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).

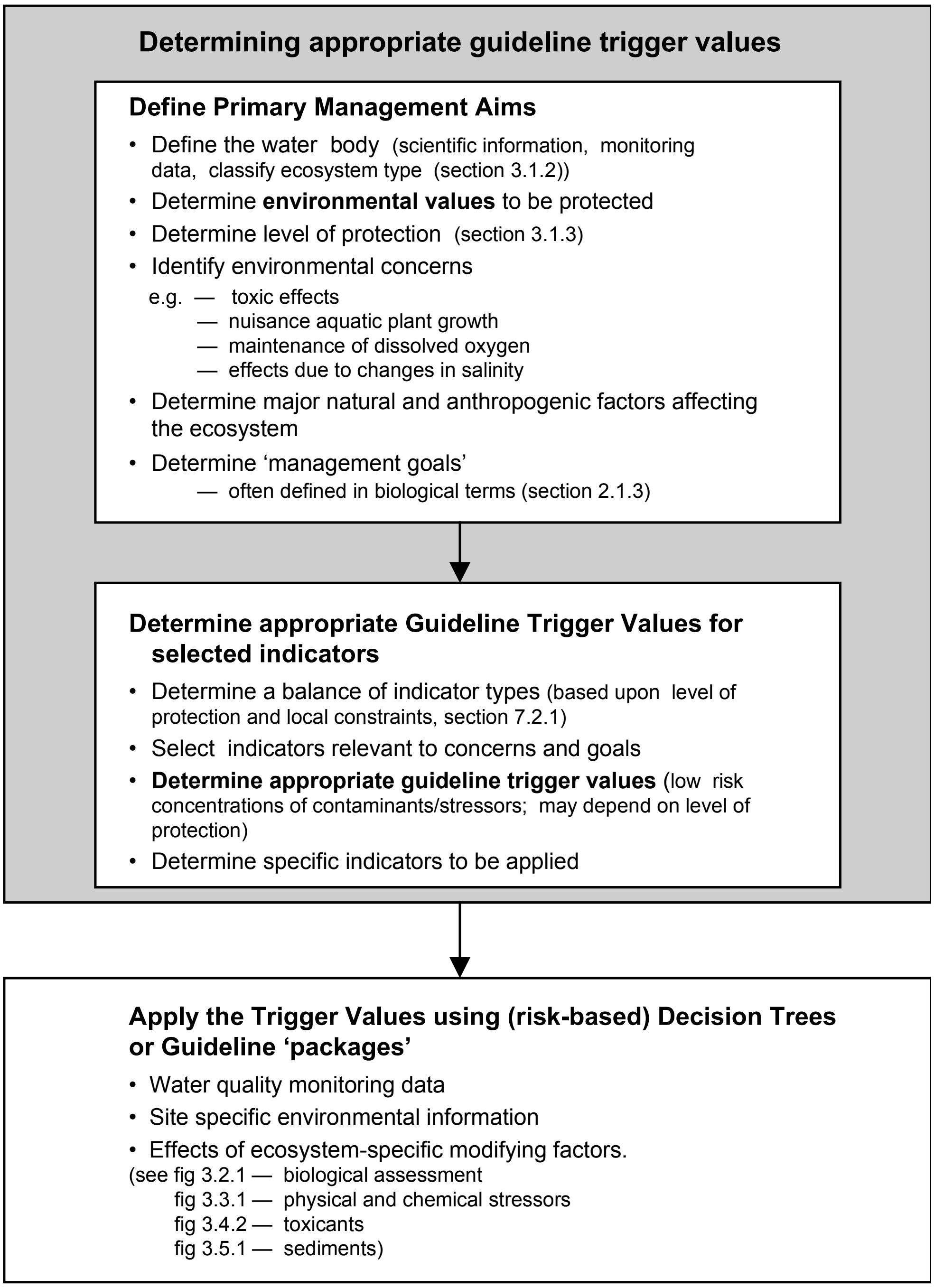

Figure 3.1.1 Flow chart of the steps involved in applying the guidelines for protection of aquatic ecosystems

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.

a See Section 2.1.3

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

b Section 3.1.3

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]

c Section 3.4 and 3.5

d Section 3.3

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.

e Section 2.1.4 and 3.2

f Section 3.3.2

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]

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:

g Section 7.2.1

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

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[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.

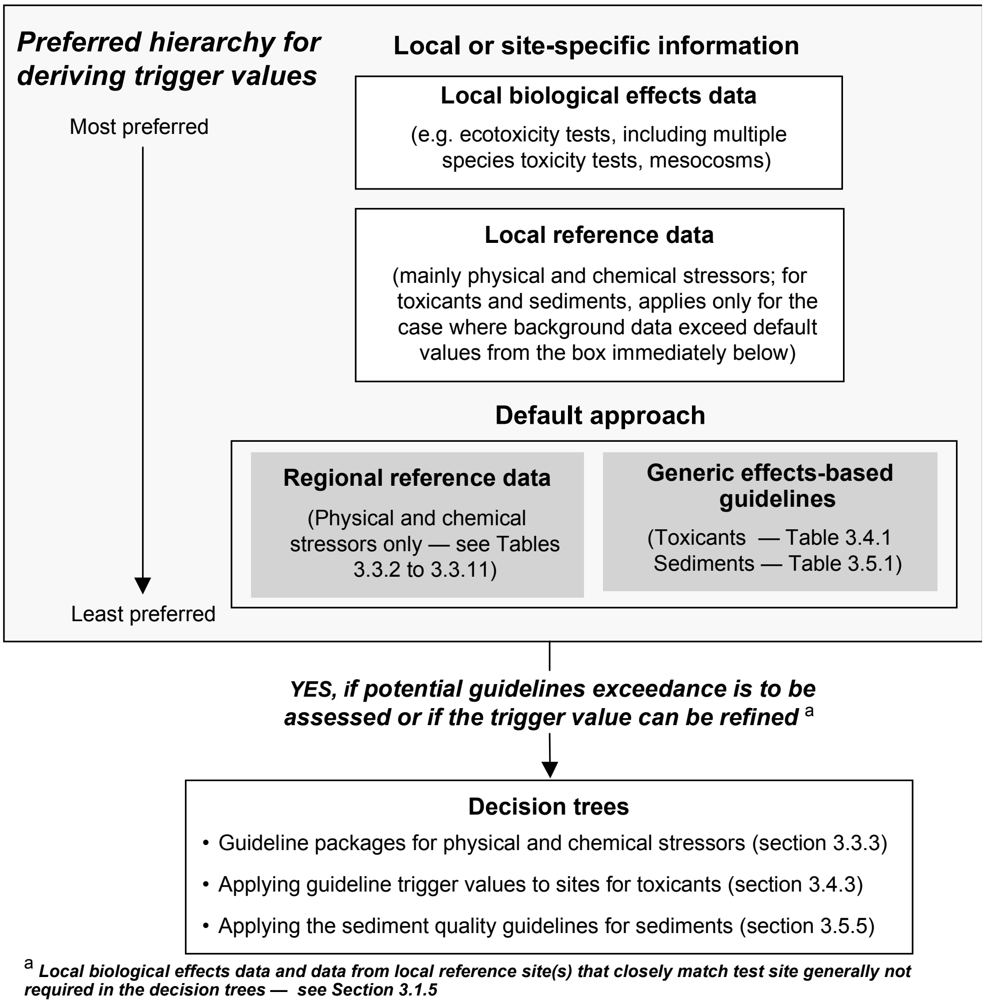

Figure 3.1.2 Procedures for deriving and refining trigger values, and assessing test sites, for physical and chemical stressors and toxicants in water and sediment. Dark grey shading indicates most likely point of entry for users requiring trigger values.

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).

a See Appendix 2 (Vol. 2)

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.

3.1.2.2 Classifying the ecosystem

b See outline in Section 3.1.5

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.

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.

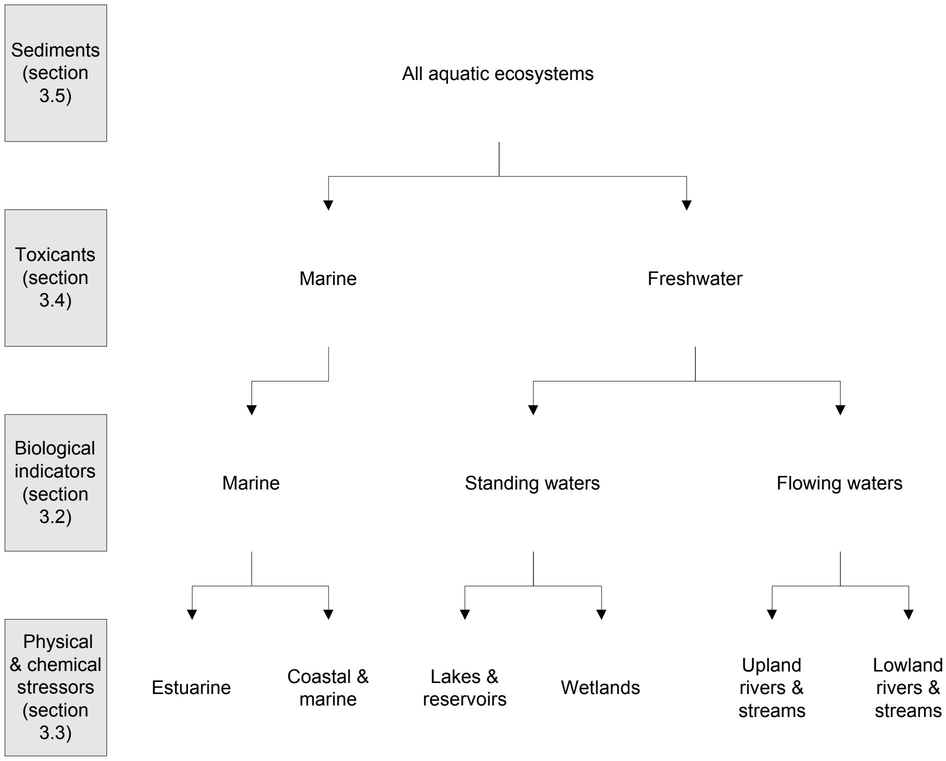

Figure 3.1.3 Classification of ecosystem type for each of the broad categories of indicators (in grey boxes at left of the diagram)

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

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[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’.

a See Section 3.2.4

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.

b Sections 3.1.8 & 3.2 to 3.5

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]

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

a See Section 3.1.1.2

b See also Sections 3.4.3.2, 7.4.4.2, 7.4.4.4

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).

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:

a See Sections 3.4.3, 3.5.5

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]
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

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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.

a See Section 2.1.3

b Section 7.4.4.3 for related discussion

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]

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'.

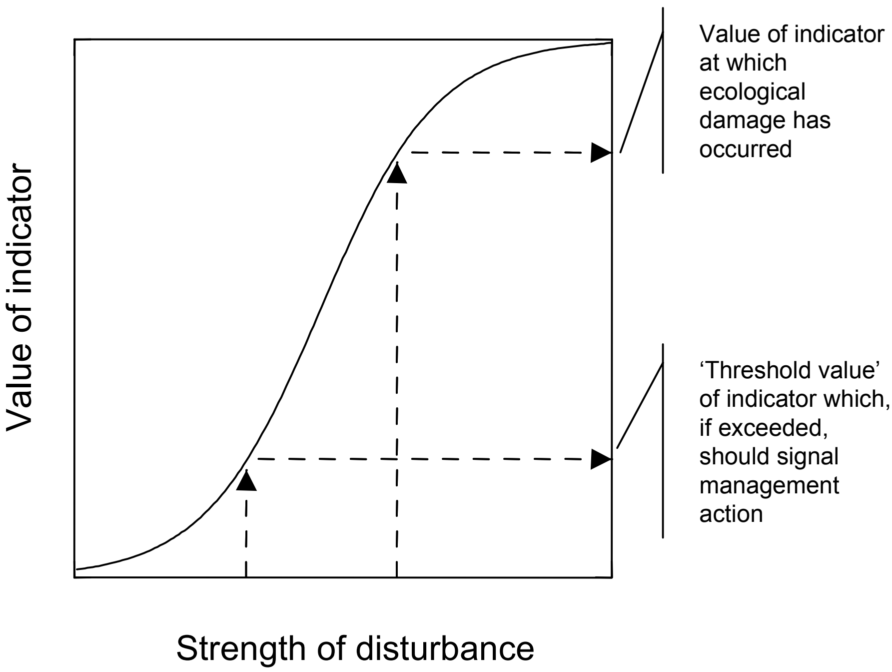


Figure 3.1.4 Graphical depiction of the relationship between indicator response and strength of disturbance, and threshold for management intervention

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.

a See Section 7.4.4

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.

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.

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

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]

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]
2. together with a balance of indicators, identify the biological assessment objectives for protection of the water resource;[c]
3. select appropriate indicators and protocols to apply to the assessment objectives;[d]
4. select the appropriate experimental design to apply to the indicator;[e]

b Section 3.1.1.1

c Sections 7.2.1 & 3.2.1.3

d Sections 3.2.2 and 3.2.3

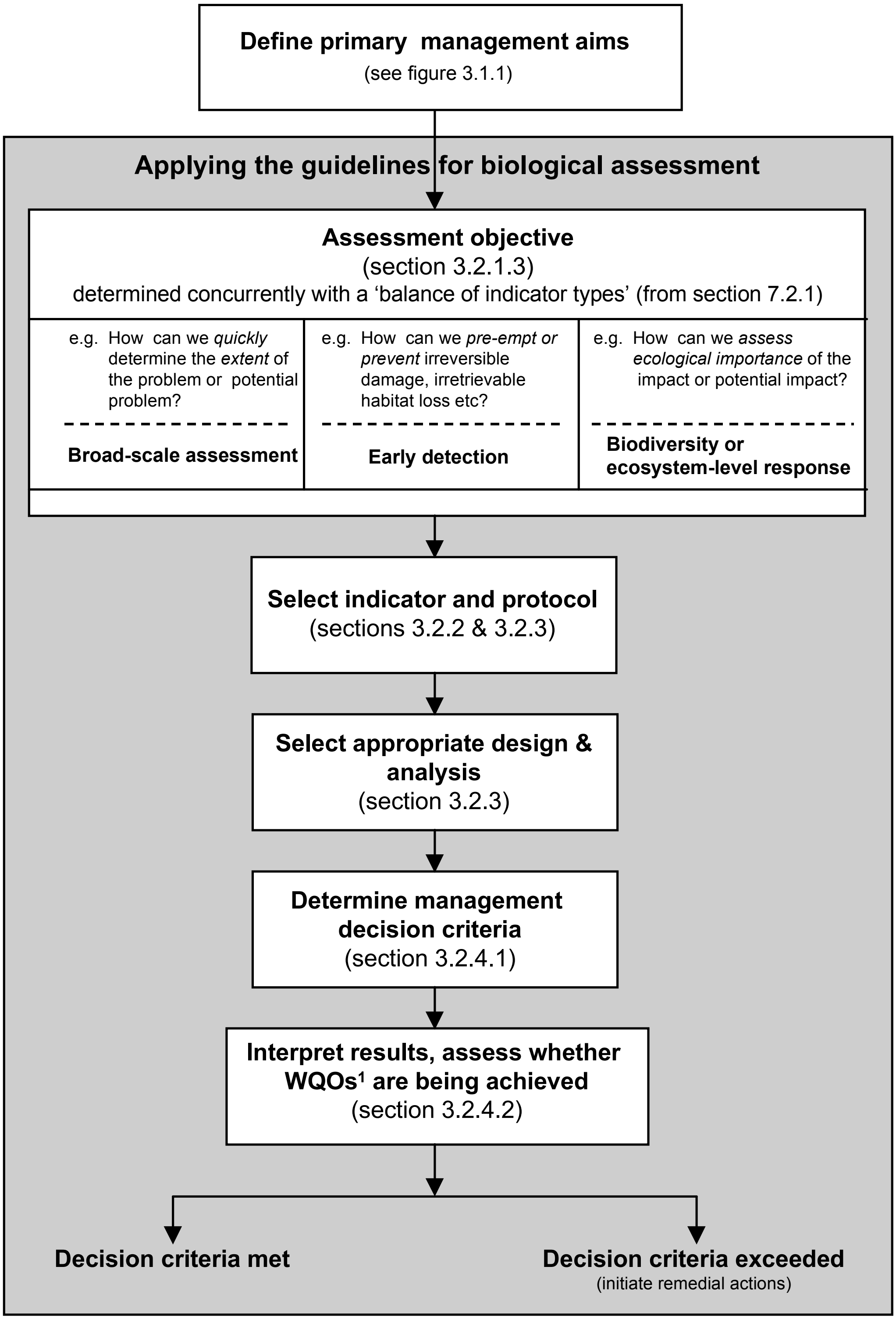
e Section 3.2.3 and Ch.7

a See Sections 3.1.7, 7.2.3.3 and 3.2.4

b Section 3.2.4.2

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

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

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.

3.2.3 Recommended experimental design and analysis procedures for generic protocols

b See Sections 7.2.2 and 7.2.3

c Sections 7.2.2 and 7.2.3

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).

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 Modifications (e.g. MBACIP, inclusion of covariates)	All protocols wherever possible Any protocol if applicable
	'Beyond BACI' designs	Any protocol if applicable.
	BACIP (single control site) Modifications to BACIP	1A, 1B 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 *a See App. 4, Vol. 2*

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.

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

b Section 3.2.4.1

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.

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)

a See Section 3.2.4.2/1

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.

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.

b Section 7.2.1.1/3

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.

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

c Section 3.2.4.1

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.

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.

d Section 3.2.5

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]

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]
- 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]
- 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]

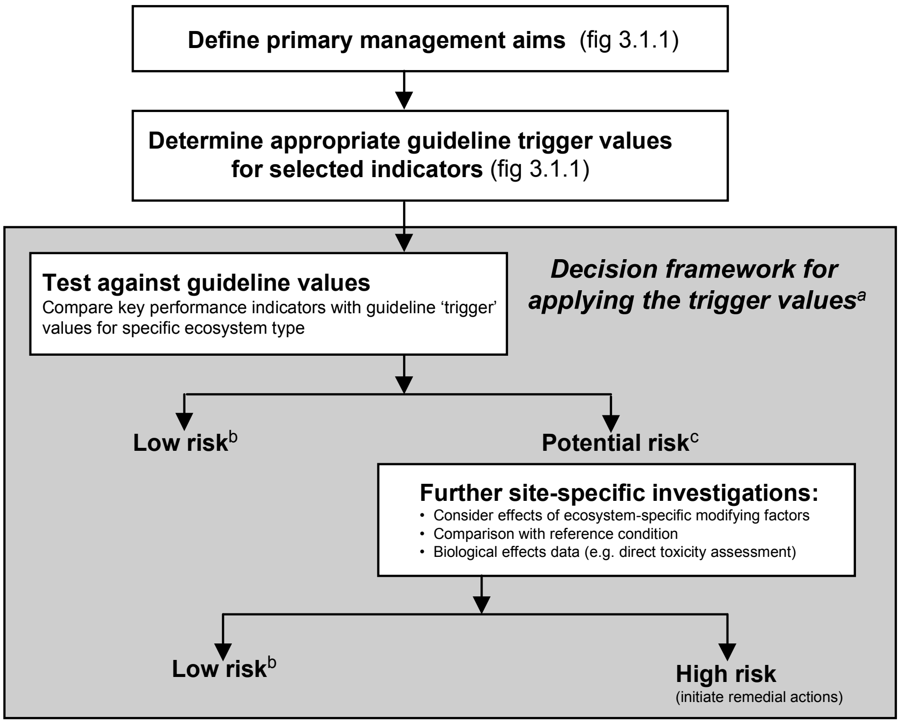
c Section 3.1.2

d Section 3.3.2.2

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.

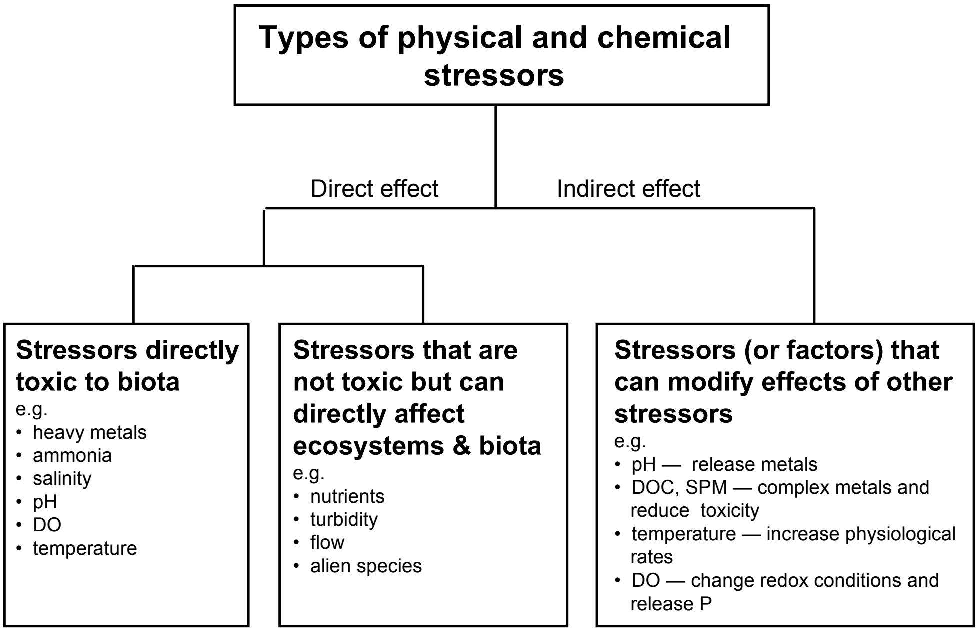


Figure 3.3.2 Types of physical and chemical stressors

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

b Section 8.5.2 (Volume 2)

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]

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 *a See box 3.3.1*

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.

Ecosystem condition

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

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]

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 80th and 20th 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* ($\mu g\ L^{-1}$)	TP ($\mu g\ P\ L^{-1}$)	FRP ($\mu g\ P\ L^{-1}$)	TN ($\mu g\ N\ L^{-1}$)	NO_x ($\mu g\ N\ L^{-1}$)	NH_4^+ ($\mu g\ N\ L^{-1}$)	DO (% saturation)[l]		pH	
							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* ($\mu g\ L^{-1}$)	TP ($\mu g\ P\ L^{-1}$)	FRP ($\mu g\ P\ L^{-1}$)	TN ($\mu g\ N\ L^{-1}$)	NO_x ($\mu g\ N\ L^{-1}$)	NH_4^+ ($\mu g\ N\ L^{-1}$)	DO (% saturation)[f]		pH	
								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* (μg L^{-1})	TP (μg P L^{-1})	FRP (μg P L^{-1})	TN (μg N L^{-1})	NO_x (μg N L^{-1})	NH_4^+ (μg N L^{-1})	DO (% saturation)[i]		pH	
								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

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[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.

a See Section 3.3.2.4

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).

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.

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]
- 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.

a Section 3.1.5

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

a See Section 3.3.2.3

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

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.

b See recommendations in Section 8.5.2, Volume 2

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

[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

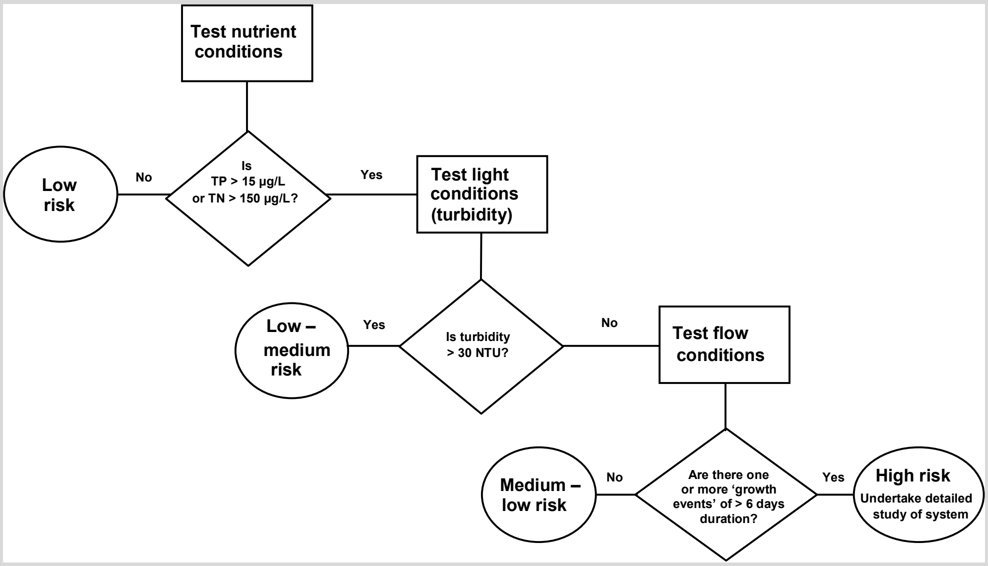
b See also Case Study 2 below

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).

Select key biological indicator and management targets

- **Chlorophyll *a* conc** — **<10μgL^{-1} for 9 in 10 years**
- **Dissolved oxygen concentration** — **>60% saturation for 9 in 10 years**

↓

Identify key stressor and key performance indicator

- **Key stressor** — **Organic matter (BOD)**
- **Key performance indicator** — **BOD loading (kg.m^{-2}.yr^{-1})**

↓

Determine trigger value for key stressor

- **Develop models relating BOD loading to water column DO concentration and sediment nutrient release for range of waterbodies and sediment types**
- **Validate model relationships using local reference and impacted sites for which data are available**
- **Use models to determine trigger values (sustainable loads) for key waterbodies throughout the catchment**

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.

a See Section 3.4.2.3

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.

3.4.2.2 Extrapolating from laboratory data to protect aquatic ecosystems

b Described in Section 8.3.3.3 in Vol. 2

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.

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.

a See Section 8.3.7

b See toxicant databases on the CD-Rom

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.

3.4.3 Applying guideline trigger values to sites

c See last paragraph of Section 2.1.4

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]

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.

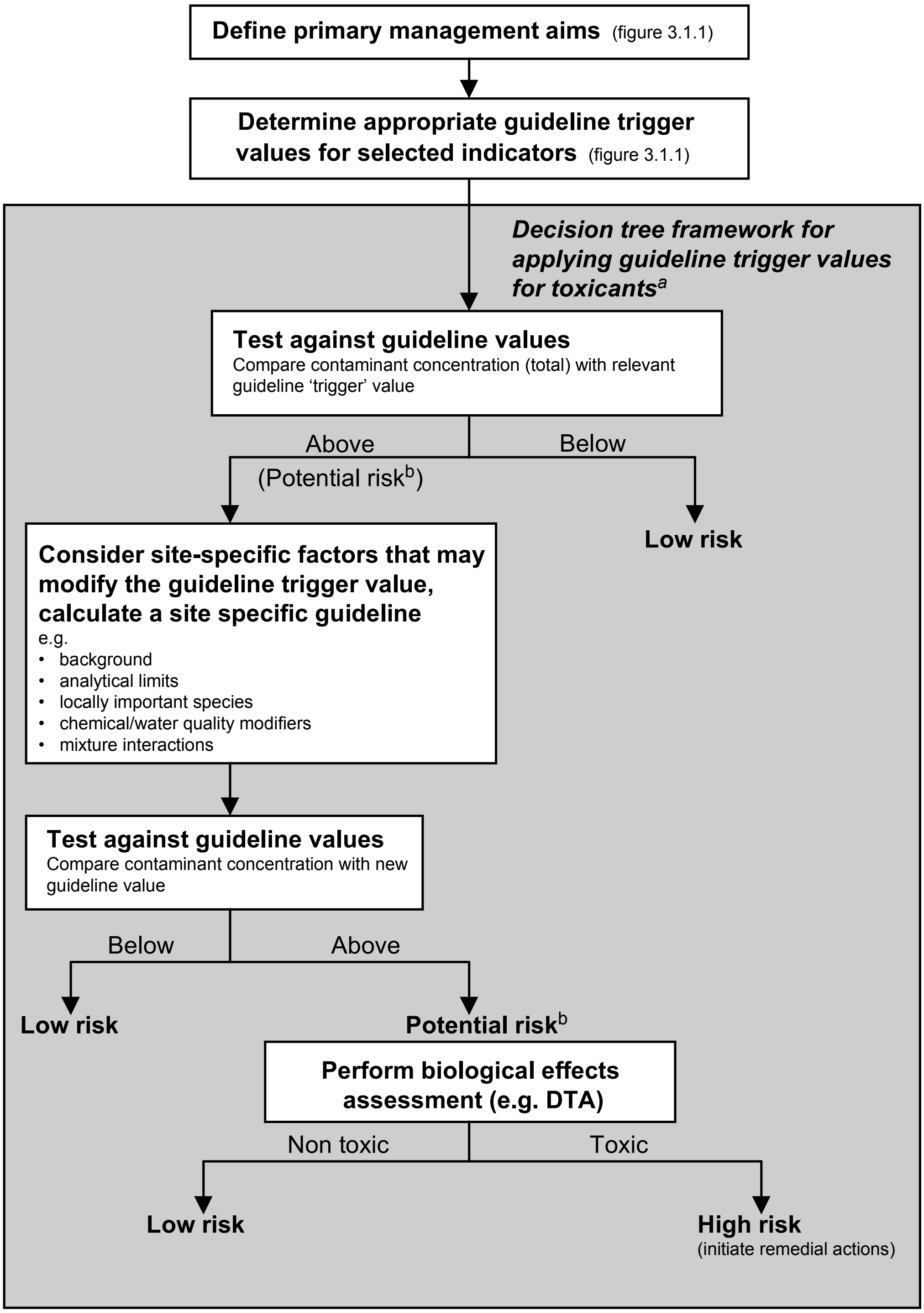
c Section 3.1.3

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

a See Section 8.3.5.2
b Section 3.4.2.4

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

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

a See Section 8.3.5.7

b Section 8.3.3.4

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.

c Section 8.3.3.4

d Section 8.3.3.4

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]

e Section 8.3.5.8

f Section 8.3.4.2

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

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.

a See Section 8.3.5.15

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.

   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).

b Section 8.3.5.16

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.

    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.

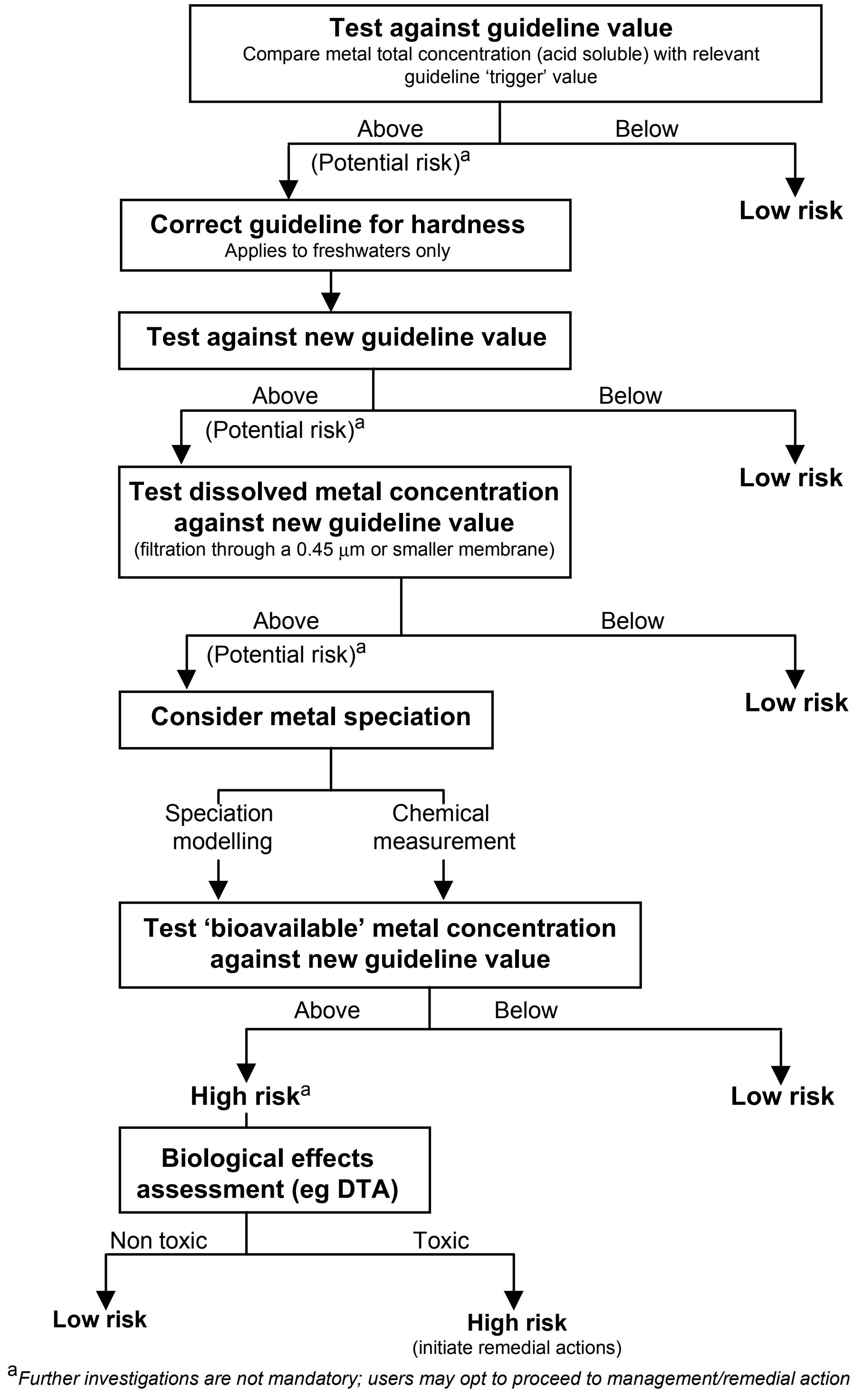


Figure 3.4.2 Decision tree for metal speciation guidelines

a See Section 8.3.5.18

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.

b Section 8.3.5.19

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).

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.

c Section 8.3.4.2

d Section 8.3.4.4

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]

e Section 8.3.6

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]

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

f Section 3.2

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]

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.

a See Section 8.4.3.2

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]

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

a See Section 8.4.3

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.

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.

b Section 8.4

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]

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 See App. 8, Volume 2

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

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.

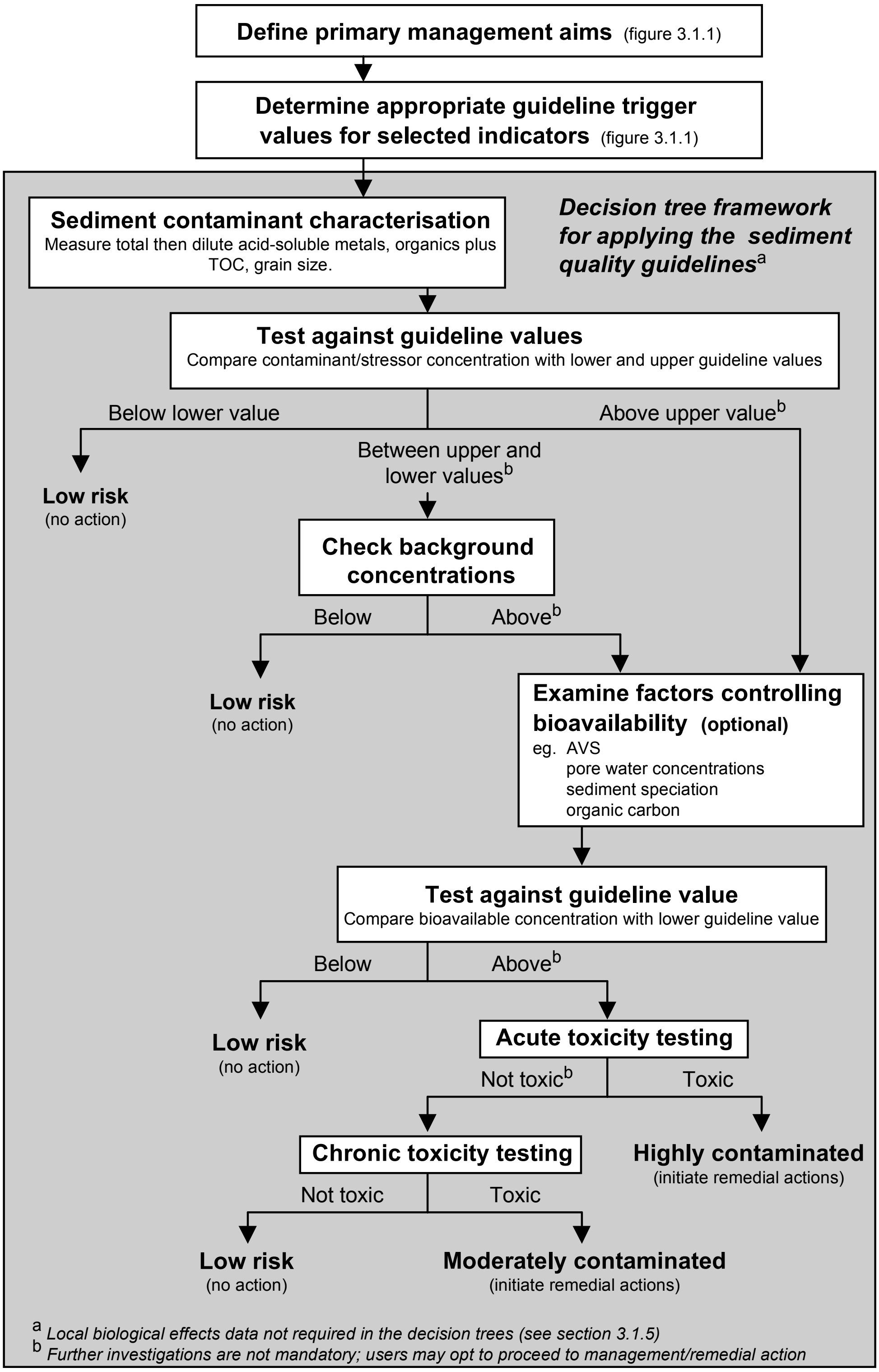


Figure 3.5.1 Decision tree for the assessment of contaminated sediments

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.