution of water quality indicators either at a specific site (preferred), or an appropriate reference system(s), and also study the ecological and biological effects of physical and chemical stressors.[a] Then define the trigger value as the level of key physical or chemical stressors below which ecologically or biologically meaningful changes do not occur, i.e. the acceptable level of change.[b] Depending on the level of protection of the water body, the trigger value can be defined more or less conservatively after consultation with stakeholders, and using professional advice.[c]

a See Sections 3.2.3, 8.1 & Monitoring Guidelines

b Sections 3.3.2.7 & 7.2.3.3

c Section 8.5.2

Using reference data

Where there is insufficient information on ecological effects to determine an acceptable change from the reference condition, use an appropriate percentile of the reference data distribution to derive the trigger value. The percentile represents a measure that can be applied to data whether they be normally or non-normally distributed.

For naturally-occurring stressors, use data from appropriate reference systems to determine the low-risk trigger value for each key indicator. For these Guidelines, data collected after two years of monthly sampling are regarded as sufficient to indicate ecosystem variability and can be used to derive trigger values.

Ideally, in ecosystems not characterised by large seasonal or event-scale effects, develop trigger values for each month, i.e. a total of 12 low-risk trigger values. However, in some ecosystems, the relationships between physical and chemical indicators and key biological responses can be influenced by strong seasonal or

event-scale effects. In these systems, it will be necessary to monitor so as to detect these seasonal influences or events. For ecosystems where seasonal or event-driven processes dominate (e.g. tropical wetlands), it is possible to group the data and derive a number of trigger values corresponding to the key seasonal periods. For example, in wet–dry tropical systems two trigger values can be derived, one for the wet season and another for the dry season. In these instances, collect, partition and compare reference and test data according to specific flow regimes and/or seasons, particularly where biological responses to a particular stressor can be identified to be more pronounced in a particular season or flow regime.[a]

a See Sections 3.3.2.9 & 3.3.3.3

Where few data are available (i.e. few reference sites or sampling times) and seasonal and event influences are poorly defined, derive a single trigger value from available data as an interim measure.

Define trigger values for physical and chemical stressors for condition 2 ecosystems, in terms of the 80th and/or 20th percentile values obtained from an appropriate reference system. This choice is arbitrary (though reasonably conservative),[b] and professional advice should be sought wherever possible in selecting an appropriate point on the distribution curve for a system. For stressors that cause problems at high concentrations (e.g. nutrients, SPM, biochemical oxygen demand (BOD), salinity), take the 80th percentile of the reference distribution as the low-risk trigger value. For stressors that cause problems at low levels (e.g. low temperature water releases from reservoirs, low dissolved oxygen in waterbodies), use the 20th percentile of the reference distribution as a low-risk trigger value. For stressors that cause problems at both high and low values (e.g. temperature, salinity, pH), the desired range for the median concentration is defined by the 20th percentile and 80th percentile of the reference distribution.[c]

b Section 7.4.4

c Section 7.4.4.1

For condition 3 waterbodies, derive trigger values from site-specific biological or ecological effects data or, when an appropriate reference system(s) has been identified and there are sufficient resources to collect the necessary information, from local reference data. In this latter case, depending on management objectives, define trigger values using a conservative percentile value (e.g. 80th percentile value) to improve water quality (preferred approach), or a less conservative percentile (e.g. 90th percentile) to maintain water quality. Use professional judgement to determine the most appropriate cutoff percentile.

For either condition 2 or condition 3 ecosystems, where there are insufficient information or resources to undertake the necessary site-specific studies, use the default values provided that are derived from regional reference data (see following section).

3.3.2.5 Default approach to deriving low-risk guideline trigger values

The default approach to deriving trigger values has used the statistical distribution of reference data collected within five geographical regions across Australia and New Zealand. Here, depending on the stressor, a *measurable perturbation* in slightly to moderately disturbed ecosystems has been defined using the 80th and/or 20th percentile of the reference data.[d]

d Section 7.4.4.1

First, New Zealand and Australian state and territory representatives used percentile distributions of available data and professional judgement to derive trigger values for each ecosystem type in their regions. Trigger values were then collated, discussed and agreed for south-east Australia (VIC, NSW, ACT, south-

east QLD, and TAS), south-west Australia (southern WA), tropical Australia (northern WA, NT, northern QLD), south central Australia — low rainfall area (SA) and New Zealand (tables 3.3.2 to 3.3.11). Summaries of the data used to derive guideline trigger values for each Australian state and territory and for New Zealand are provided in Volume 2.[a]

a See Section 8.2.2

The default trigger values in the present guidelines were derived from ecosystem data for unmodified or slightly-modified ecosystems supplied by state agenicies. However, the choice of these reference systems was not based on any objective biological criteria. This lack of specificity may have resulted in inclusion of reference systems of varying quality, and further emphasises that the default trigger values should only be used until site- or ecosystem-specific values can be generated.

Default trigger values for temperature are not provided here. Managers need to define their own upper and lower low-risk trigger values, using the 80^{th} and 20^{th} percentiles, respectively, of ecosystem temperature distribution.

Tables 3.3.2–3.3.3 South-east Australia

The following tables outline default trigger values applicable to Victoria, New South Wales, south-east Queensland, the Australian Capital Territory and Tasmania. Where individual states or territories have developed their own regional guideline trigger values, those values should be used in preference to the default values provided below. (Upland streams are defined as those at >150 m altitude, while alpine streams are those at altitudes >1500 m.)

Table 3.3.2 Default trigger values for physical and chemical stressors for south-east Australia for slightly disturbed ecosystems. Trigger values are used to assess risk of adverse effects due to nutrients, biodegradable organic matter and pH in various ecosystem types. Data derived from trigger values supplied by Australian states and territories. Chl *a* = chlorophyll *a*, TP = total phosphorus, FRP = filterable reactive phosphate, TN = total nitrogen, NO_x = oxides of nitrogen, NH_4^+ = ammonium, DO = dissolved oxygen.

Ecosystem type	Chl *a*	TP	FRP	TN	NO_x	NH_4^+	DO (% saturation)[l]		pH	
	(μg L^{-1})	(μg P L^{-1})	(μg P L^{-1})	(μg N L^{-1})	(μg N L^{-1})	(μg N L^{-1})	Lower limit	Upper limit	Lower limit	Upper limit
Upland river	na[a]	20[b]	15[g]	250[c]	15[h]	13[i]	90	110	6.5	7.5[m]
Lowland river[d]	5	50	20	500	40[o]	20	85	110	6.5	8.0
Freshwater lakes & Reservoirs	5[e]	10	5	350	10	10	90	110	6.5	8.0[m]
Wetlands	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Estuaries[p]	4[f]	30	5[j]	300	15	15	80	110	7.0	8.5
Marine[p]	1[n]	25[n]	10	120	5[k]	15[k]	90	110	8.0	8.4

na = not applicable;
a = monitoring of periphyton and not phytoplankton biomass is recommended in upland rivers — values for periphyton biomass (mg Chl *a* m^{-2}) to be developed;
b = values are 30 μgL^{-1} for Qld rivers, 10 μgL^{-1} for Vic. alpine streams and 13 μgL^{-1} for Tas. rivers;
c = values are 100 μgL^{-1} for Vic. alpine streams and 480 μgL^{-1} for Tas. rivers;
d = values are 3 μgL^{-1} for Chl *a*, 25 μgL^{-1} for TP and 350 μgL^{-1} for TN for NSW & Vic. east flowing coastal rivers;
e = values are 3 μgL^{-1} for Tas. lakes;
f = value is 5 μgL^{-1} for Qld estuaries;
g = value is 5 μgL^{-1} for Vic. alpine streams and Tas. rivers;
h = value is 190 μgL^{-1} for Tas. rivers;
i = value is 10 μgL^{-1} for Qld. rivers;
j = value is 15 μgL^{-1} for Qld. estuaries;
k = values of 25 μgL^{-1} for NO_x and 20 μgL^{-1} for NH_4^+ for NSW are elevated due to frequent upwelling events;
l = dissolved oxygen values were derived from daytime measurements. Dissolved oxygen concentrations may vary diurnally and with depth. Monitoring programs should assess this potential variability (see Section 3.3.3.2);
m = values for NSW upland rivers are 6.5–8.0, for NSW lowland rivers 6.5–8.5, for humic rich Tas. lakes and rivers 4.0-6.5;
n = values are 20 μgL^{-1} for TP for offshore waters and 1.5 μgL^{-1} for Chl *a* for Qld inshore waters;
o = value is 60 μgL^{-1} for Qld rivers;
p = no data available for Tasmanian estuarine and marine waters. A precautionary approach should be adopted when applying default trigger values to these systems.

Table 3.3.3 Ranges of default trigger values for conductivity (EC, salinity), turbidity and suspended particulate matter (SPM) indicative of slightly disturbed ecosystems in south-east Australia. Ranges for turbidity and SPM are similar and only turbidity is reported here. Values reflect high site-specific and regional variability. Explanatory notes provide detail on specific variability issues for ecosystem type.

Ecosystem type	Salinity (μScm^{-1})	Explanatory notes
Upland rivers	30–350	Conductivity in upland streams will vary depending upon catchment geology. Low values are found in Vic. alpine regions (30 μScm^{-1}) and eastern highlands (55 μScm^{-1}), and high values (350 μScm^{-1}) in NSW rivers. Tasmanian rivers are mid-range (90 μScm^{-1}).
Lowland rivers	125–2200	Lowland rivers may have higher conductivity during low flow periods and if the system receives saline groundwater inputs. Low values are found in eastern highlands of Vic. (125 μScm^{-1}) and higher values in western lowlands and northern plains of Vic (2200 μScm^{-1}). NSW coastal rivers are typically in the range 200–300 μScm^{-1}.
Lakes & reservoirs	20–30	Conductivity in lakes and reservoirs is generally low, but will vary depending upon catchment geology. Values provided are typical of Tasmanian lakes and reservoirs.
	Turbidity (NTU)	
Upland rivers	2–25	Most good condition upland streams have low turbidity. High values may be observed during high flow events.
Lowland rivers	6–50	Turbidity in lowland rivers can be extremely variable. Values at the low end of the range would be found in rivers flowing through well vegetated catchments and at low flows. Values at the high end of the range would be found in rivers draining slightly disturbed catchments and in many rivers at high flows.
Lakes & reservoirs	1–20	Most deep lakes and reservoirs have low turbidity. However, shallow lakes and reservoirs may have higher natural turbidity due to wind-induced resuspension of sediments. Lakes and reservoirs in catchments with highly dispersible soils will have high turbidity.
Estuarine & marine	0.5–10	Low turbidity values are normally found in offshore waters. Higher values may be found in estuaries or inshore coastal waters due to wind-induced resuspension or to the input of turbid water from the catchment. Turbidity is not a very useful indicator in estuarine and marine waters. A move towards the measurement of light attenuation in preference to turbidity is recommended.

Tables 3.3.4–3.3.5 Tropical Australia

The following tables outline default trigger values applicable to northern Queensland, the Northern Territory and north-west Western Australia. Where states or territories have developed regional guideline trigger values those values should be used in preference to the default values provided below. (Upland streams are defined as those at >150 m altitude.)

Table 3.3.4 Default trigger values for physical and chemical stressors for tropical Australia for slightly disturbed ecosystems. Trigger values are used to assess risk of adverse effects due to nutrients, biodegradable organic matter and pH in various ecosystem types. Data derived from trigger values supplied by Australian states and territories, for the Northern Territory and regions north of Carnarvon in the west and Rockhampton in the east. Chl *a* = chlorophyll a, TP = total phosphorus, FRP = filterable reactive phosphate, TN = total nitrogen, NO_x = oxides of nitrogen, NH_4^+ = ammonium, DO = dissolved oxygen.

Ecosystem type		Chl *a*	TP	FRP	TN	NO_x	NH_4^+	DO (% saturation) [f]		pH	
		(µg L^{-1})	(µg P L^{-1})	(µg P L^{-1})	(µg N L^{-1})	(µg N L^{-1})	(µg N L^{-1})	Lower limit	Upper limit	Lower limit	Upper limit
Upland river[e]		na[a]	10	5	150	30	6	90	120	6.0	7.5
Lowland river[e]		5	10	4	200– 300[h]	10[b]	10	85	120	6.0	8.0
Freshwater lakes & reservoirs		3	10	5	350[c]	10[b]	10	90	120	6.0	8.0
Wetlands		10	10–50[g]	5–25[g]	350–1200[g]	10	10	90[b]	120[b]	6.0	8.0
Estuaries[e]		2	20	5	250	30	15	80	120	7.0	8.5
Marine	Inshore	0.7–1.4[d]	15	5	100	2–8[d]	1–10[d]	90	no data	8.0	8.4
	Offshore	0.5–0.9[d]	10	2–5[d]	100	1–4[d]	1–6[d]	90	no data	8.2	8.2

na = not applicable

a = monitoring of periphyton and not phytoplankton biomass is recommended in upland rivers — values for periphyton biomass (mg Chl *a* m^{-2}) to be developed;

b = Northern Territory values are 5µgL^{-1} for NO_x, and <80 (lower limit) and >110% saturation (upper limit) for DO;

c = this value represents turbid lakes only. Clear lakes have much lower values;

d = the lower values are typical of clear coral dominated waters (e.g. Great Barrier Reef), while higher values typical of turbid macrotidal systems (eg. North-west Shelf of WA);

e = no data available for tropical WA estuaries or rivers. A precautionary approach should be adopted when applying default trigger values to these systems;

f = dissolved oxygen values were derived from daytime measurements. Dissolved oxygen concentrations may vary diurnally and with depth. Monitoring programs should assess this potential variability (see Section 3.3.3.2);

g = higher values are indicative of tropical WA river pools;

h = lower values from rivers draining rainforest catchments.

Table 3.3.5 Ranges of default trigger values for conductivity (EC, salinity), turbidity and suspended particulate matter (SPM) indicative of slightly disturbed ecosystems in tropical Australia. Ranges for turbidity and SPM are similar and only turbidity is reported here. Values reflect high site-specific and regional variability. Explanatory notes provide detail on specific variability issues for groupings of ecosystem type.

Ecosystem type	Salinity (μScm^{-1})	Explanatory notes
Upland & lowland rivers	20–250	Conductivity in upland streams will vary depending upon catchment geology. Values at the lower end of the range are typical of ephemeral flowing NT rivers. Catchment type may influence values for Qld lowland rivers (e.g. 150 μScm^{-1} for rivers draining rainforest catchments, 250 μScm^{-1} for savanna catchments). The first flush of water following early seasonal rains may result in temporarily high values.
Lakes, reservoirs & wetlands	90–900	Values at the lower end of the range are found in permanent billabongs in the NT. Higher conductivity values will occur during summer when water levels are reduced due to evaporation. WA wetlands can have values higher than 900 μScm^{-1}. Turbid freshwater lakes in Qld have reported conductivities of approx. 170 μScm^{-1}.
	Turbidity (NTU)	
Upland & lowland rivers	2–15	Low values for base flow conditions in NT rivers. QLD turbidity and SPM values highly variable and dependent on degree of catchment modification and seasonal rainfall runoff.
Lakes, reservoirs & wetlands	2–200	Most deep lakes and reservoirs have low turbidity. However, shallow lakes and reservoirs may have higher turbidity naturally due to wind-induced resuspension of sediments. Lakes and reservoirs in catchments with highly dispersible soils will have high turbidity. Wetlands vary greatly in turbidity depending upon the general condition of the catchment or river system draining into the wetland, recent flow events and the water level in the wetland.
Estuarine & marine	1–20	Low values indicative of offshore coral dominated waters. Higher values representative of estuarine waters. Turbidity is not a very useful indicator in estuarine and marine waters. A move towards the measurement of light attenuation in preference to turbidity is recommended. Typical light attenuation coefficients (log_{10}) in waters off north-west WA range from 0.17 for inshore waters to 0.07 for offshore waters.

Tables 3.3.6–3.3.7 South-west Australia

The following tables outline default trigger values applicable to southern Western Australia. Where regional guideline trigger values have been developed, those values should be used in preference to the default values provided below. The WA EPA is currently developing site-specific environmental quality criteria for Perth's coastal waters. (Upland streams are defined as those at >150 m altitude.)

Table 3.3.6 Default trigger values for physical and chemical stressors for south-west Australia for slightly disturbed ecosystems. Trigger values are used to assess risk of adverse effects due to nutrients, biodegradable organic matter and pH in various ecosystem types. Data derived from trigger values supplied by Western Australia. Chl *a* = chlorophyll *a*, TP = total phosphorus, FRP = filterable reactive phosphate, TN = total nitrogen, NO_x = oxides of nitrogen, NH_4^+ = ammonium, DO = dissolved oxygen.

Ecosystem type	Chl *a*	TP	FRP	TN	NO_x	NH_4^+	DO (% saturation) [i]		pH	
	(μg L^{-1})	(μg P L^{-1})	(μg P L^{-1})	(μg N L^{-1})	(μg N L^{-1})	(μg N L^{-1})	Lower limit	Upper limit	Lower limit	Upper limit
Upland river[f]	na[a]	20	10	450	200	60	90	na	6.5	8.0
Lowland river[f]	3–5	65	40	1200	150	80	80	120	6.5	8.0
Freshwater lakes & reservoirs	3–5	10	5	350	10	10	90	no data	6.5	8.0
Wetlands[d]	30	60	30	1500	100	40	90	120	7.0[e]	8.5[e]
Estuaries	3	30	5	750	45	40	90	110	7.5	8.5
Marine[g,h] Inshore[c]	0.7	20 [b]	5[b]	230	5	5	90	na	8.0	8.4
Offshore	0.3 [b]	20 [b]	5	230	5	5	90	na	8.2	8.2

na = not applicable

a = monitoring of periphyton and not phytoplankton biomass is recommended in upland rivers — values for periphyton biomass (mg Chl *a* m^{-2}) to be developed;

b = summer (low rainfall) values, values higher in winter for Chl *a* (1.0 μgL^{-1}), TP (40 μg P L^{-1}), FRP (10 μg P L^{-1});

c = inshore waters defined as coastal lagoons (excluding estuaries) and embayments and waters less than 20 metres depth;

d = elevated nutrient concentrations in highly coloured wetlands (gilven >52 $g_{440}m^{-1}$) do not appear to stimulate algal growth;

e = in highly coloured wetlands (gilven >52 $g_{440}m^{-1}$) pH typically ranges 4.5–6.5;

f = all values derived during base river flow conditions not storm events;

g = nutrient concentrations alone are poor indicators of marine trophic status;

h = these trigger values are generic and therefore do not necessarily apply in all circumstances e.g. for some unprotected coastlines, such as Albany and Geographe Bay, it may be more appropriate to use offshore values for inshore waters;

i = dissolved oxygen values were derived from daytime measurements. Dissolved oxygen concentrations may vary diurnally and with depth. Monitoring programs should assess this potential variability (see Section 3.3.3.2).

Table 3.3.7 Range of default trigger values for conductivity (EC, salinity), turbidity and suspended particulate matter (SPM) indicative of slightly disturbed ecosystems in south-west Australia. Ranges for turbidity and SPM are similar and only turbidity is reported here. Values reflect high site-specific and regional variability. Explanatory notes provide detail on specific variability issues for ecosystem types.

Ecosystem type	Salinity (μScm^{-1})	Explanatory notes
Upland & lowland rivers	120–300	Conductivity in upland streams will vary depending upon catchment geology. Values at the lower end of the range are typically found in upland rivers, with higher values found in lowland rivers. Lower conductivity values are often observed following seasonal rainfall.
Lakes, reservoirs & wetlands	300–1500	Values at the lower end of the range are observed during seasonal rainfall events. Values even higher than 1500 μScm^{-1} are often found in saltwater lakes and marshes. Wetlands typically have conductivity values in the range 500—1500 μScm^{-1} over winter. Higher values (>3000 μScm^{-1}) are often measured in wetlands in summer due to evaporative water loss.
	Turbidity (NTU)	
Upland & lowland rivers	10–20	Turbidity and SPM are highly variable and dependent on seasonal rainfall runoff. These values representative of base river flow in lowland rivers.
Lakes, reservoirs & wetlands	10–100	Most deep lakes and reservoirs have low turbidity. However, shallow lakes and reservoirs may have higher turbidity naturally due to wind-induced resuspension of sediments. Lakes and reservoirs in catchments with highly dispersible soils will have high turbidity. Wetlands vary greatly in turbidity depending upon the general condition of the catchment or river system draining into the wetland and to the water level in the wetland.
Estuarine & marine	1–2	Turbidity is not a very useful indicator in estuarine and marine waters. A more appropriate measure for WA coastal waters is light attenuation coefficient. Light attenuation coefficients (log_{10}) of 0.05–0.08 m^{-1} are indicative of unmodified offshore waters and 0.09–0.13 m^{-1} for unmodified inshore waters, depending on exposure. Light attenuation coefficients (log_{10}) for unmodified estuaries typically range 0.3–1.0 m^{-1}, although more elevated values can be associated with increased particulate loading or humic rich waters following seasonal rainfall events.

Tables 3.3.8–3.3.9 South central Australia — low rainfall area

The following tables outline default trigger values applicable to South Australia. Where regional guideline trigger values have been developed those values should be used in preference to the default values provided below. (Upland streams are defined as those at >150 m altitude.)

Table 3.3.8 Default trigger values for physical and chemical stressors for south central Australia — low rainfall areas — for slightly disturbed ecosystems. Trigger values are used to assess risk of adverse effects due to nutrients, biodegradable organic matter and pH in various ecosystem types. Data derived from trigger values supplied by South Australia. Chl *a* = chlorophyll *a*, TP = total phosphorus, FRP = filterable reactive phosphate, TN = total nitrogen, NO_x = oxides of nitrogen, NH_4^+ = ammonium, DO = dissolved oxygen.

Ecosystem type	Chl *a*	TP	FRP	TN	NO_x	NH_4^+	DO (% saturation)		pH	
	(μg L^{-1})	(μg P L^{-1})	(μg P L^{-1})	(μg N L^{-1})	(μg N L^{-1})	(μg N L^{-1})	Lower limit	Upper limit	Lower limit	Upper limit
Upland river	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Lowland river	no data	100	40	1000	100	100	90	no data	6.5	9.0
Freshwater lakes & reservoirs	no data	25	10	1000	100	25	90	no data	6.5	9.0
Wetlands	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
Estuaries	5	100	10	1000	100	50	90	no data	6.5	9.0
Marine	1	100	10	1000	50	50	no data	no data	8.0	8.5

Table 3.3.9 Ranges of default trigger values for conductivity (EC, salinity), turbidity and suspended particulate matter (SPM) indicative of slightly disturbed ecosystems in south central Australia — low rainfall areas. Ranges for turbidity and SPM are similar and only turbidity is reported here. Values reflect high site-specific and regional variability. Explanatory notes provide detail on specific variability issues for groupings of ecosystem type.

Ecosystem types	Salinity (μScm^{-1})	Explanatory notes
Lowland rivers	100–5000	Salinity can be highly variable depending on flow.
Lakes, reservoirs & wetlands	300–1000	Wetlands can have substantially higher salinity due to saline groundwater intrusion and evaporation.
	Turbidity (NTU)	
Upland & lowland rivers	1–50	Turbidity and SPM are highly variable and dependent on seasonal rainfall runoff.
Lakes & reservoirs/ wetlands	1–100	Shallow lakes and reservoirs may have higher turbidity naturally due to wind-induced resuspension of sediments. Lakes and reservoirs in catchments with highly dispersible soils will have high turbidity.
Estuarine & marine	0.5–10	Higher values are representative of estuarine waters.

Tables 3.3.10–3.3.11 New Zealand

The following tables outline default trigger values applicable to New Zealand. Where regional guideline trigger values have been developed, those values should be used in preference to the default values provided below. (Upland streams are defined as those at >150 m altitude.)

For streams and rivers, New Zealand is developing a five-category ecosystem health categorisation system (A–E, with A being desirable and E undesirable). The draft National Agenda for Sustainable Water Management (NZ Ministry for the Environment 1999) proposes as a long-term goal that all streams are in C grade or better. For lakes, New Zealand has developed a fine scale lakes trophic assessment system, that enables water managers to objectively score the trophic condition of the lake. This assessment system combines a number of physical and chemical parameters. These parameters vary considerably across New Zealand, depending, for example, on whether a lake drains a volcanic catchment, in which case nitrate is a critical parameter, or whether the lake drains a hard rock catchment, in which case phosphorus is a critical parameter. Because of this variability, and because New Zealand has developed this trophic assessment system, it is not appropriate to propose trigger values for individual parameters from lakes.

Further work is needed to develop a categorisation system for New Zealand estuarine and marine ecosystems. Consideration should be given to the use of interim trigger values for south-east Australian estuarine and marine ecosystems (tables 3.3.2–3.3.3) until New Zealand estuarine and marine trigger values are developed.

Table 3.3.10 Default trigger values for physical and chemical stressors in New Zealand for slightly disturbed ecosystems. Trigger values are used to assess risk of adverse effects due to nutrients, biodegradable organic matter and pH in various ecosystem types. Chl *a* = chlorophyll *a*, TP = total phosphorus, FRP = filterable reactive phosphate,[d] TN = total nitrogen, NO_x = oxides of nitrogen, NH_4^+ = ammoniacal nitrogen, DO = dissolved oxygen.

Ecosystem type	Chl *a*	TP	FRP	TN	NO_x	NH_4^+	DO[e] (% saturation)		pH[e]	
	(μg L^{-1})	(μg P L^{-1})	(μg P L^{-1})	(μg N L^{-1})	(μg N L^{-1})	(μg N L^{-1})	Lower limit	Upper limit	Lower limit	Upper limit
Upland river	na[a]	26[b]	9[b]	295[b]	167[b]	10[b]	99	103	7.3	8.0
Lowland river	no data	33[c]	10[c]	614[c]	444[c]	21[c]	98	105	7.2	7.8

na = not applicable

a = monitoring of periphyton and not phytoplankton biomass is recommended in upland rivers — values for periphyton biomass (mg Chl *a* m^{-2}) to be developed. New Zealand is currently making routine observations of periphyton cover.

b = values for glacial and lake-fed sites in upland rivers are lower;

c = values are lower for Haast River which receives waters from alpine regions;

d = commonly referred to as dissolved reactive phosphorus in New Zealand;

e = DO and pH percentiles may not be very useful as trigger values because of diurnal and seasonal variation — values listed are for daytime sampling.

Table 3.3.11 Default trigger values for water clarity (lower limit) and turbidity (upper limit) indicative of unmodified or slightly disturbed ecosystems in New Zealand

Ecosystem types	Upland rivers[a b]		Lowland rivers	
	Clarity (m^{-1})[c d]	Turbidity (NTU)[c d]	Clarity (m^{-1})	Turbidity (NTU)
	0.6	4.1	0.8	5.6

a = Light availability is generally less of an issue in NZ rivers and streams than is visual clarity because, in contrast to many of Australia's rivers, most NZ rivers are comparatively clear and/or shallow. Davies-Colley et al. (1992) recommend that visual clarity, light penetration and water colour are important optical properties of an ecosystem which need to be protected (see Volume 2). Neither turbidity nor visual clarity provide a useful estimate of light penetration — light penetration should be considered separately to turbidity or visual clarity. Clarity relates to the transmission of light through water and is measured by the visual range of a black disk (see NZ Ministry for the Environment (1994)) or a Secchi disk.

b = Recent work has shown that at least some NZ indigenous fish are sensitive to low levels of turbidity; however, it may also be desirable to protect the naturally high turbidities of alpine glacial lakes to prevent possible ecological impacts, such as change in predator–prey relationships.

c = Note that turbidity and visual water clarity are closely and inversely related, and the 80th percentile for turbidity is consistent with the 20th percentile for visibility and vice versa.

d = Clarity and turbidity values for glacial sites in upland rivers are lower and higher, respectively.

3.3.2.6 Comparison with the low-risk guideline trigger value

Where trigger values have been developed from reference data, it is advisable to compare the median of replicate samples from a test site with the low-risk trigger value. Statistically, the median represents the most robust descriptor of the test site data, while the reference percentile value represents the degree of excursion that the test median is permitted before triggering some action.

Two issues will influence the outcome of the comparison: the amount of data used to calculate the trigger value (minimum two years of monthly sampling); and the number of replicates used to calculate the median from the test site (minimum of a single sample). A fuller discussion of these issues, with guidance on statistical ramifications of changes in sample size, are provided in Section 7.4.4.1.

Control charting

It is best to continually compare the trigger values against the results gathered during ongoing monitoring of the physical and chemical indicators, using control charts. Control charting displays the data trends and gives early warning that the test site may be trending towards a high-risk situation. Further discussion on the applications of control charts may be found in Section 7.4.4.1 and in the Monitoring Guidelines (ANZECC & ARMCANZ 2000). Excursion of the test site value beyond the trigger value requires that further action be undertaken. This may include, simply, an examination of data for errors, comparisons with previous excursions, or the use of simple decision trees such as those outlined in the risk-based guideline packages.[a] Site specific investigations may also be required to decide if there is an issue or problem to be addressed.

a See Sections 3.3.3 & 8.2.3

3.3.2.7 Measuring acceptable ecological change

Measurement of 'acceptable' ecological change is difficult (Keough & Mapstone 1995, Mapstone 1995). In very few situations is there enough scientific knowledge to indicate if a certain minimum change from the prevailing or target condition will cause an adverse ecological effect. To define this level of change (a) water quality indicator distributions must be correlated with grades or levels of ecosystem health or integrity indicators/indices, and (b) substantiating potential cause and effect relationships must be identified through these correlations, using laboratory and field-based biological and ecological effects research.

A number of recent studies are trying to link physical and chemical stressors with ecological effects and thereby define meaningful criteria for monitoring ecosystem health:

- As mentioned above, New Zealand is developing a five-category ecosystem health classification for freshwater shingle streams draining hard rock catchments. These categories are derived by comparison with a reference condition, and are based on a number of desirable biological features such as trout spawning, presence of sensitive native fish and no growth of benthic filamentous green algae. Fifty streams have been graded, and the distribution of water quality stressors within each grade will be used to define trigger values for physical and chemical indicators (E Pyle, NZ Ministry for the Environment, pers. comm.).
- Four large-scale studies in Australia have aimed to determine the cause and effect relationships between coastal ecosystem health and physical and chemical stressors (Port Phillip Bay Study, Moreton Bay and Brisbane River

Wastewater Management Study, and two Perth studies — the Perth Coastal Water Study and the South Metropolitan Coastal Water Studies). These multidisciplinary studies have led to an understanding of the influence of key stressors on ecosystem structure (e.g. suspended sediment concentration effects on seagrass distribution) and function (e.g. nitrogen loading effects on denitrification). The design and implementation of further such studies will aid in defining acceptable levels of ecological change.[a]

a See Section 8.5.2

3.3.2.8 Load-based guidelines

Traditionally, water quality guidelines have been expressed in terms of the concentration of the stressor that should not be exceeded if problems are to be avoided (ANZECC 1992). Such concentration-based guidelines are based primarily on the prevention of toxic effects. In other situations, guidelines are better expressed in terms of the flux or loading (i.e. mass per unit time), rather than concentration.

While algal growth rate (or productivity) is related to the concentration of key nutrients in the water column, the biomass is more controlled by the total mass of these nutrients available to the growing algae (Wetzel 1975).[11] In many cases, the water column nutrient concentration is not a good indicator of algal biomass. For example, the net water column nutrient concentration could be quite small in an ecosystem with a high algal biomass but with rapid nutrient cycling. Load-based guidelines for nutrients are covered in more detail below.[b]

b Section 3.3.3.1 & case studies 1 & 2 in section 3.3.3

The dissolved oxygen concentration in a waterbody depends on the balance between the flux of bioavailable organic carbon and the rate at which heterotrophic bacteria use up oxygen in decomposing this material, and the daily inputs of oxygen by diffusion from the atmosphere (increased by mixing) and via photosynthesis by macrophytes and phytoplankton (Stumm & Morgan 1996). Load-based guidelines for bioavailable organic matter are covered below.[c]

c Section 3.3.3.2 & case study 4 in Section 8.2.3 (Vol. 2)

Load-based guidelines are applicable also for assessing the effects of sedimentation of suspended particulate matter in smothering benthic organisms. Both the rate of sedimentation and the critical depth of the deposited material are load-based.[d]

d Case study 5 (Vol. 2)

A number of case studies are presented to show the types of approaches (particularly those involving predictive modelling) that can be used to determine the sustainable load of particular materials for a particular ecosystem. We recommend that work in developing similar types of case studies be increased. A number of key research areas are identified in Section 8.5.2 of Volume 2.[e]

e Section 8.5.2 of Volume 2

3.3.2.9 Tropical ecosystems

Although the guideline packages address issues that can apply to all biogeographic regions, the case studies in Sections 3.3.3 and 8.2.3 use examples from temperate regions. There is a need for tropical, risk-based guideline packages to be developed for Australian aquatic ecosystems which are characterised by elevated seasonal temperatures and significant seasonal variability in rainfall and stream-flow patterns (Finlayson & McMahon 1988). Algal blooms may be an issue in some tropical marine and freshwater ecosystems. Extensive macrophyte assemblages can have direct (e.g. smothering) and indirect (e.g. on dissolved oxygen, nutrients and light

[11] Note: this assumes that growth is not limited by light and that losses of algae by zooplankton grazing, sedimentation and 'washout' from the system are small.

availability) effects on tropical wetlands, and risk-based guideline packages are needed to address the influences of key stressors on such systems.

Monitoring should be arranged so that it targets episodic events. For instance, seasonally-variable stream flows can cease for large parts of the year. In some streams and reservoirs, slow flowing or pooled water leads to thermal stratification, which together with autochthonous organic loading, results in naturally low and variable dissolved oxygen concentrations (MacKinnon & Herbert 1996, Townsend 1999). Seasonal rainfall events often produce 'first-flush' loads of stressors that can cause rapid changes in stressor concentrations (Hart et al. 1987, Townsend et al. 1992) that may not be captured with routine monitoring programs.

There are few data for tropical water bodies; site- or ecosystem-specific reference data need to be collected for tropical ecosystems. The approach recommended in these Guidelines[a] — studies of site-specific biological or ecological effects to develop local trigger values — is also especially appropriate in ecosystems that demonstrate such a high degree of variability in physical and chemical stressors (e.g. wet and wet–dry tropics).

a See Section 3.3.2.4

3.3.3 Guideline packages for applying the guideline trigger values to sites

3.3.3.1 Risk-based guideline packages

Ideally, a *guideline package*, consisting of low-risk trigger values and a protocol for including effects of environmental modifiers, should be developed for each ecosystem issue and each ecosystem type. At this stage, only a limited number of packages can be recommended. Guideline packages are shown and discussed here for two issues:

- nuisance growth of aquatic plants, and
- lack of dissolved oxygen.

Further guideline packages are provided in Section 8.2.3 for:

- excess suspended particulate matter (SPM),
- unnatural change in salinity,
- unnatural change in temperature,
- unnatural change in pH,
- poor optical properties,
- unnatural flow.

Each guideline package consists of two components (figure 3.3.1):

- *a set of low-risk trigger values* — A set of key stressors such as total phosphorus concentration has been identified for each issue. These are used for an initial decision about the risk of an adverse biological effect occurring. The low-risk trigger values for these key stressors need to be established as outlined in box 3.3.1. These trigger values are concentration-based, but protocols for the development of load-based guidelines are provided where these are more relevant.
- *a protocol for further investigating the risk where the trigger value is exceeded* — In these potential risk situations, ecosystem-specific modifying factors that may

alter the biological effect of the key stressor need to be considered before the final risk can be assessed. The suggested protocol involves a decision tree or predictive modelling approach where increasingly detailed investigations are undertaken (figure 3.3.1). For example, where testing of the key stressor against the appropriate trigger values suggests a potential risk of excessive cyanobacterial growth in a particular lowland river, the steps involved in further investigating this situation could be:

i. make a simple assessment of the possible effect of key ecosystem-specific modifiers on the biological effect of the stressor. A simple decision tree model for this type of assessment is provided in Case Study 1.

ii. if this simple assessment still suggests a potential risk of adverse biological effects, then undertake more sophisticated site-specific investigations and associated modelling. For example, a load-based model of the system to predict the relationship between nutrient loads, key ecosystem variables and aquatic plant growth,[a] or a more comprehensive ecosystem-based model of the system (see Case Study 4, Harris et al. 1996) could be devised.

a See Case Study 3 in Section 8.2.3, Vol. 2

In many cases there is insufficient information to allow quantification of the relationships between the key stressor and environmental factors controlling bioavailability.[b] It is essential that these relationships be clarified in the immediate future.

b Section 8.5.2 in Vol. 2

As discussed in Section 3.1.5, generally, local biological effects data and data from local reference site(s) that closely match the test site are not required in the decision trees.

3.3.3.2 Issue: Nuisance growth of aquatic plants

Background

High concentrations of nutrients, particularly phosphorus and nitrogen, and sometimes silica, can result in excessive growth of aquatic plants such as phytoplankton, cyanobacteria, macrophytes, seagrasses, and filamentous and attached algae, in a range of ecosystems, fresh and marine (AEC 1987, CSIRO & Melbourne Water 1996, WADEP 1996, DWR-NSW 1992, WAEPA 1988, Harris et al. 1996, Johnstone 1994, Jones 1992, McComb & Davis 1993, McDougall & Ho 1991, MDBC 1994, NZ Ministry for the Environment 1992).

The excessive growth can lead to a number of problems including:

- toxic effects, particularly due to cyanobacteria in fresh and brackish waters, and dinoflagellates in marine waters;
- reduction in dissolved oxygen concentrations when the plants die and are decomposed;
- reduction in recreational amenity (phytoplankton blooms and macrophytes in wetlands and lakes, seagrasses in estuaries and coastal lagoons);
- blocking of waterways and standing waterbodies by macrophytes;
- change in biodiversity.

Excessive growth of aquatic plants occurs when there are high concentrations and loads of nutrients. Other factors play a part in limiting the growth of nuisance species, particularly toxic cyanobacteria. The factors include hydraulic retention time, mixing conditions, light, temperature, suspended solids, grazing pressure and type of substrate.

Key indicators

Condition indicators	chlorophyll *a* (Chl *a*), cell numbers, species composition
Key stressors	total phosphorus (TP) and total nitrogen (TN) concentrations
Ecosystem modifiers	depend upon the ecosystem type, but will include hydraulic retention time (flows and volume of waterbody), mixing regimes, light regime, turbidity, temperature, suspended solids (nutrient sorption), grazing rates, and type of substrate.
Performance indicators	median (or mean) concentrations of Chl *a*, TP and TN measured under low flow conditions for rivers and streams and during the growth periods for other ecosystems.[12]

a See recommendations in Section 8.5, Vol. 2

Note that nutrients may also be remobilised and released from sediments. Sediment nutrient releases are influenced by the composition of the sediments (particularly their bioavailable organic matter, Fe, S, N, P, etc.), temperature, mixing regime of the water body and oxygen transfer rates. At present we cannot recommend quantitative relationships to estimate these releases. However, such relationships should become available in the next few years, and it is essential that these be incorporated into the guidelines as soon as possible.[a]

Low-risk trigger values

b Section 3.3.2.3

The method used to determine the low-risk trigger values will depend upon the desired level of protection.[b]

Slightly to moderately disturbed ecosystems (condition 2 ecosystems)

Depending upon the importance and present condition of the ecosystem, two approaches may be taken to derive the most appropriate trigger values for condition 2 ecosystems.

a) For important ecosystems, where an appropriate local reference system(s) is available, and there are sufficient resources to collect the necessary information for the reference system, the low-risk trigger concentrations for the three key performance indicators (TP, TN and Chl *a*) should be determined as the 80th percentile of the reference system(s) distribution. Where possible, the trigger value should be obtained for that part of the seasonal or flow period when the probability of aquatic plant growth is most likely.

b) The default regional trigger values contained in tables 3.3.2, 3.3.4, 3.3.6, 3.3.8 and 3.3.10 should be used for those situations where either an appropriate reference system is not available, or the scale of the operation makes it difficult to justify the allocation of resources to collect the necessary information on a reference system.

Highly disturbed ecosystems (condition 3 ecosystems)

a) For important waterbodies, and those in very poor condition, it is best to make appropriate site-specific scientific studies, and to use the information, with professional judgement and other relevant information, to derive trigger values.

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[12] In the future, it is recommended that sustainable nutrient loading rates be estimated for each major ecosystem type (see Section 8.5.2, Volume 2, for research and development recommendations).

Where local but higher-quality reference data are used, a less stringent cutoff than the 80th percentile value may be used. The 80th percentile values, however, should be used as a target for site improvement.

b) For highly disturbed waterbodies, where there is a lack of either information or resources to undertake the necessary site-specific studies, it is best to use the default, regional trigger values using professional judgement to derive a less stringent value if this is agreed upon by stakeholders.

Use of the guideline package

Figure 3.3.1 shows the recommended approach for determining the risk of nuisance aquatic plant growth occurring in a particular ecosystem. There are three steps.

- Test the three performance indicators (Chl *a*, TP, TN concentrations) for the particular ecosystem against the appropriate low-risk trigger value for that ecosystem type. Compare the trigger values with the median concentration for each performance indicator measured under low flow or high growth conditions.
- If test values are less than trigger values, there is low risk of adverse biological effects and no further action is required, except for regular monitoring of the key performance and condition indicators. If after regular monitoring a 'low risk' outcome is consistently obtained, there is scope to refine the guideline trigger value. If test values are higher than the trigger values, there is an increased risk that adverse biological effects will occur, and either management/remedial action or further ecosystem-specific investigation is required.[a]

a Section 3.1.5

- For some types of ecosystem, further investigation may be needed, to determine the influence of ecosystem-specific factors on the key stressors. Case studies 1, 2 and 3[b] illustrate how these factors might be used to modify the effect of high nutrient concentrations so that problems due to aquatic plants may not arise even though nutrient concentrations suggest otherwise. Relatively few quantitative relationships between these factors have been identified for Australian systems. More work needs to be undertaken on these relationships.

b Case Studies 1 & 2 in Section 3.3.3; Case Study 3 in Section 8.2.3 in Vol. 2

Sustainable nutrient loads

Although nutrient concentrations are responsible (together with other factors) for stimulating algal growth, it is the total load of the key nutrients in the ecosystem that controls the final biomass of aquatic plants. The balance between the nutrients (e.g. the N:P ratio) can also influence the composition of the algal community.

Transformation processes that occur in a waterbody release additional nutrients (e.g. from sediments, and suspended particles). It is difficult to account for these without a detailed knowledge of the system, and in many cases a predictive model (Lawrence 1997 a,b).

In Australia and New Zealand a number of advances now have helped define the 'sustainable nutrient loading' for particular waterbodies. For example, sustainable total phosphorus loads for the River Murray have been determined using a simplified Vollenweider model;[c] Harris et al. (1996) estimated the sustainable nutrient loads to Port Phillip Bay with particular emphasis on nitrogen; and sustainable nutrient loading rates have been recommended for several Western Australian estuaries and the coastal waters near Perth (Masini et al. 1992, 1994, WAWA 1995, WADEP 1996).

c See also Case Study 4 in Section 8.2.3, Vol. 2

Most of the models used to estimate sustainable loads rely on empirical relationships between phosphorus or nitrogen loads and chlorophyll *a* concentration. For example, Cary et al. (1995) found a significant linear relationship between the known externally-derived summer inorganic nitrogen loads to Cockburn Sound, WA, and the mean chlorophyll *a* concentration over a 13 year period. This relationship was used to define a total external nitrogen loading of 2030 kgN/d needed to sustain a target chlorophyll *a* concentration of 0.8 μg/L (WADEP 1996). Similarly, 'sustainable' total phosphorus loads in various sections of the River Murray system have been defined by relating the annual TP load to the water residence time in a particular reservoir or weir pool to estimate the TP concentration during the summer growth period. Then using published (or empirically derived) TP vs Chl *a* relationships, the chlorophyll *a* concentration that would result from a particular TP load has been predicted. Using this information, it has been possible to define a TP load for that waterbody that will sustain a particular target chlorophyll *a* concentration.

3.3.3.3 Issue: Lack of dissolved oxygen

Background

Low dissolved oxygen (DO) concentration has an adverse effect on many aquatic organisms (e.g. fish, invertebrates and microorganisms) which depend upon oxygen dissolved in the water for efficient functioning. It can also cause reducing conditions in sediments, so the sediments release previously-bound nutrients and toxicants to the water column where they may add to existing problems.

The concentration of DO is highly dependent on temperature, salinity, biological activity (microbial, primary production) and rate of transfer from the atmosphere. Under natural conditions, DO will change, sometimes considerably, over a daily (or diurnal) period, and highly productive systems (e.g. tropical wetlands, dune lakes and estuaries) can become severely depleted in DO, particularly when these systems are stratified.

Of greater concern is the significant decrease in DO that can occur when organic matter is added (e.g. from sewage effluent or dead plant material). The depletion of DO depends on the load of biodegradable organic material and microbial activity, and re-aeration mechanisms operating. A number of predictive computer models now exist for estimating the DO depletion in a particular ecosystem type, and so it should be possible to estimate sustainable loads of biodegradable organic matter for most situations.

The 1992 ANZECC Guidelines recommended that dissolved oxygen should not normally be permitted to fall below 6 mgL^{-1} or 80–90% saturation, determined over at least one diurnal cycle. These guidelines were based almost exclusively on overseas data, since there were very few data on the oxygen tolerance of Australian or New Zealand aquatic organisms. The Australian data are restricted to freshwater fish, and suggest that DO concentrations below 5 mgL^{-1} are stressful to many species (Koehn & O'Connor 1990).

Key indicators

Condition indicators:	variation in DO concentration; species composition
Key stressor indicator:	loading of biodegradable organic matter (BOM, kg m^{-2} d^{-1})

Modifiers:	depend upon the ecosystem type, and include mixing condition (atmospheric O_2 transfer), photosynthetic O_2 production, rate of microbial decomposition, flow, temperature, pre-loading DO, mass of other O_2 consuming materials (e.g. nitrate)
Performance indicators:	median (or mean) DO concentration[13] measured under low flow conditions for rivers and streams and during low flow and high temperature periods for other ecosystems.

Low-risk trigger values

The method used to determine the low-risk trigger values will depend upon the desired level of protection.[a]

a See Section 3.3.2.3

Slightly to moderately disturbed ecosystems (condition 2 ecosystems)

Depending upon the significance and present condition of the ecosystem, two approaches may be taken to derive the most appropriate trigger values for condition 2 ecosystems.

a) For important ecosystems, where an appropriate reference system(s) is available, and there are sufficient resources to collect the necessary information for the reference system, the low-risk trigger concentrations for DO should be determined as the 20th percentile of the reference system(s) distribution. Where possible the trigger value should be obtained for low flow conditions for rivers and streams and during low flow and high temperature periods for other ecosystems, when DO concentrations are likely to be at their lowest.

b) The default trigger values contained in tables 3.3.2, 3.3.4, 3.3.6, 3.3.8 and 3.3.10 should be used where either an appropriate reference system is not available, or the scale of the operation makes it difficult to justify the allocation of resources to collect the necessary information on a reference system.

Highly disturbed ecosystems (condition 3 ecosystems)

a) For important waterbodies, and those in very poor condition, it is best to make appropriate site-specific scientific studies, and to use the information, with professional judgement and other relevant information, to derive trigger values. Where local but higher-quality reference data are used, a less stringent cutoff than the 20th percentile value may be used. The 20th percentile values, however, should be used as a target for site improvement.

b) For highly disturbed waterbodies, where there is a lack of either information or resources to undertake the necessary site-specific studies, it is best to use the default, regional trigger values using professional judgement to derive a less stringent value if this is agreed upon by stakeholders.

Sustainable loading rates for biodegradable organic matter should be estimated for each major ecosystem type, and used to develop load-based trigger values.[b]

b See recommendations in Section 8.5.2, Volume 2

[13] The median DO concentration for the period should be calculated using the lowest diurnal DO concentrations.

Use of the guideline package

Figure 3.3.1 shows the recommended approach for determining the risk of dissolved oxygen depletion occurring in a particular ecosystem. The approach involves three steps.

- Test the performance indicator (DO concentration) for the particular ecosystem against the appropriate low-risk trigger value for that ecosystem type. Compare the trigger values with the median (or mean) DO concentration measured under low flow conditions for rivers and streams and during low flow and high temperature periods for other ecosystems.
- If the test values are greater than the trigger values, there is low risk of adverse biological effects occurring and no further action is required, except for regular monitoring of the key performance indicators and condition indicators. If after regular monitoring a 'low risk' outcome is consistently obtained, there is scope to refine the guideline trigger value.[a] If test values are lower than trigger values, there is an increased risk that adverse biological effects will occur, and further ecosystem-specific investigation is required.
- Investigations to determine the influence of ecosystem-specific factors on the key stressors will depend upon the ecosystem type. A possible approach to calculate the sustainable load of biodegradable organic matter to waterbodies is provided by Lawrence (1997 a,b).[b]

a See Section 3.1.5

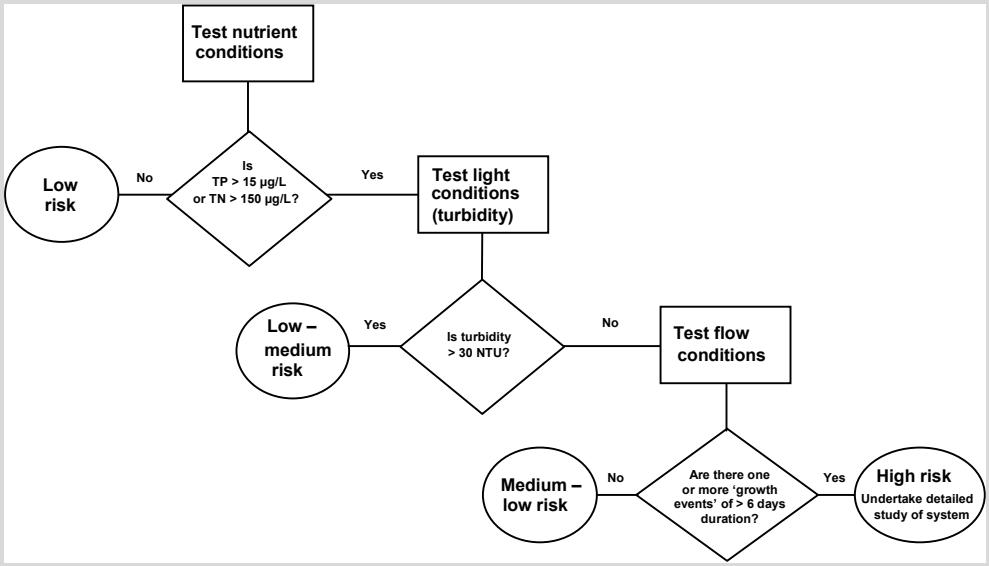
b See also Case Study 2 below

Case Study 1. Assessing the risk of cyanobacterial blooms in a lowland river

We present here an example of the use of a rather simple but effective decision tree, for assessing the risk of algal blooms arising from nutrients released to a lowland river in irrigation return drains. The protocol was initially developed as part of an environmental audit protocol developed for Goulburn-Murray Water (Hart et al. 1997; SKM 1997). More complex (and significantly more expensive) models have been developed for Port Phillip Bay (Harris et al. 1996), Hawkesbury-Nepean river (Sydney Water 1995) and the coastal waters off Perth (WAWA 1995, WADEP 1996).

The conceptual model for this case study (see figure below) assumes that algal growth in lowland rivers is controlled by three major factors:

- the concentrations of the nutrients P and N;
- the light climate (turbidity is used as a surrogate for light intensity because of a lack of data);
- the flow conditions in the river that are required for algal growth to occur.



The 'guideline package' in this case includes values for the nutrient concentrations (TP, TN) as the key stressors, and values for turbidity and flow as the modifiers. The numbers provided in the decision boxes for TP, TN and turbidity should be taken as indicative only because they will depend upon the particular ecosystem being considered.

The decision box for flow was based on the requirement that there be a sufficient period of low flow to allow algal numbers to increase to an alert level of 5000 cells mL^{-1}. A period of 6–10 days was estimated, based on an algal doubling time of 2 days and an initial algal concentration of 10–100 cells mL^{-1}. A 'growth event' was then defined as a period consisting of at least 6 consecutive days when the flow was less than the 25^{th} percentile flow obtained from the long term flow record for the system.

For the system in the figure, a high risk situation is indicated if the TP concentration is >15 μgL^{-1}, the turbidity less than 30 NTU, and there is more than one 'growth event' of >6 days duration per year. In this case, further investigation and appropriate management actions would be warranted.

Further refinement of this simple model could include:

- determining a more quantitative relationship between turbidity and the light climate for algal growth;
- validation of the assumption that the $<25^{th}$ percentile flows are the most appropriate low flow conditions to use. The present simple protocol does not take into consideration stratification that is now known to have a significant influence on cyanobacterial growth in lowland rivers (Webster et al. 1996);
- introduction of measures of the 'bioavailable' fractions of the nutrients rather than TP and TN (Hart et al. 1998);
- including the possibility that sediment release of nutrients (particularly phosphorus) may occur under low flow conditions;
- incorporation of the various decision 'rules' into a user-friendly computer program for ease of use by managers.

Case Study 2. Establishing sustainable organic matter loads for standing waterbodies

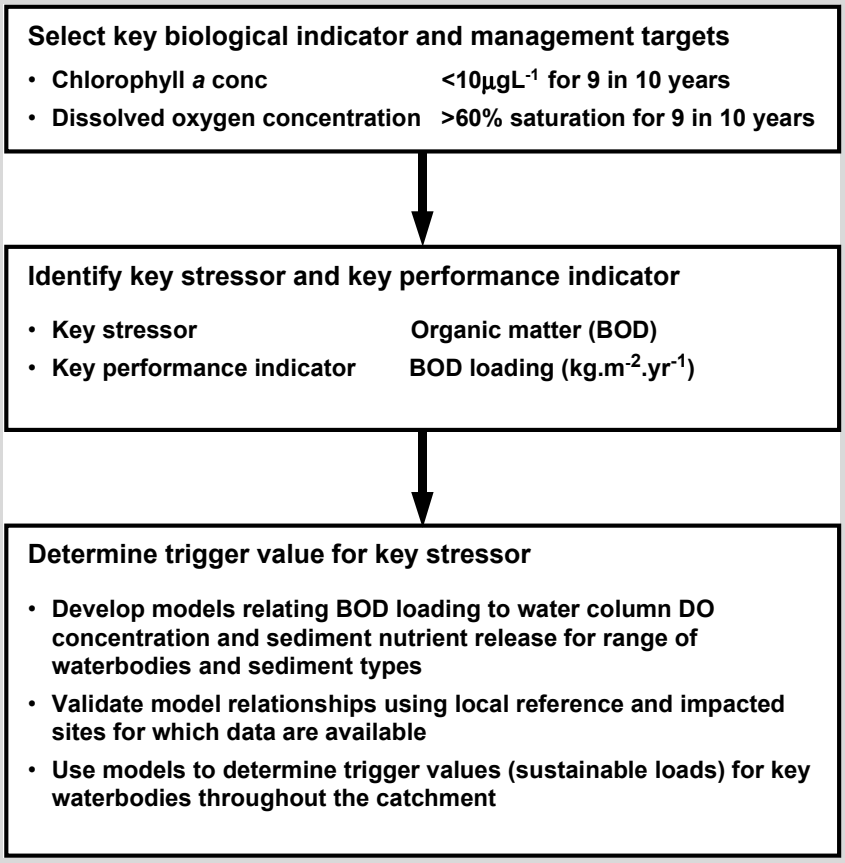
Australian research has shown that most rivers transport most water, suspended particulate matter, nutrients and organic matter during a small number of high flow events (Cosser 1989, Harris & Baxter 1996). In standing waterbodies, these event-driven loads can be augmented by point source discharges, decay of 'in-lake' algae, and releases from the sediments. High flow events are often followed by long periods of low flow conditions, when rapid decomposition of sedimented organic material by benthic bacteria can occur (Harris & Baxter 1996).

In many ecosystems, this sequence of events is quite normal and actually defines the ecosystem type. However, problems arise when an excess supply of organic material leads to de-oxygenation of the water column and to remobilisation of sediment-bound nutrients (and possibly toxic heavy metals) in bioavailable forms.

These processes may be accelerated if there is reduced transfer of oxygen from the atmosphere to the water column resulting from thermal stratification during the low flow and calm wind conditions typical of summer (Webster et al. 1996). This potential release of sediment-bound nutrients to the water column is of concern because by far the largest amount of phosphorus is stored in the sediments.

Thus, controls on the loading of organic matter to waterbodies is crucial in the effective management of the biological health and other uses of these waterbodies and, in particular, in controlling both dissolved oxygen concentrations and the remobilisation of nutrients from anaerobic sediments.

In terms of the approach proposed in these Guidelines, a possible method for establishing sustainable loads of organic matter to reservoirs, lakes and weir pools (and estuaries) is shown below (see also Lawrence 1997 a,b).



3.4 Water quality guidelines for toxicants

3.4.1 Introduction

This section provides guidance on the application of water quality guideline trigger values for toxicants. *Toxicants* is a term used for chemical contaminants that have the potential to exert toxic effects at concentrations that might be encountered in the environment. The risk-based decision scheme (Section 3.4.3) would be most commonly applied in ecosystems that could be classified as slightly to moderately disturbed (condition 2 ecosystems[a]). The decision scheme, which is optional, guides water managers on how to alter the trigger values for specific sites to account for local environmental conditions.

a See Section 3.1.3

The current NWQMS approach recommends moving away from relying solely on chemical guideline values for managing water quality, to the use of integrated approaches, comprising:

- chemical-specific guidelines coupled with water quality monitoring;
- direct toxicity assessment; and
- biological monitoring.

This approach will help to ensure that the water management focus keeps in view the goal of protecting the environment, and does not shift to merely meeting the numbers.

If more details are required, users may consult Volume 2 Section 8.3.2 on the type of data used to derive guidelines, Section 8.3.3 on the general approaches and methods used, Section 8.3.4 on the derivation procedure and requirements for data, and Section 8.3.5 on application of the decision scheme. Section 8.3.6 provides more information on direct toxicity assessment (i.e. whole effluent and ambient water toxicity testing) and Section 8.3.7 outlines the data used to derive each trigger value and summarises relevant scientific and technical information currently available.

3.4.2 How guidelines are developed for toxicants

Numerical guidelines are an essential tool for the management of receiving waters where discharge of toxicants to the environment cannot reasonably be avoided. The guidelines aim to protect ambient waters from sustained exposures to toxicants, i.e. from chronic toxicity. The derived trigger values are chemical-specific estimates to help managers achieve this aim.

Most users of these guidelines will use the trigger values (table 3.4.1) either directly or as part of the risk-based decision scheme outlined in Section 3.4.3, and in most cases will not need to know how the figures were derived. However, a brief summary is provided here.

3.4.2.1 Toxicity data for deriving guideline trigger values

The preferred data for deriving trigger values come from multiple-species toxicity tests, i.e. field or model ecosystem (mesocosm) tests that represent the complex interactions of species in the field. However, many of these tests are difficult to interpret and there were few such data available that met screening requirements.

Most of the trigger values have been derived using data from single-species toxicity tests on a range of test species, because these formed the bulk of the concentration–response information. *High reliability* trigger values[a] were calculated from chronic 'no observable effect concentration' (NOEC) data. However the majority of trigger values were *moderate reliability* trigger values, derived from short-term acute toxicity data (from tests ≤96 h duration) by applying acute-to-chronic conversion factors.

a See Section 3.4.2.3

3.4.2.2 Extrapolating from laboratory data to protect aquatic ecosystems

Most reliable trigger values (table 3.4.1) were derived using a statistical distribution approach, modified from Aldenberg and Slob (1993). This approach[b] has been adopted in The Netherlands and is recommended by the OECD (1992, 1995). The approach is based on calculations of a probability distribution of aquatic toxicity end-points. It attempts to protect a pre-determined percentage of species, usually 95%, but enables quantitative alteration of protection levels. The 95 percent protection level is most commonly applied in these Guidelines to ecosystems that could be classified as slightly to moderately disturbed.

b Described in Section 8.3.3.3 in Vol. 2

The traditional approach for extrapolating from single-species toxicity data to protect ecosystems has been to apply arbitrary *assessment factors* to the lowest toxicity value for a particular chemical (ANZECC 1992). There are deficiencies in this approach (Warne 1998), and it has been used in the current Guidelines only when there was an inadequate data set for the statistical distribution approach. The smallest assessment factors (where they were used) were applied to a comprehensive set of available chronic toxicity data, rather than acute data, when there was a high degree of confidence that the values reflected the field situation. The use of the statistically-based 95% protection provides a more defensible basis for decisions than use of assessment factors.

For chemicals such as mercury, polychlorinated biphenyls (PCBs) and organochlorine pesticides, the main issue of concern is not their direct short-term toxic effect but the indirect risks associated with their longer-term concentration in organisms and the potential for secondary poisoning. Dietary accumulation can be an important route of uptake for some chemicals, and it will need to be addressed in future revisions of the Guidelines. There is currently no formal and specific international guidance for incorporating bioaccumulation into water quality guidelines. For those chemicals that have the potential to bioaccumulate, the decision scheme provides for site-specific re-assessment of this issue if suitable data become available. Field investigations of residue levels in appropriate organisms may provide additional evidence for whether or not bioaccumulation is an issue at the site under study. In the absence of such local data, a higher level of protection is recommended (e.g. 99% protection for slightly–moderately disturbed systems instead of 95%). Chemicals that have the potential to bioaccumulate are indicated in table 3.4.1 (footnote 'B').

3.4.2.3 Procedures for deriving trigger values for toxicants

Three grades of guideline trigger values are derived: *high, moderate* or *low reliability* trigger values. The grade depends on the data available and hence the confidence or reliability of the final figures (Warne 1998). Only *high* and *moderate reliability* trigger values are reported in table 3.4.1.

- *High reliability* guideline trigger values were derived from multiple-species data or chronic NOEC data, using the risk-based statistical distribution method.
- *Moderate reliability* guideline trigger values, which reflect a lower confidence in extrapolation methods, were derived from acute toxicity data. Again, where possible, the statistical distribution method was used with the acute toxicity data. It was then necessary to convert the figure from that calculation to a chronic protection figure by application of either calculated or default acute-to-chronic ratios.
- *Low reliability* guideline trigger values were derived, in the absence of a data set of sufficient quantity, using larger assessment factors to account for greater uncertainty. These are considered as interim or indicative working levels subject to more test data becoming available. Low reliability figures should not be used as default guidelines, although it is reasonable to use them in the risk-based decision scheme to determine if conditions at the site increase or decrease the potential risk. It is important to recognise the interim nature of the low reliability figures and the inherent uncertainties in their derivation and to obtain more data where appropriate. Hence they are only reported in Section 8.3.7.

It has not been possible to derive trigger values for every chemical. Section 8.3.4.5 of Volume 2 provides some preliminary guidance for deriving preliminary working levels for such chemicals, according to international guidance (OECD 1992, 1995).

3.4.2.4 Altering the level of protection for different ecosystem conditions

The trigger values derived using the statistical distribution method were calculated at four different protection levels, 99%, 95%, 90% and 80% (table 3.4.1). Here, protection level signifies the percentage of species expected to be protected. The decision to apply a certain protection level to a specific ecosystem is the prerogative of each particular state jurisdiction or catchment manager, in consultation with the community and stakeholders. State jurisdictions or catchment managers can choose to apply different levels of protection to different ecosystem conditions if there is confidence that the disturbance is due to an overall physico-chemical disturbance and not just structural alteration.

One way of viewing the continuum of disturbance is to apply the three 'categories of ecosystem condition' for aquatic ecosystems, described in Section 3.1.3. The recommended procedure for applying the different levels of protection to the continuum of ecosystem conditions is summarised for toxicants in table 3.4.2. In most cases, the 95% protection level trigger values (table 3.4.1) should apply to ecosystems that could be classified as slightly–moderately disturbed, although a higher protection level could be applied to slightly disturbed ecosystems where the management goal is no change in biodiversity. For a few chemicals, higher levels of protection are recommended as default levels for those ecosystems, and the recommended trigger values for typical slightly–moderately disturbed ecosystems are in the shaded boxes in table 3.4.1.

The highest protection level (99%) has been chosen as the default value for ecosystems with high conservation value, pending collection of local chemical and biological monitoring data. The 99% protection levels can also be used as default values for slightly–moderately disturbed systems where local data are lacking on bioaccumulation effects or where it is considered that the 95% protection level fails

to protect key test species. This usually only occurs where trigger values have been calculated from chronic data but fail to protect against acute toxicity or vice versa. Those chemicals are shown in table 3.4.1. An example of this is endosulfan, with which key Australian species show acute toxicity at or near the 95% protection trigger value.

For ecosystems that can be classified as highly disturbed, the 95% protection trigger values can still apply. However, depending on the state of the ecosystem, the management goals and the approval of the appropriate state or regional authority in consultation with the community, it can be appropriate to apply a less stringent guideline trigger value, say protection of 90% of species, or perhaps even 80%. These values are provided as intermediate targets for water quality improvement. If the trigger values have been calculated using assessment factors, there is no reliable way to predict what changes in ecosystem protection are provided by an arbitrary reduction in the factor.

Table 3.4.1 Trigger values for toxicants at alternative levels of protection. Values in grey shading are the trigger values applying to typical *slightly–moderately disturbed systems*; see table 3.4.2 and Section 3.4.2.4 for guidance on applying these levels to different ecosystem conditions.

Chemical		Trigger values for freshwater (μgL^{-1})				Trigger values for marine water (μgL^{-1})			
		Level of protection (% species)				Level of protection (% species)			
		99%	95%	90%	80%	99%	95%	90%	80%
METALS & METALLOIDS									
Aluminium	pH >6.5	27	55	80	150	ID	ID	ID	ID
Aluminium	pH <6.5	ID	ID	ID	ID	ID	ID	ID	ID
Antimony		ID	ID	ID	ID	ID	ID	ID	ID
Arsenic (As III)		1	24	94 [C]	360 [C]	ID	ID	ID	ID
Arsenic (AsV)		0.8	13	42	140 [C]	ID	ID	ID	ID
Beryllium		ID	ID	ID	ID	ID	ID	ID	ID
Bismuth		ID	ID	ID	ID	ID	ID	ID	ID
Boron		90	370 [C]	680 [C]	1300 [C]	ID	ID	ID	ID
Cadmium	H	0.06	0.2	0.4	0.8 [C]	0.7 [B]	5.5 [B, C]	14 [B, C]	36 [B, A]
Chromium (Cr III)	H	ID	ID	ID	ID	7.7	27.4	48.6	90.6
Chromium (CrVI)		0.01	1.0 [C]	6 [A]	40 [A]	0.14	4.4	20 [C]	85 [C]
Cobalt		ID	ID	ID	ID	0.005	1	14	150 [C]
Copper	H	1.0	1.4	1.8 [C]	2.5 [C]	0.3	1.3	3 [C]	8 [A]
Gallium		ID	ID	ID	ID	ID	ID	ID	ID
Iron		ID	ID	ID	ID	ID	ID	ID	ID
Lanthanum		ID	ID	ID	ID	ID	ID	ID	ID
Lead	H	1.0	3.4	5.6	9.4 [C]	2.2	4.4	6.6 [C]	12 [C]
Manganese		1200	1900[C]	2500[C]	3600[C]	ID	ID	ID	ID
Mercury (inorganic)	B	0.06	0.6	1.9 [C]	5.4 [A]	0.1	0.4 [C]	0.7 [C]	1.4 [C]
Mercury (methyl)		ID	ID	ID	ID	ID	ID	ID	ID
Molybdenum		ID	ID	ID	ID	ID	ID	ID	ID
Nickel	H	8	11	13	17 [C]	7	70 [C]	200 [A]	560[A]
Selenium (Total)	B	5	11	18	34	ID	ID	ID	ID
Selenium (SeIV)	B	ID	ID	ID	ID	ID	ID	ID	ID
Silver		0.02	0.05	0.1	0.2 [C]	0.8	1.4	1.8	2.6 [C]
Thallium		ID	ID	ID	ID	ID	ID	ID	ID
Tin (inorganic, SnIV)		ID	ID	ID	ID	ID	ID	ID	ID
Tributyltin (as µg/L Sn)		ID	ID	ID	ID	0.0004	0.006 [C]	0.02 [C]	0.05 [C]
Uranium		ID	ID	ID	ID	ID	ID	ID	ID
Vanadium		ID	ID	ID	ID	50	100	160	280
Zinc	H	2.4	8.0 [C]	15 [C]	31 [C]	7	15 [C]	23 [C]	43 [C]
NON-METALLIC INORGANICS									
Ammonia	D	320	900 [C]	1430 [C]	2300 [A]	500	910	1200	1700
Chlorine	E	0.4	3	6 [A]	13 [A]	ID	ID	ID	ID
Cyanide	F	4	7	11	18	2	4	7	14
Nitrate	J	17	700	3400 [C]	17000 [A]	ID	ID	ID	ID
Hydrogen sulfide	G	0.5	1.0	1.5	2.6	ID	ID	ID	ID
ORGANIC ALCOHOLS									
Ethanol		400	1400	2400 [C]	4000 [C]	ID	ID	ID	ID
Ethylene glycol		ID	ID	ID	ID	ID	ID	ID	ID
Isopropyl alcohol		ID	ID	ID	ID	ID	ID	ID	ID
CHLORINATED ALKANES									
Chloromethanes									
Dichloromethane		ID	ID	ID	ID	ID	ID	ID	ID
Chloroform		ID	ID	ID	ID	ID	ID	ID	ID
Carbon tetrachloride		ID	ID	ID	ID	ID	ID	ID	ID
Chloroethanes									
1,2-dichloroethane		ID	ID	ID	ID	ID	ID	ID	ID
1,1,1-trichloroethane		ID	ID	ID	ID	ID	ID	ID	ID

Chemical	Trigger values for freshwater (μgL^{-1})				Trigger values for marine water (μgL^{-1})			
	Level of protection (% species)				Level of protection (% species)			
	99%	**95%**	**90%**	**80%**	**99%**	**95%**	**90%**	**80%**
1,1,2-trichloroethane	5400	6500	7300	8400	140	1900	5800 [C]	18000 [C]
1,1,2,2-tetrachloroethane	ID	ID	ID	ID	ID	ID	ID	ID
Pentachloroethane	ID	ID	ID	ID	ID	ID	ID	ID
Hexachloroethane B	290	360	420	500	ID	ID	ID	ID
Chloropropanes								
1,1-dichloropropane	ID	ID	ID	ID	ID	ID	ID	ID
1,2-dichloropropane	ID	ID	ID	ID	ID	ID	ID	ID
1,3-dichloropropane	ID	ID	ID	ID	ID	ID	ID	ID
CHLORINATED ALKENES								
Chloroethylene	ID	ID	ID	ID	ID	ID	ID	ID
1,1-dichloroethylene	ID	ID	ID	ID	ID	ID	ID	ID
1,1,2-trichloroethylene	ID	ID	ID	ID	ID	ID	ID	ID
1,1,2,2-tetrachloroethylene	ID	ID	ID	ID	ID	ID	ID	ID
3-chloropropene	ID	ID	ID	ID	ID	ID	ID	ID
1,3-dichloropropene	ID	ID	ID	ID	ID	ID	ID	ID
ANILINES								
Aniline	8	250 [A]	1100 [A]	4800 [A]	ID	ID	ID	ID
2,4-dichloroaniline	0.6	7	20	60 [C]	ID	ID	ID	ID
2,5-dichloroaniline	ID	ID	ID	ID	ID	ID	ID	ID
3,4-dichloroaniline	1.3	3	6 [C]	13 [C]	85	150	190	260
3,5-dichloroaniline	ID	ID	ID	ID	ID	ID	ID	ID
Benzidine	ID	ID	ID	ID	ID	ID	ID	ID
Dichlorobenzidine	ID	ID	ID	ID	ID	ID	ID	ID
AROMATIC HYDROCARBONS								
Benzene	600	950	1300	2000	500 [C]	700 [C]	900 [C]	1300 [C]
Toluene	ID	ID	ID	ID	ID	ID	ID	ID
Ethylbenzene	ID	ID	ID	ID	ID	ID	ID	ID
o-xylene	200	350	470	640	ID	ID	ID	ID
m-xylene	ID	ID	ID	ID	ID	ID	ID	ID
p-xylene	140	200	250	340	ID	ID	ID	ID
m+*p*-xylene	ID	ID	ID	ID	ID	ID	ID	ID
Cumene	ID	ID	ID	ID	ID	ID	ID	ID
Polycyclic Aromatic Hydrocarbons								
Naphthalene	2.5	16	37	85	50 [C]	70 [C]	90 [C]	120 [C]
Anthracene B	ID	ID	ID	ID	ID	ID	ID	ID
Phenanthrene B	ID	ID	ID	ID	ID	ID	ID	ID
Fluoranthene B	ID	ID	ID	ID	ID	ID	ID	ID
Benzo(a)pyrene B	ID	ID	ID	ID	ID	ID	ID	ID
Nitrobenzenes								
Nitrobenzene	230	550	820	1300	ID	ID	ID	ID
1,2-dinitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1,3-dinitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1,4-dinitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1,3,5-trinitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1-methoxy-2-nitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1-methoxy-4-nitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1-chloro-2-nitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1-chloro-3-nitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1-chloro-4-nitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1-chloro-2,4-dinitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1,2-dichloro-3-nitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1,3-dichloro-5-nitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
1,4-dichloro-2-nitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID
2,4-dichloro-2-nitrobenzene	ID	ID	ID	ID	ID	ID	ID	ID

Chemical		Trigger values for freshwater (μgL^{-1})				Trigger values for marine water (μgL^{-1})			
		Level of protection (% species)				Level of protection (% species)			
		99%	95%	90%	80%	99%	95%	90%	80%
1,2,4,5-tetrachloro-3-nitrobenzene		ID	ID	ID	ID	ID	ID	ID	ID
1,5-dichloro-2,4-dinitrobenzene		ID	ID	ID	ID	ID	ID	ID	ID
1,3,5-trichloro-2,4-dinitrobenzene		ID	ID	ID	ID	ID	ID	ID	ID
1-fluoro-4-nitrobenzene		ID	ID	ID	ID	ID	ID	ID	ID
Nitrotoluenes									
2-nitrotoluene		ID	ID	ID	ID	ID	ID	ID	ID
3-nitrotoluene		ID	ID	ID	ID	ID	ID	ID	ID
4-nitrotoluene		ID	ID	ID	ID	ID	ID	ID	ID
2,3-dinitrotoluene		ID	ID	ID	ID	ID	ID	ID	ID
2,4-dinitrotoluene		16	65 [C]	130 [C]	250 [C]	ID	ID	ID	ID
2,4,6-trinitrotoluene		100	140	160	210	ID	ID	ID	ID
1,2-dimethyl-3-nitrobenzene		ID	ID	ID	ID	ID	ID	ID	ID
1,2-dimethyl-4-nitrobenzene		ID	ID	ID	ID	ID	ID	ID	ID
4-chloro-3-nitrotoluene		ID	ID	ID	ID	ID	ID	ID	ID
Chlorobenzenes and Chloronaphthalenes									
Monochlorobenzene		ID	ID	ID	ID	ID	ID	ID	ID
1,2-dichlorobenzene		120	160	200	270	ID	ID	ID	ID
1,3-dichlorobenzene		160	260	350	520 [C]	ID	ID	ID	ID
1,4-dichlorobenzene		40	60	75	100	ID	ID	ID	ID
1,2,3-trichlorobenzene	B	3	10	16	30 [C]	ID	ID	ID	ID
1,2,4-trichlorobenzene	B	85	170[C]	220[C]	300[C]	20	80	140	240
1,3,5-trichlorobenzene	B	ID	ID	ID	ID	ID	ID	ID	ID
1,2,3,4-tetrachlorobenzene	B	ID	ID	ID	ID	ID	ID	ID	ID
1,2,3,5-tetrachlorobenzene	B	ID	ID	ID	ID	ID	ID	ID	ID
1,2,4,5-tetrachlorobenzene	B	ID	ID	ID	ID	ID	ID	ID	ID
Pentachlorobenzene	B	ID	ID	ID	ID	ID	ID	ID	ID
Hexachlorobenzene	B	ID	ID	ID	ID	ID	ID	ID	ID
1-chloronaphthalene		ID	ID	ID	ID	ID	ID	ID	ID
Polychlorinated Biphenyls (PCBs) & Dioxins									
Capacitor 21	B	ID	ID	ID	ID	ID	ID	ID	ID
Aroclor 1016	B	ID	ID	ID	ID	ID	ID	ID	ID
Aroclor 1221	B	ID	ID	ID	ID	ID	ID	ID	ID
Aroclor 1232	B	ID	ID	ID	ID	ID	ID	ID	ID
Aroclor 1242	B	0.3	0.6	1.0	1.7	ID	ID	ID	ID
Aroclor 1248	B	ID	ID	ID	ID	ID	ID	ID	ID
Aroclor 1254	B	0.01	0.03	0.07	0.2	ID	ID	ID	ID
Aroclor 1260	B	ID	ID	ID	ID	ID	ID	ID	ID
Aroclor 1262	B	ID	ID	ID	ID	ID	ID	ID	ID
Aroclor 1268	B	ID	ID	ID	ID	ID	ID	ID	ID
2,3,4'-trichlorobiphenyl	B	ID	ID	ID	ID	ID	ID	ID	ID
4,4'-dichlorobiphenyl	B	ID	ID	ID	ID	ID	ID	ID	ID
2,2',4,5,5'-pentachloro-1,1'-biphenyl	B	ID	ID	ID	ID	ID	ID	ID	ID
2,4,6,2',4',6'-hexachlorobiphenyl	B	ID	ID	ID	ID	ID	ID	ID	ID
Total PCBs	B	ID	ID	ID	ID	ID	ID	ID	ID
2,3,7,8-TCDD	B	ID	ID	ID	ID	ID	ID	ID	ID
PHENOLS and XYLENOLS									
Phenol		85	320	600	1200 [C]	270	400	520	720
2,4-dimethylphenol		ID	ID	ID	ID	ID	ID	ID	ID
Nonylphenol		ID	ID	ID	ID	ID	ID	ID	ID
2-chlorophenol	T	340 [C]	490 [C]	630 [C]	870 [C]	ID	ID	ID	ID
3-chlorophenol	T	ID	ID	ID	ID	ID	ID	ID	ID
4-chlorophenol	T	160	220	280 [C]	360 [C]	ID	ID	ID	ID
2,3-dichlorophenol	T	ID	ID	ID	ID	ID	ID	ID	ID
2,4-dichlorophenol	T	120	160 [C]	200 [C]	270 [C]	ID	ID	ID	ID

Chemical		Trigger values for freshwater (μgL^{-1})				Trigger values for marine water (μgL^{-1})			
		Level of protection (% species)				Level of protection (% species)			
		99%	95%	90%	80%	99%	95%	90%	80%
2,5-dichlorophenol	T	ID	ID	ID	ID	ID	ID	ID	ID
2,6-dichlorophenol	T	ID	ID	ID	ID	ID	ID	ID	ID
3,4-dichlorophenol	T	ID	ID	ID	ID	ID	ID	ID	ID
3,5-dichlorophenol	T	ID	ID	ID	ID	ID	ID	ID	ID
2,3,4-trichlorophenol	T	ID	ID	ID	ID	ID	ID	ID	ID
2,3,5-trichlorophenol	T	ID	ID	ID	ID	ID	ID	ID	ID
2,3,6-trichlorophenol	T	ID	ID	ID	ID	ID	ID	ID	ID
2,4,5-trichlorophenol	T,B	ID	ID	ID	ID	ID	ID	ID	ID
2,4,6-trichlorophenol	T,B	3	20	40	95	ID	ID	ID	ID
2,3,4,5-tetrachlorophenol	T,B	ID	ID	ID	ID	ID	ID	ID	ID
2,3,4,6- tetrachlorophenol	T,B	10	20	25	30	ID	ID	ID	ID
2,3,5,6- tetrachlorophenol	T,B	ID	ID	ID	ID	ID	ID	ID	ID
Pentachlorophenol	T,B	3.6	10	17	27 [A]	11	22	33	55 [A]
Nitrophenols									
2-nitrophenol		ID	ID	ID	ID	ID	ID	ID	ID
3-nitrophenol		ID	ID	ID	ID	ID	ID	ID	ID
4-nitrophenol		ID	ID	ID	ID	ID	ID	ID	ID
2,4-dinitrophenol		13	45	80	140	ID	ID	ID	ID
2,4,6-trinitrophenol		ID	ID	ID	ID	ID	ID	ID	ID
ORGANIC SULFUR COMPOUNDS									
Carbon disulfide		ID	ID	ID	ID	ID	ID	ID	ID
Isopropyl disulfide		ID	ID	ID	ID	ID	ID	ID	ID
n-propyl sulfide		ID	ID	ID	ID	ID	ID	ID	ID
Propyl disulfide		ID	ID	ID	ID	ID	ID	ID	ID
Tert-butyl sulfide		ID	ID	ID	ID	ID	ID	ID	ID
Phenyl disulfide		ID	ID	ID	ID	ID	ID	ID	ID
Bis(dimethylthiocarbamyl)sulfide		ID	ID	ID	ID	ID	ID	ID	ID
Bis(diethylthiocarbamyl)disulfide		ID	ID	ID	ID	ID	ID	ID	ID
2-methoxy-4H-1,3,2-benzodioxaphosphorium-2-sulfide		ID	ID	ID	ID	ID	ID	ID	ID
Xanthates									
Potassium amyl xanthate		ID	ID	ID	ID	ID	ID	ID	ID
Potassium ethyl xanthate		ID	ID	ID	ID	ID	ID	ID	ID
Potassium hexyl xanthate		ID	ID	ID	ID	ID	ID	ID	ID
Potassium isopropyl xanthate		ID	ID	ID	ID	ID	ID	ID	ID
Sodium ethyl xanthate		ID	ID	ID	ID	ID	ID	ID	ID
Sodium isobutyl xanthate		ID	ID	ID	ID	ID	ID	ID	ID
Sodium isopropyl xanthate		ID	ID	ID	ID	ID	ID	ID	ID
Sodium sec-butyl xanthate		ID	ID	ID	ID	ID	ID	ID	ID
PHTHALATES									
Dimethylphthalate		3000	3700	4300	5100	ID	ID	ID	ID
Diethylphthalate		900	1000	1100	1300	ID	ID	ID	ID
Dibutylphthalate	B	9.9	26	40.2	64.6	ID	ID	ID	ID
Di(2-ethylhexyl)phthalate	B	ID	ID	ID	ID	ID	ID	ID	ID
MISCELLANEOUS INDUSTRIAL CHEMICALS									
Acetonitrile		ID	ID	ID	ID	ID	ID	ID	ID
Acrylonitrile		ID	ID	ID	ID	ID	ID	ID	ID
Poly(acrylonitrile-co-butadiene-co-styrene)		200	530	800 [C]	1200 [C]	200	250	280	340
Dimethylformamide		ID	ID	ID	ID	ID	ID	ID	ID
1,2-diphenylhydrazine		ID	ID	ID	ID	ID	ID	ID	ID
Diphenylnitrosamine		ID	ID	ID	ID	ID	ID	ID	ID
Hexachlorobutadiene		ID	ID	ID	ID	ID	ID	ID	ID
Hexachlorocyclopentadiene		ID	ID	ID	ID	ID	ID	ID	ID

Chemical		Trigger values for freshwater (µgL^{-1})				Trigger values for marine water (µgL^{-1})			
		Level of protection (% species)				Level of protection (% species)			
		99%	95%	90%	80%	99%	95%	90%	80%
Isophorone		ID	ID	ID	ID	ID	ID	ID	ID
ORGANOCHLORINE PESTICIDES									
Aldrin	B	ID	ID	ID	ID	ID	ID	ID	ID
Chlordane	B	0.03	0.08	0.14	0.27 [C]	ID	ID	ID	ID
DDE	B	ID	ID	ID	ID	ID	ID	ID	ID
DDT	B	0.006	0.01	0.02	0.04	ID	ID	ID	ID
Dicofol	B	ID	ID	ID	ID	ID	ID	ID	ID
Dieldrin	B	ID	ID	ID	ID	ID	ID	ID	ID
Endosulfan	B	0.03	0.2 [A]	0.6 [A]	1.8 [A]	0.005	0.01	0.02	0.05 [A]
Endosulfan alpha	B	ID	ID	ID	ID	ID	ID	ID	ID
Endosulfan beta	B	ID	ID	ID	ID	ID	ID	ID	ID
Endrin	B	0.01	0.02	0.04 [C]	0.06 [A]	0.004	0.008	0.01	0.02
Heptachlor	B	0.01	0.09	0.25	0.7 [A]	ID	ID	ID	ID
Lindane		0.07	0.2	0.4	1.0 [A]	ID	ID	ID	ID
Methoxychlor	B	ID	ID	ID	ID	ID	ID	ID	ID
Mirex	B	ID	ID	ID	ID	ID	ID	ID	ID
Toxaphene	B	0.1	0.2	0.3	0.5	ID	ID	ID	ID
ORGANOPHOSPHORUS PESTICIDES									
Azinphos methyl		0.01	0.02	0.05	0.11 [A]	ID	ID	ID	ID
Chlorpyrifos	B	0.00004	0.01	0.11 [A]	1.2 [A]	0.0005	0.009	0.04[A]	0.3 [A]
Demeton		ID	ID	ID	ID	ID	ID	ID	ID
Demeton-S-methyl		ID	ID	ID	ID	ID	ID	ID	ID
Diazinon		0.00003	0.01	0.2 [A]	2 [A]	ID	ID	ID	ID
Dimethoate		0.1	0.15	0.2	0.3	ID	ID	ID	ID
Fenitrothion		0.1	0.2	0.3	0.4	ID	ID	ID	ID
Malathion		0.002	0.05	0.2	1.1 [A]	ID	ID	ID	ID
Parathion		0.0007	0.004 [C]	0.01 [C]	0.04 [A]	ID	ID	ID	ID
Profenofos	B	ID	ID	ID	ID	ID	ID	ID	ID
Temephos	B	ID	ID	ID	ID	0.0004	0.05	0.4	3.6 [A]
CARBAMATE & OTHER PESTICIDES									
Carbofuran		0.06	1.2 [A]	4 [A]	15 [A]	ID	ID	ID	ID
Methomyl		0.5	3.5	9.5	23	ID	ID	ID	ID
S-methoprene		ID	ID	ID	ID	ID	ID	ID	ID
PYRETHROIDS									
Deltamethrin		ID	ID	ID	ID	ID	ID	ID	ID
Esfenvalerate		ID	0.001*	ID	ID	ID	ID	ID	ID
HERBICIDES & FUNGICIDES									
Bypyridilium herbicides									
Diquat		0.01	1.4	10	80 [A]	ID	ID	ID	ID
Paraquat		ID	ID	ID	ID	ID	ID	ID	ID
Phenoxyacetic acid herbicides									
MCPA		ID	ID	ID	ID	ID	ID	ID	ID
2,4-D		140	280	450	830	ID	ID	ID	ID
2,4,5-T		3	36	100	290 [A]	ID	ID	ID	ID
Sulfonylurea herbicides									
Bensulfuron		ID	ID	ID	ID	ID	ID	ID	ID
Metsulfuron		ID	ID	ID	ID	ID	ID	ID	ID
Thiocarbamate herbicides									
Molinate		0.1	3.4	14	57	ID	ID	ID	ID
Thiobencarb		1	2.8	4.6	8 [C]	ID	ID	ID	ID
Thiram		0.01	0.2	0.8 [C]	3 [A]	ID	ID	ID	ID
Triazine herbicides									
Amitrole		ID	ID	ID	ID	ID	ID	ID	ID
Atrazine		0.7	13	45 [C]	150 [C]	ID	ID	ID	ID

Chemical	Trigger values for freshwater (μgL^{-1})				Trigger values for marine water (μgL^{-1})			
	Level of protection (% species)				Level of protection (% species)			
	99%	95%	90%	80%	99%	95%	90%	80%
Hexazinone	ID	ID	ID	ID	ID	ID	ID	ID
Simazine	0.2	3.2	11	35	ID	ID	ID	ID
Urea herbicides								
Diuron	ID	ID	ID	ID	ID	ID	ID	ID
Tebuthiuron	0.02	2.2	20	160 C	ID	ID	ID	ID
Miscellaneous herbicides								
Acrolein	ID	ID	ID	ID	ID	ID	ID	ID
Bromacil	ID	ID	ID	ID	ID	ID	ID	ID
Glyphosate	370	1200	2000	3600 A	ID	ID	ID	ID
Imazethapyr	ID	ID	ID	ID	ID	ID	ID	ID
Ioxynil	ID	ID	ID	ID	ID	ID	ID	ID
Metolachlor	ID	ID	ID	ID	ID	ID	ID	ID
Sethoxydim	ID	ID	ID	ID	ID	ID	ID	ID
Trifluralin B	2.6	4.4	6	9 A	ID	ID	ID	ID
GENERIC GROUPS OF CHEMICALS								
Surfactants								
Linear alkylbenzene sulfonates (LAS)	65	280	520 C	1000 C	ID	ID	ID	ID
Alcohol ethoxyolated sulfate (AES)	340	650	850 C	1100 C	ID	ID	ID	ID
Alcohol ethoxylated surfactants (AE)	50	140	220	360 C	ID	ID	ID	ID
Oils & Petroleum Hydrocarbons	ID	ID	ID	ID	ID	ID	ID	ID
Oil Spill Dispersants								
BP 1100X	ID	ID	ID	ID	ID	ID	ID	ID
Corexit 7664	ID	ID	ID	ID	ID	ID	ID	ID
Corexit 8667		ID	ID	ID	ID	ID	ID	ID
Corexit 9527	ID	ID	ID	ID	230	1100	2200	4400 A
Corexit 9550	ID	ID	ID	ID	ID	ID	ID	ID

Notes: Where the final water quality guideline to be applied to a site is below current analytical practical quantitation limits, see Section 3.4.3.3 for guidance.

Most trigger values listed here for metals and metalloids are *High reliability* figures, derived from field or chronic NOEC data (see 3.4.2.3 for reference to Volume 2). The exceptions are *Moderate reliability* for freshwater aluminium (pH >6.5), manganese and marine chromium (III).

Most trigger values listed here for non-metallic inorganics and organic chemicals are *Moderate reliability* figures, derived from acute LC_{50} data (see 3.4.2.3 for reference to Volume 2). The exceptions are *High reliability* for freshwater ammonia, 3,4-DCA, endosulfan, chlorpyrifos, esfenvalerate, tebuthiuron, three surfactants and marine for 1,1,2-TCE and chlorpyrifos.

* = *High reliability* figure for esfenvalerate derived from mesocosm NOEC data (no alternative protection levels available).

A = Figure may not protect key test species from acute toxicity (and chronic) — check Section 8.3.7 for spread of data and its significance. 'A' indicates that trigger value > acute toxicity figure; note that trigger value should be <1/3 of acute figure (Section 8.3.4.4).

B = Chemicals for which possible bioaccumulation and secondary poisoning effects should be considered (see Sections 8.3.3.4 and 8.3.5.7).

C = Figure may not protect key test species from chronic toxicity (this refers to experimental chronic figures or geometric mean for species) — check Section 8.3.7 for spread of data and its significance. Where grey shading and 'C' coincide, refer to text in Section 8.3.7.

D = Ammonia as TOTAL ammonia as [NH_3-N] at pH 8. For changes in trigger value with pH refer to Section 8.3.7.2.

E = Chlorine as total chlorine, as [Cl]; see Section 8.3.7.2.

F = Cyanide as un-ionised HCN, measured as [CN]; see Section 8.3.7.2.

G = Sulfide as un-ionised H_2S, measured as [S]; see Section 8.3.7.2.

H = Chemicals for which algorithms have been provided in table 3.4.3 to account for the effects of hardness. The values have been calculated using a hardness of 30 mg/L $CaCO_3$. These should be adjusted to the site-specific hardness (see Section 3.4.3).

J = Figures protect against toxicity and do not relate to eutrophication issues. Refer to Section 3.3 if eutrophication is the issue of concern.

ID = Insufficient data to derive a reliable trigger value. Users advised to check if a low reliability value or an ECL is given in Section 8.3.7.

T = Tainting or flavour impairment of fish flesh may possibly occur at concentrations below the trigger value. See Sections 4.4.5.3/3 and 8.3.7.

Table 3.4.2 General framework for applying levels of protection for toxicants to different ecosystem conditions

Ecosystem condition	Level of protection
1 High conservation/ ecological value	• For anthropogenic toxicants, detection at any concentration could be grounds for source investigation and management intervention; for natural toxicants background concentrations should not be exceeded.[a] *Where local biological or chemical data have not yet been gathered*, apply the 99% protection levels (table 3.4.1) as default values. Any relaxation of these objectives should only occur where comprehensive biological effects and monitoring data clearly show that biodiversity would not be altered. • In the case of effluent discharges, Direct Toxicity Assessment (DTA) should also be required on the effluent. • Precautionary approach taken to assessment of post-baseline data through trend analysis or feedback triggers.
2 Slightly to moderately disturbed ecosystems	• Always preferable to use local biological effects data (including DTA) to derive guidelines. *If local biological effects data unavailable*, apply 95% protection levels (table 3.4.1) as default, low-risk trigger values.[b] 99% values are recommended for certain chemicals as noted in table 3.4.1.[c] • Precautionary approach may be required for assessment of post-baseline data through trend analysis or feedback triggers. • In the case of effluent discharges DTA may be required.
3 Highly disturbed ecosystems	• Apply the same guidelines as for slightly–moderately disturbed systems. However, the lower protection levels provided in the Guidelines may be accepted by stakeholders. • DTA could be used as an alternative approach for deriving site-specific guidelines.

a This means that indicator values at background and test sites should be statistically indistinguishable. It is acknowledged that it may not be strictly possible to meet this criterion in every situation.

b For slightly disturbed ecosystems where the management goal is no change in biodiversity, users may prefer to apply a higher protection level.

c 99% values recommended for chemicals that bioaccumulate or for which 95% provides inadequate protection for key test species. Jurisdictions may choose 99% values for some ecosystems that are more towards the slightly disturbed end of the continuum.

Modified values for this lowest level of protection should not approach levels that may cause acute toxicity. Footnotes in table 3.4.1 indicate where the figures at any protection level may cause acute or chronic toxicity but it is better to view the chemical descriptions[a] to gain the full context. The data distribution curves[b] illustrate the spread of the data (either acute or chronic) used to derive each trigger value. As indicated above, the emphasis should be on improvement of the *highly disturbed* ecosystem, not just maintenance of a degraded condition.

a See Section 8.3.7

b See toxicant databases on the CD-Rom

3.4.3 Applying guideline trigger values to sites

The guideline trigger values (table 3.4.1) were mostly derived primarily according to risk assessment principles, using data from laboratory tests in clean water. They represent the best current estimates of the concentrations of chemicals that should have no significant adverse effects on the aquatic ecosystem. They focus on direct toxic effects of individual chemicals, but it is intended that they be applied at specific sites, where possible, using the decision tree. This does not imply that application of the guidelines requires a full (quantitative) risk assessment.[c]

c See last paragraph of Section 2.1.4

These trigger values should not be considered as blanket guidelines for national water quality, because ecosystem types vary so widely throughout Australia and New Zealand. Such variations, even on a smaller scale, can have marked effects on the bioavailability, transport and degradation of chemicals, and on their toxicity. The trigger values may not apply to every aquatic ecosystem in Australia or New Zealand and in some instances adequate protection of the environment may require less or in some cases more stringent values.

3.4.3.1 Underlying philosophy for applying the guidelines

The general approach to use of the decision scheme is outlined in Section 3.1.5. If a trigger value listed in table 3.4.1 is exceeded at a site, further action results. The action can be:

i. Incorporation of additional information or further site-specific investigation to determine whether or not the chemical is posing a real risk to the environment. The investigation may determine the fraction of the chemical in the water that organisms can take up (the bioavailable fraction) to use for comparing with the trigger value. The investigation and/or regular monitoring may also result in refinement of the guideline figure to suit regional or local water quality parameters and other conditions. Such refinement would occur where exceedance of the trigger value was shown to have no adverse effects upon the ecosystem; alternatively

ii. Accept the trigger value without change as a guideline applying to that site and initiate management action or remediation.

The appropriate state or regional authority can often provide guidance and direction for implementing the decision scheme. Even if no other steps of the scheme are undertaken, it is important at least to adjust the trigger values for the six hardness-related metals (tables 3.4.3 and 3.4.4) to account for the local water hardness (step 9 of the scheme below). The trigger values for these metals have been derived at low water hardness, corresponding to high toxicity. In some cases, either the potential for environmental harm or the economic importance of the chemical may be sufficiently significant to warrant more intensive work to define a concentration that would adequately protect the environment.

Although the calculated site-specific guideline figure represents a concentration of toxicant that will not degrade the environmental value at the site, it should not be inferred that the environment could be contaminated up to this level (ANZECC 1992).

Where the site-specific guideline for a toxicant is exceeded, management action will normally result. However, this should be addressed under the processes of the individual states/territories or regions. It is important that toxicant guidelines are not interpreted in isolation from other ecosystem factors such as habitat, flow etc, as well as chemical fate. If the chemical is likely to be deposited in sediment, then consult the sediment guidelines.[a]

a See Section 3.5

In practice, not all site-specific data on a particular chemical are equivalent in extent, detail or method. If a manager were to apply the strict requirements used in deriving the original guideline trigger value, much valuable local information would be lost. Differing site-specific trigger values developed using various methods can be examined and weighted according to pre-determined criteria of quality and relevance to the ecosystem. This should be done in a commonsense

manner consistent with commonly applied principles of risk assessment to arrive at an appropriate figure (e.g. Menzie et al. 1996). The result can provide water managers with a way of integrating varying information on a particular site; if it is provided during assessments by the proponent, it can assist in maintaining consistent professional judgement. Inclusion of these multiple lines of evidence strengthens the overall result.[a]

a See Section 8.3.5.1

3.4.3.2 Decision tree for applying the guideline trigger values

The decision scheme outlined below gives step by step guidance on how to assess test site data and tailor the guideline trigger values according to site-specific environmental conditions. A simplified diagrammatic version of the decision tree is shown in figure 3.4.1.[b] The decision scheme is not mandatory and at any time a water manager can default to the original trigger value or use only those steps that are relevant to the situation and chemical at hand. The scheme is designed to determine if the conditions at a specific site reduce (or occasionally, increase) the risk to the environment of the study chemical.

b Section 8.3.5.1

The process of deriving water quality guidelines for a specific site begins with determination of the management aims, including a decision on the appropriate level of protection.[c] The next step is to assess the factors at the site that modify toxicity and bioavailability of the chemical. The measured or calculated bioavailable fraction can then be compared with the trigger value, or in some cases a site-specific guideline can be developed on the basis of known relationships between some physical or chemical parameters and the original trigger value. Examples of the latter include corrections for the effects of hardness for metals, the effects of pH for ammonia, or the effects of temperature for other chemicals. In the absence of quantitative data for such relationships, it may be possible to qualitatively estimate the likely trends in toxicity of a chemical, and hence risk, at a particular site. This is where professional judgement may be necessary, strengthened by examining multiple lines of evidence.

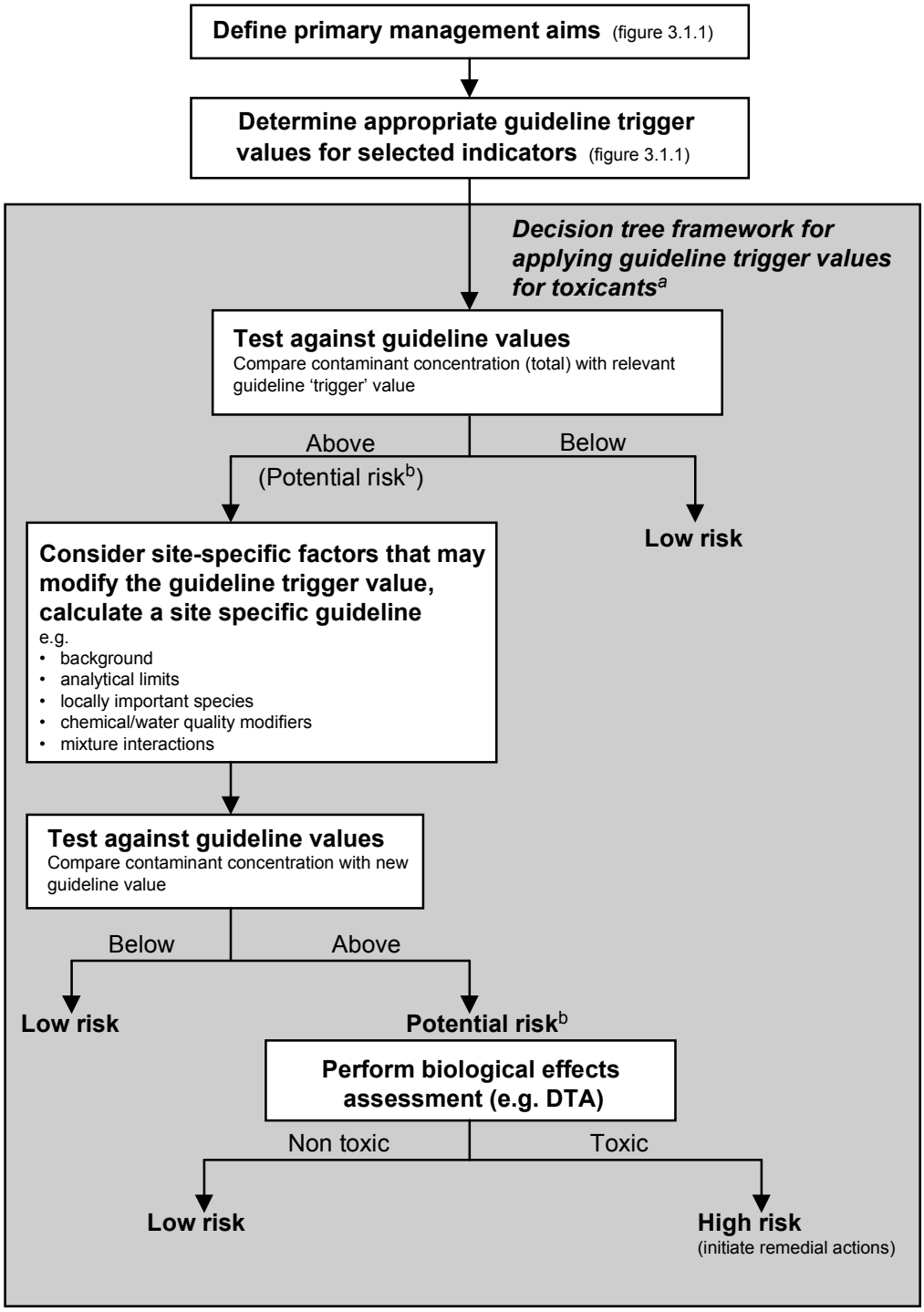
c Section 3.1.3

Ultimately, it is biological measurement that will provide confirmation of the site-specific guideline, so the decision scheme directs users to the option of direct toxicity assessment (DTA) if the guideline is exceeded or if there is low confidence in desktop assessments.[d] When no default trigger value is provided, where the trigger value is not applicable to a specific site, or if the chemical is one of a complex mixture, DTA is also useful. Further, DTA may provide the required link between chemical levels and biological effects or establish concentrations that are unlikely to cause adverse environmental effects. Field biological assessments can be undertaken also.[e]

d Section 8.3.6

e Section 3.2

The stepwise procedure for applying the decision scheme is outlined below. The cross-references to Volume 2 provide background information on each step. Site-specific trigger values can be derived at each step and compared with the concentration of chemical measured at the site. Note that at any stage the stakeholders may wish to accept the lowest original or partially modified trigger value and institute management actions to reduce contamination or pollution, if that value is exceeded. However, if a trigger value is accepted without completing the decision tree, the value may not be the most appropriate for the site.



[a] *Local biological effects data not required in the decision trees (see section 3.1.5)*
[b] *Further investigations are not mandatory; users may opt to proceed to management/remedial action.*

Figure 3.4.1 Simplified decision tree for assessing toxicants in ambient waters

Application of the decision tree

1. On advice of the water management authority, select the appropriate target *ecosystem condition* (Section 3.1.3) for the particular site or region under study.[a] This may determine which trigger value is used.[b] Alternative levels of protection are also given in table 3.4.1. The concept of three ecosystem conditions in Section 3.1.3 is for management guidance only. Users need to

a See Section 8.3.5.2

b Section 3.4.2.4

view these as examples that represent a continuum of ecosystem conditions. Table 3.4.2 summarises the approaches and default trigger values recommended for each ecosystem condition. For highly disturbed (condition 3) ecosystems, it may be appropriate to negotiate a lower level of protection for toxicants in some instances and hence to use a less stringent trigger value for ensuing calculations. Initial decisions are also made about whether the sample is freshwater or saline because different trigger values may apply, and whether the chemical is a metal, which may affect which of the following steps apply.

a See Section 3.4.3.3; see also Section 8.3.5.3 and the Monitoring Guidelines

2. Collect and analyse water samples. Design, implement and organise the logistics of sampling protocols, filter samples and mathematically process data.[a]

   Judgement on whether a chemical concentration exceeds a guideline value should not rely on results of analysis of a single sample, except possibly if the concentration is high enough to potentially cause acute toxicity. It is better to collect a number of samples and to compare the median value with the guideline value.

   Should the samples be filtered in the field? Samples do not normally need to be filtered unless the user is studying metals and considers that field filtration is cost-effective. Often, users will find it easier and most economical to compare total unfiltered concentrations initially. Comparison of total concentrations will, at best, overestimate the fraction that is bioavailable. The major toxic effect of metals comes from the dissolved fraction, so it is valid to filter samples (e.g. to 0.45 µm) and compare the filtered concentration against the trigger value. If other measurements of metal bioavailability are being pursued (e.g. step 10), filtration will be necessary but chemical preservation is not advised.

   There are few bioavailability measurements for organic chemicals and expert advice should be sought on the appropriateness of this step for organic chemicals.

b See the Monitoring Guidelines

   The present guidelines do not prescribe specific methods for chemical analyses.[b] Users must satisfy themselves that analysis methods are appropriate and sufficiently accurate, that the laboratories are suitably accredited and that quality control procedures have been adhered to.

   If users intend to follow this decision scheme, it will also be necessary to analyse for the water quality parameters that may affect the chemical toxicity and hence the site-specific trigger value. Measures of pH, organic carbon and hardness (e.g. for metals) will also assist some steps.

c Section 8.3.5.4

3. Consider the analytical practical quantitation limit (PQL)[14] using the best available technology.[c] If the PQL is *above* the trigger value (i.e. PQL >TV) there are three options, on advice of the appropriate state regulator:

   i) accept that any validated detection implies that guidelines have been exceeded; or

[14] The practical quantitation limit (PQL) is the lowest level achievable among laboratories within specified limits during routine laboratory operations. The PQL represents a practical and routinely achievable detection level with a relatively good certainty that any reported value is reliable (Clesceri et al. 1998). The PQL is often around 5 times the method detection limit.

ii) examine the decision scheme to see if site-specific factors reduce the environmental risk; or

iii) proceed directly to direct toxicity assessment (DTA) where one of the following two approaches can be adopted:

- site-specific toxicity testing of the toxicant in question, using local species under local conditions, to derive a site-specific trigger value (step 7). Note that some judgement is required (ideally, based on existing information) about whether adverse effects can be expected at concentrations below the PQL, in which case this option is not appropriate.
- DTA of the ambient water (step 12) to ascertain whether adverse effects are being observed at the present concentration of toxicant. If effects are observed, management action is initiated. This can include the use of toxicity identification and evaluation (TIE) techniques, which assist in identifying the unmeasured toxicant source (Burkhard & Ankley 1989, Manning et al. 1993).[a]

  Water regulators may also recommend DTA if the chemical cannot be measured and the issue is of high significance.

a See Section 8.3.6.3

4. Consider the natural *background* concentration (or range) of the toxicant at the site.[b] This applies mostly for metals and some non-metallic inorganics. The only organic chemicals to which this will commonly apply will be some phenols or globally distributed contaminants such as DDT. Table 8.3.2 (Volume 2) provides some general literature guidance on commonly encountered background levels. If background concentrations cannot be measured at the site, measurement at an equivalent high-quality reference site that is deemed to closely match the geology, natural water quality etc of the site(s) of interest is suggested.

   If the background concentration has been clearly established and it *exceeds* the trigger value (it is preferable to compare filtered background concentrations for metals), the 80th percentile of the background concentration can be accepted as the site-specific trigger value for ensuing steps.[c] In addition, users may apply DTA to background or reference waters (Step 12) using locally adapted species, to confirm that there is no toxicity. In the unlikely event that adverse effects are observed, management action must be initiated, and again this can include the use of TIE to begin to identify the compound(s) causing toxicity.

b See Sections 7.4.4.2, 8.3.5.5; table 8.3.2

c Section 7.4.4.2

5. Examine if *transient exposure* is relevant and if it can be incorporated into the decision scheme.[d] At present, there is little international guidance on how to incorporate brief exposures into guidelines, and it may not yet be possible to do this. A number of chemicals can cause delayed toxic effects after brief exposures, so it has been considered unwise to develop a second set of guideline numbers based on acute toxicity to account for brief exposures. Concentrations at which acute toxicity is likely to occur[e] may not necessarily bear any resemblance to the concentrations that should protect against transient exposure. New information about transient exposure, published in the peer-reviewed literature, may assist users to take transient exposure into account for some chemicals.

d Section 8.3.5.6

e Section 8.3.7

6. Determine if the chemical *bioaccumulates* in organisms and if it is likely to cause *secondary poisoning* (i.e. biomagnify).[a] For some chemicals (e.g. mercury and PCBs), this is the main issue of concern, rather than direct effects of toxicants.[b] Chemicals that have the potential to bioaccumulate and cause harm are identified by 'B' in table 3.4.1. Some metals, such as copper, can accumulate in shellfish without causing harm.

   The decision scheme provides the opportunity to examine whether the identified chemicals may actually be bioaccumulating at the study site. This can be validated by relating tissue residues in local organisms to chemical levels in water. If data are available, it may be possible to refine the trigger value to account for these phenomena.[c] Alternatively the Canadian approach (CCME 1997) can give guidance on what levels of chemicals in food may accumulate in water-associated wildlife.[d] Appendix 3, Method 1B(i) of Volume 2 may also provide some guidance here. If there are no local data for such chemicals to enable these approaches to be used, users are advised to apply the 99% protection level trigger values for ecosystems that could be classified as *slightly to moderately disturbed.* However, this derivation is precautionary, and is not directly related to bioconcentration effects.

7. Consider whether there are *locally important species* or genera, either ecologically or economically, which were not adequately evaluated in calculating the original default trigger value. It will be necessary to examine the original data set used to calculate the trigger value, available on the enclosed CD-Rom (under the title, *The ANZECC & ARMCANZ Water Quality Guideline Database for Toxicants*), insert any new and appropriate data and recalculate the trigger value by the same method as used originally.[e] If considering this step, seek expert advice. In most situations it is reasonable to accept the original suite of test species as an adequate surrogate for untested species in the environment but there may be specific instances where it is worthwhile to consider particular species. In some cases it may be valid to check whether the original trigger value has been calculated using species that are locally inappropriate *and* if these data can be substituted by new data from more appropriate species which have an equivalent role in the ecosystem. Data should only be deleted in *exceptional* circumstances. *It is important in all cases to maintain the integrity of the trigger values by adhering to the requirements for data quality and quantity. It is also important to ensure that a comprehensive overseas data set is not substituted by a native data set that does not cover the necessary breadth of taxa.*[f]

8. Consider whether *chemical or water quality parameters* at the site may increase or decrease the toxicity of the chemical and hence potentially alter the site-specific trigger value. This applies for organic or non-metallic inorganic chemicals, as the hardness calculations for metals[g] also cover all these parameters except temperature and dissolved oxygen.

   These parameters may include differences in the proprietary formulation of the chemical[h] and variations in water quality parameters[i] such as suspended matter, dissolved organic matter, salinity, pH, temperature, hardness and dissolved oxygen. Specific guidance on which parameters are known to affect toxicity of each chemical is given in Section 8.3.7. In some cases, there are simple numerical factors or algorithms linking the water quality parameter and the toxicity of the chemical. If so, this can be applied to the original data or to

a See Section 8.3.5.7

b Section 8.3.3.4

c Section 8.3.3.4

d Section 8.3.3.4

e Section 8.3.5.8

f Section 8.3.4.2

g Section 8.3.5.15

h Section 8.3.5.9

i Sections 8.3.5.10 to 8.3.5.17 for detail on respective parameters

the trigger value to derive a site-specific guideline that accounts for these parameters, as below (using temperature as an example). Thus:

- Check back to the original data and apply factors to convert all the data to a single (say) temperature that better represents the site. Re-calculate the site-specific guideline according to the method used to derive the original trigger value; or
- if all the original data have been calculated at a standard (say) temperature, apply the factor directly to the trigger value.

Remember that when the parameter *increases* toxicity, the factor is <1 and when it *decreases* toxicity, the factor is >1. Tables for temperature and/or pH conversions are available in Volume 2 for ammonia, cyanide and sulfide. If there is not, a simple quantitative relationship, seek expert advice. For instance, the equilibrium between many organic chemicals and suspended matter is poorly understood and requires well-designed studies, e.g. DTA (Step 12) under appropriate conditions. It may be possible to make a qualitative estimate of whether the parameters increase or decrease the risk.

9. For metals or metalloids in fresh waters (up to 2500 mgL^{-1} salinity), consider the effect of hardness, pH and alkalinity on toxicity and derive a hardness-modified trigger value (HMTV)[a] using the appropriate algorithm from table 3.4.3. Table 3.4.4 indicates how the trigger values vary with different ranges of hardness but extra care is needed for waters with hardness below 25 mgL^{-1} $CaCO_3$. If samples have been filtered, for comparison with the HMTV, this will also take into account suspended organic matter. The hardness algorithms (table 3.4.3) also account for pH. The recommended decision scheme for metals is illustrated in figure 3.4.2 but steps beyond the initial hardness adjustment are optional.

   a See Section 8.3.5.15

   If the total metal concentration in the unfiltered sample exceeds the HMTV, then users may choose one or more of four steps:

   (i) compare metal concentration with the HMTV after filtering the original un-acidified sample through a 0.45 µm membrane filter. An alternative is to proceed directly to measuring filtered concentrations instead of totals initially.

   (ii) proceed to more complex estimates of metal bioavailability (step 10) relating to the study site;

   (iii) accept that the guideline has been exceeded and institute management action;

   (iv) proceed to DTA (step 12).

10. Examine the concentration of the metal or metalloid to determine the concentration of the bioavailable species, i.e. the concentration that is most likely to exert a biological effect. This uses speciation modelling or chemical techniques for metal speciation analysis[b] to account for the effects of factors such as dissolved organic matter, pH and redox potential on the bioavailable fraction of the metal. Seek professional advice for this step.

    b Section 8.3.5.16

    If the bioavailable metal concentration exceeds the HMTV or the trigger value (if a hardness algorithm is not available), consider these two options, with guidance from the regulatory authority:

    - use direct toxicity assessment (DTA) to confirm the results or develop a new site-specific guideline; or
    - develop management options to reduce contamination.

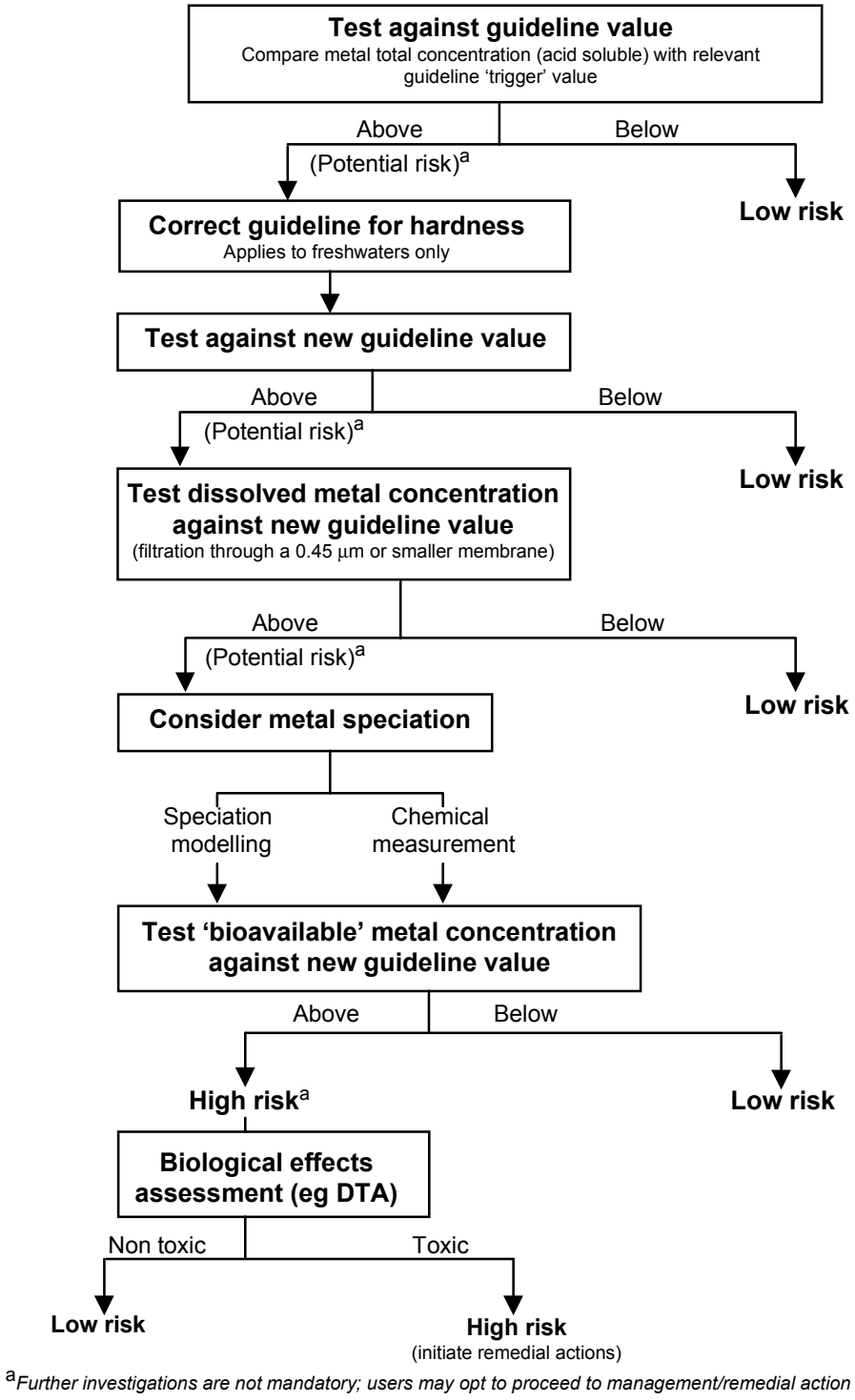


Figure 3.4.2 Decision tree for metal speciation guidelines

11. Consider the effect of *mixtures* and chemical interactions on overall toxicity.[a] If the chemical occurs as a component of a simple mixture, and the mixture interactions are simple and predictable (i.e. usually two–three components and additive toxicity) the mixture toxicity can be modelled using the mixtures equation in Section 8.3.5.18.

a See Section 8.3.5.18

12. If there is any degree of *complexity* in the mixture interactions, proceed to direct toxicity assessment (DTA) on the ambient waters at the site.[b] Use an appropriate battery of test species and chronic end-points to ascertain whether toxicity is being observed. If adverse effects are observed, initiate management action and use TIE to assist in identifying the compound(s) that are causing toxicity. Use DTA also to assess toxicity of ambient waters when background levels are high (step 3), when guideline values are lower than analytical PQLs (step 4), or to quantify the effects of water quality parameters or proprietary formulations on the chemical toxicity (step 8).

b Section 8.3.5.19

Where a chemical is to be used in an environment of particular socio-political or ecological importance, it is better to undertake toxicity testing with that chemical on species relevant to that environment (i.e. step 7). It is best to do this before the chemical is introduced. Such data can be used to develop new guideline values relevant to that region; for example, to collect a suite of tropical data for a development affecting tropical freshwaters.

When using DTA to examine toxicity of a chemical to locally important species (step 7) or for pre-release effluents (see table 3.4.2), determine chronic effects at a range of concentrations of the chemical or effluent. For dilution, use the local reference dilution waters. Determine NOEC values for the chemical or effluent and use them for calculating site-specific guidelines. The method used for these calculations will depend on the number of data points, but use the statistical distribution method if the data requirements have been met (at least five species from four different taxonomic groups).[c] Otherwise it is best to divide the lowest chronic NOEC by 10. Follow the general methods for calculation of trigger values.[d]

c Section 8.3.4.2
d Section 8.3.4.4

The DTA can comprise *in situ* field and/or laboratory ecotoxicity tests (Chapman 1995), preferably chronic or sub-chronic tests on appropriate species using local dilution waters, satisfying all sampling, test and analysis conditions.[e]

e Section 8.3.6

To aid interpretation of results, analyse the chemicals concurrently with biological assessment, unless there is a biological marker of toxicity.

For already existing discharges and for chemicals that have a high potential to disturb the environment, it will be necessary to measure and assess the biological health of potentially disturbed sites.[f]

f Section 3.2

Table 3.4.3 General form of the hardness-dependent algorithms describing guideline values for selected metals in freshwaters

Metal	Hardness-dependent algorithm
Cadmium	$HMTV = TV\ (H/30)^{0.89}$
Chromium(III)	$HMTV = TV\ (H/30)^{0.82}$
Copper	$HMTV = TV(H/30)^{0.85}$
Lead	$HMTV = TV(H/30)^{1.27}$
Nickel	$HMTV = TV(H/30)^{0.85}$
Zinc	$HMTV = TV(H/30)^{0.85}$

HMTV, hardness-modified trigger value (µg/L); TV, trigger value (µg/L) at a hardness of 30 mg/L as $CaCO_3$; H, measured hardness (mg/L as $CaCO_3$) of a fresh surface water (≤2.5‰). From Markich et al (in press).

Table 3.4.4 Approximate factors to apply to soft water trigger values for selected metals in freshwaters of varying water hardness[a]

Hardness category[b] (mg/L as $CaCO_3$)	Water hardness[c] (mg/L as $CaCO_3$)	Cd	Cr(III)	Cu	Pb	Ni	Zn
Soft (0–59)	30	TV	TV	TV	TV	TV	TV
Moderate (60–119)	90	X 2.7	X 2.5	X 2.5	X 4.0	X 2.5	X 2.5
Hard (120–179)	150	X 4.2	X 3.7	X 3.9	X 7.6	X 3.9	X 3.9
Very hard (180–240)	210	X 5.7	X 4.9	X 5.2	X 11.8	X 5.2	X 5.2
Extremely hard (400)	400	X 10.0	X 8.4	X 9.0	X 26.7	X 9.0	X 9.0

a Trigger values from table 3.4.1;

b Range of water hardness (mg/L as $CaCO_3$) for each category as defined by CCREM (1987);

c Mid-range value of each water hardness category. For example, a copper trigger value of 1.4 µg/L (from table 3.4.1) with 95% protection level chosen (e.g. slightly–moderately disturbed system) is applied to a site with very hard water (e.g. 210 mg/L as $CaCO_3$) by multiplying the trigger value by 5.2 to give a site-specific trigger value of 7.3 µg/L. If the hardness is away from the mid-range, it may be preferable to use the algorithm.

3.4.3.3 Comparing monitoring data with trigger values

Wherever there is concern about toxicants in a waterbody, data must be gathered to see if there are accompanying adverse ecological effects. This process has many steps, and it is beyond the scope of these Guidelines to address all of them in detail. Those which are not elaborated in Chapter 7 of this volume are discussed in detail in the Monitoring Guidelines (ANZECC & ARMCANZ 2000). The purpose of this section is to direct readers to the appropriate places to learn more about the necessary procedures for a chemical monitoring program.

- *The design of sampling protocols*. The Monitoring Guidelines (Chapter 3) advises on: study type, temporal and spatial considerations, site selection and identification, sampling precision, timing and frequency, and considerations for selecting indicators (measurement parameters).

- *The implementation of sampling protocols.* Chapter 4 of the Monitoring Guidelines discusses procedural issues in sample acquisition. Specifically it addresses ways for ensuring that samples are sufficiently numerous, well-documented and representative, and with appropriate analytical integrity, to enable strong inferences to be made about water quality. It also offers advice on logistical issues and OH&S considerations. Specific topics include: the mechanics of sampling; maintenance of sample integrity; field QA and QC; and OH&S requirements.

- *The elucidation of the 'biologically-relevant' (usually bioavailable) fraction.* Chapter 7 of these Guidelines provides some information on this topic. Chapter 4 of the Monitoring Guidelines makes recommendations about sample filtration, but mainly from the perspective of sample preservation. Section 7.4.2 of the present Guidelines discusses filtration with an emphasis on speciation considerations. That section also describes other steps in calculating the relevant indicator concentration, such as thermodynamic modelling, while section 8.3.5 describes the application of algorithms designed to account for the modifying effect of indicators such as water hardness.
- *The mathematical (including statistical) processing of raw or speciation-adjusted data*. Chapter 6 of the Monitoring Guidelines offers a detailed and very useful primer on data management and interpretation, including summary statistics, methods of inference, multivariate analysis, power analysis, regression techniques, trend analysis, and non-parametric statistics. It also contains useful discussions on water quality modelling, outlier detection and the treatment of data below the analytical detection limit.
- *The comparison of test data with background data and default trigger values.* Whether or not a study area has adequate water quality is decided by comparing monitoring data with a guideline recommendation.[a] This assessment of whether the guideline has been exceeded is embodied in the concept of an 'attainment benchmark'. The default trigger value can be structured as a comparison between reference (or background) and test-site data or as a comparison with a single default trigger value. Statistical decision criteria can be used to compare test data with background data or default trigger values.[b] In general, the greater the amount of reference data (if applicable) and test data gathered, the smaller will be the error rates associated with detecting change in toxicant concentrations in the field. Wherever maintenance of biological diversity is a key management goal — e.g. sites of high conservation value (condition 1) or slightly disturbed systems (condition 2), statistical decision criteria should be set as conservatively as possible. Values of the criteria as recommended for biological indicators might be used as a starting point in negotiations.[c]

a See Section 7.4.4.2

b Section 3.1.7 (statistical decision criteria); section 7.4.4.2 (default trigger values); Section 7.4.4.2 (detecting change in toxicant concentrations in the field); See also the Monitoring Guidelines Chapter 6.

c Section 3.2.4.2

3.5 Sediment quality guidelines

3.5.1 Introduction

The *Australian Water Quality Guidelines for Fresh and Marine Waters* (ANZECC 1992) provided a framework for managing receiving water quality. Those Guidelines recognised that total load and fate of contaminants, particularly to enclosed systems, should also be considered. Sediments are important, both as a source and as a sink of dissolved contaminants, as has been recognised for some time. As well as influencing surface water quality, sediments represent a source of bioavailable contaminants to benthic biota and hence potentially to the aquatic food chain. Therefore it is desirable to define situations in which contaminants associated with sediments represent a likely threat to ecosystem health. While costly remediation or restoration might not represent a management option, sediment guidelines can usefully serve to identify uncontaminated sites that are worthy of protection. Sediment quality guidelines are being actively considered by regulatory agencies worldwide.

This section reviews the current state of knowledge on environmental effects of contaminants in sediments, and the approaches being used to formulate sediment quality guidelines. On the basis of these, it outlines a procedure for the development of appropriate sediment quality guidelines for Australia and New Zealand. The guidelines would apply to slightly to moderately disturbed (condition 2) and highly disturbed (condition 3) aquatic ecosystems.[a] Consideration of sediment quality follows the decision-tree approach being adopted in these Guidelines, with a focus on identifying the issues and the protection necessary to manage them.

a See Section 3.1.3

For aquatic ecosystems considered to be of high conservation/ecological value (condition 1) a precautionary approach is recommended. In these ecosystems, chemicals originating from human activities should be undetectable, and naturally occurring toxicants (e.g. metals) should not exceed background sediment concentrations.[b] This approach should only be relaxed when there are considerable biological assessment data showing that such a change in sediment quality would not disturb the biological diversity of the ecosystem.

b Section 3.1.3.2

3.5.2 Underlying philosophy of sediment guidelines

It is important to understand why sediment guidelines are being developed and how and where they might be applied. The establishment of guidelines will serve three principal purposes:

- to identify sediments where contaminant concentrations are likely to result in adverse effects on sediment ecological health;
- to facilitate decisions about the potential remobilisation of contaminants into the water column and/or into aquatic food chains;
- to identify and enable protection of uncontaminated sediments.

Many urban and harbour sediments fall into the first category, usually being contaminated by heavy metals and hydrophobic organic compounds resulting from both diffuse and point-source inputs. They are not easily remediated. At present,

ex situ treatment or dredging and disposal are the most cost-effective options. If a site is known to have highly contaminated sediments with potential for biological uptake, it may be possible to control the collection of benthic organisms for human consumption. For the most part, because of the enormous costs involved, there is unlikely to be large-scale sediment remediation, unless it is driven by human health risk assessments. Contaminated sediments can be remediated naturally when fresh sediments, able to support viable biological populations, settle on top of them. This can occur through water column inputs and can be managed through controls on inputs via water quality guidelines. Management conflicts can arise when natural sediment accumulation restricts navigation.

It is possible to adopt measures to protect unmodified areas from further contamination by managing inputs. This is where the application of sediment quality guidelines will be of greatest value. Just as for water quality guidelines, the application of sediment guidelines will involve a decision-tree approach. It is important to reiterate that the guidelines should not be used on a pass or fail basis.

The guideline numbers are trigger values that, if exceeded, prompt further action as defined by the decision tree. The first-level screening compares the trigger value with the measured value for the total contaminant concentration in the sediment. If the trigger value is exceeded, then this triggers either management/remedial action or further investigation to consider the fraction of the contaminant that is bioavailable or can be transformed and mobilised in a bioavailable form.

In the case of metals, the dilute-acid-soluble metal concentration is likely to be a more meaningful measure than the total value. The derivation of future trigger values might ultimately be based on this measurement. Non-available forms will include mineralised contaminants that require strong acid dissolution. For metals that form insoluble sulfides, the role of amorphous iron sulfide (FeS), measured as so-called acid volatile sulfides (AVS), can be an important factor in reducing metal bioavailability. This exchangeable sulfide is able to bind released metals in non-bioavailable forms. Changes in redox potential and pH also affect the availability of metals and other contaminants, and should be considered.

It is important to consider both sediment pore waters and the sediment particles as sources of contaminants. The importance of these sources varies for various classes of sediment dwelling organisms, as discussed elsewhere.[a]

a See Section 8.4.3.2

3.5.3 Approach and methodology used in trigger value derivation

The many approaches adopted internationally to derive sediment quality guidelines are more fully described in Section 8.4 (Volume 2). By far the most widely used method is an effects database for contaminated and uncontaminated sites, based on or derived from field data, laboratory toxicity testing and predictions based on equilibrium partitioning of contaminants between sediment and pore water. There are few reliable data on sediment toxicity for either Australian or New Zealand samples from which independent sediment quality guidelines might be derived, and without a financial impetus there is little likelihood that further data will be forthcoming in the immediate future. Because of this, and as has been done in many other countries, the option selected for the sediment quality guidelines is to use the best available overseas data and refine these on the basis of our knowledge of existing baseline concentrations, as well as by using local effects data as they become available.

The recommended guideline values are tabulated as interim sediment quality guideline (ISQG) values (table 3.5.1), and the low and high values correspond to the effects range-low and -median used in the NOAA listing (Long et al. 1995).

3.5.4 Recommended guideline values

3.5.4.1 Metals, metalloids, organometallic and organic compounds

The recommended guideline values for a range of metals, metalloids, organometallic and organic sediment contaminants are listed in table 3.5.1.[a] Values are expressed as concentrations on a dry weight basis. This does not imply that samples should be dried before analysis, resulting in potential losses of some analytes, but that results should be corrected for moisture content. For organic compounds, values are normalised to 1% organic carbon, rather than being expressed as mg/kg organic carbon as is sometimes done. If the sediment organic carbon content is markedly higher than 1%, the guideline value should be relaxed (i.e. made less stringent), because additional carbon binding sites reduce the contaminant bioavailability.

a See Section 8.4.3

The issue of uncertainties is often overlooked and is worth re-emphasising. The database underpinning the guidelines (Long et al. 1995) was originally designed to rank sediments. The values represent a statistical probability of effects (10% or 50%) when tested against only one or two species, principally amphipods. This is not analogous to the Aldenberg and Slob (1993) approach to water quality guidelines that are protective of 95% of the species, based on tests on a large range of aquatic species of varying sensitivities. Note that some tests use sea urchin fertilisation, while for organic compounds the tests apply Microtox® luminescent bacteria to solvent extracts of sediments. The ecological relevance of these is questionable.

There are added uncertainties about how well the effects of multiple toxicants have been dealt with. The data do not consider antagonism or synergism between chemicals, and, as originally derived, they are based only on disturbances to biological receptors and do not relate to human health disturbances.

3.5.4.2 Ammonia, sulfide, nutrients and other sediment contaminants

No specific guideline values are provided in any of the overseas databases for ammonia or nutrients such as phosphate and nitrate, yet it is important to identify when these represent a threat to benthic communities.

The major disturbance of ammonia will be seen in pore waters, and it is best that these be sampled and the measured ammonia concentrations compared against water quality guidelines.[b]

b Section 8.4

The biological effects of sulfide in sediments are poorly understood. The decision tree acknowledges the role of sulfide in reducing metal toxicity, but sulfide can affect animal behaviour which in turn can alter the toxicity of both sulfide and also other sediment contaminants (Wang & Chapman 1999). Both sulfide and ammonia can potentially be released in any sediment studies. This may require the refining of appropriate TIE protocols for use with sediments.

Table 3.5.1 Recommended sediment quality guidelines[a]

Contaminant	ISQG-Low (Trigger value)	ISQG-High
METALS (mg/kg dry wt)		
Antimony	2	25
Cadmium	1.5	10
Chromium	80	370
Copper	65	270
Lead	50	220
Mercury	0.15	1
Nickel	21	52
Silver	1	3.7
Zinc	200	410
METALLOIDS (mg/kg dry wt)		
Arsenic	20	70
ORGANOMETALLICS		
Tributyltin (µg Sn/kg dry wt.)	5	70
ORGANICS (µg/kg dry wt) [b]		
Acenaphthene	16	500
Acenaphthalene	44	640
Anthracene	85	1100
Fluorene	19	540
Naphthalene	160	2100
Phenanthrene	240	1500
Low Molecular Weight PAHs [c]	552	3160
Benzo(a)anthracene	261	1600
Benzo(a)pyrene	430	1600
Dibenzo(a,h)anthracene	63	260
Chrysene	384	2800
Fluoranthene	600	5100
Pyrene	665	2600
High Molecular Weight PAHs [c]	1700	9600
Total PAHs	4000	45000
Total DDT	1.6	46
p.p'-DDE	2.2	27
o,p'- + p,p'-DDD	2	20
Chlordane	0.5	6
Dieldrin	0.02	8
Endrin	0.02	8
Lindane	0.32	1
Total PCBs	23	–

a Primarily adapted from Long et al. (1995);

b Normalised to 1% organic carbon;

c Low molecular weight PAHs are the sum of concentrations of acenaphthene, acenaphthalene, anthracene, fluorene, 2-methylnaphthalene, naphthalene and phenanthrene; high molecular weight PAHs are the sum of concentrations of benzo(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(a,h)anthracene, fluoranthene and pyrene.

For nutrients, the need to define sediment guidelines is debatable. In this case, the disturbance that we are seeking to protect against is algal or macrophyte blooms, whereas the proposed guidelines address biological disturbances, based in part on equilibrium partitioning to sediment pore waters and ultimately the water column. It should theoretically be possible to derive a guideline value based on the undesirable release of nutrients to the water column and their subsequent undesirable ecosystem disturbances. This would require some measure or prediction of pore water nitrogen and phosphorus and a judgement as to what concentration of bioavailable nutrient constitutes a threat, logically based on water quality guidelines.

There are methods that purport to measure bioavailable phosphorus, for example bioassays or the use of iron strips, but there are factors such as redox potential that will be important in defining this. Indeed, control of bioavailable carbon inputs is more important than the concentration of phosphorus itself. The application of water quality guidelines to pore waters is possible, although prior use of the nutrients by benthic organisms may have already reduced the pore water concentrations. It is generally thought that development of nutrient guidelines is too difficult at this stage, and must await further research developments.

3.5.4.3 Absence of guidelines

In some instances, no guidelines will be specified for a contaminant of interest. This generally reflects an absence of an adequate data set for that contaminant. An interim approach is required to provide some guidance as well as to ensure environmental protection in situations where guidelines would apply. The approach suggested is to derive a value on the basis of natural background (reference) concentration multiplied by an appropriate factor. A factor of two is recommended, although in some highly disturbed ecosystems a slightly larger factor may be more appropriate, but no larger than three. An alternative approach is to apply the water quality guideline values to sediment pore waters.

3.5.5 Applying the sediment quality guidelines

A protocol is provided to summarise key aspects of collection and laboratory analysis of sediment samples [a] while the Monitoring Guidelines provide full details.

a See App. 8, Volume 2

3.5.5.1 Sediment sampling

The use of appropriate sampling techniques is a prerequisite for chemical or toxicity testing of sediments or sediment pore waters. The depth of sampling will be dictated by the issue being investigated, and this in turn will determine whether corers or grab sampling is preferable. Full details on sampling methodology are provided in the Monitoring Guidelines.

3.5.5.2 Applications of chemical testing

It is important to recognise the limitations applicable to the guideline values in table 3.5.1 as discussed above. They nevertheless form a good basis for sediment quality assessment, if applied using a decision tree approach as illustrated in figure 3.5.1.

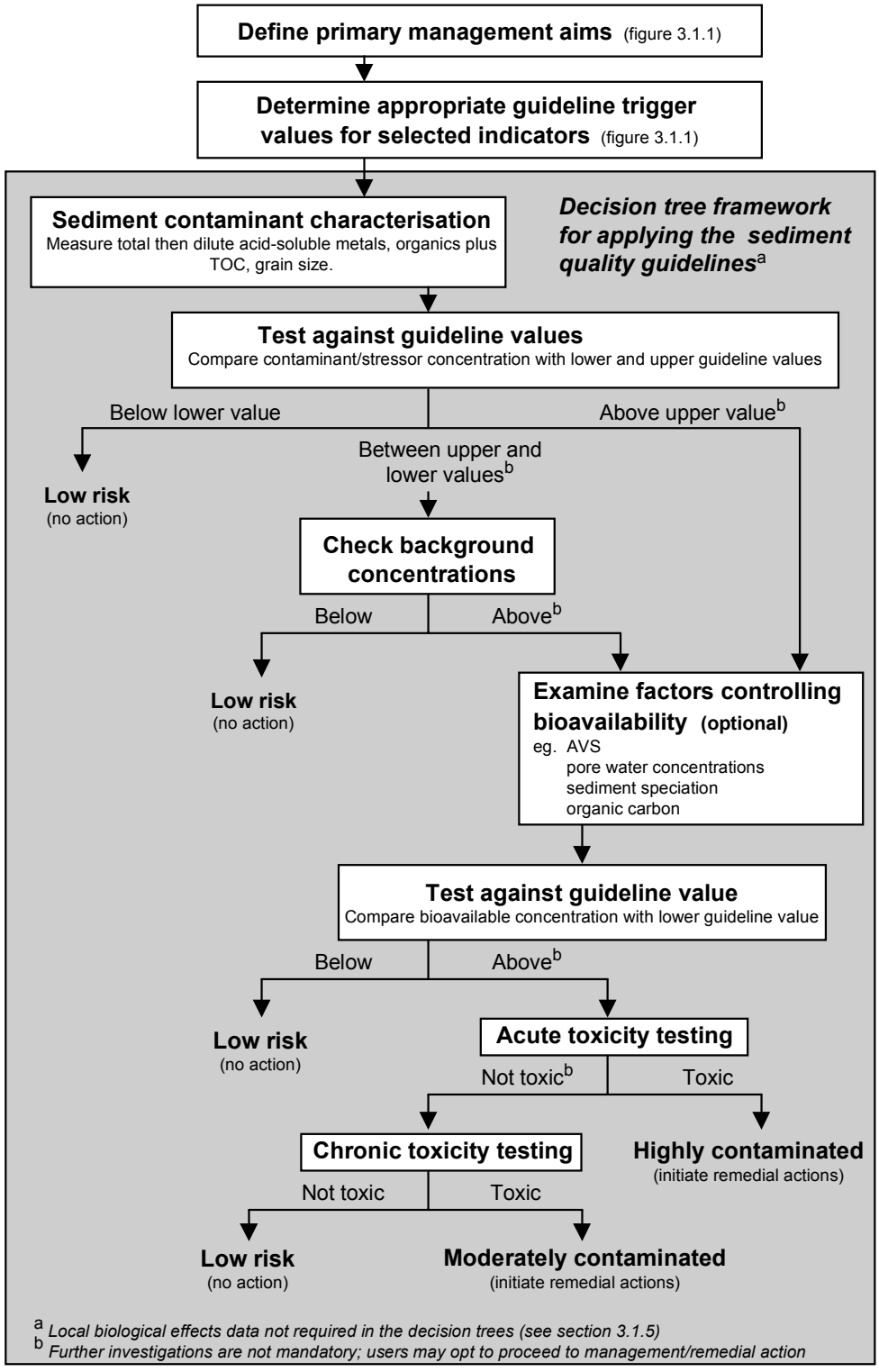


Figure 3.5.1 Decision tree for the assessment of contaminated sediments

a Section 3.1.5

The general approach to use of the decision scheme is outlined in Section 3.1.5.[a] If the lower sediment quality guideline, the trigger value, for a particular contaminant is not exceeded, it is unlikely that it will result in any biological disturbance for organisms inhabiting that sediment. If the trigger value is exceeded, either management (including remedial) action is taken, or additional site-specific studies are conducted to determine whether this exceedance poses a risk to the ecosystem.

Should a 'low risk' outcome result after continuous monitoring, there is scope to refine the guideline trigger value. Note that in the consideration of guideline values for metals, total metals concentrations are used, however, acid-soluble metals, are more representative of a bioavailable fraction and it is envisaged that ultimately trigger value compliance will be based on this measurement, as discussed later.

Comparison with background concentrations

The next step in the decision tree involves a comparison with background concentrations. Exceedance of a trigger value is acceptable if it is at or below the normal background concentration for a site. The selection of background or reference no-effects sites should, where possible, use sediments of comparable grain sizes. Similarly, the analysis of sediment cores must ensure that fluctuations in contaminant concentrations with depth are not the result of grain size changes, or in the case of organics, to changes in the organic carbon content.

For metals, a reliable determination of 'natural' levels of contaminants is best done on the basis of trace element ratios determined for a range of uncontaminated sites. Usually the contaminant element is referred to naturally occurring elements such as lithium, iron or aluminium (e.g. Loring & Rantala 1992).

The theoretical background concentration of most synthetic organic compounds is zero, but from a practical viewpoint, ubiquitous contamination has occurred far from point sources. Reference sites removed from such sources are appropriate for determining background concentrations.

Consideration of factors controlling bioavailability

If both the lower guideline trigger value and the background or reference site concentrations are exceeded, the next level evaluation will be to consider whether there are any factors which might lower the potential bioavailability of contaminants. The methods of sampling of sediments and sediment pore waters will be critical if meaningful data (especially for metals) are to be obtained, to ensure that the natural chemical conditions, especially redox conditions, salinity and pH, are not altered. If such changes are allowed to occur, erroneous analytical data on contaminant bioavailability may be obtained.[a]

a See Section 4.3.5 of the Monitoring Guidelines

For metals, the speciation considerations might be:[b]

b See discussion in Section 8.4, Vol. 2

a) *Sediment speciation* — dilute-acid-extractable metals concentrations below lower guideline value. It is recommended that this should involve treatment of the sample with 1 M hydrochloric acid for 1 hour (Allen 1993).

   Since a considerable fraction of the total metal concentration in sediments may be present in detrital mineralised phases that are not bioavailable, a better estimate of the bioavailable fraction is desirable. Although the capacity of chemical extractions to selectively remove only this fraction is limited, a dilute-acid-extraction will not remove the mineralised fractions and will therefore provide more appropriate metal concentration data for use in new effects databases. During extraction of carbonate- or sulfide-containing sediments, allowance must be made for acid consumed by reaction with these phases.

   Note that, except for spiked sediment toxicity tests where ionic metal additions are made, the field data used to derive the guidelines are likely to be based on total concentrations. Therefore a judgement against these measurements using

speciation cannot be fully justified. Rather, such considerations should be applied in new guideline values developed from an NWQMS database.

b) *Acid volatile sulfides, AVS*: Σ_i [SEM] < [AVS]

If the concentration of acid volatile sulfide (AVS), released by dilute acid treatment of the moist sediment, exceeds the sum of the heavy metal concentrations released by the same treatment (referred to as simultaneously extracted metals (SEM)), then this excess sulfide is able to bind heavy metals in insoluble and non-bioavailable forms, and therefore the metals will not cause toxicity.[a] This applies particularly to lead, zinc and cadmium. Its application to copper, nickel and possibly cobalt is suspect.

a See Section 8.4.3.2, Vol. 2

Recent reports urge caution in the application of the AVS binding model, particularly because of concern for its relevance in longer-term and community level effects (IMO 1997). Other limitations are discussed in Section 8.4. A description of the methods for measuring AVS and SEM may be found in Allen et al. (1992).

c) *Pore water*: Σ_i $[M_{i,d}]/[WQG_{i,d}]$ <1, where $[M_{i,d}]$ is the total dissolved pore water concentration for each metal and $[WQG_{i,d}]$ is the water quality guideline value for each metal.

Assuming that pore water represents the major exposure route to sediment toxicants, then if pore water concentrations for any metal are below the water quality guideline concentration, there is unlikely to be an adverse biological disturbance. The correct methods should be used for sampling pore waters, to avoid losses or changes in redox status. Note that there is the possibility of seasonal variations in pore water contaminant concentrations as well as in AVS.

For organic compounds, the use of guidelines normalised to total organic carbon (TOC) is a first stage. The effects of natural sediment and water chemistry on the equilibrium partitioning of the particular organic compounds are moderating factors requiring consideration. This may mean separate measurements of the partitioning into natural waters of appropriate salinity or the measurement of pore water concentrations. Analytical detection with the small volumes generally encountered creates problems, so this is often a difficult area. Such considerations as rates of degradation, either chemical, physical or biological, can be important for hydrophilic and for some hydrophobic organics.

If on the basis of any of the above considerations the trigger value is still exceeded, and further investigation is sought rather than management/remedial action, toxicity tests will be required. The tests will further characterise the nature of sediment as either moderately or highly contaminated. Alternatively, toxicity testing might be employed in lieu of more detailed chemical investigations when the trigger value is exceeded.

The guidelines discussed above have been derived on the basis of the toxicity of contaminants in sediments and associated pore waters, to benthic biota. An additional factor that needs to be taken into consideration, especially for riverine sediments, is mobility. Dynamic zones can be created in rivers during periods of high flow that lead to erosion and sediment mobilisation. Finer, contaminant-rich particles will be the most mobile, although larger particles will also be moved in storm flows. Two considerations arise under these conditions.

First there is the concern for enhanced contaminant release, either resulting from the disturbance of surface sediments and pore waters, or as a consequence of chemical transformations, such as oxidation of previously anoxic sediments. The former is not important, since pore water concentrations will be diluted. The possibility of oxidative release especially of metals is more a concern. In this case the kinetics of oxidation of metal sulfides is important. Elutriate tests with overlying saline or freshwaters can be used to demonstrate a worst case release scenario.

Secondly there is the possibility that the deposition process will lead to particle sorting, and if this were to result in a greater concentration of clay/silt particles at a particular site, there is a real possibility that in some cases the guideline concentrations for the whole sediment could now be exceeded because of removal of the diluent effect of coarser particles. If sorting is believed to be a possibility, it would be appropriate to assess the sediment on the basis of analyses on the <63 μm size fraction only.

In the absence of sediment guideline values for a particular contaminant, the first recourse is to the water quality guideline values. Sampling and analysis of sediment pore water can be undertaken, and water quality values can be employed to judge its acceptability. Care must be taken that the chemistry of the pore waters is not altered during the sampling process. This means squeezing, or centrifuging the sediment under nitrogen to minimise oxidation. Often it is very difficult to obtain sufficient sample to undertake a pore water analysis, especially for organic contaminants. In these cases, toxicity testing of the sediment or pore water is the only option.

In relation to water quality, different levels of protection have been considered for particular ecosystem conditions (namely high conservation value, slightly to moderately disturbed and highly disturbed). It is not appropriate at this stage to provide guidelines for different levels of protection for sediments, until more data are available. The provision of low and high guideline values, in combination with the decision-tree approach, should nevertheless provide useful guidance about the potential ecological effects of sediment contaminants that can guide management actions, as indicated in table 3.1.2.

Application of toxicity testing

The decision-tree allows for toxicity testing as the ultimate means of assessing sediment quality. Although this is shown at the bottom of the tree, mainly on the basis of its greater cost compared to chemical analyses, it may be applied at any stage. Appropriate methods may include examining the water extractable contaminants (elutriate testing), pore water testing, or whole sediment bioassays. Whole sediment testing with infaunal species has the greatest ecological relevance. Marine and freshwater testing with amphipods have been most widely used, although tests using midge larvae, insects and worms have been reported.[a]

a See Section 8.4; also Method 2A (App. 3, Vol. 2), table 3.2.2

As with chemical testing, is important that the sample used for toxicity testing has the same chemistry as it did in the field situation. Oxidation of sediments during manipulations may significantly alter metal bioavailability.

Normally toxicity testing will be used to demonstrate the absence of toxicity when the guideline for a particular contaminant is exceeded. If toxicity is observed, its origins cannot necessarily be attributed to the contaminant of interest, because of

the possibility of other contaminants either contributing to the observed toxicity or being the primary cause. Under these conditions, it will be necessary to apply TIE procedures (USEPA 1991) which successively separate classes of contaminants and identify any toxicity that they may have caused. Despite a large number of applications of the TIE approach, it is most often ammonia or common pesticides that have been found to be the source of toxicity.

4 Primary industries

4.1 Introduction

Both the quality and the quantity of water resources are critical issues for agriculture and aquaculture in Australia and New Zealand. Water quality is also of major importance for the protection of human consumers of food products. Growth of these major primary industries, together with expanding urbanisation and other industrial development, has increased the demand for good quality water but at the same time exerted escalating pressures on the quality of the water resources that are available. Therefore, to assess water quality for primary industries, not only must productivity issues be considered but also the possible adverse effects of these enterprises on downstream water quality and activities.

In recent years it has been recognised that pollution-related issues should be addressed by approaching the conservation, management and use of water resources in a holistic manner, according to the principles of integrated catchment management. Key strategies for achieving ecologically sustainable development include the involvement of stakeholders in decision-making processes and the development and adoption by industry of best management practice guidelines.

This is the first occasion on which water quality guidelines have been provided for aquaculture industries in Australia and New Zealand. Most of the guidelines presented for aquaculture should be used with some caution because few are based on a critical assessment of a wide data set.[a] This chapter also discusses issues concerning water quality guidelines for the protection of human consumers of aquatic foods. Recreational and commercial fisheries are based on wild populations of fish, crustacea and shellfish species, which are supported by natural habitats and food webs. Accordingly, for the protection of wild animal stocks, the reader is referred to the water quality guidelines for the protection and maintenance of aquatic ecosystems (Chapter 3).

a See Sections 4.4 and 9.4.4

Irrigation and livestock watering are the major agricultural uses of water. Minor amounts are used for other production purposes, such as the mixing of pesticide, fertiliser and veterinary formulations, and livestock dietary supplements. In Australia particularly, both the irrigation and livestock industries rely heavily on the use of groundwater, as well as surface water resources. Groundwater is also an important source of stock water in parts of New Zealand. Thus the guidelines provided for these industries are applicable (where appropriate) to both surface and groundwater quality.

Guidelines for general on-farm water use are included with the irrigation guidelines and cover topics such as corrosion and fouling of pipes and fittings. Certain issues concerning water quality for use by agriculture are also discussed in other documents published in conjunction with the National Water Quality Management Strategy; for example, the *Guidelines for Sewerage Systems — Use of Reclaimed Water* (ARMCANZ, ANZECC & NHMRC 2000). Note, however, that occasional discrepancies may occur in the information provided by different NWQMS documents; for example, when revision of the documents is out of step. All the

guideline documents are based on the best scientific information available at the time of publication.

For information on the quality of farmstead water supplies for domestic use in Australia, the reader is referred to Chapter 6 of these Guidelines and Section 7.7 of the *Australian Drinking Water Guidelines* (NHMRC & ARMCANZ 1996). Readers in New Zealand are referred to the *Drinking-water Standards for New Zealand* (New Zealand Ministry of Health 1995a) and the *Guidelines for Drinking-water Quality Management* (New Zealand Ministry of Health 1995b). Issues such as water quality for washing of farm produce or for dairy water supplies are outside the scope of the present guidelines and the reader is referred to local health and hygiene regulations and the proposed food safety standards of the Australian and New Zealand Food Authority.

An important first step in using these guidelines is to consider the management framework for their application. This includes defining the primary management aims, determining appropriate trigger values, defining water quality objectives, and establishing a monitoring and assessment program to address these objectives.[a]

a See Section 2.1

The type of monitoring and assessment program required will be specific to each situation, but there are several broad principles or procedures that are common to all programs. For details see Chapter 7, particularly noting figure 7.1 which gives a generic flow chart of the procedural framework for monitoring and assessment, and Section 7.4 which discusses specific issues for physical and chemical indicators.

4.2 Water quality for irrigation and general water use

Agricultural practice in Australia and New Zealand is often dependent on irrigation, because of climatic constraints on crop demand. In Australia particularly, agriculture is predominantly based in areas of limited rainfall, and there is heavy reliance on the use of surface and groundwaters for irrigation of crops and pastures. Approximately 70% (nearly 12 000 giga-litres) of Australia's developed water is used for irrigation, 21% for urban or industrial purposes and 9% for rural water supply (DEST State of the Environment Advisory Council 1996). Irrigated agriculture contributes very significantly to the Australian economy, with an annual production value of commodities such as cotton, rice, cereals, sugar, horticulture and irrigated fodder, of over $7 billion (Cape 1997).

In New Zealand irrigation is playing an increasingly important role in agricultural production. The area of irrigated land is doubling approximately every 10 years. Around 80% of allocated water in New Zealand is used for irrigation, with the remaining 20% for urban and industrial uses. Irrigated agriculture makes a significant contribution to the New Zealand economy, with irrigation being worth an extra $800 million 'at the farm gate' and possibly three times this in export earnings.

An important goal of these Water Quality Guidelines is to maintain the productivity of irrigated agricultural land and associated water resources, in accordance with the principles of ecologically sustainable development and integrated catchment management.[a] This should be a key consideration in any irrigation strategy, alongside maximum yield and economic viability.

a See Section 4.2.1

4.2.1 Philosophy

In developing the guidelines, emphasis has been placed on sustainability in agricultural practice (DEST State of the Environment Advisory Council 1996), which aims to ensure that:

- the supply of necessary inputs is sustainable;
- the quality of natural resources is not degraded;
- the environment is not irreversibly harmed;
- the welfare and options of future generations are not jeopardised by the production and consumption activities of the present generation; and
- yields and produce quality are maintained and improved.

In terms of water quality, the focus for sustainable farming systems is on adopting management practices that maintain productivity and minimise the off-farm movement or leaching of potential aquatic contaminants. Key issues include soil erosion, landscape salinity, fertiliser and pesticide management, livestock access to streams, and safe disposal of effluent from intensive animal industries (Hunter et al. 1996).

4.2.2 Scope

Soil, plant and water resource issues that have been taken into account in developing the water quality guidelines for irrigation use are summarised in table 4.2.1. Factors affecting irrigation water quality concern physical, chemical and biological characteristics that may affect the soil environment and crop growth.

Table 4.2.1 Key issues concerning irrigation water quality effects on soil, plants and water resources

	Key issues
Soil	Root zone salinity
	Soil structural stability
	Build-up of contaminants in soil
	Release of contaminants from soil to crops & pastures
Plants	Yield
	Salt tolerance
	Specific ion tolerance
	Foliar injury
	Uptake of toxicants in produce for human consumption
	Contamination by pathogens
Water resources	Deep drainage & leaching below root zone
	Movement of salts, nutrients & contaminants to groundwaters & surface waters
Important associated factors	Quantity and seasonality of rainfall
	Soil properties
	Crop and pasture species and management options
	Land type
	Groundwater depth and quality

Guidelines are also included for general on-farm water use dealing with the corrosion and fouling potential of waters. These characteristics are important for the maintenance of farm equipment (pumps, pipes, etc.). The guidelines may also be applied more widely where corrosion and fouling are of concern.

Specific irrigation water quality guidelines for intensive horticultural activities (e.g. hydroponics and glass-house growing) are not included in this document.

Guidelines for irrigation water quality are given here for biological parameters, salinity and sodicity, inorganic contaminants (i.e. specific ions, including heavy metals and nutrients), organic contaminants (i.e. pesticides) and radiological characteristics. The guidelines are trigger values below which there should be minimal risk of adverse effects. Further investigation is recommended if a trigger value is exceeded, to determine the level of risk.

A more detailed discussion of all water quality parameters included in the guidelines is given in Volume 3, Section 9.2.

4.2.3 Biological parameters

4.2.3.1 Algae

No trigger value for algae in irrigation waters is recommended; however, excessive algal growth may indicate nutrient pollution of the water supply.

Algae are commonly found in most water sources and do not generally cause problems in irrigation waters unless there is excessive growth due to factors such

as suitable flow regime, temperature, abundant nutrients and adequate sunlight. The main problem associated with excessive algal growth in irrigation waters is the blockage of distribution and irrigation equipment. This can result in reduced or uneven flow throughout the irrigation system which may reduce crop yield and increase overall maintenance costs.

4.2.3.2 Cyanobacteria (blue-green algae)

No trigger values for cyanobacteria in irrigation waters are recommended at this time.

Cyanobacteria (blue-green algae) form part of the natural microbial population in most waterbodies. Under certain natural or human-induced circumstances, toxic blooms can occur and these may adversely affect the suitability of waters for irrigation, particularly because toxin residues can potentially accumulate on produce for human or animal consumption. If an algal bloom occurs, it is recommended that an alternative source of irrigation water be used, and that the water be tested for microbial composition and (if necessary) toxicity. There is presently insufficient information available for use in deriving trigger values for cyanobacteria in irrigation water.

4.2.3.3 Human and animal pathogens

Trigger values for thermotolerant coliforms in irrigation waters are provided in table 4.2.2.

Table 4.2.2 Trigger values for thermotolerant coliforms in irrigation waters used for food and non-food crops[a]

Intended use	Level of thermotolerant coliforms[b]
Raw human food crops in direct contact with irrigation water (e.g. via sprays, irrigation of salad vegetables)	<10 cfu[c] / 100 mL
Raw human food crops not in direct contact with irrigation water (edible product separated from contact with water, e.g. by peel, use of trickle irrigation); or crops sold to consumers cooked or processed	<1000 cfu / 100 mL
Pasture and fodder for dairy animals (without withholding period)	<100 cfu / 100 mL
Pasture and fodder for dairy animals (with withholding period of 5 days)	<1000 cfu / 100 mL
Pasture and fodder (for grazing animals except pigs and dairy animals, i.e. cattle, sheep and goats)	<1000 cfu / 100 mL
Silviculture, turf, cotton, etc. (restricted public access)	<10 000 cfu / 100 mL

a Adapted from ARMCANZ, ANZECC & NHMRC (1999)

b Median values (refer to text)

c cfu = colony forming units

It is generally not feasible nor warranted to test irrigation water for the presence of the wide range of water-borne microbial pathogens that may affect human and animal health. The guidelines recommended here are based on the practicable testing of irrigation waters for the presence of thermotolerant coliforms (also known as faecal coliforms), which gives an indication of faecal contamination and thus the possible presence of microbial pathogens (NHMRC & ARMCANZ 1996). However, the test does not specifically indicate whether pathogenic organisms are present.

It is recommended that a median value of thermotolerant coliforms be used, based on a number of readings generated over time from a regular monitoring program. Investigations of likely causes are warranted when 20% of results exceed four times the median guideline value (ARMCANZ, ANZECC & NHMRC 2000).

For helminths, a trigger value of ≤1 helminth egg per litre is proposed for the protection of crop consumers in areas where helminth infections are known to be endemic. A lower value of 0.5 eggs per litre may be required to protect farm workers and their families in situations of direct exposure to the water (ARMCANZ, ANZECC & NHMRC 2000). Insufficient information is available for use in setting guidelines for protozoa and viruses in irrigation water.[a]

a See also Section 9.2.2.3

4.2.3.4 Plant pathogens

No trigger values for plant pathogens in irrigation waters are recommended at this time. As a general precaution, disinfestation treatment is advisable for water that contains plant pathogens and is to be used for irrigating potentially susceptible plants.

Agricultural crops and pastures can be affected by various plant pathogens transmitted through a number of different pathways including irrigation water, although it is believed that the risk from pathogens in irrigation water is low under most circumstances. However, plant pathogens in irrigation water used for intensive agricultural and horticultural industries (particularly where wastewaters are reused) can potentially lead to crop damage and economic loss.

A great deal of work needs to be done before guidelines can be developed, particularly regarding the efficacy of water-borne plant pathogens on a wide range of crops.

4.2.4 Irrigation salinity and sodicity

4.2.4.1 Salinity and sodicity

To assess the salinity and sodicity of water for irrigation use, a number of interactive factors must be considered. As outlined in this section, these include irrigation water quality, soil properties, plant salt tolerance, climate, landscape (including geological and hydrological features), and water and soil management.

Salinity is the presence of soluble salts in or on soils, or in waters. High salinity levels in soils may result in reduced plant productivity or, in extreme cases, the elimination of crops and native vegetation. Salinity related issues are of concern in many parts of Australia but salinisation is currently considered to be only of minor importance in New Zealand.

Sodicity is the presence of a high proportion of sodium (Na^+) ions relative to calcium (Ca^{2+}) and magnesium (Mg^{2+}) ions in soil or water. Sodicity degrades soil structure by breaking down clay aggregates, which makes the soil more erodible and less permeable to water, and reduces plant growth.

The effects of salinity and sodicity in irrigation waters are very situation-specific, making it inappropriate to set water quality trigger values for general application. Factors which need to be considered include: the type of crop being cultivated and its salt tolerance, the characteristics of the soil under irrigation, soil management and water management practices, climate and rainfall (figure 4.2.1).

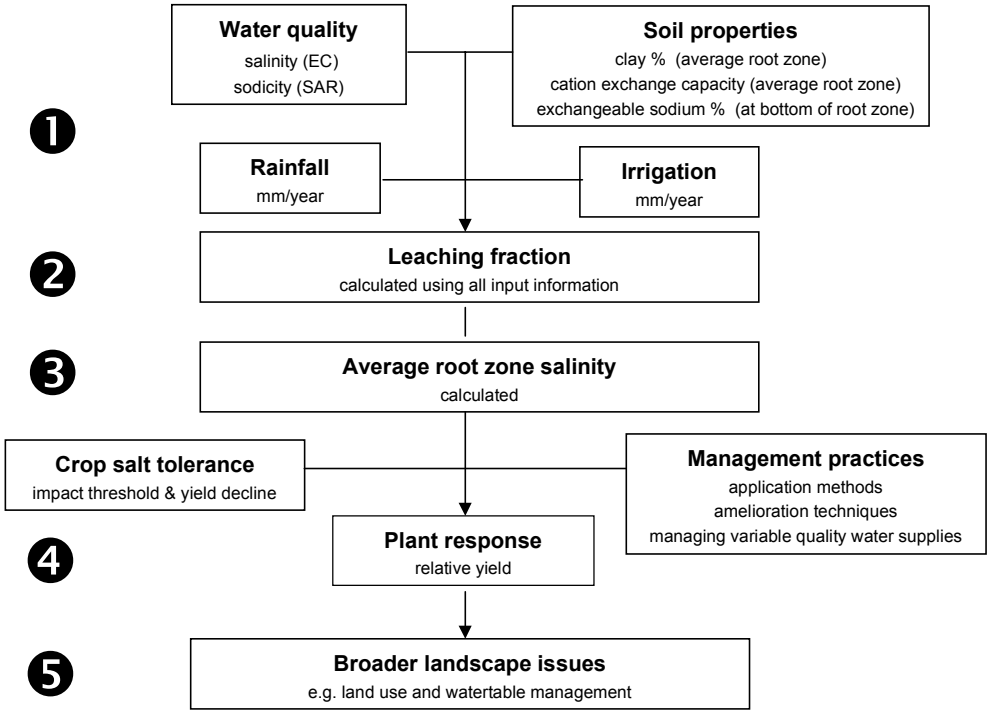


Figure 4.2.1 Flow diagram for evaluating salinity and sodicity impacts of irrigation water

a See details in Section 9.2.3

There are five key steps to determining the suitability of irrigation water with respect to salinity and sodicity (figure 4.2.1).[a]

Step 1. Identify the soil properties, water quality, climate (rainfall) and management (irrigation application rates) practices for the site in question.

Step 2. Estimate the leaching fraction under the proposed irrigation regime using approaches outlined in this section.

Step 3. Estimate the new average root zone salinity as outlined in this section. Average root zone salinity is considered the key limitation to plant growth in response to salinity and sodicity levels in irrigation water. However, poor soil structure can also reduce plant yields by limiting aeration, water infiltration and root growth. The likelihood of soil structural problems induced by irrigation can be predicted from trigger values derived in this section.

Step 4. Estimate relative plant yield (although note that the impact of salinity and sodicity can be modified by management practices as discussed later in this section).

Step 5. Consider salinity and sodicity problems within the framework of broader catchment issues such as regional watertables, groundwater pollution and surface water quality. Watertable salinity develops in response to excess water and salts accumulating in sensitive parts of the landscape. Excess water can percolate to groundwaters as a result of changing climatic patterns (e.g. frequency and duration of rainfall events), land use or land management (including irrigation). Before an irrigation scheme is developed, the planning process should include investigation of the regional hydrogeology to avoid development of watertable salinity. The guidelines given here concentrate on localised effects of irrigation, but broader salinity issues should not be ignored.

Software *SALF PREDICT* is now available. It estimates the parameters necessary for a detailed assessment of irrigation water quality in relation to soil properties, rainfall, water quality and plant salt tolerance. The software is based on summer rainfall areas and should be used with some caution in winter rainfall areas. It incorporates many of the detailed algorithms presented in Volume 3, Section 9.2.3. The software is provided on the CD ROM provided with these Guidelines and is also available from the Queensland Department of Natural Resources.

A simple initial assessment can be made by measuring the electrical conductivity (EC_i) and concentrations of sodium (Na^+), calcium (Ca^{2+}) and magnesium (Mg^{2+}) in irrigation water. Note that EC is expressed in units of dS/m throughout Section 4.2.4 (1 dS/m = 1000 μS/cm).

Determining the suitability of irrigation water salinity for a crop

Calculate the average root zone salinity (EC_{se}) from EC_i and the average root zone leaching fraction (LF), to see if a crop is likely to be affected by the irrigation water salinity. First, estimate the LF of the soil being irrigated (i.e. the proportion of applied water that leaches below the root zone). Approximate average LF values for four broad soil types are listed in table 4.2.3. Then calculate EC_{se} using the following equation:

$$EC_{se} = \frac{EC_i}{2.2 \times LF} \qquad (4.1)$$

where:
EC_{se} = average root zone salinity in dS/m
EC_i = electrical conductivity of irrigation water in dS/m
LF = average leaching fraction.

Table 4.2.3 Soil type and average root zone leaching fraction[a]

Soil type	Average root zone LF
Sand	0.6
Loam	0.33
Light clay	0.33
Heavy clay	0.2

a From DNR (1997a), adapted from DNR (1997b)

Then use the EC_{se} value to assess the general level of crop tolerance to the irrigation water salinity by comparing it with the values in table 4.2.4. Alternatively, compare the EC_{se} with the relative salt tolerances of specific crop and pasture species provided here in table 4.2.5 and in Volume 3, Section 9.2.3, table 9.2.10.

Table 4.2.4 Soil and water salinity criteria based on plant salt tolerance groupings[a]

Plant salt tolerance groupings	Water or soil salinity rating	Average root zone salinity, EC_{se} (dS/m)[b]
Sensitive crops	Very low	<0.95
Moderately sensitive crops	Low	0.95–1.9
Moderately tolerant crops	Medium	1.9–4.5
Tolerant crops	High	4.5–7.7
Very tolerant crops	Very high	7.7–12.2
Generally too saline	Extreme	>12.2

a Adapted from DNR (1997b)

b 1 dS/m = 1000 µS/cm

A list of the relative salt tolerances of a limited selection of common field crop, pasture and horticulture species is provided in table 4.2.5. Information in this table is derived from data currently available in the literature, but preference should be given to locally derived data where available. This gives approximate values of average root zone salinities at the threshold level (the level causing yield reduction). It also shows electrical conductivity of irrigation water at the threshold level for a range of soil types, but it is meant as a general guide only.[a]

a See also Section 9.2.3

If at all uncertain about salt tolerance or the effect of irrigation water quality on soil structure, submit a soil sample for analysis and seek expert advice.

Determining the risk of soil structure degradation caused by irrigation water quality

Calculate the sodium adsorption ratio (SAR) and use it (with EC_i) to predict soil structure stability in relation to irrigation water. The SAR value measures the relative concentration of sodium (Na^+) to calcium (Ca^{2+}) and magnesium (Mg^{2+}) and can be calculated from the following equation:

$$SAR = \frac{Na^+}{\sqrt{\frac{Ca^{2+} + Mg^{2+}}{2}}} \qquad (4.2)$$

Where Na^+, Ca^{2+} and Mg^{2+} are expressed in $mmole_c/L$ (where subscript c indicates change).

Evaluate the quality of the irrigation water by superimposing its EC_i and SAR values on figure 4.2.2, to see if it will affect soil structure (through clay aggregate breakdown). Water quality that falls to the right of the dashed line is unlikely to cause soil structural problems. Water quality that falls to the left of the solid line is likely to induce degradation of soil structure; corrective management will be required (e.g. application of lime or gypsum). Water that falls between the lines is of marginal quality and should be treated with caution.

Table 4.2.5 Tolerance of plants to salinity in irrigation water[a]

Common name	Scientific name	Average root zone salinity threshold (EC_{se}) (dS/m)[b]	EC_i threshold for crops growing in		
			sand	loam	clay
Field Crops					
Barley, grain	*Hordeum vulgare*	8	12.6	7.2	4.2
Cotton	*Gossypium hirsutum*	7.7	12.1	6.9	4.0
Beet, sugar	*Beta vulgaris*	7	11.0	6.3	3.7
Sorghum	*Sorghum bicolor*	6.8	9.4	5.3	3.1
Wheat	*Triticum aestivum*	6	9.4	5.3	3.1
Sunflower	*Helianthus annuus*	5.5	7.5	4.3	2.5
Oats	*Avena sativa*	5	7.0	4.0	2.3
Soybean	*Glycine max*	5	7.0	4.0	2.3
Peanut	*Arachis hypogala*	3.2	4.4	2.5	1.5
Rice, paddy	*Oryza sativa*	3	4.8	2.7	1.6
Corn, grain, sweet	*Zea mays*	1.7	3.2	1.8	1.1
Sugarcane	*Saccharum officinarum*	1.7	4.3	2.5	1.4
Fruits					
Olive	*Olea europaea*	4	5.1	2.9	1.7
Macadamia seedling		3.6	4.6	2.6	1.5
Peach	*Prunus persica*	3.2	4.7	2.7	1.6
Rockmelon	*Cucumis melo*	2.2	4.6	2.6	1.5
Grapefruit	*Citrus paradisi*	1.8	3.0	1.7	1.0
Orange	*Citrus sinensis*	1.7	2.9	1.7	1.0
Grape	*Vitis* spp.	1.5	3.3	1.9	1.1
Avocado	*Persea americana*	1.3	2.3	1.3	0.8
Apple	*Malus sylvestris*	1	2.0	1.2	0.7
Pastures					
Wheatgrass, tall	*Agropyron elongatum*	7.5	12.5	7.2	4.2
Rhodes grass, Pioneer	*Chloris gayana*	7	12.8	7.3	4.2
Couch grass	*Cynodon dactylon*	6.9	10.8	6.1	3.6
Buffel grass, Gayndah	*Cenchrus ciliaris var Gayndah*	5.5	8.2	4.7	2.7
Phalaris	*Phalaris tuberosa (aquatica)*	4.2	5.3	3.0	1.8
Fescue	*Festuca clatior*	3.9	7.3	4.2	2.4
Green panic, Petri	*Panicum maximum*	3	5.6	3.2	1.8
Townsville stylo	*Stylosanthes humilis*	2.4	3.7	2.1	1.2
Clover, Berseem Clover	*Trifolium alexandrinum*	2	3.8	2.2	1.3
Lucerne, Hunter River	*Medicago sativa*	2	4.7	2.7	1.6
Clover, strawberry (Palestine)	*Trifolium fragiferum*	1.6	3.3	1.9	1.1
Snail medic	*Medicago scutellata*	1.5	2.9	1.7	1.0
Clover, white (New Zealand)	*Trifolium repens*	1	2.5	1.4	0.8
Vegetables					
Zucchini	*Cucurbita pepo melopepo*	4.7	7.3	4.2	2.4
Beet, garden	*Beta vulgaris*	4	6.5	3.7	2.1
Broccoli	*Brassica oleracea*	2.8	4.9	2.8	1.6
Cucumber	*Cucumis sativus*	2.5	4.2	2.4	1.4
Pea	*Pisum sativum* L.	2.5	3.2	1.8	1.1
Tomato	*Lycopersicon esculentum*	2.3	3.5	2.0	1.2
Potato	*Solanum tuberosum*	1.7	3.2	1.8	1.1
Pepper	*Capsicum annum*	1.5	2.8	1.6	0.9
Lettuce	*Lactuca sativa*	1.3	2.7	1.5	0.9
Onion	*Allium cepa*	1.2	2.3	1.3	0.8
Eggplant	*Solanum melongena*	1.1	3.2	1.8	1.1
Bean	*Phaseolus vulgaris*	1	1.9	1.1	0.6
Carrot	*Daucus carota*	1	2.2	1.2	0.7

a From DNR (1997a), adapted from DNR (1997b); b 1 dS/m = 1000 µS/cm

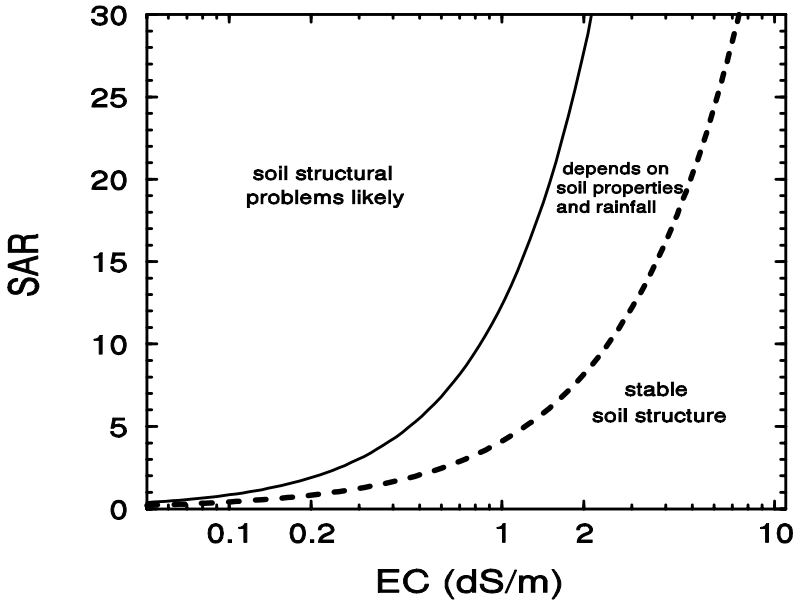


Figure 4.2.2 Relationship between SAR and EC of irrigation water for prediction of soil structural stability (from DNR 1997a, adapted from DNR 1997b; note that 1 dS/m = 1000 μS/cm)

4.2.5 Major ions of concern for irrigation water quality

4.2.5.1 Bicarbonate

No trigger value is recommended for bicarbonate in irrigation waters.

Elevated levels of bicarbonate in irrigation waters can adversely affect irrigation equipment, soil structure and crop foliage. These problems occur when the bicarbonate (or carbonate) in solution with calcium is sufficient to exceed the solubility of calcium carbonate. The precipitation of calcium carbonate can lead to white scale formation on leaves and fruit and may clog irrigation equipment.

The same process can give rise to precipitates of calcium carbonate in soil. This will effectively increase the sodium adsorption ratio (SAR) or exchangeable sodium percentage (ESP) and may lead to soil structural problems. An overview of the effect of irrigation with waters of high SAR is given in Volume 3, Section 9.2.3.

4.2.5.2 Chloride

Issues concerning chloride in irrigation waters relate to the risk of: (1) foliar injury to crops; and (2) increased uptake by plants of cadmium from soil. These are discussed more fully in Volume 3, Section 9.2.4.2.

1 Foliar injury

Trigger values for prevention of foliar injury due to chloride in irrigation water from sprinkler application are provided in table 4.2.6.

Chloride in irrigation water can also reduce the quality of tobacco leaf. Chloride concentrations >40 mg/L are considered unsuitable for irrigation of tobacco and some reduction in quality may occur with concentrations in the range 25–40 mg/L (Gill 1986).

Table 4.2.6 Chloride concentrations (mg/L) causing foliar injury in crops of varying sensitivity[a]

Sensitive <175	Moderately sensitive 175–350	Moderately tolerant 350–700	Tolerant >700
Almond	Pepper	Barley	Cauliflower
Apricot	Potato	Maize	Cotton
Citrus	Tomato	Cucumber	Sugar beet
Plum		Lucerne	Sunflower
Grape		Safflower	
		Sorghum	

a After Maas (1990)

2 Interaction between chloride in irrigation water and cadmium in soil

Trigger values for assessing chloride levels in irrigation water with respect to increased cadmium uptake by crops are provided in table 4.2.7.

Table 4.2.7 Risks of increasing cadmium concentrations in crops due to chloride in irrigation waters[a]

Irrigation water chloride concentration (mg/L)	Risk of increasing crop cadmium concentrations
0–350	Low
350–750	Medium
>750	High

a McLaughlin et al. (1999)

If high chloride concentrations are present in irrigation water, it is recommended that produce is tested for cadmium concentration in the edible portions (e.g. tubers for potatoes, leaves for leafy vegetables, grain for cereals, etc.).

4.2.5.3 Sodium

Trigger values for prevention of foliar injury due to sodium in irrigation water from sprinkler application are provided in table 4.2.8. Trigger values for specific toxicity effects are provided in table 4.2.9.

Table 4.2.8 Sodium concentration (mg/L) causing foliar injury in crops of varying sensitivity[a]

Sensitive <115	**Moderately sensitive 115–230**	**Moderately tolerant 230–460**	**Tolerant >460**
Almond	Pepper	Barley	Cauliflower
Apricot	Potato	Maize	Cotton
Citrus	Tomato	Cucumber	Sugar beet
Plum		Lucerne	Sunflower
Grape		Safflower	
		Sesame	
		Sorghum	

a After Maas (1990)

Table 4.2.9 Effect of sodium expressed as sodium adsorption ratio (SAR) on crop yield and quality under non-saline conditions[a]

Tolerance to SAR and range at which affected	Crop	Growth response under field conditions
Extremely sensitive SAR = 2–8	Avocado Deciduous fruits Nuts Citrus	Leaf tip burn, leaf scorch
Sensitive SAR = 8–18	Beans	Stunted growth
Medium SAR = 18–46	Clover Oats Tall fescue Rice Dallis grass	Stunted growth, possible sodium toxicity, possible calcium or magnesium deficiency
High SAR = 46–102	Wheat Cotton Lucerne Barley Beets Rhodes grass	Stunted growth

a After Pearson (1960); SAR = Sodium Adsorption Ratio (see Section 4.2.4.1)

4.2.6 Heavy metals and metalloids

Long-term trigger values (LTV) and short-term trigger values (STV) for heavy metals and metalloids in irrigation water are presented in table 4.2.10. Concentrations in irrigation water should be less than the recommended trigger values.

Table 4.2.10 Agricultural irrigation water long-term trigger value (LTV), short-term trigger value (STV) and soil cumulative contaminant loading limit (CCL) triggers for heavy metals and metalloids[a]

Element	Suggested soil CCL[b] (kg/ha)	LTV in irrigation water (long-term use — up to 100 yrs) (mg/L)	STV in irrigation water (short-term use — up to 20 yrs) (mg/L)
Aluminium	ND	5	20
Arsenic	20	0.1	2.0
Beryllium	ND	0.1	0.5
Boron	ND	0.5	Refer to table 9.2.18 (Volume 3)
Cadmium	2	0.01	0.05
Chromium	ND	0.1	1
Cobalt	ND	0.05	0.1
Copper	140	0.2	5
Fluoride	ND	1	2
Iron	ND	0.2	10
Lead	260	2	5
Lithium	ND	2.5 (0.075 Citrus crops)	2.5 (0.075 Citrus crops)
Manganese	ND	0.2	10
Mercury	2	0.002	0.002
Molybdenum	ND	0.01	0.05
Nickel	85	0.2	2
Selenium	10	0.02	0.05
Uranium	ND	0.01	0.1
Vanadium	ND	0.1	0.5
Zinc	300	2	5

a Trigger values should only be used in conjunction with information on each individual element and the potential for off-site transport of contaminants (Volume 3, Section 9.2.5)

b ND = Not determined; insufficient background data to calculate CCL

The *long-term trigger value* (LTV) is the maximum concentration (mg/L) of contaminant in the irrigation water which can be tolerated assuming 100 years of irrigation, based on the irrigation loading assumptions described in Volume 3, Section 9.2.5.

The *short-term trigger value* (STV) is the maximum concentration (mg/L) of contaminant in the irrigation water which can be tolerated for a shorter period of time (20 years) assuming the same maximum annual irrigation loading to soil as for LTV.

The LTV and STV values have been developed: (1) to minimise the build-up of contaminants in surface soils during the period of irrigation; and (2) to prevent the direct toxicity of contaminants in irrigation waters to standing crops. Where LTV and STV have been set at the same value, the primary concern is the direct toxicity of irrigation water to the standing crop (e.g. for lithium and citrus crops), rather than a risk of contaminant accumulation in soils and plant uptake.

The trigger value for contaminant concentration in soil is defined as the *cumulative contaminant loading limit* (CCL). The CCL is the maximum contaminant loading in soil defined in gravimetric units (kg/ha) and indicates the cumulative amount of contaminant added, above which site-specific risk assessment is recommended if irrigation and contaminant addition is continued.

Once the CCL has been reached, it is recommended that a soil sampling and analysis program be initiated on the irrigated area, and an environmental impact assessment of continued contaminant addition be prepared. As background concentrations of contaminants in soil may vary with soil type, and contaminant behaviour is dependent on soil texture, pH, salinity, etc., it should be noted that CCLs may be overly protective in some situations and less protective in others. The CCL is designed for use in soils with no known history of contamination from other sources. When it is suspected that the soil is contaminated before commencement of irrigation, background levels of contaminants in the soil should be determined and the CCL adjusted accordingly.

The trigger values assume that irrigation water is applied to soils and that soils may reduce contaminant bioavailability by binding contaminants and reducing concentrations in solution. They may not be suitable for plants grown in soil-less media (hydroponics or similar methods). The trigger values should only be used in conjunction with the discussion in Volume 3 on each individual element and the potential for off-site transport of contaminants.[a] The assumptions underlying these trigger values are recognised internationally as a basis for developing irrigation water quality guidelines.

a See Section 9.2.5 for full details of methods used

4.2.7 Nitrogen and phosphorus

Long-term trigger values (LTV) and short-term trigger values (STV) for nitrogen and phosphorus in irrigation water are presented in table 4.2.11. They are based on maintaining crop yield, preventing bioclogging of irrigation equipment and minimising off-site impacts. Concentrations in irrigation water should be less than the recommended trigger values.

Table 4.2.11 Agricultural irrigation water long-term trigger value (LTV) and short-term trigger value (STV) guidelines for nitrogen and phosphorus

Element	LTV in irrigation water (long-term — up to 100 yrs) (mg/L)	STV in irrigation water (short-term — up to 20 yrs) (mg/L)
Nitrogen	5	25–125 [a]
Phosphorus	0.05 (To minimise bioclogging of irrigation equipment only)	0.8–12 [a]

a Requires site-specific assessment (see Section 9.2.6)

The concepts of long-term trigger value (LTV) and short-term trigger value (STV) developed for metals and metalloids have also been used to develop guidelines for phosphorus (P) and nitrogen (N).

Excess quantities of N can lead to leaching of N into groundwater and surface water, over-stimulation of plant growth (decreasing yields) and stimulation of algal growth in surface water. The LTV for nitrogen has been set at a concentration low enough to ensure no decreases in crop yields or quality occur. The STV range for nitrogen has been set to minimise the risk of contaminating groundwater and surface water and requires site-specific information[a] which considers the crop that is being grown, environmentally significant concentrations, and gaseous losses.

a See Section 9.2.6

Phosphorus is often the nutrient that stimulates rapid growth of many microorganisms (i.e. algal blooms). The LTV for P has been set to prevent algal growth in irrigation water. The STV range for P has been set as an interim range due to the limited data currently available. Calculation of the interim range considers the fertiliser value of phosphorus in water, the phosphorus removed from irrigation sites through harvest, fertiliser inputs, and phosphorus sorption/retention capacities of soils.[b]

b An interim method of calculating a site-specific STV is outlined in Section 9.2.6

The trigger values provided in table 4.2.11 should only be used in conjunction with the discussion contained in Volume 3, Section 9.2.6.

4.2.8 Pesticides

Trigger values for pesticides in irrigation water are listed in table 4.2.12. They consider likely adverse effects of herbicides on crop growth but do not consider potential impacts on aquatic ecosystems. They are based on relatively limited information and include only a subset of herbicides (and no other pesticides) that might be found in irrigation waters.

4.2.9 Radiological quality of irrigation water

Trigger values for the radiological quality of agricultural waters are given in table 4.2.13.

Radioactive contaminants can originate from both natural and artificial sources and can potentially be found in surface waters and groundwaters. The main risks to human health due to radioactivity in irrigation water arise from the transfer of radionuclides to crop and animal products for human consumption. Cancer is a potential health hazard for humans associated with exposure to radionuclides in irrigation water.

Table 4.2.12 Interim trigger value concentrations for a range of herbicides registered in Australia for use in or near waters[a]

Herbicide	Residue limits in irrigation water (mg/L)[b]	Hazard to crops from residue in water[c]	Crop injury threshold in irrigation water (mg/L)
Acrolein	0.1	+	Flood or furrow: beans 60, corn 60, cotton 80, soybeans 20, sugar-beets 60. Sprinkler: corn 60, soybeans 15, sugar-beets 15
AF 100		+	Beets (rutabaga) 3.5, corn 3.5
Amitrol	0.002	++	Lucerne 1600, beans 1200, carrots 1600, corn 3000, cotton 1600, grains sorghum 800
Aromatic solvents (Xylene)		+	Oats 2400, potatoes 1300, wheat 1200
Asulam		++	
Atrazine		++	
Bromazil		+++	
Chlorthiamid		++	
Copper sulfate		+	Apparently above concentrations used for weed control
2,4-D		++	Field beans 3.5–10, grapes 0.7–1.5, sugar-beets 1.0–10
Dicamba		++	Cotton 0.18
Dichlobenil		++	Lucerne 10, corn 10, soybeans 1.0, sugar-beets 1.0–10, corn 125, beans 5
Diquat		+	
Diuron	0.002	+++	
2,2-DPA (Dalapon)	0.004	++	Beets 7.0, corn 0.35
Fosamine		+++	
Fluometuron		++	Sugar-beets, alfalfa, tomatoes, squash 2.2
Glyphosate		+	
Hexazinone		+++	
Karbutilate		+++	
Molinate		++	
Paraquat		+	Corn 10, field beans 0.1, sugar-beets 1.0
Picloram		+++	
Propanil		++	Alfalfa 0.15, brome grass (eradicated) 0.15
Simazine		++	
2,4,5-T		++	Potatoes, alfalfa, garden peas, corn, sugar-beets, wheat, peaches, grapes, apples, tomatoes 0.5
TCA (Trichloroacetic acid)		+++	
Terbutryne		++	
Triclopyr		++	

a From ANZECC (1992). These should be regarded as interim trigger values only.

b Guidelines have not been set for herbicides where specific residue limits are not provided, except for a general limit of 0.01 mg/L for all herbicides in NSW.

c Hazard from residue at maximum concentration likely to be found in irrigation water: + = low, ++ = moderate, +++ = high

Table 4.2.13 Trigger values for radioactive contaminants for irrigation water

Radionuclide	Trigger concentration
Radium 226	5 Bq/L
Radium 228	2 Bq/L
Uranium 238	0.2 Bq/L
Gross alpha	0.5 Bq/L
Gross beta (excluding K-40)	0.5 Bq/L

4.2.10 General water uses

4.2.10.1 pH

To limit corrosion and fouling of pumping, irrigation and stock watering systems, pH should be maintained between 6 and 8.5 for groundwater systems and between 6 and 9 for surface water systems.

The pH of water is a measure of its acidity or alkalinity. Generally, pH itself is not a water quality issue of concern, but it can indicate the presence of a number of related problems. The greatest hazard with high or low pH is the potential for deterioration as a result of corrosion or fouling. Values between 4 and 6 should be regarded with caution and a pH >6 should be maintained to reduce the potential for corrosion. The upper pH limit for groundwaters should be slightly lower than for surface waters because of the increased potential for encrustation and fouling. Soil and animal health will not generally be affected by water with pH in the range of 4–9.

4.2.10.2 Corrosion

Trigger values for assessing the corrosiveness of water are given in table 4.2.14.

Table 4.2.14 Corrosion potential of waters on metal surfaces as indicated by pH, hardness, Langelier index, Ryznar index and the log of chloride:carbonate ratio

Parameter[a]	Value	Comments
pH	<5 5 to 6 >6	High corrosion potential Likelihood of corrosion Limited corrosion potential
Hardness	<60 mg/L CaCO3	Increased corrosion potential
Langelier Index	<-0.5 -0.5 to 0.5	Increased corrosion potential Limited corrosion potential
Ryznar Index	<6 >7	Limited corrosion potential Increased corrosion potential
Log of chloride to carbonate ratio	>2	Increased corrosion potential

a For further information on these parameters refer to Volume 3, Section 9.2.9.1

Corrosion of pumping, irrigation and stock watering equipment is a common problem in many agricultural areas of Australia, particularly where groundwater sources are used. It often results in the deterioration of well and pumping equipment, pipelines, channels, sprinkler devices and storage tanks, leading to decreased or uneven water distribution. Corrosion can be caused by chemical, physical or microbiological processes acting on metal surfaces in contact with

water. Plastics and concrete may also deteriorate, through processes similar to corrosion, if elevated levels of certain constituents are present.

4.2.10.3 Fouling

Trigger values for assessing the fouling potential of water are given in table 4.2.15.

Table 4.2.15 Fouling potential of waters as indicated by pH, hardness, Langelier index, Ryznar index and the log of chloride:carbonate ratio

Parameter[a]	Value	Comments
pH	<7	Limited fouling potential
	7 to 8.5	Moderate fouling potential (groundwater)[b]
	>8.5	Increased fouling potential (groundwater)[c]
Hardness	>350 mg/L CaCO3	Increased fouling potential
Langelier Index	>0.5	Increased fouling potential
	-0.5 to 0.5	Limited fouling potential
Ryznar Index	<6	Increased fouling potential
	>7	Limited fouling potential
Log of chloride to carbonate ratio	<2	Increased fouling potential

a For further information on these parameters refer to Volume 3, Section 9.2.9.1
b For surface waters, pH range 7 to 9
c For surface waters, pH >9

Fouling of agricultural water systems can lead to decreased water quality and yield as a result of clogging, encrustation and scaling. All parts of the system can be affected including wells, pumping equipment, pipes and sprinklers. The main causes of fouling in agricultural water systems can be attributed to physical, chemical and biological properties of the water.

4.2.10.4 Agricultural chemical preparation

Insufficient information is available to set trigger values for water used to prepare agricultural chemicals.

Water is the most common additive and diluent used in the preparation of agricultural chemicals (e.g. pesticides, stock dips and fertilisers) for on-farm use. Although some agricultural chemicals can withstand a range of water qualities before performance is substantially affected, it is recommended that good quality water be used to ensure the desired result.

To check that a particular water is suitable for use with an agricultural chemical, it is best to make up and test a trial solution first. Specific details on water quality requirements should be noted from the product label or by contacting the manufacturer.

4.3 Livestock drinking water quality

Good water quality is essential for successful livestock production. Poor quality water may reduce animal production and impair fertility. In extreme cases, stock may die. Contaminants in drinking water can produce residues in animal products (e.g. meat, milk and eggs), adversely affecting their saleability and/or creating human health risks. Animal industries themselves may impair water quality downstream (e.g. through faecal contamination), highlighting the need for an integrated approach to land and water management in rural catchments.

Daily water intake varies widely among different forms of livestock and is also influenced by factors such as climate and the type of feed being consumed. Average and peak daily water requirements for a range of livestock are given in Volume 3, Section 9.3.1.

4.3.1 Derivation and use of guidelines

Many factors influence the suitability of waters for livestock watering. Requirements may differ between animal species (generally tolerances decrease in the order sheep, cattle, horses, pigs, poultry), and between different stages of growth and animal condition, and between monogastric and ruminant animals. Moreover, stock accustomed to good quality water can initially suffer ill effects or refuse to drink water of poorer quality, but may adjust if introduced gradually.

A review of the scientific literature reveals that most trigger values tend to be based on field observations rather than rigorous experimentation, although there are notable exceptions. In the present guidelines, several new trigger values have been calculated using data on chronic and toxic effect levels on animals. Since derivation of most trigger values for livestock drinking water needs further validation, they should be considered interim guidelines at this stage. Further details on the derivation of each trigger value and a more detailed discussion of all water quality parameters included in the guidelines are given in Volume 3, Section 9.3.

The scope of the guidelines for livestock drinking water includes biological, chemical and radiological characteristics that may affect animal health. The guidelines are trigger values below which there should be minimal risk to animal health. If the water quality exceeds a trigger value, it is advisable to investigate further to determine the level of risk.

4.3.2 Biological parameters

4.3.2.1 Cyanobacteria (blue-green algae)

An increasing risk to livestock health is likely when cell counts of Microcystis exceed 11 500 cells/mL and/or concentrations of microcystins exceed 2.3 µg/L expressed as microcystin-LR toxicity equivalents. There are insufficient data available to derive trigger values for other species of cyanobacteria.

Diagnostic procedure

The presence of an algal bloom does not necessarily mean that animals will be poisoned, so the following steps should be taken to assess the risk from such a bloom (after Carmichael & Falconer 1993).

1. Establish that animals are drinking the water or eating algal mats from the area where there is a substantial bloom.
2. Indentify the algae associated with the bloom to determine whether cyanobacteria are present in numbers large enough to constitute a risk.
3. If necessary, chemically analyse a sample of the bloom to identify and quantify toxins present.

Since all blooms of cyanobacteria have the potential to be toxic and all livestock are susceptible, it is prudent to consider all scums toxic until proven safe, as described above. In the interim, stock should be withdrawn from the water supply and an alternative source used. Where an alternative source is not available and the bloom is localised, it may be possible to allow stock to drink from an area on the upwind side of the bloom. In the long term, prevention of blooms is by far the best strategy, and water supplies should be managed so that nutrient inputs are minimal.[a]

a See also Section 9.3.3.1

4.3.2.2 Pathogens and parasites

Drinking water for livestock should contain less than 100 thermotolerant coliforms per 100 mL (median value).

It is generally not feasible nor warranted to test livestock drinking water for the presence of the wide range of water-borne microbial pathogens (bacteria, viruses and protozoa) and parasites that may affect stock health. In practice, water supplies are more commonly tested for the presence of thermotolerant coliforms (also known as faecal coliforms), to give an indication of faecal contamination and thus the possible presence of microbial pathogens (NHMRC & ARMCANZ 1996). However, the test does not specifically indicate whether pathogenic organisms are present or not. Testing for specific organisms may be necessary in these situations if animal health is affected.

It is recommended that a median value of thermotolerant coliforms is used, based on a number of readings generated over time from a regular monitoring program. Investigations of likely causes are warranted when 20% of results exceed four times the median trigger value (ARMCANZ, ANZECC & NHMRC 1999).[b]

b Section 9.3.3.2

4.3.3 Major ions of concern for livestock drinking water quality

Many inorganic salts are essential nutrients for animal health, but elevated concentrations of certain compounds may cause chronic or toxic effects in livestock. Unless otherwise stated, the trigger values relate to the total concentration of the constituent, irrespective of whether it is dissolved, complexed with an organic compound, or bound to suspended solids.[c]

c Section 9.3.4

4.3.3.1 Calcium

Stock should tolerate concentrations of calcium in water up to 1000 mg/L, if calcium is the dominant cation and dietary phosphorus levels are adequate. In the presence of high concentrations of magnesium and sodium, or if calcium is added to feed as a dietary supplement, the level of calcium tolerable in drinking water may be less.

Calcium is an essential element in the animal diet. However, high calcium concentrations may cause phosphorus deficiency by interfering with phosphorus absorption in the gastrointestinal tract.

4.3.3.2 Magnesium

Insufficient information is available to set trigger values for magnesium in livestock drinking water.

Magnesium is an essential element for animal nutrition. In high doses magnesium can cause scouring and diarrhoea, lethargy, lameness, decreased feed intake and decreased performance. Drinking water containing magnesium at concentrations up to 2000 mg/L has been found to have no adverse effects on cattle.[a]

a See Section 9.3.4.2

4.3.3.3 Nitrate and nitrite

Nitrate concentrations less than 400 mg/L in livestock drinking water should not be harmful to animal health. Stock may tolerate higher nitrate concentrations in drinking water, provided nitrate concentrations in feed are not high. Water containing more than 1500 mg/L nitrate is likely to be toxic to animals and should be avoided.

Concentrations of nitrite exceeding 30 mg/L may be hazardous to animal health.

Both nitrate and nitrite can cause toxicity to animals, with nitrite being far more toxic than nitrate. Symptoms of acute poisoning include increased urination, restlessness and cyanosis, leading to vomiting, convulsions and death.

Confusion can arise concerning trigger values for nitrate and nitrite because concentrations are sometimes reported on the basis of their respective nitrogen (N) contents, i.e. as nitrate-N and nitrite-N. Note that trigger values in the present guidelines are expressed as nitrate and nitrite. The conversions are as follows:

$$1 \text{ mg/L nitrate-N} = 4.43 \text{ mg/L nitrate}, \tag{4.3}$$

$$1 \text{ mg/L nitrite-N} = 3.29 \text{ mg/L nitrite}. \tag{4.4}$$

4.3.3.4 Sulfate

No adverse effects to stock are expected if the concentration of sulfate in drinking water does not exceed 1000 mg/L. Adverse effects may occur at sulfate concentrations between 1000 and 2000 mg/L, especially in young or lactating animals or in dry, hot weather when water intake is high. These effects may be temporary and may cease once stock become accustomed to the water. Levels of sulfate greater than 2000 mg/L may cause chronic or acute health problems in stock.

Sulfur is essential for animal nutrition. Excessive concentrations of sulfate in water typically cause diarrhoea in stock, but animals generally avoid water containing high sulfate concentrations.

4.3.3.5 Total dissolved solids (salinity)

Recommended concentrations of total dissolved solids in drinking water for livestock are given in table 4.3.1.

Table 4.3.1 Tolerances of livestock to total dissolved solids (salinity) in drinking water[a]

Livestock	Total dissolved solids (mg/L)		
	No adverse effects on animals expected	Animals may have initial reluctance to drink or there may be some scouring, but stock should adapt without loss of production	Loss of production and a decline in animal condition and health would be expected. Stock may tolerate these levels for short periods if introduced gradually
Beef cattle	0–4000	4000–5000	5000–10 000
Dairy cattle	0–2500	2500–4000	4000–7000
Sheep	0–5000	5000–10 000	10 000–13 000[b]
Horses	0–4000	4000–6000	6000–7000
Pigs	0–4000	4000–6000	6000–8000
Poultry	0–2000	2000–3000	3000–4000

a From ANZECC (1992), adapted to incorporate more recent information

b Sheep on lush green feed may tolerate up to 13 000 mg/L TDS without loss of condition or production

Total dissolved solids (TDS) is a measure of all inorganic salts dissolved in water and is a guide to water quality. For convenience, TDS is often estimated from electrical conductivity (EC). An approximate conversion of EC to TDS is:

$$\text{EC (dS/m)} \times 670 = \text{TDS (mg/L)} \text{ or,} \quad (4.5)$$

$$\text{EC } (\mu\text{S/cm}) \times 0.67 = \text{TDS (mg/L)} \quad (4.6)$$

Salinity is used as a convenient guide to the suitability of water for livestock watering. If a water has purgative or toxic effects, especially if the TDS concentration is above 2400 mg/L, the water should be analysed to determine the concentrations of specific ions.

4.3.4 Heavy metals and metalloids

Many metal elements are essential nutrients for animal health, but elevated concentrations of certain compounds may cause chronic or toxic effects in livestock. Stock can tolerate many metal elements in drinking water if they are not ingesting them in quantity in the diet, because accumulation in the body depends on the amount ingested from both food and water sources. The trigger values in table 4.3.2 are the metal concentrations below which there is a minimal risk of toxic effects. If these values are exceeded the situation should be investigated further. In some cases higher concentrations may be tolerated, depending on factors such as total dietary exposure to the metal or levels of other compensating elements.[a] Unless otherwise stated, the trigger values relate to the total concentration of the constituent, irrespective of whether it is dissolved, complexed with an organic compound, or bound to suspended solids.

a See also Section 9.3.5

Table 4.3.2 Recommended water quality trigger values (low risk) for heavy metals and metalloids in livestock drinking water [a]

Metal or metalloid	Trigger value (low risk)[a,b] (mg/L)
Aluminium	5
Arsenic	0.5 up to 5[c]
Beryllium	ND
Boron	5
Cadmium	0.01
Chromium	1
Cobalt	1
Copper	0.4 (sheep) 1 (cattle) 5 (pigs) 5 (poultry)
Fluoride	2
Iron	not sufficiently toxic
Lead	0.1
Manganese	not sufficiently toxic
Mercury	0.002
Molybdenum	0.15
Nickel	1
Selenium	0.02
Uranium	0.2
Vanadium	ND
Zinc	20

a Higher concentrations may be tolerated in some situations (details provided in Volume 3, Section 9.3.5)

b ND = not determined, insufficient background data to calculate

c May be tolerated if not provided as a food additive and natural levels in the diet are low

4.3.5 Pesticides and other organic contaminants

In the absence of adequate information derived specifically for livestock under Australian and New Zealand conditions, it is recommended that the drinking water guidelines for human health be adopted.

A major concern in rural environments is the potential for pesticide residues to contaminate water supplies by spray drift, deep percolation, surface runoff, accidental spillage, or by direct application to water supplies for controlling aquatic weeds. In the absence of guidelines derived specifically for livestock, the reader is referred to the *Australian Drinking Water Guidelines* (NHMRC & ARMCANZ 1996). Readers in New Zealand are referred to the *Drinking-water Standards for New Zealand* (New Zealand Ministry of Health 1995a) and the *Guidelines for Drinking-water Quality Management for New Zealand* (New Zealand Ministry of Health 1995b).

4.3.6 Radiological quality of livestock drinking water

Trigger values for the radiological quality of livestock drinking water are given in table 4.3.3.

Table 4.3.3 Trigger values for radioactive contaminants in livestock drinking water

Radionuclide	Trigger value
Radium 226	5 Bq/L
Radium 228	2 Bq/L
Uranium 238	0.2 Bq/L
Gross alpha	0.5 Bq/L
Gross beta (excluding K-40)	0.5 Bq/L

Radioactive contaminants can originate from both natural and artificial sources and can potentially be found in surface waters and groundwaters. For livestock, the main water-related risks due to radioactivity arise from the transfer of radionuclides from irrigation or stock drinking water to animals and animal products for human consumption. Cancer is a potential health hazard for humans associated with exposure to radionuclides.

4.4 Aquaculture and human consumption of aquatic foods

4.4.1 Background

Aquaculture involves the production of food for human consumption, fry for recreational fishing and natural fisheries, ornamental fish and plants for the aquarium trade, raw materials for energy and biochemicals, and a number of items for the fashion industry. With wild fisheries approaching maximum sustainable levels and many already being over exploited, aquaculture is increasingly important worldwide as a source of aquatic food and other products.

During 1997–98, almost 31 000 tonnes of product and around 9.3 million juveniles (mostly finfish fry and ornamental fish) were produced in Australia at an estimated farm gate value in excess of $517.4 million (O'Sullivan & Roberts 1999). This represents approximately 25% of total aquatic food production in Australia. The pearl oyster, southern bluefin tuna, salmonid, edible oyster and prawn industries represent the major commercial aquaculture sectors economically, totalling more than 90% of overall aquaculture production.

The main culture species in New Zealand are green shell mussels, Pacific salmon and Pacific oysters. According to the New Zealand Fishing Industry (Treyton Maldoc, pers. com. 1999), annual production of these species totalled almost 50 000 tonnes, with an estimated value of around $160 million. Aquaculture now contributes over 13% of all New Zealand aquatic food exports.

Within the growing aquaculture industry, it is well accepted that satisfactory water quality is needed for maintaining viable aquaculture operations. Poor water quality can result in loss of production of culture species, and can also reduce the quality of the end product. Production is reduced when influent water contains enough contaminants to impair development, growth or reproduction, with the ultimate result being death. Quality is reduced when low levels of a contaminant cause no obvious adverse effects but gradually accumulate in the culture species to the point where it poses a potential health risk to human consumers. Thus, both these issues needed to be considered if useful and usable guidelines are to be provided for the aquaculture industry.

This section provides water quality guidelines for influent (i.e. water that is entering the aquaculture operation) or source water quality, and it also addresses the safety of aquatic foods for human consumers, whether the foods be produced by aquaculture, or commercial, or recreational or indigenous fishing. It is the first set of joint guidelines to have been provided for the protection of aquaculture in Australia and New Zealand. Note that these guidelines for protecting the health of commercial fish species[a] do not apply to recreational and commercial fisheries based upon wild populations of aquatic organisms. Wild fish stocks are dependent on healthy ecosystems to support them thoughout their life cycle (e.g. for feeding, breeding, habitat). Hence, for the protection of wild fish stocks it is best to apply the water quality guidelines for managing aquatic ecosystems.[b]

a See Section 4.4.4

b Chapter 3

4.4.2 Philosophy

In developing these guidelines, the objective was to provide information and guidance that would:

- promote the quality of water necessary for use by the aquaculture industry; and
- protect human consumers of harvested aquatic food species.

4.4.2.1 Protection of cultured fish, molluscs and crustaceans

The guidelines for protecting aquaculture species have been developed to assist water managers to maintain an appropriate level of water quality for existing and future aquaculture activities. The water quality guidelines will provide a basis for aquaculture management decisions, such as:

- environmental planning and management,
- environmental assessment and monitoring requirements,
- appropriate environmental zoning and legislation,
- appropriate species and suitable site selection,
- site capacity,
- farm design criteria,
- stocking densities and feeding regimes,
- production schedules.

4.4.2.2 Protection of human consumers of aquatic foods

Standards for the protection of human consumers of aquatic foods are of paramount importance to the viability of the aquaculture industry. To maintain demand, the aquaculture and fishing industries must ensure the highest quality of their products, both from a visual and, more importantly, from a human health perspective. Under a treaty between Australia and New Zealand (ANZFA 1996), the Australia New Zealand Food Authority (ANZFA) develops and administers uniform (statutory) standards for chemical contamination in foods (including aquatic foods) that are likely to affect human health. Unlike the water quality guidelines, the ANZFA food standards are enforceable through legislation. Guidelines are also provided in this section against biological contaminants and against the tainting of aquatic animal flesh.

4.4.3 Scope

As the aquaculture guidelines for Australia and New Zealand are a new development, they have drawn extensively on recent overseas guidelines for aquaculture as well as on the personal experiences of a number of local industry specialists. The guidelines address the following issues:

- protection of the health of culture species from water-borne contaminants (chemicals, elements, microorganisms, toxins, etc.) during the growing period (pre-harvest), but not during post-harvest processes (e.g. slaughter, processing, transport, marketing);
- the effects of water quality on adult forms of cultured species, recognising that larval and juvenile stages may have lower tolerance levels than the adult stages;
- the protection of human consumers of harvested aquatic food species from the toxic effects of chemical and biological contaminants and from tainted flesh.

The guidelines do not address effluent water quality from aquaculture activities; however, aquaculturists need to manage their operations with downstream water quality in mind. Effluent water quality is regulated by state and federal government legislation and regulations in Australia, and through the Resource Management Act and Industry Agreed Implementation Standards in New Zealand. In addition, as stated above, the guidelines in Section 4.4.4 are only concerned with the protection of cultured, not wild species.

Given the limited information on contaminant accumulation in aquaculture species, it has not been possible to provide water quality guidelines that will guarantee that the Australian and New Zealand food standards will be achieved. Therefore, the guidelines for the protection of human consumers of aquatic foods are intended to be used in conjunction with the *Food Standards Code* (ANZFA 1996, and updates) to protect the health of human consumers of aquatic foods from the aquaculture industry. These standards are continually under review and can be examined on the appropriate web sites (for Australia: www.anzfa.gov.au; for New Zealand: www.anzfa.govt.nz).

a See Section 9.4.1 for more detail

Precautionary comments and discussion on the limitations of the guidelines are provided below in Section 4.4.6.[a]

4.4.4 Water quality guidelines for the protection of cultured fish, molluscs and crustaceans

4.4.4.1 Overview of approach

There are many aquaculture species in Australia and New Zealand and information is generally lacking on most of them, so all finfish, mollusc and crustacean species were divided into eight indicative groups. Then toxicity and tolerance data were reviewed for one or two representative species within those groups, with the species being chosen according to the level of production and availability of scientific data. Where discrepancies in the data were identified, the more conservative data were generally used. The species groups and representative species are summarised in table 4.4.1.

Justification for selecting the representative species is provided in Section 9.4.1.4 (Volume 3). As indicated in table 4.4.1, a range of aquatic plants, reptiles and invertebrates that are cultured were not included in the list of representative species. In 1997/98 the production of these species contributed less than 1.5% of the total value of aquaculture production in Australia (O'Sullivan & Roberts 1999), with the amount of relevant literature or information about them being correspondingly small.

Guideline values were determined in several ways, depending on the quantity and quality of information. Where they were available, appropriate guidelines for the protection of aquaculture from other countries (e.g. DWAF 1996, Zweig et al. 1999) were applied. In some cases, guideline values were based on acceptable risks, according to the value judgements or professional judgements of local aquaculture specialists. When neither of the above approaches could be used, the water quality requirements for the eight indicative species groups were reviewed to determine a guideline value.[b] Discussion of the confidence levels for these guidelines is provided in Section 9.4.1.5 (Volume 3).

b Sections 9.4.1.4, 9.4.1.5

Table 4.4.1 Representative aquaculture species, occurrence and culture status

Species group	Representative species[1]	Occurrence	Aquaculture status[2]
Freshwater fish	rainbow trout	Australia/New Zealand	commercial/none
	silver perch	Australia	commercial
Marine fish	snapper	Australia/New Zealand	commercial/commercial
	flounder/whiting	Australia/Australia	experimental/experimental
Brackish water or euryhaline fish	barramundi	Australia	commercial
	black bream	Australia	experimental
Freshwater crustaceans	marron	Australia	commercial
	yabbies	Australia	commercial
	red claw	Australia	commercial
	freshwater shrimp	Australia/New Zealand	experimental/commercial
Marine crustaceans	black tiger prawns	Australia	commercial
	kuruma prawns	Australia	commercial
Edible bivalves	Sydney rock oysters	Australia	commercial
	Pacific oysters	Australia/New Zealand	commercial/commercial
	blue mussels	Australia/New Zealand	commercial/none
	green shell mussels	New Zealand	commercial
Pearl oysters	golden lip	Australia	commercial
Gastropod/molluscs	abalone/paua	Australia/New Zealand	commercial/commercial
	trochus	Australia	experimental

1 The groups of aquaculture species not included in this list are: seaweeds and aquatic plants; crocodiles; a range of live feed and microalgal species; sea cucumbers (beche-de-mer), sponges and other invertebrates.

2 commercial = products offered for sale; experimental = production but no sales; none = species occurs but no culture is undertaken

The guidelines are provided in the following four categories:

- physico-chemical stressors,
- inorganic toxicants,
- organic toxicants,
- pathogens and biological contaminants.

General guideline values for the aquaculture of freshwater and saltwater (brackish and marine water) are recommended. In addition, specific guideline values are provided for species groups for which information is available on their water quality requirements. Information sources used to derive the water quality guidelines for protection of aquaculture species are listed in Section 9.4.1.4 (Volume 3).

4.4.4.2 Using the guidelines

The water quality guidelines can be used with reasonable confidence to assess ambient water quality for aquacultural uses. Where specific water quality guidelines cannot be given for the protection of aquaculture species, use the guidelines for the protection of aquatic ecosystems.[a]

a See Chapter 3

Many different aquaculture production systems and species are used in Australia and New Zealand across a wide range of environmental conditions, so it should not be assumed that one set of specific values will apply equally in all situations. Local, site-specific information will be needed to supplement the broad information provided in this chapter. This might include information on specific culture species, or local water quality variables that could affect the bioavailability and toxicity of metals (e.g. hardness, dissolved organic matter, pH, temperature).

Details of factors that could affect toxicant bioavailability are provided in Section 8.3.5 (Volume 2).

Figure 4.4.1 is a decision tree for determining water quality guidelines for the protection of aquaculture species; it includes a number of factors that might modify the guideline values. Specialist assistance may be required to complete the steps which involve chemical speciation/complexation, and likewise to conduct toxicity tests should they become necessary.[a]

a See Section 3.4.3, Vol. 1; Section 8.3.6, Volume 2

Note that a user can make a decision on the risk-based framework and leave the process at any level. However, the further through the process one moves, the greater the confidence in the level of risk. A worked example of the use of the decision tree for an aquaculturist planning to culture prawns is provided in Section 9.4.2 (Volume 3).

If ambient water quality exceeds the guideline value for any parameter then there could be a significant risk of an impact on aquacultural activities, and further investigations should be undertaken, in accordance with the decision framework in figure 4.4.1. If ambient water quality remains below the guideline values, risk can be deemed to be low. However, this cannot be taken as a guarantee that problems will not occur in the future.

It is unrealistic to expect an aquaculture operation to measure all of the water quality parameters. However, knowledge of activities upstream of the operation that may be contributing to contaminants in the influent water should serve to identify which of the parameters might be of particular concern.

4.4.4.3 The guideline values

Tables 4.4.2 and 4.4.3 provide the recommended water quality guideline values for physico-chemical parameters and toxicants, respectively, to be applied for use in general freshwater and saltwater (brackish and marine water) aquaculture. Where guideline values are available for some or all of the species groups outlined in table 4.4.1, they have been incorporated in Section 9.4.2 (Volume 3), and can be used where guidance is sought for a particular species group. A short summary for each category (i.e. physico-chemical, inorganic, etc.) is also provided after the tables. Section 9.4.2 (Volume 3) also contains further background information on each water quality parameter, including a description of how the recommended guideline value was determined.

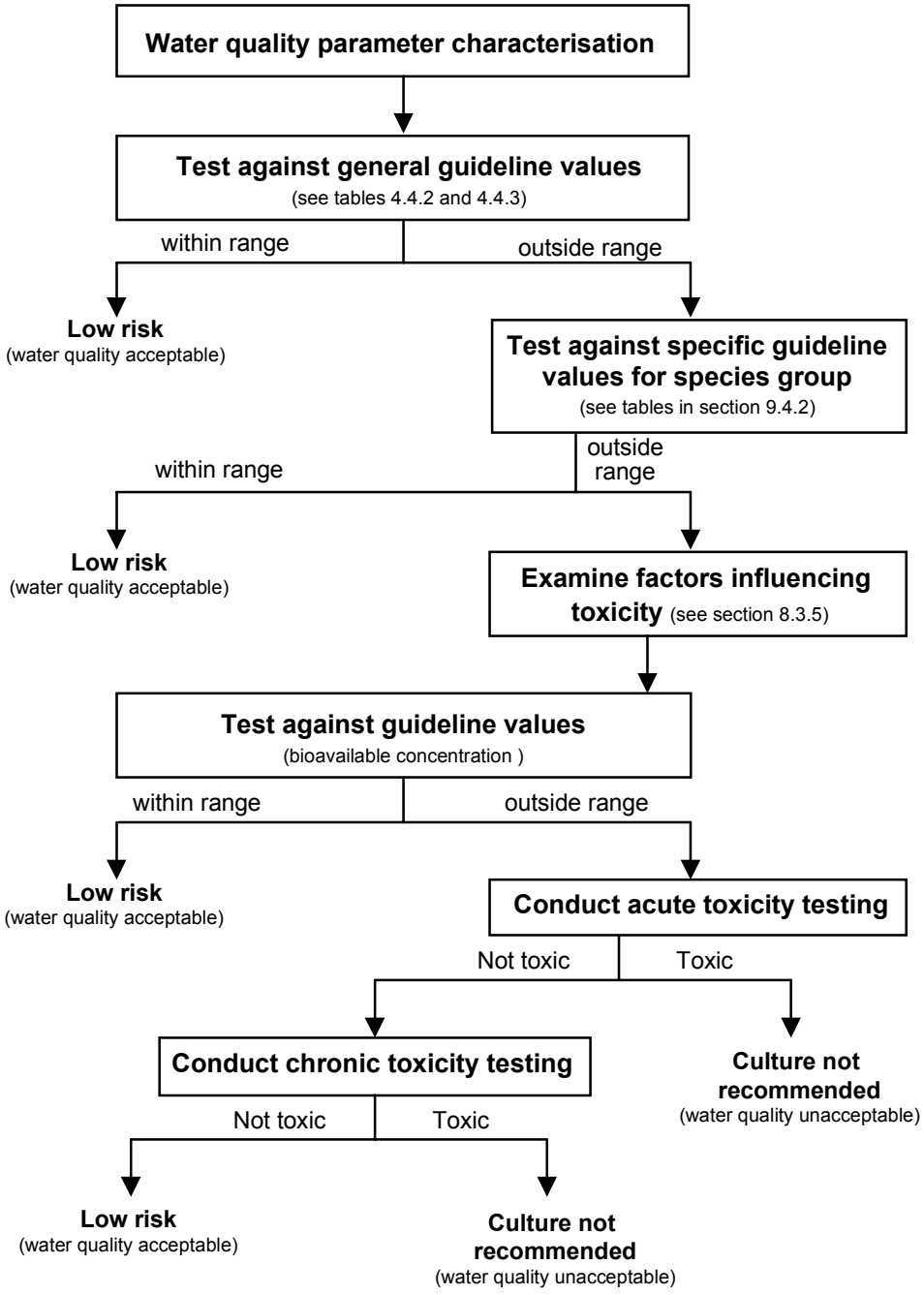


Figure 4.4.1 Decision tree for determining if water quality is acceptable for the protection of aquaculture species

Table 4.4.2 Physico-chemical stressor guidelines for the protection of aquaculture species

Measured parameter	Recommended guideline (mg/L)	
	Freshwater production	**Saltwater production**
Alkalinity	≥20[5]	>20[3]
Biochemical oxygen demand (BOD_5)	<15[1]	ND
Chemical oxygen demand (COD)	<40[1]	ND
Carbon dioxide	<10	<15
Colour and appearance of water	30–40[2] (Pt-Co units)	30–40[2] (Pt-Co units)
Dissolved oxygen	>5[3]	>5[3]
Gas supersaturation	<100%[6]	<100%[6]
Hardness ($CaCO_3$)	20–100[5]	NC[6]
pH	5.0–9.0	6.0-9.0
Salinity (total dissolved solids)	<3000[6]	33 000–37 000[6] (3000–35 000 Brackish)[6]
Suspended solids	<40	<10 (<75 Brackish)
Temperature	<2.0°C change over 1 hour[4]	<2.0°C change over 1 hour[4]

1 Schlotfeldt & Alderman (1995)

2 O'Connor pers. comm.

3 Meade (1989)

4 ANZECC (1992)

5 DWAF (1996)

6 Lawson (1995)

Others are based on professional judgements of the project team.

Table 4.4.3 Toxicant guidelines for the protection of aquaculture species

Measured parameter	Guideline (µg/L)	
	Freshwater production	**Saltwater production**
INORGANIC TOXICANTS (HEAVY METALS AND OTHERS)		
Aluminium	<30 (pH >6.5)[1] <10 (pH <6.5)	<10[1]
Ammonia (un-ionised)	<20 (pH >8.0) coldwater[2] <30 warmwater[2]	<100
Arsenic	<50[1,2]	<30[1,2]
Cadmium (varies with hardness)	<0.2–1.8[2]	<0.5–5[1]
Chlorine	<3[1]	<3[1]
Chromium	<20[2]	<20
Copper (varies with hardness)	<5[2]	<5[3]
Cyanide	<5[1]	<5[1]
Fluorides	<20[4]	ND
Hydrogen sulfide	<1[2]	<2
Iron	<10[1]	<10[1]
Lead (varies with hardness)	<1–7[4]	<1–7[4]
Magnesium	<15 000[1]	ND
Manganese	<10[1,5]	<10[1,5]
Mercury	<1	<1
Nickel	<100[1]	<100[1]
Nitrate (NO_3^-)	<50 000[6]	<100 000[3,7]
Nitrite (NO_2)	<100[1,7]	<100[1,7]
Phosphates	<100[2]	<50
Selenium	<10[1]	<10[1]
Silver	<3[1]	<3[1]
Tributyltin (TBT)	<0.026[1]	<0.01[1]
Total available nitrogen (TAN)	<1000[1]	<1000[1]
Vanadium	<100[1]	<100[1]
Zinc	<5[1]	<5[1]
ORGANIC TOXICANTS (NON-PESTICIDES)		
Detergents and surfactants	<0.1[8]	ND
Methane	<65 000[9,10]	<65 000[9,10]
Oils and greases (including petrochemicals)	<300[6]	ND
Phenols and chlorinated phenols	<0.6–1.7[6]	ND
Polychlorinated biphenyls (PCBs)	<2[1]	<2[1]
PESTICIDES		
2,4-dichlorophenol	<4.0[2]	ND
Aldrin	<0.01[2,3,8]	ND
Azinphos-methyl	<0.01[2]	ND
Chlordane	<0.01[11]	0.004[11]
Chlorpyrifos	<0.001[2]	ND
DDT (including DDD & DDE)	<0.0015[2]	ND
Demton	<0.01[11]	ND
Dieldrin	<0.005[2]	ND
Endosulfan	<0.003[2,11]	0.001[11]
Endrin	<0.002[2]	ND
Gunthion (see also Azinphos-methyl)	<0.01[11]	ND
Hexachlorobenzole	<0.00001[6]	ND
Heptachlor	<0.005[2]	ND
Lindane	<0.01[11]	0.004[11]
Malathion	<0.1[5,11]	ND
Methoxychlor	<0.03[11]	ND
Mirex	<0.001[2,11]	ND
Paraquat	ND	<0.01
Parathion	<0.04[11]	ND
Toxaphene	<0.002[2]	ND

ND: Not determined — insufficient information; NC: Not of concern; 1. Meade (1989); 2. DWAF (1996); 3. Pillay (1990); 4. Tebbutt (1972); 5. Zweig et al. (1999); 6. Schlotfeldt & Alderman (1995); 7. Coche (1981); 8. Langdon (1988); 9. McKee & Wolf (1963); 10. Boyd (1990); 11. Lannan et al. (1986). Others are based on professional judgements of the project team.

1 Physico-chemical stressors

A number of naturally-occurring physico-chemical stressors can cause adverse effects on aquaculture operations when influent water values are too high and/or too low. These guidelines address 11 physico-chemical stressors that are considered of importance to aquaculture operations. Many of these should also be regularly monitored in the culture system to ensure that the aquatic organisms are being held in conditions conducive to survival and growth. Some of the major stressors are summarised below.[a]

a See also Section 9.4.2.1

Dissolved oxygen (DO) is a basic requirement for aquaculture species (Zweig et al. 1999). The amount of oxygen required by aquatic animals is quite variable and depends on species, size, activity, water temperature, condition, and the DO concentration itself (Boyd 1990). Thus, some species are more sensitive to low levels of oxygen than others. Daily fluctuations of DO in impounded waters are much higher than those in the open sea or running waters, with low levels often occurring at dawn, and high levels in the late afternoon (Boyd 1990). The most common cause of low DO levels in an aquaculture operation is contamination by biodegradable organic substances resulting in a high BOD; the problem is further exacerbated at higher temperatures.

Water hardness, a total measure of the major cations (predominantly calcium and magnesium), is an important parameter in freshwaters, mostly because it can have a major effect on the toxicity of metals. In addition, some aquaculture species have specific calcium requirements for bone or exoskeleton formation, and calcium is also necessary for proper osmoregulation. Water hardness (measured as mg $CaCO_3$/L) can range from <1 (very soft) to >400 mg/L (very hard).

The pH of influent water refers to the $\log_{10}$ of the hydrogen ion concentration, or, more simply, how acidic or basic the water is. The pH is interdependent with a number of other water quality parameters including carbon dioxide, alkalinity and hardness. It is known to influence the toxicity of hydrogen sulfide, cyanides, heavy metals, and ammonia (Klontz 1993), and it can also be toxic in its own right. The pH levels in natural waters vary enormously and the aquaculturist should ensure that culture species are adapted to living in the conditions existing in the aquaculture operation.

Salinity is an important limiting factor in the distribution of many aquatic animals, and therefore it is an important parameter for aquaculture. In addition, salinity requirements can vary for particular species depending on their life cycle stage. Outside their natural salinity ranges, aquatic animals must expend considerable energy on osmoregulation at the expense of other processes such as growth. Salinity ranges are 0.05–1.0 gL^{-1} for freshwaters, 0.5–>30 gL^{-1} for estuarine waters, 30–40 gL^{-1} for marine waters, and can exceed 40 gL^{-1} for hypersaline/brackish waters.

Suspended solids and turbidity can have major effects on aquaculture operations. Suspended solids include phytoplankton, zooplankton and bacterial blooms, suspended organic and humic acids, and suspended silt and clay particles. All these components contribute to some extent to increased turbidity. In some instances this is advantageous, because it inhibits the growth of nuisance algae and macrophytes. However, suspended solids can cause gill irritations and tissue damage to aquatic animals, while they can also shield food organisms and clog filters (Zweig et al. 1999). Smothering effects caused by suspended solids settling on sessile

aquaculture species (e.g. mussels, oysters) can also present problems (Duchrow & Everhart 1971).

In summary, it should be highlighted that physico-chemical parameters vary widely in natural waters, and aquatic organisms have a wide range of tolerances and adaptive capacities. Thus, it is extremely difficult to recommend broadly applicable guidelines.

2 Inorganic toxicants (heavy metals and others)

A wide range of inorganic toxicants, particularly heavy metals, can be a problem in freshwater, brackish water and inshore marine aquaculture, especially in areas of human habitation that may be polluted. Trace quantities of metals are present in natural waters; however, their concentrations are generally greater in the vicinity of industrial processes (ore mining and processing, smelting plants, rolling sheet metal mills, textile and leather industries) and exhaust gases of motor vehicles and burning of other fossil fuels. These guidelines provide information on 27 inorganic toxicants. Those of greatest concern to fisheries (including aquaculture) include aluminium, arsenic, cadmium, chromium, copper, iron, lead, mercury, nickel and zinc (Svobodova et al. 1993). Other inorganic toxicants include ammonia, chlorine, cyanide, fluoride, hydrogen sulfide, nitrite, nitrate and phosphates. As mentioned above, the levels of calcium and magnesium are also important because they influence the hardness of the waters.[a]

a See Section 9.4.2.1/7

Speciation of metals is important in determining toxicity to aquatic organisms because it influences metal bioavailability. Water quality guidelines for metals in aquatic ecosystems have typically been based on total concentrations; yet it is now well established that the chemical form or speciation of metals critically influences their bioavailability (i.e. their ability to penetrate a biological cell membrane) and toxicity to aquatic organisms.[b]

b Sections 8.3.5.16 and 9.4.2.2

Most studies of the toxicity of heavy metals to fish and other aquatic organisms have shown that the free (hydrated) metal ion is the most toxic form, and that toxicity is related to the activity of the free (dissolved) metal ion (e.g. Cu^{2+} or Zn^{2+}) rather than to total metal concentration (including adsorbed, chelated or complexed forms) (Florence & Batley 1988, Boyd 1989). Heavy metal toxicity also can be affected by pH, hardness, alkalinity, dissolved oxygen, temperature and turbidity (SECL 1983). In pond water, heavy metals can be adsorbed onto clay particles and chelated by organic matter so that they remain in solution but may not have an adverse effect on fish or crustaceans (Boyd 1990). Duration of exposure, interaction with other toxic agents and species can affect the biological response to these toxic metals significantly (e.g. mercury and methane give rise to methyl mercury).

Guidelines based on total concentrations may be over-protective, since only a fraction of the total concentration will generally be bioavailable, especially in samples containing appreciable concentrations of particulate matter. Thus, it is important to measure the bioavailable metal fraction.[c] Importantly, Svobodova et al. (1993) noted that the toxic action of metals is particularly pronounced in the early stages of development of the fish.

c Section 8.3.5

3 Organic toxicants

Organic toxicants can present a problem to all types of aquaculture operations. The types of organic chemicals considered in these guidelines are detergents and surfactants, hydrocarbons derived from human activities (namely petroleum

hydrocarbons), a large number of pesticides, phenolic compounds, and polychlorinated biphenyls. Most of these originate from domestic, agricultural or industrial activities, and some are also used by aquaculture operations.

No data were available to provide guidelines for antibiotics and antimicrobials, but it is best to take due care when using such chemicals in aquaculture operations.

Detergents and surfactants are widely used in domestic and industrial operations, and can often be detected in natural waters receiving domestic and industrial effluent (Svobodova et al. 1993), while on-farm activities may also be major sources of such chemicals. There is limited toxicity information for detergents and surfactants, although a general guideline value was derived for freshwaters.

Petroleum hydrocarbons are among the most widely processed and distributed chemical products in the world (Zweig et al. 1999). Although high levels of petroleum hydrocarbons can result in mortalities and major losses of production, the major concern to the aquaculture industry is the tainting of culture animals with off-flavours (Zweig et al. 1999[a]). Given the large number of petroleum-derived hydrocarbons and their wide ranges of toxicities, it is difficult to derive meaningful guidelines (SECL 1983), although some general guidelines have been recommended.

a Also see Section 4.4.5.3/3 below

The pesticides represent a large and complex group of organic toxicants because they incorporate insecticides, acaricides, herbicides, algicides and fungicides. In addition, the behaviour (e.g. persistence, partitioning) and toxicity of pesticides varies greatly, making it difficult to generalise about risks. Pesticides generally enter water from sources in the primary industry sector, including aquaculture, but primarily agriculture. Table 4.4.2 presents guideline values only for those pesticides for which a general freshwater or saltwater value can be recommended. A more comprehensive list of pesticide guideline values for specific species groups is provided in Section 9.4.2.3/4 (Volume 3). Given the limited information on the effects of pesticides on culture species, it is also worthwhile consulting the guidelines for aquatic ecosystem protection.[b]

b Chapter 3, Volume 1

Other organic compounds of concern include phenols and polychlorinated biphenyls (PCBs). Phenolic compounds originate from the distillation of fossil fuels, the degradation of pesticides, natural (SECL 1983) and other sources. They can result in effects ranging from toxicity to the tainting of flesh. Guideline values are recommended for freshwater and saltwater, while some guideline values for specific phenols are recommended for freshwater fish culture. The PCBs are extremely persistent lipid soluble chemicals that are of great environmental concern (Svobodova et al. 1993). It is extremely difficult to recommend guidelines for PCBs because of their large number and the wide spectrum of toxicity they exhibit. However, general guideline values are recommended for freshwater and saltwater.

4 Pathogens and biological contaminants

Pathogens and biological contaminants also need to be considered for aquaculture operations, and include algal blooms and algal toxins, bacteria, viruses and parasites. As noted by Zweig et al. (1999), high concentrations of pathogenic organisms are commonly found in waters polluted by human sewage and animal wastes. No guidelines are provided for pathogens and biological contaminants because their effects can vary considerably between the type of contaminant or species of pathogen, and the culture species. Nevertheless, Section 9.4.2.4

(Volume 3) provides useful background information and some guidance on how to manage for them. Brief summary information is provided below.

Algal blooms arise from a series of processes but commonly from eutrophication (addition of excess nutrients). Direct and indirect results of algal blooms include increased pH, depleted oxygen (anoxia), the production and release of algal toxins, and gill obstruction and irritation in fish. Algal toxins can also accumulate in culture species, resulting in potential risks to human consumers.[a]

a See Section 4.4.5.3/2

It has been suggested that culture organism mortality due to disease poses a more direct threat to the aquaculture industry than pollutants (Handlinger 1996). Aquaculture source waters contain a certain number of bacteria, viruses, fungi, parasites and other organisms, which, given certain environmental conditions, can contribute to impaired health of the culture species. Thus, the maintenance of optimal water quality appears to be the best defence against infections by these organisms (DWAF 1996). Some equipment that reduces the amount of incoming potential pathogens includes inflow filters that retain particles (to which most of the bacteria will be attached) and ultra-violet (UV) sterilisers. Reducing the level of infectious organisms contributes to better culture health, reduced need to treat animals with chemicals and drugs, and lower production costs.

4.4.5 Water quality guidelines for the protection of human consumers of aquatic foods

4.4.5.1 Overview of approach

Although guidelines are provided for biological contaminants and for the tainting of animal flesh, a search of the available data has produced insufficient information for deriving water quality guidelines that will ensure the Australian and New Zealand food standards will be met. Consequently, relevant food standards from the *Food Standards Code* (ANZFA 1996, and updates) established by the Australia New Zealand Food Authority (ANZFA) are provided as guidance and discussed below.[b]

b Section 9.4.3

4.4.5.2 Using the guidelines

The guidelines for the protection of human consumers of aquatic foods are intended to be used in conjunction with the *Food Standards Code* (ANZFA 1996, and updates) to protect the health of human consumers of aquatic foods from the effects of toxicants, whether the foods be derived from aquaculture, recreational fishing, commercial fishing or indigenous fishing. Essentially, they provide useful background information and some guidance to complement the ANZFA food standards. In particular, they give detailed information on measures for predicting the tissue concentrations of contaminants before, rather than after, harvest. Such approaches may form the basis for the future development of guidelines for the protection of human consumers of aquatic foods.

The ANZFA food standards for contamination of aquatic foods are enforceable through legislation and must be adhered to. However, it is important to note that at the time of publication of these Water Quality Guidelines, the ANZFA food standards were under review and subject to change. Thus, aquaculturists and other users of these guidelines should ensure they obtain the most recent ANZFA information (for Australia: www.anzfa.gov.au; for New Zealand: www.anzfa.govt.nz).

4.4.5.3 The guidelines

The food standards developed by ANZFA and published in the *Food Standards Code* (ANZFA 1996, and updates) aim to protect consumers from chemically contaminated foods, including aquatic species. Standards for aquatic species are based on the notion of acceptable daily intake (ADI) or acceptable weekly intake (AWI). See Zweig et al. (1999) for the World Health Organization (WHO) provisional tolerable weekly intakes for selected elements, as well as import regulations for residues. Guidelines are also provided for biological contaminants and for the tainting of animal flesh.

1 Chemical contaminants (toxicants)

Chemical contaminants can be categorised into three broad groups:[a]

a See Section 9.4.3.2 (Vol. 3) for ANZFA standards

i) Inorganic toxicants (mostly heavy metals)

Inorganic toxicants (mainly heavy metals) are a potential problem for human health, particularly through bivalved molluscs in which bioaccumulation increases the concentrations of inorganic toxicants. The rate of accumulation is species-specific and depends on the mechanisms of absorption and tissue distribution.

ii) Organic toxicants (e.g. hydrocarbons, pesticides)

The broad group comprising organic toxicants such as hydrocarbons and pesticides includes synthetic compounds which through either bioaccumulation or residue concentrations are potentially toxic to human consumers of contaminated aquatic foods.

iii) Radionuclides (radioactive elements)

At present, ANZFA does not specify maximum permitted concentrations (MPCs) for radionuclides in edible tissues. Many countries have limits set on imported foods, particularly for caesium-137 (Cs-137). Environmental levels of Cs-137 are considerably lower in the southern hemisphere than in the northern hemisphere, and exporters in Australia and New Zealand should not generally experience difficulty in meeting such limits.

2 Biological contaminants

There are a number of biological contaminants that can affect human consumers of aquatic foods. The guidelines for biological contaminants are based on either a concentration of the contaminant in the water (e.g. cells/L) or the level which is considered safe in edible soft tissue of fish, crustaceans and molluscs (e.g. mg/kg, number/g). Summary information on the major biological contaminants is provided below.[b]

b Section 9.4.3.3 for ANZFA standards

i) Bacteria

Aquatic bacterial food-borne diseases in humans can originate either from bacteria naturally present in water and/or sediments, or from bacteria introduced into aquatic environments through human and/or animal faeces. Aquatic foods can become contaminated with bacteria from exposure within the aquatic environment and/or during post-harvest activities. The present guidelines only deal with exposure within the aquatic environment.

The guidelines in table 4.4.4 are provided to assist managers to minimise the exposure of human consumers of aquatic food species (e.g. recreational fishermen) to bacterial borne disease.

Table 4.4.4 Guidelines for the protection of human consumers of fish and other aquatic organisms from bacterial infection

Toxicant	Guideline *in shellfishing water*	Standard *in edible tissue*
Faecal (thermotolerant) coliforms	The median faecal coliform bacterial concentration should not exceed 14 MPN/100 mL, with no more than 10% of the samples exceeding 43 MPN/100 mL	Fish destined for human consumption should not exceed a limit of 2.3 MPN *E. coli* /g of flesh with a standard plate count of 100 000 organisms/g

MPN: Most probable number
The guideline for faecal (thermotolerant) coliforms should only be used in conjunction with the data from a sanitary survey of the shellfish harvesting areas for the purpose of harvesting area classification. Source: USEPA (1986), NAS/NAE (1973), IWBDE (1972).

A two-tiered approach is usually used to reduce bacterial loads in cultured species:

a See Section 4.4.5.3/4

- risk-based classification of waters to allow only certain waters and times for rearing or harvesting of shellfish;[a]
- treatment of shellfish to remove or destroy the bacteria (e.g. heat treatment or irradiation).

b Section 9.4.3.3/1

Depuration is an integral part of removing bacteria from shellfish, and is a statutory requirement in NSW only.[b]

ii) Viruses

Viruses that infect humans following consumption of aquatic food are of human origin, having entered aquatic ecosystems in sewage effluent. These enteric viruses are able to remain viable in the aquatic environment for long periods (Goyal et al. 1984).

c See part 4 below & Section 9.4.3.3/2

Shellfish are able to accumulate viruses in their gastrointestinal tracts, digestive glands and other tissues, but the rate of accumulation is dependent on the viral species and the shellfish species. Viruses are very difficult to detect, and other species (e.g. *Escherichia coli*, faecal coliforms) are usually used to indicate exposure to sewage-related pollution. While such sanitary surveys may not be as reliable as once thought, they are still relevant and are used in Australia and New Zealand as well as a number of other countries.[c]

Heat treatment and depuration are generally not as efficient at reducing viral loads as they are bacterial loads. Normal cooking/steaming times for shellfish may not be sufficient to inactivate viruses (University of California, Davis 1997). Similarly, depuration may not remove all viruses from shellfish (Jackson & Ogburn 1998).

iii) Parasites

There is no evidence of transmission of parasites to humans following aquatic food consumption in Australia or New Zealand. Thus, no guidelines are provided. However, the presence of parasites, cysts and necrotic tissue resulting from parasitic infections will reduce the marketability of product.

iv) Marine biotoxins

A number of marine biotoxins, most of them associated with marine algae, represent a threat to human consumers of aquatic foods. Aquatic animals accumulate the toxins when they graze on the algae or on other consumers of the algae.

There are five recognised types of microalgal toxins:

- paralytic shellfish poisoning (PSP),
- diarrhetic shellfish poisoning (DSP),
- amnesic shellfish poisoning (ASP),
- neurotoxic shellfish poisoning (NSP),
- ciguatera fish poisoning (CFP).

Three naturally-occurring toxins that are not related to algae are (University of California, Davis 1997):

- gempylotoxin,
- tetramine,
- tetrodotoxin.

Important background information on the above biotoxins is provided in Section 9.4.3.3 (Volume 3), including guidelines for water and standards for edible tissue (MBMB 1996, K Jackson pers. comm. 2000). For a detailed discussion of biotoxins in New Zealand, refer to MBMB (1996). University of California, Davis (1997) also provides useful guidance and background information.

3 Off-flavour compounds

Off-flavour compounds, also known as tainting substances, can seriously affect the palatability of aquatic food. They can result in major adverse impacts to the aquaculture and wild-capture fishing industries. Table 4.4.5 lists threshold concentrations at which tainting will occur for a variety of off-flavour compounds.[a]

a See also Section 9.4.3.4

Table 4.4.5 Guidelines for chemical compounds in water found to cause tainting of fish flesh and other aquatic organisms

Parameter	Estimated threshold level in water (mg/L)
Acenaphthene	0.02
Acetophenone	0.5
Acrylonitrile	18.0
Copper	1.0
m-cresol	0.2
o-cresol	0.4
p-cresol	0.1
Cresylic acids (meta, para)	0.2
Chlorobenzene	0.02
n-butylmercaptan	0.06
o-sec. butylphenol	0.3
p-tert. butylphenol	0.03
o-chlorophenol	0.0001–0.015
p-chlorophenol	0.0001
2,3-dinitrophenol	0.08
2,4,6-trinitrophenol	0.002
2,4-dichlorophenol	0.0001–0.014
2,5-dichlorophenol	0.02
2,6-dichlorophenol	0.03
3,4-dichlorophenol	0.0003
2-methyl-4-chlorophenol	2.0
2-methyl-6-cholorophenol	0.003
3-methyl-4-chlorophenol	0.02–3.0
o-phenylphenol	1.0
Pentachlorophenol	0.03
Phenol	1.0–10.0
Phenols in polluted rivers	0.15–0.02
2,3,4,6-tetrachlorophenol	0.001
2,3,5-trichlorophenol	0.001
2,4,6-trichlorophenol	0.002
2,4-dimethylphenol	0.4
Dimethylamine	7.0
Diphenyloxide	0.05
B,B-dichlorodiethyl ether	0.09–1
o-dichlorobenzene	<0.25
Ethylbenzene	0.25
Ethanethiol	0.2
Ethylacrylate	0.6
Formaldehyde	95.0
Gasoline	0.005
Guaicol	0.08
Kerosene	0.1
Kerosene plus kaolin	1.0
Hexachlorocyclopentadiene	0.001
Isopropylbenzene	<0.25
Naphtha	0.1
Naphthalene	1.0
Naphthol	0.5
2-Naphthol	0.3
Nitrobenzene	0.03
a-methylstyrene	0.25
Oil, emulsifiable	>15.0
Pyridine	5–28
Pyrocatechol	0.8–5
Pyrogallol	20–30
Quinoline	0.5–1
p-quinone	0.5
Styrene	0.25
Toluene	0.25
Outboard motor fuel as exhaust	7.2
Zinc	5.0

Source: Reproduced from ANZECC (1992), an adaptation of NAS/NAE (1973)

According to Zweig et al. (1999), sophisticated analytical equipment is usually not necessary for detecting tainting substances; water that tastes or smells unusual may result in off-flavours, and sensory assessments (i.e. taste, smell) are often preferable to chemical analyses.

In addition to the chemical contaminants, a number of freshwater blue-green microalgae and bacteria can cause off-flavours in native fish. The most common is the earthy or musty flavour often referred to as 'muddy' taste, which often occurs in silver perch (*Bidyanus bidyanus*). Decaying organic matter can also cause off-flavour. The incidence of off-flavours is highest in warmer months, during blooms of blue-green algae and in ponds with high stocking and feeding rates. Most off-flavours can be readily purged by placing fish in clean water such as underground or spring water, domestic (dechlorinated) or rainwater.

4 Preventative and management approaches

It is generally accepted that food species should not be grown in, or harvested from, waters likely to be exposed to contamination. If a contamination event should occur, the aquatic organisms should be regularly analysed to ensure that the ANZFA standards are not exceeded in harvested product. However, chemical analysis for the detection of contaminants in aquatic food can be an expensive process. For planning purposes a method of product quality prediction would be preferable. This problem may be illustrated by the following examples:

- The viability of the setup of an aquaculture business is being investigated. How can the investors predict whether, on harvesting, the product will be suitable for sale for human consumption?
- It is proposed to start up an industrial/sewage plant upstream of a commercial fishery. How can we predict whether effluent from the plant will have a significant adverse effect on the fishery product quality?

Section 9.4.3.5 (Volume 3) provides detailed information and guidance on several approaches for predicting water quality or safety of the aquatic food product. Due to the complexities involved, uncertainties will be associated with any prediction. Predictions cannot replace product testing, but they may enable problems to be identified and resolved before they affect an industry. Summaries of four predictive approaches are provided below.

i) Bioconcentration factor approach

Bioaccumulation can be predicted using the bioconcentration factor approach. Since circumstances will vary enormously from case to case, this approach is only intended as a general guide, not as a set of prescriptive rules; it has several limitations. The underlying principle of the bioconcentration factor approach is that where the uptake of a chemical is not controlled by the organism's metabolism, a concentration of the chemical in the organism will be proportional to the concentration of the chemical in the water or food (or sediment).

ii) Area classification approach

The area classification approach is used by the Australian Shellfish Quality Assurance Program (ASQAP) and the New Zealand Shellfish Quality Assurance Program (NZSQAP) to identify safe shellfish-growing areas to permit commercial harvesting for the domestic market and/or for export. The programs provide a risk-

based system of procedures and guidelines for regulating shellfish-growing areas, harvesting, processing and distribution of shellfish. In general, they cover:

- classification and survey of growing areas,
- relaying (relocation) and harvesting controls,
- post-harvest handling, storage, processing and transportation.

The shellfish harvesting area classification systems rely on the Sanitary Survey approach to ensure that molluscan shellfish harvested for human consumption are safe. The Sanitary Survey consists of:

- the identification and evaluation of all potential and actual pollution sources (i.e. Shoreline Survey),
- the monitoring of growing waters and shellfish to determine the most suitable classification for the shellfish harvesting area (i.e. Bacteriological Survey).

The categories of classification are based on levels of contamination from sewage, poisonous or deleterious substances, other pathogenic organisms of non-faecal origin and biotoxin-producing organisms, radionuclides, and toxic wastes (ASSAC 1997). A number of classifications can result from the Sanitary Surveys, but they differ slightly between countries.[a]

a See Section 9.4.3.5/2

iii) Phytoplankton monitoring

The purpose of phytoplankton monitoring is to predict marine biotoxins in shellfish. In New Zealand, phytoplankton monitoring is mandatory for all commercial harvested areas under the marine biotoxin monitoring program, while a similar program is operated by the Ministry of Health for all recreational shellfish harvesting sites. A combination of phytoplankton and flesh tests are used to monitor for biotoxin activity. Commercial areas are sampled weekly for biotoxin activity and if mandated trigger values are reached for a number of species, flesh testing is invoked immediately. Little such monitoring is undertaken in Australia.

Trigger values for a number of phytoplankton species under the New Zealand program (MBMB 1996) are provided in Section 9.4.3.5/3 (Volume 3).

iv) Three-phased screening approach

The three-phased screening approach is a tiered process designed for aquaculture operations to evaluate source water quality in a step-by-step process of increasing detail and complexity, in order to minimise costs (Zweig et al. 1999). Phase I screening involves the analysis of basic physico-chemical properties necessary to sustain culture species. Phase II is designed to screen source water for anthropogenic contaminants (chemical and biological). Phase III involves field assessments of the capacity of the source water to culture the selected species, using management/culture techniques similar to those of the proposed operation (i.e. a pilot study).[b]

b Section 9.4.3.5/4

4.4.6 Some precautionary comments

Section 9.4.4 (Volume 3) provides a detailed discussion of the limitations of the current guidelines for the protection of aquaculture species and human consumers of aquatic foods, and it is strongly recommended that it be read. A brief summary of the major issues is given below.

Two of the major limitations of the current guidelines are the lack of data and the variability of the data. Data variability can be attributed to several factors, including the use of different test methods (e.g. time and duration of exposure, size and age of fish, test conditions) over time, and analytical advances over time. Where differences in acceptable or tolerated concentrations are extreme between different guideline documents, it is suggested that the general/recommended guideline value provided in the current guidelines be applied, exercising some caution.

To relate laboratory toxicity studies to aquaculture operations is not a straightforward process. Many of the limitations and uncertainties are similar to those that apply when extrapolating laboratory toxicity data to natural aquatic ecosystems.[a] Some that are more specific to aquaculture operations include:

a See Section 9.4.4 for more detail

- aquaculture environments possess very different characteristics to natural environments (e.g. avoidance is not an option, feed is often derived from external sources, culture species may be regularly handled, stocking densities may be higher than in natural environments);
- very few ecotoxicological studies test aquaculture species;
- tolerance to individual contaminants is very variable between aquaculture species, even within the species groups outlined in table 4.4.1;
- toxicity test durations (i.e. usually ≤96 h) are not applicable to aquaculture operations, where organisms are constrained to an area and particular water quality for periods longer than toxicity test durations.

4.4.7 Priorities for research and development

As these guidelines are the first synthesis of water quality information for the aquaculture industry in Australia and New Zealand, a substantial number of information gaps and research needs have been identified. These are described in full in Section 9.4.5 (Volume 3).

5 Guidelines for recreational water quality and aesthetics

Water-based recreational activities are popular with Australians and New Zealanders. Although each country has an extensive coastline, much of it is inaccessible for recreational purposes, resulting in highly localised pressures on accessible coastline. The same is true for estuarine and freshwater rivers and lakes, especially those close to urban centres. Therefore, water quality guidelines are necessary to protect these waters for recreational activities such as swimming and boating, and to preserve the aesthetic appeal of water bodies. Water quality guidelines are used in the monitoring and management of a range of microbiological, physical and chemical characteristics that determine the suitability of a water resource for recreational purposes.

5.1 Guidelines for users in New Zealand

In New Zealand, water managers should refer to *Recreational Water Quality Guidelines* (NZ Ministry for the Environment 1999). This document and the draft supporting manual can be downloaded from:

http://www.mfe.govt.nz/about/publications/water_quality/beaches-guidelines.htm

The revised New Zealand guidelines were trialed over the 1999/2000 bathing season. This trial period will be followed by a consultation round similar to that carried out for the 1998 *Bacteriological Water Quality Guidelines for Marine and Fresh Water*. The extent of further revisions, if any, will depend upon the response to the revised guidelines. Any recommendation to the Minister for the Environment regarding a National Environmental Standard will be made after the round of consultation.

5.2 Guidelines for users in Australia

The material for Australian users of *Guidelines for Recreational Water Quality and Aesthetics* is currently being prepared. When completed, it will replace this section, in accordance with NWQMS requirements and National Health and Medical Research Council (NHMRC) statutory procedures. The NHMRC, ANZECC and ARMCANZ all recognise the need for a single guideline document to supplant earlier sets of guidelines for recreational water quality, published separately by the NHMRC and NWQMS (*Australian Guidelines for Recreational Use of Water* (NHMRC 1990) and ANZECC (1992) respectively).

It is intended that the new guidelines should be largely based on recommendations from the World Health Organization (WHO) including draft WHO *Guidelines for Safe Recreational-water Environments: Coastal and Fresh-waters* (WHO 1998) and WHO *Health-based Monitoring of Recreational Waters: The Feasibility of a New Approach (The 'Annapolis' Protocol)* (WHO 1999). These documents will provide the impetus to develop a single Australian guideline document. It will be part of the revised NWQMS Guidelines and will also be available as a separate NHMRC/ARMCANZ/ANZECC publication. The basis of the proposed guidelines for recreational water quality and aesthetics in Australia is provided in Appendix 5.

Until these Guidelines are revised and endorsed, users should apply the guidelines from the *Australian Water Quality Guidelines for Fresh and Marine Waters* (ANZECC 1992). These guidelines are reproduced below. While these (1992) guidelines are interim, the eventual guidelines that result from the NHMRC's current revision will be the definitive guidelines.

5.2.1 Introduction

Recreational guidelines accommodate two categories of sporting activity:

- sports in which the user comes into frequent direct contact with water, either as part of the activity or accidently; for example, swimming or surfing (primary contact);
- sports that generally have less-frequent body contact with the water; for example, boating or fishing (secondary contact).

A third recreational category concerns the passive recreational use of waterbodies, mainly as pleasant places to be near or to look at (no body contact). The relevance of the different water quality guidelines to the three recreational categories is shown in table 5.2.1. The detailed water quality guidelines for recreational water are summarised in table 5.2.2.

Table 5.2.1. Water quality characteristics relevant to recreational use

Characteristics	Primary contact (e.g. swimming)	Secondary contact (e.g. boating)	Visual use (no contact)
Microbiological guidelines	x	x	
Nuisance organisms (e.g. algae)	x	x	x
Physical and chemical guidelines:			
Aesthetics	x	x	x
Clarity	x	x	x
Colour	x	x	x
pH	x		
Temperature	x		
Toxic chemicals	x	x	
Oil, debris	x	x	x

The first part of this section on Australian guidelines provides a brief summary of the most important aspects of the above categories, while the second section contains details on the specific guidelines. Many of the guidelines necessary for the maintenance of certain aspects of recreational water quality (e.g. preservation of aquatic life and wildlife) are discussed in other chapters and will only be briefly mentioned here. The recommended guidelines rely on the guidelines developed by NHMRC (1990), with additional indicators included where appropriate.

Table 5.2.2 Summary of water quality guidelines for recreational waters

Parameter	Guideline
Microbiological	
Primary contact*	The median bacterial content in fresh and marine waters taken over the bathing season should not exceed 150 faecal coliform organisms/100 mL or 35 enterococci organisms/100 mL. Pathogenic free-living protozoans should be absent from bodies of fresh water.**
Secondary contact*	The median value in fresh and marine waters should not exceed 1000 faecal coliform organisms/100 mL or 230 enterococci organisms/100 mL.**
Nuisance organisms	Macrophytes, phytoplankton scums, filamentous algal mats, sewage fungus, leeches, etc., should not be present in excessive amounts.* Direct contact activities should be discouraged if algal levels of 15 000–20 000 cells/mL are present, depending on the algal species. Large numbers of midges and aquatic worms should also be avoided.
Physical and chemical	
Visual clarity & colour	To protect the aesthetic quality of a waterbody: • the natural visual clarity should not be reduced by more than 20%; • the natural hue of the water should not be changed by more than 10 points on the Munsell Scale; • the natural reflectance of the water should not be changed by more than 50%. To protect the visual clarity of waters used for swimming, the horizontal sighting of a 200 mm diameter black disc should exceed 1.6 m.
pH	The pH of the water should be within the range 5.0–9.0, assuming that the buffering capacity of the water is low near the extremes of the pH limits.
Temperature	For prolonged exposure, temperatures should be in the range 15–35°C.
Toxic chemicals	Waters containing chemicals that are either toxic or irritating to the skin or mucous membranes are unsuitable for recreation. Toxic substances should not exceed values in tables 5.2.3 and 5.2.4.
Surface films	Oil and petrochemicals should not be noticeable as a visible film on the water nor should they be detectable by odour.

* Refer to Section 3.3 of these revised Guidelines relating to nutrient concentrations necessary to limit excessive aquatic plant growth.

** Sampling frequency and maximum values are given in Section 5.2.3.1.

5.2.2 Recreational categories

5.2.2.1 Primary contact

Water used for primary contact activities, such as swimming, bathing and other direct water-contact sports, should be sufficiently free from faecal contamination, pathogenic organisms and other hazards (e.g. poor visibility or toxic chemicals) to protect the health and safety of the user. The general guidelines desirable for aquatic scenery are also applicable for water used for primary contact.

5.2.2.2 Secondary contact

Water used for secondary contact activities, such as boating and fishing, should also meet the guidelines suggested for aquatic scenery. Since there is less body contact with the water, the microbiological guidelines can generally be lower, although not in cases when shellfish might be taken from from the waterbody. To protect water-skiers from injury and boating vessels from damage, the water should be free from floating or submerged logs and stumps and excessive growth of algae and other aquatic plants. The quality of the water should be maintained so that there is minimal alteration of the fish habitat.[a]

a See Ch 3

5.2.2.3 Visual use

Surface waters used for visual recreational use (no-contact activity) should not be altered in any way that reduces their ability to support aesthetically valuable flora and fauna. Such alteration could be physical, such as dredging and dam construction, or could be due to the addition of wastes to the water. Visual impact of the surface waters is important; they should be free from:

- floating debris, oil, grease and other objectionable matter;
- substances that produce undesirable colour, odour, taste or foaming;
- undesirable aquatic life, such as algal blooms, or dense growths of attached plants or insects.

All these factors have to be considered in areas used for aquatic scenery.

5.2.3 Detailed water quality guidelines

5.2.3.1 Microbiological characteristics

Primary contact

The median bacterial content in samples of fresh or marine waters taken over the bathing season should not exceed:

- ***150 faecal coliform organisms/100 mL (minimum of five samples taken at regular intervals not exceeding one month, with four out of five samples containing less than 600 organisms/100 mL);***
- ***35 enterococci organisms/100 mL (maximum number in any one sample: 60–100 organisms/100 mL).***

Pathogenic free-living protozoans should be absent from bodies of fresh water. (It is not necessary to analyse water for these pathogens unless the temperature is greater than 24°C.)

Secondary contact

The median bacterial content in fresh and marine waters should not exceed:

- ***1000 faecal coliform organisms/100 mL (minimum of five samples taken at regular intervals not exceeding one month, with four out of five samples containing less than 4000 organisms/100 mL);***
- ***230 enterococci organisms/100 mL (maximum number in any one sample: 450–700 organisms/100 mL).***

There is a long international experience of disease outbreaks associated with contaminated water (McNeill 1985, Cabelli 1989). Disease-causing micro-organisms (pathogens) associated with bathing areas include salmonellae, shigellae, enteropathogenic *Escherichia coli,* cysts of *Entamoeba histolytica*, parasite ova, enteroviruses and infectious hepatitis (Hart 1974, McNeill 1985). Generally, the most common types of diseases that have been associated with swimming areas are eye, ear, nose and throat infections, skin diseases and gastrointestinal disorders. McNeill (1985) has reviewed epidemiological studies associated with recreational waters.

Direct detection of pathogens is not a feasible option for routine assessment, since they occur intermittently and are difficult to recover from water. For this reason, ‘indicator’ micro-organisms are generally used to assess the health risks associated

with pathogens in recreational waters (Elliot & Colwell 1985). A number of organisms have been considered as indicators of health risks for swimming areas (McNeill 1985, Daly 1991).

NHMRC (1990) favours the use of faecal coliform bacteria, a sub-group of the total coliform population that are easy to measure and are present in virtually all warm-blooded animals. Faecal coliform bacteria in human faeces comprise about 97% *E. coli*, around 2% *Klebsiella*, and a further 2% *Enterobacter* and *Citrobacter* together. However, McBride et al. (1991) have documented a number of deficiencies with the use of faecal coliforms as indicator organisms of health risks in recreational waters and waters used for shellfish growing. Recent epidemiological studies have shown poorer relationships between faecal coliform densities and illness rates in bathers than are obtained using enterococci (marine waters: Cabelli 1983a,b, Cabelli et al. 1982, 1983) and using either enterococci or *E. coli* (fresh waters: Dufour 1984). Further, there is now considerable evidence that faecal coliforms die off faster than pathogens under certain circumstances; therefore, they may go undetected during beach monitoring programs, resulting in the disease risks being underestimated.

New Zealand (McBride et al. 1991), Canada (CCREM 1991) and the United States (USEPA 1986) now recommend guidelines for recreational waters in terms of either enterococci or *E. coli* (or the non-faecal indicator *Pseudomonas aeruginosa*). For example, the New Zealand guidelines recommend that the median bacterial content of samples taken over the bathing season should not exceed 33 enterococci/100 mL (or 126 *E. coli*/10 mL) for fresh waters, and 35 enterococci/100 mL for marine waters (McBride et al. 1991). The guidelines recommended here are based on the levels recommended by NHMRC (1990) in terms of faecal coliforms, and those recommended by McBride et al. (1991) in terms of enterococci.

5.2.3.2 Nuisance organisms

Macrophytes, phytoplankton scums, filamentous algal mats, blue-green algae, sewage fungus and leeches should not be present in excessive amounts. Guidelines relating to nutrient concentrations necessary to limit excessive aquatic plant growth are given in Section 3.3 of these revised Guidelines.

Direct contact activities should be discouraged if algal levels of 15 000–20 000 cells/mL are present, depending upon the algal species. Large numbers of midges and aquatic worms should be avoided.

Biological factors that influence the recreational value of surface waters include those that endanger the health or physical comfort of people and animals, and those that render water aesthetically objectionable. In the first category are non-biting midges, phantom midges, caddis flies and mayflies, which can emerge in large numbers and cause serious nuisance to people picnicking, camping or living near the shoreline. More serious are biting insects that can cause irritation from their bites, respiratory allergic reactions or quite serious diseases. Common diseases transmitted by aquatic invertebrates are encephalitis, malaria and schistosome dermatitis (swimmer's itch).

Excessive growths of aquatic plants can also cause problems in recreational areas. Rooted and non-rooted macrophytes may obstruct the view of swimmers and

obscure underwater hazards. They can also entangle swimmers and induce panic if encountered unexpectedly. If the growth is very dense, boating and fishing may also be restricted. Dislodged or free-floating plants may also drift on to beaches, decay and cause objectionable odours as well as provide breeding areas for nuisance organisms.

Algal blooms, particularly if dominated by blue-green algae (cyanobacteria), can impair the recreational values of a waterbody by reducing the clarity and by accumulating along shorelines with effects similar to those cited for macrophytes. In addition, several species of blue-green algae can produce toxic substances that may kill fish, birds and domestic animals (Shilo 1981, Codd 1990, Falconer 1990). Species of blue-green algae have also been responsible for contact dermatitis in humans and influenza-like symptoms in swimmers (Codd 1990). Primary contact activities in waters containing high levels of cyanobacteria should be discouraged. Ingestion of cyanobacterial-infested water has been associated with gastrointestinal disorders in swimmers, and lipopolysaccharides found in certain cyanobacteria have been identified as causing skin irritations, dermatitis and allergy reactions observed in swimmers using cyanobacterial-infested waters (A McNeill, Victorian Rural Water Corporation, pers. comm., June 1992). As an interim guide, direct contact should be avoided when 15 000–20 000 cells/mL are present, depending on the algal species.

Periphyton growing on the bed of rivers and streams can also reduce the usefulness of these systems for contact recreation. Quinn (1991) recommended that to protect contact recreational areas:

> … the seasonal maximum cover of stream or river bed by periphyton as filamentous growths or mats (greater than about 3 mm thick) should not exceed 40%, and/or biomass should not exceed 100 mg chlorophyll *a*/m^2.

Quinn also called for additional research to define the level of periphyton that constitutes a nuisance.

Excessive aquatic plant growth is most often caused by high nutrient concentrations (mostly phosphorus and nitrogen) entering the waterbody. Guidelines for limitations on nutrients can be found in Section 3.3.

5.2.3.3 Physical and chemical characteristics

Visual clarity and colour

To protect the aesthetic quality of a waterbody:

- ***the natural visual clarity should not be reduced by more than 20%;***
- ***the natural hue of the water should not be changed by more than 10 points on the Munsell Scale;***
- ***the natural reflectance of the water should not be changed by more than 50%.***

To protect the visual clarity of waters used for swimming, the horizontal sighting of a 200 mm diameter black disc (Secchi disc) should exceed 1.6 m.

Guidelines relating to visual clarity and colour are required for two reasons: first, to ensure that the aesthetic quality of the waterbody is maintained and that there is no

obvious change in the colour or visual clarity; and second, that the visual clarity of the water is not so low that it is unsuitable for swimming.

As discussed in Section 8.2.3 (Vol. 2), the optical quality of water, primarily its colour and clarity, is determined by the attenuation of light, particularly by SPM but also by dissolved matter (Kirk 1983, 1988). Visual clarity, defined in Section 8.2.3, is of considerable importance because it affects the recreational and aesthetic quality of water.

Panel studies undertaken by Davies-Colley and Smith (1990) in New Zealand showed that almost all people can detect a change of 30% in visual clarity. Davies-Colley (1991) used these results to recommend that reduction in visual clarity should be limited to less than 20%. This value is also used here.

In addition to aesthetic values, visual clarity of water is also important so that swimmers can estimate depth and see subsurface hazards easily (Thornton & McMillon 1989, Smith et al. 1991). Most guidelines require that the substrate should be visible in areas that are of wadeable depth, the water clarity usually being specified in terms of Secchi depth (NHMRC 1990, CCREM 1991). However, as Davies-Colley (1991) points out, a just-visible Secchi disc on the bottom means that potential hazards, such as snags and broken bottles, will not be visible because the Secchi disc has a higher contrast than the hazards. Davies-Colley (1991) recommended that a better guideline for the visual clarity relevant to swimmer safety in wadeable areas would be to require that the black disc visibility should be not less than 1.6 m, which is equivalent to the bottom of the waterbody being visible at an adult chest height of around 1.2 m. For diving areas, the water clarity would need to be considerably greater than this.

Water colour is the perception of light backscattered from within the waterbody as observed when viewed downwards at a near-vertical angle. Typically, about 3% of the incident light will re-emerge from the waterbody as backscattered light, although this ratio can vary widely. Colour of water has three aspects: hue, brightness and saturation or colour purity (Davies-Colley 1991). New Zealand research has shown that people value blue and green hues in water, but not yellows and reds (Smith & Davies-Colley 1992). Davies-Colley (1991) recommended that the natural hue of a waterbody should not be changed by more than 10 points on the Munsell Scale. Further, he recommended that the natural reflectance should not be changed by more than 50% to protect the brightness of the waterbody. New Zealand studies have shown that people are not particularly sensitive to water brightness.

pH

The pH of the water should be within the range 5.0–9.0, assuming that the buffering capacity of the water is low near the extremes of the pH limits.

Ideally, the pH of the water for swimming purposes should be approximately the same as the lacrimal fluid of the eyes, which is about pH 7.4. However, lacrimal fluids have a high buffering capacity when in contact with solutions of different pH levels. They are able to maintain their pH within limits until their buffering capacity is exhausted. A deviation as small as 0.1 unit of the normal pH of the lacrimal fluid causes irritation of the eyes (Mood 1968).

Temperature

For human survival in cold water, the critical problem is to maintain body temperature. There is considerable variation from one individual to another in the rate of body cooling; it is primarily a function of body size, fat content, prior acclimatisation and overall physical fitness. Body heat is lost primarily by conduction from the inner organs through the trunk. Water cooler than 15°C is extremely stressful to swimmers not wearing appropriate protective clothing. Extended periods of continuous immersion at these temperatures may cause death. Thermal stress can be induced by temperatures exceeding the normal skin temperature of 33°C, and there is a risk of injury with prolonged exposure to temperatures above 34–35°C (Health & Welfare Canada 1983).

Toxic chemicals

Waters containing chemicals that are either toxic or irritating to the skin or mucous membranes are unsuitable for recreation. In general, toxic substances should not exceed the concentrations provided in tables 5.2.3 and 5.2.4.

In general, there are two kinds of human exposure in swimming areas: contact with the waterbody and ingestion of the water. Recreational water should contain no chemicals that can irritate the skin of the human body. To protect swimmers from harmful effects through ingestion, the guidelines from tables 5.2.3 and 5.2.4 should be applied for other toxicants. Special care must be taken to check for substances that can enter the body by absorption through the skin. Higher concentrations of toxicants may be tolerated occasionally if it is assumed that no person will ingest more than a maximum of 100 mL water during a normal swimming session (NHMRC 1990) compared with 2 L/d for potable water.

Surface films

Oil and petrochemicals should not be noticeable as a visible film on the water nor should they be detectable by odour.

The presence of oil and petrochemicals makes water aesthetically unattractive. They can form deposits on shorelines, and bottom sediments that are detectable by sight and odour. Some organic compounds can be absorbed directly from the water through the skin (CCREM 1991), making these substances even more undesirable in recreational areas.

Table 5.2.3 Summary of water quality guidelines for recreational purposes: general chemicals

Parameter	Guideline values (µg/L, unless otherwise stated)
Inorganic:	
Arsenic	50
Asbestos	NR
Barium	1000
Boron	1000
Cadmium	5
Chromium	50
Cyanide	100
Lead	50
Mercury	1
Nickel	100
Nitrate-N	10 000
Nitrite-N	1000
Selenium	10
Silver	50
Organic:	
Benzene	10
Benzo(*a*)pyrene	0.01
Carbon tetrachloride	3
1,1-Dichloroethene	0.3
1,2-Dichloroethane	10
Pentachlorophenol	10
Polychlorinated biphenyls	0.1
Tetrachloroethene	10
2,3,4,6-Tetrachlorophenol	1
Trichloroethene	30
2,4,5-Trichlorophenol	1
2,4,6-Trichlorophenol	10
Radiological:	
Gross alpha activity	0.1 Bq/L
Gross beta activity (excluding activity of ^{40}K)	0.1 Bq/L
Other chemicals:	
Aluminium	200
Ammonia (as N)	10
Chloride	400 000
Copper	1000
Oxygen	>6.5 (>80% saturation)
Hardness (as $CaCO_3$)	500 000
Iron	300
Manganese	100
Organics (CCE & CAE)	200
pH	6.5–8.5
Phenolics	2
Sodium	300 000
Sulfate	400 000
Sulfide	50
Surfactant (MBAS)	200
Total dissolved solids	1 000 000
Zinc	5000

NR = No guideline recommended at this time; MBAS Methylene blue active substances

Table 5.2.4 Summary of water quality guidelines for recreational purposes: pesticides

Compound	Maximum concentration (µg/L)	Compound	Maximum concentration (µg/L)
Acephate	20	Fenvalerate	40
Alachlor	3	Flamprop-methyl	6
Aldrin	1	Fluometuron	100
Amitrol	1	Formothion	100
Asulam	100	Fosamine (ammonium salt)	3000
Azinphos-methyl	10	Glyphosate	200
Barban	300	Heptachlor	3
Benomyl	200	Hexaflurate	60
Bentazone	400	Hexazinone	600
Bioresmethrin	60	Lindane	10
Bromazil	600	Maldison	100
Bromophos-ethyl	20	Methidathion	60
Bromoxynil	30	Methomyl	60
Carbaryl	60	Metolachlor	800
Carbendazim	200	Metribuzin	5
Carbofuran	30	Mevinphos	6
Carbophenothion	1	Molinate	1
Chlordane	6	Monocrotophos	2
Chlordimeform	20	Nabam	30
Chlorfenvinphos	10	Nitralin	1000
Chloroxuron	30	Omethoate	0.4
Chlorpyrifos	2	Oryzalin	60
Clopzralid	1000	Paraquat	40
Cyhexatin	200	Parathion	30
2,4-D	100	Parathion-methyl	6
DDT	3	Pendimethalin	600
Demeton	30	Perfluidone	20
Diazinon	10	Permethrin	300
Dicamba	300	Picloram	30
Dichlobenil	20	Piperonyl butoxide	200
3,6-Dichloropicolinic acid	1000	Pirimicarb	100
Dichlorvos	20	Pirimiphos-ethyl	1
Diclofop-methyl	3	Pirimiphos-methyl	60
Dicofol	100	Profenofos	0.6
Dieldrin	1	Promecarb	60
Difenzoquat	200	Propanil	1000
Dimethoate	100	Propargite	1000
Diquat	10	Propoxur	1000
Disulfoton	6	Pyrazophos	1000
Diuron	40	Quintozene	6
DPA	500	Sulprofos	20
Endosulfan	40	2,4,5-T	2
Endothal	600	Temephos	30
Endrin	1	Thiobencarb	40
EPTC	60	Thiometon	20
Ethion	6	Thiophanate	100
Ethoprophos	1	Thiram	30
Fenchlorphos	60	Trichlorofon	10
Fenitrothion	20	Triclopyr	20
Fenoprop	20	Trifluralin	500
Fensulfothion	20		

Sources: NHMRC & AWRC (1987), NHMRC (1989)

6 Drinking water

Drinking water for Australians and New Zealanders should be safe to use and aesthetically pleasing. Authoritative drinking water guidelines for both countries are summarised in the sections below.

6.1 Guidelines for users in New Zealand

Guidance on what constitutes good quality drinking water is provided for New Zealand by *Drinking-water Standards for New Zealand* (New Zealand Ministry of Health 1995a) and the *Guidelines for Drinking-water Quality Management* (New Zealand Ministry of Health 1995b).

6.2 Guidelines for users in Australia

In Australia guidance on what constitutes good quality drinking water is provided by the *Australian Drinking Water Guidelines* (NHMRC & ARMCANZ 1996), a companion document of the National Water Quality Management Strategy.

The Australian Drinking Water Guidelines are intended to meet the needs of consumers and apply at the point of use; for example, at the tap. They are applicable to any water intended for drinking irrespective of its source (municipal supplies, rainwater tanks, bores, point-of-use treatment devices, etc.) or where it is used (the home, restaurants, camping areas, shops, etc.).

The Guidelines provide an authoritative Australian reference on good drinking water quality, covering a wide range of the microbiological, physical, chemical and radiological characteristics that determine water quality. They are not intended as guidelines for environmental water quality, nor, as the document stresses, should they ever be seen as a licence to degrade the quality of a drinking water supply to a guideline value.

While the individual guideline values apply at the point of use, the document deals extensively with good system management. It points out that successful management of water quality in a water supply system requires an understanding of the processes and practices which can affect water quality within the system. In this context, the term 'system' is defined to include everything from the point of collection of the water, usually the catchment area, to the consumer's tap. It includes streams and rivers in the catchment, storage and service reservoirs, treatment and disinfection facilities, trunk and service mains, and consumer plumbing and appliances. Water quality can be affected at each of these points, but all are inter-related, and integrated management is essential.

The following sections summarise the key issues contained in the *Australian Drinking Water Guidelines* (NHMRC & ARMCANZ 1996).

6.2.1 Microbiological quality of drinking water

The Guidelines devote a special chapter to the microbiological quality of drinking water because the most common and widespread health risk associated with drinking water is contamination, either directly or indirectly, by human or animal excreta and the micro-organisms contained in faeces. Microorganisms, including

pathogenic organisms, can enter water supplies at every stage of the collection and distribution cycle. To ensure the microbiological safety of a water supply there should be a wide-ranging program of protection, treatment and monitoring, with barriers to the entry and transmission of pathogens throughout the system. The first of these barriers should include protection of the selected source from contamination by human or animal faeces and the maintenance of an active catchment protection program.

The Guidelines include a general section on Catchments and Raw Water Quality and a more specific section on Protection of the Water Catchment from Sources of Human and Animal Faecal Matter. It is recognised that intelligent management of land use and water resources in catchments is essential to a safe water supply. In particular, the Guidelines emphasise the need for an active watershed protection program, including an emergency plan for responding to major pollution events such as spillages or contamination. Detailed advice is given on the problems of surface and groundwater supplies, and the approaches that should be taken for their management.

6.2.2 Chemical and radiological quality of drinking water

The same principles of catchment management are critical in dealing with issues of chemical and radiological characteristics of drinking water. Many of these are difficult and expensive, if not virtually impossible, to remove by treatment of the raw source water. This applies to naturally-occurring characteristics, as well as to contaminants introduced from human activities.

Nitrate is an important example of a chemical that occurs naturally in groundwater supplies in some parts of inland Australia but that enters water as a result of intensive farming or poor waste disposal practices in more densely populated coastal settlements. The existing technologies for removing nitrate from source waters are rarely practicable in areas where nitrate is likely to be a problem. As nitrate is a health-related characteristic, the options may be to search for a better water source, or to arrange an alternative supply of water for consumption by those at risk, typically infants under three months of age.

Pesticides are an example of contaminants that can be introduced by improper use or accidental spillage in a catchment area, and can be difficult, if not impossible, to remove by practicable treatment processes. The Guidelines set out the method for control of pesticide use in Australia through a national scheme of registration, and recommend that their use in water or water catchments be authorised only where necessary. Pesticides not authorised for such use should not be present in drinking water.

6.2.3 Small water supplies

The Guidelines also contain a special chapter on the problems of small water supplies, regarded as those serving less than 1000 people. For small communities, economic constraints often mean that only untreated water can be supplied or that treatment is limited in extent. Furthermore, monitoring may be infrequent or absent. In such circumstances, sanitary assessment and the use of a clean and unpolluted water source are of paramount importance. It is therefore

recommended that small communities carry out regular sanitary inspections of their water supply.

Several measures can and should be taken to reduce the risk that supply to a small community may become unsafe. A strict protocol of practices should be established to ensure, among other things, that:

- raw water sources and storages are inspected regularly for any source of contamination (animals, birds, drainage inflows);
- cost-effective treatment is provided where the quality of raw water is poor (e.g. biological and pre-roughing filters).

Where problems occur, they should be thoroughly assessed. It may turn out that the best option for a small community is to seek an alternative source of raw water.

The Guidelines give detailed advice on the way in which regular inspections should be carried out to check for direct or potential sources of contamination. Inspection is especially important when water is obtained from streams flowing through areas developed for agricultural, industrial or residential purposes. The sources of contamination of groundwater are also discussed.

The frequency of sanitary inspections of a catchment will depend on the characteristics of each site and the source of raw water. Every catchment where there is human habitation or free public access should be comprehensively inspected at least once a year for potential sources of pollution.

6.2.4 Individual household supplies

Finally, consideration is given to the question of individual household supplies. For such supplies, the emphasis should be on selecting the best quality source water available, and on protecting its quality by the use of barrier systems and maintenance programs. Whatever the source (ground, surface or rainwater tanks), householders should assure themselves that the water is safe to drink. Information on the quality of surface and groundwater may be available from state or local governments which may monitor the particular source water as part of a state or local water monitoring program. Alternatively, the individual should consider having the water tested for any key health characteristics identified as being of local concern. Where the raw water quality does not meet the relevant guidelines, a point-of-use device may be used to treat water.

6.2.5 Guideline values

The individual guidelines cover a wide range of measurable characteristics, compounds or constituents that can potentially be found in water and affect its quality. They fall into the following categories:

- microorganisms, including
  - bacteria
  - protozoa
  - toxic algae
  - viruses;
- physical characteristics
  - radionuclides;

- chemicals, including
  - inorganic chemicals
  - organic compounds
  - organic disinfection by-products
  - pesticides.

A **health-related guideline value** is the concentration or measure of a water quality characteristic that, based on present knowledge, does not pose any significant risk to the health of the consumer over a lifetime of consumption.

An **aesthetic guideline value** is the concentration or measure of a water quality characteristic associated with good quality water.

The guideline values are intended for use in two separate but complementary ways:

- as 'action levels': that is, if the guideline value is exceeded, some form of action is initiated. This will generally be short-term and immediate. For example, if the guideline value for a health-related characteristic were exceeded, the response should be to take immediate action to reduce the risk to consumers, and, if necessary, to advise the health authority and consumers of the problem and the action taken. If the characteristic were not related to health, the action might be to advise the community of a deterioration in water quality;
- as a basis for assessing how well a water supply system meets, over time, levels of service agreed with the community ('performance assessment' as presented, for example, in an annual report). When used in this way, the data are largely of historical rather than immediate interest, and any resulting action to improve the quality of the supply will generally be longer-term.

In the case of pesticides, two values are provided:

- a *guideline value*, intended for use by regulatory authorities for surveillance and enforcement purposes;
- a *health value*, intended for use by health authorities when managing health risks associated with inadvertent exposure such as from a spill or misuse of a pesticide.

The document emphasises that health-related guidelines define water which, based on current knowledge, is *safe* to drink over a lifetime: that is, it constitutes no significant risk to health. For most water quality characteristics covered by the Guidelines, there is a grey area between what is clearly safe and clearly unsafe, and the latter has often not been reliably demonstrated. Thus the guidelines always err on the side of safety, and it follows that, for most characteristics, occasional excursions beyond the guideline values are not necessarily an immediate threat to health. The amount by which, and the duration for which, any health-related guideline value can be exceeded without raising public health concern depends on the particular circumstances. Exceedance of a guideline value should be a signal to investigate the cause and, if appropriate, to take remedial action. If the characteristic is health-related, the relevant health authority should be consulted.

For the individual guideline values, the reader is referred to the *Australian Drinking Water Guidelines* (NHMRC & ARMCANZ 1996). This document can be downloaded from:

http://www.nhmrc.health.gov.au/publicat/pdf/eh19.pdf

7 Monitoring and assessment

7.1 Introduction

This chapter deals with the practicalities of collecting and analysing data for the measurement and evaluation of water quality — on the one hand, by measuring biological indicators; on the other hand, by measuring the more traditional physical and chemical indicators, including toxicants. Much of this chapter presupposes a good background knowledge of the issues involved with selecting sample sites, the timing and frequency of sampling events, and some basic principles of statistics and the design of experiments and surveys. Much of this background is provided in the companion document *Australian Guidelines for Water Quality Monitoring and Reporting* (ANZECC & ARMCANZ 2000), the Monitoring Guidelines.

The Monitoring Guidelines lays out the framework and general principles for a water quality monitoring program. Though the present chapter is self-contained in terms of its coverage of monitoring and assessment, its principal aim is to complement, not duplicate, the Monitoring Guidelines. To this end, this chapter highlights some key issues for the users of the Water Quality Guidelines that are either very specific to their needs, or that expand upon some of the general topics introduced in the Monitoring Guidelines. Sections 7.1 and 7.2 generally follow the layout of the Monitoring Guidelines while Sections 7.3 and 7.4 provide more specialist information for monitoring using biological and physical-chemical indicators respectively. The chapter is structured as follows:

- Section 7.1: Introduction, issues associated with integrated assessment, and the framework for a monitoring and assessment program with reference to the introductory steps that set the monitoring program objectives.
- Section 7.2: This section describes the remainder of the monitoring framework. Firstly, recommendations are provided for combinations of biological and physico-chemical indicators to apply to different situations. Then, some generic issues that are common to both biological and physico-chemical approaches are discussed. For example, the choice of design for a monitoring or assessment program depends partly on whether or not there are data that pre-date a putative impact and on whether or not there are appropriate control sites. Section 7.2 also recapitulates the steps needed for defining objectives and selecting candidate indicators.
- Section 7.3: A description of issues that are specific to biological indicators.
- Section 7.4: An outline of issues for physical and chemical stressors and toxicants in water and sediment. For many of the non-biological indicators, the first step is to compare test data with a guideline trigger value; the procedure is detailed in Section 7.4.4.

7.1.1 Integrated monitoring strategies

Traditionally, physical and chemical methods alone are used to assess water quality by indirectly estimating ecological impairment. Numerical guidelines are set according to the response of biota from different taxa to individual chemicals, derived from single-stressor toxicity tests conducted under controlled laboratory

conditions. The derivation of 'global' guideline values, though conceptually simple, faces a major challenge in that data derived under experimental conditions may not be relevant to complex *real world* ecosystems. Nevertheless, direct measurement of physical and chemical water quality parameters as a surrogate for ecological health has the advantages of:

- conceptual simplicity,
- established technology,
- explicit numerical objectives,
- the ability to acquire meaningful quantities of data relatively quickly,
- comparatively low costs.

Biological indicators have a shorter history of use in monitoring in Australia and New Zealand. Their development has been intellectually challenging and has evoked considerable debate. This explains in part the slower acceptance of biological indicators in environmental monitoring even though the principle is inherently sound. Biological monitoring programs, and, to a lesser extent, monitoring with physical and chemical parameters, can be labour intensive, prone to quality control failures unless special care is taken, and may require data collection over an extended period, depending on the statistical design requirements. Environmental monitoring generally, however, has developed with improvements in the way sampling is conducted and in application of appropriate statistical techniques. Appendix 4, Volume 2 contains a case study that illustrates the importance of fully optimised designs in terms of spatial and temporal controls applied to indicators.[a] This case study concludes by considering the balance that negotiating parties may be faced with in applying optimised designs to early detection and biodiversity indicators in an essentially unmodified aquatic ecosystem (a condition 1 ecosystem).[b]

a See Sections 7.3 and 7.4 for more detail; also Chapters 3, 4 and 6 of the Monitoring Guidelines

b Section 3.1.3

As discussed in earlier chapters, these Guidelines emphasise an integrated approach to monitoring, using an appropriate mix of indicators suited to the primary management aims. Physico-chemical and biological indicators should be regarded as complementary to each other. Two issues involved in this integration are firstly, the rationale for integrated monitoring and assessment and ways to achieve integration; and secondly consideration of ways to defray costs. These are summarised briefly in turn.

7.1.1.1 Enhancing inferences

1. As discussed elsewhere,[c] it is widely acknowledged that only studies that include the biota can define or be used to assess the overall effect of waste waters on these organisms and the ecological health of ecosystems. Management goals are typically biologically-based, so organisms are the management end-point. This position holds even if the methods used for determining global numerical guidelines, including surrogates for biological end-points such as water chemical analytes, are acknowledged as having broad validity. A combination of biological and physico-chemical assessment enhances the confidence in correctly attributing causes to any observed change in water quality: biological variables integrate effects of past and present exposure and directly assess progress in achieving the management goals;

c Sections 3.1.6 & 3.2.1.1

physico-chemical variables are the explanatory variables in the cause–effect relationship.

2. Efforts should be made, wherever possible, to examine and incorporate the results of similar types of study conducted in the region. Whether the results of the additional studies are examined alone or are combined with those from the study in question, inferences can be enhanced.[a]

a See box 7.1.1

3. Sometimes samples may be gathered and processed in a manner that allows the results to be used for different purposes, each providing additional interpretative information. An example of this is provided below[b] where the advantages of combining stream macroinvertebrate samples and data from quantitative and rapid biological assessment studies are outlined.

b Section 7.2.1.1/1

4. Users need to be aware always of standard operating procedures that may be in place at the regional scale and beyond. Comparison of results with those from other studies is always enhanced where a common sampling and measurement protocol is used.

Box 7.1.1 Enhancing inferences and defraying costs in environmental monitoring programs

Whatever the indicators used in a monitoring program, savings in resources can be made in the experimental design if data from control sites are shared amongst different bodies conducting similar monitoring programs in the region. Apart from the advantage of cost sharing, combined results can then be included in formal meta-analyses (analyses which combine the results of many similar studies) and thereby allow stronger inferences to be drawn (see also Section 7.2.5).

7.1.1.2 Defraying costs

The availability of resources is recognised as a major constraint in meeting the level of monitoring recommended in these Guidelines. Ways to defray costs must always be considered. Some examples to consider in this respect include:

1. As far as possible, ensure that there is a common sampling program for collection of data on different indicators. Other than providing greater interpretative value for the data gathered, this will reduce logistical costs (e.g. transport etc.).

2. Share costs with similar monitoring programs being conducted in adjacent areas.[c]

c See box 7.1.1

3. Incorporation of biological assessment in environmental monitoring programs may lead to cost-savings for industry if 'no-observable-effects' in biological responses are found, despite values for physico-chemical indicators that might be 'high' or which may exceed the recommended guidelines.[d] Use of the decision trees for physico-chemical indicators can also lead to cost-savings for industry; the first of the two case studies included in the *Introduction* to the Water Quality Guidelines provides such an example.

d Section 3.1.3.2

Section 7.2.1.1 recommends the type and number of indicators that should be incorporated in a water resource monitoring and assessment program, depending upon ecosystem condition (condition 1, 2 or 3 ecosystems).[a] These recommendations need to be augmented in two special cases as described in Section 7.2.1.2. These special cases are situations where there is inadequate baseline data[b] and situations that call only for a broad-scale assessment of ecosystem health.[c]

a See Section 3.1.3

b Section 7.2.1.2/1

c Section 7.2.1.2/2

The final balance of indicators to be measured at a site will rest with local jurisdictions and stakeholders after they have considered factors such as the nature of contaminants, the ecosystem type, the issues of concern, level of protection, availability of baseline data and resource constraints. While the constraints of resources are acknowledged, local jurisdictions still have a responsibility to ensure their water quality monitoring programs are sufficiently adequate to give unambiguous results from which confident conclusions can be drawn.

7.1.2 Framework for a monitoring and assessment program

Although water quality monitoring with physical and chemical indicators differs in philosophy and techniques from monitoring with biological indicators, the approaches both rely on sound practice in environmental science, including:

- explicit written definition of the sampling site, project objectives, a hypothesis and the sampling protocol that will support the work;
- the definition of sampling sites, sampling frequency, and spatial and temporal variability that will permit appropriate statistical methods to be used;
- rigorous attention to field and laboratory quality control and assurance;
- incorporation of a pilot study to test the sampling protocol and determine spatial and temporal variability.

Figure 7.1.1 outlines the basic steps involved in developing a program for monitoring and assessing both biological and physico-chemical aspects of water quality. This figure is consistent with the framework for the Monitoring Guidelines, as portrayed in figure 1.1 of those Guidelines. The framework figure shown in the Monitoring Guidelines is necessarily general in nature while figure 7.1.1 of the current Guidelines has adapted the Monitoring Guidelines framework to incorporate aspects of the management framework outlined in Chapter 2.[d]

d See Figure 2.1.1

The first step of the framework, determining the primary management aims, has been described in earlier chapters of these Guidelines.[e] Determining these aims will enable stakeholders to develop an appropriate conceptual model of key ecosystem processes and interactions. By doing this they can identify assumptions against which monitoring outcomes can be tested, and develop appropriate working hypotheses whose predictions can be tested using the data that the program collects — Step 2 of the monitoring framework (figure 7.1.1). Step 2, developing a hypothesis, is discussed earlier in these Guidelines[f] and in Chapter 2 of the Monitoring Guidelines. Step 1 of the Monitoring Guidelines framework (figure 1.1), 'Monitoring Program Objectives', combines the first two steps from the Water Quality Guidelines framework of figure 7.1.1. The remainder of this chapter is concerned with the other steps in figure 7.1.1. Background information that supports the material presented here is provided in the Monitoring Guidelines.

e Chapter 2 and Section 3.1.1.1

f Section 2.2.3

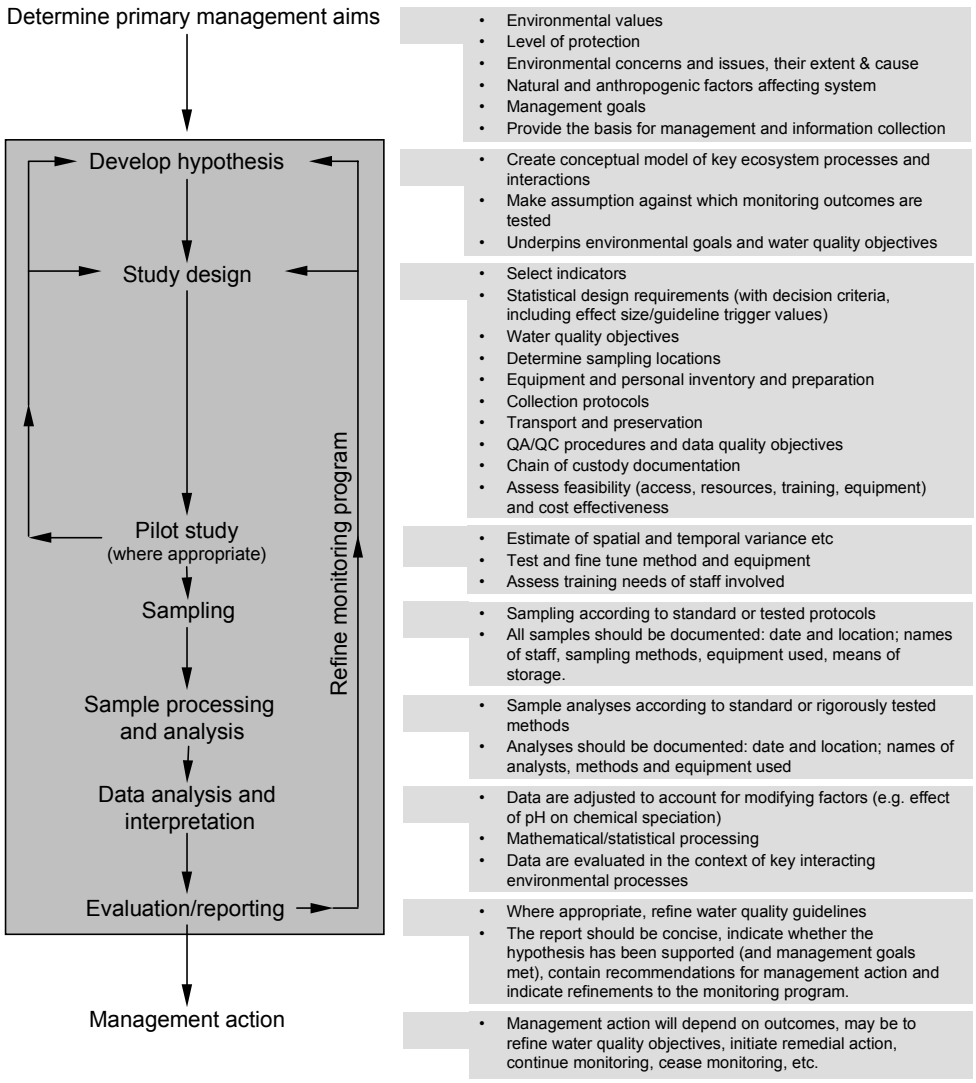


Figure 7.1.1 Procedural framework for the monitoring and assessment of water quality (the shaded area). (Adapted from the Monitoring & Reporting Guidelines and the framework for designing a wetland monitoring program adopted by the Ramsar Wetland Convention (Ramsar Convention 1996, Finlayson 1996))

7.2 Choosing a study design

This next step of the monitoring framework (figure 7.1.1) includes the selection of indicators and requirements for experimental design, including the determination of guideline values. General descriptions are provided in Chapter 3 of the Monitoring Guidelines. However, the earlier chapters of the Water Quality Guidelines and its two support volumes are the main reference sources for indicator selection and determining guideline values, and these aspects are not discussed further. This section recommends a balance of indicators to apply to different situations for aquatic ecosystem protection (Section 7.2.1) and provides specific advice for experimental design using indicators from all environmental values (Sections 7.2.2 and 7.2.3).

7.2.1 Recommendations for combinations of indicators for aquatic ecosystems

7.2.1.1 Recommendations for each ecosystem condition

This section makes some basic recommendations for the number and mix of indicators that should be used in integrated monitoring for each of the ecosystem conditions.

1. Sites of high conservation value (condition 1 ecosystems)

For high conservation value sites, the goal for a water quality assessment program should include four–six of the following aspects: (i) for contaminants other than nutrients, 'whole effluent' toxicity testing to determine a safe dilution at which effluent may be discharged; (ii) water and sediment physico-chemistry; (iii) an 'early detection' indicator for either water or sediment (whichever is deemed to harbour greater risks to aquatic organisms arising from the fate and persistence of waste substances); (iv) a quantitative biodiversity indicator; and (if applicable and available) (v) a community metabolism indicator and (vi) a rapid biological assessment (RBA) indicator (see rationale below).

Ideally, for early detection (item (iii) above) a biological indicator of the type described in Section 3.2 (in particular, table 3.2.2) would be used for monitoring. It is acknowledged, however, that such indicators have at present been developed for only a relatively narrow range of conditions and regions. Until such indicators have been further developed and are more widely available, it is important, nevertheless, to adhere to the *principle* of early detection in monitoring and to consider alternative approaches to meeting this important assessment objective. For example, in some situations, adherence and responsiveness to very conservative chemical criteria and their trends may be more protective of ecosystems than even very sensitive biological tests. Alternatively or in addition, in Section 3.2.1.3/2 it was suggested that early detection and predictive capabilities would be enhanced by placing additional sampling sites for any indicator in 'mixing zones' — effectively measuring gradients of spatial disturbance.

The quantitative biodiversity indicator (item (iv) above) should be selected from Section 3.2. It would normally be expected that some species-level data would be gathered for relevant biodiversity indicators in regions of high conservation value. As a complement to the measurement of the quantitative biodiversity indicator, there could be situations where it would be advisable to also collect data for an RBA indicator. In some respects, results gathered for RBA can be better than

results from many quantitative approaches because they provide information about the ecological importance of effects. As stated in Section 3.2.1.3/3, RBA programs that have regional coverage and that encompass a full disturbance gradient can provide regional context for the gathered data. Data gathered for RBA indicators would not normally be expected to detect minor or subtle impacts, and for this reason they should never be measured in isolation from quantitative indicators at sites of high conservation value (nor, in most cases, at sites in slightly–moderately disturbed systems).

Measurement of quantitative *and* RBA indicators need not add significantly to the costs of a monitoring program. For example, replicate quantitative samples from stream macroinvertebrate communities at a site could initially be processed as prescribed for the AUSRIVAS RBA approach (e.g. live-sorted, see Method 3A(iii), Appendix 3 of Volume 2) and then the residue could be preserved for later laboratory processing in the usual (quantitative) manner. An initial pilot study could be required to reconcile the sampling effort needed in the field to serve both RBA and quantitative approaches. RBA data gathered from several sites would be incorporated into, and assessed against, broader regional or state/territory AUSRIVAS models.

2 Slightly to moderately disturbed systems (condition 2 ecosystems)

For slightly–moderately disturbed sites, the recommended water quality assessment program has the same four–six aspects prescribed in Section 7.2.1.1/1. For measurement of biodiversity indicators, species-level data may not be necessary.

3 Highly disturbed systems (condition 3 ecosystems)

For highly disturbed sites, it is recommended that a monitoring program includes (i) water and sediment physico-chemistry, (ii) a rapid broad-scale and/or quantitative biodiversity indicator, depending upon the nature and degree of contamination and level of sensitivity to impact required (selected from Section 3.2.2), and (iii) (if applicable and available) a community metabolism indicator.

7.2.1.2 Combinations of indicators for two likely special cases

a See Section 7.2.1.2/1
b Described in Section 7.2.2 & in Ch 3, Monitoring Guidelines
c Section 7.2.1.2/2

In addition to choosing an appropriate set of indicators for an integrated program according to the ecosystem type, there are two situations that are likely to arise in many applications. In the first situation,[a] there are insufficient baseline (i.e. 'pre-impact') data to implement 'before–after' type sampling designs.[b] The second situation applies where broad-scale assessment of ecosystem health is the goal of the program.[c]

1 Sites where an insufficient baseline sampling period is available

d Section 3.2.4.1

If it is not possible to gather sufficient baseline data, the Guidelines recommend additional monitoring, including a greater number of indicators and/or sites for 'early detection' and biodiversity measurement (i.e. the 'multiple lines of evidence' concept[d]). Some recent proposals to help formalise the use of 'multiple lines of evidence' are described in Chapter 3 (Section 3.2.3) of the Monitoring Guidelines.

e Section 7.2.1.1

i. For sites where development is planned, it is recommended that more extensive biological assessment procedures be incorporated than those outlined above.[e] This would include, for contaminants other than nutrients, a 'whole effluent' toxicity testing program for determining a safe dilution at which effluent could be discharged. For such situations, further protocols for early detection and biodiversity indicators will recommend the collection of data from a larger

number of 'control' and 'to-be-disturbed' sites than would otherwise be gathered, so that stronger inferences may be drawn about impact by way of disturbance gradients.[a]

a See also Sections 7.2.2 & 7.2.3 below

ii. At sites where there are existing developments, adequate baseline data were never gathered; the project approval phase pre-dated more stringent discharge licensing conditions that have subsequently been imposed by regulators. Use the same water quality assessment indicators as for Part (i) above, modified for *a posteriori* conditions.

iii. For *a posteriori* monitoring of accidental discharges, use the same water quality assessment indicators as for Part (i) above, modified for *a posteriori* conditions.

2 Broad-scale assessment of ecosystem health

Applications of broad-scale monitoring procedures include assessments of biological water quality for planning purposes, the setting of goals for remediation and rehabilitation programs, and the monitoring and assessment of broad-scale impacts such as diffuse pollution. For such sites, it is recommended that a monitoring program includes (i) water and (if appropriate) sediment physico-chemistry, and (ii) data compatible with national RBA programs (e.g. AUSRIVAS).

7.2.2 Broad classes of monitoring design

This section describes the choices of broad classes of designs of monitoring programs which are available under different scenarios. Note that for the majority of the physical and chemical stressors and toxicants, the initial step in assessment is to compare data from the test waterbody or system with guideline trigger values.[b]

b Section 7.4.4

The design of a program for monitoring or assessing water quality is crucial. As described above, this step presupposes well articulated primary management aims and appropriate working hypotheses whose predictions can be tested using the data that the study collects.[c]

c Section 7.1.2

However, as described below, the types of program design depend on the context within which the investigation is taking place. The context can limit the choices and inferential strength of the program design.

There are five broad classes of program design (figure 7.2.1; modified after Green 1979). The choice depends on whether the disturbance (putative environmental impact) has already occurred, and whether any control sites are available for inclusion in the program. When designing any program for monitoring and assessment, professional statistical advice should be sought before the data are collected. All the designs outlined in this section have assumptions, and often involve sophisticated statistical procedures.[d]

d See the Monitoring Guidelines, Chapters 3, 4, 5 and 6

Most water quality assessment and monitoring will take place relative to a definable event, which is called a *disturbance* in figure 7.2.1. This will often be a potential environmental impact (e.g. construction of a new outfall, change in land-use), but may correspond to change in activity to improve water quality (e.g. installation of a new treatment plant, initiation of controls on fertiliser use). If the disturbance has not already occurred, then there is scope to collect appropriate data before the disturbance; furthermore, if there are control areas or sites, then this leads to the strongest class of monitoring and assessment designs, called the 'Before–After

Control–Impact' family of designs (BACI) (case A in figure 7.2.1). Wherever possible these designs should be used, especially where the opportunity exists to incorporate appropriate controls (the so-called MBACI designs of Keough & Mapstone 1995). The logic that underpins this family of designs is described in Section 3.2.2.1 of the Monitoring Guidelines. In general terms the MBACI design (where multiple control sites are included) provides the strongest inferences. A potentially important embellishment for systems with unidirectional flow is to use matched pairs of sites (upstream and downstream) in disturbed and control locations (MBACI-P of table 7.2.1). In this scenario it is the differences between upstream and downstream sites that are compared.[a]

a More detail provided in Section 7.3.2 and figure 7.2.1 for comparisons of similarity measures

However, two common situations often arise. Either there are no appropriate control sites, in which case inferences about the event need to be based on changes through time alone (case B in figure 7.2.1); or the program has to commence after the event, in which case inferences need to be based on spatial pattern alone (case D in figure 7.2.1). Inferences based on spatial pattern alone will usually need to include reference sites or sites that provide a yardstick against which to compare the site that is being disturbed. AUSRIVAS, the rapid biological assessment procedure based on stream macroinvertebrates, can be viewed as a special case of this class of design.[b] Similarly, a variety of techniques can be used for basing inferences on changes through time alone.

b See the Monitoring Guidelines Section 3.2 and Section 7.3.3 below

A further case, called *a posteriori* sampling, can arise; see case B in table 7.2.1. Some chemicals and toxicants are so unusual that they can only come from human activity (e.g. some specialised pesticides, some unusual isotopes). Detection of these substances after a disturbance has occurred may be sufficient to infer environmental impact, without the need to collect any data from before the disturbance or from spatial control or reference sites. This is likely to be highly unusual, and exceptional care would need to be taken to convince all stakeholders that the substance concerned was unequivocally linked to the disturbance.[c] Moreover, very good evidence would need to be assembled from auxiliary studies to establish that concentrations of the substance below the detection level of the laboratory analysis were ecologically harmless.

c This is described in more detail in Section 7.3.3 below

Occasionally, monitoring or assessment programs are initiated when the timing or location of the disturbance is unknown. This leads to two further types of study design that are not considered in any further detail in these Guidelines. *Baseline studies* (case C of figure 7.2.1) refer to those carried out before an event has occurred, where the goal is to attempt to detect unanticipated changes or trends in the environment. Broad-scale water quality monitoring networks as well as well-planned developments exemplify this approach. *Investigative studies* (case E of figure 7.2.1) are made in response to a perception that some environmental change has occurred; their goal is to determine the timing or nature of the change. Examples include studies carried out after unexpected fish kills or research programs investigating the extent and severity of acid rain.[d]

d See also the Monitoring Guidelines Section 3.2

Finally, management for rehabilitating or restoring disturbed sites has some special problems that need to be taken into account when designing a monitoring and assessment program. They are outlined in box 7.2.1 below, 'Issues for restoration and rehabilitation', while box 7.2.3 outlines the related procedure of 'bioequivalence testing' which is appropriate for hypothesis testing in these programs.

7.2.3 Checklist of issues in refining program design

Once the broad category of design has been selected (Section 7.2.2 above), there are a number of issues that need to be addressed, preferably in consultation with a statistician, to refine the design and ensure that data will be collected properly for the valid application of the chosen statistical methods. (See the Monitoring Guidelines Chapter 6 for a discussion of the basic statistical issues, and Chapter 3 for discussion of site selection and the scope of the sampling program in space and time.) The following sections seek to highlight the most prominent issues.

7.2.3.1 Site selection and temporal and spatial scales

To detect impacts reliably, the size and relationship of sampling areas and the pattern of sampling in space through time need careful consideration. The assessment objective and the nature of the disturbance also affect sampling design, as well as site-specific and regional factors. It is difficult to be prescriptive, but some general guidance on the issues that need to be addressed is summarised in this section; see also discussion in the Monitoring Guidelines Chapter 3.

Independence of control and impact sites for the indicators being measured is important for all the BACI-type and spatially-based procedures (cases A and D of figure 7.2.1). If control and impact sites are too close, cross-contamination can occur which can mask changes in the indicator. What constitutes *too close* depends both on the nature of the indicator and dispersion of the pollutant. Where independence cannot be ensured, there may be procedures which can take intercorrelations between sites into account. Such procedures need expert statistical input before the data are collected.

Information on water movements is essential for planning the extent and separation of control and impact sites. Climatic and water velocity data can be combined with information on discharge and morphometry in inland waters, or data on tidal movements and oceanic circulation in marine situations, to estimate the direction and extent of mixing and dispersion of effluents. Sometimes sophisticated computer simulation models are available to assist in predicting these aspects of water movement.

Spatial variation within the site(s) to be sampled can also affect the precision of estimates of that site, which in turn can affect the outcome of any formal significance tests. Often there are distinct habitats or *strata* within the sites, and variation within the strata should be quantified in any sampling area; a single sample unit from each stratum is inadequate. Several sample units should be taken within the smallest scale of systematic variation, and often sites are sufficiently large that they require several levels of successively finer spatial resolution to be nested within each of the control and impact sites (e.g. Morrisey et al. 1992). Such sub-sampling improves the precision of the estimates of interest, and a good pilot study using a thorough, hierarchical design is essential for estimating which scales of variation are important and, consequently, the most cost effective sampling strategy likely for the final design[a] (theory: Sokal & Rohlf (1981), Andrew & Mapstone (1987), McPherson (1990); examples: Morrisey et al. (1992), Downes et al. (1993)). In addition, the behaviour of data is likely to be better at higher levels in a sampling hierarchy: data are more likely to be normally distributed and the influence of zeroes in the data is diminished as a result of the central limit theorem (Keough & Mapstone 1995).

a These issues and sampling strategies to deal with them are described in Section 3.4 of the Monitoring Guidelines

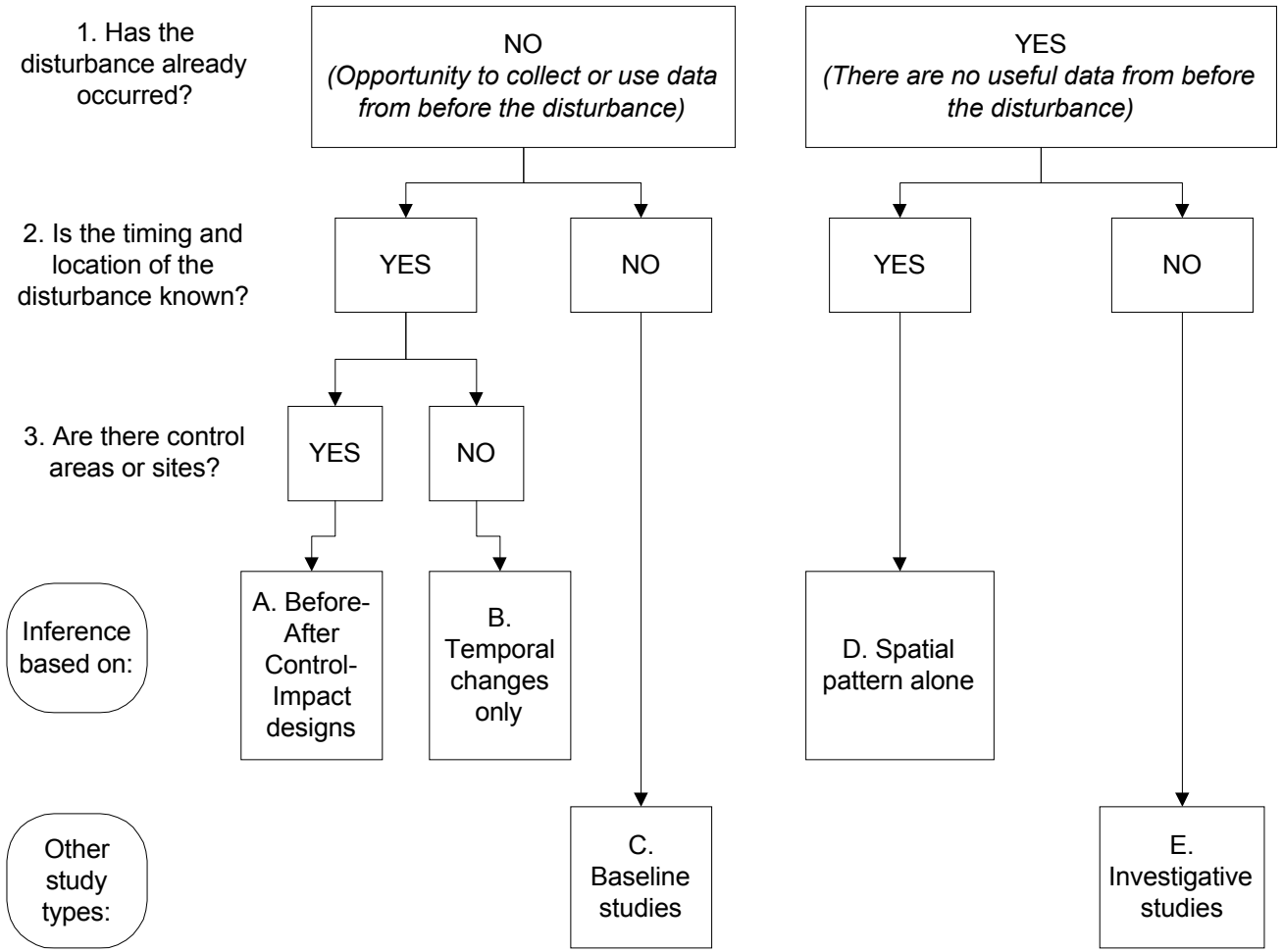


Figure 7.2.1 Flow chart depicting the broad categories of designs for monitoring and assessment that apply in different contexts. Only categories A, B and D are discussed in detail in this document. See also table 7.2.1 and the Monitoring Guidelines Section 3.2.

Table 7.2.1 Broad categories of design from figure 7.2.1 relevant to the Guidelines listed together with the assessment objectives that could be fulfilled by each category. Possible designs within each of the three broad categories (A, B and D) are tabulated with a brief description and commentary together with examples and references to other publications.

A. Inference based on the BACI (Before–After Control–Impact) family of designs

These designs are suitable for the following assessment objectives:

- early detection,
- biodiversity or ecosystem-level response.

Where comparable control sites exist and there is sufficient lead time before the disturbance, the MBACI design should be preferred unless the prevailing situation requires one of the other BACI designs described here. The general logic of the BACI family of designs is outlined in Section 3.2 of the Monitoring Guideines.

Possible designs	Description and notes	Examples and references
MBACI	Before–After Control–Impact design with Multiple control areas and (if possible) >1 impact area. Preferred design because of increased confidence that differences between control and impact areas are not due to peculiarities between single control and impact areas. – May be modified to MBACI-P (where P stands for pairing of sites) if indicator is best expressed in terms of differences between paired sites. – Short-term and long-term impacts require careful planning of frequency of sampling. Variation/trends amongst areas/times may be modelled using regression, covariates, or dynamic simulation and permutation methods.	The general principles behind this design are outlined in Section 3.2 of the Monitoring Guidelines (see also figure 3.3 in that document). Keough & Mapstone (1995; 1997) provide a full description and discussion. Faith et al. (1995) discuss principles of MBACI-P designs.
'Beyond BACI' designs	Elaboration of MBACI designs with additional nested components in time and/or space. Appropriate where the spatial and/or temporal scale of the impact is unknown or where changes in the pattern of *variation* of the indicator are more important than detecting changes in the average value of the indicator.	Underwood (1994) describes the most elaborate models based on ANOVA; general principles could be extended to other statistical techniques with more flexible assumptions (e.g. general linear models).
BACIP (single control site) Modifications	Before–After Control–Impact, Paired differences. Applicable if there is limited scope for spatial replication (e.g. one 'control' and one 'impact' site). Required if seasonal or other temporal changes in response are known to occur *OR* if temporal behaviour of response is unknown. Differences may consist of multivariate dissimilarities. – Set of differences before and after impact compared using Student's t-test (or equivalent) (Appendix 4, Vol 2). – Modelling trends or thresholds and/or inclusion of covariates (Appendix 4, Vol 2).	Illustrated in figure 3.4 of the Monitoring Guidelines. Described in detail by Stewart-Oaten et al. (1986; 1992). Example provided in Humphrey et al. (1995). Faith et al. (1995) discuss multivariate modification.
Simple BACI	Before–After Control–Impact; only one sampling event prior to impact. Applicable only if seasonal or other temporal changes in the indicator have been demonstrated to not occur. – Of dubious value because only one sampling event prior to the disturbance leads to a high chance of confounding with natural changes unrelated to the disturbance.	Described by Green (1979) and in figure 3.2 of the Monitoring Guidelines. This design critiqued by Hurlbert (1984) and Stewart-Oaten et al. (1986).

Table 7.2.1 (continued)

B. Inference based on temporal change alone

These designs are suitable for the following assessment objectives:

- early detection,
- biodiversity or ecosystem-level response.

These designs should be used if no comparable control sites exist. They assume that any changes in the behaviour of the indicator after the disturbance are solely attributable to the disturbance (see Section 3.2 of the Monitoring Guidelines). Other lines of evidence, such as would be gathered under an integrated monitoring program, would strengthen inferences from these designs (see table 3.2 in the Monitoring Guidelines).

Possible designs	Description and notes	Examples and references
Intervention analysis	Disturbance is regarded as an intervention and applicable when a long time series of data has been collected before the supposed impact which can be used as a baseline to compare to data collected after the disturbance. Applicable when no suitable control sites can be found that are comparable with the supposed impact site.	Welsh & Stewart (1989) and Thompson et al. (1982) exemplify intervention analysis applied to chemical and biological indicators respectively.
Trend analysis	Objective is to describe any trend in the chosen indicator. There are several methods that can be used to estimate trends, including those below. – Time series analysis in which, if the data sequence is long enough and sampling sufficiently frequent, temporal autocorrelations can be modelled and treated appropriately. – Control charting and allied techniques derived from statistical process control can be used to measure changes of means and variance of the values of an indicator relative to notional 'action thresholds'. – GAMs (Generalised Additive Models) relatively new, advanced group of procedures which replace linear functions with unspecified 'smoothers' that are suggested by the data themselves. Robust smoothing is useful for displaying trends in data with extreme values or outliers.	See Section 6.3 of the Monitoring Guidelines for brief, general descriptions and references for all the techniques listed here. Gilbert (1987) and Galpin & Basson (1990) provide overviews of the complexities of applying trend analyses to water quality data.
A posteriori sampling	Applicable only if measured response (especially chemical or biochemical marker) is unequivocally related to the effluent (Section 3.2.3; otherwise not elaborated upon).	

Table 7.2.1 (continued)

D. Inference based on spatial pattern alone

These designs are suitable for the following assessment objectives:

- biodiversity or ecosystem-level response
- broad-scale assessment

These designs assume that disturbed sites and undisturbed sites had similar values of the indicator before the disturbance (see Section 3.2 of the Monitoring Guidelines). Other lines of evidence, such as would be gathered under an integrated monitoring program, would strengthen inferences from these designs; see table 3.2 in the Monitoring Guidelines.

Possible designs	Description and notes	Examples and references
Conventional statistical designs (e.g. ANOVA, ANCOVA)	Comparisons are made between disturbed and undisturbed sites. – Pairing of sites upstream and downstream of disturbance and comparison of these differences with differences from matched pairs in undisturbed water bodies can strengthen the inference. – Matching disturbed site(s) with undisturbed site(s) is essential but sometimes difficult. Use of covariates can assist in adjusting for moderate background differences between sites.	Discussed by Underwood (1993); examples described by Green (1979) Davies & Nelson (1994) provide an example comparing differences between matched paired sites on streams subjected to different forestry operations.
	– For multivariate indicators (e.g. measures of community similarity) analysis of similarity (ANOSIM) techniques are available for some basic designs. Future developments in permutation and randomisation testing are likely to expand the complexity of designs that can be analysed.	Legendre & Legendre (1998) provide a general overview of a variety of multivariate techniques appropriate for similarity data. Clarke and Green (1988) and Clarke and Warwick (1994) explain ANOSIM and give some examples.
Analysis of 'disturbance gradients'	Several sites can be identified with a range of severity of the disturbance. Inferences are drawn from correlation of disturbance (or surrogate disturbance) variables with values of the indicator. A variety of techniques can be used including those below. – Regression relates the strength of disturbance to the response of the indicator. – Spatial statistical designs and methods can be useful where the inference is based on estimating parameters collected over a contiguous area. The sampling intensity for such methods is often demanding. – For multivariate indicators, spatial statistical techniques are becoming available based on permutation and randomisation tests. Again the sampling intensity can be demanding.	Basic description provided in Section 6.5 of the Monitoring Guidelines. Cressie (1993) and Rossi et al. (1992) detail some of the conventional spatial statistical techniques. Thrush et al. (1994) provide an example from marine benthos. Legendre & Legendre (1998) provide a recent overview of a range of promising techniques with copious references to published applications.
Predictive models based on spatial controls only	Detection and assessment by predictive modelling (e.g. AUSRIVAS). – At present only AUSRIVAS for macroinvertebrates in rivers and streams has been developed. This method relies on a network of reference sites against which test sites (those thought to have been disturbed) are compared. Test sites may not have been sampled contemporaneously with reference sites, so this method makes a large assumption that there is low inter-annual variation in the family-level composition of macroinvertebrate communities. Because inferences are based on family-level spatial data, this method is likely to be sensitive to only moderate to large impacts.	AUSRIVAS is outlined in Section 7.3.3 and by Schofield & Davies (1996); it is derived from the British RIVPACS system, the mechanics of which are described by Wright (1995). The applications and limitations of AUSRIVAS in the context of these Guidelines are described in Sections 3.2 and 7.3.3.

Box 7.2.1 Issues for restoration and rehabilitation

Many of the principles identified in the accompanying sections also apply to designing programs for monitoring and assessing the extent of biological recovery after an environmental impact has occurred. The formal process of setting criteria for making decisions (Section 7.2.3.3) often receives little attention in rehabilitation and restoration programs and is an area meriting further exploration (e.g. Maguire 1995).

In the majority of rehabilitation and restoration programs, there will not be reliable data collected over long time periods before the environmental impact. Thus the main problem in setting the criteria for making decisions for such programs lies in defining appropriate targets for the chosen indicators by which the success of a program can be judged. If there are no pre-disturbance data at all, then the sampling program should include appropriate undisturbed sites that can act as reference sites for the disturbed area. This, of course, entails making assumptions about similarity in behaviour of the indicator over time in the affected area and the control areas in the absence of the disturbance (Section 7.2), and there is a danger that the reference sites will not represent a realistic target for the affected area (Wiens & Parker 1995). Furthermore, there are likely to be situations where there are no appropriate reference sites, and the target reference condition will need to be set by other means (Section 3.1.4). Setting targets in these situations is difficult, and will often involve subjective judgements from expert panels and/or stakeholders. For example, suppose a target value is set for an indicator and, after the prescribed time since rehabilitation, the indicator has still not reached the target value; there are no logical grounds for determining whether the rehabilitation has failed or the target was set too high.

In all cases, there will need to be extensive liaison between managers and stakeholders to ensure that appropriate indicators are selected and that targets are appropriate for the constraints and context of the impact under consideration (Maguire 1995). Within the framework provided in Chapter 7, the following four issues need to be considered.

First, the indicators selected will need to accurately reflect the nature of the change desired. Rehabilitation programs sometimes can concentrate on obvious, but inappropriate indicators. Norris (1986) provides a salutary example where remediation of a disused mine site focused on obvious terrestrial and riparian works (as indicators of remediation success) which did not result in any improvement in the biological attributes of the river. The nature of the desired change will also depend on the time-lags between implementing a management action and the response of the indicator. For example, changes to land use on a catchment may take longer to result in a change in algal community composition than closing a sewage outfall; thus sampling programs and decision criteria will need to be geared towards gradual changes in the former and relatively abrupt changes in the latter.

Second, the context of the desired change needs to be considered in concert with the size of the effect that needs to be detected so that timely alterations to the management of the remediation program can be made (Section 7.2.3.3). For example, a program to assess the success of a clean-up operation after an accidental oil spill will need tightly specified effect sizes and timelines if legal action about compensation payments depends on the success of this operation. By contrast, the rehabilitation of a large mine area that has been a source of serious pollution for many decades may need intermediate goals as various phases of the rehabilitation process are implemented and their success is assessed. As a result, timelines and targets may need to be re-set as rehabilitation proceeds.

Third, the relative risks and cost of committing a Type I or Type II error need to be considered carefully (Section 7.2.3.3), especially in circumstances where pre-impact baseline data are limited and/or control or reference areas are few (see box 7.2.3, ‘Application of bioequivalence testing’, for the meaning of Type I and Type II errors under this form of hypothesis testing).

Fourth, the choice of analytical procedures and the scope of the conclusions (Sections 7.2.2 and 7.2.6) will be limited by the availability of appropriate reference or control data. In some cases, where multiple sites are to be rehabilitated over long time spans, MBACI designs could be implemented (Section 7.2.2), although the use of bioequivalence testing procedures under such complex designs may need further statistical development (McDonald & Erickson 1994). Conversely, monitoring the recovery of an indicator after an isolated accident such as a toxic chemical spill limits the potential analytical options and strength of the inferences (Wiens & Parker 1995, see also Section 7.2.2).

Just as spatial patterns must be considered in sampling design, so must patterns in time. These patterns may be predictable (e.g. periodic behaviour of tides) or episodic (e.g. floods), and range in scale from many years (e.g. El Niño–Southern Oscillation) to diurnal or even shorter time-scales. Again, as with spatial variation, failure to account for temporal patterns can confound impacts with natural events, and similar sampling strategies are called for to estimate these patterns.[a]

a Section 3.4, Monitoring Guidelines, also discusses these issues

7.2.3.2 The importance of good pilot data

Sampling programs can be costly, and it is important to try to optimise the sampling program so as to address the hypotheses posed by the program (the feedback loop in figure 7.1.1). Good pilot data collected before the monitoring program commences are therefore highly desirable in the absence of a validated historical database. Note that the number of samples acquired in pilot programs should be as large as is feasible to provide accurate estimates of variation; pilot data using small sample numbers yield unreliable information that may lead to poor decisions in optimising the sampling program. The design of the pilot sampling protocol must be as detailed and thoughtful as for the main project, though it should be remembered that to refine a sampling protocol is one of the principal objectives of a pilot study. Optimisation decisions based on a well designed pilot study will be more soundly based and hence defensible. Another advantage of a pilot study is that it gives field staff site-specific training, and allows anticipation of potential hazards and logistical problems. Most practitioners recommend that a significant fraction of total project resources should be dedicated to a pilot study; Keith (1991) recommends 10–15%.

7.2.3.3 Setting criteria for decisions

The values of indicator variables usually respond to disturbances in a continuous fashion (e.g. the 'dose–response relationship' of toxicology). As explained in Section 3.1.7, somewhere along the continuum a value of an indicator needs to be chosen which forms the criterion for making a decision which will precipitate some management response.

This section outlines the procedures for setting such decision criteria, in three steps. First it explains the use of hypothesis testing in this process; then it describes the three stages that need to be addressed when setting decision criteria. In outline, the first stage involves deciding what sort of change to look for in the indicators, in the context of the environmental assessment objectives. The second stage involves translating this change in the indicator into a quantifiable effect size. The third stage involves assessing the risk of making a Type I error (giving false alarm) or a Type II error (giving false sense of security)[b] in the light of the consequences or costs of making either of those errors (in a purely scientific and/or social value sense). It is important that these three interconnected stages are discussed and iterated with the stakeholders interested in the results of any monitoring or assessment program. The negotiations should be undertaken before implementing a monitoring or assessment program so that effect sizes, error rates and costs are identified explicitly. It is also best to discuss several indicators simultaneously in this process because, inevitably, some indicators may prove to be more cost-effective than others in detecting change.

b See Sections 3.1.7 and 3.2.4

The use of hypothesis testing

These Guidelines generally adopt a statistical hypothesis testing approach to determine whether the values of the chosen indicators have exceeded guideline values. Users should be aware that hypothesis testing is not the only statistical procedure that can be used in making inferences from water quality data.[a] (*Note that this is a separate issue to the requirement for general working hypotheses such as those described in Section 7.1.2 above that identify key assumptions against which monitoring outcomes can be tested.*) Some background on the criticisms of hypothesis testing and the rationale for using it in water quality monitoring are given in box 7.2.2, 'Hypothesis testing in environmental monitoring and assessment'.

a See also the Monitoring Guidelines Sections 2.4.2 and 6.4.2; the latter touches on alternative procedures

Box 7.2.2 Hypothesis testing in environmental monitoring and assessment

There has been some argument against the use of hypothesis testing tools in environmental assessment programs (e.g. Suter 1996). While hypothesis testing is not always either necessary or appropriate, much of the argument about its use (or misuse) is mis-directed. The argument is, in part, that hypothesis tests will only tell us after the event that something 'dreadful' has happened. However, the real issue is not whether hypothesis testing is appropriate, but whether the criteria by which tests are made (and for which sampling programs are designed) are sufficient or appropriate. An appropriately designed and executed sampling program intended to detect early warning signals will provide early warning whether analysed through hypothesis testing models or other procedures. Therefore it is important to make a satisfactory definition of objectives and decision criteria for each monitoring program. In real life there is a continuum or spectrum of conditions from undisturbed to disturbed; defining statistical boundaries along this spectrum to specify changes that are 'acceptable' and unacceptable to the stakeholders (as is done by the AUSRIVAS model bands) is strongly advocated, especially where clear 'break-points' in the meaning of ecological variables are not well documented.

A related issue is whether the inferences of impacts or changes should be based on dichotomous alternatives or a continuum of conditions. Suter (1996) and Stewart-Oaten (1996a,b) infer that hypothesis tests are constrained to test only two alternatives. However, there is no reason why those alternatives cannot be but two of a range of conditions along a continuum, the test being to (perhaps progressively) detect whether a response variable has moved from one condition to the next. In this case, the dichotomous test would be used only to test whether a particular threshold along a continuum had been crossed.

In summary, it is most important to choose appropriate 'performance criteria' for impact assessments or monitoring programs. If the criteria by which a management action will be triggered are inappropriate or insensitive or too coarse, then the issue of which tool to choose for statistical analysis becomes irrelevant.

For hypothesis testing to be useful in making decisions, the user needs to negotiate how much change in the indicator represents 'background noise'. In formal terms this means stipulating the null hypothesis ('no change') in terms of an effect size, as explained in the next section. That is, the null hypothesis is best thought of as the condition representing no *important* change in the value of the indicator, where 'importance' is determined by the context of the problem being monitored or assessed. Similarly, Type I and Type II errors are minimised by setting a suitable level of statistical significance when testing differences or change.

Some management programs will be oriented towards restoration or rehabilitation. In these circumstances the monitoring program will be seeking to prove that the values of the indicators are similar to those defined by the reference conditions.[a] In formal terms such programs will be trying to prove the null hypothesis (no 'important change' from reference conditions), although this is formally impossible. Hypothesis testing frameworks have been developed for such situations (they are sometimes called 'bioequivalence tests' in medicine and toxicology) and these are outlined briefly in box 7.2.3, 'Application of bioequivalence testing'.

a See Section 3.1.4

Box 7.2.3 Application of 'bioequivalence testing' for environmental restoration

Where statistical hypothesis-testing procedures are being used to analyse the data, it may be useful to re-cast the test using the framework of 'bioequivalence testing'. This procedure has been used in medical contexts (e.g. Westlake 1988, Chow & Liu 1992) and has recently been applied to environmental restoration in the USA. It is clearly explained by McDonald and Erickson (1994).

The problem with testing for recovery using a conventional hypothesis test is that the investigator is attempting to 'prove' the null hypothesis that the selected indicator in the disturbed site(s) has the same value as in the control or reference sites. However, failure to reject a null hypothesis does not constitute proof.

Tests of bioequivalence solve this problem by recasting the question so that the undesirable outcome, that the disturbed site differs substantially from the reference (i.e. the sites are *not* 'bioequivalent'), becomes the 'null hypothesis'[15] and evidence is sought to reject this hypothesis in favour of the alternative, that the impacted site is similar to the reference (i.e. the sites *are* 'bioequivalent'). Formally, the hypotheses are framed in terms of the ratio of the values of the indicator in the disturbed site and the reference site. If recovery has been achieved, the ratio should be sufficiently close to 1, and there should be strong evidence against the 'null hypothesis' which is then rejected in favour of the alternative after conducting the appropriate statistical test.

Under bioequivalence testing, a Type I error results in incorrectly deciding that the sites are bioequivalent when they still differ by an important amount (i.e. inadequate recovery, a false sense of security), whereas a Type II error results in deciding that the sites still differ when in fact they are similar (i.e. adequate recovery, false alarm). Note that with this technique stakeholders still must negotiate an effect size; users need to stipulate how different sites can be before they are declared 'non-bioequivalent'. In formal terms a critical value of the ratio of the indicator between the sites needs to be stipulated.

Stage 1: The nature of the change and its context; the use of hypothesis testing

The criteria used for making a decision depend on the level of protection assigned. As explained in Section 3.1.3 the level of protection depends on the condition of the ecosystem (condition 1, condition 2 or condition 3); specific guidance on how the level of protection affects decision criteria is given in Section 3.1.3.2 and table 3.1.2, while Section 3.1.8 elaborates on condition 3 ecosystems.

[15] Technically, the term 'null hypothesis' is usually reserved for the equivalence of a test statistic under different conditions, whereas in a bioequivalence test the investigator is quantifying the evidence against a proposition of non-equivalence under the different conditions.

In real life there is a continuum or spectrum of conditions from 'undisturbed' to 'disturbed'. Any indicator's response to disturbance is likely to vary according to the strength of that disturbance, whereas the decision about whether or not an impact has occurred is a point on that continuum. In other words, managers and stakeholders need to determine how much change from the unimpacted or pre-impact condition is acceptable.

The environmental assessment objectives (table 3.2.1) determine the point along the continuum at which an environmental impact is deemed to have occurred. For example, monitoring based on early detection of impact will have a different emphasis from monitoring geared towards assessing the ecological importance of an impact that has already happened. For early detection, a decision must be made *before* the level of change becomes harmful; otherwise the change may be irreversible. By contrast, to assess the importance of, say, an accidental ecological impact, the monitoring team must decide whether the level of acceptable change has been exceeded and by how much. In this situation the decision criterion is at the point of harmful change rather than some smaller value. In general, however, the emphasis will be on setting the decision criteria at a level that prevents harmful effects from occurring in the first place.

a See Section 2.1.3 for management goals

Thus a very important part of setting decision criteria is knowing what management actions will be taken if an impact is detected. The management goals[a] that managers have established provide most of this context, and some of the issues that may affect these goals are outlined in Section 3.1.3.3. For example, if a condition 2 ecosystem is being managed to conserve the population of a recreationally important fish, and the threat is a persistent contaminant with the potential to reduce the fecundity of the fish, then the decision criteria for the water quality indicators need be set at values which are smaller than those which begin to affect the reproduction of the fish; this will allow sufficient lead time for managers to act before the population of fish are affected. Much scientific judgement is involved in this process, and the actual values used as decision criteria will depend on a number of modifying factors (e.g. chemical speciation of toxicants); such matters are covered in more detail for each of the broad classes of indicators in Sections 3.2–3.5.

b Section 3.1.3.1 and table 3.1.2

c, d See also footnote 2, page 2–9

For ecosystem condition 1 (high conservation/ecological value) a criterion of 'no change beyond natural variability' is prescribed for biological indicators, physical and chemical stressors and sediments.[b] Operationally, this still requires users to stipulate how much change can be expected under 'natural' conditions, because this natural variation constitutes an acceptable level of change in the ecosystem.[c] Note that it is still necessary to decide on an effect size (see the next sub-section) explicitly and to ensure that sampling is intensive enough to detect effects larger than the acceptable natural changes in the chosen indicators, and avoid Type II errors. Note that the determination of the acceptable level of change may have both scientific and social elements.

For those who are new to environmental assessment, defining an acceptable level of change may seem weak, especially when management insists there must be 'no change' in the indicator. A criterion of 'no change' cannot be used operationally because it requires the user to prove the null hypothesis — an impossibility, as mentioned above. However, it is possible to state some level of change in an indicator below which it is not important to reject the null hypothesis of 'no change'.[d] This requires stakeholders to be explicit about what level of change in

the indicator is regarded as harmless or acceptable. In formal terms this process involves specifying an effect size, which is described in the next sub-section.

Stage 2: Specifying the size of the effect

The values of all the indicators used in these Guidelines vary naturally in space and time, and estimates of their true values can only be made via samples. Accordingly some observed changes in the indicators are likely to be ecologically trivial. The problem for water quality monitoring is to detect *non-trivial* changes in the chosen indicators soon enough to allow management to act. This means that monitoring programs need to be sensitive enough to detect modest rather than large changes in the indicators.

In formal terms, therefore, we need to identify the maximum amount of change in the indicator that is tolerable before we reject the null hypothesis (no important change) in favour of the alternative hypothesis (important or unacceptable change). This level of ecologically important change is sometimes called the *critical effect size*,[a] but for brevity we refer to it as the *effect size*.[b] Some of the procedures in these Guidelines have an implicit effect size; the relationship of guideline trigger values to the concept of effect size is described at the end of this subsection.

a See box 2.3 in Section 2.2.1

b See box 7.2.4

Box 7.2.4 Effect sizes are implicit in some procedures

For some procedures, the effect size and error rates tend to be implicit in the methods and are less amenable to the procedure of using scalable decision criteria described in this section.

For example, when comparing test data to a guideline trigger value, the 'effect size' may be implied by the choice of percentiles used in the comparison. See Section 7.4.4 for a full discussion of this and the trade-offs between Type I and Type II errors made in this procedure.

Similarly, in the AUSRIVAS procedure for rivers, notions of effect size and error rates are inherent in the way the summary indices are compared with the reference conditions. See Section 7.3.3 for more discussion.

There are two components of effect size: its form and its magnitude (Cohen 1988, Mapstone 1995). The form of an effect is the statistical measure (e.g. mean, variance) that is expected to differ between control and impact sites, and the pattern of differences or trends that it is necessary to detect (Stewart-Oaten et al. 1986, Green 1989, Underwood 1991a,b). The magnitude of an effect is the size of the difference or change in mean, say, or variance that would be considered important.

It is difficult to be prescriptive about effect sizes in ecological assessment for two reasons. Firstly, there is little information about the relationships between contaminants and biological indicators in field conditions, especially in Australia and New Zealand. Secondly, the degree of change that is important depends on the environmental and social values that stakeholders are seeking to protect. Strategies for setting an effect size are discussed in box 7.2.5, 'Some suggestions for setting effect size'. This is not an exhaustive list, and other strategies may arise as experience in planning programs with these procedures increases.

Box 7.2.5 Some suggestions for setting an effect size

Where an indicator has intrinsic socio-economic value (e.g. it is a commercially or recreationally important species), then effect sizes can be set to ensure sustainable use of that indicator. However, many biological indicators have been selected because they are more sensitive than commercial species or because they are thought to be *ecologically* important rather than of economic value. For example, seagrass is not used directly by humans in Australia and New Zealand, but is an important indicator because of the habitat it provides and the number of species it supports.

Existing research, or similar impacts, preferably in comparable regions, can provide information about the relationship between the indicator and size of potential impact, especially if existing impacts can be found on a gradient from mild to extreme. For example, a variety of sewage treatment plants may be present in a river basin with differing degrees of sewage treatment. Pilot data relating indicator levels and type of treatment could be used in stakeholder consultations to correlate stakeholders' expectations of acceptable sewage treatment with change in the indicator. In some cases, simulation models can use these data to estimate how much an indicator might change under different scenarios.

For many indicators in ecosystems in Australia and New Zealand, however, such background data are unlikely to be available. This will inevitably involve some judgement by the planners of a program, and an arbitrary but conservative effect size will need to be specified (e.g. Humphrey et al. 1995). This should be done explicitly, and at the beginning of the program. Any change to the effect size later in the program must be openly and explicitly negotiated and fully justified on scientific grounds.

Once the level of acceptable change has been negotiated, the degree of change in the indicator may need to be set to a smaller value so that management actions can be implemented before harmful and irreversible effects occur. When the effect size is being set, such issues as the fate and persistence of the contaminant and time-lags between a contaminant event and a measurable change in the biological indicator should be considered. Allied to these issues are selection of appropriate indicator(s),[a] and assessment of the relative costs of erroneously missing an effect of the stipulated size (Type II error) and erroneously concluding an impact occurred when, in fact, it did not (Type I error) (see next subsection).

a See Section 8.1

For the non-biological indicators in Sections 3.3–3.5, the guideline trigger values listed are the best currently available estimates of ecologically low-risk levels for those indicators.[b] These trigger values make an implicit statement about effect size: data from the test waterbody which are lower than the trigger value are thought to pose little risk to the ecosystem. Depending upon the management goals, stakeholders may need to negotiate different trigger values. There will also be situations where trigger values are exceeded. In these cases, more complex monitoring designs are called for,[c] and the steps outlined here for negotiating effect sizes will need to be followed.

b Section 7.4.4

c Section 3.1.5

Stage 3: Specifying the error rates relative to the costs of those errors

Having stipulated an effect size, the stakeholders then need to minimise the risk of two potential outcomes — in statistical terms, the Type I and Type II errors. These errors can arise because the indicators we use are sampled rather than measured completely, meaning that we are working with information which is necessarily incomplete. The first potential error is to declare that an impact has occurred

(i.e. the effect size has been exceeded), when really there has been no actual change that was ecologically important. The second potential error is to miss an ecologically important change. The probabilities of each error are conventionally denoted by the Greek letters α (for Type I) and β (for Type II).[a]

a See box 2.3 in Section 2.2.1

The challenge is to ensure that sufficient data are collected to detect the change stipulated in the effect size while, on the other hand, not expending too many resources on sample sizes that will detect ecologically trivial changes in the indicator. Inevitably, resources are scarce, so all monitoring programs will need to balance these two errors.

Conventionally, α has been set at 0.05 or smaller and few programs have stipulated β (Toft & Shea 1983, Fairweather 1991, Mapstone 1995). Although some recommendations for α and β are conservative default values for ecosystem conditions 1 and 2 (for biological indicators in Section 3.2.4), it must be emphasised that ideally these error rates should be *negotiated* rather than accepted uncritically. The most important part of this negotiation is to ensure that the *balance* between these two types of errors is acceptable to stakeholders in the process. To this end, these Guidelines recommend Mapstone's (1995, 1996) proposal that the *ratio* of these two errors is negotiated as part of refining the design of a monitoring program. This process requires iteration between stakeholders, but should be transparent, accountable and, above all, should take place before the final monitoring or assessment program is put in place (Mapstone 1995).

In outline, the choice of α and β involves four steps. First, establish the relative importance or cost of the consequences of each type of error. Second, set the ratio of the critical Type I and Type II errors relative to their costs (if there is insufficient information to estimate the costs of the errors, Mapstone suggests they should be weighted equally). Third, negotiate desired values of α and β with reference to the ratio established in the previous step with the stakeholders. Fourth, design a sampling program to meet the desirable Type II error rate, β, established in the previous step, given the effect size which has been specified earlier; this allows the sample size and details of the design to be finalised. Mapstone (1995) details two alternative decision procedures that can be followed once data have been collected and analysed.

Ideally, these negotiations should include a number of potential indicators simultaneously. In the process of balancing Type I and Type II errors, some indicators will inevitably prove much more costly than others if the two error rates are to be kept low. In such cases, stakeholders are faced with a choice: either discard the costly indicators in favour of those that will detect the stipulated effect size more cheaply, or, if the costly indicators have to be included in the program for some reason, increase the sizes of the two errors while maintaining the *ratio* between the errors. The only way to reduce the sizes of these errors is to increase the sampling intensity. Maintaining the ratio between the errors ensures that Type I errors are not minimised at the expense of increasing Type II errors — i.e. that the monitoring program does not lose power to detect an important change at the expense of being conservative about the probability of incorrectly declaring that an important change has occurred.

In two situations in these Guidelines, this negotiation of the balance between Type I and Type II errors is implicit; these are outlined in box 7.2.4.

The choice of the best sampling program is not a trivial issue. The strongest evidence will come from designs that have extensive baseline data collected before the suspected or potential impact takes place, and will involve simultaneous monitoring in multiple control sites. The weakest evidence will result from programs with limited or no pre-impact data. In all situations, inferences and assessments can be strengthened by including multiple lines of evidence.[a] The power of any statistical tests employed may be improved by including multiple indicators in a multivariate analysis, depending on the pattern of responses amongst the indicators (Green 1989).

a See Section 3.2.4.1

7.2.4 Sampling protocols and documentation

From figure 7.1.1, once the sampling program has been finalised, sampling can begin. This should take place according to standard or tested protocols. Section 8.1 and Appendix 3 of Volume 2 provide a list of protocols for biological indicators, while Section 8.3.6 outlines sources for protocols to be used in direct toxicity tests for toxicants. Procedures for sediment toxicity testing seem to be less well developed, but references to and guidance through the recent literature are provided in Section 8.4.3. Protocols for measuring physical, chemical, biological and ecotoxicological parameters of sediments are described in general terms in Section 3.5 and Appendix 8 of Volume 2, and Chapter 4 of the Monitoring Guidelines, with references to detailed literature.

Quality assurance and quality control (QA/QC) procedures should be part of any sampling protocol. Quality control (QC) and quality assurance (QA) are different but related concepts. In the context of these Guidelines, *quality control* means devising and implementing safeguards to minimise the corruption of data. These safeguards must be installed at every step of the process from project definition to the decision on whether measured concentrations compare acceptably with the guidelines. *Quality assurance* means testing the effectiveness of these safeguards.

In any QA/QC program, chain of custody documentation is essential to ensure that errors can be traced. Chapter 4 of the Monitoring Guidelines discusses QA/QC in some depth for key points for chemical, physical and toxicant indicators; Section 7.4.3 below refers to that source.

7.2.5 Sample processing and analysis

Analysis here refers to the processing of sample units (e.g. field or laboratory measurement of analytes in a water sample, counting and identifying invertebrates in a benthic sample) rather than the statistical analysis of the resulting data. As with sampling, standard or rigorously tested protocols should be used; many protocols also detail methods of analysis. Because of their reliance on often complex, rigorous laboratory procedures, more specific guidance on analytical procedures is provided for physical and chemical stressors, toxicants and sediments.[b]

b See also Sections 7.4 and 7.4.3

Again, QA/QC procedures are often described in protocols, and QA/QC is also discussed in Chapter 5 of the Monitoring Guidelines. The monitoring team should document at least the analytical steps and the date and location of the analyses, the identities of the analysts, the methods used and the type and status of any equipment used for the analysis.

Similar care in QA/QC should also be used during data entry and data management. Most modern database software packages provide value-checking routines, and clear procedures should be established to manage, track, back-up and archive data files. Clear documentation of the features of the data (e.g. the units that the data are entered in, codes used for missing or 'below detection limit' data) need to be kept with the data files.

7.2.6 Data analysis, evaluation and reporting

The first step in evaluating the data will be the formal statistical analysis. For some indicators, the data may need to be adjusted to account for modifying factors (e.g. effect of pH on chemical speciation). The process of analysing the data is also iterative, with the first step being to examine the distributions of the variables and to check for outliers,[a] to see whether the data meet the assumptions of the intended analysis. Sometimes transformation of the data can solve distributional problems. Most statistical procedures have a second diagnostic stage after the procedure has been applied (e.g. examination of residuals after fitting a regression or general linear model). If these diagnostics show that the assumptions of the procedure have been violated, alternative statistical models may need to be developed. Chapter 6 of the Monitoring Guidelines discuss these issues, while the involvement of professional statisticians is invaluable in ensuring the rigour of these analyses.

a See Chapter 6 of the Monitoring Guidelines

Once the statistical analyses have been completed, the results need to be interpreted in the context of the key interacting environmental processes and the environmental assessment objectives of the program. Reporting of the results needs to clear, concise, unambiguous and timely to allow management to act on the results. It is essential to disseminate the results to stakeholders in a form that is readily understandable, and some general recommendations are given in Chapter 7 of the Monitoring Guidelines. Reporting will often include recommendations on modifications to the program if it is a continuing program, thereby closing the feedback loop in figure 7.1.1 (this chapter).

7.3 Specific issues for biological indicators

This section addresses issues specific to biological indicators that need to be borne in mind when designing monitoring and assessment programs. Section 7.3.1 outlines the issues for *uni*variate indicators: these consist of a *single* response variable such as the density or biomass of phytoplankton, measures of community metabolism, or chemical/biochemical markers in aquatic organisms. Multivariate indicators[a] refers to measures of community composition or structure where the response variable is usually based on some measure of community similarity which, in turn, is computed from the abundance (structure) or presence or absence (composition) of many taxa within the ecosystem. Examples include measures of the community structure of diatoms, macroalgae and invertebrates. AUSRIVAS, the rapid biological assessment technique for Australian rivers, is also based on multivariate community composition data, but is a special case of a design class where inferences are based exclusively on spatial controls alone. Section 7.3.3 discusses how the outputs of AUSRIVAS relate to the issues raised in Section 7.2.

a See Section 7.3.2

7.3.1 Issues for univariate indicators

Most of the key issues are raised in Section 7.2 and in Chapters 3 and 4 of the Monitoring Guidelines. Univariate indicators are easily analysed using conventional and novel statistical procedures, provided the key assumptions are met.[b] However, two issues are worth emphasising.

b Ch 6 of the Monitoring Guidelines

First, many of the classical techniques of statistical analysis assume independence of sample units through space and time. Biological indicators may violate these assumptions temporally because of the longevity of indicators or spatially because of dispersal or behaviour of indicators. If these phenomena are likely within the monitoring program, then professional statistical advice should be sought to either adjust the sampling regime or select statistical modelling tools that can accommodate these spatial and/or temporal autocorrelations (Legendre & Legendre 1998).[c]

c See also the Monitoring Guidelines Sections 6.5.2, 6.6.1

Second, data which consist of counts of organisms sometimes result in a large number of zero values (i.e. when there are no organisms in the sampling unit). The frequency distributions of such data are typified by a 'spike' at zero, then a mode at some larger, non-zero value. Assuming that the sampling unit or device is appropriate for the size and behaviour of the organism (most of the protocols recommended in Chapter 8 give advice on sizes of sampling units; for a more thorough discussion see, for example, Andrew & Mapstone 1987), such data are usually problematic for most statistical techniques. Some recent advances have been made in this area; as this is still an active developing area of applied statistics, professional advice should be sought when choosing and using these techniques.

7.3.2 Issues for multivariate indicators

Multivariate data for biological indicators in these Guidelines typically consist of either the presence or absence of taxa or their abundances across the sample units. These data can then be summarised as similarities (or dissimilarities) between each pair of sample units. The Bray–Curtis measure, among a few others, has been demonstrated to be the best choice for such biological data (Faith et al. 1987), and

sometimes transformation of the data is desirable before the similarity measure is computed, as described in the protocols in Appendix 3 (Vol 2).

The result of these computations is a triangular matrix of similarity values. These are not easily analysed in conventional statistical procedures such as analysis of variance or regression. However, one situation which is amenable to the use of similarity measures in more conventional procedures is where the 'control' and 'impact' sites can be paired, on rivers for example. In figure 7.3.1, there is a pair of sites on each tributary river which are comparable in terms of habitat and separated by similar distances on each river. The similarities between the upstream and downstream sites on each river could be computed for a number of times before the disturbance; if the similarities computed between the paired sites decreased after the start of the impact relative to the similarities between paired sites on the control rivers, then an impact is likely to have occurred. Examples using this design are Faith et al. (1995) and Davies and Nelson (1994).

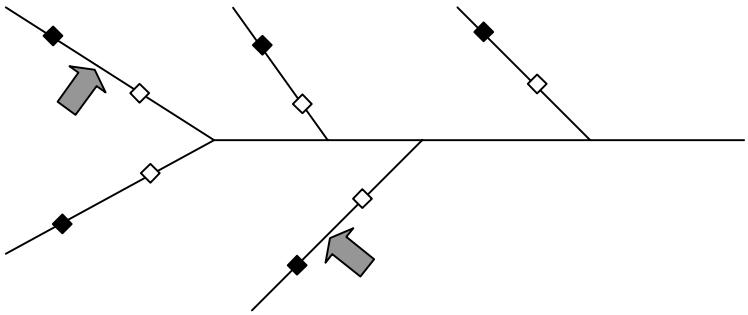


Figure 7.3.1 Schematic diagram of a river system with paired upstream (black diamonds) and downstream (white diamonds) sites on each tributary. Grey arrows indicate locations of disturbances.

Pairing of sites in this way is not always possible, however. Permutation tests that are analogues of some of the simpler conventional techniques (Smith 1998, the ANOSIM of Clarke & Green 1988, Clarke & Warwick 1994) have been used (e.g. Smith 1994). Significance testing of multivariate data based on similarity measures using permutation tests is rapidly developing (see Legendre & Legendre 1998 for an overview; Legendre & Anderson 1999 for an attempt at analysing multifactorial data). It is likely that methods for the analysis of similarity data in complex designs will become available in the near future.

The more conventional methods of analysing dissimilarity data have focused on displays of the data via such techniques as multidimensional scaling, principal components analysis and correspondence analysis (review: Legendre & Legendre 1998; brief description of principal components analysis and multidimensional scaling in Sections 6.5.4 and 6.6.3 of the Monitoring Guidelines). Inferences have been made purely on the basis of striking patterns in such displays, and Green (1979) argued that obvious patterns in such graphs were likely to correspond to large impacts. Such a procedure obviously lacks the sensitivity required for some assessment objectives. However, these displays remain an important tool for interpretation and communication after a formal hypothesis test via a randomisation or permutation procedure (Clarke & Warwick 1994).

7.3.3 Use of AUSRIVAS

7.3.3.1 Outline of AUSRIVAS

AUSRIVAS is a rapid biological assessment procedure developed for rivers and streams (Schofield & Davies 1996) similar to the British RIVPACS system (Wright et al. 1993). It currently uses macroinvertebrate data, but the use of other taxa (e.g. diatoms and fish) is being researched. Its applicability to wetlands and to New Zealand rivers is also being investigated.

AUSRIVAS is a specialised example of a monitoring design that relies on spatial information alone to infer whether a disturbance has caused an impact. In general terms the problem can be stated thus: to judge whether a particular site has been disturbed by some activity or event, other, apparently undisturbed, sites that are similar in their environmental attributes must be found to act as a standard or *control* for comparison. The site suffering the supposed disturbance is designated the *test site*, while the sites acting as controls are called *reference sites*.

In AUSRIVAS a large number of reference sites with as high an environmental quality as possible have been identified across a wide variety of river types and ecosystems, sampled for their macroinvertebrates and had their habitats characterised by a standard set of physical and chemical variables that are largely unrelated to likely pollutants. This set of reference sites has then been classified according to their biota to produce groups of sites containing similar fauna. A numerical analysis has then been used to identify the environmental attributes which describe each group of reference sites. Now, any test site requiring assessment has its environmental attributes compared with those of the reference sites to determine which group or groups of reference sites it resembles most closely. The fauna of these corresponding reference sites is then compared with the test site: if the test site supports fewer taxa than are predicted by the reference sites, it is judged to be disturbed.

7.3.3.2 Sampling protocol and issues about effect size and sensitivity

The AUSRIVAS protocols modify much of the advice given in Sections 7.2.3 to 7.2.6 above. Site selection, the methods of stratifying habitats within sampling sites, the timing of sampling and the analytical methods and outputs are all specified in protocols for each state and territory in Australia. The summary indices and recommendations for reporting procedures are also standardised, and the sampling, sorting and identification steps are subject to QA/QC programs. The software for analysing the data is maintained and developed at a central location accessible via the AUSRIVAS home page.

Decisions about effect size are implicit in the procedure. The degree of impact upon a site is judged by the values of summary indices relative to a stipulated percentile of the reference sites that act as spatial controls. If a site scores a value on these indices that is smaller than 90% of the values recorded for reference sites, the fauna is deemed to be lacking some of the families of invertebrates that could be expected at that site. Although designation of such a percentile threshold expresses an effect size in terms of how deviant a site is from reference conditions, it is analogous to but not exactly equivalent of the process of setting Type I and II error rates for conventional statistical procedures.

Nevertheless, the issue of how far a potentially-disturbed site can deviate from reference or control conditions before an impact is deemed to have occurred must still be resolved with stakeholders. Strategies similar to those described above for procedures based on statistical methods can be employed: i.e. use of existing information, examining the response of the indicator variable to known impacts of different degrees of severity, and use of pilot data in simulation modelling. Note that indicators used in rapid, broad-scale methods are often quite coarse (e.g. use of family-level rather than species-level identifications). Thus the threshold value at which the decision is made that an impact has occurred should take account of the harmfulness of the potential impact, its reversibility and the time-lags between an event and the implementation of actions to prevent harm. The threshold value may need to be set at a more conservative value than that deemed acceptable by stakeholders so that management has time to react and prevent irreversible harm.

7.3.3.3 Application and cautions

AUSRIVAS has been promoted in these Guidelines as ideally suited for the rapid, cost-effective, first-pass determination of the *extent* of a problem or potential problem, e.g. as applied to broad-scale land-use issues. Earlier, a note of caution was provided for use of the method in applications other than these,[a] particularly for detecting impacts of a minor nature and for site-specific assessments where the method requires additional testing and the addition of more data. A perspective on some of the limitations of the approach is provided below, together with comments on ongoing data collection and proposed research and development aimed at improving the sensitivity and broadening the application of this procedure.

a See box 3.2.1, Section 3.2.2.1/3

An important aspect of AUSRIVAS is the availability and selection of suitable reference sites. In some regions of Australia it is easy to find reference sites on rivers and streams draining relatively intact catchments. Unfortunately, large regions of Australia have been subject to broad-scale impacts and there are no 'near-pristine' sites from which to select biogeographically relevant reference sites (e.g. wheat belt of Western Australia, lowland reaches of the Murray-Darling Basin). Thus, in AUSRIVAS, the least impacted sites of such regions have been targeted to provide reference sites for setting targets for rehabilitation of the more degraded sites; however, this does not solve the problem of assessing the degree of degradation of the reference sites themselves. Without pre-impact data, this task is outside the ambit of routine prescriptive procedures and would require a variety of situation-specific case studies to arrive at some assessment. The issue is being addressed as part of the current Australia Wide Assessment of River Health (AWARH), which aims to report on the ecological condition of around 4000 Australia river sites by the year 2000 using AUSRIVAS.

A related but more tractable problem results when a test site has no close environmental equivalents in the reference database. Therefore, an important initial step in evaluating a test site is a statistical comparison between its environmental attributes and those of all the reference sites: if it has no sites with similar attributes in the reference set, no further assessment can be made, i.e. there are insufficient sites in the database that can be regarded as a 'control' (Furse et al. 1987). The current AUSRIVAS software (available from the AUSRIVAS homepage) contains a testing routine to assess whether test sites fall within the 'domain' of the existing reference site set on which the bioassessment models are based. It then must be

decided whether the test site warrants the added expense of adding sufficient comparable reference sites to the database to enable an assessment to be made.

A potential drawback is the relatively large number of reference sites that must be sampled to build reliable models for predicting the presence or absence of the target organisms. This is particularly relevant to site-specific assessments, where adequate characterisation of the local reference condition is critical. The development of AUSRIVAS is predicated on the collection of a large amount of reference site data nationally, and it is anticipated that the geographic spread, as well as the spatial density of sites, will gradually increase to improve the applicability of the predictive models.

Another important aspect for some Australian streams is the natural inconstancy of animal community composition amongst years. Thus, 'high' temporal variability of macroinvertebrate communities over large parts of Australia, particularly semi-arid and northern regions prone to drought and/or cyclonic disturbance (Humphrey et al. 2000), may reduce to some (as yet unknown) extent the sensitivity of 'static' models derived for these locations. To this end, it is recommended that test site assessment using the AUSRIVAS protocol should be done in parallel with reference ('control') site assessment to assess the degree of natural temporal change in macroinvertebrate community composition and compare it with the summary index value for the test site.

In some regions of Australia it is clear that some reference sites are naturally depauperate; that is, the number of macroinvertebrate taxa is low. For procedures such as AUSRIVAS, where the final reporting indices are based on the ratio of the number of taxa observed to the number of taxa expected, this poses potential problems for the sensitivity and robustness of the final assessment, even at species level.

Finally, AUSRIVAS and related procedures (Reynoldson et al. 1995) are rapid assessment tools and will only detect impacts that are severe enough to eliminate taxonomic groups of organisms. The formal hypothesis testing associated with conventional statistical methods has no clear analogue here. This procedure uses a suite of reference sites to predict the expected composition of families of invertebrates at a test site; if the test site has fewer families than expected based on the distribution of reference site values, it is deemed to be disturbed. Nevertheless, several basic considerations of survey design (sample and site replication, etc.) still apply to assessments or surveys conducted with AUSRIVAS. These considerations become particularly important at small spatial scales (i.e. a specific activity, development, point-source disturbance, within a catchment) where stronger inference and greater sensitivity to impact may be required. If AUSRIVAS is to be adopted in these situations, it must be conducted in a design framework that has adequate sample and site replication to enable the study objectives to be met. If necessary, aspects of the rapid biological assessment protocol may need to be adapted or modified so that the data gathered are amenable to both AUSRIVAS and quantitative assessment.[a]

[a] Complementary roles for quantitative and rapid assessment in monitoring programs are recommended in Section 7.2.1.1 above

7.4 Specific issues for physical and chemical indicators (including toxicants) of water and sediment

This section outlines issues specific to physical and chemical indicators (including toxicants) of water and sediment that need to be borne in mind when designing monitoring and assessment programs. The issues in Sections 7.4.1 and 7.4.3 are comprehensively discussed in Chapters 3–6 of the Monitoring Guidelines — see table 7.4.1 for a checklist of these issues and appropriate cross-reference to the Monitoring Guidelines.

Table 7.4.1 Checklist for sampling and analysis of physical and chemical indicators with cross-reference to details provided in the Monitoring Guidelines.

Issue	Chapter or section from the Monitoring Guidelines
Representative sampling	Chapter 3, Chapter 4
spatial boundaries	3.3.1
scale	3.3.2
duration	3.3.3
patterns of sampling	3.4.1
selection of sites	3.4.2
frequency of sampling	3.4.3
numbers of samples	3.4.4, A5.1.10
Surface water sampling	4.3.1, 4.3.2
hydrology, flow variations, runoff	3.4.3, 3.4.3.2, 4.3.1
stratification	3.4.1.2, 3.4.2.1
human effects on contaminant loads and timing	3.4.3.2
automatic samplers	3.4.3.2, 4.3.2
time of day	3.4.3, 3.4.3.2
Sediment sampling and sediment sample handling	4.3.1, 4.3.5, 3.4.2.1, 5.5.8
potential for contamination	4.3.5, 4.3.1
suspended sediments	4.3.5
Sample storage and handling	4.5, 4.6, 4.3
Chemical speciation	5.5.8.2, Tables 4.5 & 5.2
Bioavailable concentration vs total concentration	3.5
Quality assurance/Quality control in the field	4.6 and subsections
chain of custody	4.6.1
training of staff	4.7.2
quality assurance samples: blanks	4.6.3.1
quality assurance samples: replicates	4.6.3.2
quality assurance samples: spiked samples	4.6.3.3
pilot trial	3.4, 3.4.2, 3.4.4
equipment	4.6, 4.6.2, 4.6.3.1
sample transport	4.6.1, 4.6.2, 4.6.3.1
site access	3.4.2, 4.2, 4.7.1, 4.7.3
occupational health and safety	4.7 and subsections
analytes	5.3
cleaning and calibration	4.3.1, 4.3.2.1, 4.3.6, 4.6.1, 4.6.2
protocols	4.3.7, 4.6.2
Quality assurance/Quality control in the laboratory	5.5 and subsections
chain of custody	5.4.1.2
occupational health and safety	5.6 and subsections
training of staff	5.6.3
quality assurance program	5.5.5 and subsections
quality assurance samples	subsections of 5.5.5
matrix compatibility	5.5.5.1
accurate recording of data	5.4, 6.2

In addition to the cross-references provided to sampling and analysis of sediments in table 7.4.1, a protocol describing key aspects of collection and laboratory analysis of sediment samples is provided in Appendix 8 of Volume 2, while advice on comparing sediment 'test' data with default guideline values is provided in Section 7.4.4.4 below.

a See also the Monitoring Guidelines Section 6.4.3

These Guidelines emphasise the use of guideline trigger values for assessing the environmental significance of physical and chemical indicators. The statistical procedure for comparing test data and a trigger value is described in Section 7.4.4.[a] The generic considerations for sampling design given in Section 7.2 also apply to physical and chemical indicators.

7.4.1 Hydrology and representative sampling

b See Monitoring Guidelines Chapters 3 & 4

Sampling of waters and sediments must be representative.[b] The challenge is to sample in enough detail to outline a picture of the natural variations in time and space and to reliably detect deviations from this natural 'background' variation.

Natural variations in surface waters and groundwaters, whether flowing or standing, can affect the values of physical and chemical indicators. For example, all water bodies can form vertical or horizontal layers of differing temperature or salinity that may or may not need to be sampled separately, according to the sampling plan. Currents and the lateral and vertical movements of different water masses also need to be considered during the planning of field sampling, analyses and study design. Natural periodicity, and the timing of industrial discharges into water bodies, and the considerable effects of runoff in inland waters can make large differences to the loads and concentrations of physical and chemical indicators, and must also be planned for.

c See Vol 2, App 8 and Monitoring Guidelines Sections 4.3, 4.3.5

In sediments,[c] the sampling plan and study design must consider the effects of natural layering, mixing, and variations in particle size and porosity on the indicator being sampled. The likelihood of disturbance and cross-contamination during sampling must not be forgotten. Suspended sediments need to be collected in a representative manner (Batley 1989), as do sediment pore waters.

For all samples, precautions must be planned and taken to prevent the values of the indicators changing during storage and transport.

7.4.2 Chemical speciation in water samples

The issue of the chemical form of physical and chemical indicators (that is, the compound(s) of the indicator present in the sample) are relevant regardless of the use envisaged for the water. Speciation (the form of the chemical) assumes critical importance where the environmental value concerns ecosystem protection or human health. The form of the indicators needs to be determined and those chemical species that are likely to affect the environmental value must be identified. In the past, total (i.e. unfiltered) concentrations were measured and compared with guideline values on the understanding that this approach probably overestimates the amount of deleterious form(s) of the indicator. This approach to protection may be overconservative. A refinement is to measure and compare *total filtered* concentrations. This, too, is a conservative approach (though less so) because the diversity of chemical forms of a physical and chemical indicator in the solution may have different effects on an environmental value.

There are at least two ways to resolve the speciation problem.

- Determine the indicator using an analytical method that is specific to the chemical species. While this is an improvement on using total filtered concentrations, it requires the species or range of species detected by the method to be arbitrarily defined as a surrogate for the species affecting (usually detrimentally) the environmental value. An example of this approach is the use of anodic stripping voltametry in the determination of copper. The fraction determined under operationally defined conditions is identified as labile forms which, in turn, are believed to be the forms in which copper is most bioavailable.
- Use *thermodynamic speciation modelling*. One requirement of this mathematical tool is that all aqueous chemical species that may be important to the chemical form of indicators be measured. This usually increases the analytical requirements because of the inclusion of chemical species that would not otherwise be determined. The technique requires that the system measured is in equilibrium, and that the equilibrium is the same as that existing at the time of sampling. This has implications for preservation, transport and storage of samples. The specification and interpretation of thermodynamic speciation models is complex and requires considerable facility in the use of computers, and in the interpretation of chemical data. A more detailed discussion of speciation modelling is beyond the scope of these Guidelines.

7.4.3 Quality Assurance and Quality Control (QA/QC)

Quality control and quality assurance were defined generically in Section 7.2.4. A specific formal statement of quality control for physical and chemical indicators is this:

> The overall objective of quality control in the measurement of physical and chemical variables is the determination of the *exact* indicator concentration that existed at a specifically defined location at the time the sample was taken. In most cases this requirement extends to the chemical speciation of the indicator.

Neglect of QA/QC is probably the most important reason for the unreliability of most historical chemical data.

Protocols for field and laboratory aspects of sampling must be followed carefully, as discussed in the Monitoring Guidelines.[a] QA/QC begins with the choice and training of competent field and laboratory staff; it includes the choice and maintenance of field and laboratory equipment and vehicles. It extends to the checking of analytical methods and analytical performance, the tracking of each sample throughout sampling and analysis, and the accurate recording of data in the final database.

a See Monitoring Guidelines Chapters 4, 5 and 6

7.4.4 Comparing test data with guideline trigger values

7.4.4.1 Physical and chemical stressors

This section provides a summary of the approach recommended for comparing results from a test site with a guideline trigger value. Details of the method are contained in Appendix 7 of Volume 2; Section 6.4.3 of the Monitoring Guidelines touches on it also. There are a number of common statistical methods that are

potentially applicable for this purpose, although experience suggests that the assumptions underpinning many 'conventional'[16] statistical tests are often violated by water quality data (and sometimes quite seriously so). Section 6.3.4 of the Monitoring Guidelines recommends transformations to correct specific problems, although the action required will depend very much on the characteristics of the data at hand. This lack of consistency in the way site-specific data may be processed and interpreted is an impediment to the development of a simple, straightforward trigger rule.

Compounding this difficulty is the usual requirement to specify the magnitude of change in a particular statistical parameter (e.g. mean, variance, percentile) that is deemed to be 'significant' — either ecologically or statistically or both. The quantification of a *minimum* effect size that can be claimed to be ecologically important is difficult. With respect to the trigger rule outlined in this section, this issue of ecological importance is discussed further below and more generally in Section 7.2.3.3. The important observation to note at this stage, however, is that exceedances of the trigger values are an 'early warning' mechanism to alert managers of a potential problem. ***They are not intended to be an instrument to assess 'compliance' and should not be used in this capacity***.

During the development of a suitable trigger mechanism, considerable attention was given to the following design requirements:

- explicit recognition of the inherent (and usually large) variability of natural systems;
- robustness under a wide range of operating conditions and environments;
- no, or only weak, distributional assumptions about the population of values from which the test and reference data are obtained;
- known statistical properties, consistent with and supporting the monitoring objectives of this document;
- ease of implementation and interpretation;
- suitability for visual display and analysis;
- intuitive appeal.

The recommended trigger-based approach for physico-chemical stressors may be stated as follows.

A trigger for further investigation will be deemed to have occurred when the median concentration of *n* independent samples taken at a test site exceeds the eightieth percentile of the same indicator at a suitably chosen reference site. Where suitable reference site data do not exist, the comparison should be with the relevant guideline value published in this document.

This rule satisfies the first dot point above since it is statistically-based and acknowledges natural background variation by comparison to a reference site. Its robustness derives from the fact that it accommodates site-specific anomalies and uses a robust statistical measure as the basis for triggering. No assumptions are

[16] In this context, the term conventional is used to denote statistical procedures based on the *general linear statistical model* having normally distributed errors.

required to be made about the distributional properties of the data obtained from either the test or reference sites. The computational requirements of the approach are minimal and can be performed without the need for statistical tables, formulae, or computer software. As demonstrated later in this section, the temporal sequence of trigger events is readily captured in a simple plot.

It should be understood that the trigger protocol is responsive to shifts in the *location* (i.e. 'average') of the distribution of values at the test site. While differences in shape of the reference and test distribution may be important in some instances, this is a secondary consideration that is not specifically addressed by this protocol. It is also important to note that the role of the 80th percentile at the reference site is to simply quantify the notion of a 'measurable perturbation' at the test site. The protocol is not a statistical test of the equivalence of the 50th and 80th percentiles *per se*. The advantages of using a percentile of the reference distribution are 1) it avoids the need to specify an absolute quantity and 2) because the reference site is being monitored over time, the trigger criterion is being constantly updated to reflect temporal trends and the effects of extraneous factors (e.g. climate variability, seasonality).

Implementation of the trigger criterion is both flexible and adaptive. For example, the user can identify a level of routine sampling (through the specification of the sample size *n*) that provides an acceptable balance between cost of sampling and analysis and the risk of false triggering. The method also encourages the establishment and maintenance of long-term reference monitoring as an alternative to comparisons with the default guideline values provided in Section 3.3 that do not account for site-specific anomalies.

The remainder of this section addresses sampling issues, data requirements, computational procedures and statistical properties associated with the proposed method. The mathematical detail associated with computation of Type I and Type II errors may be found in the Annex of Appendix 7 of Volume 2. Worked examples of the computations and performance aspects of the trigger rule are provided in Appendix 7 (Volume 2).

1 Data requirements at the reference sites

Prior to implementing the trigger rule, the user will need to have addressed some data collection issues.

- *Reference site selection*: selection of (a) suitable reference site(s) has been addressed in Section 3.1.4.
- *Minimum data requirements at the reference site*: *a minimum* of two years of contiguous monthly data at the reference site is required before a valid trigger value can be established. Until this minimum data requirement has been established, comparison of the test site median should be made with reference to the default guideline values identified in Section 3.3 of this document.

2 Computation of the 80th percentile at the reference site

The computation of the 80th percentile at the reference site is always based on the *most recent* 24 monthly observations. The procedure is as follows:

(i) arrange the 24 data values in *ascending* (i.e. lowest to highest) order,

(ii) take the simple average (mean) of the 19th and 20th observations in this ordered set.

3 Updating the reference site data and 80th percentile

Each month, a new reading at the reference (and test) site is obtained. The reference site observation is appended to the end of the original (i.e. unsorted) time sequence. Steps (i) and (ii) from 2 above are applied to the most recent 24 data values. Note, even though only the most recent two years of data is used in the computations, no data should be discarded.

Maintenance of the complete data record will allow longer-term statistics to be computed. For example, after five years of monthly monitoring, *all* sixty observations could be used to compute the overall 80th percentile. This could serve as a useful benchmark against which the 'rolling' monthly percentiles could be compared for evidence of trends.

4 Data requirements at the test site

A feature of the method is the flexibility it provides the user for the allocation of resources to the sampling effort. As previously mentioned, there is no fixed requirement to monitor at a reference location (i.e. the default guideline values can be applied). Similarly, the choice of sample size at the test site is arbitrary, although there are implications for the rate of false triggering. For example, a minimum resource allocation would set n=1 for the number of samples to be collected each month from the test site. It is clear that the chance of a *single* observation from the test site exceeding the 80th percentile of a reference distribution which is *identical* to the test distribution is precisely 20%. Thus the Type I error in this case is 20%. This figure can be reduced by increasing n. For example, when n=5 the Type I error rate is approximately 0.05. The concomitant advantage of larger sample sizes is the reduction in Type II error (the probability of a false no-trigger). So-called 'power curves' are provided in Appendix 7 (Volume 2) to assist in understanding the consequences upon error rates of a particular sampling strategy at the test location.

5 Computation of the median at the test site.

The median is defined to be the 'middle' value in a set of data such that half of the observations have values numerically greater than the median and half have values numerically less than the median. For small data sets, the sample median is obtained as either the single middle value after sorting in ascending order when n is *odd*, or the average of the two middle observations when n is *even*.

6 Ecological importance

The proposed trigger rule does not purport to define or represent an ecologically important change. As previously explained, the trigger approach is an early warning mechanism to alert the resource manager of a *potential* or *emerging* change that should be followed up. Whether or not the actual change in condition at the test site has biological and/or ecological ramifications can only be ascertained by a much more comprehensive investigation and analysis. To make this distinction clear, the concept of a **measurable perturbation** is introduced. Our *de facto* definition of a measurable perturbation is that it is the magnitude of the shift between the 50th and 80th percentiles *at a reference site*. While the definition is arbitrary, it does have broad acceptance and intuitive appeal among experts. It should also be noted that the *statistical significance* associated with a change in condition equal to or greater than a measurable perturbation would require a separate analysis.

7 Performance characteristics

It is important that the statistical performance characteristics of any test or decision-making rule are documented and understood to avoid unduly conservative or liberal triggering.

The foregoing discussion makes no assumptions regarding the shape of the reference and test distributions. Without this knowledge, a formal calculation of Type I and Type II errors is not possible. However, as a general principle, increasing the frequency of collection of independent samples will reduce the magnitude of both errors. A more complete discussion of the performance characteristics of the recommended approach is provided in Appendix 7 of Volume 2.

8 On-going monitoring — the control chart

The foregoing has been provided to assist with the month-by-month comparisons. It is suggested that these monthly results be plotted in a manner indicated in figure 7.4.1 below. This provides a visual inspection of all results and helps identify trends, anomalies, periodicities and other phenomena. The methods in Chapter 6 of the Monitoring Guidelines can be used to model trends and other data behaviour if required.

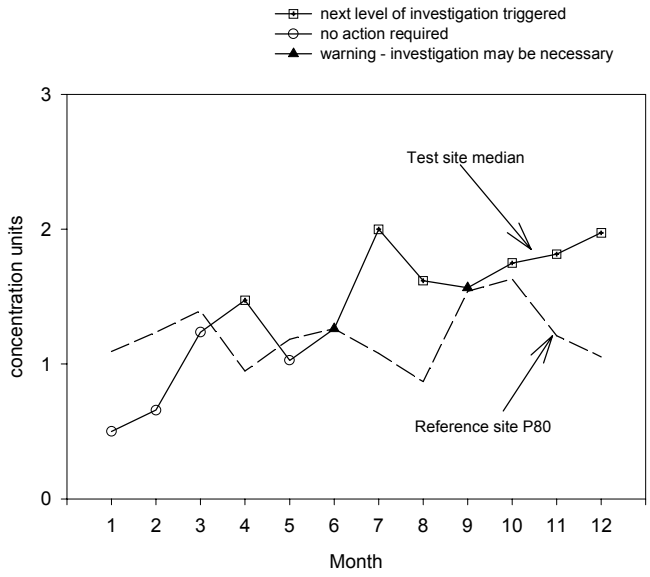


Figure 7.4.1 Control chart showing physical and chemical data (Y axis) for test and reference sites plotted against time, and recommended actions

9 Comparing test data against single guideline (default values)

In the absence of suitable reference site data (as defined in step 1 above), the median of the test site data is to be compared with the default guideline value identified in Section 3.3.2.5 of this document. This guideline value has been computed as the 80th percentile of the amalgamation of a number of historical data sets across broad geographical regions. Unlike the comparison with a locally-derived 80th percentile, the guideline value is static and will not reflect any local spatial and/or temporal anomalies. Reference site monitoring is strongly advocated

if these effects are considered to represent a significant source of departure from the guideline value.

Figure 7.4.2 below illustrates the difference in control charting procedures when the guideline value is used in place of a trigger obtained using the 80th percentile from reference site monitoring.

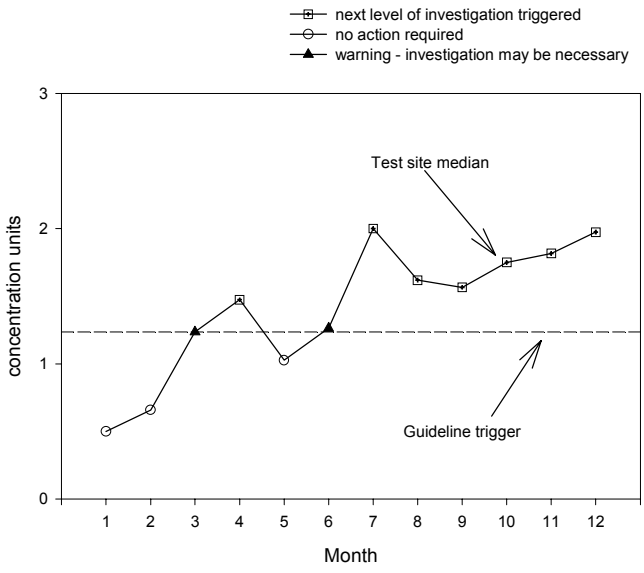


Figure 7.4.2 Control chart showing physical and chemical data (Y axis) for test site plotted against default trigger value, time and recommended actions

7.4.4.2 Toxicants

This section describes the general needs for comparing toxicant test data with guideline trigger values. Conceptually, toxicants and 'physical and chemical stressors' are subcategories of the same class of potentially hazardous indicators, being properties or (usually) constituents of the aquatic environment. However, the treatment of these groups for guideline purposes is different. Specifically, toxicants are usually compared with a single default trigger value, less commonly with a background or reference distribution. The default values are prepared by analysis of a comprehensive set of available ecotoxicological data. Physical and chemical stressors at a test site are usually compared with those at a reference site. The latter reference-comparison approach, however, has its parallels in measurement programs for toxicants, as described in 1 below.

1 Background data that may supplant guideline default trigger values

Some surface waters will contain concentrations of toxicants that may naturally exceed the default guideline trigger values tabulated in Section 3.4. Where this is the case and as recommended in Section 3.4.3.2, new trigger values should be based on background (or baseline) data. (Note that 'background' in this case, refers to *natural* toxicant concentrations that are unrelated to human disturbance.) As a matter of course, gathering of background data is always recommended, at least in

the initial stages of a water quality management program, to establish whether or not concentrations of toxicants are naturally high.

Toxicant concentrations may vary seasonally. Because of this and the need to be confident about the best estimate of background concentrations, it is recommended that background data be gathered on a monthly basis for at least two years. In all respects, data requirements and collection are the same as for physical and chemical stressors, as described above.[a] Until this minimum data requirement has been established, comparison of the test site median should be made with reference to the default guidelines identified in Section 3.4.3 of this document.

a See step 1, Section 7.4.4.1

For those months, seasons or flow periods that constitute logical time intervals or events to consider and derive background data, the 80th percentile of background data (from a minimum of 10 observations) should be compared with the default guideline value. This 80th percentile value is used as the new trigger value for this period if it exceeds the default guideline value provided in Section 3.4.3 of this document. Test data are compared with the new trigger values using the same principles as outlined in steps 2–8 above for physical and chemical stressors.

Where background toxicant values fall consistently below default trigger values, sampling intensity at these sites could be reduced after a suitable period (e.g. two years).

2 Comparing test data with default guideline values

In practical terms, the method for comparing toxicant test data with default guideline values should be similar to the approach recommended in step 9 above for physical and chemical stressors.[b] However, it is recommended that a more conservative approach should apply to the comparison of toxicant test data with default guideline values. Specifically, it is recommended that action is triggered if the 95th percentile of the test distribution exceeds the default value (or stated differently, no action is triggered if 95% of the values fall below the guideline value). The more stringent approach is recommended here because, unlike physical and chemical stressors, toxicant default values are based upon actual biological effects data and so by implication, exceedance of the value indicates the potential for ecological harm. Note that because the proportion of values required to be less than the default trigger value is very high (95%), a single observation greater than the trigger value would be legitimate grounds for action in most cases, even early in a sampling program.

b Section 7.4.4.1

7.4.4.3 Physical and chemical (including toxicant) data gathered from surface waters 'upstream' of the test site

In many situations, particularly where additional human use activities are present 'upstream' of the test site of interest, the regular collection of data from upstream of the test site will be necessary. These data will be compared with the test data of interest to assist in determining the source and cause of any possible elevated toxicant concentrations found at the test site. Where there are multiple sources of toxicants along a water-course, catchment managers will need to establish appropriate data analysis and assessment procedures to apply.

7.4.4.4 Sediments

The application of the decision tree[a] reverts to reference or background site concentrations if these exceed the trigger values. The selection of an acceptable reference site for water quality studies has been discussed earlier.[b] Basically the same considerations apply to sediments, with the additional option, that a reference or background condition can also be established from measurements at depths in sediment cores below observed concentration excursions.

a See Section 3.5

b Section 3.1.4

While temporal variability is used to characterise water quality parameters at a reference site, this is clearly inappropriate for sediments where the accumulation rates are typically below 1 cm/y. It is more appropriate to use spatial variability, either based on depth profiles at a test site or an appropriate number of surface sediment samples, to characterise a site. Sites will typically contain a range of grain sizes, and determining median concentrations and 80^{th} or 95^{th} percentile values may distort any comparison. It is important that in comparing test and reference sites, samples with a similar grain-size distribution are used. Normalising to a fine grain size (e.g. <63 μm) is inappropriate, as the normalised value will have less of an impact on biota when diluted with coarser sediments that usually contain lower contaminant concentrations.

The spatial scale over which the reference and test site measurements are taken is a matter for decision by stakeholders, based on sound scientific judgement. The heterogeneity of sediment samples with respect to contaminants largely mirrors the differences in grain size. Defining the size of the test site will be a regulatory responsibility, in terms of the spatial extent of contaminated sediment that is acceptable in the region of interest. As a guide, the spatial extent of a test site may be a geographical feature, for example, a delta or an embayment within a harbour. Alternatively, a test site may comprise a recognised ecological habitat, for instance a riffle zone in a stream or a defined area of fine sediment in a lake. In a large water body the test site might be larger than in a narrow river or creek, where biota might have difficulty in avoiding the contamination. The area of any reference site should be comparable to that of the test site, and the grain size must be similar.

Because of the poor reliability of the sediment trigger values it is difficult to be prescriptive about how these can be compared with test values. The same applies to the comparison of reference site values with test sites, where comparisons of reference median or 80^{th} percentile with the test site median may be equally appropriate in giving an estimate of the relative concentrations, which is really all that is required in the case of sediments. However, where sediment samples within a test site clearly exceed trigger values, or are reasonably inferred to be ecologically hazardous, these Guidelines recommend additional sampling to more precisely delineate contaminated zones within the site.

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Appendix 1 Acronyms and glossary of terms

Acronyms

ACC	Acceptable Contaminant Concentration
ACR	Acute-to-chronic ratios
AE	Alcohol ethoxylated surfactants
AES	Alcohol ethoxyolated sulfate surfactants
AGPS	Australian Government Publishing Service
ANCA	Australian Nature Conservation Agency
ANZECC	Australian and New Zealand Environment and Conservation Council
ANZFA	Australia New Zealand Food Authority
AQUIRE	Aquatic Toxicity Information Retrieval Database
ARMCANZ	Agricultural and Resource Management Council of Australia and New Zealand
ASSAC	Australian Shellfish Sanitation Advisory Committee
ASSCP	Australian Shellfish Sanitation Control Program
ASQAP	Australian Shellfish Quality Assurance Program
ASTM	American Society for Testing and Materials Designation
AUSRIVAS	Australian River Assessment Scheme
AVS	Acid volatile sulfide
BACI	Before– After, Control–Impact
BACIP	Before–After, Control–Impact Paired
BCF	Bioconcentration factor
BEDS	Biological effects database
BOD	Biological oxygen demand
BOM	Biodegradable organic matter
CCL	Cumulative Contaminant Loading Limit
CCME	Canadian Council for Ministers of the Environment
CCREM	Canadian Council for Resource and Environment Ministers
CEC	Cation exchange capacity
CFU	Colony forming units
COAG	Council of Australian Governments
COD	Chemical oxygen demand
CSIRO	Commonwealth Scientific & Industrial Research Organisation
DASET	Department of Arts, Sport, Environment and Territories
DCC	Desirable contaminant concentration
DEST	Department of Environment, Sport and Territories
DISR	Department of Industry, Science and Resources
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DTA	Direct toxicity assessment
DWAF	Department of Water Affairs and Forestry
DUAP	Department of Urban Affairs and Planning

EC	Electrical conductivity
ECLs	Environmental concern levels
EEZ	Exclusive Economic Zone
EIA	Environmental Impact Assessment
EIS	Environmental Impact Statement
ENSO	El Nino Southern Oscillation
ERIN	Environmental Resource Information Network
eriss	Environmental Research Institute of the Supervising Scientist
ES	Effect size
ESD	Ecologically sustainable development
ESP	Exchangeable sodium percentage
EV	Environmental value
FNARH	First National Assessment of River Health
GESAMP	Joint Group of Experts on the Scientific Aspects of Marine Pollution
ICM	Integrated catchment management
ICPMS	Inductively coupled plasma mass spectrometry
IMO	International Maritime Organisation
ISQG	Interim sediment quality guideline
LAS	Linear alkylbenzene sulfonates
LOEC	Lowest observed effect concentration
LWRRDC	Land and Water Resources Research and Development Corporation
MATC	Maximum acceptable toxicant concentration
MBACI	Multiple Before–After, Control–Impact
MBACIP	Multiple Before–After, Control–Impact, Paired
MDBC	Murray Darling Basin Commission
MHSPE	Ministry for Housing, Spatial Planning and the Environment
MPC	Maximum permitted concentration
NATA	National Association of Testing Authorities of Australia
NHMRC	National Health and Medical Research Council
NICNAS	National Industrial Chemicals Notification and Assessment Scheme
NIWA	National Institute of Water and Atmospheric Research
NOAA	US National Oceanic and Atmospheric Administration
NOEC	No observable effect concentration
NPDES	National Pollutant Discharge Elimination System
NRC	National Research Council
NRHP	National River Health Program
NSSP	US National Shellfish Sanitation Program
NSWDWR	NSW Department of Water Resources
NSWEPA	NSW Environmental Protection Authority
NWQMS	National Water Quality Management Strategy
OECD	Organisation for Economic Co-operation and Development
PAR	Photosynthetically available radiation
PCBs	Polychlorinated biphenyls

PQL	Practical quantitation limit
QA/QC	Quality assurance/quality control
RBA	Rapid biological assessment
RIVPACS	Riverine Invertebrate Prediction and Classification System
SACC	State Algal Coordinating Committee
SAR	Sodium adsorption ratio
SCARM	ARMCANZ Standing Committee for Agricultural and Resource Management
SCEP	ANZECC Standing Committee on Environmental Protection
SEM	Simultaneously extracted metals
SoE	State of the Environment
SPM	Suspended particulate matter
TAN	Total ammonia nitrogen
TCM	Total catchment management
TDS	Total dissolved solids
TIE	Toxicity identification & evaluation
TTM	Total toxicity of mixtures
UNESCO	United Nations Education Scientific and Cultural Organization
USEPA	United States Environmental Protection Agency
VEPA	Victoria Environment Protection Authority
WADEP	Western Australian Department of Environmental Protection
WAEPA	WA Environmental Protection Authority
WAWA	WA Water Authority (now split between the Water Corporation Western Australia and Waters and Rivers Commission (WA)
WET	Whole effluent toxicity
WHO	World Health Organization
WQG	Water quality guideline
WWW	World Wide Web

Glossary

Term	Definition
Abalone/paua	*Haliotis* spp. of shellfish
Abiotic	The non-living components of a system (see biota)
Absorption	In chemistry: Penetration of one substance into the body of another In biology: The act of absorbing (i.e. to take in as fluids or gases through a cell membrane). To take a substance (e.g. water, nutrients) into the body through the skin or mucous membranes or, in plants, through root hairs.
Acceptable Contaminant Concentration (ACC)	The ACC is the maximum concentration (mg/L) of contaminant in irrigation water which can be tolerated for a shorter period of time (20 years) assuming the same maximum annual irrigation loading as DCC
Acclimation	Short-term adaptation of individual organisms to specific environmental conditions
Acid-soluble metal	The concentration of the metal that passes through a 0.45 µm membrane filter after the sample is acidified to pH 1.5–2.0 with nitric acid
Acidic	Having a high hydrogen ion concentration (low pH)
Acid volatile sulfides (AVS)	Sulfides in sediments that liberate hydrogen sulfide on reaction with cold dilute acid (mainly FeS or MnS in sediments)
Acute toxicity	Rapid adverse effect (e.g. death) caused by a substance in a living organism. Can be used to define either the exposure or the response to an exposure (effect).
Acute–chronic ratio	The species mean acute value divided by the chronic value for the same species
Additive toxicity	The toxicity of a mixture of chemicals that is approximately equivalent to that expected from a simple summation of the known toxicities of the individual chemicals present in the mixture (i.e. algebraic summation of effects).
Adsorption	The taking up of one substance at the surface of another
Aeration	Any process where a substance becomes permeated with air or another gas. The term is usually applied to aqueous liquids being brought into intimate contact with air by spraying, bubbling or agitating the liquid.
Aerobic	Of organisms requiring oxygen for respiration or conditions where oxygen is available
Aesthetic	Aspects of, say, a water body, that can be considered beautiful or pleasant to the senses
'Aggressive' carbon dioxide	The amount of dissolved carbon dioxide in excess of that required to stabilise the bicarbonate ion present in water
Algae	Comparatively simple chlorophyll-bearing plants, most of which are aquatic and microscopic in size
Alkalinity	The quantitative capacity of aqueous media to react with hydroxyl ions. The equivalent sum of the bases that are titratable with strong acid. Alkalinity is a capacity factor that represents the acid-neutralising capacity of an aqueous system.
Alkaloids	Nitrogenous organic bases found in plants
Allochthonous	Organic material that is developed or derived outside a particular waterbody
Ambient waters	All surrounding waters, generally of largely natural occurrence

Amphipods	Invertebrates belonging to the order Crustacea
Anaerobic	Conditions where oxygen is lacking; organisms not requiring oxygen for respiration
Analytes	The physical and chemical species (indicators) to be determined
Anion	Negatively charged ion
Anionic	Characteristic behaviour or property of an ion that has a negative charge. Anions move to the anode in electrolysis.
Anode	The electrode where oxidation occurs
Antagonism	A phenomenon in which the effect or toxicity of a mixture of chemicals is less than that which would be expected from a simple summation of the effects or toxicities of the individual chemicals present in the mixture (i.e. algebraic subtraction of effects)
Anthropogenic	Produced or caused by humans
A posteriori	Identifying causes by inductive reasoning based on actual effects, consequences or facts (i.e. from observation, experience or experiment)
A priori	Predicting effects by deductive reasoning based on causes rather than actual observation, experience or experiment
Aquaculture	Commonly termed fish farming, but it broadly refers to the commercial growing of marine (mariculture) or freshwater animals and aquatic plants
Aquatic ecosystem	Any watery environment from small to large, from pond to ocean, in which plants and animals interact with the chemical and physical features of the environment
Aquifer	An underground layer of permeable rock, sand or gravel that absorbs water and allows it free passage through pore spaces
Assessment factors	A unitless number applied to the lowest toxicity figure for a chemical to derive a concentration that should not cause adverse environmental effects; also called 'application factor' or 'safety factor', the size of the AF varies with the type of data (section 8.3.3.2)
Assimilation	The incorporation of absorbed substances into cellular material
Assimilative capacity	The maximum loading rate of a particular pollutant that can be tolerated or processed by the receiving environment without causing significant degradation to the quality of the ecosystem and hence the environmental values it supports
Ataxia	Inability to coordinate voluntary movement
Autochthonous	Organic material that is developed or produced within a particular waterbody
Autotrophy	The nutrition of organisms that produce their own organic constituents from inorganic compounds, using energy from sunlight or oxidation processes (e.g. most plants and some bacteria)
Avoidance threshold	The lowest concentration of a substance that causes a fish to actively move away from the source
Barramundi	*Lates calcarifer*
Baseline data (studies)	Also called pre-operational data (studies); collected (undertaken) before a development begins
Benthic	Referring to organisms living in or on the sediments of aquatic habitats (lakes, rivers, ponds, etc.)
Benthos	The sum total of organisms living in, or on, the sediments of aquatic habitats

Binding sites	Sites on a substrate where chemical interaction with an indicator (qv) may occur. The interaction may be electrostatic, polar, hydrogen bonding or covalent bonding.
Bioaccumulation	General term describing a process by which chemical substances are accumulated by aquatic organisms from water, either directly or through consumption of food containing the chemicals
Bioassay	A test that exposes living organisms to several levels of a substance that is under investigation, and evaluates the organisms' responses
Bioavailable	The fraction of the total of a chemical in the surrounding environment that can be taken up by organisms. The environment may include water, sediment, soil, suspended particles, and food items.
Biochemical (or biological) oxygen demand	The decrease in oxygen content in mg/L of a sample of water in the dark at a certain temperature over a certain of period of time which is brought about by the bacterial breakdown of organic matter Usually the decomposition has proceeded so far after 20 days that no further change occurs. The oxygen demand is measured after 5 days (BOD_5), at which time 70% of the final value has usually been reached.
Bioclogging	Clogging of irrigation infrastructure due to excessive algae or microbial growth
Bioconcentration	A process by which there is a net accumulation of a chemical directly from water into aquatic organisms resulting from simultaneous uptake (e.g. by gill or epithelial tissue) and elimination
Bioconcentration factor (BCF)	A unitless value describing the degree to which a chemical can be concentrated in the tissues of an organism in the aquatic environment At apparent equilibrium during the uptake phase of a bioconcentration test, the BCF is the concentration of a chemical in one or more tissues of the aquatic organisms divided by the average exposure concentration in the test.
Biocorrosion	Corrosion caused by microorganisms through formation of biofilms on the metal surface
Biodiversity (biological diversity)	The variety of life forms, including the plants, animals and micro-organisms, the genes they contain and the ecosystems and ecological processes of which they are a part
Biofilm	Layer of materials created by microorganisms on an underwater surface
Biological assessment	Use and measurement of the biota to monitor and assess the ecological health of an ecosystem
Biological community	An assemblage of organisms characterised by a distinctive combination of species occupying a common environment and interacting with one another
Biomagnification	Result of the processes of bioconcentration and bioaccumulation by which tissue concentrations of bioaccumulated chemicals increase as the chemical passes up through two or more trophic levels The term implies an efficient transfer of chemicals from food to consumer, so that residue concentrations increase systematically from one trophic level to the next.
Biomass	The living weight of a plant or animal population, usually expressed on a unit area basis
Biosolids	Sewage sludge, organic residuals remaining after domestic sewage treatment

Biota	The sum total of the living organisms of any designated area
Biotoxins	A toxin (a poison) which originates from a living thing (a plant, animal, fungi, bacteria, etc.)
Bioturbation	The physical disturbance of sediments by burrowing and other activities of organisms
Bivalve	A mollusc with a hinged double shell
Black bream	*Acanthopagrus butcheri*
Black tiger prawn	*Penaeus monodon*
Bloom	An unusually large number of organisms per unit of water, usually algae, made up of one or a few species
Blue mussel	*Mytilus edulis*
Buffer	A solution containing a weak acid and its conjugate weak base, the pH of which changes only slightly on the addition of acid or alkali
Buffering capacity	A measure of the relative sensitivity of a solution to pH changes on addition of acids or base
°C	Degrees Celsius
Carcinogen	A substance that induces cancer in a living organism
Catchment	The total area draining into a river, reservoir, or other body of water
Cathode	The electrode where reduction occurs
Cation	Positively charged ion
Cation exchange capacity (CEC)	A measure of a soil's ability to retain cations
Cationic	The characteristic behaviour or property of an ion with a positive charge. Cations move to the cathode in electrolysis.
Chelate	The type of co-ordination compound in which a central metal ion is attached by co-ordinate links to two or more non-metal atoms in the same molecule, called ligands
Chemical oxygen demand	The amount of oxygen required to oxidise all organic matter that is susceptible to oxidation by a strong chemical oxidant
Chlorination	1) The process of introducing one or more chlorine atoms into a compound 2) The application of chlorine to water, sewage or industrial wastes for disinfection
Chronic	Lingering or continuing for a long time; often for periods from several weeks to years. Can be used to define either the exposure of an aquatic species or its response to an exposure (effect). Chronic exposure typically includes a biological response of relatively slow progress and long continuance, often affecting a life stage.
Chronic value	The geometric mean of the lower and upper limits obtained from an acceptable chronic test or by analysing chronic data using a regression analysis A lower chronic limit is the highest tested concentration that did not cause an unacceptable amount of adverse effect on any of the specified biological measurements, and below which no tested concentration caused unacceptable effect An upper chronic limit is the lowest tested concentration that did cause an unacceptable amount of adverse effect on one or more biological measurements and above which all tested concentrations also caused such an effect
Cladoceran	Water flea; zooplankton belonging to the fourth order of the Branchiopoda, the Cladocera

Colloid	Material in solution typically 1 nm–100 nm in diameter. Colloidal particles do not settle out of solution through the force of gravity. Organic colloidal matter is considered especially important in the transport of inorganic substances such as P through the soil profile.
Community	An assemblage of organisms characterised by a distinctive combination of species occupying a common environment and interacting with one another
Community composition	All the types of taxa present in a community
Community metabolism	The biological movement of carbon in an ecosystem, involving two processes, production (*via* photosynthesis) and respiration
Community structure	All the types of taxa present in a community and their relative abundances
Complexation	The formation of a compound by the union of a metal ion with a non-metallic ion or molecule called a ligand or complexing agent
Compliance	Action in accordance with upholding a 'standard' (water quality)
Concentration	The quantifiable amount of chemical in, say, water, food or sediment
Condition indicators or targets	Indicators of the condition or state of the ecosystem. They are normally biological indicators (e.g. species composition, species abundance), but may also be physical or chemical indicators (e.g. dissolved oxygen concentration, temperature, flow duration). These often represent the *targets*, or *water quality objectives*, that need to be met in order to actually achieve the desired level of ecosystem protection.
Contaminant	Biological (e.g. bacterial and viral pathogens) and chemical (see Toxicants) introductions capable of producing an adverse response (effect) in a biological system, seriously injuring structure or function or producing death
Control	That part of an experimental procedure which is like the treated part in every respect except that it is not subjected to the test conditions. The control is used as a standard of comparison, to check that the outcome of the experiment is a reflection of the test conditions and not of some unknown factor.
Corrosion	Deterioration of surfaces through erosion processes such as the conversion of metals to oxides and carbonates
Cresylic	Acidic commercial mixture of phenolic materials boiling above the cresol range (greater than 240°C)
Criteria (water quality)	Scientific data evaluated to derive the recommended quality of water for different uses
Crop quality	With regard to inorganic contaminants, increased concentration of contaminant in plant tissue that while not phytotoxic, reduces the economic value of the crop due to increased residues
Cumulative	Resulting from successive additions at different times or in different ways
Cumulative Contaminant Loading Limit (CCL)	The CCL is the maximum contaminant loading in soil defined in gravimetric units (kg/ha); it indicates the cumulative amount of contaminant added, above which site-specific risk assessment is recommended if irrigation and contaminant addition is continued.
Cyanobacteria	A division of photosynthetic bacteria, formerly known as blue-green algae, that can produce strong toxins
Cyanosis	A blueness in the appearance of surficial tissues, generally due to a deficiency of oxygen bound to haemoglobin
Cytotoxic	Having an adverse impact on cells
Decision criteria	Criteria by which decisions will be made as a result of monitoring for potential impacts. (See also effect size, Type I error, Type II error)

Decision framework or decision tree	A series of steps for tailoring guideline trigger values to a specific site or region and for assessing water quality by considering the local or regional environmental factors that will modify the effect of the particular water quality parameter The decision frameworks or trees begin with the simplest steps and finish with the most difficult and expensive.
Depuration	Process that uses a controlled aquatic environment to reduce the level of pathogenic organisms that may be present in live shellfish
Desirable Contaminant Concentration (DCC)	The DCC is the maximum concentration (mg/L) of contaminant in irrigation water which can be tolerated assuming 100 years of irrigation based on stated irrigation loading assumptions
Detection limit	The smallest concentration or amount of a substance that can be reported as present with a specified degree of certainty by definite complete analytical procedures
Detritus	Unconsolidated sediments composed of both inorganic and dead and decaying organic material
Dinoflagellates	Major class of marine algae that move by flagella. They are often red in color, and can produce strong toxins that can kill many fish and other marine organisms.
Direct toxicity assessment (DTA)	The use of toxicity tests to determine the acute and/or chronic toxicity of waste water discharges or total pollutant loads in receiving waters. (Assesses the toxicity of mixtures of chemicals rather than individual chemicals.)
Diuresis	Increased discharge of urine
Diurnal	Daily
Divalent	Having a valence (combining power at atomic level) of two (e.g. calcium, Ca^{2+})
Dose	The quantifiable amount of a material introduced into an animal
Dysphagia	Difficulty in swallowing
Early detection	Measurable biological, physical or chemical response in relation to a particular stress, prior to significant adverse affects occurring on the system of interest.
Early life-stage test	28-day to 32-day exposures (60-day post-hatch for salmonids) of the early life stages of a species of fish from shortly after fertilisation through embryonic, larval and early juvenile development. Data are obtained on survival and growth.
EC_{50} (median effective concentration)	The concentration of material in water that is estimated to be effective in producing some lethal response in 50% of the test organisms. The LC_{50} is usually expressed as a time-dependent value (e.g. 24-hour or 96-hour LC_{50}).
EC_{se}	The electrical conductivity of the soil saturation extract
EC_s	The electrical conductivity of the soil solution at maximum field water content
$EC_{1:5}$	The electrical conductivity of a 1:5 soil:water extract
Ecological integrity (health)	The ‘health’ or ‘condition’ of an ecosystem The ability of an ecosystem to support and maintain key ecological processes and organisms so that their species compositions, diversity and functional organisations are as comparable as possible to those occurring in natural habitats within a region
Ecologically sustainable development	Development that improves the total quality of life, both now and in the future, in a way that maintains the ecological processes on which life depends
Ecosystem condition	Current or desired status of health of an ecosystem, as affected by human disturbance
Ecosystem health	In this document synonymous with ‘ecological integrity’ (qv)

Ecosystem-specific modifying factor	A factor that can influence (mostly reduce) the biological effects caused by a Particular toxicant or stressor
Effect size	The size of impact that would cause concern (or constitute an early warning). Often defined as a level of (ecological) change that is acceptable in comparison to a defined reference.
Effluent	A complex waste material (e.g. liquid industrial discharge or sewage) that may be discharged into the environment
Electrical conductivity	The ability of water or soil solution to conduct an electric current
Encrustation	Accumulation of material on surfaces through chemical or biological processes
End-points	Measured attainment response, typically applied to ecotoxicity or management goals
Endemic, endemism	Confined in occurrence to a local region
Enterococci	Any streptococcal bacteria normally found in the human intestinal tract; usually nonpathogenic
Environmental values	Particular values or uses of the environment that are important for a healthy ecosystem or for public benefit, welfare, safety or health and that require protection from the effects of pollution, waste discharges and deposits. Several environmental values may be designated for a specific waterbody.
Epilimnion	The uppermost layer of water in a lake, characterised by an essentially uniform temperature that is generally warmer than elsewhere in the lake, and by relatively uniform mixing by wind and wave action
Epilithon	Organisms attached to rocks, such as algae and lichens
Epiphyte	A plant that grows on the outside of another plant, using it for support only and not obtaining food from it
ESP	The exchangeable sodium content of a soil expressed as a percentage of the cation exchange capacity
Eukaryotes	An organism characterised by the presence of membrane-bound organelles (see prokaryote)
Euphotic	Of surface waters to a depth of approximately 80–100 m; the lit region that extends virtually from the water surface to the level at which photosynthesis fails to occur because of reduced light penetration
Euryhaline	Describes organisms that are capable of osmo-regulating over a wide range of salinities
Eutrophic	Abundant in nutrients and having high rates of productivity frequently resulting in oxygen depletion below the surface layer of a waterbody
Eutrophication	Enrichment of waters with nutrients, primarily phosphorus, causing abundant aquatic plant growth and often leading to seasonal deficiencies in dissolved oxygen
Evapotranspiration	The combined loss of water from a given area during a specified period of time by evaporation from the soil or water surface and by transpiration from plants
Exchangeable sodium percentage (ESP)	The sodium adsorbed onto clay mineral surfaces as a proportion of the total cation exchange capacity of those surfaces
Exposure	The amount of physical or chemical agent that reaches a target or receptor
Fate	Disposition of a material in various environmental compartments (e.g. soil or sediment, water, air, biota) as a result of transport, transformation and degradation
Field capacity	The greatest amount of water that it is possible for a soil to hold in its pore spaces after excess water has drained away

Flocculation	(1) The process by which suspended colloidal or very fine particles coalesce and agglomerate into well-defined hydrated floccules of sufficient size to settle rapidly (2) The stirring of water after coagulant chemicals have been added to promote the formation of particles that will settle
Flounder	*Rhombosolea* spp.
Flow-through system	An exposure system for aquatic toxicity tests in which the test material solutions and control water flow into and out of test chambers on a once-through basis either intermittently or continuously
Fluorosis	Chronic poisoning by fluorine
Fouling	Accumulation of material through chemical, physical or biological processes
Free carbon dioxide	The amount of dissolved carbon dioxide in excess of that required to stabilise the bicarbonate ion present in water
Freshwater shrimp	A decapod crustacean, including the genus *Macrobrachium* spp.
Gastropod	A mollusc of the Class Gastropoda, with a locomotive organ placed ventrally (e.g. snail and limpet)
Gilvin	The coloured component of dissolved organic matter in water. It is composed mainly of humic, fulvic and tannic compounds.
Green shell mussel	*Perna canaliculus*
Gross alpha (activity)	A measure of the concentration of alpha-particle emitting radionuclides in water. This is determined by standard techniques involving the evaporation of a water sample and measurement of the alpha activity of the residue.
Gross beta (activity)	A measure of the concentration of beta-particle emitting radionuclides in water. This is determined by standard techniques involving the evaporation of a water sample and measurement of the beta activity of the residue.
Groundwater	Water stored underground in rock crevices and in the pores of geologic materials that make up the earth's crust; water that supplies springs and wells
Guideline package	Decision trees that are applied to physical and chemical stressors and/or associated issues for aquatic ecosystems
Guideline trigger values	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.
Guideline (water quality)	Numerical concentration limit or narrative statement recommended to support and maintain a designated water use
Habitat	The place where a population (e.g. human, animal, plant, microorganism) lives and its surroundings, both living and non-living
Half-life	Time required to reduce by one-half the concentration of a material in a medium (e.g. soil or water) or organism (e.g. fish tissue) by transport, degradation, transformation or depuration
Hardness	The concentration of all metallic cations, except those of the alkali metals, present in water. In general, hardness is a measure of the concentration of calcium and magnesium ions in water and is frequently expressed as mg/L calcium carbonate equivalent.
Hazard	The potential or capacity of a known or potential environmental contaminant to cause adverse ecological effects

Helminth	Helminths are worms; the helminths discussed in this document are human and animal parasites
Hepatotoxin	Toxic substances which adversely affect the liver
Heterotrophy	The nutrition of plants and animals that are dependent on organic matter for food
High reliability guideline trigger values	Trigger values that have a higher degree of confidence because they are derived from an adequate set of chronic toxicity data (section 8.3.4) and hence require less extrapolation from the data to protect ecosystems
Humic substances	Organic substances only partially broken down that occur in water mainly in a colloidal state. Humic acids are large-molecule organic acids that dissolve in water.
Hydrogeology	Study of subsurface waters and with related geologic aspects of surface water
Hydrograph	Graphical representation of river or stream discharge or of water-level fluctuations in a well
Hydrolysis	(1) The formation of an acid and a base from a salt by the ionic dissociation of water (2) The decomposition of organic compounds by interaction with water
Hydrophilic	Having an affinity for water, readily absorbs water
Hydrophobic	Having little or no affinity for water, repels or does not absorb water
Hypolimnion	The region of a waterbody that extends from below the thermocline to the bottom of the lake; it is thus removed from much of the surface influence
Hypothesis	Supposition made from known facts as a starting-point for further investigation
Hypoxia	Deficiency of oxygen in tissues or in blood; anoxia
Incipient LC_{50}	The concentration of a chemical that is lethal to 50% of the test organisms as a result of exposure for periods sufficiently long that acute lethal action has essentially ceased. The asymptote (part of the toxicity curve parallel to the time axis) of the toxicity curve indicates the value of the incipient LC_{50} approximately.
Indicator	A parameter that can be used to provide a measure of the quality of water or the condition of an ecosystem
Ingestion	The swallowing or taking in of food material
Inorganic carbon	Generally, simple ions and molecules that contain carbon bonded only to inorganic atoms. Carbonates are the most common group, although the cyanide ion is also considered to be inorganic.
Interstitial	Occurring in interstices or spaces; applied to water and to flora and fauna living between sand grains and soil particles
Invertebrates	Animals lacking a dorsal column of vertebrae or a notochord
In vitro	Outside the intact organism; generally applied to experiments involving biochemical events occurring in tissue fragments or fractions in a laboratory
Ion	An electrically charged atom
Kuruma prawn	*Penaeus japonicus*
Langelier Saturation Index (SI)	An index based on the tendency of water to deposit or dissolve calcium carbonate. It relates the actual pH of water with the pH at which water is saturated with calcium carbonate ($SI = pH - pH_S$).
LC_{100}	Lowest concentration of a toxicant that kills all the test organisms

LC_{50} (median lethal concentration)	The concentration of material in water that is estimated to be lethal to 50% of the test organisms. The LC_{50} is usually expressed as a time-dependent value, e.g. 24-hour or 96-hour LC_{50}, the concentration estimated to be lethal to 50% of the test organisms after 24 or 96 hours of exposure.
LD_{50} (median lethal dose)	The dose of material that is estimated to be lethal to 50% of the test organisms. Appropriate for use with test animals such as rats, mice and dogs, it is rarely applicable to aquatic organisms because it indicates the quantity of a material introduced directly into the body by injection or ingestion rather than the concentration of the material in water in which aquatic organisms are exposed during toxicity tests.
Leachate	Water that has passed through a soil and that contains soluble material removed from that soil
Leaching	Where referred to in the salinity and sodicity section of Chapter 4, the downward movement of water and solutes below the root zone
Leaching fraction (LF)	The proportion of water applied to the surface of a soil (e.g. as irrigation or rainfall) that drains below the root zone in the soil profile
Lentic	Standing body of water such as a lake or pond
Lethal	Causing death by direct action. Death of aquatic organisms is the cessation of all visible signs of biological activity.
Level of protection	A level of quality desired by stakeholders and implied by the selected management goals and water quality objectives for the water resource
Life-cycle study	A chronic (or full chronic) study in which all the significant life stages of an organism are exposed to a test material. Generally, a life-cycle test involves an entire reproductive cycle of the organism. A partial life-cycle toxicity test includes the part of the life cycle observed to be especially sensitive to chemical exposure.
Ligand	A molecule, ion or atom that is attached to the central atom of a co-ordination compound, a chelate or other complex. May also be called complexing agent.
Liveweight	Weight of the living animal
LOEC (Lowest observed effect concentration)	The lowest concentration of a material used in a toxicity test that has a statistically significant adverse effect on the exposed population of test organisms as compared with the controls. When derived from a life-cycle or partial life-cycle test, it is numerically the same as the upper limit of the MATC.
LOEL (Lowest observed effect level)	The lowest concentration that produces an observable effect in a test species. Below this concentration there are no observed effects in the test species.
Long-term trigger value (LTV)	The maximum concentration of contaminant in irrigation water which can be tolerated assuming 100 years of irrigation, based on key irrigation loading assumptions
Lotic	Flowing waters (e.g. rivers and streams)
Low reliability guideline trigger values	Trigger values that have a low degree of confidence because they are derived from an incomplete data set (section 8.3.4.1). They are derived using either assessment factors or from modelled data using the statistical method. They should only be used as interim indicative working levels.
Macrophyte	A member of the macroscopic plant life of an area, especially of a body of water; large aquatic plant

Management goals	Long-term management objectives that can be used to assess whether the corresponding environmental value is being maintained. They should reflect the desired levels of protection for the aquatic system and any relevant environmental problems. Management goals will mostly be narrative statements focusing management on the relevant water quality objectives.
Marron	*Cherax tenuimanus*
MATC (Maximum acceptable toxicant concentration)	The maximum concentration of a toxic substance that a receiving water may contain without causing significant harm to its productivity or uses as determined by chronic toxicity tests
Maximum tolerable daily level (MTDL)	The dietary level that when fed for a limited period, will not impair animal performance and should not produce unsafe residues in produce for human consumption
Median	Middle value in a sequence of numbers
Mesotrophic	Water bodies or organisms which are intermediate between nutrient-rich and nutrient-poor
Metabolite	Any product of metabolism
Methylation	The introduction of methyl (CH_3) groups into organic and inorganic compounds
Methyl mercury	The most common form is the cation CH_3Hg^+ although $(CH_3)_2Hg$ also occurs. Both are extremely potent toxicants and can lead to secondary poisoning through biomagnification. They are usually formed in anoxic sediments.
Mixing zones	An explicitly defined area around an effluent discharge where effluent concentrations may exceed guideline values and therefore result in certain environmental values not being protected. The size of the mixing zone is site specific.
Moderate reliability guideline trigger values	Trigger values that have a moderate degree of confidence because they are derived from an adequate set of acute toxicity data (section 8.3.4) and hence require more extrapolation than high reliability trigger values, including an acute-to-chronic conversion
Monomeric	A chemical compound comprising single molecules
Morphometry	The form, shape and dimensions of an entity, e.g. waterbody or animal
Multiple lines of evidence	Weight of the evidence based on different types of information from a variety of different sources and studies
Munsell Scale	A means of expressing the colour of a soil by matching it against a colour chart
Necrotic	Localised dying tissue
Neurotoxin	Toxic substances which adversely affect the nervous system
NOEC (No observed effect concentration)	The highest concentration of a toxicant at which no statistically significant effect is observable, compared to the controls; the statistical significance is measured at the 95% confidence level
Not detectable	Below the limit of detection of a specified method of analysis
Nutrient solution	Plant growth medium providing all essential elements for plant growth in the absence of soil or other support media. Also referred to as solution culture.
Octanol:water partition coefficient (P_{OW})	The ratio of a chemical's solubilities in *n*-octanol and water at equilibrium. The logarithm of P_{OW} is used as an indication of a chemical's propensity for bioconcentration by aquatic organisms.
Off-flavour	Result of the accumulation of certain pollutants such as petroleum hydrocarbons in fish or shellfish to a level that affects the flavour, making the product undesirable for human consumption; also known as tainting

Oligotrophic	Waters with a small supply of nutrients
Organic carbon	Generally carbon which is chemically bonded to other carbon atoms, although methane (one carbon atom only) and its derivatives are considered organic
Organism	Any living animal or plant; anything capable of carrying on life processes
Osmoregulation	The biological process of maintaining the proper salt concentration in body tissues to support life
Osmosis	Diffusion of a solvent through a semi-permeable membrane into a more concentrated solution, tending to equalise the concentrations on both sides of the membrane
Oxidation	The combination of oxygen with a substance, or the removal of hydrogen from it or, more generally, any reaction in which an atom loses electrons
Oxygenation	The process of adding dissolved oxygen to a solution
Pacific oyster	*Crassostrea gigas*
PAH	Polycyclic aromatic hydrocarbons
Parameter	A measurable or quantifiable characteristic or feature
Partition coefficient	A ratio of the equilibrium concentration of the chemical between a non-polar and polar solvent
Pathogen	An organism capable of eliciting disease symptoms in another organism
Pelagic	Term applied to organisms of the plankton and nekton which inhabit the open water of a sea or lake
Percentile	Division of a frequency distribution into one hundredths
Performance indicators	These are the indicators used to assess the risk that a particular issue will occur (they are used in the guideline packages to compare against the trigger values). They are generally median (or mean) concentrations in the ambient water, and may be stressor and/or condition indicators.
Periphyton	The organisms attached to submerged plants
Pesticide	A substance or mixture of substances used to kill unwanted species of plants or animals
pH	Value that represents the acidity or alkalinity of an aqueous solution. It is defined as the negative logarithm of the hydrogen ion concentration of the solution.
pH ($CaCl_2$)	Measurement of soil pH in a 1:2.5 solution of soil:0.01M $CaCl_2$. The $CaCl_2$ solution is used because it has an ionic strength similar to that of soil water.
Phenols	Phenol is a benzene ring with one -OH radical replacing hydrogen. Phenols are compounds which contain additional chemical groups bound to this basic structure (each replacing hydrogen).
Photodegradation	Breakdown of a substance by exposure to light; the process whereby ultra-violet radiation in sunlight attacks a chemical bond or link in a chemical structure
Photolysis	The decomposition of a compound into simpler units as a result of the absorption of one or more quanta of radiation
Photosynthesis	The conversion of carbon dioxide to carbohydrates in the presence of chlorophyll using light energy
Physico-chemical	Refers to the physical (e.g. temperature, electrical conductivity) and chemical (e.g. concentrations of nitrate, mercury) characteristics of water
Physiology	The study of the functioning of organisms and their parts
Phytoplankton	Small (often microscopic) aquatic plants suspended in water

Phytotoxicity	Toxicity of contaminants to plants
Pilot program	A field investigation similar in design to a sampling program, but less ambitious in scope. It is used to assess preliminary indicator values, spatial and temporal variability and logistic issues before definitive sampling.
Plankton	Plants (phytoplankton) and animals (zooplankton), usually microscopic, floating in aquatic systems
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
Polychlorinated biphenyls	These are highly toxic and persistent compounds derived from the replacement by Cl radicals of numerous H radicals on biphenyl, which consists of two benzene rings joined by a covalent bond, with the elimination of two H radicals ($C_{12}H_{10}$).
Potable water	Water suitable, on the basis of both health and aesthetic considerations, for drinking or culinary purposes
Practical Quantitation Limit (PQL)	The Practical Quantitation Limit (PQL) is the lowest level achievable among laboratories within specified limits during routine laboratory operations. The PQL represents a practical and routinely achievable detection level with a relatively good certainty that any reported value is reliable (Clesceri et al. 1998). The PQL is often around 5 times the method detection limit.
Precipitation	(1) The formation of solid particles in a solution; generally, the settling out of small particles (2) The settling-out of water from cloud, in the form of rain, hail, snow, etc.
Primary production	The production of organic matter from inorganic materials
Producers	Organisms that are able to build up their body substance from inorganic materials
Prokaryotes	Organisms characterised by the absence of membrane-bound organelles (opposite to eukaryotes)
Prolarvae	Newly hatched larvae during the first few days when they feed on their supply of embryonic yolk
Protocol	A formally agreed method and procedure for measuring an indicator; it defines the sampling, sample handling procedures and sample analysis
Protozoans	Single-celled, animal-like organisms of the kingdom Protista
Quality assurance (QA)	The implementation of checks on the success of quality control (e.g. replicate samples, analysis of samples of known concentration)
Quality control (QC)	The implementation of procedures to maximise the integrity of monitoring data (e.g. cleaning procedures, contamination avoidance, sample preservation methods)
Rainbow trout	*Oncorhynchus mykiss*
Radiological	Pertaining to nuclear radiation
Rapid biological assessment	A form of biological assessment, best developed using stream macroinvertebrate communities, that uses standardised, cost-effective protocols to provide rapid sample processing, data analysis, reporting and management response. The results from such assessments are likely to be reliable to detect large impacts but not small or minor impacts.
Recruitment	In these Guidelines, the replenishment or addition of individuals of an animal or plant population through reproduction, dispersion and migration
Red claw	*Cherax quadricarinatus*

Redox potential	An expression of the oxidising or reducing power of a solution relative to a reference potential. This potential is dependent on the nature of the substances dissolved in the water, as well as on the proportion of their oxidised and reduced components.
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
Relaying	Transfer of shellfish from restricted areas and conditional approved areas (when closed due to harvesting criteria being exceeded) to approved or conditional approved areas for natural biological cleansing using the ambient environment as a treatment system
Risk	A statistical concept defined as the expected likelihood or probability of undesirable effects resulting from a specified exposure to known or potential environmental concentrations of a material. A material is considered safe if the risks associated with its exposure are judged to be acceptable. Estimates of risk may be expressed in absolute or relative terms. Absolute risk is the excess risk due to exposure. Relative risk is the ratio of the risk in the exposed population to the risk in the unexposed population.
Ryznar (Stability) index	Index relating the pH of water (pH) to the pH of water just saturated with calcium carborate (pH_s)
Salinity	The presence of soluble salts in or on soils or in water
Sediment	Unconsolidated mineral and organic particulate material that settles to the bottom of aquatic environment
Sediment pore waters	Water that occupies the space between particles in a sediment, as distinct from overlying water which is the water above the sediment layer
Sewage fungus	A thick filamentous growth that develops in water contaminated with sewage. The filamentous material is composed predominately of the bacterium *Sphaerotilus natans*.
Short-term trigger value (STV)	The maximum concentration of contaminant in irrigation water which can be tolerated for a shorter period of time (20 years) assuming the same maximum annual irrigation loading to soil as for the *long-term trigger value* (qv)
Silver perch	*Bidyanus bidyanus*
Simultaneously extracted metals	The sum of the molar concentrations of heavy metals (excluding iron and manganese) that are solubilised with cold dilute acid (usually measured simultaneously with the measurement of AVS).
Snapper	*Pagrus auratus*
Sodicity	The presence of a high proportion of sodium ions relative to other cations in a soil
Sodium adsorption ratio (SAR)	The concentration of sodium relative to calcium and magnesium in the soil solution
Solution concentration	Concentration of solutes in the soil water phase. The solutes, which may be contaminants, in the soil water are generally regarded as being highly available to organisms.
Sorption	Process whereby contaminants in soils adhere to the inorganic and organic soil particles
Speciation	The intimate chemical environment of the indicator (qv), that is the compound or ion of which it forms a part
Species	A group of organisms that resemble each other to a greater degree than members of other groups and that form a reproductively isolated group that will not produce viable offspring if bred with members of another group

Species richness	The number of species present (generally applied to a sample or community)
SPM — suspended particulate matter	This is insoluble material which resides in the water column, or is dispersed in a sample upon agitation
Stakeholder	A person or group (e.g. an industry, a government jurisdiction, a community group, the public, etc.) who have an interest or concern in something
Standard (water quality)	An objective that is recognised in enforceable environmental control laws of a level of government
Standing crop	The weight of organic material that can be sampled or harvested by normal methods at any one time from a given area
Static system	An exposure system of aquatic toxicity tests in which the test chambers contain solutions of the test material or control water that are not usually changed during the test. Depending upon conditions, a static system may or may not be in equilibrium.
Steady state or dynamic equilibrium	The state at which the competing rates of uptake and elimination of a chemical within an organism or tissue are equal. An apparent steady state is reached when the concentration of a chemical in tissue remains essentially constant during a continuous exposure.
Stressors	The physical, chemical or biological factors that can cause an adverse effect in an aquatic ecosystem as measured by the condition indicators (see Section 3.3.2)
Sub-lethal	Involving a stimulus below the level that causes death
Supersaturation	Refers to a solution containing more solute than equilibrium conditions will allow
Survival time	The time interval between death and the initial exposure of an aquatic organism to a harmful substance
Suspension	A system in which very small particles (solid, semi-solid, or liquid) are more or less uniformly dispersed in a liquid or gaseous medium. If the particles are small enough to pass through filter membranes, the system is termed a colloidal suspension. If the particles are of larger than colloidal dimensions they will tend to precipitate, if heavier than the suspending medium, or to agglomerate and rise to the surface, if lighter.
Sydney rock oyster	*Saccostrea commercialis*
Synergism	A phenomenon in which the effect or toxicity of a mixture of chemicals is greater than that to be expected from a simple summation of the effects or toxicities of the individual chemicals present in the mixture
Tainting	See 'Off-flavour'
Taxa richness	Number of taxa present
Taxon (Taxa)	Any group of organisms considered to be sufficiently distinct from other such groups to be treated as a separate unit (e.g. species, genera, families)
Taxonomic (group, resolution)	An organism's location in the biological classification system used to identify and group organisms with similar physical, chemical and/or structural composition.
Teratogen	An agent that increases the incidence of congenital malformations
Thermodynamic equilibrium	Property of a system which is in mechanical, chemical and thermal equilibrium
Thermotolerant coliform	Also known as faecal coliforms. In tropical and sub-tropical areas, thermotolerant coliforms may on some occasions include microorganisms of environmental rather than faecal origin.

Threshold concentration	A concentration above which some effect (or response) will be produced and below which it will not
Tolerance	The ability of an organism to withstand adverse or other environmental conditions for an indefinitely long exposure without dying
Total dissolved solids (TDS)	A measure of the inorganic salts (and organic compounds) dissolved in water
Total metal	The concentration of a metal in an unfiltered sample that is digested in strong nitric acid
Toxicant	A chemical capable of producing an adverse response (effect) in a biological system at concentrations that might be encountered in the environment, seriously injuring structure or function or producing death. Examples include pesticides, heavy metals and biotoxins (i.e. domoic acid, ciguatoxin and saxitoxins).
Toxicity	The inherent potential or capacity of a material to cause adverse effects in a living organism
Toxicity identification and evaluation (TIE)	Toxicity characterisation procedures involving use of selective chemical manipulations or separations and analyses coupled with toxicity testing to identify specific classes of chemicals and ultimately individual chemicals that are responsible for the toxicity observed in a particular sample
Toxicity test	The means by which the toxicity of a chemical or other test material is determined. A toxicity test is used to measure the degree of response produced by exposure to a specific level of stimulus (or concentration of chemical).
Trigger values	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.
Trochus	*Trochus niloticus*
True colour	The colour of water resulting from substances that are totally in solution; not to be mistaken for apparent colour resulting from colloidal or suspended matter
Turbulence	Unorganised movement in liquids and gases resulting from eddy formation
Type I error	Probability of concluding that an impact has occurred when, in fact, an impact has not occurred
Type II error	Probability of concluding that an impact has not occurred when, in fact, an impact has occurred
Univariate	Statistical analysis concerned with data collected on one dimension of the same organism
Uptake	A process by which materials are absorbed and incorporated into a living organism
Value judgements	A decision involving basic issues of fairness, reasonableness, justice, or morality
Volatile	Having a low boiling or subliming pressure (a high vapour pressure)
Water quality criteria	Scientific data evaluated to derive the recommended quality of water for various uses
Water quality guideline	See 'Guideline (water quality)'
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.

Watertable	The level of groundwater; the upper surface of the zone of saturation for underground water.
Whiting	*Sillago* spp. of marine fish
Whole effluent toxicity testing	The use of toxicity tests to determine the acute and/or chronic toxicity of effluents
Xenobiotic	A foreign chemical or material not produced in nature and not normally considered a constituent of a specified biological system. This term is usually applied to manufactured chemicals.
Yabby	*Cherax destructor*
Zooplankton	The animal portion of the plankton

Appendix 2 The National Water Quality Management Strategy

The Australian and New Zealand Environment and Conservation Council (ANZECC) and the Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ) are working together to develop a National Water Quality Management Strategy (NWQMS).

The guiding principles for the Strategy are set out in *National Water Quality Management Strategy: Policies and Principles — A Reference Document* (NWQMS Paper 2, ANZECC & ARMCANZ, 1994) which emphasises the importance of:

- ecologically sustainable development
- integrated (or total) catchment management
- best management practices, including the use of acceptable modern technology, and waste minimisation and utilisation
- the role of economic measures, including user pays and polluter pays.

The process of implementing the National Water Quality Management Strategy involves the community working in concert with government in setting and achieving local environmental values, which are designed to maintain good water quality and to progressively improve poor water quality. It involves development of a plan for each catchment and aquifer, which takes account of all existing and proposed activities and developments, and which contains the agreed environmental values and feasible management options.

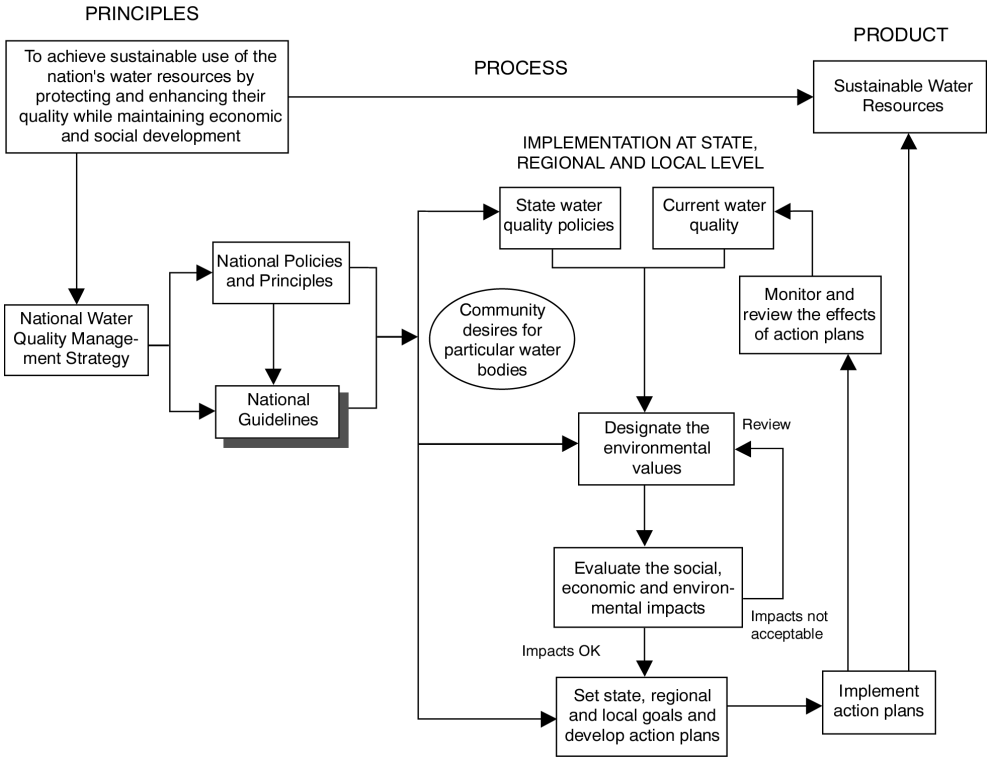

Figure A1 National Water Quality Management Strategy

Documents of the National Water Quality Management Strategy

Paper No.	Title
Policies and Process for Water Quality Management	
1	*Water Quality Management — An Outline of the Policies*
2	*Policies and Principles — A Reference Document*
3	*Implementation Guidelines*
Water Quality Benchmarks	
4	*Australian and New Zealand Water Quality Guidelines for Fresh and Marine Waters*
5	*Australian Drinking Water Guidelines — Summary*
6	*Australian Drinking Water Guidelines*
7	*Australian Guidelines for Water Quality Monitoring and Reporting*
Groundwater Management	
8	*Guidelines for Groundwater Protection*
Guidelines for Diffuse and Point Sources*	
9	*Rural Land Uses and Water Quality*
10	*Guidelines for Urban Stormwater Management*
11	*Guidelines for Sewerage Systems — Effluent Management*
12	*Guidelines for Sewerage Systems — Acceptance of Trade Waste (Industrial Waste)*
13	*Guidelines for Sewerage Systems — Sludge (Biosolids) Management*
14	*Guidelines for Sewerage Systems — Use of Reclaimed Water*
15	*Guidelines for Sewerage Systems — Sewerage System Overflows*
16a	*Effluent Management Guidelines for Dairy Sheds*
16b	*Effluent Management Guidelines for Dairy Processing Plants*
17	*Effluent Management Guidelines for Intensive Piggeries*
18	*Effluent Management Guidelines for Aqueous Wool Scouring and Carbonising*
19	*Effluent Management Guidelines for Tanning and Related Industries in Australia*
20	*Effluent Management Guidelines for Australian Wineries and Distilleries*

* The guidelines for diffuse and point sources are national guidelines which aim to ensure high levels of environmental protection that are broadly consistent across Australia.

Appendix 3 Recent water quality documents of the NZ Ministry for the Environment

- *Flow Guidelines for Instream Values* (NZ Ministry for the Environment 1995)
- *New Zealand Drinking Water Guidelines* (NZ Ministry of Health 1995).
- *Water Quality Guidelines No. 1: Biological Growths* (NZ Ministry for the Environment 1992)
- *Water Quality Guidelines No. 2: Colour and Clarity* (NZ Ministry for the Environment 1994)
- *Periphyton Guidelines* (NZ Ministry for the Environment, in press)
- *Recreational Water Quality Guidelines* (NZ Ministry for the Environment 1999)
- *Monitoring the Trophic Status of New Zealand's Lakes* (NZ Ministry for the Environment, in press)
- *Managing Waterways on Farms* (NZ Ministry for the Environment, in press)
- *A discussion on reasonable mixing in water quality management, Resource Management Ideas No. 10* (NZ Ministry for the Environment 1994)
- *Reducing the Impacts of Agricultural Runoff on Water Quality: A discussion of policy approaches* (NZ Ministry for the Environment 1997)

Appendix 4 Development of the revised guidelines

This Appendix outlines the revision process for the Guidelines, and the various instrumentalities involved.

The revision program

In March 1993 the ANZECC Standing Committee on Environmental Protection (SCEP) made a decision to review the *Australian Water Quality Guidelines for Fresh and Marine Waters*, periodically, with the following two primary objectives:

- to incorporate current scientific, international and national information which is appropriate to Australian and New Zealand conditions; and
- to produce a document that is sufficiently clear and understandable for the relevant authorities to use for consultation.

The Environmental Research Institute of the Supervising Scientist (***eriss***) was given the responsibility for managing the review of the Guidelines on behalf of ANZECC.

The review strategy was endorsed by the ANZECC SCEP at its meeting on 15 May 1996 which involved key government and non-government groups in recognition of the need for an open and transparent process with broad community consultation. As part of the strategy, a project committee was established and given very broad representation to oversee and facilitate the process. Membership included state environment agencies, a number of agencies and representatives of the Agricultural and Resource Management Council of Australia and New Zealand (ARMCANZ), and representatives of the National Health and Medical Research Council (NHMRC), the National Farmers Federation, the Australian Seafood Industry Council, and peak conservation organisations (full membership listed below).

The outcomes from a preliminary workshop, together with the issues raised in early public submissions on the Guidelines, were used to assist in identifying and defining the scope of the tasks necessary for reviewing the 1992 report. Expert groups from Australia and New Zealand were then commissioned to review the technical aspects of the report and a draft of the revised guidelines was compiled by ***eriss***.

After consideration by the project committee, the draft document was referred firstly to the ARMCANZ/ANZECC Contact Group (listed below), and then to the ARMCANZ subcommittee — the Sustainable Land & Water Resource Management Committee (SLWRMC) — for feedback and subsequent endorsement prior to its release. The Guidelines were released for a three-month public comment period in July 1999 by the Australian Commonwealth Minister for the Environment & Heritage, without the endorsement of SCEP and the ARMCANZ Standing Committee for Agricultural and Resource Management (SCARM).

eriss received and collated 96 public submissions. On the basis of these comments, the Guidelines were redrafted in close consultation with the Contact Group and its working parties (listed below). The revised Guidelines were endorsed by the Contact Group in May 2000 and by SCEP in June 2000, with the final document then referred to ANZECC who approved the Guidelines for publication under the National Water Quality Management Strategy in July 2000.

The Project Committee

Name	Organisation	Name	Organisation
Graeme Batley	ARMCANZ — CSIRO Energy Technology	Kevin McAlpine (Secretary)	Com. Environmental Research Institute of the Supervising Scientist
John Chapman	NSW Environmental Protection Authority — Centre for Ecotoxicology	Bill Maher	University of Canberra
John Cugley	SA Department of Environment and Natural Resources	Scott Markich	Australian Nuclear Science and Technology Organisation
Lisa Dixon	Victorian Environmental Protection Authority	Greg Miller	peak conservation organisations
Barry Hart	Monash University — Water Studies Centre	Andrew Moss	Queensland Department of Environment
Chris Humphrey	Com. Environmental Research Institute of the Supervising Scientist	Barry Noller	Northern Territory Department of Mines and Energy
Heather Hunter	Queensland Department of Natural Resources	Eric Pyle	New Zealand Ministry for the Environment
Arthur Johnston (Chairman)	Com. Environmental Research Institute of the Supervising Scientist	Nigel Scullion	Australian Seafood Industry Council
Warren Jones	Tasmanian Department of Environment and Land Management	Graham Skyring	ARMCANZ — Skyring Environment Enterprises
David Klessa	Com. Environmental Research Institute of the Supervising Scientist	Victor Talbot	WA Department of Environmental Protection
Mike Lawton	Northern Territory Department of Land Planning and Environment	Alan Thomas	Com. Environment Protection Group
Chris leGras	Com. Environmental Research Institute of the Supervising Scientist	Pam Waudby	National Farmers Federation
Richard Lugg	National Health and Medical Research Council	Rosalyn Vulcano	Northern Territory Power and Water Authority

Contributing members of the Contact Group and/or proxies (period 1996–2000)

Name	Organisation	Name	Organisation
Paul Bainton & Alan Thomas	Com. Environment Australia, Environment Protection Group	Rachel Gregson, Ross Dalton, Dennis Alyliffe, David Lambert, Stephen Clark	Agriculture, Fisheries and Forestry — Australia
Barbara Richardson & Carolyn Davies	NSW Environment Protection Authority	Bruce Cooper	New South Wales Department of Land and Water Conservation
Chris Bell	Victorian Environment Protection Authority	Anne Woolley & Peter Thompson	Queensland Department of Natural Resources
John Cugley	SA Environment Protection Authority	Peter Scott	Melbourne Water Corporation
Stephen Fisher (EPA) & Ian Eskdale (DE)	Queensland Environment Protection Agency & Queensland Department of Environment	Alan Maus & Barry Sanders	Water Corporation of Western Australia
Victor Talbot	Western Australian Department of Environmental Protection	Michael Lawton	Northern Territory Department of Lands, Planning & Environment
Greg Dowson & Warren Jones	Tasmanian Department of Environment and Land Management	Robert Neil	Environment ACT
Bob Zuur & Eric Pyle	New Zealand Ministry of the Environment	Philip Callan	National Health and Medical Research Council

Working Parties to the Contact Group

1. Aquaculture: David Cunliffe, Pauline Semple, Victor Talbot, Rob Cordover, Christine Cowie, Kerry Jackson and Michelle Burford
2. Agriculture: Liz Rogers, Greg Dowson, Karen Benn and Karina Watkins
3. Physico-chemical stressors: Klaus Koop, Greg Dowson, John Cugley, Peter Scott, Andrew Moss and Bob Humphries
4. Toxicants and sediments: Peter Thompson, Bob Humphries, John Cugley, Bruce Cooper, Munro Mortimer, Victor Talbot, Karina Watkins and Gus Fabris

Appendix 5 Basis of the proposed guidelines for recreational water quality and aesthetics in Australia

The draft World Health Organization (WHO) *Guidelines for Safe Recreational-water Environments: Coastal and Fresh-waters* (WHO 1998), in referring to the different types of recreational usage of water, give the following examples:

- no contact, where enjoyment is of aesthetic beauty of the water environment;
- limited contact, e.g. boating, rowing, fishing;
- meaningful direct contact that involves a negligible risk of swallowing water, e.g. wading;
- extensive direct contact with full body immersion and a meaningful risk of swallowing water, e.g. swimming.

The WHO *Health-based Monitoring of Recreational Waters: The Feasibility of a New Approach (The 'Annapolis' Protocol)* (WHO 1999) considers the adequacy and effectiveness of present approaches to the monitoring and assessment of recreational water, particularly where the monitoring is linked to the effective management of microbiological hazards in coastal and freshwater areas.

A number of types of hazards that can be encountered in recreational water are dealt with in the WHO Guidelines; they include:

- poisoning and toxicoses, including stings of poisonous and venomous animals, ingestion or inhalation of, or contact with, chemically contaminated water or blooms of toxic cyanobacteria or dinoflagellates;
- physiological effects, including chilling, thermal shock;
- exposure to pathogenic bacteria, viruses, fungi or parasites;
- aesthetic quality including visual clarity, colour, odour, surface scum.

The WHO Guidelines also include guidance on assessment and control measures, public health advice and intervention requirements when guideline values are exceeded.

References

WHO 1998. Guidelines for safe recreational-water environments: Coastal and fresh-waters. Draft for Consultation, EOS/DRAFT/98.14, World Health Organization, Geneva.

WHO 1999. Health-based monitoring of recreational waters: The feasibility of a new approach (The 'Annapolis Protocol'), WHO/SDEW/WSH/99.1, World Health Organization, Geneva.

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Note: Page entries beginning with 'A' refer to appendices