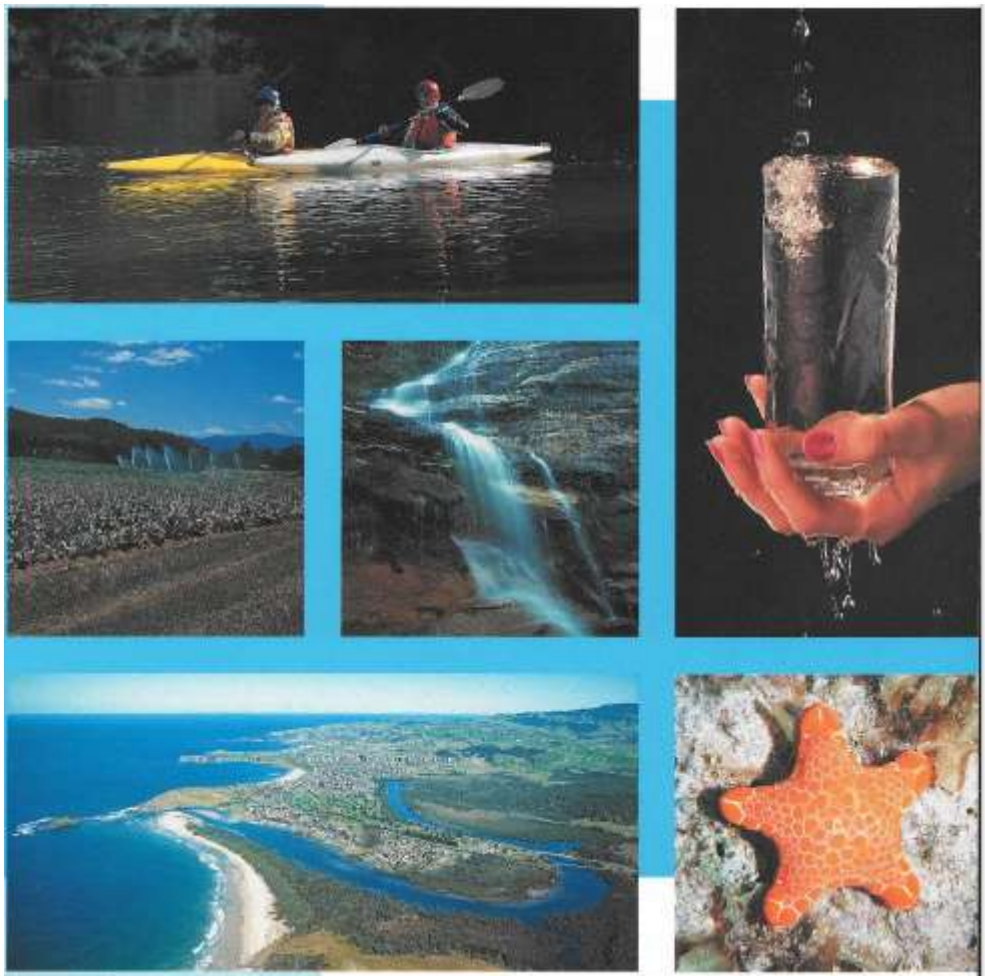
Australian Water Quality Guidelines for Fresh and Marine Waters

National Water Quality Management Strategy

November 1992



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Preface

The management of water resources is an integral part of environmental management and an essential requirement for supporting the economic, social and environmental objectives of our society. Over the years of European settlement of Australia, water usage has accelerated t the point where most economically viable sources of water have been harnessed, and water resource managers are faced with the challenge of maintaining and enhancing the quality of a heavily used resource.

The Australian and New Zealand Environment and Conservation Council and the Australian Water Resources Council have recognised the deterioration in the quantity and quality of Australian waters and are co-operatively developing the *National Water Quality Management Strategy*. The aim of this strategy is to pursue the sustainable use of the nation's water resources by protecting and enhancing their quality wile maintaining economic and social development.

The *Australian Water Quality Guidelines for Fresh and Marine Waters* is a component of that strategy. It collates a vast body of scientific information and management experience on the water quality required to sustain the range of environmental values that Australian waters may support. It represents an important reference tool for the development of catchment management plans and policies, allowing governments and the community to make informed decisions about water quality requirements and the consequences of management decisions.

Such a document cannot hope to apply to the whole range of water environments across Australia without modification to address local conditions. This becomes obvious when considering the differences between, for example, tropical marine waters and central Australian intermittent drainage systems. In particular, there is a need for local assessment of nutrient levels to minimise the incidence and severity of algal blooms, and of such variables as water clarity, temperature and oxygen levels.

Not all these guideline values are being achieved across Australia at present. In some cases this reflects natural differences in water quality, but in others it represents a degradation of waters through a lack of awareness of environmental implications associated with historical water and land use management practices. The guideline values here may be applied in a wide range of circumstances but should be treated, in many cases, as long-term goals to be achieved by a concerted management program rather than as immediately achievable outcomes. In fact, maintaining the full range of environmental values may no longer be attainable in some areas without disproportionate expense.

The choice of specific management goals for waters should reflect an informed community selection of options with a full awareness of environmental, social and financial costs. This is central to the approach adopted in the National Water Management Strategy by the Australian and New Zealand Environment and Conservation Council and the Australian Water Resources Council Ministerial Councils.

Acknowledgements

The objective of the Australian and New Zealand Environment and Conservation Council is to provide a forum for consultation and co-ordination between State, territory and Commonwealth governments of Australia and the Government of New Zealand on environmental and conservation issues.

The Australian Water Quality Guidelines for Fresh and Marine Waters has been prepared with the assistance of a large number of organisations and individuals over a period of two years.

The Australian and New Zealand Environment and Conservation Council would like to acknowledge the efforts of the working group set up to oversee the development of the guidelines and co-ordinate input from Commonwealth, State and Territory environmental and resource management agencies, and the co-operation and comments provided by the Australian Water Resources Council.

Particular thanks are extended to the scientific consultants Barry Hart, Connie Angehrn-Bettinazzi and Ian Campbell of the Water Studies Centre, Monash University, and Michael Jones of NSR Environmental Consultants, Hawthorn; and also to Jane Finlay for editing this document.

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1 Introduction

The Australian community has clearly indicated that the adequate long-term protection and enhancement of all facets of the Australian environment will be one of the most important issues of the 1990s and beyond. The Federal Government has responded to this concern by developing a policy for 'ecologically sustainable development' (Commonwealth of Australia 1990) that will assist in ensuring that orderly economic, industrial and urban development is achieved without long-term degradation of Australia's resources and environment.

The Australian and New Zealand Environment and Conservation Council (ANZECC) and the Australian Water Resources Council (AWRC) are developing a national water quality management strategy that seeks to manage the nation's water resources on a sustainable basis. There are three main elements to this strategy:

- an overview document entitled 'National Water Quality Management Strategy: Policies and Principles—A Draft Reference Document' (ANZECC/AWRC 1992);
- guidelines on specific water quality issues, including rural water quality, sewerage systems, groundwater protection and drinking water;
- the Australian Water Quality Guidelines for Fresh and Marine Waters.

The Australian Water Quality Guidelines provide numerical and narrative criteria to assist in managing water resources in a sustainable manner. The approach adopted in developing these water quality guidelines is consistent with the philosophy given in the overview document, 'National Water Quality Management Strategy: Policies and Principles'.

The overview document describes the policy framework for achieving ecologically sustainable management of the nation's water resources. It draws on the five key principles set out in 'Ecologically Sustainable Development—A Commonwealth Discussion Paper' (Commonwealth of Australia 1990). These principles are the requirements for national policies to:

- integrate economic and environmental goals
- recognise the value of the asset
- provide for inter-generational equity
- adopt a precautionary approach
- recognise the global dimension.

The overall objective of the water quality management policy (ANZECC/AWRC 1992) is to:

> …achieve the sustainable use of the nation's water resources by protecting and enhancing their quality while maintaining economic and social development.

To achieve this objective, the environmental values of water resources need to be defined and protected from the effects of degradation, including pollution.

Each waterbody may have a number of environmental values, including recreational use, potable water supply, ecosystem protection and irrigation, that need to be measured in order to test whether they are being protected. Water quality criteria or reference values provide the means to

make such measurement. Each environmental value is given a set of numerical or narrative criteria that must be met to ensure that the particular environmental value can be fully protected. The criteria are stringently set to minimise the risk of detrimental effects on the aquatic environment; however, significant environmental benefit may still be achieved in some ecosystems without fully meeting the criteria.

The approach of protecting specifically recognised environmental values is widely used internationally, although terminology may differ from country to country. Environmental values are also commonly referred to as 'beneficial uses'. ANZECC considers that the latter terminology suggests an exploitative approach to the management of environmental resources that could be seen to be at odds with the goal of sustainable use. For this reason, the term 'environmental values' will be used throughout this document.

There can be conflict between the various environmental values; for example, using the water resource for irrigation may mean that it is not available for recreational purposes. The National Water Quality Management Strategy recognises the role of the community in resolving these conflicts and in determining what values a water resource should support. It is envisaged that the water quality guidelines or reference values for a particular waterbody will assist the community in making informed choices about the value and use of that waterbody.

1.1 Management of water quality

In developing these national water quality guidelines, account has been taken of the following features of the Australian water environment:

- the great variability of the water environment in Australia, particularly in flow and ecosystem types;
- the wide range of environmental values that may be recognised for different waterbodies, leading to the adoption of a range of desired water quality conditions;
- the need for interstate co-operation for shared watersheds (e.g. the Murray–Darling system);
- the recognition that some aquatic systems can never support particular environmental values because they have been degraded naturally or irreversibly changed (e.g. long-term increase in salinity). However, other waterbodies may be improved with remedial action or restoration programs so that additional environmental values are achieved or existing values are upgraded.

It should be recognised that some of the environmental values of aquatic ecosystems may not be protected merely by maintaining water quality. The total load and fate of pollutants to enclosed systems should be considered, particularly where nutrients and cumulative toxicants are discharged. In addition, issues such as the maintenance of fringing vegetation, the diversion of flow and the presence of barriers to migration may significantly influence the quality of associated ecosystems.

The water quality management approach involves:

- identifying the environmental values of particular waterbodies that are to be protected;
- establishing the objectives that will achieve the required level of protection. These objectives are established in terms of key indicators of quality (physico-chemical and biological) using the collated scientific information relating to each indicator and each environmental value;

- establishing water quality management strategies (e.g. policies covering receiving water, effluent, non-point source and catchment management) that will provide the instruments for achieving the objectives;
- developing and initiating a monitoring and surveillance program to ensure that the water quality (or environmental) objectives are being maintained;
- initiating a research program to fill in the unknowns and to refine scientific information relating to each particular aquatic system.

In Australia, this process is best implemented by integrating national, State and regional powers and responsibilities, and by using complementary water quality planning and policy tools. The administrative process would be as follows:

- Each State would use its own water quality planning and environmental policy tools to set water quality objectives and goals consistent with the agreed national guidelines.
- Regional communities would be encouraged to participate in the identification of local environmental values to be protected and the associated water quality guidelines to achieve that protection.
- Local management strategies would be developed and implemented by relevant stakeholders.

The formation of the National Environment Protection Authority (NEPA) provides significant opportunities to achieve effective integration of these processes.

1.1.1 Environmental values

Identification of the needs and wants of the community (e.g. agricultural water, swimming, commercial or recreational fishing, protection of the ecosystem) is an essential step in defining the environmental values of a particular waterbody. Once these are established, the water quality required to achieve or maintain these values can be determined and compared with the existing water quality and the community requirements. In some cases, the values needed to maintain or achieve the desired water quality may not be attainable; for example, a stream may have a naturally high salinity that limits its use for maintaining particular fish species. However, it is important to determine whether the background values reflect the 'natural' conditions or an already significantly modified system.

It should be recognised that there will be costs associated with managing a waterbody to the water quality level desired by the community. The community may modify its needs and wants when balancing the inherent costs and benefits associated with attaining particular water qualities. The objectives and environmental values to be maintained or achieved need ultimately to be formalised and made available to the whole community.

Until recently, management of Australia's water resources was primarily focused on protecting human health-oriented environmental values, such as quality of drinking water, agricultural water and water in which seafoods are grown. Maintenance of water quality to protect aquatic ecosystems was often included, but the approach was based on a very deterministic view of ecosystems that assumed that the factors controlling the functioning of ecosystems could be easily identified and would not lead to problems if maintained below certain levels. However, it is now well recognised that the relationships between key ecological processes and their components are complex, variable (probabilistic) and cannot be determined precisely. The water quality guidelines provided in this document attempt to take these factors into consideration.

Five environmental values are considered in these national guidelines:

- ecosystem protection (both inland and marine), including protection of waters used for shellfish and fish production and by wildlife;
- recreation and aesthetics;
- raw water for drinking water supply;
- agricultural water;
- industrial water.

This document provides up-to-date scientific information on water quality to enable local communities to make informed choices about the environmental values for a particular waterbody (and the feasibility of achieving these values), and to assist resource managers to develop water quality management strategies that will meet community expectations.

1.1.2 Defining objectives

Each environmental value requires a certain level of water quality to be maintained. The water quality required to sustain environmental values may be defined by establishing water quality objectives that become the goals for management action. This is essentially a complex process, depending on factors such as the feasibility and costs of achieving the desired water quality and the opportunity costs to the community if these environmental goals are not reached.

The distribution of costs and benefits will vary according to different community sectors and interests. A cost to one sector of the community will often be a benefit to another sector. A simple example of this variation would be a stream that has swimming as a designated environmental value but has been rendered unsuitable for swimming by wastewater discharge.

Loss of the swimming resource is a cost borne by the swimmers, while the discharger of the waste enjoys an economic benefit from the disposal of the wastewater to the stream. If swimming were to be restored, the discharge would need to be improved or removed entirely from the stream. This would lead to a redistribution of costs and benefits within the community, with the swimmers benefiting from the improved water quality and the discharger faced with additional costs to treat the wastewater. If the discharger were a public body, such as a sewerage authority, these costs would be levied in the form of higher rates to the local community, which would then need to decide whether or not it wished to retain that particular stream for swimming in light of the increase in rates.

The distribution of costs and benefits can be much more complex than in this simple example. Much degradation of the water environment has resulted from the effects of land-use patterns and the associated runoff. Beneficiaries may lie outside a local catchment area if national or State priorities are pursued or if downstream users are considered. A clear and effective process of consultation is therefore required.

Key indicators of quality

Sets of key indicators of water quality need to be established to provide a means of identifying and measuring change in each environmental value. Three types of indicators of environmental quality can be identified:

- those that are normally present in the water and that can be usefully monitored for a change in concentration, quantity or quality, some or all of which can be linked to a change in the environmental value;
- those that are not normally present, but if detected in certain concentrations or quantities can be used to identify a change in or effect on an environmental value;
- indicators that are normally present but the absence of which reflects a change in an environmental value.

Colman et al. (1991) identified the following desirable features of key indicators:

- they should be able to be unambiguously related to environmental change in the identified environmental value;
- they must have sufficient statistical power to be monitored with confidence of detecting meaningful change;
- monitoring of the indicator must be logistically achievable, of an assured quality and capable of being audited in the long term (the latter is particularly important for ecosystem protection).

The use of indicators to provide for adequate protection is based on the premise that, firstly, it is possible to specify particular (preferably measurable) indicators of quality and, secondly, there are particular concentrations of these indicators that can be specified below which either adverse effects will not occur (i.e. a threshold level) or the risk of such effects is low.

The key indicators used in defining water quality are still largely physico-chemical; for example pH, conductivity, nutrient concentrations and toxicant concentration. For drinking water, microbial and radiological indicators are also included to ensure adequate protection of human health. However, major deficiencies with the present guidelines for ecosystem protection are the general lack of biological and ecological indicators and the general difficulty in taking account of the dynamic nature of aquatic ecosystems that, as natural systems, are subject to a range of factors (e.g. floods, drought) that can cause major fluctuations in almost any indicators selected. These deficiencies are further discussed in Chapter 2 and in Hart (1992).

1.1.3 Water quality management strategies

It should be noted that the water quality objectives defined by the process described in the previous section apply to ambient water quality and not to effluent quality.

The water quality objectives defined by the described process will require action to attain and/or maintain the desired quality and achieve the values identified by the community. However, even when the most stringent effluent limits are set and strict waste minimisation is practised, effluents may be of poorer quality than the receiving water quality objectives. In these cases, it has been the practice of water quality managers to use the concept of the 'mixing zone', an explicitly defined area around an effluent discharge where certain environmental values are not protected. Effective discharge controls that consider both the concentration and the total mass of pollutants, combined with *in situ* dilution and waste treatment, should insure that the area of a mixing zone is limited and the values of the waterbody as a whole are not prejudiced. The environmental conditions within a mixing zone, and its size, are important community concerns, particularly because degraded areas around effluent discharges reduce the environmental benefits to the community.

Some waters (e.g. many of those in national parks or reserves) are highly valued for their pristine state and outstanding natural values. In many other countries, and 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. Even where some degradation is permitted, water quality objectives may be set at a level that is more stringent than a direct application of the guidelines would imply.

Development of a water quality management strategy is dependent on the quality of available information and the capacity to predict the effects of various actions on water quality. It is common to use models to predict the effects of discharges on the environment. These models may be conservative and rely on worst-case conditions of dilution and degradation of effluents, or may attempt more complex analysis of cause and effect. Because of the unique features of each system (particularly the ecosystems), it has generally been found that models developed for a particular waterbody cannot be used for other waterbodies without significant modification.

One model that is raised frequently in the context of water pollution control is that of 'assimilative capacity'. The underlying philosophy of this concept is that a waterbody has the potential to process pollutants up to some defined limit without causing harm to the environment or significant degradation of water quality. This is a very controversial issue in environmental and resource management literature. In the past it has been applied to all types of contaminants (i.e. naturally occurring and synthetic toxicants and biostimulants) with a limited understanding of ecological processes and theory.

Recently, the concept of assimilative capacity has been redefined and applied to prevent and control the undesirable effects of nutrient enrichment of waterbodies (Masini et al. 1992). This approach is based on establishing linkages between total nutrient loadings to an ecosystem and the response of the most sensitive/important component of that ecosystem. Once these relationships have been quantified, and the desired environmental quality objective defined, regulatory agencies can use this information to set ecologically-based maximum nutrient loadings to that ecosystem that are consistent with maintaining the desired environmental quality.

This ecologically-based management approach takes into account the degree of error or uncertainty associated with the relationship between nutrient loading and ecological effect, and maximum nutrient loadings are set accordingly to offer adequate protection to the ecosystem. This approach can be used to both manage the recovery of degraded systems and to prevent incremental degradation of healthy systems in the face of urban, rural and industrial expansion. This is further discussed in Section 2.1.2.

1.1.4 Monitoring and assessing water quality

It is of limited value to recommend water quality guidelines without coupling these guidelines to the methods and frequency of sampling and analysis required to adequately monitor and assess the water quality. For example, in the case of heavy metal toxicants, decisions are needed on whether to analyse filtered or unfiltered samples and on the type of treatment of these samples (e.g. acidified or acid digested). The ultimate decision on how to treat each sample can have a major influence on the actual concentration recorded. For heavy metal toxicants, it is the 'biologically available' fraction that is of concern; however, the current state of knowledge is insufficient to permit a single analytical

technique or instrument to be used to measure this fraction. For this reason, it is recommended that toxicant concentrations be measured in unfiltered samples to ensure a conservative approach.

The factors that need to be considered when assessing the physico-chemical or biological quality of a water resource are covered in Chapter 7. Traditionally, physico-chemical indicators have almost exclusively been used to assess water quality and, for this reason, sampling and analysis techniques are generally well developed, although not necessarily consistently used throughout the country. Increasingly, biological (or ecological) assessment of waterbodies is being undertaken in Australia, although there is still some way to go before the procedures being developed are accepted nationally.

There is an urgent need to develop national protocols for the physico-chemical and biological assessment of water quality. Many of the guidelines recommended in this document (particularly those for ecosystem protection, Chapter 2) are provided as ranges of values and presume that additional site-specific information will be obtained by State agencies to identify the specific levels to be adopted.

1.2 Approach adopted for the Australian Guidelines

The approach adopted in preparing these Australian Water Quality Guidelines comprised:

- a review of the most recent overseas criteria documents, particularly those produced by the United States (USEPA 1986), Canada (CCREM 1991) and the World Health Organisation (WHO 1980, 1984);
- a broad review of the relevant Australian information (both published and unpublished) to supplement or modify overseas information where this was judged to be inappropriate;
- a detailed public review of the draft.

For each environmental value, specific guidelines have been formulated in terms of the key indicators. A summary of the relevant scientific and technical information, together with adequate referencing of the source information, is provided as the rationale for each guideline.

Guidelines or criteria

Water quality criteria comprise the scientific and technical information used to provide an objective means for judging the quality needed to maintain a particular environmental value. Specifically, criteria are the quantitative (or qualitative) data that predict the chance or magnitude of the effects of a contaminant on a defined receptor (e.g. human, aquatic organism) under specific environmental conditions. These criteria are described in terms of physical, chemical, radiological, microbial and biological indicators of water quality.

Water quality guidelines translate the criteria into a form that can be used for management purposes. In many cases this will involve some value judgement on an acceptable risk of human health or ecosystem impairment. Standards are what guidelines (perhaps further modified by social, political and/or economic considerations) become when compliance is enforced by law.

Different approaches have been used to develop guidelines related to different indicators. In the case of toxicants (e.g. heavy metals, pesticides), the approach generally adopted to provide adequate protection for aquatic ecosystems has been to estimate or predict an 'acceptable safe concentration' based on toxicity test results. Much of the available toxicity data relates to acute lethal concentrations (e.g. the concentration required to kill 50% of the test organisms in four days, referred to as the LC50), with a considerably smaller amount of data relating to chronic tests. The common practice is to reduce the acutely toxic concentration by a

factor of 0.1–0.01 to obtain a 'safe concentration'. Sometimes it is possible to provide quantitative information on factors that can modify the toxic effects. For example, the toxicity of a number of heavy metals is significantly modified by the hardness and concentration of dissolved organic matter in the water. Because there are few data on the effects of toxicants on Australian native aquatic animals and plants, it has been necessary to place considerable reliance on overseas data.

It is difficult to establish guidelines for indicators that are naturally present in a waterbody (e.g. nutrients, temperature, colour, salinity), but that cause adverse effects when present in excessive amounts, particularly with the wide range of ecosystem types that must be considered in these national guidelines. Sometimes it is possible to express a guideline in terms of a maximum per cent change in the natural values of the particular indicator. This approach has been adopted by the Victorian Environment Protection Authority (VicEPA) for salinity (no greater that a 2% change; Vic. Govt 1988), and is being considered by New Zealand in relation to the optical properties of waters required to support the standard referring to 'conspicuous change in the colour or visual clarity' in the recent New Zealand Resource Management Act 1991 No. 69 (NZRMA 1991). Difficulties arise with this approach in determining just what is the 'natural concentration', given the variability known to occur in most aquatic systems.

This document recognises the natural variability that occurs both within particular aquatic systems and among different ecosystems (e.g. alpine streams compared with tropical billabongs). Reference values, or guidelines, are provided to assist the community in making choices concerning water quality and to provide guidance for water resource managers. In most cases a single reference value has been given; however, a range of concentrations for a particular indicator has been provided where appropriate. In the special case of nutrients, no specific reference values are provided, because a number of factors in addition to nutrients can influence the growth of nuisance aquatic plants.

In all cases where broad national guidelines are provided, it is essential that site-specific information is obtained to determine more precisely the range of values that will provide the required level of protection for that system.

It is important to recognise that these Australian guidelines have been developed on the basis of information currently available. A process for updating them as new and relevant information becomes available must be put in place.

In all cases an attempt has been made to provide guidance on the range of concentrations or levels of each key indicator required to provide adequate protection of the environmental value. It is important that these guidelines are not considered as blanket values for national water quality. There are a wide range of ecosystem types throughout Australia, and to assume that one set of specific values could apply equally to all would be ill advised. Local, site-specific information will be needed to supplement the broad information provided in this document, particularly for ecosystem protection.

In the case of toxicants, guidance is provided on the typical maximum concentrations permissible for adequate protection. However, this should not be taken as an indication that the environment can be contaminated up to these levels in systems where the existing levels are lower than the recommended values. For those indicators where guideline ranges are provided, it is the expectation that State environmental and resource management agencies will undertake local, site-specific investigations of their own systems to confirm the specific levels to be adopted. Guidance on the types of physico-chemical and biological survey methods that may be used is provided in Chapter 7.

1.2.1 Limitations of the present guidelines

The approach adopted in developing these Australian Water Quality Guidelines has limitations that must form the focus of future investigations. First, the water quality indicators used are almost exclusively physico-chemical and not biological. ANZECC has identified the development of biological guidelines as key indicators of aquatic ecosystem health, and some suggestions have been made in this document about potential biological indicators. Second, regarding toxicants, it is assumed that if certain key biological species (those used for bioassays) are protected from toxic effects, this will be sufficient to provide protection for the whole ecosystem. Third, these guidelines allow protection of the aquatic environment by considering the quality of the water compartment. While this is important, it is not necessarily adequate for protection of the whole ecosystem. Other factors, such as serious pollution of sediments, reduction in stream flow (from damming and building of barriers) or removal of habitat (de-snagging, draining wetlands), could equally cause significant deterioration in the ecosystem and need to be addressed.

There is a strong case for a new approach to be developed if Australia is to successfully tackle the task of ecologically sustainable development and the maintenance and improvement of its water resources (inland and marine) through the 1990s and beyond. The main focus of this new approach must be towards ecologically-based management. In the present context, this would involve extending the concept of water quality guidelines to ecosystem or environmental guidelines, where the maintenance of adequate water quality is seen as only one (albeit important) component of protecting the resource. This concept (discussed more fully in Section 2.1) is seen as central to any water quality (environmental) management policy developed for the 1990s, and must be reflected in the water quality guidelines established as a backup to such a policy. It should be noted that environmental requirements are a central feature of the new Water Law recently promulgated in Victoria (Vic. Govt 1990).

There is increasing public concern that objectives aimed solely at protecting the quality of the water may be insufficient to ensure that the total aquatic system is adequately protected. It is essential that other components of the aquatic system (e.g. biota, sediments, riparian vegetation, catchment use) are also considered. Pressure is mounting for an expansion of the concept of water quality guidelines—perhaps to broader 'environmental' guidelines for water resources—where the present emphasis on physico-chemical indicators of water quality would be supplemented with information on biological indicators, broad ecosystem 'health' indicators, sediment indicators and 'habitat' indicators. There is insufficient information presently available to define these latter indicators. This will require further discussions between scientists and managers and probably further research into particular aspects.

1.3 Structure of the document

In addition to this introductory chapter, the Australian Water Quality Guidelines for Fresh and Marine Waters comprises five chapters, each covering one of the environmental values. Chapter 2 covers ecosystem protection (both inland and marine), including protection of waters used for shellfish and fish production and for wildlife. Chapter 3 deals with water for recreation and aesthetic uses, Chapter 4 with raw water for drinking water supplies, Chapter 5 with agricultural water supplies, and Chapter 6 with industrial water supplies. Chapter 7 includes information on physico-chemical and biological methods for assessing water quality. A list of references is provided in

Appendix A. Appendix B contains a glossary of terms used in the document, and Appendix C provides a list of abbreviations and acronyms.

2 Protection of aquatic ecosystems

Aquatic ecosystems comprise the plant, animal and microbial communities that live in water and the physical environment and climate regime with which they interact. Humans have caused profound changes in Australian aquatic ecosystems, particularly in the 200 years since European settlement of the continent (e.g. Senate Select Committee 1970; Wood 1975; SER 1988). Community concern about the lack of attention given to the ecological degradation associated with development undertaken in pursuing material and social welfare has lead the national government to adopt a policy of ecologically sustainable development (Hawke 1989). In particular, this policy will aim at maintaining sustainable ecosystems and preserving genetic diversity. The new national water quality management policy developed jointly by ANZECC and AWRC reflects this ecologically sustainable development thrust. The policy objective is to:

> ...achieve sustainable use of the nation's water resources by protecting and enhancing their quality while maintaining economic and social development. *(ANZECC/AWRC 1992)*

The establishment of national water quality guidelines is an integral part of the national water quality management strategy.

This chapter specifies guidelines for the protection of freshwater and marine aquatic ecosystems. A summary of the guidelines is given in Table 2.1, and a full rationale for each indicator is given in the relevant section. The chapter comprises five sections: Section 2.1 covers the background philosophy and the procedures used to develop the guidelines; Section 2.2, the biological guidelines; Section 2.3, physico-chemical guidelines; Section 2.4, toxicant guidelines; Section 2.5, additional guidelines to protect those waters used for the production of fish and shellfish for human consumption; and Section 2.6, the guidelines for wildlife protection.

The guidelines required to protect aquatic ecosystems are often the most stringent, and generally ensure that other related environmental values, such as edible fish and shellfish, and wildlife, are also protected. For this reason, the guidelines for edible fish and crustacea, shellfish culture and harvesting, and maintenance of wildlife have been included in this chapter. It is assumed that the general guidelines relevant to protection of an ecosystem (freshwater or marine) will be maintained for each of these environmental values. Where it has been assessed that extra indicators specific for protection of the ecosystem are needed, these have been provided.

2.1 Development of guidelines

There is now little argument that the long-term survival of the human race is dependent on maintaining the integrity of the earth's biological systems. In addition, aquatic ecosystems are worthy of protection for their own intrinsic value. Many of the benefits Australians obtain from aquatic ecosystems can only be maintained if the ecosystems themselves are protected from degradation. Commercial and recreational harvests of fish and shellfish can only be obtained from waters where there are ecosystems providing the food and habitat to support the growth and reproduction of the harvested species. Conservation of endangered species can only be achieved by conserving the ecosystems that support them. Many water-linked recreational activities, such as swimming and scuba diving, aesthetic enjoyment and appreciation of nature, require or benefit from healthy aquatic ecosystems (O'Brien et al. 1983). The need to maintain balanced aquatic ecosystems

was highlighted by the many problems, including the unsuitability of the water for drinking or stock watering, that resulted from a massive algal bloom that occurred in the Darling River in the latter part of 1991.

Traditional approaches to the management of water quality grew from concerns about human health. Management and monitoring practices were initially established by sanitary engineers, microbiologists and health workers to assess and manage contamination of water by sewage and materials toxic to humans. The health of the human community was the ultimate indicator of the success of a program. This approach was superseded by the recognition of the need to protect aquatic ecosystems using toxicity test data to derive chemical water quality criteria. However, the problem of ensuring the protection of aquatic ecosystems is less tractable than that of protecting human health because many more species are involved, all with their unique sensitivities and ecological requirements. For aquatic ecosystems, the ultimate indicator of success must be the health of the biological community. As with the health of human communities, there is no simple numerical measure of the health of aquatic ecosystems, nor is one likely to be developed. However, it is obvious that biological evaluation and monitoring of ecosystems is necessary to ensure their protection.

2.1.1 Philosophy

The national guidelines for the protection of aquatic ecosystems are based on the ecologically sustainable development philosophy, where, in terms of the ANZECC/AWRC water quality management policy, the goal is to protect biological diversity (biological diversity) and maintain ecological processes and systems. However, establishing what is an adequate level of biodiversity protection is not simple. 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 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.

Thus, the task is to maintain the 'ecological integrity' of each marine and freshwater resource.

2.1.2 Ecosystem response and ecosystem management

Pearce et al. (1989) suggest that ecological integrity is maintained when the productivity, stability and resilience of the system are sustained; that is, when the system is 'ecologically healthy' and has the capacity to perform all essential ecological processes. In addition, a system has ecological integrity when it has the ability to maintain evolutionary potential in the long term. When the ecological integrity of a natural system is reduced, the capacity of that system (and species within it) to survive change is also reduced.

Table 2.1 Summary guidelines for protection of aquatic ecosystems

Type of indicator	Indicator	Units	Fresh waters	Marine waters
Biological			It is premature to recommend specific values for these indicators. The need for biological evaluation is recognised, and these indicators are identified as important characteristics of ecosystem function (Section 2.2)	
Physico-chemical	Colour & clarity	–	< 10% change in euphotic depth[1]	< 10% change in euphotic depth

Type of indicator	Indicator	Units	Fresh waters	Marine waters
	Dissolved oxygen[2]	mg/L	> 6 (> 80–90% saturation)	> 6 (> 80–90% saturation)
	Nutrients/nuisance growths	–	(Section 2.3.3)	(Section 2.3.3)
	pH	–	6.5–9.0	< 0.2 pH unit change
	Salinity	mg/L	< 1000 (about 1,500 μS/cm)	–
	Suspended particulate matter/turbidity	–	< 10% change seasonal mean concentration (see also colour & clarity)	< 10% change seasonal mean concentration (see also colour & clarity)
	Temperature[3]	–	< 2°C increase	< 2°C increase
Toxicants				
Inorganic toxicants	Aluminium	μg/L	< 5.0 (if pH < = 6.5)	NR
	Aluminium	μg/L	< 100.0 (if pH > 6.5)	–
	Ammonia	μg/L	20.0–30.0 (Table 2.3)	NR
	Antimony	μg/L	30.0	500.0
	Arsenic	μg/L	50.0	50.0
	Beryllium	μg/L	4.0[4]	NR
	Cadmium	μg/L	0.2–2.0[5]	2.0
	Chromium	μg/L	10.0	50.0
	Copper	μg/L	2.0–5.0[5]	5.0
	Cyanide	μg/L	5.0	5.0
	Iron	μg/L	1,000.0[6]	NR
	Lead	μg/L	1.0–5.0[5]	5.0
	Mercury	μg/L	0.1	0.1
	Nickel	μg/L	15.0–150.0[5]	15.0
	Selenium	μg/L	5.0	70.0
	Silver	μg/L	0.1	1.0
	Sulfide	μg/L	2.0	2.0
	Thallium	μg/L	4.0	20.0
	Tin (tributyltin)	μg/L	0.008	0.002
	Zinc	μg/L	5.0–50.0[6]	50.0
Organic toxicants	Acrylonitrile	μg/L	NR	NR
	Benzidine	μg/L	NR	NR
	Dichlorobenzidine	μg/L	NR	NR
	Diphenylhydrazine	μg/L	NR	NR
Halogenated aliphatic compounds	Hexachlorobutadiene	μg/L	0.1	0.3
	Halogenated ethers	μg/L	NR	NR
	Isophorone	μg/L	NR	NR
Monocyclic aromatic compounds	Benzene	μg/L	300.0	300.0
	Chlorinated benzenes	μg/L	(Table 2.8)	NR

Type of indicator	Indicator	Units	Fresh waters	Marine waters
	Chlorinated phenols	μg/L	(Table 2.9)	(Table 2.9)
	Phenol	μg/L	50.0	50.0
	Toluene	μg/L	300.0	NR
	Nitrosamines	μg/L	NR	NR
Pesticides	Organochlorine	μg/L	(Table 2.10)	(Table 2.10)
	Organophosphate	μg/L	(Table 2.10)	(Table 2.10)
	Acrolein	μg/L	0.2	0.2
Phthalate esters	di-n-butylphthalate	μg/L	4.0	NR
	di(2-ethylhexyl)phthalate	μg/L	0.6	NR
	other phthalate esters	μg/L	0.2	NR
Polyaromatic hydrocarbons	Chlorinated naphthalenes	μg/L	NR	NR
	Polychlorinated biphenyls	μg/L	0.001	0.004
	Polychlorinated dibenzo-p-dioxins	μg/L	NR	NR
	Polycyclic aromatic hydrocarbons	μg/L	3.0	3.0

SPM: Suspended particulate matter; NR: no recommendation made at this time

Notes

1. For systems where depth is greater than 0.5 x euphotic depth (z_{eu}). For waters shallower than 0.5 z_{eu}, the maximum reduction in light at the sediment bed should not exceed 20%
2. Measured over at least one, but preferably several, diurnal cycles
3. Or use formula in Section 2.3.7; no data for temperature reductions
4. Higher values may be acceptable in hard waters
5. Depends upon hardness of water
6. Provided iron not present as Fe(II)

An indicator of ecological integrity is the degree to which ecosystems have been altered from their natural state. However, defining 'natural state' is problematic. In Australia, the natural state is often taken to be that existing before European settlement. However, it is now increasingly recognised that traditional Aboriginal land uses had significant impact on the terrestrial environment (e.g. structure and composition of vegetation communities and the distribution and abundance of fauna). The effects on the aquatic environment are not known, but would probably have been somewhat less. Nevertheless, the ecological changes resulting from European occupation have been significantly more rapid and far-reaching, such that few natural reference points now exist. Those that do exist are generally labelled as 'pristine' systems.

A wide range of human activities can adversely affect aquatic ecosystems, including:

- pollution from industrial, urban, agricultural and mining sources
- siltation and sedimentation from land clearance, forestry and road building
- salinisation
- river regulation via dams and weirs

- clearance of stream bank vegetation
- over-exploitation of fisheries resources
- introduction of exotic species.

It has been suggested that the single biggest threat to the maintenance of biodiversity is habitat destruction (Biodiversity Working Party 1991).

The need for a broader, more holistic approach to ecosystem management was foreshadowed in Introduction. Such an approach would require consideration of all changes, not just those affecting the quality of the water compartment. Such changes could include seriously polluted sediments, reduction in stream flow (from damming and building of barriers), removal of habitat (de-snagging, draining wetlands) or significant catchment land use changes, any of which could cause significant deterioration of the ecosystem. The water quality guidelines documented here are a necessary, but only partially sufficient, tool for ecosystem management.

Biological systems are known to be inherently variable and this, coupled with the marked differences in sensitivity of different ecosystems and biological communities to particular pollutants and other stressors, makes it essential that management must occur on an ecosystem-by-ecosystem basis. Rarely will it be possible to predict whole ecosystem effects with a high degree of confidence; indeed, most often such predictions will be made with a relatively high degree of uncertainty. Recognition of this variability and uncertainty has lead to the development of new predictive and adaptive, yet holistic, ecosystem-based management methods.

'Assimilative capacity' has been used as a tool for managing environmental degradation for several decades. The underlying assumption of this approach is that an ecosystem has a capacity to absorb and modify wastes without altering the quality of that ecosystem. This concept, as defined, has been largely discredited by the scientific community (Campbell 1981, 1986), since it has been applied to pollutants, such as toxicants and salinity, that are not assimilated by the system; rather, these toxicants may be tolerated by individual biological species or the entire ecosystem when present below certain concentrations. Therefore, protection of aquatic ecosystems from the effects of naturally occurring toxic substances, which act according to their concentrations in solution, is best achieved by utilising water quality criteria based on toxicological studies.

Nutrients, on the other hand, are not toxic (except at very high concentrations) and require a different management approach. These substances stimulate plant growth, resulting in larger than normal standing crops of algae and organic matter in the ecosystem. The amounts that will grow are dependent on specific characteristics of the ecosystem in question (e.g. flushing/retention times, dominant biota) and on the total nutrient loading to that ecosystem. Excessive amounts of algae, be they phytoplankton, epiphytes or macroalgae, can cause serious and sometimes irreversible long-term damage to an ecosystem. To ensure that this does not occur, the critical links between nutrient loadings and environmental response must be determined to develop appropriate ecologically-based management strategies.

In recent years the concept of assimilative capacity has been redefined (Masini et al. 1992) and proposed as a management tool to set ecosystem-specific, ecologically-based limits on cumulative nutrient loads. The approach is to define the ecosystem boundary, quantify the total cumulative load of nutrients to that ecosystem from all sources, identify the key pathway of nutrient conversion into

organic matter and link this with the most sensitive/important component of the ecosystem in question. It is this final relationship, the link between the nutrient load to the system and the ecological response of the system, that provides the information required to set appropriate loading limits. This approach accommodates the error associated with the assimilative capacity estimate by using the lower error boundary (the working assimilative capacity; Masini et al. 1992) as the initial maximum permissible load, coupled with tactical monitoring programs and periodic review. This combination of prediction, acknowledgement of uncertainty, monitoring and review allows management strategies the flexibility to be adapted to an improved knowledge base provided by well-planned monitoring programs.

Given the historical use and connotations applied to the term 'assimilative capacity', modern usage of this term should be restricted to the management of nutrient pollution and used within the context of cumulative impact and acceptable ecological change. This latter concept acknowledges that the system will change no matter how small the additional anthropogenic nutrient loading.

The concept of Adaptive Environmental Assessment Management (AEAM), developed first by Holling (1978) and then by Walters (1986), is a promising ecosystem management tool that can integrate a range of management strategies and responses. AEAM introduces adaptive and flexible management methodologies that establish procedures to manage in conditions of uncertainty and develop more robust predictive tools of environmental assessment and bio-economic analysis. AEAM recognises the importance of accumulating systematic and detailed knowledge, but not the need to 'know everything' before effective management can occur. Rather, this methodology seeks to put in place a flexible, adaptive management system that can adjust as further information becomes available. It can also play an important role in pin-pointing more precisely those areas where additional scientific and other information is required. The AEAM methodology is being used increasingly in Australia: for example, in Macquarie Marshes, New South Wales (Gilmour & Geering 1989); Latrobe River Water Quality Management Strategy, Victoria (Blake T., VicEPA, pers. comm., July 1992); and Beenyup Outfall, Western Australia (Walters C.J., University of British Columbia, pers. comm., May 1992).

2.1.3 Levels of protection

Some aquatic ecosystems have been significantly and permanently modified by human activity and can never be returned to 'natural' conditions, and it has been a common practice to designate lower levels of protection for these ecosystems. For example, the Victorian EPA (VicEPA 1983) stipulates two levels of protection, one for maintenance of aquatic ecosystems and one for the maintenance of modified aquatic ecosystems. However, there are philosophical difficulties associated with designating lower levels of protection for some ecosystems based on the nomination of arbitrarily selected criteria.

Two categories of aquatic ecosystems can be distinguished. The first category includes those pristine ecosystems not subject to human interference through discharges or activities within the catchment. Such systems are now largely restricted to national parks, and it is appropriate that the existing water quality of pristine ecosystems be maintained through strict management.

The second category includes all modified ecosystems subject to human interference. Some modified ecosystems have been permanently altered physically, for example through channelling of the stream or port construction. Others may be changed through long-term chemical toxicity caused by

contaminated sediments. Still others can be modified because of changed flow regimes resulting from, for example, the regulation of a river system.

The intention of the guidelines provided here is to ensure that water quality is not a constraint on the development and long-term maintenance of a healthy biological community, although it is recognised that, despite the quality of the water, changes such as those outlined previously may limit the biological communities that can develop and persist. It is not yet possible to specify with any degree of certainty just what constitutes a 'healthy' or 'acceptable' aquatic ecosystem—this is partially a scientific question and partially one related to what society considers is acceptable.

It is important that water quality managers recognise that the values provided in this document represent long-term goals, and that it is expected that less stringent standards may be set as interim goals.

2.1.4 Physico-chemical indicators

The establishment of physico-chemical water quality guidelines is necessary to protect ecosystems, to facilitate discharge controls and to identify problem contaminants in water. However, the establishment of appropriate guidelines for ecosystem protection is far more difficult in this case than for many other environmental values because of the complexity and diversity of the systems that must be protected.

For the Australian Water Quality Guidelines, the physico-chemical indicators have been separated into two groups: toxic and bioaccumulated chemicals and other physico-chemical indicators.

Toxic and bioaccumulated chemicals

Although there are often large differences in sensitivity to toxicants among different species, such differences are not systematic and, as a result, the Australian biota as a whole appears about as sensitive as biota from elsewhere (Skidmore & Firth 1983). Any given ecosystem is, *a priori*, as likely to contain sensitive species as any another. As a result, national guidelines for toxicants can be set with some degree of confidence that they will be applicable in most situations. The approach adopted here involved the following methodology:

- All components of the aquatic ecosystem (e.g. phytoplankton, zooplankton, benthos, macrophytes, fish) were considered where data were available. When data were limited, tentative guidelines were deemed preferable to no guidelines.
- For all toxicants, an attempt was made to obtain a minimum set of acute and/or chronic toxicity data (defined by CCREM 1991, Appendix IX). In very few cases was it possible to obtain any toxicity data for indigenous Australian aquatic plants or animals, since there has been very little toxicity testing done here.
- The Canadian approach was followed to the extent that the guidelines are set:

  … to protect all forms of aquatic life and all aspects of the aquatic life cycle… The intention is to protect all life stages during indefinite exposure to the water. *(CCREM 1991)*

- Overseas toxicity data were not considered if based on a single unsubstantiated value or a non-standard test.
- When only acute toxicity data were available, the following application factors were used to obtain 'safe' levels:
  - 0.05 x LC_{50} for materials that are non-persistent or are not accumulated;

  - 0.01 x LC_{50} for materials that are persistent or require additional caution because data are limited.

- Although it was recognised that natural variability should be considered in establishing guidelines for toxicants, it was considered that the detailed site-specific information needed to establish a statistical compliance level (e.g. 80 percentile, 95 percentile) must be obtained locally for each system.
- Analysis of the toxicant in an unfiltered sample is recommended. This approach is protective of the environment because it includes the measurement of forms of the toxicant that are unlikely to become biologically available. Analytical methods that measure the biologically active fractions directly are not yet available (Section 7.1.4). Again, there will be specific situations (e.g. extremely turbid rivers) where this approach may be overly cautious, but these should be assessed in the local context.

Other physico-chemical indicators

Depending upon geographical location, substantial differences can be found in the natural range of concentrations of indicators (such as nutrients, dissolved and suspended solids and temperature) that occur naturally in aquatic systems. Local biological communities are adapted to local conditions; for example, the high altitude streams of the Snowy Mountains have lower concentrations of dissolved solids, suspended solids and nutrients and lower temperatures in comparison with the lower reaches of the Darling River, and much of the high-altitude biota would be unable to survive in the Darling River. Thus, the geographic variation makes it inappropriate to define guidelines for such water quality indicators without reference to local conditions.

The approach adopted for the Australian Water Quality Guidelines has been to specify *ranges of values* for each of the physico-chemical indicators. The major factors that should be considered when setting appropriate local guidelines are also outlined. To develop appropriate local guidelines it will be necessary to improve the data base at the local level and to synthesise that data, both at the local scale, to understand how local systems work, and at larger scales, to develop state and national water quality perspectives.

2.1.5 Biological indicators

Biological water quality assessment must become an essential tool of resource managers responsible for protecting aquatic ecosystems, as only these biological techniques can demonstrate that the integrity of the ecosystem is being maintained. However, the development of biological assessment techniques applicable to the protection of aquatic ecosystems is in its infancy in Australia (Marchant & Chessman 1989; Underwood 1991a, 1991b), and indeed elsewhere around the world (USEPA 1990; Metcalfe-Smith 1992).

The problems of natural spatial and temporal variability evident in physico-chemical indicators are even more pronounced with biological indicators. Since Kolkwitz and Marsson (1902) first enunciated biological methods for the assessment of sewage pollution in European rivers, almost a century of research has served to emphasise the need to develop regionally appropriate techniques based on an appreciation of local aquatic ecosystems. While Australian aquatic ecosystems are still generally poorly described, much less understood, their regional diversity is, nonetheless, obvious. Thus, it will never be possible to propose meaningful simple numerical indicators (such as diversity index values or biotic index values) and expect that the numerical values generated can be used as absolute indicators. Rather, any biological assessment must depend on local comparisons to assess the

relative quality of two or more sites, or of a single site at a series of different times. Care must be taken to ensure that the data collected to facilitate such analyses can be validly interpreted statistically. For these reasons, it is appropriate here to recommend biological assessment methods but not to recommend the absolute values of indicator summary statistics.

The lack of information about, and understanding of, Australian aquatic systems, provides a strong argument for the implementation of adaptive management strategies using approaches such as those described by Walters (1986) and discussed in more detail earlier in this chapter. These techniques have been developed specifically to facilitate the management of ecosystems for which there is a high degree of uncertainty, and their implementation has much to offer aquatic resource managers in Australia. However, for such strategies to be successfully employed, the aquatic ecosystems being managed must be effectively monitored so that the impacts of management decisions can be confidently assessed. Historically, ecological monitoring of Australian aquatic environments has been rare and, more often than not, has been carried out with insufficient intensity or scientific rigour to draw more than a few, if any, conclusions about the direction or extent of ecosystem change (Underwood 1991b).

As noted above, the development of biological indicators for aquatic ecosystems protection is in its infancy around the world. In the United States, the EPA is attempting to produce a regional framework for resource management, primarily to facilitate water quality management (Gallant et al. 1989). The various 'ecoregions' have been identified using relevant information and a geographic information system. The system has been used to design monitoring and survey programs, conduct assessments of aquatic ecosystem quality and interpret chemical water quality monitoring data. The mapping of chemical water quality indicators, such as conductivity and total nitrogen (total-N) concentration, should allow more meaningful regional guidelines to be developed. Moreover, sites or areas differing from the regional pattern can be identified and investigated to establish whether they are being influenced by anthropogenic or natural disturbance.

Two methods that have been used with some success to assess the condition of aquatic ecosystems—the 'Index of Biological Integrity' (Karr 1991) and the 'Rapid Bioassessment Protocols' (Plafkin et al. 1989)—both rely on the ecoregions defined by the USEPA framework to identify 'control' sites. These are regional reference sites, selected to represent healthy ecological communities typical of the region. Such an approach has clear applications to Australia, both at the national and State levels.

Natural water quality conditions vary widely across the spectrum of inland and coastal waters of Australia. The Australian States are all large, therefore each contains a broad range of types of aquatic environments that must be considered separately for water quality management.

The comparison of Snowy Mountains' streams with the lower Darling River illustrated the differences between two types of stream environments within New South Wales. Such natural differences in water quality in different aquatic ecosystems lead to the biota of those systems having different tolerances to environmental stressors. For example, the Darling River was naturally subject to high salinities prior to significant impacts by Europeans. When Sturt first encountered the river in January 1829, he found it 'extremely nauseous, and strongly impregnated with salt', to the extent that neither the humans nor the stock in his party could drink from it (Sturt 1833). The biological communities of the Darling River may be expected to be more tolerant of high salinity than those of

the Snowy Mountains' streams, where salinities are always extremely low. As another example, in marine ecosystems, seagrass beds are more sensitive to turbidity (which reduces light penetration) than are mangrove systems.

This variation in tolerances in individual ecosystems may require the establishment of particular water quality guidelines for particular ecosystems. Such guidelines may correspond to regional guidelines, as would be expected in the case of salinity tolerances in inland waters, or may be far more site-specific, as is probably the case for seagrass beds.

2.1.6 Assessing ecosystem condition

Recent efforts to develop measures of aquatic ecosystem condition, or 'health', have focused on the biological integrity of ecosystems (Karr 1991). Biological integrity in this context has been defined as:

> …the ability to support and maintain a balanced, integrative, adaptive community of organisms having a species composition, diversity and functional organisation comparable to that of natural habitat of the region. *(Karr & Dudley 1981)*

Thus, the extent to which the integrity of an ecosystem is being maintained can only be assessed when the characteristic biological communities of a region are known or, since this will rarely be the case in Australia, by comparison of the biological community at the site(s) of interest with unimpacted communities in similar habitats elsewhere in the region.

Selection of appropriate biological indicators and the procedures to adopt in undertaking a biological water quality assessment are discussed in Section 7.2. The raw data collected in biological water quality assessments generally consist of lists of species and their abundances. A number of methods can been used to interpret these data, and these may be broadly grouped under two headings: community composition methods and community function methods.

Community composition methods

Community composition methods compare biological communities by using diversity measures, indicator taxa, or similarity measures.

Diversity measures

Diversity measures include species diversity indices, such as the Shannon Index or Simpson's Index (e.g. Magurran 1988; Pielou 1975), or various other measures of species richness or evenness, including general indices, such as those proposed by Hurlbert (1971), and indices specifically designed for water quality studies, such as Cairns' Sequential Comparison Index (Cairns et al. 1968) and Average Species Richness proposed by Campbell et al. (1982). The general assumption is that high levels of diversity are desirable and equate with high levels of biological integrity.

Indicator taxa

Measures such as the saprobian system (e.g. Sladecek 1973) or the various biotic indices, including the Trent Biotic Index and the Chandler Biotic Score (Hellawell 1986), use the presence or absence or the relative abundance of particular taxa believed to be either sensitive to or tolerant of poor water quality to evaluate biological integrity.

Indices of biological and community integrity (Karr 1991; Plafkin et al. 1989) use a range of metrics incorporating indicator taxa, indicators of ecosystem function (trophic or functional groups) and condition of individual organisms (frequency of deformities) in the final index value. The value of

each individual metric may be graded by comparison with values from an unimpacted site in the same region.

Similarity measures
Similarity measures use various multivariate techniques (analysis of variance using multiple species) to group sampling sites and/or times by their similarity or dissimilarity. The groups are then interpreted by reference to physical and chemical parameters, and by the key biological differences that led to the grouping structure.

Community function methods
Ecosystem function can be assessed by measuring rates of processes such as respiration, primary production and nutrient cycling, or by measures of trophic structure such as the flow of energy through or biomass of various trophic levels. Several authors have advocated measures of ecosystem function as indicators of ecosystem condition (Plafkin et al. 1989) but, with the possible exception of primary production, these have rarely been used. The main impediment has been the difficulty of making the required measurements.

The indices of biological and community integrity discussed previously attempt to incorporate some indicators of community function by including metrics on relative numbers of fish from various trophic levels or invertebrates from functional feeding groups. Section 2.2 recommends that four biological factors be used to assess the condition of ecosystem health: species richness, species composition, primary production and ecosystem function (estimated from the change in the production to respiration [P:R] ratio). These should be regarded as interim guidelines at this stage; however, their introduction should produce additional information on a wide range of ecosystem types that will assist in updating the guidelines in the future.

2.2 Biological factors

2.2.1 Species richness

In any waterbody, the species richness of the predominant macrophyte, periphytic, phytoplanktonic, benthic and planktonic invertebrate or vertebrate assemblages, as measured by an appropriate standardised index, should not be altered.

The preservation of biodiversity is a significant international concern, and the focus of a major Australian Government initiative (Biodiversity Working Party 1991). A decrease in species richness has long been considered as an indicator of ecosystem stress (Magurran 1988), and is both conceptually simpler and less unequivocally measured than species diversity (Hurlbert 1971). Since different components of an ecosystem may respond differentially to stress, it is important that all the major biological groups (plants and animals, water column dwellers and attached or benthic dwellers) be evaluated. The measurement of species richness at several unimpacted and several impacted sites is necessary to determine whether differences between the two are statistically significant.

2.2.2 Species composition

In any waterbody, impacts that result in significant changes in species composition compared with those in similar, local unimpacted systems should not be permitted.

Again, this requirement must be based on multiple samples from multiple impacted and unimpacted sites, or quadrats, to establish, first, that there is a significant change in species composition and, second, the mean extent of that change. It is possible, although probably unlikely, that ecosystems could maintain species richness while still changing markedly in species composition.

2.2.3 Primary production

In any waterbody, net primary production should not vary from the levels encountered in similar, local unimpacted habitats under similar light, temperature and nutrient loading regimes.

Primary production forms the basis of many aquatic food chains, particularly in limnetic systems where detritus from aquatic macrophytes and/or terrestrial sources is low or absent. Primary production is known to be sensitive to light, temperature and nutrients, among other factors (Section 2.3.3), and hence is likely to be significantly affected by changes in water clarity, changes in the extent of shading when stream-side vegetation is cleared, and warm-water and nutrient-rich discharges.

2.2.4 Ecosystem function

In any waterbody, changes that vary the relative importance of the detrital and grazing food chains should be minimised. Production to respiration ratios should not vary significantly from those of similar, local unimpacted systems.

Some ecosystems, such as large standing waterbodies, have autochthonous primary production as their major energy source. Others, including forest streams and some wetland systems derive most of their energy from allochthonous detritus. Aquatic systems should be managed such that the relative balance between these two major energy pathways is maintained, and that natural detritus-driven aquatic systems are not converted to autochthonous primary production driven systems and vice versa.

2.3 Physico-chemical factors

2.3.1 Clarity and colour

In fresh waters that are deeper than 0.5 z_{eu} the natural euphotic depth (z_{eu}) should not be permitted to change by more than 10%. In waters shallower than 0.5 z_{eu} the maximum reduction in light at the sediment bed should not exceed 20% to protect the light climate of benthic plants.

In marine waters, the natural euphotic depth should not be permitted to change by more than 10%.

The optical quality of water, primarily its clarity and colour, is determined by the attenuation of light, particularly by suspended particulate matter but also by dissolved matter (Kirk 1983, 1988). Light in waterbodies can be attenuated by two main processes: absorption and scattering (Kirk 1983). Davies-Colley (1991) recommends that two different aspects of water clarity (the transparency of water), and the water colour should be protected. These are described as follows:

- *Visual clarity*: This is the maximum distance at which objects can be viewed through the water. Visual clarity is best measured using a black disc (Secchi disc) viewed horizontally through the water. A black disc is the preferred target because it reflects no light, and therefore its sighting range is almost independent of sunlight conditions.

- *Light penetration*: A useful index of the penetration of diffuse sunlight into the waterbody is the 'euphotic depth' (z_{eu}), the depth at which photosynthetically available radiation (PAR) is reduced to 1% of the level at the water surface. This is measured with an appropriate light sensor such as a PAR sensor. Generally, aquatic plants cannot grow at depths greater than the euphotic depth because of light limitation. There is no simple relationship between euphotic depth and visual clarity.
- *Colour*: This term describes the light emanating from within the waterbody due to a combination of absorption and scattering processes. Typically, about 3% of the incident light will re-emerge from the waterbody as backscattered light, although this ratio can vary widely.

Protection of the optical properties of aquatic ecosystem waters is required for three reasons. First, a number of predatory fish (and sight-predatory birds) rely upon the clarity of the water to be able to see their prey. Davies-Colley (1991) reviewed the scientific literature on the importance of the visual clarity of waters in habitat for sight-predators and concluded:

> …little scientific work seems to have been carried out …and no criteria appear to be available on which to base guidelines. This is a major gap in the literature and one deserving urgent research.

Second, a reduction in the light penetration into a waterbody will result in a reduction in photosynthesis and hence primary production, with possible deleterious effects on phytoplankton, macrophytes and benthic plants. For example, in a study of macrophytes in a range of New Zealand lakes, Vant et al. (1987) found the depth limit of macrophytes was closely correlated with (and approximately numerically equal to) the euphotic depth. Finally, the colour of water may also affect aquatic ecosystems by influencing the spectral distribution of underwater light available for photosynthesis and illumination.

In addition to influencing the optical properties of a waterbody, suspended particulate and colloidal matter may directly affect aquatic ecosystems by degrading benthic habitats, smothering benthic organisms or exerting an oxygen demand. These direct effects are covered more fully in Section 2.3.6.

Many water quality criteria documents recommend the use of compensation depth (the depth below which plants cannot grow because their gross production is balanced by their respiration) as the indicator for the protection of primary production in aquatic ecosystems (e.g. Hart 1974; USEPA 1986). For example, USEPA (1986) restricts the change in the compensation depth to less than 10% of the seasonally established norm. However, a major disadvantage of this indicator is the difficulty in measuring the 'normal' compensation depth in variable systems.

Davies-Colley (1991) has recommended using euphotic depth as the indicator to protect primary production in aquatic ecosystems. Euphotic depth is usually similar to the compensation depth, but is considerably more easily measured. The euphotic depth is inversely related to the average diffuse attenuation coefficient (K_{av}) for downwelling light—$z_{eu} = 4.6/K_{av}$ (Kirk 1983).

Davies-Colley (1991) recommends that changes to the euphotic depth should be restricted to less than 10% in waters deeper than 0.5 z_{eu}. This would result in a maximum 10% reduction in the depth range of benthic plants. A 10% reduction in euphotic depth corresponds to a substantial 40% reduction in PAR at the euphotic depth. In shallow waterbodies, such as rivers, guidelines expressed in terms of a relative change in the euphotic depth are inappropriate, since the (virtual) euphotic depth may be greater than the water depth. For waters shallower than 0.5 z_{eu}, Davies-Colley (1991)

recommends that the maximum permissible reduction in PAR should be limited to 20% at the greatest depth to provide adequate protection of the light climate of benthic plants. In small streams in New Zealand, where discharge of clays from alluvial gold mines caused reduction of between 12% and 73% (mean 44%) in the stream-bed lighting, Davies-Colley et al. (1992) found benthic algal production was reduced almost proportionally to the light reduction.

2.3.2 Dissolved oxygen

Dissolved oxygen should not normally be permitted to fall below 6 mg/L or 80–90% saturation, this being determined over at least one diurnal cycle. Wherever possible, dissolved oxygen should be measured over the full diurnal cycle for a period of a few days to establish the diurnal range in concentration.

Dissolved oxygen levels are most often reduced in aquatic ecosystems directly by the addition of organic material or indirectly through the addition of plant nutrients. The addition of organic material stimulates the activity of aerobic heterotrophs, primarily bacteria, which utilise dissolved oxygen from the water as they mineralise the organic material. The addition of plant nutrients may influence dissolved oxygen, either through the utilisation of oxygen by the plants at night, when they cease to photosynthesise, or through the decomposition of the plant material that has grown due to the nutrients.

Dissolved oxygen is possibly the most abused water quality parameter. The concentration varies with water temperature and salinity and is greatly influenced by biological activity. Photosynthetic activity produces oxygen during periods of sufficient light, while respiration by aquatic organisms removes oxygen from the water. Consequently, the concentrations of dissolved oxygen in a waterbody may vary widely in the course of twenty-four hours, particularly in streams where there is significant nutrient enrichment. For example, Sutherland (1981) found diurnal variations of about 10 mg/L in dissolved oxygen in the Yarrowee River, Victoria, downstream of a sewage treatment plant.

Where photosynthetic activity is significantly affecting levels of dissolved oxygen in a system, the diurnal levels may vary greatly from day to day, depending on the weather and, particularly, the amount of available light. The rate of re-aeration will also influence the concentration to which dissolved oxygen may drop as a result of the respiratory activities of the biota (including microbiological activity) with turbulent waterbodies being more rapidly aerated and thus less subject to very low concentrations.

As a consequence of the factors discussed above, spot measurements of dissolved oxygen are not particularly useful. The full diurnal range of dissolved oxygen must be known before the data can be interpreted, and preferably the diurnal range over a number of days, including both sunny and dull days. In streams with turbulent sections, dissolved oxygen should be measured in the least turbulent stretches.

Any reduction in dissolved oxygen concentrations in water reduces the physiological efficiency of fish and non-airbreathing invertebrates. While quite extensive data exista on the effects of reduced dissolved oxygen concentrations on fish elsewhere in the world (e.g. Alabaster & Lloyd 1982; USEPA 1986), there are few Australian data. Koehn and O'Connor (1990) review the data on freshwater fish found in Victoria, and suggest that dissolved oxygen concentrations below 5 mg/L are stressful to

many species. The authors of these guidelines are not aware of any published data on the effects of dissolved oxygen on Australian aquatic invertebrates.

The dissolved oxygen concentration in water is dependent upon temperature and salinity (and biological activity). For this reason, the dissolved oxygen guideline is expressed in terms of both absolute concentration and percentage of saturation so that it will be more meaningful for the variety of waters throughout Australia. Dissolved oxygen should not normally be permitted to fall below 6 mg/L or 80–90% saturation, this being determined over at least one diurnal cycle.

2.3.3 Nutrients, chlorophyll-a and/or nuisance plant growth

It is not possible to recommend a single set of nitrogen and phosphorus concentrations that will prevent nuisance algal problems in Australian fresh and marine waters because many other factors (e.g. poor light, high turbidity, temperature, high grazing rates, poor attachment substrates) can also limit the development of nuisance growths. Nuisance algal growths may also occur downstream in a river system because of some limiting factor in an upstream section. Therefore, it is strongly recommended that site-specific studies be undertaken to determine appropriate concentrations for each particular system. The concentration values or ranges for the surface waters listed in the following text are provided primarily as an indication of levels at or above which problems have been known to occur, dependent on a range of other limiting factors.

In some aquatic ecosystems, the natural levels of nitrogen and phosphorus may fall into the ranges defined in this section but cause no environmental damage to the system. In other, similar ecosystem types, the same levels of these indicators can contribute significantly to nuisance algal growth. Therefore, it is expected that, following site-specific studies, guideline values for nitrogen and phosphorus may be determined for specific waterbodies that are higher or lower than the values or ranges indicated in the text.

General

Plant growth (via photosynthesis) is primarily dependent on sunlight and certain inorganic nutrients, and can be summarised by the following simplified equation:

$$\text{Sunlight} + 106\ CO_2 + 16NO_3^- + HPO_4^{2-} + 122\ H_2O + 18H^+ + (\text{trace elements}) \rightarrow$$
$$\{(CH_2O)_{106}(NH_3)_{16}(H_3PO_4)\}\ (\text{plant biomass})$$

Light input or the supply of biologically available nitrogen and/or phosphorus usually limits biomass production. The most biologically available (bioavailable) form of phosphorus is orthophosphate (PO_4^{3-}) and the most bioavailable forms of nitrogen are ammonia (NH_3) and nitrate (NO_3^-).

The largest amount of these nutrients is normally derived from the catchment, via either wastewater discharges or diffuse runoff. A number of species of cyanobacteria (blue-green algae) are capable of fixing molecular nitrogen (Round 1981). The significance of diffuse sources in any given situation depends on the yield of nutrient generated by the particular land-use activity (i.e. nutrient kg/ha/a) and the area or proportion of the catchment devoted to that activity. Generally, the highest yields are from urban areas, with successively lower yields from agricultural and forested catchments (Campbell & Doeg 1989).

Since controls on the concentrations and loads of these two nutrients are often the main way in which nuisance growths of these aquatic plants are managed, further discussion is included in the sections following relating to the three main types of waterbodies.

Despite the heavy emphasis on nutrient controls to manage eutrophication, consideration needs to be given to the other factors that can influence the growth of aquatic plants (Reynolds 1984; Harris 1986; Quinn 1991). These include the following:

- Light, which is essential for plant growth. In clear waterbodies, such as many of those in North America and Europe, aquatic plant growth is unlikely to be limited by lack of light. However, under the more turbid conditions that exist in many Australian rivers, lakes and reservoirs, light penetration is reduced and may limit plant growth. Ganf (1980, 1982) found that light, and not nutrient concentrations, limits phytoplankton growth in Mount Bold Reservoir, South Australia.
- Temperature, which influences maximum rates of photosynthesis and growth, and the species composition of the phytoplankton and periphyton populations in particular. Different species have different optimum temperatures. As a general rule, maximum growth rates of algae (under light-saturated conditions) approximately double with a 10˚C temperature increase (Quinn 1991).
- Current velocity, which is particularly relevant for periphyton and macrophytes. Moderate current velocities can increase periphyton biomass (when growth is nutrient limited) by increasing the nutrient supply to the cell surfaces. High current velocities can result in reduced biomass by sloughing off the periphyton. Short water residence time (less than one to two weeks) in reservoirs and estuaries may remove phytoplankton before the standing crop reaches a maximum.
- Substrate stability, which is particularly relevant for periphyton and macrophytes. Macrophytes grow best in rivers and lakes in areas of fine sediment and low current velocity (Wright & McDonnell 1986).
- Grazing. Zooplankton grazing can influence both the biomass and species composition of phytoplankton in lakes and reservoirs, and grazing by macro-invertebrates can limit periphyton biomass in flowing waters. Filtering shellfish may limit phytoplankton in some estuaries. Macrophytes are grazed by invertebrate shredders, crayfish and waterfowl.

Rivers and streams

Periphyton and macrophytes commonly cause problems in nutrient-enriched shallow rivers and streams, while phytoplankton are normally the cause of problems in larger (and particularly slow-flowing) rivers. All can result in diurnal fluctuation in pH and dissolved oxygen (DO) that can stress or eliminate sensitive species. In addition, periphyton can form dense mats that cover the stream bed and reduce habitat quality for fish and invertebrates, macrophytes can reduce light penetration and provide a physical barrier to fish migration, and phytoplankton (particularly cyanobacteria) can cause taste, odour and toxicity problems.

A wide range of nutrient concentrations have been reported for Australian rivers and streams. For example, total-P concentrations can vary from less than 10 μg/L in small, near pristine mountain streams, such as Myrtle Creek (Hart et al. 1992), to over 1,000 μg/L in heavily polluted rivers. Equally, total-N can vary from as low as 100–200 μg/L to in excess of 10,000 μg/L in heavily polluted rivers. In many turbid Australian rivers, a significant proportion of the total-P and total-N concentrations, particularly that bound to suspended particulate matter or complexed with organic molecules, may not be available for plant growth. Therefore, the use of total-P and total-N concentrations to limit

nutrient concentrations in flowing waters represents a conservative approach to the management of nutrients in flowing waters.

In many systems, potential nuisance aquatic plants are less of a problem in a river than in a downstream lake, reservoir or estuary (Henley et al. 1980). Information on the total load of nutrients transported by the river is required to assess the possible downstream problems. However, obtaining this information is not simple, since nutrient concentrations in rivers and streams can vary markedly with flow, maximum loads being transported during flood events (Cullen et al. 1978; Cosser 1979). For example, during an eighteen-month study of Lake Burley Griffin, Cullen et al. (1978) found that floods that occurred during 9% of the study time transported 69% of the phosphorus entering the lake. Two main sources contribute nutrients during flood events:

- increased amounts of nutrients (dissolved and particulate) are washed into the river or stream from the land;
- increased erosion of stream beds and banks occurs at higher discharge, contributing increased particulate material and associated nutrients.

High-frequency sampling over flood events is essential if the annual total load of nutrients transported by a river is to be reliably estimated.

On the basis of limited data, Quinn (1991) suggested that dissolved reactive phosphorus (DRP) concentrations need to be below about 15–30 μg/L, and dissolved inorganic nitrogen (DIN = NO_3-N + NH_3-N) concentrations below about 40–100 μg/L, for nutrients to exert any significant control on periphyton biomass in New Zealand streams. There has been insufficient study of nutrient limitations on periphyton growth in Australian rivers to assess how relevant these New Zealand data are to Australian conditions. However, limited information from Chessman and Hutton (1989) suggests that these ranges can probably be used for Australian rivers and streams. Models relating nutrient concentrations and periphyton biomass are another tool available to river managers. A potentially useful model for predicting periphyton growth in phosphorus-limited streams has been published by Welsh et al. (1989, 1992).

Cyanobacterial blooms have also occurred in a number of slow-moving rivers (e.g. the Murray and the Darling) in recent years. It has been hypothesised that, despite the fact that nutrient concentrations appear to be sufficiently high to support considerable phytoplankton biomass, under most circumstances growth in these systems is controlled by light limitation. However, when river flows are low (or non-existent), turbidity is reduced and light penetration is increased, thus providing conditions that result in greater phytoplankton growth. Such rivers resemble linear lakes when phytoplankton blooms occur in them, and the reader is referred to the next section for a detailed discussion on the relationships between nutrients and algal growth in lakes and reservoirs.

The blanket imposition of national nutrient limits to prevent undesirable periphyton and phytoplankton growths in rivers and streams is not justified, since some systems cope naturally with relatively high nutrient loads without excessive aquatic plant growth occurring because other factors (e.g. poor light, high turbidity, high grazing rates, poor attachment substrates) limit the development of nuisance growths. Rather, it is strongly recommended that site-specific studies be undertaken to determine the potential for undesirable aquatic plant growths occurring in each particular system. The following nutrient values or concentration ranges are provided as an indication of levels at or

above which problems have been known to occur, depending upon a range of other factors. It is expected that, following site-specific studies, guideline values for nitrogen and phosphorus may be determined for specific waterbodies that are higher or lower than the values or ranges indicated in the text. The indicative concentration values or ranges are:

- Total-P 10–100 μg/L
- Total-N 100–750 μg/L.

Lakes and reservoirs

Excessive growths of phytoplankton (algal blooms) cause most problems in nutrient-enriched lakes and reservoirs. Ecologically, the main problems relate to excessive diurnal fluctuation in pH and dissolved oxygen that can stress or eliminate sensitive species, depletion of oxygen in bottom waters resulting in death of fish and release of phosphorus, and reduced light penetration leading to macrophyte decline. In addition, certain algal groups can cause problems with tastes and odours (e.g. diatoms, cyanobacteria), toxins (e.g. cyanobacteria, dinoflagellates) and filtration (e.g. diatoms). In northern Australia, aquatic plants such as water hyacinth (*Eichhornia crassipes*) and *Salvinia molesta* cause many problems (Mitchell 1986).

Cyanobacteria (blue-green algae) are the main bloom-causing algal group in fresh waters. During a cyanobacterial bloom there is generally a massive increase in cell numbers (to greater than 10^6 cell/mL), with most of the biomass comprising one or, at most, two species forming a surface scum. Paerl (1988) has summarised the evidence suggesting that the formation of surface blooms by cyanobacteria represents a highly adaptive ecological and physiological strategy aimed at minimising light (PAR) and CO_2 limitations, and perhaps also 'shading' subsurface phytoplankton species, thus restricting their growth. Many, but not all, cyanobacteria (e.g. *Anabaena*, *Nodularia*) are able to fix molecular nitrogen; that is, convert nitrogen into organic matter such as proteins.

The following physical, chemical and biological conditions appear to favour the formation of cyanobacterial blooms: confined waterbody; vertically stratified water mass (non-turbulent); warm weather; high incident light; enhanced allochthonous inorganic nutrient (nitrogen and/or phosphorus) loadings; adequate bio-available trace metals; sediments physically and nutritionally suitable as 'seed beds' for storing and supplying spores or akinetes; close algal-bacterial interactions leading to enhanced nutrient cycling; close interaction between algae and micrograzers (rotifers, protists), which leads to enhanced nutrient availability among filamentous and colonial nuisance algae; and the absence or reduced activity of macrograzers such as crustaceans, zooplankton and fish (Paerl 1988).

Physically stable, fresh and brackish waterbodies receiving excessive amounts of both nitrogen and phosphorus are often found to have blooms of *Microcystis* and *Oscillatoria*. Those receiving relatively large phosphorus inputs have blooms of *Anabaena*, *Aphanizomenon* and *Gloeotrichia*, the minor species in these cases being *Microcystis and Oscillatoria* (Paerl 1988). Suttle and Harrison (1988) have provided evidence showing that nitrogen-fixing cyanobacteria are not particularly good competitors for phosphorus, and are actually quite poor at obtaining phosphorus at low concentrations. Therefore, although nitrogen may be limiting in a particular lake, nitrogen-fixers may remain essentially absent because of their inability to obtain an adequate supply of phosphorus.

There is considerable uncertainty in what constitutes an 'undesirable biological growth' of phytoplankton, since the background levels of phytoplankton abundance can vary widely and there are a number of ways in which excessive growth can degrade the environmental values of a waterbody. A number of indicators have been used to define bloom conditions, including cell numbers of particular algal groups, chlorophyll-a concentration and primary productivity (OECD 1982). The levels of these indicators judged to indicate a bloom will differ considerably depending on the type of waterbody. In an oligotrophic lake, for example, a bloom might be considered to occur if the Secchi depth decreased from 10 m to 5 m; while in a turbid eutrophic lake, a bloom condition might be considered to occur only when there was a perceptible surface scum of algae.

The classifications provided in Table 2.2 to assist Australian reservoir and lake managers are based on chlorophyll-a concentrations and are adapted from work reported by Quinn (1991) for New Zealand lakes. It is recommended that limitations on chlorophyll-a be developed for each lake and reservoir, based on site-specific information and the environmental values to be protected. AEC (1987) recommends that the chlorophyll-a concentrations should be kept below 5 μg/L in reservoirs used for drinking waters, and below 20 μg/L for recreational lakes.

Table 2.2 Annual mean and maximum chlorophyll-a concentrations for reservoirs and lakes

Annual mean (μg/L)	Annual maximum (μg/L)	Lake conditions
< 2	< 5	Oligotrophic, aesthetically pleasing, very low phytoplankton levels
2–5	5–15	Some algal turbidity, reduced aesthetic appeal, some oxygen depletion
5–15	15–40	Obvious algal turbidity, reduced aesthetic appeal, oxygen depletion
> 15	> 40	Eutrophic, high levels of phytoplankton growth, significantly reduced aesthetic appeal, serious oxygen depletion in bottom waters, reduction in other uses

Common management practice is to limit the concentrations and loads of nitrogen and phosphorus entering a waterbody on the assumption that there is a direct causal relationship between these nutrients and phytoplankton biomass. There is, however, considerable confusion as to whether nutrients limit the rate of growth or the total amount (biomass) of phytoplankton or both. For example, it is often stated that phosphorus limits phytoplankton growth in lakes because the concentration of dissolved inorganic phosphorus (DIP) is very low. This wrongly assumes that there is a simple relationship between nutrient concentration and the rate of algal growth, an idea that grew out of batch culture work where the initial nutrient concentration represented the entire available nutrient supply.

In real systems, where nutrients can be recycled, it is necessary to know both the rate of uptake from the soluble pool and the rate of regeneration of the nutrient from the algal pool (and also from the sediment pool if this is rapid enough). The particular concentration of dissolved inorganic nitrogen (DIN) or DIP in the water column reflects the net effect of the rate at which nitrogen or phosphorus is taken up by the phytoplankton and the rate at which it is regenerated. A very low nutrient concentration could indicate that a particular nutrient is essentially depleted from the water column and is therefore limiting phytoplankton growth, but equally could simply be the net result of a very

rapid uptake and release of the nutrient. If the uptake and regeneration of a nutrient are not balanced (and there is not another input source), the soluble pool may become depleted in that nutrient.

The rates of turnover of DIP and DIN are generally found to be very much faster in oligotrophic waters than in eutrophic waters, although eutrophic waters generally have much larger soluble pools sizes than oligotrophic waters (Harris 1986). Oligotrophic waters appear to be more dependent upon internal recycling of nutrients than eutrophic waters.

The actual biomass of phytoplankton present at a particular time is difficult to predict (Lane & Levins 1977; Harris 1986), since it will be controlled by a number of biological factors (grazing, death, microbial regeneration of nutrients), in addition to the light climate, temperature and the nutrient availability discussed above. Hence, there is no particular reason why the phytoplankton biomass (often estimated by chlorophyll-a concentration) should be directly related to the dissolved or total nutrient concentration in a particular aquatic system.

The success of the Vollenweider approach (Vollenweider 1968, 1975, 1976) to predicting annual algal biomass (as chlorophyll-a) from the annual total-P and total-N loadings suggests that in those systems for which this approach works (deep, relatively clear lakes and reservoirs), the small-scale, rapid processes of DIN and DIP turnover, grazing, growth and death average out over the longer term. (The time scales of many of these processes can be quite different; for example, turnover of DIN and DIP pools may be measured in minutes, algal growth processes occur over periods of days or weeks, and loading rates of total-N and total-P may be seasonal.) A number of modifications to the original Vollenweider approach (close coupling between biomass and total-P supply) have been produced over the years to make the concept fit other types of waterbodies, particularly the turbid Australian and South African waters (Pridmore 1987; White 1989; Stauffer 1991).

Natural phytoplankton communities are made up of populations of different species, and it is simplistic to assume that all the species in a community will be limited (i.e. prevented from increasing in numbers) by a single factor. Hecky and Kilham (1988) have reviewed evidence showing that, because of their different life histories, the different phytoplankton species making up the natural assemblage may be limited by different factors (e.g. nutrients, light, temperature, grazing, turbulence) operating at the same time. Ultimately, these species will persist in the waterbody only if their cellular growth rate is greater than losses due to sedimentation, grazing or other causes of death.

Smalls and Cannon (1983) identified a total-P of 10 μg/L as the critical concentration above which algal problems occurred for Prospect Reservoir in New South Wales. This was also the concentration nominated by Vollenweider (1976) as distinguishing oligotrophic from mesotrophic waters for phosphorus-limited lakes and reservoirs. In other still, inland waters, the critical levels of phosphorus have been identified as somewhat higher, often around 20 μg/L (AEC 1987). Considerably higher concentrations of phosphorus occur in Mount Bold Reservoir, South Australia, without resulting algal problems, probably because this reservoir is relatively turbid and therefore light-limited (Ganf 1980, 1982).

In recent years, management agencies have placed increasing reliance on the nitrogen–phosphorus ratio as a management tool to control nuisance algal blooms—cyanobacteria in fresh waters and

dinoflagellates in marine waters (Rhee 1978). The hypothesis that the nitrogen–phosphorus (N:P) ratio will control the type of phytoplankton community is based on the following:

- The observation that the cellular composition of phytoplankton is relatively constant and that the two major nutrients (nitrogen and phosphorus) are in the atomic ratio of 16–20:1 for optimum algal growth.
- If the N:P ratio is greater than about 20:1, the system is limited by phosphorus, and if less than about 16:1, it is limited by nitrogen.
- In nitrogen-limited systems, phytoplankton species that can obtain nitrogen from another source (Paerl 1988) will have a competitive advantage (e.g. some species of cyanobacteria can fix atmospheric nitrogen).

The range in the elemental composition of algae is rather narrow, presumably because all algal cells have to perform similar metabolic functions. The average cellular composition of phytoplankton is often taken as C:N:P = 106:16:1 (the 'Redfield Ratio', Redfield 1958), although more recently Harris (1986) has suggested that a ratio of 166:20:1 is probably more representative of the average phytoplankton composition.

There are now an increasing number of examples where the use of N:P ratios is either inappropriate or the limitations have not been sufficiently considered. The first limitation is the assumption that the cellular composition of phytoplankton is relatively constant and that the cellular N:P ratio is about 16–20:1 for optimum growth. However, a review of the literature made by Hecky and Kilham (1988) showed that the optimum N:P ratios for fourteen freshwater and marine phytoplankton for which data were available ranged from seven to eighty-seven.

The second limitation is the considerable uncertainty over just what measures of nitrogen and phosphorus should be used in calculating the N:P ratio. Most commonly, the ratio of the total-N and total-P concentrations (TN:TP) is used, but ratios of DIN:DIP and DIN:DRP have also been used. These are all surrogates for what is really required, namely the ratio of algal available nitrogen to algal available phosphorus (aN:aP). The ratio TN:TP will generally overestimate aN:aP because organic phosphorus is often more readily available than organic nitrogen (Jannson et al. 1988), and DIN:DRP will underestimate aN:aP because DRP will include some organic phosphorus compounds in addition to orthophosphate (Tarapchak et al. 1982). In turbid Australian systems, estimation of the proportion of the total-P and total-N that are available to algae, and under what conditions, is even more difficult to assess.

A third limitation is that phytoplankton community structure is most sensitive to the N:P ratio in systems where phytoplankton growth is controlled exclusively by nutrient supply and both nitrogen and phosphorus can limit different species simultaneously (Suttle & Harrison 1988). For this to occur, the *in situ* N:P ratio must be in the range of the critical N:P ratios for phytoplankton (i.e. about 7:1 to 45:1).

It is not possible to recommend a single set of nitrogen and phosphorus concentrations that will prevent phytoplankton problems in lakes and reservoirs throughout Australia. Rather, it is strongly recommended that site-specific studies be undertaken to determine the appropriate concentrations for each particular system. The following nutrient values or concentration ranges are provided as an indication of levels at or above which problems have been known to occur, depending upon a range

of other factors. It is expected that, following site-specific studies, guideline values for nitrogen and phosphorus may be determined for specific waterbodies that are higher or lower than the values or ranges indicated in the text. The indicative concentration values or ranges are:

- Total-P: 5–50 μg/L
- Total-N: 100–500 μg/L
- Chlorophyll-a: 2–10 μg/L.

Estuaries and coastal waters

Algal problems in estuarine and coastal regions generally occur in the upper and lower estuarine areas and in confined embayments and coastal lakes. There have been reports of cyanobacterial problems in the upper, freshwater regions of several Australian estuaries; for example, a microcystis bloom in the upper Hawkesbury estuary (AEC 1987). Algal blooms are rare in the mid-estuarine regions of many estuaries because of light limitation due to the relatively high turbidity (Moss 1987; Moss & Bennett 1992).

Algal problems have also been reported in a number of coastal lakes or embayments around Australia, for example *Nodularia* in the Gippsland Lakes, Victoria, and in the Harvey Estuary, Western Australia; dinoflagellate blooms (red tides) in Port Phillip Bay, Victoria, and Morton Bay, Queensland; seagrasses in Port Phillip Bay, Victoria; and *Cladophora* in the Peel Inlet, Western Australia (Moss 1987; Lukatelich & McComb 1986; AEC 1987).

There are few published surveys of nutrient concentrations in estuarine and coastal regions of Australia. A survey of nutrient concentrations in Cockburn Sound, Western Australia, during the summer of 1989–90 found mean PO_4-P concentrations in the range 1–7 μg/L, mean NO_3-N concentrations in the range 5–11 μg/L, and mean NH_3-N concentrations in the range 2–24 μg/L (Cary et al. 1991). The high values are probably influenced by an industrial effluent discharged to the Sound. Moss (1987) lists data for a number of Queensland estuaries, and some information is available on the nutrient concentrations in marine waters off the coast of Australia. Considerable spatial variation in the nutrient concentrations in Australian marine waters has been reported; PO_4-P concentrations in surface waters range from 1 μg/L to 16 μg/L, NO_3-N concentrations from 16 μg/L to 56 μg/L, and NH_3-N concentrations less than 3 μg/L (Rochford 1980, 1984; Gibbs et al. 1986, 1991; Sorokin 1990). Chlorophyll-a concentrations have been reported in the range 0.1 μg/L to greater than 1.0 μg/L, with the highest values generally found closest to land (Gabric et al. 1990; Gibbs et al. 1991).

As was the case for lakes and reservoirs, there is still no wide-spread agreement on the best indicator to use in assessing what constitutes a 'nuisance growth' of algae in estuarine or coastal waters. Moss (1987) recommended limits on the chlorophyll-a concentrations in Queensland estuaries to control phytoplankton blooms; the suggested limit was a ninety percentile of less than 15 μg/L chlorophyll-a. Because of a lack of detailed information, limitations on other problem algae were specified in qualitative terms; for example, dinoflagellate populations should not reach levels where they give rise to damaging red tides, floating macrophytes should be prevented from forming mats covering extensive proportions of the water surface, macroalgal biomass should be prevented from building up to nuisance proportions.

It is not possible to recommend a single set of nitrogen and phosphorus concentrations that will prevent phytoplankton problems in estuaries and coastal waters throughout Australia. Rather, it is strongly recommended that site-specific studies be undertaken to determine appropriate concentrations for each particular system. The concentration values or ranges listed below are provided as an indication of levels at or above which problems have been known to occur, depending upon a range of other factors. It is expected that, following site-specific studies, guideline values for nitrogen and phosphorus may be determined for specific waterbodies that are higher or lower than the values or ranges indicated in the text. The indicative concentration values or ranges are:

- PO_4-P: 5–15 μg/L (estuaries & embayments); 1–10 μg/L (coastal waters)
- NO_3-N: 10–100 μg/L (estuaries & embayments); 10–60 μg/L (coastal waters)
- NH_4-N: < 5 μg/L (estuaries & embayments); < 5 μg/L (coastal waters)
- Chlorophyll-a: 1–10 μg/L (estuaries & embayments); < 1 μg/L (coastal waters).

2.3.4 pH

The pH of fresh waters should not be permitted to vary beyond the range 6.5 to 9.0. It is also recommended that changes of more than 0.5 pH units from the natural seasonal maximum or minimum should be investigated.

In marine waters, the pH should not be permitted to vary by more than 0.2 units from the normal values.

Most natural fresh waters have a pH close to 7.0, and marine waters close to 8.2. In many waters the pH is controlled by the carbonate–bicarbonate buffer system. The buffer capacity of a waterbody is related to the alkalinity and has a major influence on the pH changes that may occur in a particular waterbody due to the discharge of acidic or alkaline wastewaters. Marine waters are strongly buffered, and even a small pH change in seawater indicates a major change to the system. Total alkalinity of seawater is 115–120 mg (as $CaCO_3$)/L (Morel 1983).

There are a number of reviews of the effects of pH changes on freshwater aquatic biota (Fromm 1980; Haines 1981; Alabaster & Lloyd 1982). It is generally accepted that pH 5–9 is the range that is not acutely lethal to fish; however, the toxicity of several pollutants (e.g. ammonia, cyanide) can be markedly affected by pH changes within this range. Chronic effects have been reported below pH 5, with harmful effects on the eggs and fry of sensitive fish. Loss of fish populations have been attributed to spawning failure and diminished hatching success at moderate (less than 6.0) pH levels (Fromm 1980; CCREM 1991).

In acidified streams, macroinvertebrate communities often have lower densities, fewer species and altered taxonomic composition compared with nearby sites where stream water pH is higher (Simpson et al. 1985; Collier & Winterbourn 1987). A number of possible reasons for these observations have been advanced, including the direct physiological effect of the low pH and indirect effects of pH (e.g. toxicity due to aluminium released by the low pH, and changes in food supply caused by the pH). In a study of acidic (pH 4.3–5.7) and alkaline (pH 6.6–8.0) streams in New Zealand, Collier and Winterbourn (1987) found higher numbers of invertebrate taxa (sixty-four compared with forty-three) and greater mean densities of benthic invertebrates (up to five times higher) in the alkaline streams. Despite the elevated concentrations of aluminium in the acidic streams, Collier and

Winterbourn (1987) suggested that changes in the food supply (e.g. epilithon) was responsible for the depauperate fauna in the acidic streams.

The commonly accepted lower limit for pH (6.5) is based on bioassay experiments that showed a decrease in egg production and hatchability of the fathead minnow and in emergence of aquatic insects at pH values below 6.6 (Mount 1973; CCREM 1991). Alabaster and Lloyd (1982) have summarised the evidence used to establish the commonly accepted upper limit for pH of 9.0. A number of guidelines or criteria also suggest controls on the maximum changes in alkalinity due to waste discharges. For example, USEPA (1973) recommends that the total alkalinity should not be permitted to decrease by more than 25% below the natural level.

Less is known of the direct physiological effects of pH changes on marine organisms. The range of pH values in marine waters is considerably less than in most fresh waters, typically being 8.0 to 8.3. This range can be extended in biologically active coastal waters. NAS/NAE (1973) have reported that marine plankton and benthic invertebrates are more sensitive to changes in pH than marine fish.

2.3.5 Salinity

In fresh waters, the salinity (conductivity) should not be permitted to increase above 1,000 mg/L (about 1,500 μS/cm). This concentration may need to be reduced depending upon other uses of the water.

In estuarine and coastal waters, salinity changes should be less than 5% from background levels.

There is considerable confusion over the reporting of total dissolved substances in waters, with terms such as total dissolved salts, total dissolved solids, salinity, conductivity and filterable residue all being used. Hart (1974) considers these terms to be rather non-specific indicators of water quality and has questioned their use. However, because the broad characterisation of the amount of dissolved mineral material is still widely used to characterise drinking and agricultural waters in particular, filterable residue (APHA 1990) and conductivity have also been included in this document. It is assumed that:

$$\text{filterable residue (mg/L)} = 0.68 \times \text{conductivity (}\mu\text{S/cm)}$$

Consideration of the possible adverse effects of salinity has assumed a greater importance in Australia over recent years because of the increases in the salinity of many waterbodies due to the widespread dry-land and irrigation salinity problems, and proposals that saline wastewaters (mostly from remediation works associated with irrigation areas) be discharged to rivers and wetlands.

Aquatic organisms are classified as stenohaline (able to adapt to only a narrow range of salinities) or euryhaline (able to adapt to a wide salinity range—up to 10,000–15,000 mg/L), with most organisms being stenohaline. Broadly, salinity changes may affect aquatic organisms either directly, through changes in total osmolality, the relative proportions of solutes and the solubility of dissolved gases, or indirectly, by modifying the species composition of the ecosystem.

Hart et al. (1990, 1991) have reviewed the biological effects of saline discharges, with special reference to Australia. Their review covered the lethal and sub-lethal effects of salinity on microbes (mainly bacteria), macrophytes and micro-algae, riparian vegetation, invertebrates, fish, amphibians, reptiles, mammals and birds. The review highlighted the dearth of information on the sensitivity of Australian freshwater organisms to increases in salinity, and found few studies on sub-lethal or long-

term effects, or on possibly more-sensitive life stages. Hart et al. (1990, 1991) concluded that direct adverse biological effects would be expected in Australian river, stream and wetland ecosystems if salinity were allowed to increase to around 1,000 mg/L. Considerations other than the possible adverse effects on the receiving water biota may limit the salinity in wastewater discharges to significantly less than 1,000 mg/L; for example, an increase in salinity to this level would significantly affect the use of water for drinking purposes, and may also have an adverse effect on its use for irrigation.

The Victorian EPA requires variations in salinity (as total dissolved solids) in estuarine and coastal waters to be less than 5% from background (Vic. Govt 1988).

2.3.6 Suspended particulate matter and turbidity

Increases in SPM should be limited such that the optical guidelines (Section 2.3.1) are maintained and that the seasonal mean nephelometric turbidity does not change by more than 10%.

SPM or suspended solids may arise from point sources such as industrial and sewage discharges (especially effluents from pottery and brick-making plants), but by far the largest contribution in Australia comes from diffuse sources such as soil and stream bank erosion. Sedimentation from these sources may be the greatest single human impact in Australian streams. Most sedimentation and suspended solids in marine systems is contributed via rivers and streams, and most of this material is deposited in estuaries or coastal waters, in some cases with offshore impact, particularly on reef ecosystems.

The levels of SPM encountered in all types of waters may vary dramatically with time. In rivers, the concentrations increase with discharge on the rising limb of the hydrograph as sediment is washed into the river due to rainfall and deposited sediment is resuspended. In still waters, the levels of SPM vary with turbulence as deposited sediment is resuspended, and also with wind. Estuaries are influenced by the loads transported by the inflowing rivers. Because of this temporal variation, investigations of SPM concentrations in waterbodies must be carried out with frequent sampling during those periods when peak concentrations may be expected in the system; single 'spot' values are uninformative.

Australian inland waters are sometimes described as being naturally highly turbid, with high levels of SPM. This may be be true for systems such as shallow lakes in which the sediments are easily resuspended due to wind-driven turbulence; however, it is not possible to draw any firm conclusions for Australian rivers. The early explorers did not comment on the turbidity of the rivers they encountered (although Sturt did find the Darling too salty for human or stock consumption), and by the time explorers such as Sturt reached the Murray, the headwaters of the Murray–Darling system were already subject to extensive clearing, which may have produced elevated levels of SPM. Given the large variations in SPM concentrations and turbidity in Australian waters, it seems appropriate to establish guidelines in terms of acceptable changes to existing levels; however, in particular degraded systems it may be more appropriate to aim for a reduction in the present levels.

Ryan (1991) has recently reviewed the impact of sediment on New Zealand streams, and Campbell and Doeg (1989) reviewed much of the Australian literature, particularly with reference to the effects of forestry operations. SPM may influence aquatic ecosystems both when in suspension and as it settles out. In suspension, the main impact of particulate matter is optical, as it can can reduce light

penetration and thus affect primary production (Kirk 1985; Davies-Colley 1991). SPM can also interfere with the feeding mechanisms of filter-feeding organisms. When the particulate matter settles out, it may smother sessile organisms and/or change the nature of the substrate, filling the interstices between stones and converting solid substrata to silty substrata. These impacts are most pronounced in waters with very low natural concentrations of SPM, where increases of only 5 mg/L may measurably reduce photosynthesis (Ryan 1991). Doeg and Milledge (1991) and Quinn et al. (1992) have recently reported on the effects of SPM on stream benthic macroinvertebrates. The levels of SPM at which acutely lethal effects occur to fish are very much higher than those at which aquatic ecosystems are seriously degraded.

It should be recognised that SPM measurement is a gross oversimplification because the character of the SPM (e.g. size distribution, particle shapes, fall velocity, optical character, specific surface area, organic content) is almost as equally important as the mass concentration as regards environmental effects.

Previous United States and European water quality criteria recommended a permissible level of suspended solids of 25 mg/L (e.g. USEPA 1973; Alabaster & Lloyd 1982). The Canadian water quality guidelines (CCREM 1991) recommend that the change in suspended particulate matter concentration should not be permitted to exceed 10 mg/L in systems where the background concentration is equal to or less than 100 mg/L, and should not exceed 10% of background when the background concentration is greater than 100 mg/L.

In a review of the optical and non-optical effects of SPM, Davies-Colley (1991) concluded that guidelines in terms of mass concentration of SPM are not useful. When suspended, the main impact of SPM is optical, so that guidelines protecting the optical water quality should negate the need for guidelines in terms of SPM mass concentration. R.J. Davies-Colley (Water Research Centre, Hamilton, New Zealand, pers. comm., June 1992) has suggested that it may simpler to express the guideline relating to SPM in terms of a relative change (e.g. 10%) in turbidity; nephelometric turbidity is easier and cheaper to measure and would permit on-site measurements to be made by water managers. Davies-Colley and Close (1990) found that turbidity was closely correlated (inversely) with visual clarity. Non-optical impacts occur mainly when SPM settles on the sediment bed or on other surfaces, but scientific criteria on which to base guidelines for settled fine solids are not yet available.

2.3.7 Temperature

The maximum permissible increase in the natural temperature of any inland or marine waters should be either 2°C or that set by the formula relating maximum permissible temperature for long-term exposure (T_{lt}) to the temperature for optimum growth and the incipient lethal temperature, whichever is the least.

There are insufficient data to establish a guideline for acceptable reductions in temperature.

Water temperature has a substantial effect on the functioning of aquatic ecosystems and the physiology of the biota (e.g. Thorp & Gibbons 1978). Physiological processes have thermal optima, and alterations to ambient temperatures may affect the species exposed in a variety of ways. Growth and metabolism, timing and success of reproduction, mobility and migration patterns and production may all be altered by changes in ambient temperature regimes. Effects may be direct, through

changes to the metabolism (especially of poikilothermic organisms), or indirect, through influence on the solubility of oxygen in water or as a result of the release of colder bottom waters from reservoirs.

In contrast to the situation in some other countries (USEPA 1973; Alabaster & Lloyd 1982), there is relatively little information available on the thermal tolerances of Australian aquatic organisms or their responses to temperature change. The few published Australian studies primarily consist of investigations into the effect of water temperature on the egg development times of mayflies (Suter & Bishop 1990; Brittain & Campbell 1991) and stoneflies (Brittain 1991). Brittain and Campbell (1991) have suggested that the distribution of *Coloburiscoides*, a mayfly genus common in upland streams of south-eastern mainland Australia, may be influenced by low winter water temperatures, since the eggs fail to develop below 5°C, which suggests that the genus could also be adversely affected by the reduced water temperature caused by releases of cold water from reservoirs.

The key variables that should be considered before a natural thermal regime is altered are the:

- lethal tolerances of key species within the affected area and their relation to length of exposure;
- influence on the rate of primary production in the system. This is important because studies have demonstrated greater sensitivity of algal growth to temperature increase than to nutrient increase, which can lead to algal blooms;
- influence on the rates of secondary production of key species within the system. Thermal changes can lead to increased production of undesirable species and decreases in the production of desirable species;
- tolerances of the various life stages of the species that occur within the affected area. Not all life stages of a given species are equally sensitive, and reproductive stages are often the most sensitive to thermal disruption;
- likely impact on species richness and natural community composition in the affected area.

The two activities that may cause substantial changes in water temperatures are the discharge of heated effluents or cooling waters and, in streams, the discharge of cold water from reservoirs. The former has received some attention in Australia, particularly the discharge of cooling waters from power stations, but the impacts of the latter have been largely ignored. The lack of data on thermal responses of the Australian aquatic biota is of great concern, since it effectively precludes meaningful predictions of the effects of thermal alterations on Australian aquatic ecosystems. USEPA (1973, 1986) has adopted a formula to determine the upper thermal limits for heated effluent discharges based on known thermal optima, as follows:

$$T_{lt} = T_{og} + \frac{T_l - T_{og}}{3}$$

where: T_{lt} = maximum permissible temperature for long-term exposure

T_{og} = temperature for optimum growth

T_l = incipient lethal temperature.

USEPA (1973, 1986) recommends that at least nine species (three fish, three invertebrates and three plants) are used to determine T_{lt}, with the value determined for the most sensitive species being adopted. As an interim guideline, Hart (1974) recommended that to ensure adequate protection of aquatic life, artificial heat sources should not be permitted to increase the ambient water

temperature by more than 2°C. There are insufficient data to recommend a guideline relating to reductions of temperature.

2.4 Toxicants

The guidelines in this section relate to median concentrations of toxicants in unfiltered samples. They should be taken as guidance values only and, where possible, should be modified to reflect the actual conditions existing in the specific ecosystem. For example, if the waters contain high concentrations of SPM, the use of unfiltered samples could lead to a possible overestimation of the bioavailable concentration of a toxicant, and the concentrations of total toxicant may need to be modified to reflect this. Equally, waters with high concentrations of hardness or dissolved organic matter will have a greater capacity to complex many heavy metals and to form less toxic forms. Detailed site-specific information will be needed to establish a statistical compliance level (e.g. 80 percentile, 95 percentile) if required.

In addition, it is known that certain mixtures of heavy metals can have a toxicity greater than the added individual toxicities (synergism), and other combinations a reduced toxicity (antagonism). The present guidelines do not consider the possibility of these effects. If all toxicants were present at close to their guideline values, significant combined effects could be expected (Enserink et al. 1991). Where possible, waste dischargers should be required to undertake toxicity testing of their effluents.

2.4.1 Inorganic indicators

Aluminium

The total aluminium concentration in fresh waters should not exceed 5 µg/L if the pH is 6.5 or less, and should not exceed 100 µg/L if the pH is greater than 6.5.

No guideline is recommended for marine waters because of a lack of information.

The bioavailability and toxicity of aluminium is generally greatest in acid solutions (Campbell & Stokes 1985). Maximum toxicity has been found to occur around pH 5.0–5.2 (Schofield & Trojnar 1980). Aluminium in acid habitats has been observed to be toxic to fish (Dillon et al. 1984), amphibians (Andren et al. 1988) and phytoplankton (Folsom et al. 1986; Claesson & Tornqvist 1988). Aluminium is generally more toxic over the pH range 4.4–5.4, with a maximum toxicity occurring around pH 5.0–5.2. The inorganic single unit aluminium species ($AlOH^+$) is thought to be the most toxic (Driscoll et al. 1980).

Under very acid conditions, the toxic effects of the high H^+ concentration appear to be more important than the effects of low concentrations of aluminium; at approximately neutral pH values, the toxicity of aluminium is greatly reduced (CCREM 1991). The solubility of aluminium is also enhanced under alkaline conditions, due to its amphoteric character, and some researchers found that the acute toxicity of aluminium increased from pH 7 to pH 9 (Freeman & Evert 1971; Hunter et al. 1980). However, the opposite relationship was found in other studies (Boyd 1979). Complexing agents such as fluoride, citrate and humic substances reduce the availability of aluminium to organisms, resulting in lower toxicity. Among freshwater aquatic plants, single-celled plants are generally found to be the most sensitive to aluminium (USEPA 1988a).

In a study of acidic (pH 4.3–5.7) and alkaline (pH 6.6–8.0) streams in New Zealand, Collier and Winterbourn (1987) found higher numbers of invertebrate taxa (sixty-four compared with forty-

three) and greater mean densities of benthic invertebrates (up to five times higher) in alkaline streams. Despite the elevated concentrations of aluminium in acidic streams (total reactive aluminium of 123–363 µg/L compared with 84 µg/L in alkaline streams), they suggested that changes in the food supply (e.g. epilithon, changed because of pH) and not aluminium were responsible for the depauperate fauna in those streams. However, they did note that much of the toxic inorganic single unit aluminium present in the acidic streams was probably complexed by dissolved humic matter.

The guidelines for freshwater systems recommended by CCREM (1991) have been used. The guideline for acid waters is based on the no-effect concentration for the toad *Bufo americanis* (Clark & LaZerle 1985), whereas the guideline for waters with pH greater than 6.5 is based on the recommendation of USEPA (1973), which reported that aluminium concentrations greater than 100 µg/L would be deleterious to the growth and survival of fish. No guideline is recommended for aluminium in marine waters because of the lack of adequate data regarding aluminium toxicity to saltwater species.

Ammonia

The concentration of undissociated ammonia in fresh waters should be limited to 0.02–0.03 mg/L (as NH_3). The concentration of total ammonia corresponding to this limit as a function of pH and temperature is given in Table 2.3*.*

There are insufficient data to allow a guideline to be recommended for ammonia in marine waters.

The most common sources of ammonia entering surface waters and groundwaters are domestic sewage and industrial effluents. Ammonia is very soluble in water, the solubility being around 100,000 mg/L at 20°C.The toxicity of ammonia is dependent on the concentration of the undissociated form (NH_3), which is controlled by the pH and temperature of the solution. Ammonia is a non-persistent and non-cummulative toxicant to aquatic life.

There have been many reviews of ammonia toxicity (e.g. Alabaster & Lloyd 1982; Thurston & Russo 1983; USEPA 1985g; CCREM 1991). USEPA (1985g) found that ammonia was acutely toxic to freshwater organisms at concentrations (uncorrected for pH and temperature) ranging from 0.5 mg/L to 23 mg/L for nineteen invertebrate species and from 0.08 mg/L to 4.6 mg/L for twenty-nine fish species. Invertebrates are generally more tolerant to ammonia than fish, and phytoplankton and aquatic vascular plants more tolerant again (USEPA 1986; CCREM 1991). Salmonid fish appear to be particularly sensitive to ammonia. Acute toxicity to fishes may cause loss of equilibrium; hyperexcitability; increased breathing rate, cardiac output and oxygen uptake; and, in extreme cases, convulsions, coma and death. Chronic effects of ammonia include a reduction in hatching success, reduction in growth rate and morphological development, and pathological changes in gill, liver and kidney tissue (USEPA 1986).

A limit of 0.02–0.03 mg/L undissociated ammonia has been adopted in a number of criteria documents (e.g. USEPA 1985g; CCREM 1991), based largely on the toxicity of ammonia to sensitive cold-water fish. The concentration of total ammonia corresponding to this limit of undissociated ammonia is shown as a function of pH and temperature in Table 2.3. The relevance of this very low undissociated ammonia concentration to a number of Australian waters is questionable; however, in the absence of any more-definitive data, these overseas values are recommended.

USEPA (1986) found that there were insufficient data for marine species to permit an ammonia criterion to be recommended. The use of the above guideline for fresh waters is probably inappropriate for marine waters in view of the fact that the recommended limit is based on protecting sensitive cold-water fishes, and that the high salinity of marine waters will significantly modify the ammonia concentrations listed in Table 2.3.

Table 2.3 Recommended guidelines for total ammonia concentration (mg/L as NH_3) at different temperatures

pH	0°C	5°C	10°C	15°C	20°C	25°C	30°C
6.5	2.5	2.4	2.2	2.2	1.49	1.04	0.73
6.75	2.5	2.4	2.2	2.2	1.49	1.04	0.73
7.0	2.5	2.4	2.2	2.2	1.49	1.04	0.74
7.25	2.5	2.4	2.2	2.2	1.5	1.04	0.74
7.5	2.5	2.4	2.2	2.2	1.5	1.05	0.74
7.75	2.3	2.2	2.1	2.0	1.4	0.99	0.71
8.0	1.53	1.44	1.37	1.33	0.93	0.66	0.47
8.25	0.87	0.82	0.78	0.76	0.54	0.39	0.28
8.5	0.49	0.47	0.45	0.44	0.32	0.23	0.17
8.75	0.28	0.27	0.26	0.27	0.19	0.16	0.11
9.0	0.16	0.16	0.16	0.16	0.13	0.1	0.08

Source: USEPA (1985g)

Antimony

The concentration of total antimony in fresh waters should not exceed 30 µg/L.

The concentration of total antimony in marine waters should not exceed 500 µg/L.

Two forms of antimony are found in natural water: antimony (III) occurs under 'moderately oxidising conditions', whereas antimony (V) predominates in 'highly oxidising' environments (Callahan et al. 1979). Water-borne antimony can result from natural weathering of geological formations and minerals as well as from anthropogenic sources such as effluents from mining, manufacturing and municipal wastes (USEPA 1988b). There are no known biological functions of antimony (Wood & Wang 1985). There are few studies on the bioaccumulation of antimony in the aquatic environment. Chapman et al. (1986) reported bioconcentration factors of 40 and 16,000 for freshwater fish and invertebrates respectively, whereas no detectable bioconcentration of antimony was found in bluegill sunfish (*Lepomis macrochirus*) during twenty-eight day's exposure (USEPA 1978).

Available data for antimony indicate that chronic toxicity to freshwater aquatic life occurs at concentrations as low as 1,600 µg/L (USEPA 1986). Aquatic freshwater plants may be even more sensitive than fish or invertebrate species. Toxicity to algae occurs at concentration of 600 µg/L (USEPA 1986). According to USEPA (1988b), freshwater aquatic organisms should not be affected by antimony(III) if the four-day average concentration does not exceed 30 µg/L.

Little information is available regarding toxicity of antimony to marine organisms. USEPA (1988b) tested the acute toxicity of antimony (III) in eleven species, with acute values ranging from 3.7 µg/L for sea urchin to values greater than 1,000,000 µg/L for mummichog fish. USEPA concluded that

saltwater aquatic organisms should not be affected unacceptably if the four-day average concentration does not exceed 500 µg/L (USEPA 1988b).

Arsenic

The total concentration of arsenic in fresh and marine waters should not exceed 50 µg/L.

Arsenic is released into the environment naturally by weathering of arsenic-containing rocks and volcanic activity. The estimated amount of arsenic released as a result of human activities is about twice that from weathering (Ferguson & Gavis 1972). Several forms of arsenic occur in natural waters, depending upon the redox potential and pH, the two most common being arsenic (III) and arsenic (V). Both arsenic (III) and arsenic (V) form stable bonds with carbon, resulting in numerous organo-arsenical compounds, some of which are very toxic (e.g. methylarsine).

Both inorganic forms of arsenic appear to have similar toxicity in fresh waters (CCREM 1991). Acute toxicity of arsenic (III) to invertebrates occurred at concentrations as low as 812 µg/L (Sanders & Cope 1968; Call et al. 1983; Lima et al. 1984). The lowest concentration of arsenic (V) causing acute toxicity was 850 µg/L for a cladoceran *Bosmina longirostris* (Passino & Novak 1984). Adult freshwater fish are generally less sensitive to arsenic. Concentrations of arsenic (III) causing an acute toxic response in fish ranged upwards from 13,300 µg/L (CCREM 1991). The lowest acute toxic concentration of arsenic (V) for fish (rainbow trout) was 10,800 µg/L (Hale 1977).

The chronic toxicities of arsenic (III) and arsenic (V) to fish are in the same concentration range as acute toxicities to invertebrates (CCREM 1991). Arsenic (V) seems to be more toxic to plants than arsenic (III). Toxicity for the alga *Scenedesmus obliquus* occurred at concentrations from 48 µg/L upwards (Vocke et al. 1980).

USEPA (1986) recommended a criterion (four-day average) for arsenic (III) in fresh water of 190 µg/L. Not enough data were available to recommend a criteria for any other arsenic compound (USEPA 1986). CCREM (1991) and Hart (1982) recommended a guideline of 50 µg/L for the protection of freshwater aquatic life.

Few data are available on the toxicity to saltwater organisms of any form of arsenic other than inorganic arsenic (III). According to USEPA (1986), acute toxicity of arsenic (III) to saltwater animals ranged from 230–16,000 µg/L. USEPA (1986) recommended a criterion (four-day average) for arsenic (III) in saltwater of 36 µg/L.

Beryllium

The concentration of total beryllium in fresh waters should not exceed 4 µg/L; higher values are possibly acceptable in hard water.

No guideline is recommended for marine waters.

The major source of beryllium in the environment is the combustion of fossil fuels. It can also enter natural waters through weathering processes, atmospheric fall-out and discharges from industrial and municipal operations (Tepper 1972).

The acute toxicity of beryllium to freshwater fish is dependent on water hardness, with beryllium being more toxic in soft water (CCREM 1991). Acute toxicity to the fathead minnow (*Pimephales promelas*), guppy (*Poecilia reticulata*) and bluegill (*Lepomis macrochirus*) over a hardness range of

20–400 mg/L decreased about two orders of magnitude (USEPA 1980a). Acute toxicity in soft water (hardness of 23 mg/L) for the most sensitive species, the guppy, ranged from 130 µg/L to 450 µg/L (Slonim 1973). Acute toxicity to the only invertebrate species studied (*Daphnia magna*) was comparable to the acute toxicity to fish. No chronic tests have been conducted with freshwater fish; however, chronic effects of beryllium on *D. magna* showed adverse effects on survival at concentrations of 36 µg/L (USEPA 1980a).

The recommended guideline for beryllium in fresh waters is based on the adverse chronic effect on *D. magna* with a safety factor of 0.1. USEPA (1986) recommended a concentration of 5.3 µg/L (four-day average) for fresh waters. Very limited data are available regarding beryllium toxicity in saltwater animals and no guideline can be recommended for marine systems at this stage.

Cadmium

The concentration of total cadmium in fresh waters should not exceed 0.2–2 µg/L, depending on water hardness.

The concentration of total cadmium in marine waters should not exceed 2 µg/L.

In natural surface waters, cadmium occurs predominantly in the divalent form, comprising several inorganic and organic compounds (Reeder et al. 1979). Cadmium may be accumulated by a number of aquatic organisms, with bioconcentration factors in the order of 100–100,000 (Reeder et al. 1979).

Acute toxicity of cadmium to freshwater animal species in forty-four genera ranged from 1 µg/L for rainbow trout to 28,000 µg/L for mayfly (USEPA 1986). Aquatic plants were affected by cadmium concentrations as low as 2 µg/L. The acute toxicity of cadmium to aquatic biota is affected by water hardness, pH, water temperature and the presence of organic compounds (CCREM 1991; Giesy et al. 1977). The antagonistic effect of water hardness on acute toxicity has been demonstrated with five fish species (USEPA 1986).

Chronic toxicity values for cadmium exposure of twelve fish and four invertebrates species ranged from 0.15 µg/L for *Daphnia magna* to 156 µg/L for Atlantic salmon (USEPA 1986), with salmonids being generally the most sensitive fish species (CCREM 1991). Water hardness also affects the chronic toxicity, with cadmium being less toxic in hard water (Sauter et al. 1976).

USEPA (1986) established a criterion for fresh waters based on a formula requiring a value for water hardness. CCREM (1991) recommended a concentration in the range of 0.2–1.8 µg/L, depending on water hardness. Hart (1982) established a criterion of 0.2 µg/L for the protection of freshwater aquatic life.

The acute toxicity of cadmium to five species of marine fishes ranged from 577 µg/L for larval Atlantic silverside to 114,000 µg/L for juvenile mummichog fish. The acute values for thirty invertebrates ranged from 15.5 µg/L upwards (USEPA 1985a). The acute toxicity of cadmium generally increases as salinity decreases, whereas temperature effects seem to be species-specific (USEPA 1986). The few chronic date available appears to reflect the effects of salinity and temperature (USEPA 1986). USEPA recommended a criterion of 9.3 µg/L for saltwater; Hart (1982) established a criterion of 2 µg/L.

Chromium

The total concentration of chromium in fresh waters should not exceed 10 µg/L, on the assumption that all is in chromium (VI) form.

The total concentration of chromium in marine waters should not exceed 50 µg/L, on the assumption that all is in chromium (VI) form.

In natural waters, chromium is present mainly in the trivalent, chromium (III), and hexavalent, chromium (VI), forms (Hart 1982). The form of chromium present appears to significantly affect toxicity to aquatic organisms and the behaviour of chromium in the aquatic environment. Chromium (III) is least soluble in the pH range of natural waters (Taylor et al. 1979). Precipitation of chromium hydroxide is thought to be the dominant removal mechanism for chromium (III) in natural water.

Studies in lake water showed that the ratio of chromium (III) to chromium (VI) is affected by the amount of organic matter and dissolved oxygen (Benes & Steinnes 1975). Chromium (VI) is quite soluble, existing in solution as a complex anion (CCREM 1991). Chromium is bioaccumulated by aquatic organisms, with bioconcentration factors ranging between 100 and 1,000 (CCREM 1991). Chromium (VI) is able to penetrate cell membranes to a much greater extent than chromium (III), which is one reason why chromium (VI) is considered to be the more toxic form of the metal (USEPA 1973; Anderson 1987).

Chromium (VI)

USEPA (1985b) compiled acute toxicity data for chromium (VI) in freshwater animal species in twenty-seven genera. Values ranged from 23 µg/L for a cladoceran to 1,870 µg/L for a stonefly, whereas all five species of daphnids tested were especially sensitive. Acute toxicity of chromium (VI) appears to decrease as hardness and pH increase, but the data were insufficient for USEPA (1985b) to develop a guideline based on water hardness.

Chronic values for three freshwater fish (rainbow trout, brook trout and fathead minnow) ranged from 265 µg/L to 2,000 µg/L (USEPA 1986). In all three chronic tests, a temporary reduction in growth occurred at low concentrations of chromium (VI). Daphnids were more sensitive, with chronic values ranging from 2.5 µg/L to 40 µg/L (Trabalika & Gehrs 1977; USEPA 1985b). USEPA (1986) recommended a criterion of 11 µg/L (four-day average) for chromium (VI).

Acute toxicity of chromium (VI) to twenty-three saltwater animals ranged from 2,000 µg/L for a polychaete worm to 105,000 µg/L for the mud snail (USEPA 1986). Chronic values for a polychaete ranged from 13 µg/L to 37 µg/L, whereas for a mysid the value was 132 µg/L. USEPA (1986) recommended a criterion of 50 µg/L (four-day average) for chromium (VI) in marine waters.

Chromium (III)

Acute values for chromium (III) are available for twenty freshwater animal species in eighteen genera. These values ranged from 2,221 µg/L upwards (USEPA 1986). A concentration of 70 µg/L was reported for chronic exposure of rainbow trout (CCREM 1991), and the chronic value for fathead minnow in hard water was 1,025 µg/L (USEPA 1986). Hardness has a significant influence on the toxicity, with chromium (III) being more toxic in soft water, and USEPA (1986) established a guideline based on a formula requiring a value for water hardness.

Only two values are available for the acute toxicity of chromium (III) to saltwater animals; 10,300 µg/L for the eastern oyster and 31,500 µg/L for the mummichog fish. Hart (1982) recommended a criterion of 10 µg/L for total chromium to protect aquatic communities; CCREM (1991) established a guideline of 2 µg/L for total chromium in fresh waters.

Copper

The concentration of total copper in fresh waters should not exceed 2–5 µg/L, depending on water hardness.

The concentration of total copper in marine waters should not exceed 5 µg/L.

Natural sources of copper in aquatic environments include weathering of copper minerals and native copper; however, anthropogenic activities can release significant amount of copper to the environment (McNeely et al. 1979). Copper is commonly found in the +II state in natural waters. The composition of the various copper species depends on pH and the presence of inorganic and organic ligands in the water. As an essential element, copper is readily accumulated by plants and animals; bioconcentration factors ranging from 100 to 26,000 have been recorded for various species of phytoplankton, zooplankton, macrophytes, macroinvertebrates and fish (Spear & Pierce 1979).

The toxicity of copper in fresh waters increases with decreasing water hardness and dissolved oxygen concentration. High concentrations of chelating agents (e.g. humic acids, amino acids) and suspended solids lead to lower copper toxicity, presumably by complexation forming less available forms of copper (Alabaster & Lloyd 1982). Acute toxicity data for freshwater species in forty-one genera are available (USEPA 1985c). At a hardness of 50 mg/L, the values ranged from 17 µg/L for *Ptychocheilus* to 1,000 µg/L for *Acroneuria.* Skidmore and Firth (1983) found the acute toxicity of copper for ten Australian species ranged from 200 µg/L to 7,800 µg/L, and Bacher and O'Brien (1990) reported that acute toxicities for Australian species ranged from 40 µg/L to 21,000 µg/L.

Chronic values for fifteen freshwater species ranged from 4 µg/L upwards (USEPA 1986). Changes in fish behaviour have been demonstrated at concentrations as low as 4 µg/L (CCREM 1991). A size effect has been shown for bluegill, guppy and rainbow trout, with juveniles being more sensitive than adults (Chakoumakos et al. 1979; Tsai & Chang 1981). Fish and invertebrates seem to be about equally sensitive to the chronic toxicity of copper in fresh waters. The sensitivity of a number of species of freshwater plants that were tested was similar to those of animals (USEPA 1986). Protection of animal species appears to offer adequate protection of plants.

CCREM (1991) recommended a guideline for copper based on hardness, with concentrations ranging from 2 µg/L to 6 µg/L. USEPA (1986) recommended a criterion based on a formula requiring a value for water hardness. Hart (1982) established a criterion of 5 µg/L for filterable copper in soft waters with low complexing capacity.

The acute toxicity for saltwater animals ranged from 5.8 µg/L for blue mullet to 600 µg/L for green crab (USEPA 1986). Chronic life-cycle tests with a mysid showed adverse effects at 77 µg/L, leading to an acute–chronic ratio of 3.4. Toxic effects on saltwater algae were observed at copper concentrations between 5 µg/L and 100 µg/L (USEPA 1985c). USEPA (1986) recommended a criterion of 2.9 µg/L for the protection of marine life.

Cyanide

The concentration of free cyanide in fresh and marine waters should not exceed 5 µg/L.

Cyanides can be introduced into water from cleaning, electroplating and chemical industries as well as from effluents of mining industries (Hart 1974; CCREM 1991). Cyanides may also be released into the aquatic environment through decomposition of plants that synthesise cyanoglycosides and micro-organisms that produce free cyanide as a result of their metabolic processes (CCREM 1991). Cyanides comprise a diverse group of organic and inorganic compounds characterised by the C≡N group. It is generally considered that HCN is the most toxic form of cyanide; however, most toxicity data are reported in terms of free cyanide, which is the sum of molecular HCN and the cyanide ion (CN^-) expressed as CN.

No pronounced correlation was found between acute toxicity of cyanide to fish and alkalinity, hardness or pH below 8.3 (USEPA 1985d); however, pH will greatly modify the toxicity of metallocyanide complexes (Leduc et al. 1982). Iron–cyanide complexes may also dissociate to form HCN in the presence of sunlight (CCREM 1991). There is also evidence that a number of metal–cyanide complexes may have inherent toxicity (Leduc et al. 1982).

Acute toxicity data for free cyanide are available for a wide variety of freshwater species, with invertebrates generally found to be more tolerant than fish. Acutely toxic concentrations to invertebrates ranged from 95 µg/L to 2,500 µg/L (Cairns et al. 1978), and concentrations from about 50 µg/L to 200 µg/L were fatal to juveniles of most of the more sensitive fish species (USEPA 1985d). The toxicity of cyanide towards freshwater fish is affected by temperature (Wuhrmann & Woker 1955; Smith et al. 1978), dissolved oxygen (Doudoroff 1976; Doudoroff et al. 1979; Smith et al. 1978), the presence of other contaminants (Speyer 1975; Broderius & Smith 1979) and pH (Clark et al. 1984).

Chronic toxic concentrations for the exposure of bluegill, brook trout and fathead minnow were 14 µg/L, 8 µg/L and 16 µg/L respectively (USEPA 1985d); chronic values for two freshwater invertebrates were 18 µg/L and 34 µg/L. Freshwater plants showed a wide range of sensitivities to cyanide, and were affected at concentrations ranging from 30 µg/L to 26,000 µg/L (USEPA 1986). CCREM (1991) recommended a concentration of 5 µg/L free cyanide to protect freshwater aquatic life; USEPA established a criterion of 5.2 µg/L (four-day average); and Hart (1982) recommended a criterion of 10 µg/L.

Acute toxicity of free cyanide to saltwater species ranged from 4.9 µg/L to 10,000 µg/L, with invertebrates being both the most and least sensitive species (USEPA 1986). Long-term survival in early life-stage tests with the sheepshead minnow resulted in a chronic value of 36 µg/L, whereas long-term survival tests with a mysid gave a chronic concentration of 70 µg/L (USEPA 1986).

Iron

In most bodies of fresh water, it is unlikely that toxic effects will occur at total iron concentrations below around 1,000 µg/L. However, if the ferrous iron concentration is above 500 µg/L, problems may occur due to floc formation when the ferrous iron is oxidised.

There are insufficient data to recommend a guideline for iron in marine waters.

Iron is the fourth most abundant element in the earth's crust and may be present in natural waters in varying quantities depending upon the geology of the area and other chemical components of the waterway (USEPA 1986). The most common oxidation states of iron in water are the ferrous (Fe^{2+}) and the ferric (Fe^{3+}) states, although other forms may be present in organic and inorganic wastewater streams. In surface waters, iron is generally present in the ferric state; in reducing waters, the ferrous form can persist.

Iron is an essential trace element for both plants and animals; however, acute toxicity to aquatic insects has been reported at iron concentrations ranging from 320 µg/L to 16,000 µg/L (Warnick & Bell 1969). The three-week LC_{50} for *Daphnia magn*a was 5,900 µg/L (Biesinger & Christen, 1972). A reduction of 50% in the hatchability of fathead minnow eggs occurred at iron concentrations of 1,500 µg/L (Sykora et al. 1972), and the safe concentration for exposure of juvenile brook trout ranged between 7,500 µg/L and 12,500 µg/L (CCREM 1991). CCREM (1991) recommended a guideline concentration of 300 µg/L for the protection of freshwater aquatic life.

Iron hydroxyoxide flocs resulting from the oxidation of ferrous iron in leachates from rock and disused mines and, in bottom waters, from dams and reservoirs can cause problems as a result of smothering effects (Hickey C.W., Water Research Centre, Hamilton, New Zealand, pers. comm., June 1992). No adequate data on iron toxicity to saltwater species were available.

Lead

The concentration of total lead in fresh waters should not exceed 1–5 µg/L, depending on water hardness.

The concentration of total lead in marine waters should not exceed 5 µg/L.

Anthropogenic outputs to the environment outweigh all natural sources (e.g. weathering of sulfide ores, especially galena), and lead reaches the aquatic environment through precipitation, fall-out of lead dust, street runoff and industrial and municipal wastewater discharges (USEPA 1976; Jaques 1985). Lead is generally present in very low concentrations in natural waters. In fresh waters, the main species of lead are $PbCO_3$ and lead-organic complexes, with very much smaller amounts of free lead ions. In marine waters, lead carbonate is the predominant form (Hart 1982).

The acute toxicity of lead to several species of freshwater animals was greater in soft water than in hard water. At a hardness of 50 mg/L (as $CaCO_3$) the acute sensitivities of ten species ranged from 143 µg/L for an amphipod to 236 µg/L for a midge (USEPA 1985e). Acute toxicities for Australian freshwater species ranged from 180 µg/L to 500 µg/L (Bacher & O'Brien 1990). The chronic toxicity also decreased with increasing water hardness (CCREM 1991). Reproduction of *Daphnia magna* was impaired 16% by 30 µg/L lead in soft water, and 44% of trout developed spinal deformities at lead concentrations of 31 µg/L in soft water. However, in hard water, none of the rainbow trout showed deformities at concentrations of 190 µg/L (Biesinger & Christen 1972). Freshwater algae were affected by concentrations of lead above 500 µg/L, based on data for four species (USEPA 1986). Bioconcentration factors for four species of invertebrates and two species of fish ranged from 499 to 1,700 (USEPA 1985e).

CCREM (1991) recommended a guideline for lead based on hardness, with concentrations ranging from 1 µg/L to 7 µg/L. USEPA (1986) recommended a criterion based on a formula requiring a value for water hardness. Hart (1982) established a criterion of 5 µg/L for filterable lead in soft waters.

The acute toxicity for thirteen marine animal species ranged from 315 µg/L (mummichog) to 27,000 µg/L (soft-shell clam). Fewer data are available for chronic toxicity in marine waters. Unacceptable effects for a mysid were observed at 37 µg/L, and macroalgae were affected at 20 µg/L (USEPA 1986). USEPA (1986) recommended a criterion of 5.4 µg/L (four-day average) for the protection of marine species.

Mercury

The total concentration of mercury in fresh and marine waters should not exceed 0.1 µg/L, on the assumption that only a small percentage (less than 10%) is present as methylmercury.

Mercury in the aquatic environment exists mainly as complexes of mercury (II) and as organomercurials (Hart 1982). Of particular concern to the aquatic environment is that inorganic forms of mercury that have a relatively low toxicity and availability to concentrate in tissues may be converted *in situ* into organomercury complexes (particularly methylmercury), which are more toxic and tend to bioaccumulate. The guideline for mercury is based mainly on limiting the bioaccumulation in seafood that may be eaten by humans.

Acute toxicity of mercury (II) to invertebrates species ranged from 2.2 µg/L for *Daphnia pulex* (Canton & Adema 1978) to 2,00 µg/L for a mayfly (Warnick & Bell 1969). Acute values for fish raged from 30 µg/L for a guppy to 1,000 µg/L for *Tilapia* (Deshmukh & Marathe 1980; Quereshi & Saksena 1980). Few data are available regarding acute toxicity of organomercury compounds, but they all appear to be four to thirty-one times more toxic than mercury (II), (USEPA 1986).

Methylmercury appears to be the most chronically toxic of the tested mercury compounds, with chronic toxicity occurring at less than 0.04 µg/L for *D. magna* and 0.52 µg/L for brook trout (McKim et al. 1976; Biesinger et al. 1982). The most sensitive plant species generally appear to be less sensitive than sensitive animal species to both mercury (II) and methylmercury (CCREM 1991). Bioconcentration factors of 5,000 have been reported for mercury (II); factors for methylmercury ranged from 4,000 to 85,000 (USEPA 1986). CCREM (1991) and Hart (1982) recommended a criterion of 0.1 µg/L in fresh waters. USEPA (1986) established a criterion of 0.012 µg/L (four-day average) for the protection of freshwater aquatic life, based on the assumption that all the mercury is discharged as methylmercury.

USEPA (1985f) summarised data on the acute toxicity of mercuric chloride in saltwater, with values ranging from 3.5 µg/L to 1,700 µg/L. Generally, fish tend to be more resistant than molluscs and crustaceans. Concentrations of mercury (II) that affect the growth and photosynthetic activity of saltwater plants ranged from 10 µg/L to 160 µg/L. Bioconcentration factors of 10,000 to 40,000 resulted for mercuric chloride and methylmercury with an oyster (USEPA 1986). USEPA (1986) established a criterion of 0.025 µg/L (four-day average) for the protection of marine aquatic life, again based on all the mercury being present as methylmercury.

Nickel

The concentration of total nickel in fresh waters should not exceed 15–150 µg/L, depending on water hardness.

The concentration of total nickel in marine waters should not exceed 15 µg/L.

Nickel can enter the environment naturally through weathering of minerals and rocks and through anthropogenic sources. More than 90% of the nickel in the aquatic environment is associated with particulate matter or sediments, and only about 7% is in the dissolved phase (Hart 1982). The chemistry of nickel in aqueous solution is essentially that of a divalent cation (CCREM 1991).

USEPA (1986) compiled data on the acute toxicity of nickel to twenty-one freshwater species in eighteen genera. Acute values ranged from 1,100 µg/L for a cladoceran to 43,000 µg/L for a fish. Data available for two species indicated that chronic toxicity decreases as hardness increases. The measured chronic concentrations ranged from 15 µg/L for *Daphnia magna* in soft water to 530 µg/L for fathead minnow in hard water (USEPA 1986). Toxicity appears to be also pH dependent: the higher the acidity, the higher the toxicity (CCREM 1991). Reduced growth was noted in several freshwater algae at concentrations as low as 50 µg/L (USEPA 1986). Bioconcentration is not a significant problem in the aquatic environment; bioconcentration factors ranged from 0.8 for fish muscle to 193 for a cladoceran (USEPA 1986).

CCREM (1991) recommended a guideline for nickel based on hardness, with concentrations ranging from 25 µg/L to 150 µg/L. USEPA (1986) recommended a criterion based on a formula requiring a value for water hardness. Hart (1982) established a criterion of 25 µg/L for filterable nickel in soft waters.

The acute toxicity of nickel for twenty-three saltwater species in twenty genera has been tested (USEPA 1986). The values ranged from 152 µg/L for juveniles of a mysid to 1,100 mg/L for clams. Few data are available regarding chronic toxicity of nickel in saltwater species. Exposure of *Mysidopsis bahia* to 141 µg/L nickel resulted in reduced survival and reproduction. The measured acute–chronic ratio was 5.5 (USEPA 1986). USEPA (1986) recommended a criterion of 8.3 µg/L (four-day average) for the protection of saltwater species.

Selenium

The concentration of total selenium in fresh waters should not exceed 5 µg/L.

The concentration of total selenium in marine waters should not exceed 70 µg/L.

Although the major source of selenium in the environment is weathering of rocks and soils (Rosenfeld & Beath 1964), anthropogenic sources such as emissions from burning fossil fuels may also contribute selenium to natural waters (Hart 1982;
Health & Welfare Canada 1980). In the aquatic environment, selenium exists in two common oxidation states: selenium (IV), known as selenite (SeO_3^{2-}); and selenium (VI), known as selenate (SeO_4^{2-}). Most selenites are less water-soluble than are selenates (CCREM 1991). Selenium can be methylated by various organisms (Stadtman 1974; Chau et al. 1976). According to USEPA (1987a), selenium (IV) is generally two to four times more toxic than selenium (VI) to freshwater and saltwater fish and invertebrates species. In contrast to the effect on aquatic animals, selenium (VI) is either as toxic as or more toxic to aquatic plants than selenium (IV).

Selenium (IV)

USEPA (1987a) compiled acute data from freshwater fish and invertebrates species with values ranging from 340 µg/L for an amphipod to 203,000 µg/L for a leech. Chronic concentrations were

available for two fish and two invertebrates, and ranged from 47 µg/L to 690 µg/L. The acute toxicity of selenium (IV) for nine species of freshwater algae ranged from 500 µg/L to 30,000 µg/L (USEPA 1987a). Acute toxicity to sixteen saltwater animals, including eight invertebrates and eight fishes, ranged from 600 µg/L to 17,300 µg/L, whereas chronic values for mysid and sheepshead minnow ranged from 222 µg/L to 675 µg/L, resulting in acute–chronic ratios of 7 and 11 (USEPA 1987a).

Selenium (VI)
USEPA (1987a) summarised the acute toxicity of selenium (VI) in twelve freshwater animal species. Acute values ranged from 75 µg/L for an amphipod to 44,000 µg/L for a leech. Chronic toxicity tests were conducted with *Daphnia magna*, fathead minnow and rainbow trout. The reported values ranged from 565 µg/L to 1,200 µg/L, with an acute–chronic ratio range of 3 to 16 respectively. Fewer data are available concerning the toxicity of selenium (VI) to saltwater species. Acute toxicity concentrations of prolarvae and juveniles of striped bass, *Morane saxatilis*, resulted in a ninety-six hour LC_{50} of 9,800 µg/L and 85,800 µg/L respectively (USEPA 1987a).

CCREM (1991) and Hart (1982) recommended critera for total selenium in fresh waters of 1 µg/L and 10 µg/L respectively. USEPA (1987a) established a criterion for total selenium of 5 µg/L for fresh waters and 71 µg/L for marine waters.

Silver
The concentration of total silver in fresh waters should not exceed 0.1 µg/L.

The concentration of total silver in marine waters should not exceed 1.0 µg/L.

Silver is among the less common but most widely distributed elements in the earth's crust (CCREM 1991). Primary sources of anthropogenic silver in surface waters include industrial and smelting wastes, and wastes from jewellery manufacture and the production and disposal of photographic materials (USEPA 1987b). Silver exists in aqueous systems primarily in the univalent state Ag(I).

Silver is one of the most toxic metals to aquatic life. Silver nitrate and silver iodide are particularly toxic, whereas silver chloride is 300 times less acutely toxic (CCREM 1991). The acute toxicity of silver is related to the water hardness; toxicity decreases as hardness increases. Acute toxicity values for both freshwater microinvertebrates and fishes ranged from 0.9 µg/L to 29 µg/L for the most sensitive species (USEPA 1987b). Chronic toxicity concentrations for silver in fresh waters were derived from a test with rainbow trouts and fathead minnows, the values being 0.2 µg/L and 0.5 µg/L respectively (USEPA 1987b). Bioconcentration factors in fresh waters ranged from not detectable to 150. CCREM (1991) and USEPA (1987b) recommended criteria for the protection of freshwater aquatic life of 0.1 µg/L and 0.12 µg/L (four-day average) respectively.

USEPA (1987b) compiled a data base of acute toxicity values for silver exposure to twenty-one species of saltwater animals. The concentrations ranged from 3 µg/L for the eastern oyster to greater than 100,000 µg/L for the mummichog fish. Chronic toxicity has been determined in five life-cycle tests with the saltwater mysid. Decreases in reproduction occur at concentrations of 15 µg/L to 88 µg/L (McKenney 1982). USEPA (1987b) recommended a criterion of 0.92 µg/L (four-day average) for the protection of marine aquatic life.

Sulfide—hydrogen sulfide

The concentration of undissociated hydrogen sulfide in fresh and marine waters should not exceed 2 µg/L.

Sulfides are present in many industrial wastes, such as those from tanneries, paper mills, chemical plants and gas works. A major source of hydrogen sulfide in aquatic systems is anaerobic decomposition of sewage, sludge, algae and other naturally deposited organic materials (USEPA 1986). The most toxic form is H_2S, which is the predominant form (99%) at pH 5, whereas sulfide (HS^-) concentrations increase with higher pH values (USEPA 1986).

The toxicity of sulfide to aquatic animals is dependent on the temperature, pH and dissolved oxygen concentration (USEPA 1986). Fish usually exhibit a strong avoidance reaction to sulfide (Jones 1964). Safe levels for walleyes and fathead minnows were found to vary from 2.9 µg/L to 12 µg/L, with eggs being the least sensitive and juveniles the most sensitive in short-term tests (Smith 1971). On the basis of chronic tests evaluating growth and survival, the safe H_2S concentration for bluegill (*Lepomis macrochirus*) juveniles and adults was 2 µg/L (USEPA 1986). According to USEPA (1986), water containing 2 µg/L undissociated H_2S would not be hazardous to most fish and other aquatic wildlife.

Thallium

The concentration of total thallium in fresh waters should not exceed 4 µg/L.

The concentration of total thallium in marine waters should not exceed 20 µg/L.

Thallium is introduced into the environment by natural weathering and as waste from the production of other metals (CCREM 1991), and is present in trace amounts in fresh waters (McNeely et al. 1979). Thallium (I) is the predominant form of thallium in most aerobic waters; however, in waters with high oxygen content some thallium (III) may be present (USEPA 1979a). In reducing environments, thallium may be precipitated in the elemental form or as the insoluble sulfide if sulfur is present (Lee 1971; Magorion et al. 1974). Thallium has been reported to exhibit chronic toxicity to freshwater aquatic life at concentrations of 40 µg/L (USEPA 1986). The guideline value for fresh waters has been derived by multiplying this chronic toxicity value by 0.1. According to USEPA (1986), acute toxicity to saltwater aquatic life occurred at concentrations of 2,130 µg/L, and would probably occur at lower concentrations among species that are more sensitive than those tested.

No data are available regarding the chronic toxicity of thallium to sensitive saltwater animals. The guideline value for marine waters has been derived from the acute toxicity using an application factor of 0.01.

Tributyltin

The concentration of tributyltin in fresh waters should not exceed 8 ng/L.

The concentration of tributyltin in marine waters should not exceed 2 ng/L.

Inorganic tin is generally considered to be non-toxic; however, the attachment of alkyl or aryl groups to the tin atom greatly increases the toxicity (CCREM 1991, Appendix X). Tributyltin, which has been used as an additive in antifouling paints for many years, seems to be one of the most toxic of the organotin compounds, and its use is either banned or is proposed to be banned in all Australian States.

CCREM (1991) has summarised the toxicity data for tributyltin for freshwater animals: the ninety-six hour LC_{50} to freshwater fish ranged from 2.6 µg/L to 13 µg/L. In a study of the chronic toxicity of tributyltin, Brooke et al. (1986) found a significant reduction in the number of surviving young of *Daphnia magna* at a concentration of 0.2 µg/L after twenty-one days' exposure. Brooke et al. (1986) estimated that the lowest observed effect level (LOEL) for tributyltin was 0.08 µg/L. Tributyltin is bioaccumulated; Gibbs et al. (1988) reported a maximum bioaccumulation factor of 250,000 in the snail *Nucella lapillus* after fifty-four days' exposure at concentrations of 1–2 ng/L tributyltin (as Sn). USEPA proposed that freshwater organisms should not be affected if the four-day average concentration of tributyltin did not exceed 26 ng/L more than once every three years on average (USEPA 1989).

Zabel et al. (1988) have reviewed the effects of tributyltin on marine organisms. On the basis of this review, they recommended that the level in marine waters should be set at 1 ng/L. They also recommended that the level for fresh waters also be set at this value, on the grounds that the ecotoxicological effects have been well established for marine molluscan species and the indications are that freshwater species could be equally as sensitive. Abel et al. (1987) reported that the United Kingdom adopted a target value for marine waters of 20 ng/L in 1985, with an improved target of 2 ng/L for 1989. ANZECC passed a resolution in 1990 that antifoulants should not release more than 5 µg tributyltin/cm^2/d; however, this rate may be reduced to 1 µg/cm^2/d.

The Canadian guideline has been adopted for freshwater aquatic life. This was derived by reducing the LOEL by a safety factor of 10 (to allow for possible bioaccumulation) to give a concentration of 8 ng/L. The United Kingdom value of 2 ng/L has been adopted for marine waters.

Zinc

The concentration of total zinc in fresh waters should not exceed 5–50 µg/L depending on water hardness.

The concentration of total zinc in marine waters should not exceed 50 µg/L.

Zinc can enter the environment through both natural (e.g. weathering and erosion) and anthropogenic (e.g. zinc production, waste incineration) processes (CCREM 1991). The toxicity of zinc is influenced by water hardness and pH. Generally, the acute toxicity of zinc is lower in waters with higher water hardness and lower pH (Mount 1966; Holcombe & Andrew 1978).

USEPA (1987c) compiled acute toxicity values of zinc for forty-three freshwater species. At a hardness of 50 mg/L, the concentrations ranged from 51 µg/L to 81,000 µg/L. Bacher and O'Brien (1990) found the acute toxicities for Australian freshwater species ranged from 140 µg/L to 6,900 µg/L, and Skidmore and Firth (1983) found a range of 340–9600 µg/L for ten Australian species. Chronic toxicity values in fish and invertebrates ranged from 36 µg/L to greater than 5,200 µg/L (USEPA 1987c). Zinc was found to bioaccumulate in freshwater animal tissues fifty to 1,130 times. CCREM (1991) recommended a guideline of 30 µg/L for total zinc in fresh waters; Hart (1982) established a criterion of 50 µg/L for zinc in soft water with low complexing capacity. USEPA (1987c) established a criterion for fresh water based on a formula requiring a value for water hardness.

Acute toxicity concentrations for zinc in thirty-three saltwater animals (twenty-six invertebrates and seven fish) ranged from 190 µg/L for cabezon (*Scarpaenichthys mormoratus*) to 320,000 µg/L for *Macoma balthica*. During a life-cycle test, unacceptable effects for a mysid were found at a

concentration of 120 µg/L, leading to an acute–chronic ratio of 3. Saltwater plants were affected at concentrations between 19 µg/L and 10,100 µg/L (USEPA 1987c). USEPA (1987c) recommended a criterion of 86 µg/L for the protection of marine aquatic life.

2.4.2 Organic compounds

This section summarises the available data on toxic impact of organic compounds on aquatic life. The main sources were CCREM (1991), Nicholson (1984) and USEPA documents (USEPA 1980b–1980ao). Guidelines could not be established for all organic parameters due to insufficient toxicity data.

Acrylonitrile

There are insufficient data available to recommend a guideline for acrylonitrile in fresh and marine waters.

The major use of acrylonitrile (CH_2=CHCN) is in the manufacture of copolymers for the production of acrylic fibres. Other uses include the manufacture of resins, elastomers and other chemicals (USEPA 1980b). The toxic effects of acrylonitrile are similar to cyanide poisoning (USEPA 1980b). Few data are available regarding toxicity of acrylonitrile to aquatic life. Acute toxicity to *Daphnia magna* occurred at 7,550 µg/L (USEPA 1980b). No definitive data are available on chronic toxicity to freshwater and saltwater animals.

Benzidine

There are insufficient data available to recommend a guideline for benzidine in fresh and marine waters.

Benzidine is used in the manufacture of dyes and as an analytical reagent. It can enter the aquatic environment mainly through discharges from dye plants (USEPA 1980c). Few data are available regarding the fate and toxicity of benzidine in aquatic systems. Acute toxicity to freshwater animals can occur at concentrations as low as 2,500 µg/L (USEPA 1986). No data are available concerning the toxicity of benzidine to saltwater organisms (USEPA 1986).

Detergents and oil dispersants

The maximum concentration of surfactant or oil dispersant compounds permitted should be 0.05 times the ninety-six hour LC_{50} values determined in the receiving water in question, using the most sensitive important species in the region.

The primary toxic components of detergents are the surfactants; for example, linear alkylate sulfonates (LASs) or alkyl benzene sulfonates (ABSs). LASs are more toxic than ABSs (at least to some fish species), but are also more biodegradable. LAS compounds are generally found to be most toxic to fish, with LC_{50} values ranging from 0.2 mg/L to 10 mg/L having been reported; however the actual toxicity depends on the structure of the LAS or ABS compounds in the detergent formulation.

Modern oil dispersants are relatively non-toxic, with LC_{50} values generally greater than 1,000 mg/L. Given the large number of surfactants and oil dispersants in use, it is not possible to provide a single reference value; rather, it is recommended that the maximum concentration permitted should be 0.05 times the ninety-six hour LC_{50} determined in the receiving water in question, using the most sensitive important species in the region.

Dichlorobenzidine

There are insufficient data available to recommend a guideline for dichlorobenzidine in fresh and marine waters.

Dichlorobenzidine is used in the production of dyes and pigments and as a curing agent for polyurethanes (USEPA 1980d). There are few data available on the bioconcentration and bioaccumulation of dichlorobenzidine in the aquatic environment. Dichlorobenzidine has been shown to bioconcentrate in fish to a significant degree, approximately 1,150 fold (USEPA 1980d). Little information is available regarding the toxicity of dichlorobenzidine to freshwater and saltwater organisms. Acute toxicity to bluegill sunfish occurs at concentrations as low as 500 µg/L (USEPA 1980d).

Diphenylhydrazine

There are insufficient data available to recommend a guideline for diphenylhydrazine in fresh and marine waters.

1,2-diphenylhydrazine is used in organic synthesis and has a major use as a starting material in the manufacture of benzidine. In aerated, aqueous solutions, 1,2-diphenylhydrazine occurs in equilibrium with azobenzene (Griffiths 1972; Rao & Hayon 1976) and the environmental fate of both forms must be considered. According to the octanol–water partition coefficient, both forms should be relatively strongly adsorbed by organic particulates (CCREM 1991). Experimental data indicate that 1,2-diphenylhydrazine is toxic to freshwater aquatic organisms. The LC_{50} ranged from 27 µg/L to 4,100 µg/L (USEPA 1980e). Data on acute and chronic toxicity in saltwater organisms are not available.

Halogenated aliphatic compounds

Chlorinated ethanes

There are insufficient data available to recommend a guideline for chlorinated ethanes in fresh and marine waters.

Chlorinated ethanes are widely used as industrial solvents, in the production of other organochlorine compounds, as dry-cleaning agents, as anaesthetics and in the manufacturing of plastics and textiles (CCREM 1991). The main sources to the environment are industrial emissions and discharge of liquid industrial wastes (USEPA 1980f). All chlorinated ethanes are sufficiently soluble to be of potential concern as water pollutants, and all are at least mildly toxic to aquatic organisms, with toxicity increasing greatly with the degree of chlorination (USEPA 1986). Chronic LOEL concentrations of chlorinated ethanes are given in Table 2.4.

Chlorinated ethanes do not bioconcentrate significantly; however, they do exhibit a greater bioconcentration potential with increasing chlorination (USEPA 1980f). Detailed information about the toxicity of chlorinated ethanes is given by USEPA (1980f, 1986).

Table 2.4 Chronic LOEL for chlorinated ethanes

Parameter	Fresh water chronic LOEL (µg/L)	Salt water chronic LOEL (µg/L)
1,2-Dichloroethane	20,000	113,000*
1,1,1-Trichloroethane	–	31,200*
1,1,2-Trichloroethane	9,400	–
1,1,2,2-Tetrachloroethane	2,400	9,020*

Parameter	Fresh water chronic LOEL (µg/L)	Salt water chronic LOEL (µg/L)
Pentachloroethane	1,100	281
Hexachloroethane	540	940*

– Insufficient data

* Acute LOEL, no chronic LOEL concentration available

Source: USEPA (1986)

Chlorinated ethylenes

There are insufficient data available to recommend a guideline for chlorinated ethylenes in fresh and marine waters.

Chlorinated ethylenes (chloroethylenes) are produced in large quantities and are widely used in industry as chemical intermediates, as anaesthetics and as solvents (CCREM 1991). The major route for entry to the environment is volatilisation during production and use; their presence in water may result from atmospheric deposition and industrial discharge. Although little information is available on the fate of chloroethylenes in the aquatic environment, volatilisation is considered to be the primary removal mechanism (CCREM 1991).

Few data are available on bioconcentration in the aquatic environment; most chloroethylenes are considered to be persistent with respect to biodegradation (Pearson & McConnell 1975). Because most of the chloroethylenes have small octanol–water partition coefficients, bioaccumulation is expected to be low (CCREM 1991). Chronic LOEL concentrations of chloroethylenes are given in Table 2.5. More information on the toxicity of chloroethylenes is given in CCREM (1991) and USEPA (1980g, 1980h, 1980i, 1986).

Table 2.5 Chronic LOEL for chlorinated ethylenes

Parameter	Fresh water chronic LOEL (µg/L)	Salt water chronic LOEL (µg/L)
Dichloroethylene	11,600	224,000*
Trichloroethylene	45,000*	2,000*
Tetrachloroethylene	840	450

* Acute LOEL, no chronic LOEL concentration available

Source: USEPA (1986)

Dichlorinated propanes and propenes

There are insufficient data available to recommend a guideline for dichlorinated propanes and propenes in fresh and marine waters.

The principal uses of dichlorinated propanes and propenes are as soil fumigants for the control of nematodes, as solvents, and in dry-cleaning and degreasing processes (Verschueren 1983; Worthingen 1983). These compounds can enter aquatic environments as discharges from industrial effluents, as runoff from agricultural land and as municipal effluents (USEPA 1980k). In general, they are water-soluble, volatile, and have a slight affinity for organic matter (CCREM 1991). Volatilisation plays a major role in their removal from water. Chronic LOEL concentrations of dichlorinated propanes and propenes are given in Table 2.6. More information regarding the toxicity of these compounds is given in USEPA (1980k; 1986).

Table 2.6 Chronic LOEL for dichlorinated propanes and propenes

Parameter	Fresh water chronic LOEL (µg/L)	Salt water chronic LOEL (µg/L)
Dichloropropane	5,700	3,040
Dichloropropene	224	790*

* Acute LOEL, no chronic LOEL concentration available
Source: USEPA (1986)

Halogenated methanes
There are insufficient data available to recommend a guideline for halogenated methanes in fresh and marine waters.

Halogenated methanes are organic compounds in which one or more of the halogens are incorporated into the methane molecule. They enter the environment from both natural and anthropogenic sources (CCREM 1991). Toxicological data regarding aquatic life are available for carbon tetrachloride (USEPA 1980l), chloroform (USEPA 1980m) and methylene chloride, methyl chloride, bromoform and methyl bromide (USEPA 1980n). The brominated compounds are more toxic to fish than the chlorinated analogs (USEPA 1980n). Some chronic LOEL for halogenated methanes are given in Table 2.7.

Table 2.7 Chronic LOEL for halogenated methanes

Parameter	Fresh water chronic LOEL (µg/L)	Salt water chronic LOEL (µg/L)
Carbon tetrachloride	35,200*	50,000*
Chloroform	1,240	–
Halogenated methanes**	11,000*	6,400

– Insufficient data
* Acute LOEL, no chronic LOEL concentration available
** Methylene chloride, methyl chloride, bromoform, methyl bromide; LOEL for the most toxic halogenated methane
Source: USEPA (1986)

Hexachlorobutadiene
The concentration of hexachlorobutadiene in fresh waters should not exceed 0.1 µg/L.

The concentration of hexachlorobutadiene in marine waters should not exceed 0.3 µg/L.

Hexachlorobutadiene (hexachloro-1,3-butadiene) is a by-product of the production of chlorinated hydrocarbons. It is used as a solvent in chemical industries and as a heat transfer fluid in electrical transformers (CCREM 1991). The presence of hexachloro-butadiene in the environment results from anthropogenic sources such as volatilisation and solubilisation from industrial wastes. Sorption to sediments is considered to be an important mechanism for the removal of hexachlorobutadiene from the water column (CCREM 1991). Little information is available regarding biodegradation.

Hexachlorobutadiene is acutely toxic to freshwater fish and invertebrates over a relative narrow range of 90–326 µg/L (USEPA 1980o). Chronic toxicity to the fathead minnow occurred at 9.3 µg/L. Bioconcentration factors for aquatic animals ranged from 29 to 7,000 (USEPA 1980o; Ahmad et al. 1984). The recommended guideline for freshwater species is adopted from CCREM (1991), and is based on the known mean chronic effect concentration of 9.3 µg/L for fish with an application factor of 0.01, which is applied because there is insufficient information to firmly establish a safe

concentration. Acute toxicity to saltwater life occurred at concentrations of 32 µg/L (USEPA 1986). No chronic toxicity data for saltwater animals are available.

Hexachlorocyclopentadiene
There are insufficient data available to recommend a guideline for hexachlorocyclo-pentadiene in fresh and marine waters.

Hexachlorocyclopentadiene is a chemical intermediate used in the production of organochlorine pesticides (e.g. aldrin, chlordane, heptachlor, endrin, endosulfan) and in fire-retardant substances (CCREM 1991). It can enter the environment from industrial discharges and emissions from pesticides. Since hexachlorocyclo-pentadiene is highly photoreactive, photolysis is expected to be the primary removal mechanism in the aquatic environment; however, sorption and bioaccumulation may also play a significant role (CCREM 1991).

According to USEPA (1986), acute and chronic toxicity to freshwater aquatic life occurred at concentrations as low as 7.0 µg/L and 5.2 µg/L respectively. A bioconcentration factor for freshwater fish of less than 11 was reported (Spehar et al. 1979). Acute toxicity to saltwater aquatic life occurred at 7.0 µg/L. No data are available concerning chronic toxicity to sensitive saltwater animals.

Halogenated ethers
There are insufficient data available to recommend a guideline for halogenated ethers in fresh and marine waters.

Halogenated ethers (haloethers) are used in industrial organic synthesis, textile manufacture, pesticide manufacture, and as solvents for polymerisation reactions (USEPA 1980p, 1980q). Chloroethers appear to be the most important haloethers used commercially (USEPA 1980p). Haloethers may enter the aquatic environment in the discharge from industrial and manufacturing processes (CCREM 1991). The fate of these compounds in the aquatic environment is not well understood, and little information is available on either biodegradation or bioaccumulation.

The only toxicity data for haloethers other than chloroalkyl ethers are for 4-bromo-phenylphenyl ether. The chronic LOEL concentration for 4-bromophenylphenyl ether in fresh water is 12,000 µg/L (USEPA 1980p), whereas the acute LOEL concentration for chloroalkyl ethers in fresh water is 238,000 µg/L (USEPA 1980q).

Isophorone
There are insufficient data available to recommend a guideline for isophorone in fresh and marine waters.

Isophorone (3,5,5-trimethyl-2-cyclohexen-1-one) is an industrial chemical used as a solvent or co-solvent for finishes, lacquers, resins, pesticides, fats, oils and gums (USEPA 1980r). Bio-oxidation of isophorone in domestic wastewater reached 42% after twenty days; in synthetic salt water, 90% was oxidised during the same time (USEPA 1980r). Nevertheless, little or no information is available concerning bioconcentration and the fate of isophorane under environmental conditions.

Isophorane has been reported to be toxic to aquatic life, particularly saltwater invertebrates (USEPA 1980r). USEPA (1986) reported acute LOEL for freshwater and marine aquatic life of 117,000 µg/L and 12,900 µg/L respectively.

Monocyclic aromatic compounds

Benzene

The concentration of benzene in fresh and marine waters should not exceed 300 µg/L.

Benzene is used as an intermediate in chemical and pharmaceutical industries for the preparation of styrene, cyclohexane, detergents and pesticides. It is also used as a thinner, degreasing and cleaning agent; a solvent in the rubber industry; and as a fuel additive (CCREM 1991; USEPA 1980s). Benzene is released into the aquatic environment from spills and release during manufacturing and other uses and from combustion of fuel (CCREM 1991). Volatilisation is expected to play a significant role in the removal of benzene from the waterbody, especially from turbulent, well-mixed waters. Some species of bacteria are able to use benzene as a sole source of carbon (Gibson 1976); however, it has been suggested that benzene is resistant to biodegradation in dilute aqueous systems (Helfgott et al. 1977).

The acute toxicity of benzene to six freshwater fish species has been tested (USEPA 1980s), with LC_{50} values ranging upward from 5,300 µg/L for rainbow trout (De Grave et al. 1980). Fish species appeared to be more sensitive than invertebrates. Acute toxicity concentrations for two *Daphnia* species were 380,000 µg/L and 300,000 µg/L (USEPA 1986). Saltwater invertebrates also appear to be less sensitive than saltwater fish (USEPA 1980s).

No definite chronic toxicity concentrations are available regarding freshwater and saltwater aquatic life. The recommended guideline is based on the guideline level of CCREM (1991), using the acute concentration for the most sensitive species with an safety factor of 0.05.

Chlorinated benzenes

The concentration of chlorinated benzenes in fresh waters should not exceed the concentrations listed in Table 2.8*.*

There are insufficient data available to recommend a guideline for chlorinated benzenes in marine waters.

Chlorinated benzenes (chlorobenzenes) are used as industrial solvents, pesticides, dielectric fluids, deodorants and chemical intermediates (CCREM 1991), and can enter the aquatic environment as a result of their use. They are prevalent in both solid and liquid industrial effluents and in atmospheric discharges. Chlorobenzenes comprise twelve different chlorinated isomers. The dynamics of the individual chlorobenzene isomers will depend on their individual physical and chemical properties. Chlorobenzenes are usually relatively resistant to abiotic and biotic degradation and they tend to persist in the environment. The major removal mechanisms for chlorobenzenes from the water column are considered to be sorption, volatilisation and bioaccumulation (CCREM 1991).

Table 2.8 Recommended guidelines for chlorinated benzenes in fresh waters

Chlorinated benzene	Guideline (µg/L)
Monochlorobenzene	15.0
1,2-dichlorobenzene	2.5
1,3-dichlorobenzene	2.5
1,4-dichlorobenzene	4.0
1,2,3-trichlorobenzene	0.9

1,2,4-trichlorobenzene	0.5
1,3,5-trichlorobenzene	0.7
1,2,3,4-tetrachlorobenzene	0.1
1,2,3,5-tetrachlorobenzene	0.1
1,2,4,5-tetrachlorobenzene	0.2
Pentachlorobenzene	0.03
Hexachlorobenzene	0.007

Source: adapted from CCREM (1991)

There is a direct relationship between toxicity to fish, invertebrates and plant species and the degree of chlorination of benzene. Monochlorobenzene is the least toxic chlorobenzene, with pentachlorobenzene being the most toxic (USEPA 1980t, 1980u). The real toxicity of hexachlorobenzene is difficult to determine because of the low solubility of this substance.

No marked difference in the sensitivity between freshwater fish and invertebrate species was evident for chlorobenzene. The guidelines for freshwater aquatic life (Table 2.8) were adapted from CCREM (1991), and are based on acute fish and early life stage chronic data. According to USEPA (1986), available data for toxicity of chlorobenzenes to saltwater animals occurred at concentrations as low as 160 µg/L and 129 µg/L respectively. There were insufficient data to establish numerical limits for the different chlorobenzenes in salt water (USEPA 1986).

Chlorinated phenols
The concentration of chlorinated phenols in fresh and marine waters should not exceed the concentrations listened in Table 2.9*.*

Chlorinated phenols (chlorophenols) are used as disinfectants, biocides, preservatives, dyes, pesticides and industrial and medical chemicals (CCREM 1991). They are released into the aquatic environment through industrial effluents from pulp and paper operations and wood treatment plants, and in agricultural runoff. Various chlorophenol compounds are also produced as intermediate metabolites in the microbial breakdown of pesticides (e.g. lindane, 2,4-D). The environmental behaviour of individual chlorophenol compounds is related to their physical and chemical properties. Depending on their degree of chlorination and their molecular weight, photolysis, sorption, biodegradation and bioaccumulation account for their environmental fate.

Generally, the toxicity of chlorophenols increases with increasing chlorine substitution, but individual compounds vary widely in toxicity (CCREM 1991). USEPA (1980v) concluded that a single numerical limit for all chlorophenols was inappropriate because of the wide variability in toxicity. Recommended guidelines are provided in Table 2.9.

Chlorophenols have also been shown to impair the flavour of the edible portions of fish, sometimes at concentrations lower than those at which toxicity occurs to

Table 2.9 Recommended guidelines for chlorinated phenols in fresh and marine waters

Chlorinated phenol	Fresh water guideline (µg/L)	Salt water guideline (µg/L)
Monochlorophenol	7.0	–
2,4-dichlorophenol	0.2	–

Chlorinated phenol	Fresh water guideline (µg/L)	Salt water guideline (µg/L)
Trichlorophenol (total)	18.0	–
2,4,5-trichlorophenol	–	8.0
Tetrachlorophenol	1.0	–
Pentachlorophenol	0.05	0.2

– Insufficient data
Sources: CCREM (1991), USEPA (1986, 1987d)

aquatic organisms. A guideline based on tainting threshold concentrations was applied for 2,4-dichlorophenol (the most critical isomer in terms of flavour impairment) and monochlorophenol in fresh water. The guidelines for the other chlorophenols are based on toxicological data (CCREM 1991; USEPA 1980w, 1987d). Little information is available concerning toxicity of chlorophenols in salt water, and only two guidelines can be recommended for saltwater aquatic life (Table 2.9).

Few bioaccumulation factors are available for chlorophenols. USEPA (1980x) reported a bioconcentration factor of 214 for 2-chlorophenol in bluegill; a factor of 130 was estimated for 2,4-dichlorophenol (USEPA 1980y); factors up to 1,000 were reported for pentachlorophenol (USEPA 1980w); and bioaccumulation factors of 47–71 were determined in inland silverside for 2,4,5-trichlorophenol (USEPA 1987d). For pentachlorophenol, the freshwater acute toxicity is 4.4 µg/L for larval carp (USEPA 1987d) which, with an application factor of 0.01, gives a figure of 0.05 µg/L. For marine waters, the lowest acute toxicity figure is 23 µg/L, which gives a safe level of 0.2 µg/L.

2,4-dimethylphenol
There are insufficient data available to recommend a guideline for 2,4-dimethylphenol in fresh and marine waters.

2,4-dimethylphenol (2,3-xylenol) is a naturally occurring substituted phenol derived from fractions of petroleum and coal tar distillation (USEPA 1980z). The available data for freshwater aquatic life indicate that acute toxicity occurs at concentrations as low as 2120 µg/L (USEPA 1986). No data are available concerning the chronic toxicity of pentachlorophenol to sensitive freshwater animals. No saltwater organisms have been tested with 2,4-dimethyl-phenol and therefore no toxicity data are available for saltwater species (USEPA 1986).

Dinitrotoluenes
There are insufficient data available to recommend a guideline for dinitrotoluenes in fresh and marine waters.

Dinitrotoluenes are used in the production of explosives, in the manufacture and formulation of urethane polymers and surface coatings, and in the preparation of dyes and organic chemicals (USEPA 1980aa; Verschueren 1983). They may enter the aquatic environment through discharges into waters by manufacturing industries that produce dyes, isocyanates, polyurethanes and munitions (CCREM 1991).

There are six possible isomers of dinitrotoluene; however, environmental aquatic data are available only for 2,4-dinitrotoluene and 2,6-dinitrotoluene. Available data for freshwater organisms indicate that 2,3-dinitrotoluene is two orders of magnitude more toxic to invertebrates and fish than is 2,4-dinitrotoluene. USEPA (1980aa) reported that acute toxicity concentrations in fresh water ranged

within 330–660 µg/L for 2,3-dinitrotoluene and 31,000–35,000 µg/L for 2,4-dinitrotoluene. A chronic value of 116 µg/L for 2,3-dinitrotoluene was derived from an embryo-larval test with fathead minnow (USEPA 1980aa).

Only acute toxicity data are available for saltwater species. The ninety-six hour LC_{50} for the exposure of *Mysidopsis bahia* to 2,3-dinitrotoluene was 590 µg/L (USEPA 1980aa). No measured bioaccumulation data are available for dinitrotoluene.

Ethylbenzene
The concentration of ethylbenzene in fresh waters should not exceed 140 µg/L.

There are insufficient data available to recommend a guideline for ethylbenzene in marine waters.

Ethylbenzene (phenylethane) has a broad environmental distribution due to its widespread use in commercial products and is present in various petroleum combustion processes (USEPA 1980ab). Ethylbenzene may be released to the environment through emission to the atmosphere and discharge into a waterbody during production and use (CCREM 1991). Little is known about the environmental fate of ethylbenzene, but volatilisation is probably its primary fate. Some species of bacteria have been found to be able to utilise ethylbenzene as a sole source of carbon (Claus & Walkner 1964; Gibson et al. 1973).

Although no information was found on bioaccumulation in natural systems, some accumulation may be expected based on its octanol–water partition coefficient (CCREM 1991). USEPA (1980ab) estimated a bioaccumulation factor of 1,400. According to CCREM (1991), the most sensitive freshwater fish is rainbow trout, with acute toxicity occurring at a concentration of 14,000 µg/L (Johnson & Finley 1980). The available data for saltwater aquatic life indicate that acute toxicity occurs at concentrations as low as 430 µg/L (USEPA 1986). No data are available regarding the chronic toxicity in fresh and salt water. CCREM (1991) recommended a tentative guideline of 700 µg/L for ethylbenzene in fresh water, based on the acute toxicity of rainbow trout and using a safety factor of 0.05. However, given that ethylbenzene is bioaccumulated, the use of an application factor of 0.01 is more appropriate.

Nitrobenzenes
There are insufficient data available to recommend a guideline for nitrobenzenes in fresh and marine waters.

Nitrobenzenes are used in the manufacture of rubber, photographic chemicals, and the production of aniline and dyestuffs (CCREM 1991), and little is known about their fate in the aquatic environment. Because of their relatively low octanol–water partition coefficient, nitrobenzenes are not expected to bioaccumulate to an appreciable extent (CCREM 1991). Acute toxicity concentrations in fresh water for *Daphnia magna*, the bluegill and fathead minnow were 27,000 µg/L, 43,000 µg/L and 117,000 µg/L respectively (LeBlanc 1980; Buccafusco et al. 1981; Holcombe et al. 1984). Acute toxicity to saltwater aquatic life occurs at concentrations as low as 6,680 µg/L (USEPA 1980ac). No data are available regarding chronic toxicity of nitrobenzene to freshwater and saltwater organisms (USEPA 1986).

Nitrophenols

There are insufficient data available to recommend a guideline for nitrophenols in fresh and marine waters.

Nitrophenols are mono, di and trivalent derivatives of phenol and are used as dyes, pigments, pharmaceuticals, chemical intermediates, explosives and fungicides (CCREM 1991). Their major routes of entry to the aquatic environment are through the industrial effluents of production plants and chemical firms where these
compounds are used as intermediates (CCREM 1991). Microbial degradation or photodegradation of compounds containing some form of nitrophenol may also release nitrophenols to the environment (USEPA 1980ad). Little information is available on the fate of nitrophenols in the aquatic environment. The few studies available on the biodegradation of nitrophenols by natural communities of micro-organisms indicate that nitrophenols appear to be more resistant to degradation than other phenols (CCREM 1991). Nitrophenols have low octanol–water partition coefficients and, therefore, bioaccumulation in most aquatic organisms is not expected to be significant (Pearce & Simpkins 1968).

The five nitrophenols for which acute toxicity data in fresh water are available are 2,4-dinitro-6-methylphenol, 2,4-dinitrophenol, 4-nitrophenol, 2,4,6-trinitrophenol and 2-nitrophenol (listed in decreasing order of toxicity). Acute LC_{50}s ranged upwards from 230 µg/L for bluegill exposed to 2,4-dinitro-6-methylphenol (USEPA 1980ad, 1986). The available acute toxicity data for saltwater species concerning 4-nitrophenol, 2,4-dinitrophenol and 2,4,6-trinitrophenol indicate that toxicity occurs at concentrations as low as 4,850 µg/L (USEPA 1986, 1980ad). No data are available regarding chronic toxicity in freshwater and saltwater aquatic life (USEPA 1986).

Phenol

The concentration of phenol in fresh and marine waters should not exceed 50 µg/L.

Phenol is a large-volume industrial chemical produced as an intermediate for the preparation of other chemicals (USEPA 1980ae). Phenol or phenolic wastes are also produced during coking of coal, distillation of wood and in oil refineries (USEPA 1980ae). Photo-oxidation, oxidation and microbial degradation are probably the major fate of phenol in the aquatic environment (CCREM 1991). Several laboratory studies have demonstrated the use of phenol as the sole carbon source for a number of micro-organisms. In addition, phenol has been shown to degrade in natural waters (CCREM 1991; Visser et al. 1977). Phenol does not accumulate to any significant extent in exposed organisms, and is rapidly eliminated on termination of exposure (Swift 1979).

USEPA (1980ae) compiled data of phenol toxicity in fresh water. The available data indicate that acute and chronic toxicity occurs at concentrations as low as 10,200 µg/L and 2,560 µg/L respectively. Acute toxicity to saltwater animals occurs at concentrations of 5,800 µg/L (USEPA 1986). The recommended guideline is based on the acute toxicity of phenol to *Daphnia* (Nicholson 1984) with an safety factor of 0.05. USEPA (1986) did not establish a numerical limit for phenol. CCREM (1991) established a guideline for freshwater protection of 1 µg/L for total monohydric and dihydric phenols. Nicholson (1984) recommended a criterion of 100 µg/L, based on acute toxicity of phenol to *Daphnia* with a safety factor of 0.1.

Toluene
The concentration of toluene in fresh waters should not exceed 300 µg/L.

There are insufficient data available to recommend a guideline for toluene in marine waters.

Toluene is used in the production of chemicals, as a gasoline component and as a solvent in formulations of rubber cements, paints, inks and pesticides (CCREM 1991). Sources of toluene in the aquatic environment include industrial effluents,
spills, discharges of oil and gas from boats, municipal waste treatment facilities, agricultural runoff and atmospheric deposition (USEPA 1980af). Juttner and Henatsch (1986) have shown that toluene can also be generated naturally. Little information is available on the biodegradation of toluene in natural systems. Toluene may be bioaccumulated to some extent under continuous exposure (CCREM 1991).

Concentrations of toluene causing acute toxicity to freshwater fish species ranged from 54,600 µg/L for coho salmon to 240,000 µg/L for channel catfish (Moles et al. 1981; Johnson & Finley 1980). Chronic toxicity of toluene to aquatic invertebrates may occur at concentrations below 4,300 µg/L (CCREM 1991). Acute toxicity to fish in salt water occurred at 6,300 µg/L. A chronic value of 2,200 µg/L has been obtained from an embryo-larval test with the sheepshead minnow (USEPA 1980af). USEPA (1986) did not recommend a numerical limit for toluene because of insufficient available data. CCREM (1991) recommended a guideline of 300 µg/L for the protection of freshwater aquatic life based on the LC_{50} for coho salmon with a safety factor of 0.05.

Nitrosamines
There are insufficient data available to recommend guidelines for nitrosamines in fresh and marine waters.

Nitrosamines have potential uses as solvents in the fibre and plastic industries, antioxidants in fuels, additions to fertilisers, softeners for copolymers, insect repellants, insecticides, fungicides and bactericides (CCREM 1991). Nitrosamines may be formed in the environment via interaction of nitrosating agents and secondary amines. Their formation is pH-dependent, with a maximum in the range of pH 3–4; however, dimethyl nitrosamine has been formed from dimethylamine and nitrite at pH values as high as 7.7 in samples of soil, sewage and lake water (CCREM 1991).

USEPA (1980ag) tested N-nitrosodiphenylamine on three fish and invertebrate species. Acute toxicity in fresh water occurred at concentrations as low as 5,850 µg/L. Feeding studies with dimethyl nitrosamine and rainbow trout demonstrated a dose-related carcinogenic response (USEPA 1980ag). The only data available for nitrosamines in marine water systems is the acute toxic effect of N-nitrosodiphenyl-amine on the mummichog, with a concentration of 3,300,000 µg/L (USEPA 1980ag).

Pesticides
Pesticides are applied directly to the environment and can find their way into natural waters via accidental spillage, spray drift, agricultural runoff after rain, or via atmospheric deposition (e.g. Ritter 1990). Other sources of pollution are accidental releases. The main pesticide groups discussed in this section are the organochlorine and organophosphate insecticides.

The developed guidelines given in Table 2.10 are primarily based on those proposed by Nicholson (1984), USEPA (1986) and CCREM (1991). Where no guidelines were available, the lowest acute or

chronic toxicity level was reduced by factors of 0.001 and 0.01 respectively to establish a guideline. More-specific information about pesticides in the aquatic environment is given by Nicholson (1984). A summary of the acute toxicity of thirteen organic pesticides to three Australian freshwater organisms is provided by O'Brien (1991).

Table 2.10 Recommended maximum concentrations for pesticides in unfiltered water samples

Type of pesticide	Pesticide	Fresh water guidelines (ng/L)	Salt water guideline (ng/L)
Organochlorines	Aldrin	10	10
	Chlordane	4	4
	DDE	14	14
	DDT	1	1
	Dieldrin	2	2
	Endosulfan	10	10
	Endrin	3	3
	Heptachlor	10	10
	Lindane	3	3
	Methoxychlor	40	40
	Mirex	1	1
	Toxaphene	8	8
Organophosphates	Chlorpyrifos	1	1
	Demeton	100	100
	Guthion (Azinphos-methyl)	10	10
	Malathion	70	100
	Parathion	4	4
Other pesticides	Acrolein	200	200

Sources: adapted from Nicholson (1984), USEPA (1986), CCREM (1991)

Organochlorine pesticides
The concentration of organochlorines in fresh and marine waters should not exceed the concentrations listed in Table 2.10*.*

Most organochlorine pesticides have low water-solubility but high chemical and biological stability. The persistence of organochlorine in the environment results in a greater chance of contact with non-target organisms. Because of their fat solubility they tend to accumulate in the fat tissue of organisms. In addition to accumulation through direct contact with water, organochlorines can also accumulate along the food chain. The persistence and accumulation of these substances in the environment has resulted in their use being curtailed or banned in Australia. More specific information about the current use of the different organochlorines is given in Chapter 4.

The longevity of organochlorine pesticides is shown by a recent study of the Ovens and King rivers where, despite the fact that DDT and dieldrin were banned in Victoria in 1987, DDT, dieldrin, heptachlor, aldrin and lindane concentrations in water samples exceeded the safe drinking water and ecosystem protection values on some occasions (VicEPA 1990).

The acute toxicity of organochlorine pesticides to aquatic organisms has been generally well documented (Nicholson 1984; USEPA 1986; CCREM 1991). Lethal concentrations depend markedly on the organochlorine compounds and the organisms tested.

Endosulfan is the main organochlorine pesticide still in use in Australia, most of the others having now been banned. Most of the endosulfan is used in the cotton-growing areas of New South Wales, and considerable concern has been expressed about the contamination of surface waters and the effects on aquatic organisms in these areas (Napier 1989; Novak 1989). CCREM (1991) recommended a maximum concentration of 20 ng/L, based on the acute toxicity of endosulfan for freshwater fish (generally the most sensitive species) of 0.34 μg/L and an application factor of 0.05. The lowest acute toxicity of endosulfan for a freshwater fish (bony bream, *Nematolosa erebi,* from the Mehi River) was 0.2 μg/L (Chapman J., NSWEPA, pers. comm., June 1992). Applying an application factor of 0.05 produces a level of 10 ng/L, which is the value recommended.

Organophosphate pesticides
The concentration of organophosphates in fresh and marine waters should not exceed the concentrations listened in Table 2.10*.*

In contrast to the organochlorine pesticides, organophosphate pesticides tend to be unstable; they generally degrade readily and do not accumulate. The degradation to non-toxic substances occurs usually within a few days or weeks (Hart 1974), and there is little evidence that these compounds can accumulate through the food chain. They also tend to be less toxic to fish than organochlorines (Nicholson 1984).

Acrolein
The concentration of acrolein in fresh and marine waters should not exceed 200 ng/L.

Acrolein has a substantially greater acute toxicity to fish than many other herbicides used for aquatic weed control (CCREM 1991). The lowest acute toxic concentration to freshwater aquatic organisms reported by the USEPA (1986) was 68 μg/L. There was little apparent difference between acute and chronic toxicity in fresh water. Chronic toxicity for the fathead minnow was observed at 21 μg/L (USEPA 1986). The available data for acute toxicity in salt water indicate that aquatic life is affected at concentrations of 55 μg/L; however, lower concentrations may affect species that are more sensitive (USEPA 1986). No data are available regarding chronic toxicity of acrolein for sensitive saltwater species. The recommended guideline is based on the chronic toxicity to the fathead minnow with a safety factor of 0.01.

Phthalate esters
The recommended guidelines for phthalate esters for the protection of freshwater aquatic life are:

- di-butylphthalate not to exceed 4 μg/L
- di(2-ethylhexyl)phthalate not to exceed 0.6 μg/L
- other phthalate esters not to exceed 0.2 μg/L.

There are insufficient data available to recommend a guideline for phthalate esters in marine waters.

Phthalate esters represent a large family of chemicals widely used as plasticisers, primarily in the production of polyvinyl chloride (PVC) resins (USEPA 1980ah). Other applications are found in

cosmetics, rubbing alcohol, insect repellant, insecticides and tablet coatings (CCREM 1991). Although phthalate esters are insoluble in pure water, they may be transported in the aquatic environment in solubilised forms by fulvic and humic acids. This solubilisation has been found to be pH-dependent (Ogner & Schnitzer 1970; Matsuda & Schnitzer 1971). Depending upon specific conditions in aquatic ecosystems, bioaccumulation and biodegradation will also be significant (CCREM 1991; Pierce et al. 1980).

Phthalate esters are a diverse group of organic compounds and toxicity in aquatic ecosystems varies with the ester tested. The insolubility of some phthalate esters in water makes it difficult to determine the actual concentrations used in toxicological tests (Pierce et al. 1980). Appropriate acute toxicity data for freshwater organisms are available for five esters: butylbenzyl phthalate, diethyl phthalate, dimethyl phthalate, di-n-butyl phthalate and di-2-ethylhexylphthalate (USEPA 1980ah). The sensitivity of fish and invertebrates was generally similar, with most values exceeding 1,000 µg/L. Concentrations causing chronic toxicity in freshwater animals were as low as 3 µg/L (USEPA 1986).

Toxicity data for saltwater organisms are available for three phthalate esters (USEPA 1980ah). LC_{50} concentrations for butylbenzyl, diethyl and dimethyl phthalates for the mysid shrimp were 9,630 µg/L, 7,590 µg/L and 7,000 µg/L respectively. No data are available concerning chronic toxicity of phthalate esters to sensitive aquatic life (USEPA 1986). USEPA (1986) did not recommend a guideline for phthalate esters because their minimum data base requirements were not met. CCREM (1991) recommended guidelines in fresh water of 4 µg/L for di-n-butylphthalate, 0.6 µg/L for di(2-ethylhexyl)phthalate and 0.2 µg/L for other phthalate esters.

Polyaromatic compounds

Chlorinated naphthalenes

There are insufficient data available to recommend a guideline for chlorinated naphthalenes in fresh and marine waters.

Chlorinated naphthalenes consist of two aromatic carbon rings with any or all of the eight hydrogen atoms replaced with chlorine. The commercial products are usually mixtures with various degrees of chlorination. Mixtures of trichloronaphthalenes and tetrachloronaphthalenes comprise the bulk of commercial use as paper impregnates in automobile capacitors. Other mixtures are used as oil additives for engine cleaning and in fabric dying (USEPA 1980ai).

Limited data exist on the toxicity of these compounds for aquatic organisms. The available data indicate that acute toxicity occurs at concentrations of 1,600 µg/L for *Daphnia magna* exposed to 1-chloronaphthalene (USEPA 1980ai). Acute toxicity to saltwater aquatic life occurred at concentrations as low as 7.5 µg/L (USEPA 1986). No data are available concerning the chronic toxicity of chlorinated naphthalenes (USEPA 1986).

Polychlorinated biphenyls

The concentration of polychlorinated biphenyls in fresh waters should not exceed 0.001 µg/L.

The concentration of polychlorinated biphenyls in marine waters should not exceed 0.004 µg/L.

Polychlorinated biphenyls (PCBs) are widely used in industrial applications because they have excellent thermal stability, strong resistance to both acid and base hydrolysis, general inertness, excellent dielectric properties and non-flammability (CCREM 1991). Sources of PCBs entering the

environment are the open burning or incomplete combustion of PCBs or wastes containing PCBs (Klein & Weisgerber 1976).

There are 209 theoretically possible chlorinated biphenyls (Cook 1972). Individual PCBs vary widely in their physical and chemical properties according to the degree and the position of the chlorination (CCREM 1991). Most PCBs are slightly soluble in water and the solubility decreases with the chlorine content (CCREM 1991). Sorption to sediments is probably the predominant mechanism for removing PCBs from the water column (USEPA 1979b). PCBs are soluble in the lipids of biological systems; bioconcentration factors of 200,000 and greater have been reported in the fathead minnow (Neely 1977).

The acute toxicity of PCBs appears to be similar for freshwater fish and invertebrates. Concentrations causing acute toxicity to freshwater invertebrates ranged from 10 µg/L to 400 µg/L (CCREM 1991). Chronic toxicity to aquatic freshwater life occurs at concentrations as low as 0.2 µg/L (USEPA 1980ak). Acute toxicity concentrations for saltwater invertebrates ranged from 12.5 µg/L upwards (USEPA 1980ak).

USEPA (1986) recommended that criteria for PCBs in fresh and salt water use concentrations of 0.014 µg/L and 0.030 µg/L respectively. According to USEPA (1986), these values are probably too high because they are based on bioconcentration factors measured in laboratory studies. CCREM (1991) recommended a maximum concentration of 0.001 µg/L for PCBs in fresh water. Nicholson (1984) recommended a criterion of 0.004 µg/L for PCBs to protect aquatic life.

Polychlorinated dibenzo-p-dioxins
*There are insufficient data available to recommend a guideline for polychlorinated dibenzo-*p*-dioxins in fresh and marine waters.*

Polychlorinated dibenzo-*p*-dioxins (PCDDs) result from various synthetic or pyrolytic reactions. PCDDs are known to exist in a variety of chemicals, including pesticides, the wood preservative pentachlorophenol, and chlorinated phenols (CCREM 1991). They can be formed by combustion processes, including the burning of fossil fuels, wood and garbage. PCDDs are a group of chemicals composed of seventy-five chemically-related compounds.

In general, little information is available on the fate of PCDDs in the aquatic environment. Most of the available data refer to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD). TCDD exhibits a delayed biological response in many species, and is highly
lethal at low concentrations to aquatic organisms (USEPA 1980al). The available information indicates that acute concentrations for some freshwater animals species exposed to TCDD are greater than 1.0 µg/L. Chronic concentrations are less than 0.01 µg/L, with a chronic concentration for rainbow trout of less than 0.001 µg/L (USEPA 1986). Measured bioaccumulation factors for TCDD ranged from 3,000 to 900,000 (USEPA 1986).

Polycyclic aromatic hydrocarbons
The concentration of polycyclic aromatic hydrocarbons in fresh and marine waters should not exceed 3 µg/L.

Polycyclic aromatic hydrocarbons (PAHs) are formed by incomplete combustion of organic material, diagenesis and biosynthesis. Natural sources include forest fires, volcanic activity, diagenesis and,

possibly, production by some plants and micro-organisms; however, a significant fraction of PAHs is the result of anthropogenic combustion processes (CCREM 1991). Atmospheric deposition is believed to be a significant route of entry into the aquatic environment, but materials containing PAHs may also directly enter the water system via release of crude oil and petroleum products (CCREM 1991).

Concentrations of PAHs in aquatic ecosystems are generally highest in sediments, intermediate in aquatic biota and lowest in the water column (Neff 1979; NRCC 1983). In field studies, sorption to suspended particles and bed sediments was found to be the primary removal mechanism for high-molecular weight PAHs, whereas volatilisation and transport were the primary mechanisms for low-molecular weight PAHs (Herbes 1976; Knap & Williams 1982; Readman et al. 1982). Mixed microbial population in sediment water systems may degrade some PAHs, with degradation progressively decreasing with increasing molecular weight (CCREM 1991).

USEPA prepared documents on ambient water quality guidelines for naphthalene (1980am), fluoranthene (1980an), phenanthrene (1988c) and polynuclear aromatic hydrocarbons (1980ao); however, except for phenanthrene, insufficient data were available to recommend numerical limits. The acute toxicities of fluoranthene and naphthalene to freshwater aquatic life were around 4,000 µg/L and 2,300 µg/L respectively (USEPA 1980an, 1980am). Acute toxicity to saltwater aquatic life occurs at concentrations as low as 40 µg/L for fluoranthene. USEPA (1988c) developed guidelines (four-day average) for phenanthrene in fresh and salt water, resulting in concentrations of 6.3 µg/L and 4.6 µg/L respectively. Benzo(*a*)pyrene is highly lipophilic, and bioconcentration factors ranged from 930 in the mosquito fish to 134,240 in *Daphnia pulex* (Lu et al. 1977). Mixtures of polycyclic aromatic hydrocarbons have been found to cause tumours in fish (IJC 1983).

Nicholson (1984) recommended a criterion of 3 µg/L for PAHs to protect aquatic life. He based this on the acute toxicity (twenty-four hour LC_{50} of 316 µg/L) of trimethyl-naphthalene to the copepod, *Eurytemora affinis*, reduced by an application factor of 0.01 because PAHs can accumulate. However, Nicholson (1984) did note that this value '. . . should be used with caution because of the limited toxicity data on which it is based'. CCREM (1991) suggested that there were insufficient data to recommend a guideline for PAHs. The guideline recommended for Australia is 3 µg/L.

2.5 Water quality guidelines for the production of edible fish, crustacea and shellfish

Guidelines for the protection of edible fish, crustacea and shellfish may be divided into two categories: those for the protection of the aquatic organisms and those for the protection of the human consumer.

2.5.1 Guidelines for the protection of fish, crustacea and shellfish

Water quality guidelines are necessary to determine optimum environmental conditions for the growth and reproduction of edible fish, crustacea and shellfish. The guidelines for the protection of these species are generally those discussed in Section 2.1.

2.5.2 Guidelines for the protection of the human consumer

Toxicants

Minimal risk concentrations in the water are required to protect consumers from toxicants that may accumulate in the tissue of fish, crustacea and shellfish, either directly from the water or by biomagnification in the food chain. If the guidelines for the protection of aquatic ecosystems given in

Section 2.1 do not protect the human consumer, lower concentration levels of the toxicant concerned are listed in Table 2.11. If a toxicant is not listed in Table 2.11, the value given for the protection of aquatic ecosystems will also protect the human consumer.

Bacteria

In addition to toxicants, bacterial guidelines may also be important, especially if the organisms are eaten raw; for example, shellfish consumption has been implicated in transmitting infectious hepatitis in humans. Guidelines for bacteria are listed under biological indicators in Table 2.11. Biotoxins that can cause poisoning of the consumer are also listed in Table 2.11.

Tainting substances

Deterioration of the palatability of fish, crustacea and shellfish could have serious economic impacts on the fishing and harvesting industries. The chemical compounds found to cause tainting of the flesh of fish and other aquatic organisms are summarised in Table 2.12. The values given provide information on possible sources of tainting and the concentrations at which tainting will occur. The concentrations quoted should not be used as guideline levels for ecosystem protection.

2.6 Guidelines for the protection of water-associated wildlife

In this document, wildlife is defined as all species of vertebrates other than fish and humans that depend on aquatic environments for drinking water, food or habitat requirements.

Table 2.11 Guidelines for the protection of human consumers of fish and other aquatic organisms

Type of indicator	Indicator	Guideline (µg/L, if not otherwise stated)
Biological Indicators	Algae	No guideline. Toxins may be present in cyanobacteria and may be accumulated in other aquatic organisms
	Biotoxins:	
	Gonyaulax shellfish toxins	< 0.8 µg/g shellfish
	Ciguatera-like toxins	< 20 mouse units/100 g shellfish
	Faecal coliforms	The median faecal coliform bacterial concentration should not exceed 14 MPN/100 ml, with no more than 10% of the samples exceeding 43 MPN/100 ml.
Toxicants	Arsenic	0.02*
	Beryllium	0.1*
	Manganese	100.0
	Nickel	100.0
	Acrylonitrile	0.7*
	Benzidine	0.0005*
	Dichlorobenzidine	0.02*
	Diphenylhydrazine	0.6*
	Halogenated aliphatic compounds	–
	Chlorinated ethanes:	–
	1,2-dichloroethane	240.0*
	1,1,2-trichloroethane	40.0*

Type of indicator	Indicator	Guideline (µg/L, if not otherwise stated)
	1,1,2,2-tetrachloroethane	11.0*
	Hexachloroethane	9.0
	Chlorinated ethylenes	–
	Chloroethylene (vinyl chloride)	530.0*
	1,1-dichloroethylene	2.0*
	Trichloroethylene	80.0*
	Tetrachloroethylene	9.0*
	Halogenated methanes	–
	Carbon tetrachloride	7.0*
	Chloroform	16.0
	Other halogenated methanes	16.0*
	Halogenated ethers	–
	bis(chloromethyl) ether	0.002*
	bis (2-chloroethyl) ether	1.0*
	Monocyclic aromatic compounds	–
	Benzene	40.0*
	Chlorinated benzene	–
	1,2,4,5-tetrachlorobenzene	50.0
	Pentachlorobenzene	80.0
	Hexachlorobenzene	0.0007*
	Chlorinated phenols	–
	2,4,6-trinitrophenol	4.0*
	Dinitrotoluene	9.0*
	Nitrosamines	–
	N-nitrosodiethylamine	1.0*
	N-nitrosodimethylamine	16.0*
	N-nitrosodibuthylamine	0.6*
	N-nitrosopyrrolidine	90.0*
	N-nitrosodiphenylamine	16.0*
	Pesticides	–
	Aldrin	0.08 ng/L*
	Chlordane	0.5 ng/L*
	DDT	0.03 ng/L*
	Dieldrin	0.08 ng/L*
	Heptachlor	0.3 ng/L*
	PAH	0.03*
	2,3,7,8-tetrachlorodibenzodioxin	0.00001 ng/L*
	Radionuclides	0.4 Bq/L

MPN: Most probable number

* Potential carcinogen, risk level 1:1,000,000

Sources: adapted from USEPA (1986), NAS/NAE (1973), IWBDE (1972)

Table 2.12 Guidelines for chemical compounds in water found to cause tainting of fish flesh and other aquatic organisms

Parameter	Estimated threshold level in water (µg/L)
Acenaphthene	0.02
Acetophenone	0.5
Acrylonitrile	18.0
Copper	1.0
m-cresol	0.2
o-cresol	0.4
p-cresol	0.1
Cresylic acids (meta, para)	0.2
Chlorobenzene	0.02
n-butylmercaptan	0.06
o-sec. butylphenol	0.3
p-tert. butylphenol	0.03
o-chlorophenol	0.0001–0.015
p-chlorophenol	0.0001
2,3-dinitrophenol	0.08
2,4-dichlorophenol	0.0001-0.014
2,5-dichlorophenol	0.02
2,6-dichlorophenol	0.03
3,4-dichlorophenol	0.0003
2-methyl-4-chlorophenol	2.0
2-methyl-6-chlorophenol	0.003
3-methyl-4-chlorophenol	0.02-3
o-phenylphenol	1.0
Pentachlorophenol	0.03
Phenol	1-10
Phenols in polluted rivers	0.15-0.02
2,3,4,6-tetrachlorophenol	0.001
2,3,5-trichlorophenol	0.001
2,4,6-trichlorophenol	0.002
2,4-dimethylphenol	0.4
Dimethylamine	7.0
Diphenyloxide	0.05
B,B-dichlorodiethyl ether	0.09-1
o-dichlorobenzene	< 0.25
Ethylbenzene	0.25
Ethanethiol	0.2

Parameter	Estimated threshold level in water (µg/L)
Ethylacrylate	0.6
Formaldehyde	95.0
Gasoline	0.005
Guaicol	0.08
Kerosene	0.1
Kerosene plus kaolin	1.0
Hexachlorocyclopentadiene	0.001
Isopropylbenzene	< 0.25
Naphtha	0.1
Naphthalene	1.0
Naphthol	0.5
2-Naphthol	0.3
Nitrobenzene	0.03
a-methylstyrene	0.25
Oil, emulsifiable	> 15.0
Pyridine	5-28
Pyrocatechol	0.8-5
Pyrogallol	20-30
Quinoline	0.5-1
p-quinone	0.5
Styrene	0.25
Toluene	0.25
Outboard motor fuel as exhaust	7.2
Zinc	5.0

Source: adapted from NAS/NAE (1973)

2.6.1 Protection of drinking water for wildlife

Generally, the guidelines given for the protection of aquatic ecosystems (Section 2.1) will be sufficient to protect wildlife from detrimental effects associated with drinking contaminated water.

2.6.2 Protection of food for wildlife

Many wildlife species are predators and, therefore, very vulnerable to substances that can bioaccumulate along the food chain. In these instances, environmental levels that are safe for fish and invertebrates do not necessarily convey safety for predators or even for scavengers that consume aquatic organisms. Stringent guidelines may therefore be required for the protection of wildlife from pollutants that are able to concentrate along the food chain. Many toxicants are known to bioconcentrate (Section 2.1), but little is known regarding the effects on the predator organisms. Table 2.13 summarises some well-known toxicants that can accumulate along the food chain.

Table 2.13 Guidelines for toxicants that can accumulate along the food chain

Parameter	Guideline (concentration in food organisms, µg/L)
DDT	1.0

Parameter	Guideline (concentration in food organisms, µg/L)
PCB	0.5
Mercury	0.5

Sources: NAS/NAE (1973), USEPA (1976)

2.6.3 Protection of habitat requirements for wildlife

Generally, the guidelines given in Section 2.1 will be sufficient to protect wildlife habitat requirements.

3 Recreational water quality and aesthetics

Water-based recreational activities are highly regarded by Australians. Although Australia has about 20,000 km of coastline, much of it is inaccessible for recreational purposes, resulting in highly localised pressures on accessible coastline. The same is true for many lakes, especially those close to urban centres. Water quality guidelines are therefore necessary to protect these waters for recreational activities, such as swimming and boating, and to preserve the waters' aesthetic appeal.

Sporting activities can be divided into two categories:

- sports in which the user comes into frequent direct contact with water, either as part of the activity or accidently; for example, swimming or surfing (primary contact);
- sports that generally have less-frequent body contact with the water; for example, boating or fishing (secondary contact).

A third recreational category concerns the passive recreational use of waterbodies, mainly as pleasant places to be near or to look at (no body contact). The relevance of the different water quality guidelines to the three recreational categories is shown in Table 3.1. The detailed water quality guidelines for recreational water are summarised in Table 3.2.

Table 3.1 Water quality characteristics relevant to recreational use

Characteristics	Primary contact (e.g. swimming)	Secondary contact (e.g. boating)	Visual use (no contact)
Microbiological guidelines	x	x	–
Nuisance organisms (e.g. algae)	x	x	x
Physical and chemical guidelines:	–	–	–
Aesthetics	x	x	x
Clarity	x	x	x
Colour	x	x	x
pH	x	–	–
Temperature	x	–	–
Toxic chemicals	x	x	–
Oil, debris	x	x	x

The first section of this chapter provides a brief summary of the most important aspects of the above categories, while the second section contains details on the specific guidelines. Many of the guidelines necessary for the maintenance of certain aspects of recreational water quality (e.g. preservation of aquatic life and wildlife) are discussed in other chapters and will only be briefly mentioned here. The recommended guidelines rely on the guidelines developed by NHMRC (1990), with additional indicators included where appropriate.

Table 3.2 Summary of water quality guidelines for recreational waters

Parameter	Guideline
Microbiological	–
Primary contact*	The median bacterial content in fresh and marine waters taken over the bathing season should not exceed 150 faecal coliform organisms/100 mL or 35 enterococci organisms/100 mL. Pathogenic free-living protozoans should be absent from bodies of fresh water.**
Secondary contact*	The median value in fresh and marine waters should not exceed 1,000 faecal coliform organisms/100 mL or 230 enterococci organisms/100 mL.**
Nuisance organisms	Macrophytes, phytoplankton scums, filamentous algal mats, sewage fungus, leeches etc. should not be present in excessive amounts.* Direct contact activities should be discouraged if algal levels of 15,000–20,000 cells/mL are present, depending on the algal species. Large numbers of midges and aquatic worms should also be avoided.
Physical and chemical	–
Visual clarity & colour	To protect the aesthetic quality of a waterbody: • the natural visual clarity should not be reduced by more than 20%; • the natural hue of the water should not be changed by more than 10 points on the Munsell Scale; • the natural reflectance of the water should not be changed by more than 50%. To protect the visual clarity of waters used for swimming, the horizontal sighting of a 200 mm diameter black disc should exceed 1.6 m.
pH	The pH of the water should be within the range 5.0–9.0, assuming that the buffering capacity of the water is low near the extremes of the pH limits.
Temperature	For prolonged exposure, temperatures should be in the range of 15–35°C.
Toxic chemicals	Water containing chemicals that are either toxic or irritating to the skin or mucous membranes are unsuitable for recreation. Toxic substances should not exceed levels given for untreated drinking waters.
Surface films	Oil and petrochemicals should not be noticeable as a visible film on the water nor should they be detectable by odour.

* Refer to Section 2.3.3 for guidelines relating to nutrient concentrations necessary to limit excessive aquatic plant growth.

** Sampling frequency and maximum values are given in Section 3.2.1.

3.1 Recreational categories

3.1.1 Primary contact

Water used for primary contact activities, such as swimming, bathing and other direct water-contact sports, should be sufficiently free from faecal contamination, pathogenic organisms and other hazards (e.g. poor visibility or toxic chemicals) to protect the health and safety of the user. The general guidelines desirable for aquatic scenery are also applicable for water used for primary contact.

3.1.2 Secondary contact

Water used for secondary contact activities, such as boating and fishing, should also meet the guidelines suggested for aquatic scenery. Since there is less body contact with the water, the microbiological guidelines can generally be lower, although not in cases when shellfish might be taken from from the waterbody. To protect water-skiers from injury and boating vessels from damage, the water should be free from floating or submerged logs and stumps and excessive growth

of algae and other aquatic plants. The quality of the water should be maintained so that there is minimal alteration of the fish habitat (Chapter 2).

3.1.3 Visual use

Surface waters used for visual recreational use (no-contact activity) should not be altered in any way that reduces their ability to support aesthetically valuable flora and fauna. Such alteration may be physical, such as dredging and dam construction, or may be due to addition of wastes to the water. Visual impact of the surface waters is important; they should be free from:

- floating debris, oil, grease and other objectionable matter;
- substances that produce undesirable colour, odour, taste or foaming;
- undesirable aquatic life, such as 'algal blooms', or dense growths of attached plants or insects.

All these factors have to be considered in areas used for aquatic scenery.

3.2 Detailed water quality guidelines

3.2.1 Microbiological characteristics

Primary contact

The median bacterial content in samples of fresh or marine waters taken over the bathing season should not exceed:

- 150 faecal coliform organisms/100 mL (minimum of five samples taken at regular intervals not exceeding one month, with four out of five samples containing less than 600 organisms/100 mL);
- 35 enterococci organisms/100 mL (maximum number in any one sample: 60–100 organisms/100 mL).

Pathogenic free-living protozoans should be absent from bodies of fresh water. (It is not necessary to analyse water for these pathogens unless the temperature is greater than 24°C.)

Secondary contact

The median bacterial content in fresh and marine waters should not exceed:

- 1,000 faecal coliform organisms/100 mL (minimum of five samples taken at regular intervals not exceeding one month, with four out of five samples containing less than 4,000 organisms/100 mL);
- 230 enterococci organisms/100 mL (maximum number in any one sample 450–700 organisms/100 mL).

There is a long international experience of disease outbreaks associated with contaminated water (McNeill 1985; Cabelli 1989). Disease-causing micro-organisms (pathogens) associated with bathing areas include salmonellae, shigellae, enteropathogenic *Escherichia coli,* cysts of *Entamoeba histolytica*, parasite ova, enteroviruses and infectious hepatitis (Hart 1974; McNeill 1985). Generally, the most common types of diseases that have been associated with swimming areas are eye, ear, nose and throat infections, skin diseases and gastrointestinal disorders. McNeill (1985) has reviewed epidemiological studies associated with recreational waters.

Direct detection of pathogens is not a feasible option for routine assessment, since they occur intermittently and are difficult to recover from water. For this reason, 'indicator' micro-organisms are

generally used to assess the health risks associated with pathogens in recreational waters (Elliot & Colwell 1985). A number of organisms have been considered as indicators of health risks for swimming areas (McNeill 1985; Daly 1991).

NHMRC (1990) favours the use of faecal coliform bacteria, a sub-group of the total coliform population that are easy to measure and are present in virtually all warm-blooded animals. Faecal coliform bacteria in human faeces comprise about 97% *E. coli*, around 2% *Klebsiella*, and a further 2% *Enterobacter* and *Citrobacter* together. However, McBride et al. (1991) have documented a number of deficiencies with the use of faecal coliforms as indicator organisms of health risks in recreational waters and waters used for shellfish growing. Recent epidemiological studies have shown poorer relationships between faecal coliform densities and illness rates in bathers than are obtained using enterococci (marine waters: Cabelli 1983a, 1983b; Cabelli et al. 1982, 1983) and using either enterococci or *E. coli* (fresh waters: Dufour 1984). Further, there is now considerable evidence that faecal coliforms die off faster than pathogens under certain circumstances; therefore, they may go undetected during beach monitoring programs, resulting in the disease risks being underestimated.

New Zealand (McBride et al. 1991), Canada (CCREM 1991) and the United States (USEPA 1986) now recommend guidelines for recreational waters in terms of either enterococci or *E. coli* (or the non-faecal indicator *Pseudomonas aeruginosa*). For example, the New Zealand guidelines recommend that the median bacterial content of samples taken over the bathing season should not exceed 33 enterococci/100 mL (or 126 *E. coli*/10 mL) for fresh waters, and 35 enterococci/100 mL for marine waters (McBride et al. 1991). The guidelines recommended here are based on the levels recommended by NHMRC (1990) in terms of faecal coliforms, and those recommended by McBride et al. (1991) in terms of enterococci.

3.2.2 Nuisance organisms

Macrophytes, phytoplankton scums, filamentous algal mats, blue-green algae, sewage fungus and leeches should not be present in excessive amounts. Guidelines relating to nutrient concentrations necessary to limit excessive aquatic plant growth are given in Section 2.3.3.

Direct contact activities should be discouraged if algal levels of 15,000-20,000 cells/mL are present, depending upon the algal species. Large numbers of midges and aquatic worms should be avoided.

Biological factors that influence the recreational value of surface waters include those that endanger the health or physical comfort of people and animals, and those that render water aesthetically objectionable. In the first category are non-biting midges, phantom midges, caddis flies and mayflies, which can emerge in large numbers and cause serious nuisance to people picnicking, camping or living near the shoreline. More serious are biting insects that can cause irritation from their bites, respiratory allergic reactions or quite serious diseases. Common diseases transmitted by aquatic invertebrates are encephalitis, malaria and schistosome dermatitis (swimmer's itch).

Excessive growths of aquatic plants can also cause problems in recreational areas. Rooted and non-rooted macrophytes may obstruct the view of swimmers and obscure underwater hazards. They can also entangle swimmers and induce panic if encountered unexpectedly. If the growth is very dense, boating and fishing may also be restricted. Dislodged or free-floating plants may also drift on to beaches, decay and cause objectionable odours as well as provide breeding areas for nuisance organisms.

Algal blooms, particularly if dominated by blue-green algae (cyanobacteria), can impair the recreational values of a waterbody by reducing the clarity and by accumulating along shorelines with effects similar to those cited for macrophytes. In addition, several species of blue-green algae can produce toxic substances that may kill fish, birds and domestic animals (Shilo 1981; Codd 1990; Falconer 1990). Species of blue-green algae have also been responsible for contact dermatitis in humans and influenza-like symptoms in swimmers (Codd 1990). Primary contact activities in waters containing high levels of cyanobacteria should be discouraged. Ingestion of cyanobacterial-infested water has been associated with gastrointestinal disorders in swimmers, and lipopolysaccharides found in certain cyanobacteria have been identified as causing skin irritations, dermatitis and allergy reactions observed in swimmers using cyanobacterial-infested waters (McNeill, A., Victorian Rural Water Corporation, pers. comm., June 1992). As an interim guide, direct contact should be avoided when 15,000-20,000 cells/mL are present, depending on the algal species.

Periphyton growing on the bed of rivers and streams can also reduce the usefulness of these systems for contact recreation. Quinn (1991) recommended that to protect contact recreational areas:

> …the seasonal maximum cover of stream or river bed by periphyton as filamentous growths or mats (greater than about 3 mm thick) should not exceed 40%, and/or biomass should not exceed 100 mg chlorophyll-a/m^2.

He also called for additional research to define the level of periphyton that constitutes a nuisance.

Excessive aquatic plant growth is most often caused by high nutrient concentrations (mostly phosphorus and nitrogen) entering the waterbody. Guidelines for limitations on nutrients can be found in Section 2.3.3.

3.2.3 Physical and chemical characteristics

Visual clarity and colour
To protect the aesthetic quality of a waterbody:

- the natural visual clarity should not be reduced by more than 20%;
- the natural hue of the water should not be changed by more than 10 points on the Munsell Scale;
- the natural reflectance of the water should not be changed by more than 50%.

To protect the visual clarity of waters used for swimming, the horizontal sighting of a 200 mm diameter black disc (Secchi disc) should exceed 1.6 m.

Guidelines relating to visual clarity and colour are required for two reasons: first, to ensure that the aesthetic quality of the waterbody is maintained and that there is no obvious change in the colour or visual clarity and, second, that the visual clarity of the water is not so low that it is unsuitable for swimming.

As discussed in Section 2.3.1, the optical quality of water, primarily its colour and clarity, is determined by the attenuation of light, particularly by SPM but also by dissolved matter (Kirk 1983, 1988). Visual clarity, defined in Section 2.3.1, is of considerable importance because it affects the recreational and aesthetic quality of water.

Panel studies undertaken by Davies-Colley and Smith (1990) in New Zealand showed that almost all people can detect a change of 30% in visual clarity. Davies-Colley (1991) used these results to recommend that reduction in visual clarity should be limited to less than 20%. This value is also used here.

In addition to aesthetic values, visual clarity of water is also important for swimmers to be able to estimate depth and to see subsurface hazards easily (Thornton & McMillon 1989; Smith et al. 1991). Most guidelines require that the substrate should be visible in areas that are of wadeable depth, the water clarity usually being specified in terms of Secchi depth (NHMRC 1990; CCREM 1991). However, as Davies-Colley (1991) points out, a just-visible Secchi disc on the bottom means that potential hazards, such as snags and broken bottles, will not be visible because the Secchi disc has a higher contrast than the hazards. Davies-Colley (1991) recommended that a better guideline for the visual clarity relevant to swimmer safety in wadeable areas would be to require that the black disc visibility should be not less than 1.6 m, which is equivalent to the bottom of the waterbody being visible at an adult chest height of around 1.2 m. For diving areas, the water clarity would need to be considerably greater than this.

Water colour is the perception of light backscattered from within the waterbody as observed when viewed downwards at a near-vertical angle. Typically, about 3% of the incident light will re-emerge from the waterbody as backscattered light, although this ratio can vary widely. Colour of water has three aspects: hue, brightness and saturation or colour purity (Davies-Colley 1991). New Zealand research has shown that people value blue and green hues in water, but not yellows and reds (Smith & Davies-Colley 1992). Davies-Colley (1991) recommended that the natural hue of a waterbody should not be changed by more than 10 points on the Munsell Scale. Further, he recommended that the natural reflectance should not be changed by more than 50% to protect the brightness of the waterbody. New Zealand studies have shown that people are not particularly sensitive to water brightness.

pH

The pH of the water should be within the range 5.0–9.0, assuming that the buffering capacity of the water is low near the extremes of the pH limits.

Ideally, the pH of the water for swimming purposes should be approximately the same as the lacrimal fluid of the eyes, which is about pH 7.4. However, lacrimal fluids have a high buffering capacity when contacted with solutions of different pH levels. They are able to maintain their pH within limits until their buffering capacity is exhausted. A deviation as small as 0.1 unit of the normal pH of the lacrimal fluid causes irritation of the eyes (Mood 1968).

Temperature

For human survival in cold water, the critical problem is to maintain body temperature. There is considerable variation from one individual to another in the rate of body cooling; it is primarily a function of body size, fat content, prior acclimatisation and overall physical fitness. Body heat is lost primarily by conduction from the inner organs through the trunk. Water cooler than 15°C is extremely stressful to swimmers not wearing appropriate protective clothing. Extended periods of continuous immersion at these temperatures may cause death. Thermal stress can be induced by temperatures exceeding the normal skin temperature of 33°C, and there is a risk of injury with prolonged exposure to temperatures above 34–35°C (Health & Welfare Canada 1983).

Toxic chemicals

Waters containing chemicals that are either toxic or irritating to the skin or mucous membranes are unsuitable for recreation. In general, toxic substances should not exceed the concentrations given for untreated drinking waters (Chapter 4).

In general, there are two kinds of human exposure in swimming areas: contact with the waterbody and ingestion of the water. Recreational water should contain no chemicals that can irritate the skin of the human body. To protect swimmers from harmful effects through ingestion, the guidelines for raw water for drinking water supply (Chapter 4) should be applied for other toxicants. Special care must be taken for substances that can enter the body through skin absorption. Higher concentrations of toxicants may be tolerated occasionally if it is assumed that a person will ingest a maximum of 100 mL water during a normal swimming session (NHMRC 1990) compared with 2 L/d for potable water.

Surface films

Oil and petrochemicals should not be noticeable as a visible film on the water nor should they be detectable by odour.

The presence of oil and petrochemicals makes water aesthetically unattractive. They can form deposits on shorelines, and bottom sediments that are detectable by sight and odour. Some organics can be absorbed directly from the water through the skin (CCREM 1991), making these substances even more undesirable in recreational areas.

4 Raw water for drinking water supply

The most authoritative guidelines for Australian drinking waters are contained in the document 'Guidelines for Drinking Water Quality in Australia', produced jointly by NHMRC and AWRC in 1987. These guidelines are presently being updated, and it is expected that the updating will be completed in 1993. It is proposed that, when they become available, a review of the current values listed in this section will be undertaken.

The new NHMRC/AWRC drinking water guidelines are expected to be based largely on the most recent WHO guidelines, but with some changes to make them more relevant to Australian conditions. Some specific changes are expected in the area of pesticide concentrations in drinking waters, where NHMRC and AWRC have indicated that they still intend to rely on the Maximum Residue Level procedure, which was the basis for the existing guidelines (NHMRC/AWRC 1987). In addition, the range of pesticides considered will be extended. It must be emphasised that the NHMRC/AWRC drinking water guidelines relate to 'at tap water quality' while the ANZECC guidelines in this chapter relate to 'raw water quality'.

Many water supplies in Australia require treatment to make them either drinkable or suitable for domestic and industrial use. In this chapter, 'raw water for drinking water supply' refers to water that is used as the intake source for public use. The majority of Australians obtain their drinking water from piped water supplies, most of which include some form of treatment between the raw water supply and delivery to the user. The purpose of the treatment process is to provide the user with drinking water that is safe, palatable and aesthetically pleasing. A major reason to fully treat surface waters for drinking purposes is to improve the aesthetic characteristics rather than for direct health reasons.

4.1 Raw water quality

The raw water quality required for a water supply system depends on the quality of the water to be delivered to the consumers and on the characteristics of the system. For a particular water supply system, the raw water quality may be determined by starting with the water quality guideline for the water delivered to consumers, working back through the water supply system and determining the water quality criteria at key points in the system. The guidelines will allow for changes in water quality between the offtake point and the delivery to the consumer. Final water quality reflects the treatment process itself and incidental changes within the distribution system that lead to either improvements or deterioration in water quality. By working back through the complete water supply system the required quality of raw water entering the system can be defined.

Although the raw water limits presented in this chapter represent drinking water of acceptable quality, there is no inference that better quality water supplies should be allowed to be degraded to these limits. The maintenance of good-quality drinking water can be achieved both by protecting the raw water and by water treatment. It is possible to protect the raw water supply by means of pollution control measures that prevent undesirable constituents entering the raw water and by good catchment management practices. A wide range of treatment technologies are available that enable production of acceptable drinking water from almost any raw water.

Two types of raw water are considered in this chapter: raw water subjected to coarse screening only and raw water subjected to coarse screening and disinfection. Given the wide range of treatment methods that could be used in particular situations (e.g. coagulation, flocculation, filtration, ion exchange, reverse osmosis, carbon adsorption columns), it has not been possible to specify raw water quality guidelines for the many types of water quality that could be involved.

4.1.1 Raw water subjected to coarse screening only

The guidelines listed in Table 4.1 apply to raw water that is not treated prior to consumption apart from the removal of coarse debris. These 'raw water' guidelines need to serve two purposes: firstly, they should protect people who consume untreated water and, secondly, they should provide guidance for catchment managers who require values against which they can evaluate the water quality in their particular area. Untreated water used for drinking water supplies that contains substances at concentrations higher than those given in Table 4.1 may result in deleterious health effects or objections from consumers on aesthetic grounds.

4.1.2 Raw water subjected to coarse screening and disinfection

Slightly poorer quality, primarily due to microbiological contamination, may be acceptable in raw water that is to be disinfected prior to delivery to the consumer. Additional treatment technology, such as coagulation, flocculation and filtration prior to chlorination, or alternative disinfection methods, have not been considered.

Turbidity and dissolved organic carbon (DOC) are the two major features that need special consideration in raw waters to be chlorinated only. Turbidity or suspended particulate matter can interfere with the efficiency of the disinfection process, while chlorination of DOC can result in the formation of chlorinated organic compounds. Although adequate disinfection can occur where the raw water turbidity is elevated, this depends upon the chlorine concentration and the contact period used. In such cases, the disinfection efficiency should be determined on a site-specific basis. Since chlorination efficiency also depends on pH, it is recommended that the pH range be the same as that for raw waters not being treated. Insufficient information is available at this time to allow an appropriate guideline to be recommended for DOC concentrations in raw waters.

4.1.3 Catchment management

Where possible, raw water for drinking purposes should be protected by appropriate management of the catchment supplying the water. Water supplies of a better quality than that described in Table 4.1 should not be allowed to deteriorate to the guideline levels. Where the raw water quality is less than that specified in the guidelines, the preferred option is to improve catchment management practices so that water quality also improves. The alternative is to supply adequate treatment prior to delivery of the water to the consumer, with the degree and type of treatment required depending on the extent to which the existing water quality does not meet the guidelines. Although it is possible to provide suitable treatment for almost any standard of raw water, this will not necessarily be the most preferable option.

Table 4.1 Summary of quality guidelines for raw waters for drinking purposes subjected to coarse screening

Parameter	Guideline values (mg/L, unless otherwise stated)
Biological parameters	–
Micro-organisms:	–

Parameter	Guideline values (mg/L, unless otherwise stated)
Total coliforms	Up to ten coliform organisms may be occasionally accepted in 100 mL. Coliform organisms should not be detectable in 100 mL of any two consecutive samples. Throughout any year, 95% of samples should not contain any coliform organisms in 100 mL
Faecal coliforms	No sample should contain any faecal coliforms in 100 mL
Algae	Up to 5,000 cells/mL may be tolerated; levels of 1,000–2,000 cells/mL of cyanobacteria may result in problems
Toxic parameters	–
Inorganic:	–
Arsenic	0.05
Asbestos	NR
Barium	1.0
Boron	1.0
Cadmium	0.005
Chromium	0.05
Cyanide	0.1
Lead	0.05
Mercury	0.001
Nickel	0.1
Nitrate-N	10.0
Nitrite-N	1.0
Selenium	0.01
Silver	0.05
Organic:	–
Benzene	10.0 µg/L
Benzo(*a*)pyrene	0.01µg/L
Carbon tetrachloride	3.0 µg/L
1,1-Dichloroethene	0.3 µg/L
1,2-Dichloroethane	10.0 µg/L
Pentachlorophenol	10.0 µg/L
Pesticides	(Table 4.2)
Polychlorinated biphenyls	0.1 µg/L
Tetrachloroethene	10.0 µg/L
2,3,4,6-Tetrachlorophenol	1.0 µg/L
Trichloroethene	30.0 µg/L
2,4,5-Trichlorophenol	1.0 µg/L
2,4,6-Trichlorophenol	10.0 µg/L
Radiological:	–
Gross alpha activity	0.1 Bq/L
Gross beta activity (excluding activity of ^{40}K)	0.1 Bq/L

Parameter	Guideline values (mg/L, unless otherwise stated)
Aesthetic parameters	–
Physical:	–
Colour	15.0 Pt-Co
Taste & odour	Not objectionable*
Turbidity	Site-specific determinant
Chemical:	–
Aluminium	0.2
Ammonia (as N)	0.01
Chloride	400.0
Copper	1.0
Oxygen	> 6.5 (> 80% saturation)
Hardness (as $CaCO_3$)	500.0
Iron	0.3
Manganese	0.1
Organics (CCE & CAE)	0.2
pH	6.5–8.5
Phenolics	0.002
Sodium	300.0
Sulfate	400.0
Sulfide	0.05
Surfactant (MBAS)	0.2
Total dissolved solids	1,000.0**
Zinc	5.0

NR No guideline recommended at this time; MBAS Methylene blue active substances

* Engineering & Water Supply Department suggests combined concentration of geosmin and methylisoborneol should be less than 20 ng/L

** Levels in excess of 500 mg/L cause a deterioration in taste

4.2 Guidelines for toxicants (health related)

The following sections contain guidelines for health-related toxicants in raw water for drinking water supply, under the categories of biological parameters and toxic chemicals.

4.2.1 Biological parameters

Pathogenic organisms

Raw water for drinking water supplies subjected to coarse screening only:

- total coliforms:
  - up to 10 coliform organisms may be occasionally accepted in 100 mL;
  - coliform organisms should not be detectable in 100 mL of any two consecutive samples;
  - throughout any year, 95% of samples should not contain any coliform organisms in 100 mL;
- faecal coliforms:
  - no sample should contain any faecal coliforms in 100 mL.

A wide variety of bacterial, viral, protozoan and helminthic pathogens excreted in faeces are capable of initiating water-borne infections. Spread of infections by pathogenic agents depends on factors such as pathogen survival in water and the dose required for establishing infections in particular individuals. Many major disease epidemics (including cholera, typhoid, infectious hepatitis and various forms of gastrointestinal diseases) are known to have occurred as a result of contaminated water being used for drinking purposes (NHMRC/AWRC 1987). However, with the introduction of effective disinfection techniques the incidence of water-borne disease is rare.

The recognition that microbial infections can be water-borne has led to the development of methods for routine examination to ensure that drinking water is free from excremental pollution. Although it is now possible to detect the presence of many pathogens in water, the methods of isolation and enumeration are often complex and time consuming (WHO 1984; McNeill 1985). It is therefore impracticable to monitor water for a wide range of microbial pathogens, and preferable to use indicator systems that can reliably index the presence of pathogens and the related health risk. Ideal indicators should fulfil the following criteria (McNeill 1985):

- pathogen source specific;
- simultaneous presence with bacterial, viral, protozoan and helminthic pathogens and in sufficient numbers to provide an accurate density estimate when pathogen levels are associated with unacceptable health risk;
- no regrowth in the environment;
- applicable to all water types and geographical areas;
- quantification to be accurate, precise, sensitive, cost-effective and simple, with results being available within a short time period.

Currently, there is no indicator that complies with all the above criteria, although many of them are fulfilled by coliform organisms, especially *Escherichia coli*. Historically, the coliform group, faecal streptococci and sulfite-reducing *Clostridium perfringens* have been used as indicators of faecal contamination. More recently, the faecal coliform group and *E.coli* have been differentiated from the total coliforms as more specific indicators of faecal pollution.

Total coliforms have long been used as a microbial indicator of drinking water quality because these organisms are easy to detect and enumerate in water. Coliforms should not be detectable in treated water supplies and, if they are found, their presence would suggest that the treatment was inadequate. In this sense, total coliform tests can be used as an indicator for treatment efficiency, although total coliforms may not be directly health related. The absence of coliforms in disinfected water does not necessarily indicate freedom from cysts of *Giardia*, amoebae and other parasites, since these parasites are known to be more resistant to disinfection (WHO 1984). Furthermore, total coliform bacteria are not specific for faecal contamination, since they also derive from vegetation and soil (Geldreich et al. 1964).

Faecal coliforms are coliform organisms that are able to ferment lactose at 44°C or 44.5°C. They comprise *E.coli* and, to a lesser extent, *Enterobacter, Citrobacter* and *Klebsiella.* Of all these organisms, *E. coli* is specifically of faecal origin and is therefore the most accurate indicator for faecal pollution. *E. coli* is recommended as an indicator organism in the German water quality criteria (Trinkwasserverordnung 1990). A detailed discussion of other indicator organisms, including viral

pathogens, is given in McNeill (1985). NHMRC/AWRC (1987) have recommended guidelines for drinking water based on total coliform and faecal coliform organisms.

Algae
Up to 5,000 cells/mL may be tolerated; however, levels of 1,000–2,000 cells/mL of cyanobacteria should be regarded as the level at which problems may arise.

Blue-green algae, also known as cyanobacteria, normally occur in small numbers in fresh waters and estuaries and in the sea. Under favourable environmental conditions they can reproduce rapidly and form a 'bloom'. The most abundant bloom-forming blue-green algae in Australia are *Microcystis* and *Anabaena* in fresh waters and *Nodularia* in brackish waters (Falconer 1988). The major public health concern with these algal species is the production and release of toxins. There are two main groups of toxins produced by blue-green algae: *Microcystis* and *Nodularia* produce hepatotoxins, which cause liver damage (Runnegar et al. 1988; Falconer et al. 1983), whereas *Anabaena* produces neurotoxins, which cause symptoms of neuro-muscular and respiratory disorders (Jackson et al. 1985; Falconer 1991).

Water supplies are also potentially at risk from these organisms. Blooms of *Microcystis* occurred in drinking water reservoirs of the city of Armidale, New South Wales, in 1981. The bloom was destroyed by treating the reservoir with copper sulfate; however, the copper caused disintegration of the cyanobacteria within a few hours and released toxic cellular compounds into the water (Hawkins et al. 1985). This was thought to be the cause of liver damage found among residents of this city (Falconer et al. 1983). It is essential to monitor algae on a regular basis in any water supply reservoir known to have a history of blue-green algal blooms. Methods for controlling, monitoring and sampling of blue-green algae in drinking-water supplies are given by the Health Department, Victoria (1990). Analytical procedures for the analysis of cyanobacterial toxins are given by Flett and Nicholson (1991).

There is considerable uncertainty as to the actual algal levels in a drinking water supply reservoir that will cause problems. Until recently, monitoring of algal numbers and species in most reservoirs was inadequate. From the small amount of data available, it seems that problems may arise from drinking water containing more than about 5,000 cells/mL of non-toxic algae. However, problems could be expected at lower numbers (about 1,000–2,000 cells/mL) if potentially toxin-forming species are present. It is essential that algae in water supply reservoirs be monitored regularly, and that this monitoring be increased if the algal numbers increase. It is recommended that weekly monitoring is undertaken when cell numbers are 1,000–5,000 cells/mL, increasing to bi-weekly when numbers are in excess of 5,000 cells/mL. Monitoring should also be conducted regularly if cyanobacteria make up a significant proportion of the algal numbers.

4.2.2 Toxic chemicals

Inorganic parameters
Arsenic
The concentration of total arsenic in raw water should not exceed 0.05 mg/L.

Many arsenic compounds are water soluble and thus contamination of water may occur. The trivalent (arsenite), pentavalent (arsenate) and organic forms of arsenic have been found in natural waters. The form of arsenic affects its absorption into the human body; elemental arsenic is very

poorly absorbed, while trivalent and pentavalent inorganic compounds, as well as organic arsenic, are generally readily absorbed (Underwood 1977; WHO 1984;).

There is no evidence that arsenic is essential to man, although it is known that some organic arsenic compounds stimulate growth in animals. Arsenic in high doses is toxic to humans (WHO 1984), and the International Agency for Research on Cancer (IARC) has now confirmed that arsenic is a human carcinogen. Acute arsenic poisoning involves the central nervous system, resulting in coma and death (Vallee 1972), whereas chronic poisoning is characterised by general muscular weakness, loss of appetite, and inflammation of the mucous membranes in the eye, nose and larynx (WHO 1984).

Asbestos
No guideline for asbestos in raw water is recommended at this time.

A study of Canadian drinking water supplies found asbestos concentrations ranging from less than 10^5 to greater than 10^9 fibres/L in raw water (CCREM, 1991). Limited information exists on asbestos levels in Australian waters; typical levels reported are around 10^6 fibres/L with a range between less than 10^6 to 10^7 fibres/L (Sherman P., Queensland Department of Water Resources, pers. comm., April 1992). CCREM (1991) estimated that the daily intake of asbestos from the air was around 300 times greater than from drinking water (600 ng/d compared with 2 ng/d).

Although a number of epidemiological studies have been carried out, only one study has shown a marginally significant association between asbestos levels in drinking water and cancer of the digestive tract (McCabe & Millette 1979). The Canadian water quality criteria (CCREM 1991) contain no values for asbestos in drinking water, as it is believed that the ingestion of asbestos at the concentrations found in drinking water do not constitute a health hazard (Health & Welfare Canada 1979). USEPA (1989) has indicated that it will recommend a maximum containment goal of 7×10^6 fibres/L in drinking water. There is insufficient information at this time to recommended a guideline for asbestos in raw water to be used for drinking water supplies.

Barium
The concentration of total barium in raw water should not exceed 1.0 mg/L.

There is no evidence that barium is essential for humans. The normal source of barium in water is natural mineral matter (WHO 1984). Barium sulfate, the predominate form, is only slightly soluble in water; however, the presence of other anions can enhance the solubility of barium. Barium sulfate is very poorly absorbed by the gastrointestinal tract and has a low toxicity, whereas soluble barium salts are readily absorbed (NRC 1977; USEPA 1977).

Ingestion of soluble barium compounds may result in effects on the gastrointestinal tract and the central nervous system (WHO 1984). Barium salts are considered to be muscle stimulants, especially of the heart (Hart 1974). Excretion of barium occurs readily and accumulation of barium in the bones or other tissue is unlikely (USEPA 1986).

Boron
The concentration of total boron in raw water should not exceed 1.0 mg/L.

Boron may enter water from natural sources or in effluents from industries. Boron is rapidly absorbed into the human body, but is also quickly excreted into the urine. Excessive intake of boron

may cause a variety of physiological distress symptoms, including digestive difficulties and toxicity to the central nervous system (Hart 1974).

Cadmium
The concentration of total cadmium in raw water should not exceed 0.005 mg/L.

The solubility of cadmium in water is influenced by the nature of the cadmium compounds and pH of the water (WHO 1984). Concentrations of cadmium in natural waters in Australia are usually well below 2 µg/L (NHMRC/AWRC 1987). Cadmium is fairly readily absorbed into the human body through ingestion, the absorption rate depending on the chemical form of the cadmium and the age of the person, as well as on dietary factors such as iron, calcium and protein deficiencies (WHO 1984).

The estimated acute oral lethal dose of cadmium for humans is several hundred milligrams (Gleason 1969); however, the evidence of carcinogenity to humans is rather weak (WHO 1984). Cadmium is accumulated in the liver and in the kidney tissue of humans (Hart 1982. Long-term exposure has been associated with anaemia, anosmia, osteomalacia and cardiovascular diseases. Typical features of chronic cadmium poisoning are renal glomerular and tubular damage (Robards & Worsfold 1991).

Chromium
The concentration of total chromium in raw water should not exceed 0.1 mg/L.

Generally, the levels of chromium found in water are low due to the low solubility of chromium compounds. Drinking water usually contains very low concentrations of chromium, that is 5 µg/L or less (NRC 1974). Chromium (III) at low levels is essential to humans, whereas the hexavalent form, chromium (VI), is toxic. It is suggested (WHO 1984) that absorption of water-borne chromium (VI) is at least nine times higher than for chromium (III).

The largest stores of chromium in humans occur in skin, muscle and fat. Chromium accumulated at high levels can generate problems such as nausea and skin ulcerations and, as concentrations reach 0.1 mg/g bodyweight, it can ultimately become lethal (Richard & Bourg 1991), although these levels are unlikely to occur from drinking water only.

The toxicology of chromium has been reviewed by Anderson (1987) and Golden and Karch (1989). WHO (1984) states that chromium (VI) in high doses has been implicated as the cause of digestive tract cancers in humans, although USEPA (1989) and Golden and Karch (1989) state that neither chromium (III) nor chromium (VI) have been found to be carcinogenic following oral ingestion. NHMRC/AWRC (1987) and USEPA (1986) recommend a maximum concentration of 0.05 mg/L total chromium in drinking water; however, USEPA (1989) proposes that the drinking water guideline be raised to 0.1 mg/L. A risk assessment by Golden and Karch (1989) concluded that 0.37 mg/L of chromium (VI) in drinking water for a lifetime was not a significant risk to health.

Cyanide
The concentration of cyanide in raw water should not exceed 0.1 mg/L.

Low exposures to cyanide (2.9–4.7 mg/d) are not fatal to human beings due to the human body's highly efficient detoxification system that converts the cyanide ion into the relatively non-toxic thiocyanate (SCN^-) ion, which is slowly eliminated from the body in the urine (WHO 1984). There is no evidence that cyanide ions are stored in the human body. Lethal toxic effects occur at levels

above 50–60 mg, when the detoxification mechanism is overwhelmed and cyanide blocks oxidative processes in the cell (Hart 1974).

Lead
The concentration of total lead in raw water should not exceed 0.05 mg/L.

Lead is not considered to be an essential trace element. Reticulation systems incorporating lead pipes and fittings are the principal sources of lead in drinking water, but these are uncommon in Australia (NHMRC/AWRC 1987). Lead tends to accumulate in the bones, and the half-lives of lead in blood, soft tissue and bone have been estimated to be two to four weeks, four weeks and twenty-seven-and-a-half years respectively (WHO 1984). Signs of lead toxification include gastrointestinal disturbances, fatigue, anaemia, muscular paralysis and encephalopathy (Hart 1974).

Mercury
The concentration of total mercury in raw water should not exceed 0.001 mg/L.

Mercury is toxic to humans and serves no beneficial physiological functions (Health & Welfare Canada 1980). The mercury concentrations in most potential drinking waters are very low, being mostly less than 0.1 µg/L (Hart 1982). Absorption of inorganic mercury from water into the human body may be 15% or less (WHO 1980), whereas methylmercury is almost completely absorbed. Inorganic mercury compounds are rapidly accumulated in the kidneys. Absorbed methylmercury appears in the blood, where 80–90% is bound to red cells (WHO 1984). The higher toxicity of methylmercury is due to its lipid solubility, which permits the crossing of biological membranes more easily, especially into the brain, spinal cord and peripheral nerves as well as across the placenta (WHO 1984).

The range between the no-effect level and the first signs of toxicity is very narrow in humans; ingestion of 0.2 mg/d showed no toxic symptoms, whereas 0.3 mg/d resulted in toxic symptoms and 0.9 mg/d was lethal (Huetter 1988). The effects of mercury poisoning are neurological and renal disturbance (Swedish Expert Group 1971). Mercury may also cause gonadotoxic and mutagenic effects (WHO 1984). Unborn foetuses are considered to be the main section of the population at risk (WHO 1984).

Nickel
The concentration of total nickel in raw water should not exceed 0.1 mg/L.

Nickel is probably an essential trace element for humans, although it may be toxic in higher concentrations (Hart 1982; WHO 1984). Absorption through the gastrointestinal tract seems to be in the range 1–10% (CEC 1979; Underwood 1977). No significant accumulation of nickel by various tissues of the human body has been observed (USEPA 1977). Nickel seems to be a relatively non-toxic element; high doses up to 1,000 mg/kg in the diet of animals did not cause any adverse effects (NAS 1975; Underwood 1977). Animal experiments have shown certain nickel compounds to be carcinogenic (CEC 1979); however, soluble nickel compounds are not regarded as human carcinogens (WHO 1984).

Nitrate and nitrite
The concentration of nitrate-N in raw water should not exceed 10 mg/L. For nitrite-N, the recommended concentration is 1 mg/L.

Significant nitrate concentrations are widespread in Australian groundwaters (NHMRC/AWRC 1987). Elevated nitrate levels can occur naturally or as a result of agricultural practices or from waste disposal. A very important consideration is that nitrate can be readily converted *in vivo* to nitrite as a result of bacterial reduction (NRC 1977).

The conversion of nitrate to nitrite is high in infants, where the stomach acidity can be about pH 4. The absorbed nitrite can combine with haemoglobin to form methaemoglobin, which results in a reduction of the oxygen transport capacity of the blood. The problem of methaemoglobinaemia as a result of nitrate in water does not arise in adults (WHO 1984). There is also evidence that nitrate may be converted to suspected carcinogenic nitrosamines in the human digestive tract (Bouwer 1990). Therefore, the concentration of nitrate (as N) in raw water for drinking water supply should not exceed 10 mg/L, and the level of nitrite should be below 1 mg/L if used for infant feeding. Concentrations up to 23 mg/L of nitrate (as N) may be acceptable if used only for adults (Hart 1974).

Selenium
The concentration of total selenium in raw water should not exceed 0.01 mg/L.

It has been recognised that trace amounts of selenium in the diet are essential for a number of species, and there is growing evidence that it is also an important element for human health (WHO 1984). According to Hansen (1991), selenium also has a protective effect on methylmercury toxicity. Only limited data are available regarding selenium toxicity in humans. Long-term exposure to high selenium levels may result in gastrointestinal disturbance, discolouration of the skin and bad teeth (WHO 1984).

Silver
The concentration of total silver in raw water should not exceed 0.05 mg/L.

Silver is a non-essential element (Health & Welfare Canada 1980) but relatively little is known about its absorption and metabolism in humans, except that individual organs appear to absorb the metal selectively (NRC 1977). In humans, more than 50% of the body burden was found in the liver sixteen days after exposure (Newton & Holmes 1966). Cases of silver toxicity have been recorded, but only after extremely high doses. The main effects of silver are discolouration of skin, hair and fingernails (WHO 1984).

Organic parameters: Pesticides
Pesticides in raw water should not exceed the concentrations listed in Table 4.2.

Water quality guidelines are needed for pesticides that are persistent or easily leached from soils, as well as pesticides that are applied near or in water supplies. Table 4.2 gives an overview of the different pesticides used in Australia with their recommended maximum acceptable limits according to NHMRC/AWRC (1987) and NHMRC (1989). The guidelines are based on maximum acceptable daily intake levels. This section discusses some of the more persistent and/or ubiquitous pesticides in detail. More specific information on other pesticides is given by Nicholson (1984).

Aldrin and dieldrin
The concentration of aldrin and dieldrin in raw water should not exceed 1 µg/L.

Aldrin belongs to the group of cyclodiene insecticides, which are persistent and accumulate in the food chain. Dieldrin, the epoxide of aldrin, is used in its own right but can also be formed from aldrin

by metabolic oxidation in animals and by chemical oxidation in soils. Both insecticides were mainly used for the control of termites (Nicholson 1984), and both have been found widely in the aquatic environment.

Fish can build up concentrations of dieldrin amounting to several mg/kg bodyweight from concentrations of a few ng/L in water (WHO 1984). Toxic effects of dieldrin in humans are associated with the stimulation of the central nervous system, causing death in acute poisoning cases (WHO 1967).

Results of various carcinogenicity tests on mice indicate that there is a species-specific effect of aldrin and dieldrin on the mouse liver resulting in liver tumours (FAO 1978). Epidemiological studies carried out on humans (occupationally exposed workers) did not allow any conclusion to be made concerning the risk of developing cancer in humans (IARC 1974). According to WHO (1984), other toxicological data support the view that dieldrin and aldrin are not carcinogenic to humans.

Chlordane
The concentration of chlordane in raw water should not exceed 6 µg/L.

Pure chlordane is composed of a mixture of stereoisomers, with cis(alpha) and trans(gamma) isomers predominating. Chlordane is still a widely used insecticide for the control of termites and ants (Nicholson 1984). Like most other organochlorine insecticides, chlordane is persistent and accumulates in the food chain. A study of the persistence of technical grade chlordane in river water showed 85% remaining after eight weeks, whereas cis and trans chlordane were completely stable over the eight-week period (Brookers 1974).

Table 4.2 Guideline values for pesticides in raw water

Compound	Maximum concentration (µg/L)	Compound	Maximum concentration (µg/L)
Acephate	20.0	Fenvalerate	40.0
Alachlor	3.0	Flamprop-methyl	6.0
Aldrin	1.0	Fluometuron	100.0
Amitrol	1.0	Formothion	100.0
Asulam	100.0	Fosamine (ammonium salt)	3,000.0
Azinphos-methyl	10.0	Glyphosate	200.0
Barban	300.0	Heptachlor	3.0
Benomyl	200.0	Hexaflurate	60.0
Bentazone	400.0	Hexazinone	600.0
Bioresmethrin	60.0	Lindane	10.0
Bromazil	600.0	Maldision	100.0
Bromophos-ethyl	20.0	Methidathion	60.0
Bromoxynil	30.0	Methomyl	60.0
Carbaryl	60.0	Metolachlor	800.0
Carbendazim	200.0	Metribuzin	5.0
Carbofuran	30.0	Mevinphos	6.0
Carbophenothion	1.0	Molinate	1.0

Compound	Maximum concentration (µg/L)	Compound	Maximum concentration (µg/L)
Chlordane	6.0	Monocrotophos	2.0
Chlordimeform	20.0	Nabam	30.0
Chlorfenvinphos	10.0	Nitralin	1,000.0
Chloroxuron	30.0	Omethoate	0.4
Chlorpyrifos	2.0	Oryzalin	60.0
Clopzralid	1,000.0	Paraquat	40.0
Cyhexatin	200.0	Parathion	30.0
2,4-D	100.0	Parathion-methyl	6.0
DDT	3.0	Pendimethalin	600.0
Demeton	30.0	Perfluidone	20.0
Diazinon	10.0	Permethrin	300.0
Dicamba	300.0	Picloram	30.0
Dichlobenil	20.0	Piperonyl butoxide	200.0
3,6-Dichloropicolinic acid	1,000.0	Pirimicarb	100.0
Dichlorvos	20.0	Pirimiphos-ethyl	1.0
Diclofop-methyl	3.0	Pirimiphos-methyl	60.0
Dicofol	100.0	Profenofos	0.6
Dieldrin	1.0	Promecarb	60.0
Difenzoquat	200.0	Propanil	1,000.0
Dimethoate	100.0	Propargite	1,000.0
Diquat	10.0	Propoxur	1,000.0
Disulfoton	6.0	Pyrazophos	1,000.0
Diuron	40.0	Quintozene	6.0
DPA	500.0	Sulprofos	20.0
Endosulfan	40.0	2,4,5-T	2.0
Endothal	600.0	Temephos	30.0
Endrin	1.0	Thiobencarb	40.0
EPTC	60.0	Thiometon	20.0
Ethion	6.0	Thiophanate	100.0
Ethoprophos	1.0	Thiram	30.0
Fenchlorphos	60.0	Trichlorofon	10.0
Fenitrothion	20.0	Triclopyr	20.0
Fenoprop	20.0	Trifluralin	500.0
Fensulfothion	20.0	–	–

Sources: NHMRC/AWRC (1987), NHMRC (1989)

According to WHO (1984), pure chlordane showed a negative result in the Ames Mutagenity Test, whereas technical grade chlordane was mutagenic in the test. Carcinogenity of chlordane has been demonstrated in only one animal, the mouse (WHO 1984). Consequently, the maximum level of chlordane in raw water is based on toxicity data and not on carcinogenicity data.

DDT
The concentration of DDT in raw water should not exceed 3 µg/L.

The structure of DDT (dichlorodiphenyltrichloroethane*)* permits several isomeric forms. The commercial product DDT consists predominantly of *p,p'*-DDT together with some *o,p'*-DDT and some smaller amounts of other isomers. DDT was first used during World War II to protect military areas against malaria, typhus and certain other vector-borne diseases.

During the last two decades, DDT has become the most controversial organochlorine insecticide due to its extreme persistence and accumulation throughout the environment. Concern over the accumulation of these residues and their ubiquitous occurrence has resulted in the use of DDT being curtailed or banned in some countries. However, DDT is still used extensively in tropical regions of Third World countries.

The main toxic effects of DDT are on the membranes of the nervous system, and it can significantly affect the liver (WHO 1984). According to WHO (1984), no teratogenic effects were observed in several animal studies and DDT was not found to be mutagenic in bacterial test systems. Although a large number of epidemiological studies have been carried out involving workers exposed to DDT, no conclusive data were obtained regarding the carcinogenic potential of DDT to humans.

Heptachlor and heptachlor epoxide
The concentration of heptachlor in raw water should not exceed 3 µg/L.

Heptachlor is a broad-spectrum insecticide of the group of cyclodiene insecticides, and is very effective against soil insects (Nicholson 1984). It rapidly forms, either biologically or photochemically, the more toxic compound heptachlor epoxide. Like most of the organochlorine insecticides, heptachlor and its epoxide are persistent and accumulate in the environment. These undesirable properties have resulted in it being restricted in many developed countries (Nicholson 1984), although heptachlor is still approved in Australia for use against termites.

Heptachlor is readily metabolised by mammalian systems to heptachlor epoxide, which is stored in the adipose tissue, liver, kidney and muscle (WHO 1967). Because of its high lipid content, milk is one of the major excretion routes for organochlorine compounds, and heptachlor epoxide is one of the most frequently found pesticides in human milk.

Although evaluation of the carcinogenic risk to humans produced inconclusive results (Nicholson 1984), heptachlor and heptachlor epoxide have been shown to be carcinogenic in mice; therefore, heptachlor must be considered a possible human carcinogen.

Lindane
The concentration of lindane in raw water should not exceed 10 µg/L.

Lindane (γ-HCH) is a broad insecticide of the group of cyclic chlorinated hydrocarbons. In Australia, its main uses are on pineapples and pets; in the United States, it is used for the treatment of animals, buildings, clothes, water (for mosquitos), plant seeds and soils (USEPA 1979c). Lindane can be isomerised to alpha and delta isomers by micro-organisms and plants.

The bioaccumulation factors of lindane and other HCH isomers are low compared with other organochlorine insecticides like DDT and dieldrin (Nicholson 1984); nevertheless, lindane residues

have been found in the aquatic environment of Australia. A survey of corals, fish and molluscs of the Great Barrier Reef for organochlorines revealed lindane to be the most common pollutant from this group (Olafson 1978). This high frequency of detection reflects the widespread use of this pesticide.

Lindane is fairly soluble in water (10 mg/L), which contributes to rapid absorption and excretion in animals and man (USEPA 1979c; WHO 1984). Symptoms of poisoning include headache, vertigo and irritation of the skin, eyes and respiratory tract mucosa. People occupationally exposed to HCH for eleven to twenty-three years revealed biochemical manifestations of toxic hepatitis (WHO 1984), and animal experiments showed an increased incidence of liver tumours when animals were fed with lindane. Therefore, lindane needs to be considered a human carcinogen (WHO 1984).

2,4-dichlorophenoxyacetic acid
The recommended maximum level for 2,4-dichlorophenoxyacetic (2,4-D) acid in raw water is 100 µg/L.

2,4-D is used as a herbicide for the control of broad-leafed plants and as a plant growth regulator (WHO 1984). This herbicide is readily degraded in soils and sediments by microbial processes (Bowmer 1987), and there is reportedly no accumulation. If 2,4-D is absorbed by humans or animals, it is distributed in the various tissues, but not stored, and is rapidly excreted virtually unchanged (WHO 1984). Individuals exposed to 2,4-D through use or manufacture have complained of fatigue, headache, liver pains and loss of appetite. Studies on the carcinogenic properties of this compound have been inconclusive; however, WHO (1984) suggests that 2,4-D is not a potential carcinogen.

Other organic parameters

Benzene and lower alkylbenzenes
Benzene and lower alkylbenzenes, such as toluene, xylene and ethylbenzene, are widely used in chemical processes as intermediates in the production of various chemicals and as solvents. Concentrations of these compounds in groundwater can exceed the levels in surface waters because of the suppression of the evaporation process. According to WHO (1984), except for benzene, there seems to be no potential health risk from the levels of these compounds usually found in drinking water.

Benzene: The concentration of benzene in raw water should not exceed 10 µg/L.

Benzene is a volatile, colourless liquid produced mainly from petroleum or coal tar distillation. The major sources of benzene in water are atmospheric deposition (rain and snow) and chemical-plant effluents (WHO 1984). Due to its volatility, benzene has an average half-life of around four minutes in surface waters (WHO 1984).

Acute exposure to benzene results in central nervous system depression. Epidemiological studies suggest a relationship between chronic benzene exposure and leukaemia, and animal experiments have shown induced chromosome damage following exposure to benzene (WHO 1984; USEPA 1980a). The guideline for benzene in raw water is based on cancer data, with a cancer risk of 1:100,000 per lifetime.

Chlorinated alkanes and alkenes
Chlorinated alkanes are produced in large quantities, the major use of these compounds being as intermediates in the production of other organochlorine substances. Chlorinated alkenes are widely

used as solvents, softeners, paint thinners, dry-cleaning fluids and as intermediates. They usually occur at low concentrations in surface waters due to their high volatility. According to NHMRC/AWRC (1987), these compounds are not considered to be a significant problem in Australia.

Carbon tetrachloride: The concentration of carbon tetrachloride in raw water should not exceed 3 µg/L.

Carbon tetrachloride (CCl_4) has a wide range of industrial and chemical applications, including its use as an intermediate in the manufacture of fluorocarbons (aerosol propellants). Carbon tetrachloride may be exceedingly stable under certain environmental conditions, with a half-life of 70,000 years (USEPA 1980b), and evaporation seems to be the major process of removal from surface waters (WHO 1984). The compound is almost completely absorbed through the gastrointestinal tract, and simultaneous ingestion of fat and alcohol enhances the absorption (Nielson & Larson 1965). It can also enter the body through the skin.

Acute toxicity of carbon tetrachloride results in liver injury within a few days, and changes in blood parameters, visual acuity and the pancreas have also been noted (WHO 1984). Studies performed by the US National Cancer Institute (National Cancer Institute 1976) have found carbon tetrachloride to be carcinogenic in mice. Given these findings, the recommended guideline is based on a cancer risk of 1:100,000 per lifetime.

1,1-dichloroethene: The concentration of 1,1-dichloroethene in raw water should not exceed 0.3 µg/L.

Of the three isomers of dichloroethene, 1,1-dichloroethene is the most widely used in the chemical industry as an intermediate for the synthesis of methylchloroform and polyvinyldiene chloride copolymers (CCREM, 1991). Based on studies of related compounds, such as trichloroethene, it appears that virtually 100% of ingested 1,1-dichloroethene is absorbed by the body (WHO 1984). In studies with rats, the largest concentrations of 1,1-dichloroethene were found in the kidneys, followed by liver, spleen, heart and brain. Research data suggest substantial binding of this compound to macromolecules and associated lipids (WHO 1984). 1,1-dichloroethene has been shown to be mutagenic in the Ames test, and it produced mammary tumours in both mice and rats, as well as kidney adenocarcinomas in mice. The guideline for 1,1-dichloroethene in raw water is based on a cancer risk of 1:100,000 per lifetime.

1,2-dichloroethane: The concentration of 1,2-dichloroethane in raw water should not exceed 10 µg/L.

1,2-dichloroethane is a liquid that is widely used as a solvent, an intermediate in the chemical industry and an insecticide (not in Australia). Few data are available concerning the metabolism of 1,2-dichloroethane. It is only known that this compound is readily soluble in the lipids of the brain, which probably promotes the influence of 1,2-dichloroethane on the nervous system (Zoeteman 1978). 1,2-dichloroethane is a known mutagen. Epidemiological studies have not shown a relationship between exposure to this compound and cancer (WHO 1984).

Tetrachloroethene: The concentration of tetrachloroethene in raw water should not exceed 10 µg/L.

Tetrachloroethene (perchloroethylene, PCE) is a colourless liquid used primarily as a dry-cleaning solvent and, to a lesser extent, as a degreasing solvent in metal industries (WHO 1984). Due to its high volatility, the concentrations of PCE in surface water are normally small; however, higher

concentrations can be found in groundwaters. There is evidence that PCE is completely absorbed from the gastrointestinal tract (WHO 1984). Like the other chloroethenes, acute effects of PCE are dominated by central nervous system depression. PCE has been demonstrated to be a liver carcinogen in mice, but results in rats were negative. A conservative approach was adopted for the recommended raw water guidelines, with PCE treated as if it were a carcinogen. The guideline is based on a cancer risk of 1:100 000 per lifetime.

Trichloroethene: The concentration of trichloroethene (TCE) in raw water should not exceed 30 µg/L.

TCE, a colourless, volatile liquid, is mainly used as a degreasing solvent in the metal industry. TCE is readily absorbed into the body by all routes of exposure (WHO 1984) and it is known as a central nervous system depressant; the compound has been used medically as a general anaesthetic (Defalque 1961). TCE has been reported to be mutagenic to a number of bacterial strains and it produced liver cancer in mice; however, it is not clear if this substance is a tumour inducer or a promoter (WHO 1984). The recommended maximum concentration in raw water is based on the demonstrated carcinogenity in mice and an acceptable risk level of less than one additional case of cancer per 100,000 population per lifetime.

Chlorophenols

Chlorinated phenols (chlorophenols) are used as disinfectants, biocides, preservatives, dyes and pesticides. Chlorophenols are also formed by chlorination of drinking water that contains phenols (USEPA 1979a; WHO 1984). Chlorophenols are widely recognised for their low taste and odour threshold, which can be as low as 1 µg/L (WHO 1984). Where phenols are present in high concentrations in raw water, they should be reduced as far as possible before chlorination takes place.

Pentachlorophenol: The concentration of pentachlorophenol (PCP) in raw water should not exceed 10 µg/L.

PCP is the most widely used chlorophenol in industry, and is commonly used as a wood preservative. It may occur in drinking water at concentrations that are objectionable on both health and taste grounds. The taste threshold for PCP in drinking water is 30 µg/L (Nicholson 1984). The recommended guideline of 10 µg/L is based on toxicological data (damage of kidney and liver) suggesting an acceptable daily intake of 3 µg/kg bodyweight. A PCP concentration of 10 µg/L in drinking water would account for an average of 10% of this acceptable daily intake (WHO 1984).

2,3,4,6-tetrachlorophenol: The concentration of 2,3,4,6-tetrachlorophenol in raw water should not exceed 1 µg/L.

2,3,4,6-tetrachlorophenol is the only one of the three tetrachlorophenol isomers that is commercially used (Jones 1981). Together with PCP, it is usually added to wood preservatives as an active ingredient (CCREM 1991). The taste threshold for 2,3,4,6-tetrachlorophenol in drinking water is reported to be 1 µg/L (Nicholson 1984). The guideline based on toxicological data would be twenty times higher than this (Nicholson 1984).

2,4,5-trichlorophenol: The concentration of 2,4,5-trichlorophenol in raw water should not exceed 1 µg/L.

Of the six isomers of trichlorophenol only 2,4,5-trichlorophenol and 2,4,6-trichlorophenol are commercially used, both as ingredients in wood preservatives. 2,4,5-trichlorophenol is also used in the production of hexachlorophene, a disinfectant and sanitary product for domestic and hospital use (CCREM 1991). A guideline based on toxicological data would suggest a maximum level of 700 µg/L in raw water, whereas a guideline based on organoleptic data would be as low as 1 µg/L (Nicholson 1984).

2,4,6-trichlorophenol: The concentration of trichlorophenol in raw water should not exceed 10 µg/L.

2,4,6-trichlorophenol, together with pentachlorophenol, is an active ingredient in wood preservatives. The main concern regarding 2,4,6-trichlorophenol in drinking water is that this substance has been shown to induce leukaemia and is mutagenic in yeast, a result that also indicates the carcinogenic properties of the compound (WHO 1984). On the basis of these findings, 2,4,6-trichlorophenol has to be considered as a compound that might increase the cancer rate in humans if present in sufficient quantities in drinking water. In view of the possible carcinogenic properties, the recommended guideline, based on toxicological–carcinogenic data and assuming a lifetime cancer risk of 1 per 100,000, is 10 µg/L.

Polychlorinated biphenyls
The concentration of polychlorinated biphenyls (PCBs) in raw water should not exceed 0.1 µg/L.

PCBs are a complex mixture of compounds manufactured by the reaction of biphenyl with chlorine (Nicholson 1984). PCBs have been widely used because they have excellent thermal stability, general inertness, excellent dielectric properties and non-flammability (CCREM 1991). The two main uses for PCBs are capacitors and transformers. According to Nicholson (1984), no PCBs are manufactured in Australia, and usage is restricted to closed system applications. Once discharged into the environment, PCBs persist; being inert and highly fat soluble they accumulate in the lipids of exposed organisms resulting in bioconcentration.

A number of studies have dealt with the toxicity of PCBs (Kimbrough 1974; USEPA 1979b), and carcinogenity and co-carcinogenity have been reported. The IARC has classified PCBs as probable carcinogens. In order to protect human health, a guideline of 0.1 µg/L is recommended, assuming an acceptable cancer risk of 1:1,000,000 (Nicholson 1984).

Polycyclic aromatic hydrocarbons
*A maximum concentration of 0.01 µg/L for benzo(*a*)pyrene (indicator for polycyclic aromatic hydrocarbons) is recommended.*

Polycyclic aromatic hydrocarbons (PAHs) are a large group of organic compounds consisting of two or more benzene rings, with non-aromatic rings present in some instances. PAHs are formed by incomplete combustion of organic material from both natural (e.g. forest fire, volcanic activity) and anthropogenic (e.g. burning of fuels and refuse) sources. PAHs may also be synthesised by some bacteria, algae and higher plants.

Atmospheric deposition is responsible for much of the background concentrations of PAHs in the aquatic environment, but water-based and land-based discharges may also contribute significantly to PAH concentrations at specific locations (CCREM 1991). Because of their low solubility in water, PAHs tend to adsorb strongly to particulate matter (WHO 1984).

A number of PAHs have been shown to be mutagenic and carcinogenic (Cerniglia 1984; Santodonato et al. 1980). According to NHMRC/AWRC (1987) benzo(*a*)pyrene is an indicator for the pollution of PAHs. The guideline value is based on toxicological experiments, which showed a statistically significant dose-related increase in the incidence of stomach tumours after adding benzo(*a*)pyrene to the diet of mice (Neal & Rigdon 1967).

Radiological parameters
It is recommended that the gross alpha activity and the gross beta activity (excluding the activity of ^{40}K) should not exceed 0.1 Bq/L.

Radioactive materials are introduced into the environment from a number of sources, both natural and anthropogenic. The natural sources include those substances produced by cosmic rays, which may find their way to water sources via rainfall and runoff, and those present in rocks and soils, such as ^{238}uranium and its daughters ^{226}radium and ^{222}radon. Human-made radionuclides result from fall-out from nuclear testing, nuclear power production, and medical and other uses of radioactive materials.

The effects of exposure of humans to excessive radiation are well known. The detrimental health effects include those that are manifested in the exposed individual (e.g. carcinogenesis) or in that individual's descendants. The guidelines are designed to cover the intake of the most toxic radionuclides. According to NHMRC/AWRC (1987), the analysis for specific isotopes is recommended when gross levels are exceeded.

4.3 Aesthetic Guidelines (not directly health related)

4.3.1 Physical

Colour
A limit of 15 Pt-Co is recommended for the true colour of raw water.

Colour in drinking water is generally caused by natural organic substances (e.g. humic substances), although dissolved iron, manganese and highly coloured industrial wastes, such as pulp and paper and textile wastes, can also contribute to this colour. A colour greater than 15 Pt-Co can be detected in a glass of water by the consumer (NHMRC/AWRC 1987). The removal of excessive colour (if present as organic matter) prior to chlorination will reduce the production of trihalomethanes (Fitzgerald et al. 1990). Limiting colour in drinking water can also limit the concentration of undesirable substances that can be adsorbed or complexed by the organic material.

Taste and odour
The taste and odour of raw water should not lead to levels in drinking water that are objectionable to most consumers.

Odours in drinking water may come from natural sources (microscopic organisms, particularly algae; decaying vegetation; and other organic matter) or from anthropogenic activities (sewage, industrial effluents). In general, the sense of taste is more useful in detecting inorganic constituents rather than organic compounds in drinking water (WHO 1984). The presence of objectionable taste in a public water supply system may cause consumers to seek alternative sources of drinking water that may be less 'safe'. Management should ensure that sources of objectionable tastes and odours (often from algae) are kept to a minimum in raw water supplies.

A quantitative maximum acceptable limit cannot be established because of the lack of an objective measurement method. However, the Engineering and Water Supply Department in South Australia is using the analytical determination of two of the main taste and odour forming compounds, geosmin and methylisoborneol, as an interim threshold, the combined concentration being kept to less than 20 ng/L.

Turbidity
Turbidity in water is caused by suspended and colloidal inorganic and/or organic particles. In addition to being aesthetically displeasing, turbidity can protect pathogenic micro-organisms from the effect of disinfection processes and promote disinfection deficiencies (NHMRC/AWRC 1987). Therefore, turbidity should be lower in raw water subjected to coarse screening only. The amount of allowable turbidity depends on the chlorine concentration used and the contact period and is, therefore, site specific.

4.3.2 Chemical

Aluminium
The concentration of total aluminium in raw water should not exceed 0.2 mg/L.

Aluminium salts are used extensively in water treatment for the removal of colour and turbidity. Some natural waters also contain significant levels of colloidal and dissolved aluminium (NHMRC/AWRC 1987). Aluminium is not essential to humans. Aluminium salts are normally absorbed from food and water into the human body, but are complexed with phosphate and excreted in the faeces (Thienes & Haley 1972).

Administration of aluminium to rats showed no effect on longevity and incidence of tumours (Crapper et al. 1973). Aluminium has been associated with Alzheimer's disease, although it is not clear whether the presence of aluminium causes such conditions or is simply an indicator of other factors (WHO 1984; Kawachi & Pearce 1991). Aluminium is health-related for kidney dialysis patients, and precautions are needed to ensure that water receives sufficient treatment to remove aluminium (Wills & Savory 1983).

Ammonia (as N)
The concentration of ammonia in raw water should not exceed 0.01 mg/L.

Ammonia enters surface waters and groundwaters from decomposition of nitrogenous organic matter (e.g. domestic sewage) and effluents from industries. Ammonia in the amount present in natural or polluted waters is not physiologically damaging (Hart 1974). However, the presence of ammonia in water supplies may indicate recent sewage pollution, particularly if the free ammonia present greatly exceeds the organic nitrogen concentration.

Chloride
The concentration of chloride in raw water should not exceed 400 mg/L.

Chloride is the most abundant anion in the human body and contributes significantly, along with its associated cations, to the osmotic activity of the extracellular fluid (WHO 1984). Because water is a relatively minor contributor of chloride, health impacts with respect to chloride in water are not significant. The taste threshold for chloride is dependent on the associated cation, but is usually between 200 mg/L and 300 mg/L. Most agencies (WHO 1984; CCREM 1991; CEC 1980;

Trinkwasserverordnung 1990) recommend chloride concentrations of 250 mg/L as the maximum desirable level. NHMRC recommends a maximum of 400 mg/L, in recognition of the higher level of chloride in many Australian waters (NHMRC/AWRC 1987).

Copper
The concentration of total copper in raw water should not exceed 1.0 mg/L.

Copper is a beneficial and essential element for humans (Health & Welfare Canada 1980). Copper poisoning is rare in humans; the dose usually considered lethal being 15 g (single dose as $CuSO_4$; Hart 1974). Copper in water has an unpleasant, astringent taste, the taste threshold being detectable in water at 2.6 mg/L (Cohen et al. 1960). Staining of laundry and plumbing fixtures occurs when copper concentrations in water exceed 1 mg/L.

Dissolved oxygen
The dissolved oxygen concentration should not fall below 6.5 mg/L or 80% saturation.

The presence of dissolved oxygen has no adverse physiological effect. A supersaturated concentration of dissolved oxygen may arise from an algal bloom and can increase corrosion of metal pipes. Of more concern in public water supplies are concentrations that are too low; if no dissolved oxygen is present, anaerobic decomposition may occur resulting in a foul odour and taste (Hart 1974). Anaerobic conditions may also promote the growth of potentially toxic bacteria.

Hardness (as $CaCO_3$)
The concentration of hardness (as $CaCO_3$) in raw water should not exceed 500 mg/L.

The principal hardness-causing ions in fresh water are calcium and magnesium. Other multivalent cations like strontium, iron, barium and manganese may also contribute. The sum of the individual components of the hardness are expressed in terms of an equivalent quantity of calcium carbonate. Hardness is mainly a problem in water supplies using groundwater (NHMRC/AWRC 1987). In areas with very hard water, household pipes can become choked with deposited scale and an increased soap consumption for efficient washing is observed. There is no evidence of adverse health effects attributable to high levels of calcium and magnesium (WHO 1984).

Iron
The concentration of total iron in raw water should not exceed 0.3 mg/L.

Iron is an essential element in human nutrition, but food sources generally provide the minimum requirements (CCREM 1991). Iron in public water supplies is highly undesirable, mainly due to aesthetic considerations; it affects the taste of drinking water and stains laundry. Iron compounds that settle-out in the distribution system gradually reduce the flow of water. A slimy coating on piping may also result due to the presence of iron bacteria, which derive their energy from the oxidation of ferrous iron to ferric iron (WHO 1984).

Manganese
The concentration of total manganese in raw water should not exceed 0.1 mg/L.

Manganese is an essential element in animals and humans (WHO 1984). Guidelines for manganese concentrations in drinking water are governed by aesthetic considerations rather than possible health effects. The presence of manganese in drinking water supplies may be objectionable for a

number of reasons; levels above 0.15 mg/L result in an undesirable taste in beverages and stain plumbing fixtures and laundry (Griffin 1960). Levels higher than 0.05 mg/L can also cause growth of slimes, which may result in taste, odour and colour problems (NHMRC/AWRC 1987).

Organics
The concentration of total organics in raw water should not exceed 0.2 mg/L.

Organic contaminants can enter a water supply from agricultural runoff (natural substances, insecticides, herbicides, other agricultural chemicals), from partially treated domestic waste and from industrial waste. Some of these compounds are present in very low concentrations and are not degraded in the water environment. It is relatively difficult to detect, identify and quantify these organic substances; the test procedures are complex, expensive and usually not practicable in routine examinations.

A general technique used for quantifying organic pollutants in water is the carbon adsorption technique, where the organic compounds are absorbed on to activated carbon under prescribed conditions; desorption occurs with specific solvents also under prescribed conditions. 'Organics-carbon adsorbable' are divided into two classes depending on the solvent used: the 'carbon-chloroform extract' (CCE) and the 'carbon-alcohol extract' (CAE). Because of the undefined nature of the CCE and CAE, it may be desirable that extracts exceeding the above limits be further analysed to determine the precise organic compounds present.

pH
The pH of raw water should be within the range of 6.5-8.5.

The pH of an aqueous system is a measure of the acid-base equilibrium achieved by various dissolved compounds. Provided that the pH is not extreme, it can generally be assumed that the pH is controlled by the $CO_2/HCO_3^-/CO_3^{2-}$ equilibrium system. Major sources of pH changes in water are industrial effluents, which may be strongly acid or alkaline. The acceptable range for drinking water is primarily based on minimising corrosion and encrustation, with consideration of the effectiveness of chlorine disinfection which is impaired above pH 8. According to NHMRC/AWRC (1987), new concrete tanks and cement mortar-lined pipes can significantly increase pH, and a value up to 9.2 is acceptable in such circumstances.

Phenolics
The concentration of total phenols in raw water should not exceed 2 µg/L.

Although phenols in high concentrations are toxic to man, taste and odour thresholds occur at much lower concentrations and therefore aesthetic considerations prevail. Unacceptable taste and odour are generated when waters with concentrations of phenols exceeding 5 µg/L are chlorinated (Health & Welfare Canada 1980).

Sodium
The concentration of total sodium in raw water should not exceed 300 mg/L.

Sodium is a common component of drinking waters in Australia, particularly those of groundwater origin (NHMRC/AWRC 1987). Because the human body has very effective methods of controlling sodium levels, sodium is not an acutely toxic element (WHO 1984). WHO (1984) recommends a guideline value of 200 mg/L, which is based on the taste threshold of sodium and not on health

considerations. NHMRC/AWRC (1987) recommend a value of 300 mg/L as a consequence of the considerable difficulties in reducing the high sodium content of Australian waters.

Sulfate
The concentration of sulfate in raw water should not exceed 400 mg/L.

The major concerns related to high concentrations of sulfate in drinking water are catharsis, gastrointestinal irritation and unpleasant taste. Usually the human body adapts rapidly to higher concentrations (NHMRC/AWRC 1987). The taste threshold for sodium sulfate is 200–500 mg/L (WHO 1984), and is higher for most other sulfates (CCREM 1991). The guideline of 400 mg/L is based on taste considerations.

Sulfide (as H_2S)
The concentration of sulfide in raw water should not exceed 0.05 mg/L.

Hydrogen sulfide (H_2S) can be present in underground water and, under anaerobic conditions, in surface waters. The most sensitive effects of sulfides in drinking water are disagreeable taste and odours. Because of the low taste and odour threshold, the acceptable concentration was set at 0.05 mg/L. Sulfide may also lead to the growth of sulfur-oxidising bacterial slimes (Hart 1974).

Surfactants (methylene blue active substances)
The concentration of surfactants in raw water should not exceed 0.2 mg/L.

Foaming is an undesirable property of drinking water. Many substances in water will cause foaming when the water is agitated or air is entrained as in a tap. The methylene-blue method is used to measure anionic surfactants; however, this method will detect more than just anionic surfactants as interference occurs with other compounds (Hart, 1974).

Total dissolved solids
The concentration of total dissolved solids (TDS) in raw water should not exceed 1,000 mg/L.

Many Australian waters have characteristically high levels of dissolved solids (NHMRC/AWRC 1987). TDS influence other water quality parameters such as taste, hardness, corrosion properties and the tendency to encrustation. There is no evidence of deleterious physiological reactions occurring in humans consuming drinking water that exceeds 1,000 mg/L (WHO 1984). The guideline is based on taste considerations; above 1,500 mg/L, taste generally renders water unacceptable to human consumers (NHMRC/AWRC 1987). Most urban consumers would reject drinking water with TDS above approximately 500 mg/L.

Zinc
The concentration of total zinc in raw water should not exceed 5 mg/L.

The guideline for zinc is based on aesthetic considerations. Zinc is considered to be essential for humans; more than seventy zinc metallo-enzymes are known (WHO 1984). Zinc gives water an undesirable astringent taste and water containing zinc in excess of 5 mg/L may appear opalescent and develops a greasy film when boiled (WHO 1984). These aesthetic problems develop at concentrations below those related to any human health concerns.

5 Agricultural water uses

Water supply for agricultural purposes is a significant determinant of agricultural productivity in many areas of Australia and, as such, indirectly influences important export industries. By world standards, agricultural communities in Australia often face severely limited quantities of water of suitable quality. The problem is compounded as the increasing pressures of urbanisation, industrialisation and agricultural practices themselves threaten the quality of these water resources.

5.1 Irrigation

Because of its low relief and latitude, two-thirds of Australia is arid or semi-arid. Irrigation using both surface water and groundwater plays an important role in satisfactory agricultural use of this land, and constitutes about 70% of the water use in Australia (Department of Primary Industries & Energy 1987). However, this use of water resources has, in many cases, introduced salt and other chemical and microbiological contaminants into soils and plants. These contaminants can cause alteration of the soil, death and disease of livestock and contamination of food products. In the long term, if the irrigation water adversely affects the soil's physical and chemical properties, crop yields will also not be sustained.

Factors influencing irrigation water quality guidelines

Specific water quality guidelines depend on a complex interaction of different factors. Three important factors should be considered in applying the guidelines for irrigation water (Table 5.1):

- ***Soil***: Soil texture, structure and organic matter determine percolation of water, holding capacity and exchange capacity. Therefore, the degree to which the irrigation water and its components will be leached out, remain available to plants or become fixed and unavailable to plants, depends largely on the soil characteristics. Nevertheless, insufficient rationale has been published in the scientific literature to establish soil categories as a standard part of water quality guidelines.
- ***Crops***: Crops vary widely in their sensitivity to toxic substances. The guidelines contained in Table 5.1 are set to protect the most sensitive crop.
- ***Climate and management***: Evapotranspiration and rainfall determine the frequency of irrigation required. In general, the potential toxicity of the substances in the irrigation water increases as more frequent irrigation is required. Nevertheless, application of water in excess of crop needs may provide protection of the crop through leaching of salts from the plant root zone when drainage is unrestricted. The type of irrigation method used is also important (e.g. flood, furrow or sprinkler methods) for the sensitivity of crops to toxic substances in the irrigation water (VIRASC 1980).

Table 5.1 Summary of guidelines for irrigation water quality

Parameter	Guideline (mg/L, unless otherwise stated)	Comment
Biological parameters	–	–
Plant pathogens	–	*
Human and animal pathogens	1,000 faecal coliforms/100 mL	Tentative value. Geometric mean of not less than 5 water samples taken

		per month; no more than 20% should exceed 4,000 organisms/100 mL
Algae	Should not be visible	–
BOD_5	–	No guideline recommended
Major ions	–	–
Bicarbonate	–	No guideline recommended due to interaction with other factors
Chloride	30–700 (Table 5.2, Table 5.3, Table 5.4)	Maximum concentration should be set according to sensitivity of crop
Sodium	Soils: Figure 5.1; crops Table 5.5	–
Total dissolved solids	Table 5.6	–
*Heavy metals and trace ions***	–	–
Aluminium	5.0	High toxicity in acid soils
Arsenic	0.1	–
Beryllium	0.1	–
Boron	0.5–6.0	Table 5.8
Cadmium	0.01	Higher toxicity in acid soils
Chromium	1.0	Limit chromium (VI) concentration to 0.1 mg/L
Cobalt	0.05	–
Copper	0.2	–
Fluoride	1.0	–
Iron	1.0	–
Lead	0.2	–
Lithium	2.5	Citrus: 0.075 mg/L
Manganese	2.0	If acid soils, limit to 0.2 mg/L
Mercury	0.002	–
Molybdenum	0.01	–
Nickel	0.2	–
pH ($CaCl_2$)	4.5–9.0	–
Selenium	0.02	–
Uranium	0.01	–
Vanadium	0.1	–
Zinc	2.0	1 mg/L is recommended for sandy soil below pH 6
Pesticides		–
Insecticides	–	No guidelines recommended
Herbicides	Table 5.9	–
Radioactivity	Gross Alpha 0.1 Bq/L	–
–	Gross Beta 0.1 Bq/L	–

* All surface waters used in WA plant nurseries in *Phytophthora* susceptible areas must be chlorinated to 2 mg/L residual chlorine

** Higher maximum concentrations may be recommended in neutral to alkaline soils, as discussed in the detailed guidelines in the paragraphs following

Guidelines for irrigation waters

The recommended water quality guidelines (Table 5.1) rely heavily on the criteria developed by NAS/NAE (1973) and Hart (1974), but are supplemented with more recent information where available. These criteria assume an annual application rate of irrigation water of 1,000 mm and retention of trace ions in the surface in the top 15 cm of the soil. Under these conditions, the recommended concentration of ions in the irrigation water should allow irrigation for a minimum 100 years before any phytotoxic levels are reached in the soil. In some parts of Australia, application rates may be significantly higher or lower than this figure. In these areas it is recommended that the guideline values may be adjusted to accommodate different loading rates for contaminants and different leaching characteristics of the soils. Guidelines for irrigation using wastewater are given by VicEPA (1991). Some States have developed information on the effects of salt on soils and crops (Gill 1986).

5.1.1 Biological parameters

Plant pathogens

A variety of pathogens can be contracted and distributed through irrigation water, including nematodes, fungi, viruses and bacteria (NAS/NAE 1973). The risk of infection is lower if the irrigation water is not recycled from fields and reused on other crops or if wastewater is used for irrigation, with the exception of *Phytophthora* in surface waters that are used in Western Australia for irrigation of susceptible horticultural crops. However, insufficient data are available to allow the formulation of guidelines for plant pathogens. NHMRC (1987) should be consulted for a discussion of the application of reclaimed water in irrigation.

Human and animal pathogens

A tentative guideline is set of a geometric mean (log) of no more than 1,000 faecal coliforms/100 mL based on not less than five water samples taken per month. No more than 20% of these samples should exceed 4,000 organisms/100 mL.

The above criterion is described as tentative due to the discussion on the use of appropriate indicator organisms (e.g. *Escherichia coli).*

Many micro-organisms that are pathogenic to animals and humans can be carried in irrigation water and may contaminate field crops. These organisms may be transferred to the consumer on the surface of the produce and can cause bacillary and amoebic dysentery, *Salmonella gastroenteritis*, typhoid and paratyphoid fevers (Hart 1974). Faecal coliforms are recommended as indicator organisms for the presence of pathogens in most water quality criteria documents (NAS/NAE 1973; Hart 1974; VicEPA 1983; CCREM 1991). However, it should be recognised that this is somewhat controversial as *E. coli* has been recommended as a more appropriate indicator (CCREM 1991). The Victorian water quality criteria (VicEPA 1983) are recommended for use throughout Australia.

Algae

No quantitative objectives have been recommended for algae. Nevertheless, to protect the irrigation equipment from clogging and to avoid soil oxygen depletion, algae should not be visible.

Biochemical oxygen demand (BOD_5)
Soil aeration and oxygen availability present no problem in well-structured soils with good water quality. Problems may arise when drainage is poor and the reuse of irrigation effluent becomes more popular. At this point, no guideline is recommended.

5.1.2 Major ions

Bicarbonate
No quantitative guideline for bicarbonate is recommended because the potential hazard of bicarbonates is influenced by other soil and water characteristics. The bicarbonate hazard is high in low-salinity waters applied to sandy and silty loam soils.

In arid and semi-arid regions of Australia, irrigation water containing high concentrations of bicarbonate is frequently used. Long-term use of such irrigation water can lead to a high concentration of bicarbonate in the soil water due to evapotranspiration, and there is an increased tendency for calcium and magnesium to precipitate as insoluble carbonates. Over time, this reduction of the calcium and magnesium concentration will result in an increased sodium adsorption ratio (SAR), which adversely affects soil structure (refer to the section 'Sodium'). Hence, a sodium hazard may result where previously one did not exist.

Eaton (1950) and Wilcox (1958) developed the concept of residual sodium carbonate ($[HCO_3^- + CO_3^{2-}] - [Ca^{2+} + Mg^{2+}]$) in order to characterise water quality with respect to bicarbonate hazard. This concept did not take into consideration that the precipitation of calcium and magnesium is not quantitative (Bower et al. 1968; Wilcox 1958), and that the two ions can also be released into the soil solution by mineral weathering processes (Rhoades 1968).

The calculation of adjusted SAR (Rhoades 1968, 1971) tends to over-compensate for a bicarbonate hazard, as shown by experiences in Canada (CCREM 1991). The bicarbonate hazard is high in low-salinity waters applied to sandy and silty loam soils (Director, Qld Bureau of Sugar Experiment Stations, pers. comm., January 1992).

Chloride
*The maximum chloride concentration should be set according to the sensitivity of a crop (*Table 5.2*,* Table 5.3 *and* Table 5.4*). Irrigation water containing more than 100 mg Cl/L should not be used for sensitive crops.*

Chloride is essential to the growth of plants. However, in excess it can have a toxic effect depending on the sensitivity of the crop and the irrigation method chosen (Table 5.2, Table 5.3 and Table 5.4). In general, most woody plant species (stone-fruit, citrus, avocados) are sensitive to low concentrations of chloride, whereas most vegetable, grain, forage and fibre crops are less sensitive (Chapman 1966; Oster and Rhoades 1984).

There are two ways of inducing chloride damage. First, the chloride ion can be taken up by the roots and moved upwards to accumulate in the leaves. Excessive accumulation may cause burning of leaf tips or margins, bronzing and premature yellowing of leaves (Hart 1974). Second, direct foliar absorption of chloride from sprinkler irrigation can cause damage especially on fruit trees, which are most sensitive. Generally these effects are minimised with night-time sprinklers and water applied at a rapid continuous rate, providing that care is taken to prevent soil erosion.

Table 5.2 Chloride tolerance of fruit and woody crops by root uptake

Rootstocks	Chloride in irrigation water (mg/L)	Cultivars	Chloride in irrigation water (mg/L)
Grapes	710–960	Boysenberry	250
Stone-fruits (peaches, plums, etc.)	180–600	Blackberry, raspberry	–
Strawberries	110–180	–	–

Sources: Westcot and Ayers (1984); CCREM (1991)

Table 5.3 Chloride concentrations in irrigation water causing foliar damage

Sensitivity	Chloride (mg/L)	Affected crop
Sensitive	< 178	Almond, apricot, plum
Moderately sensitive	178–355	Grape, pepper, potato, tomato
Moderately tolerant	355–710	Alfalfa, barley, corn, cucumber
Tolerant	> 710	Cauliflower, cotton, safflower, sesame, sorghum, sugar-beet, sunflower

Source: Westcot and Ayers (1984)

Table 5.4 Tolerance of chloride sensitive crops to chloride in irrigation water

Crop	Irrigation method	Maximum chloride concentrations (mg/L)
Citrus	Overhead sprinklers	100
–	Under-tree sprinkler	265
Stone-fruit	Overhead sprinklers	70
–	Under-tree sprinkler	175
Vines	–	350
Tobacco	Overhead sprinklers	30

Sources: Callinan (1970), Jones (1972), AWRC (1969)

Sodium

The major problem associated with sodium in irrigation water is its tendency to adversely affect soil structure. The magnitude of this effect can be related to the relative proportions of sodium ions to calcium and magnesium ions in the irrigation water (SAR). Figure 5.1 can be used to classify a possible sodium hazard to the soil structure. The direct toxic effects of sodium concentrations in irrigation water (expressed as SAR) on different plants are shown in Table 5.5.

Sodium damage due to irrigation water can occur in two ways: either through alteration of the soil structure or via direct toxicity to the crops.

Table 5.5 Tolerance of crops to sodium

Tolerance	SAR of irrigation water	Crop	Condition
Very sensitive	2–8	Deciduous fruits, nuts, citrus, avocado	Leaf tip burn, leaf scorch
Sensitive	8–18	Beans	Stunted, soil structure favourable

Tolerance	SAR of irrigation water	Crop	Condition
Moderately tolerant	18–46	Clover, oats, tall fescue, rice	Stunted due to nutrition and soil structure
Tolerant	46–102	Wheat, lucerne, barley, tomatoes, beets, tall wheat grass, crested grass, fairway grass	Stunted due to poor soil structure

Source: Hart (1974)

Soil structure

Excessive sodium in irrigation water relative to calcium and magnesium can adversely affect soil structure and reduce the rate at which water moves into and through the soil, as well as reduce soil aeration (Figure 5.1). The relation of sodium to calcium and magnesium is expressed as:

$$SAR = \frac{[Na^+]}{\sqrt{\frac{[Ca^{2+}] + [Mg^{2+}]}{2}}}$$

where $[Na^+]$, $[Ca^{2+}]$, $[Mg^{2+}]$ are concentrations in milli-equivalent per litre.

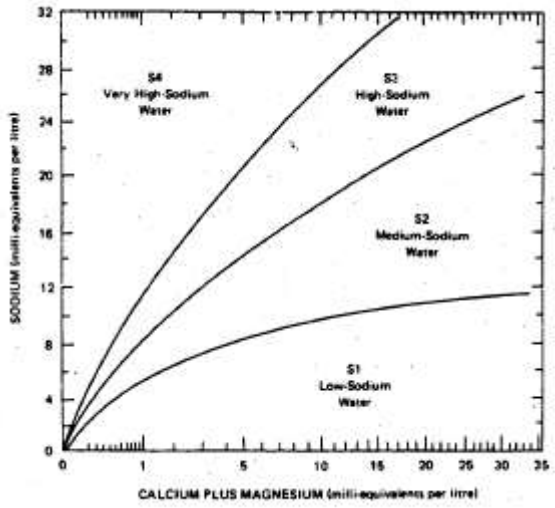
If calcium is the predominant adsorbed cation, the soil tends to have a granular structure, which is easily worked and readily permeable. However, when adsorbed sodium exceeds 10–15% of the total exchange capacity of the soil, the clay becomes dispersed and puddled when wet, lowering permeability and forming a hard impermeable crust when dry. Most researchers agree that the problems of soil permeability increase when SAR approaches 10 (CCREM 1991).

Quirk and Schofield (1955) showed that there is a clear advantage in increasing the salinity in irrigation water in order to maintain soils in a flocculated and permeable condition. The potential permeability problem of irrigation water with a high SAR can often be reduced by high salt concentrations. Hence irrigation water relatively high in sodium and low in total salt may result in poor soil physical condition, whereas waters high in sodium and high in total salt provide stable conditions.

Plants

Sodium is required in limited amounts for most plant growth. However, some plants are sodium-sensitive and can be affected by low concentrations of exchangeable sodium (Table 5.6). Bernstein (1962) has reported that sodium toxicity can occur in sensitive fruit crops (avocados, stone-fruit and citrus) when SAR is as low as 5.5.

Figure 5.1 Water quality guidelines for sodium water



Notes:
Low-sodium water (S1) can be used for irrigation on almost all soils, with little danger of the development of a sodium problem. However, sodium-sensitive crops, such as stone-fruit trees and avocados, may accumulate harmful amounts of sodium in the leaves.
Medium-sodium water (S2) may present a moderate sodium problem in fine-textured (clay) soils unless there is gypsum in the soil. This water can be used on coarse-textured (sandy) or organic soils that take water well.
High-sodium water (S3) may produce sodium problems in most soils and requires special management, good drainage, high leaching and additions of organic matter. If there is plenty of gypsum in the soil, a serious problem may not develop for some time. If gypsum is not present it, or some similar material, may have to be added.
Very high-sodium water (S4) is generally unsatisfactory for irrigation except at low-salinity or medium-salinity levels, where the use of gypsum or some other additives makes it possible to use such water.

5.1.3 Total dissolved solids (salinity)

General guidelines for the salinity of irrigation water and crop tolerances are given in Tables 5.6 and Table 5.7 *respectively.*

The salinity or total dissolved solids (TDS) concentration of irrigation water is an extremely important water quality consideration. An increase in salinity causes an increase in the osmotic pressure of the soil solution, resulting in a reduced availability of water for plant consumption and possible retardation of plant growth. Table 5.6 contains the recommended guidelines for salinity in irrigation water. These guidelines are influenced by soil characteristics, crop tolerance, climate and irrigation practices.

Table 5.6 General guidelines for salinity of irrigation water

Class	Comment	Electrical conductivity (µS/cm)	TDS (mg/L)*
1	Low-salinity water can be used with most crops on most soils and with all methods of water application with little likelihood that a salinity problem will develop. Some leaching is required, but this occurs under normal irrigation practices except in soils of extremely low permeability	0–280	0–175

Class	Comment	Electrical conductivity (µS/cm)	TDS (mg/L)*
2	Medium-salinity water can be used if moderate leaching occurs. Plants with medium salt tolerance can be grown, usually without special measures for salinity control. Sprinkler irrigation with the more-saline waters in this group may cause leaf scorch on salt-sensitive crops, especially at high temperatures in the daytime and with low application rates	280–800	175–500
3	High-salinity water cannot be used on soils with restricted drainage. Even with adequate drainage, special management for salinity control may be required, and the salt tolerance of the plants to be irrigated must be considered	800–2,300	500–1,500
4	Very high-salinity water is not suitable for irrigation water under ordinary conditions. For use, soils must be permeable, drainage adequate, water must be applied in excess to provide considerable leaching, and salt-tolerant crops should be selected	2,300–5,500	1,500–3,500
5	Extremely high-salinity water may be used only on permeable, well-drained soils under good management, especially in relation to leaching and for salt-tolerant crops, or for occasional emergency use	> 5,500	> 3,500

* TDS (mg/L) = 0.68 x electrical conductivity (µS/cm)

Source: Hart (1974)

With adequate drainage, salt accumulation in the soil can be controlled to an extent by the rate of application of water. If the sum of applied irrigation water and rainfall is lower than evaporation and plant consumption, an accumulation of salts in the main root zone will result. Proper irrigation management will allow application of sufficient excess water (leaching fraction) to move a portion of the salts out of the root zone, while not causing excessive increases in the groundwater table.

Plants vary in their tolerance to soil salinity (Table 5.7). In general, most fruit crops are sensitive, followed by vegetable, field and forage crops. Tolerances vary with different stages of growth; usually germinating seedling are the most sensitive (Hart 1974).

5.1.4 Heavy metals and trace ions

Aluminium

The concentration of total aluminium in irrigation water should not exceed 5 mg/L.

Toxicity of aluminium to field crops is an important cause of reduced productivity on acid soils, because the soluble aluminium content in the soil solution increases due to the enhanced solubility of aluminium oxides and the destruction of clay minerals and other silicates that occurs at low soil pH values. Thus, aluminium toxicity may develop without the introduction of aluminium in the irrigation water. In this case, lime must be added to increase the soil pH. Several crops show aluminium toxicity at concentrations as low as 0.1–0.5 mg/L in soil solution (Schachtschabel et al. 1989). These values cannot be applied directly to irrigation waters because of the capacity of soils to adsorb and complex aluminium ions and hence reduce the toxicity of the Al^{3+} cation, the species most harmful to plants (Wright et al. 1987). However, these values do indicate that aluminium is toxic to plants at relatively low concentrations.

Table 5.7 Relative tolerance of suggested crop plants to saline irrigation water

Water class	EC (µS/cm)	TDS (mg/L)	Pastures and fodders	Fruit	Vegetables	Ornamentals	Precautions for irrigation uses
1/2	0–800	0–500	Ladino clover, red clover, Alsike clover, white Dutch clover, subterranean clover	Persimmon, loquat, passionfruit, strawberry, avocado, almond, apricot, peach, plum, lemon, grapefruit, orange, grape, walnut	Parsnips, green beans, celery, radish, cucumber, squash, peas, onion, carrot, potatoes, sweet corn, lettuce, French beans	Violet, African violet, primula, gardenia, begonia, azalea, camellia, magnolia, fuchsia, dahlia	Avoid wetting leaves on hot, dry days
3	800–2,300	500–1,500	Cocksfoot, perennial ryegrass	Mulberry, apple, pear, raspberry, quince	Cauliflower, bell pepper, cabbage, broccoli, tomato, broad beans, field beans, sweet potato, artichoke	Geranium, gladiolus, bauhinia, zinnia, rose, aster, poinsettia, musa, podocarpus	Avoid wetting leaves during daytime. Avoid light, frequent waterings. Water quickly and use continuous-wetting sprinklers if wetting the leaves.
4	2,300–5,500	1,500–3,500	Oats (hay), wheat (hay), rye (hay), lucerne, Sudan grass, *Paspalum dilatatum*, strawberry clover, sweet clovers, millet, Wimmera ryegrass, Rhodes grass, couch grass, barley, birdsfoot trefoil	Olive, fig, pomegranate, cantaloupe	Spinach, asparagus, kale, garden beets, gherkins	Stock, chrysanthemum, carnation, hibiscus, oleander, bougainvillea, vinca, Aust. hop bush, coprosma (green and variegated), Japanese pepper, *Ficus* spp. in gen., *Ficus hillii*, false acacia, Qld pyramid ree, NZ Christmas bush, flase mahogany, Rottnest cyprus, *Acacia longifolia*, buffalo grass, kikuyu grass, portulaca, mesembryanthemum, boobyalla, morrel, swamp yate, York gum, couch grass, bamboo, Kondinin blackbutt	Avoid wetting leaves of most plants where possible. Adeqaute leaching necessary.

Water class	EC (µS/cm)	TDS (mg/L)	Pastures and fodders	Fruit	Vegetables	Ornamentals	Precautions for irrigation uses
5	> 5,500	3,500	Seashore paspalum, *Puccinella cilata*, saltwater couch	date plum	–	Canary palm, Paspalum vaginatum, salt sheoaks, salt river gum, tamarisks (evergreen and deciduous), saltbushes	Do not wet leaves where possible. Excellent drainage and leaching essential.

EC: Electrical conductivity

Note: The plant and water groupings are not meant to be rigid, but merely provide a general guide. Plants are arranged in approximate order of salt tolerance in each column, with the least tolerant at the top. Soil texture and drainage may be extremely important. Plants listed as suitable for saline water will grow better with less-saline water.

Arsenic

The concentration of total arsenic in irrigation water should not exceed 0.1 mg/L.

Agricultural soils can have elevated concentrations of arsenic due to the past use of organo-arsenic pesticides, which remain as long-lasting residues in the soil (NAS 1977a). Generally, arsenate (arsenic [V]) and arsenite (arsenic [III]) are the primary forms of arsenic in the soil. Both arsenate and arsenite are subjected to chemically and/or microbiologically mediated oxidation reduction and methylation reactions in soils (Masscheleyn et al. 1991). Uncontaminated soils have concentrations in the range 1–20 mg/kg (Hart 1982). Woolson (1973) reported that vegetable crops did not grow in soils treated with 500 mg arsenic/kg, and crop growth was reduced proportionally at rates of 10 mg/kg, 50 mg/kg and 100 mg/kg. The main effect of toxic amounts of arsenic appears to be the destruction of chlorophyll in the foliage, a consequence of inhibition of reductase enzymes (McKee & Wolf 1963).

Nutrient solutions containing 0.5–10 mg/L (depending on plant species) can result in toxic effects on crops (NAS/NAE 1973). However, soils have a high capacity to reduce the toxicity of applied dissolved arsenic through adsorption to clay and reaction with phosphorous (CCREM 1991). The edible parts of plants do not generally accumulate arsenic to levels dangerous to consumers. Instead, experience shows that yield reductions and crop failure are the main effects of arsenic-contaminated soils (Hart 1974; NRCC 1978a).

Beryllium

The concentration of total beryllium in irrigation water should not exceed 0.1 mg/L.

Beryllium is toxic to both animals and plants. Romney and Childress (1965) reported that 2 mg/L in nutrient solutions reduced the growth of different plants. The toxicity was found to be greater in acid soils (Williams & LeRiche 1968). The translocation of beryllium from the root of the plants to the foliage does not occur readily, either from nutrient solutions or from soil (Gough et al. 1979).

Boron

The concentration of total boron in irrigation water should not exceed 0.5 mg/L for sensitive plants. Recommended maximum concentrations for tolerant species are given in Table 5.8*.*

Boron in relatively small amounts is essential to the normal growth of all plants; however, this element can be toxic when present in excess. Crop species vary both in their boron requirement and in their tolerance to excess boron. A compilation of the tolerances of different plants is provided in Table 5.8.

Table 5.8 Relative tolerance of agricultural crops to boron

Tolerance*	Concentration of boron in soil water (mg/L)**	Agricultural crop
Very sensitive	< 0.5	Blackberry
Sensitive	0.5–1.0	Peach, cherry, plum, grape, cowpea, onion, garlic, sweet potato, wheat, barley, sunflower, mung bean, sesame, lupin, strawberry, Jerusalem artichoke, kidney beans, lima beans
Moderately sensitive	1.0–2.0	Red pepper, pea, carrot, radish, potato, cucumber
Moderately tolerant	2.0–4.0	Lettuce, cabbage, celery, turnip, Kentucky bluegrass, oat, corn, artichoke, tobacco, mustard, clover, squash, musk melon

Tolerant	4.0–6.0	Sorghum, tomato, alfalfa, purple, vetch, parsley, red beet, sugar-beet
Very tolerant	6.0–15.0	Asparagus

* Tolerance will vary with climate, soil conditions and crop varieties; values are to be used as a guideline only

** Maximum concentrations tolerated in irrigation water without reduction in yield or vegetative growth are approximately equal to soil water values

Source: Westcot and Ayers (1984).

Boron sorption plays an important role in determining soil solution concentrations (Jame et al. 1982). Concentrations of 1–2 mg/L usually occur in the soil solution only when the adsorptive capacity of the soil is saturated. Sensitive crops may therefore be grown in natural to alkaline soil without harm from irrigation water containing 2 mg/L of boron for a longer period (NAS/NAE 1973).

For reasonable crop protection, it is recommended that the boron concentrations listed in Table 5.8 be used for irrigation water. Leaching of boron from soils containing water-soluble boron can cause problems due to increases in the soil water concentration above the tolerance limits of the crop being irrigated. High-boron soils have caused problems in areas of northern Victoria (Penman & McAlpine 1949; Sauer 1958). In Western Australia, boron concentrations of up to 0.7 mg/L in the saturation extract were consistent with normal plant growth (Malcolm 1971). Generally, toxic concentrations of boron in irrigation waters are often associated with groundwater or secondary wastewaters, rather than surface water (Ayers & Westcot 1976).

Cadmium

The concentration of total cadmium in irrigation water should not exceed 0.01 mg/L.

Cadmium is toxic to both animals and plants in low concentrations. Reported cases of cadmium poisoning in Japan from 1947 to 1965 (Itai-Itai disease) led to increasing concern regarding cadmium in the environment, and much research done in recent years indicates that even carcinogenity may be a possibility (Merian 1984). Uncontaminated soils generally contain around 0.06 mg/kg cadmium (Hart 1974). Higher concentrations may be due to addition of phosphate fertiliser (an average of 0.04 g cadmium/kg P_2O_5; Sauerbeck 1985) or sewage sludge. In rural areas, input via atmospheric deposition may also contribute to elevated concentrations of cadmium in the soil.

Although it is not required for metabolism, cadmium is readily taken up by plants and uptake increases with acidity and the total content of cadmium in the soil system (Herms & Brummer 1980, 1984). Because cadmium is similar to zinc (an essential element for plant growth), it can readily interfere with metabolic processes within the plants. The resorption of cadmium by the plant can also be minimised by a good supply of zinc, manganese and copper. Cadmium in nutrient solutions reduces growth of a diversity of plants at levels ranging from 0.1 mg/L to 1 mg/L (NAS/NAE 1973; Hart 1974). Dugdale (1978) reported a total of 50 mg/kg of cadmium in soil as the threshold for toxicity to plants, although first symptoms of toxicity may occur in sensitive plants at lower levels (Miller et al. 1976; Haghiri 1973).

Chromium

The concentration of total chromium in irrigation water should not exceed 1 mg/L for continuous use. The concentration of chromium (VI) should be limited to 0.1 mg/L.

There is no evidence that chromium is essential to plants, although traces of chromium are essential for humans and animals (Anderson 1987; Schachtschabel et al. 1989). Concentrations of 1–10 mg/L in nutrient solutions reduce crop yield, depending on the tolerance of different plant species (NAS 1974). Chromium (III) and chromium (VI) in nutrient solutions are about equally available to plants. However, when added to the soil chromium (VI) remains mobile and available to plants, whereas chromium (III) is adsorbed or complexed and therefore immobile (Breeze 1973).

The toxicity limits for chromium (VI) range from 5 mg/kg to 500 mg/kg, while toxic effects of chromium (III) occur at 50–5,000 mg/kg, depending on plant species and soil type (NRCC 1976). Because translocation of chromium within the plant does not readily occur, most of the absorbed chromium remains in the roots. Therefore, chromium concentrations in plant material are usually found in the following order: the greatest amount in roots, then leaves, then fruit (Schachtschabel et al. 1989). In general, there should be few problems associated with discharges to land of wastewaters (e.g. from tanneries) containing chromium (III) because this form of chromium is relatively non-mobile.

The maximum concentration recommended for chromium in irrigation waters in the United States is 0.1 mg/L for continuous use on all soils, and 1 mg/L for use for up to twenty years on neutral and acid soils (NAS/NAE 1973). The Canadian limit has been set at 0.1 mg/L (CCREM 1991), but CCREM suggests that future guidelines should be written in terms of the more available form, chromium (VI). Hart (1982) recommended a concentration of total chromium of 1.0 mg/L for continuous use.

Cobalt

The concentration of total cobalt in irrigation water should not exceed 0.05 mg/L.

Field occurrence of cobalt toxicity is rare (Hart 1974). Hodgson (1960) has reported strong interaction between cobalt and most soils at neutral and alkaline pH values, and Vanselow (1966) showed that high concentrations of cobalt (100 mg/kg) had little effect on citrus crops, probably also due to adsorption of cobalt by soil particles. However, cobalt in nutrient solution has been found to be toxic to a variety of food crops at concentrations of about 0.1–5 mg/L.

Copper

The concentration of total copper in irrigation water should not exceed 0.2 mg/L.

Copper is an important component of several plant enzymes and, therefore, essential in small concentrations for plant growth. For healthy plant growth, the copper content in soil should not fall below 6 mg/kg, although higher copper concentrations are required in organic soils or soils rich in phosphate, manganese, iron or zinc (CCREM 1991).

The concentrations of copper in uncontaminated soils are generally 1–50 mg/kg (Demayo & Taylor 1981). Higher concentrations can occur due to application of sewage sludge, copper-based fungicides (vineyards), and manure produced by animals fed with copper-rich diets (100–850 mg copper/kg dry food; Sauerbeck 1985). Atmospheric deposition in mining and smelting areas may also contribute to elevated levels of copper in soils.

Delas (1963) provided first evidence of copper toxicity in sensitive plants at concentrations of 25–50 mg/kg soil. However, according to Baker (1974), copper toxicity is associated with higher concentrations in soils ranging from 150 mg/kg to 400 mg/kg. Plant uptake of copper occurs more

readily in soils with pH ($CaCl_2$) less than 5 (Herms & Brummer 1984; Sanders 1982), and toxicity is therefore related to the pH of the soil.

Fluoride
The concentration of total fluoride in irrigation water should not exceed 1.0 mg/L.

Serious effects of fluoride occur mostly to animal and human consumers of fluoride-contaminated plants, but not to the plants themselves. The fluoride content of soils usually ranges from 20 mg/kg to 400 mg/kg (Schachtschabel et al. 1989), depending on the parent rock.

Fluoride ions are strongly adsorbed on iron and aluminium oxides as well as on clay minerals (Peek & Volk 1985) and the amount of fluoride taken up from the soil by plants is generally unrelated to the fluoride content of the soil or the irrigation water. Soil type, calcium and phosphorus content and soil pH seem to be the predominant controlling factors (Hart 1974). On neutral and alkaline soils, it is probable that higher concentrations of fluoride in irrigation water can be tolerated without subsequent problems arising, due to the capacity of this soils to deactivate fluoride (NAS/NAE 1973; Hart 1974).

Iron
The concentration of total iron in irrigation water should not exceed 1.0 mg/L.

Iron is essential for plant growth. Dissolved iron in irrigation water is relatively common in Australia; it precipitates on aeration and may produce light-brown spotting on plants (Hart 1974). This coating may interfere with photosynthesis and normal transpiration and respiration. Iron in irrigation water may clog irrigation equipment and hence reduce the efficiency of irrigation. Problems may occur for trickle irrigation systems at concentrations as low as 0.25 mg/L (Director, Tas. Dept of Primary Industry, pers. comm., May 1992).

Lead
The concentration of total lead in irrigation water should not exceed 0.2 mg/L.

Lead seems to bind very effectively to most soils, and total lead concentrations in agricultural soils range from 2 mg/kg to 200 mg/kg, with a mean around 16 mg/kg (Hart 1982). Plant uptake may be facilitated by lower soil pH (less than 4.5) and lower organic content (Herms & Brummer 1984; Jorgenson 1976).

Lead is not readily transported within the plant system, therefore concentrations of lead in plant material are usually found in the following order: the greatest amount in roots, then leaves, then fruit. High concentrations in leaves (e.g. lettuce) are mostly correlated to high atmospheric deposition of lead on to the plant due to industrial (smelter) or traffic emissions, rather than to uptake of lead via roots (Page et al. 1971; Schachtschabel et al. 1989).

The phytotoxicity of lead is relatively low compared with other trace elements; reduction in the yield of lettuces resulted at concentrations of 1,000 mg/kg (John & Van Laerhoven 1972), and poor growth of beans was reported at 820 mg/kg (Berg 1970). The recommended guideline of 0.2 mg/L for continuous use on all soils would result in a maximum level of lead in the soil of 88.9 mg/kg in 100 years, assuming that lead is retained in the top 15 cm (applied rate of 1,000 mm/m^2/a).

Lithium

The concentration of total lithium in irrigation water should not exceed 2.5 mg/L, except when used for citrus when the recommended value is 0.075 mg/L.

Lithium is very mobile within soils and is not precipitated by any known process (Hart 1974). Usually crops sensitive to sodium are also affected by high lithium concentrations. Except for citrus trees, most crops can tolerate up to 5 mg/L in nutrient solution (NAS/NAE 1973). Citrus trees begin to show slight toxicity at concentrations of 0.06–0.1 mg/L in water (Bradford 1963). Lithium concentrations of 0.1–0.25 mg/L in irrigation water produced severe toxicity symptoms in grapefruit, and concentrations of 3.5 mg/L were toxic to sugar-beets (Hilgeman et al. 1970; El-Sheikh et al. 1971).

Manganese

The concentration of total manganese in irrigation water should not exceed 2.0 mg/L, unless the soil is acidic, in which case the manganese concentration should be limited to 0.2 mg/L.

Manganese is an essential element for plant growth, apparently as an enzyme activator, but is deficient in many Australian soils (Graham et al. 1988). Manganese toxicity to crops may occur in acid soils. Literature summarised in Hart (1974) indicates that toxicity of manganese varies considerably within different crops; toxicity has been observed due to concentrations as low as 0.5 mg/L.

Clogging of irrigation pipelines may arise at concentrations as low as 0.03 mg/L (Director, Tas. Dept of Primary Industry, pers. comm., May 1992). Awad (1984) recommended that manganese at concentrations of up to 2 mg/L is generally acceptable in irrigation waters for continuous use. CCREM (1991) recommended that the concentration of manganese in irrigation waters should not exceed 0.2 mg/L for continuous use on all soils, and 10.0 mg/L for up to twenty years' use on neutral and alkaline fine-textured soils. Hart (1974) recommended a maximum concentration of 0.5 mg/L.

Mercury

The concentration of total mercury in irrigation water should not exceed 0.002 mg/L.

Mercury is strongly retained by soils, especially by those high in organic matter. Most plants do not readily take up mercury due to the low availability within the soil solution (Hart 1982; Schachtschabel et al. 1989). Lettuce grown on contaminated soil (7 mg mercury/kg) showed only a small increase in mercury absorption (MacLean 1974a); however, carrots and mushrooms can accumulate mercury from soils. Stijve and Besson (1976) reported accumulation factors of mercury of up to 33 in mushrooms when compared with the soil.

Molybdenum

The concentration of total molybdenum in irrigation water should not exceed 0.01 mg/L.

Plants absorb molybdenum predominantly as the MoO_4^{2-} anion from the soil solution and can concentrate it in tissue, apparently without adverse effects. Accumulation of molybdenum by crops is facilitated in alkaline soil through increased anion exchange, leading to higher MoO_4^{2-} concentrations in the soil solution.

Molybdenum accumulation in plant tissue may be harmful to livestock consuming contaminated feed. Molybdenum surplus in livestock diet may produce copper deficiency; therefore, a good copper supply may reduce molybdenum toxicity in livestock. Toxic effects of molybdenum in forage crops are considered to occur at above 5–6 mg/kg for cattle and 10–11 mg/kg for sheep (Dye 1962).

Nickel
The concentration of total nickel in irrigation water should not exceed 0.2 mg/L.

Nickel concentrations in most soils range from 5 mg/kg to 500 mg/kg, with an average around 100 mg/kg (Chapman 1966). Soils developed from serpentine rocks contain much higher quantities of nickel (400–500 mg/kg). Soil nickel concentrations around 500 mg/kg seem to be toxic to plants, although this depends on the soil conditions, particularly pH. Below pH 6 the concentration of soluble and exchangeable nickel increases considerably (Herms & Brummer 1984). Nickel concentrations in nutrient solutions of 1–2 mg/L are toxic to a number of plants (Foroughi et al. 1988).

pH
Generally, crops and soils can tolerate irrigation waters ranging in pH from 4.5 to 9.0.

Irrigation waters with extreme pH values may cause indirect problems. Values below 4.8 can cause solubilisation of aluminium, manganese or heavy metals in concentrations large enough to be toxic to plants if applied to acid soils over a long period. Waters having high pH values (greater than 8.3) may contain high concentrations of sodium, carbonate and bicarbonate. These ions may affect growth and soil conditions, as discussed in previous sections.

Selenium
The concentration of total selenium in irrigation water should not exceed 0.02 mg/L.

The guideline for selenium in irrigation water is based on the toxicity of selenium to animals rather than plants, which can absorb relatively large amounts of selenium without apparent harm. The actual selenium concentration in a plant depends on factors such as plant species and age and the concentration of soluble selenium.

The availability of selenium within the soil system depends on the amount of soluble selenite (SeO_3^{2-}) and selenate (SeO_4^{2-}). In acid soils and moderately oxidising soil conditions where selenite is predominant with high fractions of iron and aluminium oxides, the availability of selenite is low due to high adsorption on to the soil. High availability occurs in alkaline soils with strong oxidising conditions where selenate is predominant due to low adsorption (Schachtschabel et al. 1989). According to Horvath (1976), selenium concentrations of over 5 mg/kg in feed are toxic to livestock.

Uranium
The concentration of total uranium in irrigation water should not exceed 0.01 mg/L.

Only a small fraction of the uranium in soil is available to plants due to adsorption on soil particles and organic matter (Harmsen & de Haan 1980). Uranium taken up by plants usually accumulates in the roots (Hamilton 1974). Zhukov and Zudilkin (1971) reported wheat yields were not affected by the addition of 10 mg/kg uranyl nitrate to soil, whereas yield was reduced by 50% when adding 50 mg/kg. Vegetables can accumulate uranium to levels 100 times those in irrigation waters (Morishima et al. 1977).

Vanadium
The concentration of total vanadium in irrigation water should not exceed 0.1 mg/L.

Depending on soil type and species of plant, vanadium concentrations of 10 mg/kg soil may inhibit crop growth. Vanadium interferes with the adsorption of essential elements such as calcium, copper, iron, manganese and phosphorous (Warrington 1955; Cannon 1963; Wallace et al. 1977).

Zinc

The concentration of total zinc in irrigation water should not exceed 2.0 mg/L. For sandy soils below pH($CaCl_2$) 6, 1.0 mg/L is recommended.

Zinc is an essential element for plants and animals; however, high concentrations in soils may have toxic effects on plants and micro-organisms (Schachtschabel et al. 1989). Toxicity to plants generally seems to start at concentrations in nutrient solutions around 0.4 mg/L (Hart 1982). Toxic signs for zinc include iron chlorosis, reduction in leaf size, necrosis of tips and distortion of foliage (Chapman 1966). Zinc is more readily available to plants in acid soils; that is, pH($CaCl_2$) less than 6 (MacLean 1974b; MacLean & Dekker 1978; Hornburg & Brummer 1989).

5.1.5 Pesticides

Herbicide residues in irrigation water should not exceed the concentrations listed in Table 5.9*.*

No guidelines are recommended for insecticides because they are unlikely to be deleterious to crops.

The most important organic compounds in irrigation water are herbicides. These can enter irrigation water through treatment to control algae and submerged aquatic weeds or treatment of irrigation channels and ditches to control terrestrial weeds (CCREM 1991). The herbicides registered in Australia for these purposes are listed in Table 5.9. A review concerning the use of aquatic herbicides is given by Bowmer (1987).

Insecticides in surface waters used for irrigation are unlikely to cause deleterious effects on crops. Thus, the major concern regarding insecticides in irrigation water or soil runoff water is the possible adverse effect on aquatic life and wildlife subsequently using this water. These effects have been discussed in Chapter 2.

The concentrations of insecticides in surface water are generally below 1 µg/L (NAS/NAE 1973), and it is unlikely that these low concentrations can be accumulated by plants to concentrations that would be dangerous to humans and livestock (NAS/NAE 1973).

5.1.6 Radioactivity

Water containing less than 0.1 Bq/L Gross Alpha activity and less than 0.1 Bq/L Gross Beta activity (excluding ^{40}K) is acceptable without taking into account other radioactive sources.

Plants are generally more resistant to radiation than animals. Vegetation can accumulate radionuclides, and contamination of the plants to a level harmful to humans can be reached long before the plants themselves are damaged.

Plants can be contaminated by radionuclides in two ways: first, through direct adsorption on the plant surface through sprinkler irrigation and, second, through root uptake and translocation within the plant system. Myttenaere et al. (1969) showed that the adsorption of the radionuclides caesium and cobalt from irrigation water on plant surfaces was much greater than the uptake via the root system.

Important radionuclides that can be absorbed and concentrated are strontium, caesium, barium, iodine, calcium, chromium, potassium, ruthenium, zirconium and zinc (McKee & Wolf 1963). Relatively abundant radionuclides with long half-lives, such as ^{90}strontium and ^{137}caesium, may be highly concentrated in the surface layer of the soil and may initiate a sequence of increasing concentrations in the plant–animal food chain (Hart 1974).

5.2 Livestock

Groundwater is a major source of drinking water for livestock over a large area of Australia. It may contain large quantities of dissolved salts, depending on the soil and parent rock of the surrounding area and many other factors including rainfall, evaporation, vegetation and topography. Fertiliser and individual effluents may also be a major problem in certain areas.

Table 5.9 Herbicides registered for use in or near waters (mg/L)

Herbicide	Residue limits in irrigation water	Hazard to crops from residue in water**	Crop injury threshold in irrigation water (mg/L)
Acrolein	0.1	+	Flood or furrow: beans 60, corn 60, cotton 80, soybeans 20, sugar-beets 60. Sprinkler: corn 60, soybeans 15, sugar-beets 15
AF 100	*	+	Beets (rutabag) > 3.5, corn 3.5
Amitrol	0.002	++	Lucerne 1,600, beans 1,200, carrots 1,600, corn 3,000, cotton 1,600, grains soghum > 800, oats 2,400, potatoes 1,300, wheat 1,200
Aromatic solvents (Xylene)	*	+	–
Asulam	*	++	–
Atrazine	*	++	–
Bromazil	*	+++	–
Chlorthiamid	*	++	–
Copper sulfate	*	+	Apparently above concentrations used for weed control (see irrigation criterion for copper)
2,4-D	*	++	Field beans 3.5–10, grapes 0.7–1.5, sugar beets 1.0–10
Dicamba	*	++	Cotton 0.18
Dichlobenil	*	++	Lucerne 10, corn > 10, sugar-beets 1.0–10, corn 125, beans 5
Diquat	*	+	–
Diuron	0.002	+++	–
2,2-DPA (Dalapon)	0.004	++	Beets > 7.0, corn < 0.35
Fosamine	*	+++	–
Fluometuron	*	++	Sugar-beets, alfalfa, tomatoes, squash > 2.2
Glyphosate	*	+	–
Hexazinone	*	+++	–
Karbutilate	*	+++	–

Herbicide	Residue limits in irrigation water	Hazard to crops from residue in water**	Crop injury threshold in irrigation water (mg/L)
Molinate	*	++	–
Paraquat	*	+	Corn > 10, field beans 0.1, sugar-beets < 1.0
Picloram	*	+++	–
Propanil	*	++	Alfalfa 0.15, brome grass (eradicated) 0.15
Simazine	*	++	–
2,4,5-T	*	++	Potatoes, alfalfa, garden peas, corn, sugar-beets, wheat, peaches, grapes, apples, tomatoes > 0.5
TCA	*	+++	–
Terbutryne	*	++	–
Triclopyr	*	++	–

* Guideline not set except as a general limit (0.1 mg/L) for specific herbicides in Tasmania and all herbicides in New South Wales.

** Hazard from residue at the expected maximum concentration: + = low, ++ = moderate, +++ = high

> Damage may occur at higher than this level

Sources: NHMRC (1985), Hart (1974), CCREM (1991), Demint et al. (1975), Bruns et al. (1972), Comes and Kelly (1979)

Good water quality is an essential component of successful livestock production. Poor quality water may reduce production by, and interfere with the reproduction of, livestock. In extreme cases stock may die. Animal products, particularly milk, may become contaminated so that their consumption by humans must be restricted. A summary of recommended water quality guidelines is presented in Table 5.10. The recommended guidelines have been largely determined from field observations and not from rigorous experimentation.

The guidelines for drinking water for livestock must take into account the type of livestock, including age and condition; the daily water requirements, which depend on climate; and the concentrations of certain elements in the animal feed. A detailed discussion of these factors is given by Hart (1974) and VIRASC (1980). If drinking water for livestock contains high concentrations of certain compounds, the diet of the animals may require adjustment. Average daily water requirements are listed in Table 5.11.

5.2.1 Biological parameters

Pathogens and parasites

Drinking water for livestock should contain no more than 1,000 faecal coliforms/100 mL (geometric mean [log] based on not less than five water samples taken per month). No more than 20% of these samples should exceed 5,000 faecal coliforms/100 mL.

A number of pathogens and parasites (viruses, bacteria, protozoa, nematodes, trematodes) can be transmitted by contaminated water to livestock, which may result in reduced growth, morbidity or mortality (Smith et al. 1974). Management of sanitation and manure should be emphasised to prevent contamination of water.

One of the most significant bacterial infections is leptospirosis (Carbould 1972). Infection of healthy animals comes from direct contact with urine or with an infected environment. *Leptospira* can

survive for several weeks in moist areas, but cannot survive desiccation. Enteric diseases caused by *Escherichia coli* (e.g. collibactillosis) are probably more important than leptospirosis in Australia (Kabay J., WA Dept Agriculture, pers. comm., May 1992). *Salmonella* may be transmitted to animals by drinking waters containing effluents from abattoirs. Sewage is another important source of this and other pathogens. Anaerobic conditions may be required for growth of certain organisms, including various species of *Clostridium*. Aeration of water aids in preventing the spread of this organism (NCWQ 1968).

A number of serious pathogenic conditions in livestock can be caused by viruses. Pathogenic viruses are not usually transmitted by water, but there is some evidence that water supplies contaminated by faeces may transmit viruses to healthy animals (Hart 1974).

Different types of parasites can live in various organs, the intestinal tracts or muscle tissue of livestock. The majority of livestock usually have a relatively low incidence of parasitic infections. Most of the infections do not cause mortality directly, but reduction of growth rate and vitality is observed and susceptibility to fatal infectious disease organisms increases (CCREM 1991). A number of stock parasites spend part of their life-cycles in water, and faecal contamination is the usual means of introduction into the water. One of the parasitic diseases of concern in Australia is beef measles, a disease that may be associated with sewage sludge contamination.

Table 5.10 Water quality guidelines for livestock watering (mg/L, unless otherwise stated)

Parameter	Guidelines	Comment
Biological parameter	–	–
Pathogens and parasites	1,000 faecal coliforms/100 mL	Geometric mean for not less than 5 water samples taken per month; no more than 20% should exceed 5,000 organisms per 10 mL.
Algae	Up to 10,000 cells/mL may be tolerated, depending on the algal species present.	–
Major ions and nutrients	–	–
Calcium	1,000.0	–
Nitrate-N	30.0	30 (horses), 40 (cattle), 60 (sheep)
Nitrite-N	10.0	–
Sulfate	1,000.0	–
Total dissolved solids	3,000.0	(Table 5.12)
Trace elements	–	–
Aluminium	5.0	–
Arsenic	0.5	–
Beryllium	0.1	–
Boron	5.0	–
Cadmium	0.01	–
Chromium	1.0	–
Cobalt	1.0	–
Copper	0.5	0.5 sheep, 1.0 pigs and poultry, 5.0 cattle

Parameter	Guidelines	Comment
Fluoride	2.0	–
Iron	–	No guideline recommended
Lead	0.1	–
Magnesium	–	(Table 5.13)
Manganese	–	No guideline recommended
Mercury	0.002	–
Molybdenum	0.01	–
Nickel	1.0	–
Selenium	0.02	–
Uranium	0.2	–
Vanadium	0.1	–
Zinc	20.0	–
Pesticides	See guidelines raw water for drinking water supply (Chapter 4)	
Radioactivity	See guidelines raw water for drinking water supply (Chapter 4)	
Other organic toxicants	See guidelines raw water for drinking water supply (Chapter 4)	

Table 5.11 Average daily water requirements for livestock

Class of livestock	Daily water requirement (L)
Dairy cattle	46–91
Beef cattle	32–68
Horses	36–91
Pigs	9–23
Sheep	3–7
Chickens, per 100 birds	18–46

Source: Hart (1974)

Chlorination of drinking water should destroy most bacterial and viral pathogens and some parasites (CCREM 1991). However, adequate management is a better means of ensuring that drinking waters for livestock are not contaminated with manure, sewage or surface runoff containing significant sediment load. The recommended guideline for faecal coliforms is that proposed by VicEPA (1983).

Algae

Up to 10,000 cells/mL of algae may be tolerated in drinking water for livestock, depending on the species present.

Under some circumstances, a bloom of blue-green algae (cyanobacteria) may occur in dams and ponds used for livestock watering. Most of these blooms develop during late summer and autumn. At least six species of freshwater blue-green algae are known to be toxic, with the species *Microcystis aeruginosa* and *Anabaena circinalis* being responsible world-wide for livestock death (Jackson et al. 1985). Animals known to be affected include all common domestic animals (Schwimmer & Schwimmer 1968).

Algal blooms are linked to liver damage and hemorrhage as well as cyanosis and pulmonary oedema (Jackson et al. 1985), and death of livestock can be quite rapid, in some cases occurring in minutes. Results of experimental intoxification determined that 1.44 L of a bloom of *Anabaena* would be lethal to a 60 kg calf (Carmichael et al. 1977), and 0.65 L of a bloom of *Microcystis* would be lethal for a 30 kg lamb (Jackson et al. 1985). Very few quantitative data are available on actual algal levels that have caused stock deaths. It is considered that levels in excess of 10,000 cells/mL may cause problems, depending on the actual species present. If an algal bloom is found in a farm dam, the water supply should be isolated from stock until the algae are identified and the levels of infestation and health risk to the animals are known.

5.2.2 Major ions

Calcium

The concentration of total calcium in drinking water for livestock should not exceed 1,000 mg/L, if calcium is the dominant cation. In the presence of high concentrations of magnesium and sodium, the level of calcium should be adjusted downwards.

Calcium is an essential element for animals, however, high calcium concentrations may cause phosphorus deficiency and calcious formation in the body (Mulhearn 1964). Long-term intake of water containing around 1,100 mg/L calcium by sheep had no adverse effect on health and wool production, but the sodium concentration of the plasma decreased while the calcium content increased (Peirce 1960).

Nitrate and nitrite

The concentration of nitrate-N in drinking water for livestock should not exceed 30 mg/L. For nitrite-N, the recommended value is 10 mg/L.

Both nitrate (NO_3) and nitrite (NO_2) can cause toxicity. To cause toxic symptoms, nitrate must first be reduced to nitrite by bacteria in the digestive system. Nitrite can be absorbed into the bloodstream, where it converts haemoglobin to methaemoglobin, thus reducing the oxygen-carrying capacity of the blood (Hart 1974). As ingestion of nitrite leads to a more rapid onset of toxic effects than nitrate, the guideline value for nitrite must be correspondingly lower than that for nitrate.

Winks (1963) reported death of calves and cattle in Queensland from drinking water containing 2,200 mg/L of nitrate. He suggested that the dangerous concentration of nitrate for cattle is somewhere between 300 mg/L and 2,200 mg/L. Seerely et al. (1965) reported that drinking water containing 330 mg/L did not affect pigs' health through two breeding seasons, whereas Winks (1963) suggested that there may be a risk to pigs from water containing 30–50 mg/L nitrate. The values for horses, cattle and sheep given by Ward (1954) are listed in Table 5.10. Nevertheless, there seem to be insufficient data available to provide detailed, no-effect levels for nitrate in drinking water for different animal species.

Sulfate

The concentration of sulfate in drinking water for livestock should not exceed 1,000 mg/L.

Sulfate causes diarrhoea in young animals at concentrations of 1,000 mg/L (Church 1979). Higher concentrations of sulfate may be tolerated depending on the species of livestock, age, and principal cations associated with the sulfate ion, but loss in production should be expected (CCREM 1991). An

improvement was reported in productivity and health of dairy cattle when their source of drinking water was changed from deep-well water containing 1,500–2,500 mg/L sulfate to surface water containing less than 1,000 mg/L sulfate (CCREM 1991). Hereford cattle showed decreased water and food consumption, weight loss and diuresis when consuming drinking water containing 3,380 mg/L sulfate (Weeth & Hunter 1971).

Total dissolved solids (salinity)

The tolerable concentration of total dissolved solids (TDS) in drinking water for livestock varies considerably depending upon the type of livestock and the mix of dissolved ions (e.g. magnesium). Guidelines are provided in Table 5.12 *(TDS) and* Table 5.13 *(magnesium).*

Highly mineralised waters can cause physiological upset and sometimes death in the majority of terrestrial animals, including humans. Animals under physiological stress, such as reproduction, lactation or rapid growth, are particularly susceptible to mineral imbalances. Salinity is used throughout Australia as a convenient guide to the suitability of water for livestock watering. Table 5.12 summarises the commonly accepted salinity tolerances of livestock.

Table 5.12 Total dissolved solids concentrations for drinking water for livestock (mg/L)*

Stock	Desirable maximum concentration for healthy growth	Maximum concentration at which good condition might be expected	Maximum concentration that may be safe for limited periods
Sheep, dry feed	6,000	13,000	**
Beef cattle	4,000	5,000	10,000
Dairy cattle	3,000	4,000	6,000
Horses	4,000	6,000	7,000
Pigs	2,000	3,000	4,000
Poultry	2,000	3,000	4,000

* Refer also to Table 5.13

** Level depends on type of feed

Source: Hart (1974)

Livestock will normally avoid water of high salinity. Water of marginal quality can cause gastrointestinal symptoms, a reduction in weight gain and a reduction in milk or egg production. However, livestock can acclimatise physiologically to some extent to water of high salinity when the level is adjusted over several weeks.

While total soluble salts may give a guide to water quality, there may also be a need to assess the concentrations of specific ions causing purgative or toxic effects, especially if the salt concentration ranges from 2,500 mg/L to 14,000 mg/L. For example, it is usual to determine the magnesium concentration if the water is to be used by all stock (Saul & Flinn 1978, 1985).

5.2.3 Toxic metals

Aluminium

The concentration of total aluminium in drinking water for livestock should not exceed 5.0 mg/L.

Aluminium is usually present in natural waters in concentrations below 1 mg/L, except in areas with low soil pH where the aluminium content may be increased due to the increased solubility of soil

aluminium oxides and clay minerals. No adverse effects were observed when aluminium sulfate was fed to sheep and cows at concentrations of 15,000 mg/kg of diet (Bailey 1977), or when aluminium chloride was added to steer feed at concentrations of 1,200 mg/kg (Valdivia et al. 1978).

Arsenic

The concentration of total arsenic in drinking water for livestock should not exceed 0.5 mg/L. If arsenic is not provided as a food additive and natural levels of arsenic are low in the diet, a level of 5 mg/L in drinking water may be tolerated.

Arsenical chemicals have been used in the past in sheep dips. The application of arsenic sheep dip is no longer permitted and organophosphates and synthetic pyrethroids are used instead (Turner M., Vic. Dept Agriculture, pers. comm., May 1992). Organic arsenic compounds are used as feed additives to enhance growth in pigs and poultry (Gough et al. 1979).

The toxicity of arsenic depends to a large extent on the form in which it occurs; inorganic arsenic is more toxic than organic arsenic, trivalent inorganic arsenic (arsenite) is more hazardous than the pentavalent form (arsenate). NAS (1980) gave a maximum tolerable dietary level for livestock of 50 mg/kg in feed for inorganic forms and 10 mg/kg for organic forms.

Beryllium

To provide a degree of safety for animal health, the concentration of total beryllium in drinking water for livestock should not exceed 0.1 mg/L. The value is tentative until further research data are available.

Usually beryllium is poorly absorbed from the gastrointestinal tract, and toxicity via this route of entry is low (WHO 1984). However, as reported by USEPA (1980), beryllium may be carcinogenic and has the potential to bioaccumulate. Mice and rats fed over their life-span with a concentration of 0.43 mg Be/L as beryllium sulfate showed no affect in growth and longevity, but some leukemias and tumours were observed (Schroeder & Mitchner 1975a, 1975b). In another study, rats were fed with beryllium in the diet at levels of 5 mg/kg, 50 mg/kg and 500 mg/kg of feed, and no evidence of carcinogenic response related to beryllium was found (WHO 1984).

Boron

The concentration of total boron in drinking water for livestock should not exceed 5.0 mg/L.

Boron may be present in groundwater, however the concentrations normally found in water are below 4 mg/L (Hart 1974). As reported by Green and Weeth (1977), boron concentrations of 150 mg/L in drinking water for cattle resulted in a decrease in hay consumption and a loss of weight. The tolerance concentration of boron was estimated to be between 40 mg/L and 150 mg/L. NAS (1980) suggested a maximum tolerable level of 150 mg/kg boron (as borax) in the diet of cattle, and presumed that this value should be reasonable for other species of livestock.

Cadmium

The concentration of total cadmium in drinking water for livestock should not exceed 0.01 mg/L.

Usually only a small amount of the cadmium intake by livestock comes from drinking water, most being from food. Nevertheless, restrictions need to be placed on cadmium concentrations in drinking water for livestock because of toxic and possibly teratogenic, mutagenic and carcinogenic effects (CCREM 1991).

Miller (1971) reported that only a small part of the ingested cadmium in ruminants was absorbed, with most of that going to the kidney and liver. Anaemia, abortions, stillbirth and reduced growth were observed in animals given cadmium in doses of 1–160 mg/kg bodyweight (Powell et al. 1964; Miller et al. 1967; Doyle et al. 1974; Supplee 1961). Due to the accumulation of cadmium in the liver and kidneys of livestock, and the possible consumption of these organs by humans, toxic levels of cadmium can be passed directly to the consumer.

Chromium

The concentration of total chromium in drinking water for livestock should not exceed 1.0 mg/L.

Chromium is an essential element to mammals; it exists in two natural forms—the trivalent, chromium (III), and the hexavalent, chromium (VI). Trivalent chromium salts are poorly absorbed by the gastrointestinal tract, whereas the absorption rate of hexavalent chromium is much higher. Generally chromium (VI) is more toxic than chromium (III), (WHO 1984; CCREM 1991).

Studies with rats and dogs showed that water containing 5–6 mg/L chromium (VI) did not cause tissue damage; whereas concentrations of 10 mg/L resulted in accumulation of chromium in the tissue, but no toxic effects were detected (NRCC 1976). Rats showed no obvious toxic effects at chromium concentrations (as potassium chromate) of 0.5 mg/L (Romoser et al. 1961), and at 25 mg/L (MacKenzie et al. 1958) in their drinking water.

Cobalt

The concentration of total cobalt in drinking water for livestock should not exceed 1.0 mg/L.

Cobalt is an essential element for animals, and is important in several enzyme systems. Generally cobalt has a low toxicity in animals. Underwood (1977) reported reduced appetite and some weight loss when cobalt was administered daily at concentrations of 1.1 mg/kg bodyweight to the diet of calves. According to CCREM (1991), drinking water for calves would have to contain at least 10 mg/L before the symptoms observed by Underwood would be evident. Pigs, cattle and poultry may tolerate cobalt in concentrations of 10 mg/kg in their diet (NAS 1980).

Copper

The concentration of total copper in drinking water for livestock should not exceed 0.5 mg/L for sheep, 1 mg/L for pigs and poultry, and 5 mg/L for cattle. If the animals' diet is high in copper, the levels in drinking water should be revised downwards.

Copper is essential to animals; copper deficiency can result in morbidity and, in some cases, death (NAS 1977b). Intake of copper in excess can lead to copper toxicosis in livestock, which would be expected to be feed-related rather than water-related. The wide agricultural use of various forms of copper creates a possibility of copper contamination of water supplies, and for this reason it is necessary to establish recommended concentrations of copper in drinking water for livestock.

Toxic effects of copper depend largely on the type of livestock, but also on the form of copper. For example, copper chloride is two to four times more toxic to sheep than is copper sulfate (CCREM 1991). Demayo and Taylor (1981), who reviewed maximum levels of dietary copper intake by livestock, suggested that, to avoid toxicosis, maximal copper concentration in the diet should not exceed 5–20 mg/kg for sheep, 100 mg/kg for cattle, 150–400 mg/kg for pigs and 250–500 mg/kg for chickens.

Fluoride

The concentration of total fluoride in drinking water for livestock should not exceed 2.0 mg/L. If the livestock feed contains fluoride, the drinking water limit should be reduced to 1.0 mg/L.

Groundwater may be contaminated by fluoride leached from fluoride-containing strata. For example, groundwater at Carnarvon, Western Australia, contains around 5 mg/L fluoride. Fluoride accumulates in bones rather than in soft tissue, and excess uptake of fluoride can result in tooth damage to growing animals and bone lesions in older animals (Rose & Marier 1978; CPHA 1979). In Queensland, fluoride in drinking water for livestock at concentrations greater than 2 mg/L has been observed to affect the teeth of young animals (VIRASC 1980).

Another source of excessive ingestion of fluoride may be the diet if the vegetation is contaminated by aerial deposition in industrial areas (NAS 1971). According to NAS (1971), no toxic effects resulted from dietary concentrations of 30–50 mg/kg for cattle, 70–100 mg/kg for sheep and pigs and 150–400 mg/kg for poultry. Van Hensburn and de Vos (1966) have shown that levels of fluoride above 5 mg/L in drinking water adversely affect breeding efficiency in cattle, whereas Hibbs and Thilsted (1983) reported erosion of teeth at concentrations of 3.3 mg/L.

Iron

No guideline has been established for iron in drinking water for livestock.

Iron is essential to animal life and has a low toxicity, only being harmful to livestock if ingested in fairly large amounts. Coup and Campbell (1964) reported slight scouring and blackening of the faeces after administering a daily dose of 30 g of iron as ferric hydroxide. At a dosage of 60 g, scouring and blackening were pronounced and associated with a decline in bodyweight, milk and fat yield and a general worsening in the condition of the coat. No adverse effects were noticed from a daily dosage of 15 g of iron.

Iron-contaminated water does not contain enough iron to cause the above-mentioned problems, but toxic effects can occur when cows are grazed on pastures heavily irrigated with groundwater containing 17 mg/L iron (Hart 1974). Nevertheless, pasture irrigated with water managed according to the Australian water quality guidelines for irrigation water (1 mg/L iron) should not cause problems for grazing stock.

Lead

The concentration of total lead in drinking water for livestock should not exceed 0.1 mg/L.

The toxicity of lead depends on the type of animal (including its age), the form of lead and the rate of lead ingestion (Hart 1982). Lead is accumulated in the skeleton to a critical maximum level, after which circulating concentrations increase until poisoning occurs (Hatch 1977; Jaworski 1979).

Hammond and Aronson (1964) suggested that 6–7 mg ingested lead/kg bodyweight is the minimum concentration that causes poisoning to cattle. Calves were killed by accidental exposure to an estimated lead dose of 5–8 mg/kg/d for thirty days (Osweiler & Ruhr 1978). According to James et al. (1966), sheep died following dietary exposure to 5.7 mg lead/kg bodyweight. Horses appear to be among the animals most sensitive to lead poisoning; chronic poisoning occurred after receiving drinking water and grass contaminated with lead at concentrations of 0.5–1 mg/L and 5–20 mg/kg (dry weight) respectively (Singer 1976).

Reduced resistance to diseases has been caused even by low-level intake of lead (Schroeder & Balassa 1967; Hemphill et al. 1971). The lead guideline for drinking water for livestock has been set at 0.1 mg/L.

Magnesium

The recommended guidelines for total magnesium in drinking water for livestock are related to the TDS levels of the water and are given in Table 5.13.

Certain magnesium salts, particularly magnesium sulfate, can cause scouring and diarrhoea in livestock. According to VIRASC (1980), the tolerance for pigs and poultry is unknown but could be less than 250 mg/L. Peirce (1960) reported that water of approximately 13,000 mg/L total salinity containing up to 250 mg/L magnesium as magnesium chloride had no deleterious effects, but with 506 mg/L and 1,263 mg/L magnesium, food intake decreased and some diarrhoea occurred.

Table 5.13 Magnesium and TDS concentrations in drinking water for livestock*

Category	Guideline	Comments
1	TDS below 5,000 mg/L Magnesium below 600 mg/L	Suitable for sheep & cattle of all ages
2	TDS of 5,000–10,000 mg/L, Magnesium below 600 mg/L	Generally unsuitable for lambs, calves & weaners. Caution needed with lactating stock if unaccustomed. Suitable for dry, mature sheep & cattle
3	TDS of 10,000–15,000 mg/L Magnesium below 600 mg/L	Suitable for dry, mature sheep. Caution needed with cattle if unaccustomed
4	TDS above 15,000 mg/L, Any magnesium level	Generally unsuitable for all stock
5	Any TDS level Magnesium above 600 mg/L	Generally unsuitable for all stock

* Refer also to Table 5.12

Source: Flinn (1984)

According to Saul and Flinn (1978, 1985), cattle can tolerate up to 500 mg/L magnesium. Magnesium concentrations in drinking water for livestock are less important if the level of TDS exceeds 6,000 mg/L (Flinn 1980).

Manganese

No guideline has been established for manganese in drinking water for livestock.

Manganese is an essential element to animals and man, but only about 3% of ingested manganese is absorbed (NAS 1979). Manganese is low in toxicity unless ingested in large amounts (NAS/NAE 1973).

Mercury

The concentration of total mercury in drinking water for livestock should not exceed 0.002 mg/L.

In establishing guidelines for mercury in drinking water for livestock, both the toxic effects of mercury on the animals and the possible accumulation in animal tissue must be considered. The accumulation occurs primarily in the kidney and liver (NRCC 1979). The toxicity of mercury depends mostly on its chemical form, with alkylmercury compounds, particularly methylmercury, being the most toxic.

Signs of poisoning were observed at 2 mg/kg in turkey, 8 mg/kg in cattle and 10 mg/kg in sheep (Palmer et al. 1973). Cattle receiving only 0.48 mg/kg of methylmercury compound per day accumulated 100 mg/kg in the kidney within twenty-seven days; sheep accumulated 120–210 mg/kg under the same conditions (Palmer et al. 1973).

Reeder et al. (1979) suggested that guidelines for mercury should be based on a maximum acceptable level of 0.5 mg/kg in edible animal tissue. Using chicken as a model, Reeder et al. (1979) calculated the maximum allowable intake of mercury in drinking water for stock as being 3 µg/l, assuming a maximum concentration of 0.2 mg/kg in edible animal tissue.

Molybdenum
The concentration of total molybdenum in drinking water for livestock should not exceed 0.01 mg/L.

The toxicity of molybdenum is closely related to the dietary intake of copper and inorganic sulfate. According to NAS (1977), cattle require a copper–molybdenum ratio of 2:1 to prevent molybdenum poisoning, unless copper exceeds 13–16 mg/kg in feed. Pigs, sheep and poultry are more tolerant to elevated molybdenum concentrations (NAS, 1980). Adding ammonium molybdate to drinking water of calves at concentrations of 50 mg/L caused reductions in the copper concentration in liver; molybdenum concentrations lower than 10 mg/L had no effect (Kincaid 1980).

The principal detrimental effects of molybdenum in livestock result from contaminated pasture eaten by animals rather than from molybdenum in drinking water, since molybdenum may be concentrated in plants.

Nickel
The concentration of total nickel in livestock water should not exceed 1.0 mg/L.

Nickel is an essential element in the nutrition of animals (NRCC 1981), and nickel levels of 0.05–0.08 mg/kg in the diet are regarded as essential (Hart 1982). Nickel deficiency can cause pigmentation changes and dermatitis of the shank skin in chickens. Effects on reproduction in pigs were also reported (Nielsen & Ollerich 1974; Anke at al. 1974). Growth reduction in calves was induced by adding nickel salts to the diet at concentrations of 250 mg/kg (O'Dell et al. 1970).

A concentration of 5 mg/L (as nickel acetate) in the drinking water of mice applied over a lifetime was not toxic (Schroeder et al. 1964), whereas nickel chloride at 5 mg/L in the drinking water of rats through three generations resulted in increased mortality among newborn pups and an increased number of runts (Schroeder & Mitchener 1971).

Selenium
The concentration of total selenium in drinking water for livestock should not exceed 0.02 mg/L.

Selenium is both essential and toxic to animals. Diets containing less than 0.02–0.04 mg/kg of selenium resulted in deficiency symptoms in cattle, sheep, pigs and poultry (Oldfield et al. 1974; Underwood 1977). The threshold level of dietary selenium required to induce toxicity is estimated to be 5 mg/kg (Horvath 1976). Acute selenosis results in blindness and often paralysis (Hart 1982). Poisoning of livestock has occurred following ingestion of forage grown in selenium-rich soil (Johnson 1976).

The transmission of selenium into milk, resulting in selenomethionine proteins, is an additional problem. Milk from cows in areas where selenium poisoning occurred was reported to have contained 0.3–1.2 mg/L selenium; normal concentrations range from 0.003–0.007 mg/L (Underwood 1971).

Uranium
The concentration of total uranium in drinking water for livestock should not exceed 0.2 mg/L.

According to Gardner (1963), the minimum concentration of uranium to cause poisoning is 50 mg/d for sheep and 40 mg/d for cattle. Phosphorous supplements fed to dairy cattle may contribute 16 mg/d uranium, depending on the source of phosphorus (Reid et al. 1977).

Vanadium
The concentration of total vanadium in drinking water for livestock should not exceed 0.1 mg/L.

Some experiences with rats and chicks suggest that vanadium is required for lipid, tooth and bone metabolism (Hopkins & Mohr 1971). Concentrations of 2 mg/L (as NH_4VO_3) in drinking water improved the development of growing chicks. According to Van Zinderen et al. (1980), reduced growth rate resulted in chickens and rats given diets containing 13 mg/kg and 25 mg/kg vanadium respectively.

Zinc
The concentration of total zinc in drinking water for livestock should not exceed 20 mg/L.

Zinc is an essential element for both animals and man, and is necessary for the function of various enzyme systems (Parisic & Vallee 1969). Zinc deficiency leads to growth retardation, disorders of bones and joints, skin diseases and low fertility (Farnsworth & Kline 1973). Dietary requirements range from 50 mg/kg to 100 mg/kg of diet (Underwood 1971). According to Neathery and Miller (1977), the estimated maximum safe levels of zinc, expressed as concentrations in the diet, are 500 mg/kg for calves, 600 mg/kg for sheep, 1,000 mg/kg for chicks, pigs and mature cattle, and 2,000 mg/kg for turkeys.

5.2.4 Pesticides

It is recommended that the guidelines set for raw water for drinking water supply (Chapter 4) be adopted.

Pesticides can enter surface waters by adsorption to soil particles in runoff water, adsorption to wind-blown soil particles, direct application to water for control of biting flies or aquatic weeds, drift of pesticide spray and careless disposal of waste pesticides or containers (CCREM 1991).

Usually surface waters contain less than 0.1 mg/L pesticides due to the low concentrations resulting from runoff, aerial drift or direct deposition, and because of the rapid breakdown of most of the newly registered pesticides. The acute toxicity of commonly used herbicides is given in Table 5.14, and Table 5.15 summarises the toxicity of insecticides to livestock. The recommendation that drinking water guidelines be adopted for the maximum limits of pesticides in drinking water for livestock should provide a margin of safety for livestock and prevent unacceptable pesticide residues in animal products.

Table 5.14 Acute toxicity of common herbicides (active ingredient)

Herbicide	Acute oral LD_{50} (mg/kg)*	Reference
Bromoxynil	190	Pimentel (1971)
2,4 D	375–1,200	NRCC (1978)
Dicamba	2,900	Pimentel (1971)
Diclofop-methyl	557–580	Weed Science Society of America (1983)
Difenzoquat	270	Weed Science Society of America, (1983)
Glyphosate	5,600	Weed Science Society of America (1983)
MCPA	700–1,200	NRCC 1978
Propanil	1,384	Weed Science Society of America (1983)
Triallate	1,100	Weed Science Society of America (1983)
Trifluralin	> 10,000	Weed Science Society of America (1983)

* Data for adult laboratory rats

Source: CCREM (1991)

Table 5.15 Summary of laboratory feeding studies on toxicity of insecticides (active ingredient) to livestock

Insecticide	Acute oral LD_{50} (mg/kg)*	Highest dosage at which no effect observed	Reference
Carbaryl	540	Pheasants: 350 mg/kg/bodyweight/d for 30 days	Hudson et al. (1984)
–	–	Japanese quail: 300 mg/kg feed for 14 weeks	Bursian & Edens (1979)
Carbofuran	11	Cattle: 12–147 mg/kg feed**	Miles et al. (1971)
Chlorpyrifos	135	Chickens: 20 mg/L in drinking water for 28 days	Marshall & Roberts (1978)
Fenitrothion	250	Cattle: 100 mg/kg feed for 28 days	Johnson & Bowman (1972)
Methoxychlor	5,000–6,000	Cattle: 0-14 mg/kg feed for 113 days	Gardner & Bailey (1975)

* Data for adult laboratory rats (Pimentel 1971)

** Minor symptoms of nervousness occurred during feeding and disappeared with subsequent feedings

Source: CCREM (1991)

5.2.5 Radioactivity

It is recommended that the guidelines set for raw water for drinking water supply (Chapter 4) be adopted.

All unnecessary exposure to radiation should be avoided. Drinking water standards for livestock watering are set to ensure that the intake of radioactivity will not be harmful to animals and man. According to J. Cooper (Aust. Radiation Laboratory, pers. comm., May 1992), values ten times higher than these may be tolerated.

5.2.6 Other organic toxicants

It is recommended that the guidelines set for raw water for drinking water supply (Chapter 4) be adopted.

Organic toxicants and their degradation products may be accumulated in animal products consumed by humans. It is therefore recommended that the levels of toxic organic substances in drinking water for livestock be limited to the same level as domestic water supplies.

5.3 Farmstead water supplies

On many farms throughout Australia, reticulated water is not available and water is usually obtained from rain-water tanks, streams, irrigation systems, farm dams or groundwater. Rain-water from tanks is generally of good quality but in short supply. Water from other sources may vary from satisfactory to unusable with respect to bacterial levels, TDS, toxic substances and/or turbidity.

Water can be used in the following areas:

- domestic use (including drinking water, washing, hot water supplies)
- dairy water supplies (washing, cooling etc.)
- water for produce preparation (e.g. washing of vegetables).

In order to protect people living on farms and the consumers of farm products, it is recommended that water of the quality outlined in Chapter 4 (domestic supply) should be used. Obviously, in some cases lower quality water will be used due to the lack of water of desirable quality. The possible dangers associated with the use of lower quality water are discussed in Chapter 4 and by VIRASC (1980).

Raw water supplies not meeting the requirements in Chapter 4 should be treated to yield a finished quality comparable to drinking water.

6 Appendix A: References

Chapter 1

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Chapter 2

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7 Appendix B: Glossary of terms

Term	Definition
Abiotic	The non-living components of a system (see biota).
Absorption	In chemistry: Penetration of one substance into the body of another. In biology: The act of absorbing; i.e. to take in as fluids or gases through a cell membrane. To take a substance (e.g. water, nutrients) into the body through the skin or mucous membranes or, in plants, through root hairs.
Acid-soluble metal	The concentration of the metal that passes through a 0.45 µm membrane filter after the sample is acidified to pH 1.5–2.0 with nitric acid.
Acute–chronic ratio	The species mean acute value divided by the chronic value for the same species.
Acute toxicity	Rapid adverse effect (e.g. death) caused by a substance in a living organism. Can be used to define either the exposure or the response to an exposure (effect).
Additive toxicity	The toxicity of a mixture of chemicals that is approximately equivalent to that expected from a simple summation of the known toxicities of the individual chemicals present in the mixture (i.e. algebraic summation of effects).
Adsorption	The taking up of one substance at the surface of another.
Aeration	Any process where a substance becomes permeated with air or another gas. The term is usually applied to aqueous liquids being brought into intimate contact with air by spraying, bubbling or agitating the liquid.
Aesthetic	Those aspects of water that are perceivable by the senses.
Algae	Comparatively simple chlorophyll-bearing plants, most of which are aquatic and microscopic in size.
Alkalinity	The quantitative capacity of aqueous media to react with hydroxyl ions. The equivalent sum of the bases that are titratable with strong acid. Alkalinity is a capacity factor that represents the acid-neutralizing capacity of an aqueous system.
Allochthonous	Organic material that is developed or derived externally from a particular waterbody.
Amphipod	Invertebrates belonging to the order of Crustacea, including the freshwater shrimps.
Antagonism	A phenomenon in which the toxicity of a mixture of chemicals is less than that which would be expected from a simple summation of the toxicities of the individual chemicals present in the mixture (i.e. algebraic subtraction of effects).
Anthropogenic	Produced or caused by humans.
Application factor (AF)	A numerical, unitless value, calculated as the threshold chronically toxic concentration of a chemical divided by its acute toxic concentration. An AF is generally calculated by dividing the limits (no observed effect concentration [NOEC] and lowest observed effect concentration [LOEC]) on the maximum acceptable toxicant concentration (MATC) by the time-independent LC_{50}, if available, or the 96-h LC_{50} (or 48-hour EC_{50}, or 48-h LC_{50} for daphnids) from a flow-through acute toxicity test. The AF is usually reported as a range and is multiplied by the median lethal concentration of a chemical as determined in a short-term (acute) toxicity test to estimate an expected no-effect concentration under chronic exposure.
Assimilation	The incorporation of absorbed substances into cellular material.
Autochthonous	Organic material that is developed or produced within a particular waterbody.
Avoidance threshold	The lowest concentration of a substance that causes a fish to actively move away from the source.

Term	Definition
Benthic	Referring to organisms living in or on the sediments of aquatic habitats (lakes, rivers, ponds, etc.).
Benthos	The sum total of organisms living in, or on, the sediments of aquatic habitats.
Bioaccumulation	General term describing a process by which chemical substances are accumulated by aquatic organisms from water directly or through consumption of food containing the chemicals.
Bioassay	Test used to evaluate the relative potency of a chemical by comparing its effect on a living organism with the effect of a control without the test chemical that is run under identical conditions.
Bioavailable	The fraction of the total chemical in the surrounding environment that can be taken up by organisms. The environment may include water, sediment, suspended particles, and food items.
Biochemical oxygen demand (BOD)	The decrease in oxygen content in mm/L of a sample of water in the dark at a certain temperature over a certain period of time, which is brought about by the bacterial breakdown of organic matter. Usually the decomposition has proceeded so far after 20 days that no further change occurs. The oxygen demand is measured after 5 days (BOD_5), at which time 70% of the final value has usually been reached.
Bioconcentration	A process by which there is a net accumulation of a chemical directly from water into aquatic organisms resulting from simultaneous uptake (e.g. by gill or epithelial tissue) and elimination.
Bioconcentration factor (BCF)	A unitless value describing the degree to which a chemical can be concentrated in the tissues of an organism in the aquatic environment. At apparent equilibrium during the uptake phase of a bioconcentration test, the BCF is the concentration of a chemical in one or more tissues of the aquatic organisms divided by the average exposure concentration in the test.
Biological monitoring	The direct measurement of changes in the biological status of a habitat based on evaluations of the number and distribution of individuals or species before and after a change.
Biomagnification	Result of the processes of bioconcentration and bioaccumulation by which tissue concentrations of bioaccumulated chemicals increase as the chemical passes up through two or more trophic levels. The term implies an efficient transfer of chemicals from food to consumer, so that residue concentrations increase systematically from one trophic level to the next.
Biomass	The living weight of a plant or animal population, usually expressed on a unit area basis.
Biota	The sum total of the living organisms of any designated area.
Bioturbation	The physical disturbance of sediments by burrowing and other activities of organisms.
Bloom	An unusually large number of organisms per unit of water, usually algae, made up of one or a few species.
Buffer	A solution containing a weak acid and its conjugate weak base, the pH of which changes only slightly on the addition of acid or alkali.
Calcium sulfate product	A scaling index used to estimate the upper limit of calcium and sulfate concentrations in the circulating water of a cooling system and expressed as $\{Ca^{2+}\} \times \{SO_2^{4-}\} = 5{,}000{,}000$.
Carcinogen	A substance that induces cancer in a living organism.

Term	Definition
Cation-exchange capacity	The total quantity of cations that a soil can absorb by cation exchange, usually expressed as milli-equivalents per 100 grams. Measured values of cation-exchange capacity depend somewhat on the method used for the determination.
Chelate	The type of co-ordination compound in which a central metal ion is attached by co-ordinate links to two or more non-metal atoms in the same molecule, called ligands.
Chlorination	(1) The process of introducing one or more chlorine atoms into a compound. (2) The application of chlorine to water, sewage or industrial wastes for disinfection or other biological or chemical results.
Chlorophyll	The green pigment in plants.
Chronic	Involving a stimulus that is lingering or continues for a long time; often signifies periods from several weeks to years, depending on the reproductive life cycle of the aquatic species. Can be used to define either the exposure or the response to an exposure (effect). Chronic exposure typically induces a biological response of relatively slow progress and long continuance.
Chronic value	The geometric mean of the lower and upper limits obtained from an acceptable chronic test or by analyzing chronic data using a regression analysis. A lower chronic limit is the highest tested concentration that did not cause an unacceptable amount of adverse effect on any of the specified biological measurements, and below which no tested concentration caused unacceptable effect. An upper chronic limit is the lowest tested concentration that did cause an unacceptable amount of adverse effect on one or more biological measurements and above which all tested concentrations also caused such an effect.
Cladoceran	Water flea. Zooplankton belonging to the fourth order of the Branchiopoda, the Cladocera.
Coagulation	The process of converting a finely divided or colloidally dispersed suspension of a solid into particles of such size that settling takes place.
Colloid	A state of matter intermediate between a true solution and a suspension where the material is typically 0.1 µm to 1 nm in diameter. Colloids (colloidal particles) cannot settle out of a circulating medium through the force of gravity. Colloidal particles cannot diffuse through membranes that do allow ordinary molecules and ions to pass freely.
Community	An assemblage of organisms characterised by a distinctive combination of species occupying a common environment and interacting with one another.
Compensation point	The depth at which assimilation and dissimilation are equal (see assimilation).
Complexation	The formation of a compound by the union of a metal ion with a non-metallic ion or molecule called a ligand or complexing agent.
Concentration	The quantifiable amount of chemical in the surrounding water, food or sediment.
Control	A treatment in a toxicity test that duplicates all the conditions of the exposure treatments but contains no test material. The control is used to determine the absence of toxicity in the basic test conditions (e.g. health of test organisms, quality of dilution water).
Corrosion	The electrochemical degradation of metals or alloys due to reaction with their environment; it is accelerated by the presence of acids or bases.
Cresylic	Acidic commercial mixture of phenolic materials boiling above the cresol range (greater than 240°C).
Criteria (water quality)	Scientific data evaluated to derive the recommended limits for water uses.
Cumulative	Brought about, or increased in strength, by successive additions at different times or in different ways.

Term	Definition
Cyanide, weak acid dissociable	The concentration of HCN following distillation of the sample in the presence of acetate buffer at pH 4.5. This method measures free, simple and weak-acid-dissociable metal cyanides.
Depauperate	To reduce in quality, vigour or capacity.
Diagenesis	During generation or through generation.
Detection limit	The smallest concentration or amount of a substance that can be reported as present with a specified degree of certainty by a definite, complete analytical procedure.
Detritus	Unconsolidated sediments composed of both inorganic and dead and decaying organic material.
Dissolved constituent	The constituents in a water sample that will pass through a 0.45 µm membrane filter.
Diurnal	Daily
Diurnal cycling	Having a period of variation of one day.
Dose	The quantifiable amount of a material introduced into an animal.
EC_{50}	See median effective concentration.
Early life-stage test	28-day to 32-day (60-day post-hatch for salmonids) exposures of the early life stages of a species of fish from shortly after fertilisation through embryonic, larval and early juvenile development. Data are obtained on survival and growth.
Effluent	A complex waste material (e.g. liquid industrial discharge or sewage) that may be discharged into the environment.
Environmental values	Particular values or uses of the environment that are conducive to public benefit, welfare, safety or health and that require protection from the effects of pollution, waste discharges and deposits. Several environmental values may be designated for a specific waterbody.
Enteropathogenic	Capable of producing disease in the intestines.
Epilimnion	The uppermost layer of water in a lake, characterised by an essentially uniform temperature that is generally warmer than elsewhere in the lake, and by relatively uniform mixing by wind and wave action.
Epilithon	Organisms attached on rocks, such as algae and lichens.
Epiphyte	A plant that grows on the outside of another plant, using it for support only and not obtaining food from it.
Euphotic zone	Surface waters to a depth of approximately 80–100 m; the lit region that extends virtually from the water surface to the level at which photosynthesis fails to occur because of reduced light penetration.
Eutrophic	Abundant in nutrients and having high rates of productivity frequently resulting in oxygen depletion below the surface layer of a waterbody.
Eutrophication	Enrichment of waters with nutrients, primarily phosphorus, causing abundant aquatic plant growth.
Evapotranspiration	The combined loss of material from a given area during a specified period of time by evaporation from the soil or water surface and by transpiration from plants.
Exposure	The amount of physical or chemical agent that reaches a target or receptor.
Fate	Disposition of a material in various environmental compartments (e.g. soil or sediment, water, air, biota) as a result of transport, transformation and degradation.

Term	Definition
Field capacity	The greatest amount of water that is possible for a soil to hold in its pore spaces after excess water has drained away.
Flocculation	(1) The process by which suspended colloidal or very fine particles coalesce and agglomerate into well-defined hydrated floccules of sufficient size to settle rapidly. (2) The stirring of water after coagulant chemicals have been added to promote the formation of particles that will settle.
Flow-through system	An exposure system for aquatic toxicity tests in which the test material solutions and control water flow into and out of test chambers on a once-through basis either intermittently or continuously.
Fulvic	Dull yellowish brown or tawny.
Guideline (water quality)	Numerical concentration limit or narrative statement recommended to support and maintain a designated water use.
Half-life	Time required to reduce by one-half the concentration of a material in a medium (e.g. soil or water) or organism (e.g. fish tissue) by transport, degradation, transformation or depuration.
Hardness	The concentration of all metallic cations, except those of the alkali metals, present in water. In general, hardness is a measure of the concentration of calcium and magnesium ions in water and is frequently expressed as mm/L calcium carbonate equivalent.
Hazard evaluation	Identification and assessment of the potential adverse effects that could result from manufacture, use and disposal of a material in a specified quantity and manner.
Heterotrophy	The nutrition of plants and animals that are dependant on organic matter for food.
Humic substances	Organic substances only partially broken down that occur in water mainly in a colloidal state. Humic acids are large-molecule organic acids that dissolve in water.
Hydrograph	Graphical representation of either surface stream discharges or water-level fluctuations in wells.
Hydrolysis	(1) The formation of an acid and a base from a salt by the ionic dissociation of water. (2) The decomposition of organic compounds by interaction with water.
Hypolimnion	The region of a waterbody that extends from below the thermocline to the bottom of the lake; it is thus removed from much of the surface influence.
Hypoxia	Deficiency of oxygen in tissues or in blood. Anoxia.
In vitro	Outside the intact organism; generally applied to experiments involving biochemical events occurring in tissue fragments or fractions.
In vivo	Within an intact animal or organism.
Incipient LC_{50}	The concentration of a chemical that is lethal to 50% of the test organisms as a result of exposure for periods sufficiently long that acute lethal action has essentially ceased. The asymptote (part of the toxicity curve parallel to the time axis) of the toxicity curve indicates the value of the incipient LC_{50} approximately.
Ingestion	The swallowing or taking in of food material.
Interstitial	Occurring in interstices or spaces; applied to water found between and to flora and fauna living between sand grains and soil particles.
Invertebrates	Animals lacking a dorsal column of vertebrae or a notochord.
Joint action	Two or more chemicals exerting their effects simultaneously.

Term	Definition
Langelier Saturation Index (SI)	Index relating the actual pH of water (pH) to the pH at which water is just saturated with calcium carbonate (pH_s). SI = $pH–pH_s$.
Leachate	Water that has passed through a soil and that contains soluble material removed from that soil.
Leaching	The downward movement of a material in solution through soil.
Lethal	Causing death by direct action. Death of aquatic organisms is the cessation of all visible signs of biological activity.
Lentic	Standing body of water such as a lake or pond.
Life-cycle study	A chronic (or full chronic) study in which all the significant life stages of an organism are exposed to a test material. Generally, a life-cycle test involves an entire reproductive cycle of the organism. A partial life-cycle toxicity test includes the part of the life cycle observed to be especially sensitive to chemical exposure.
Ligand	A molecule, ion or atom that is attached to the central atom of a co-ordination compound, a chelate or other complex. May also be called complexing agent.
Limnetic	The open water region of a lake; zone of deep water between surface and compensation depth.
Limnology	The study of fresh water, including biological, geological, physical and chemical aspects.
Lipophilic	Having an affinity for fats or other lipids; substances that concentrate in fatty tissues of organisms.
Littoral zone	The interface region between the land of the draining basin and the open water of lakes.
Lowest observed effect concentration (LOEC)	The lowest concentration of a material used in a toxicity test that has a statistically significant adverse effect on the exposed population of test organisms as compared with the controls. When derived from a life-cycle or partial life-cycle test, it is numerically the same as the upper limit of the MATC.
Lowest observed effect level (LOEL)	The lowest concentration that produces an observable effect in a test species. Below this concentration there are no observed effects in the test species.
Macrograzer	An organism feeding on relatively large particles of food.
Macrophyte	A member of the macroscopic plant life, especially of a waterbody.
Macroscopic	Large enough to be observed by the naked eye.
Maximum acceptable toxicant concentration (MATC)	The concentration of a toxic substance that may be present in a receiving water without causing significant harm to its productivity or uses as determined by chronic toxicity tests.
Measured flow-through test	A toxicity test with constant flow or continuous flow of water where the concentration of the substance in the water is measured.
Median effective dose (ED_{50})	The dose of material estimated to be effective in producing some sub-lethal response in 50% of the test organisms. Appropriate for use with test animals such as rats, mice and dogs, it is rarely applicable to aquatic organisms because it indicates the quantity of a material introduced directly into the body by injection or ingestion rather than the concentration of the material in water in which aquatic organisms are exposed during toxicity tests.
Median effective concentration (EC_{50})	The concentration of material in water to which test organisms are exposed that is estimated to be effective in producing some lethal response in 50% of the test organisms. The LC_{50} is usually expressed as a time-dependant value (e.g. 24-hour or 96-hour LC_{50}).

Term	Definition
Median lethal concentration (LC_{50})	The concentration of material in water to which test organisms are exposed that is estimated to be lethal to 50% of the test organisms. The LC_{50} is usually expressed as a time-dependent value (e.g. 24-hour or 96-hour LC_{50}; the concentration estimated to be lethal to 50% of the test organisms after 24 or 96 hours of exposure).
Median lethal dose (LD_{50})	The dose of material that is estimated to be lethal to 50% of the test organisms. Appropriate for use with test animals such as rats, mice and dogs, it is rarely applicable to aquatic organisms because it indicates the quantity of a material introduced directly into the body by injection or ingestion rather than the concentration of the material in water in which aquatic organisms are exposed during toxicity tests.
Median tolerance limit (TL_m or TL_{50})	The concentration of material in water at which 50% of the test organisms survive after a specified time of exposure. The TL_{50} (equivalent to the TL_m) is usually expressed as a time dependent value (e.g. 24-hour or 96-hour TL_{50}; the estimated concentration at which 50% of the test organisms survive after 24 or 96 hours of exposure). Unlike lethal concentration and lethal dose, the term 'tolerance limit' is applicable in designating a level of any measurable lethal condition (e.g. extremes in pH, temperature, dissolved oxygen). TL_m and TL_{50} have been replaced by median lethal concentration (LC_{50}) and median effective concentration (EC_{50}).
Mesotrophic	Organisms providing a moderate amounts of nutrition.
Metabolism	The sum of all chemical processes occurring in an organism or living cell.
Metabolite	Any product of metabolism.
Methylation	The introduction of methyl (CH_3) groups into organic and inorganic compounds.
Micrograzer	An organism feeding on small particles of food.
Mineralise	To convert to a mineral substance; to impregnate with mineral material.
Monomeric	A chemical compound comprising single molecules.
Multivariate	Statistical analysis concerned with data collected on several dimensions of the same organism.
Munsell Scale	A means of expressing the colour of a soil by matching it against a colour chart.
Mutagenesis	The alteration of the genetic material of a cell in such a manner that the alteration is transmitted to subsequent generations of cells.
Nannoplankton	Those organisms suspended in open water that, because of their small size, cannot be collected by nets. They can be recovered by sedimentation or centrifugation.
Nekton	The powerful swimmers among the freshwater animals that to a large degree are capable of moving about voluntarily from place to place.
Neuston	The collective term for microscopic components of the pleuston that are adapted to the interface habitat between air and water. The neuston are classified as those organisms adapted to living on the upper surface of the interface film (the epineuston) and those living on the underside of the surface film (the hyponeuston).
Not detectable	Below the limit of detection of a specified method of analysis.
Objective (water quality)	A numerical concentration limit or narrative statement that has been established to support and protect the designated uses of water at a specified site.
Octanol–water partition coefficient (P_{ow})	The ratio of a chemical's solubility in *n*-octanol and water at equilibrium. The logarithm of P_{ow} is used as an indication of a chemical's propensity for bioconcentration by aquatic organisms.
Oligotrophic	Waters with a small supply of nutrients.

Term	Definition
Organism	Any living animal or plant; anything capable of carrying on life processes.
Organoleptic	Pertaining to or perceived by a sensory organ.
Osmolality	Measure of osmotic concentration. The total number of osmotically active particles per litre of solvent.
Osmosis	Diffusion of a solvent through a semi-permeable membrane into a more concentrated solution, tending to equalise the concentrations on both sides of the membrane.
Oxidation	The combination of oxygen with a substance, or the removal of hydrogen from it or, more generally, any reaction in which an atom loses electrons.
Oxygenation	The process of adding dissolved oxygen to a solution.
Parameter	A measurable or quantifiable characteristic or feature.
Partition coefficient	A ratio of the equilibrium concentration of the chemical between a non-polar and polar solvent.
Pathogen	An organism capable of eliciting disease symptoms in another organism.
Pelagic	Term applied to organisms of the plankton and nekton which inhabit the open water of a sea or lake.
Periphyton	The organisms attached to submerged plants.
Pesticide	A substance or mixture of substances used to kill unwanted species of plants or animals.
pH	Value taken to represent the acidity or alkalinity of an aqueous solution. It is defined as the negative logarithm of the hydrogen ion acidity of the solution.
Photodegradation	Breakdown of a substance by exposure to light. The process whereby ultra-violet radiation in sunlight attacks a chemical bond or link in a chemical structure.
Photolysis	The decomposition of a compound into simpler units as a result of absorbing one or more quanta of radiation.
Photo-oxidation	Oxidation induced by radiant energy.
Photosynthesis	The conversion of carbon dioxide to carbohydrates in the presence of chlorophyll using light energy.
Physiology	The study of the functioning of organisms and their parts.
Phytophthora	A root parasite.
Plankton	Plants (phytoplankton) and annuals (zooplankton), usually microscopic, floating in aquatic systems.
Potable water	Water suitable, on the basis of both health and aesthetic considerations, for drinking or culinary purposes.
Polycyclic	An organic compound having four or more ring structures, which may be the same or different.
Precipitation	(1) The formation of solid particles in a solution. Generally, the settling out of small particles. (2) The settling-out of water from cloud, in the form of rain, hail, snow, etc.
Primary production	The production of organic matter from inorganic materials.
Producers	Organisms that are able to build up their body substance from inorganic materials.
Prolarvae	Newly hatched larvae during the first few days when they feed on their supply of embryonic yolk.
Raw water	Surface or groundwater that is available as a source of drinking water but has not received any treatment.

Term	Definition
Risk	A statistical concept defined as the expected frequency or probability of undesirable effects resulting from a specified exposure to known or potential environmental con-centrations of a material. A material is considered safe if the risks associated with its exposure are judged to be acceptable. Estimates of risk may be expressed in absolute or relative terms. Absolute risk is the excess risk due to exposure. Relative risk is the ratio of the risk in the exposed population to the risk in the unexposed population.
Ryzner Stability Index (RSI)	Index relating the pH of water (pH) to the pH of water just saturated with calcium carborate (pH_s).
Safety factor	A number used to provide an extra margin of safety beyond the known or estimated sensitivities of aquatic organisms. Often applied when sufficient information about the toxicity, particularly the chronic toxicity, of a particular substance is not well known.
Saprobian system	Community of organisms that feed on decaying organic matter.
Scale	A calcareous deposit in water tubes or steam boilers resulting from deposition of mineral compounds present in the water.
Sorption	A surface phenomenon that may be either absorption or adsorption, or a combination of the two. The term is often used when the specific mechanism is not known.
Species	Generally regarded as a group of organisms that resemble each other to a greater degree than members of other groups and that form a reproductively isolated group that will not normally breed with members of another group.
Standard (water quality)	An objective that is recognised in enforceable environmental control laws of a level of government.
Standing crop	The weight of organic material that can be sampled or harvested by normal methods at any one time from a given area.
Static system	An exposure system of aquatic toxicity tests in which the test chambers contain solutions of the test material or control water that are not usually changed during the test. Depending upon conditions, a static system may or may not be in equilibrium.
Steady state or dynamic equilibrium	The state at which the competing rates of uptake and elimination of a chemical within an organism or tissue are equal. An apparent steady state is reached when the concentration of a chemical in tissue remains essentially constant during a continuous exposure.
Stoichiometric weight	The relative quantities of elements in a chemical compound according to their combining weights.
Sub-lethal	Involving a stimulus below the level that causes death.
Sub-littoral	The shore zone from the lowest water level to the lower boundary of plant growth.
Survival time	The time interval between initial exposure of an aquatic organism to a harmful parameter and death.
Suspended constituent	The constituents in a water sample (the residue) that are retained on a filter medium. The type of filter must be specified.
Suspension	A system in which very small particles (solid, semi-solid, or liquid) are more or less uniformly dispersed in a liquid or gaseous medium. If the particles are small enough to pass through filter membranes, the system is termed a colloidal suspension. If the particles are larger than colloidal dimensions they will tend to precipitate, if heavier than the suspending medium, or to agglomerate and rise to the surface, if lighter.
Synergism	A phenomenon in which the toxicity of a mixture of chemicals is greater than that to be expected from a simple summation of the toxicities of the individual chemicals present in the mixture.

Term	Definition
Teratogen	An agent that increases the incidence of congenital malformations.
Threshold concentration	A concentration above which some effect (or response) will be produced and below which it will not.
Tolerance	The ability of an organism to withstand adverse or other environmental conditions for an indefinitely long exposure without dying.
Total metal	The concentration of a metal in an unfiltered sample that is digested in strong nitric acid.
Total recoverable metal	The concentration of a metal in an unfiltered sample following treatment with hot dilute mineral acid.
Toxicant	An agent or material capable of producing an adverse response (effect) in a biological system, seriously injuring structure or function or producing death.
Toxicity	The inherent potential or capacity of a material to cause adverse effects in a living organism.
Toxicity test	The means by which the toxicity of a chemical or other test material is determined. A toxicity test is used to measure the degree of response produced by exposure to a specific level of stimulus (or concentration of chemical).
True colour	The colour of water resulting from substances that are totally in solution; not to be mistaken for apparent colour resulting from colloidal or suspended matter.
Turbulence	Unorganized movement in liquids and gases resulting from eddy formation.
Univariate	Statistical analysis concerned with data collected on one dimension of the same organism.
Uptake	A process by which materials are absorbed and incorporated into a living organism.
Volatile	Having a low boiling or subliming pressure (a high vapour pressure).
Xenobiotic	A foreign chemical or material not produced in nature and not normally considered a constitutive component of a specified biological system. This term is usually applied to manufactured chemicals.
Zooplankton	The animal portion of the plankton.

8 Appendix C: Abbreviations and acronyms

Some scientific abbreviations not included here may be found in the Glossary of Terms (Appendix B).

Abbreviation	Term
AEAM	Adaptive environmental assessment management
AEC	Australian Environment Council
ABS	alkyl benzene sulfonate
aN	available nitrogen
ANZECC	Australian and New Zealand Environment and Conservation Council
aP	available phosphorus
APHA	American Public Health Association
AS	Australian Standard
ASTM	American Society for Testing Materials
AWRC	Australian Water Resources Council
BOD	biological oxygen demand
CCREM	Canadian Council of Resource and Environment Ministers
CEC	Commission of the European Communities
DDT	dichlorodiphenyltrichloroethane
DIN	dissolved inorganic nitrogen
DIP	dissolved inorganic phosphorus
DOC	dissolved organic carbon
DRP	dissolved reactive phosphorus
EC	electrical conductivity
EPA	Environment Protection Authority
FAO	Food and Agriculture Organization
IARC	International Agency for Cancer Research
IJC	International Joint Commission
IWBDE	Inland Water Branch of the Department of the Environment
LAS	linear alkylate sulfonate
LOEL	lowest observed effect level
Abbreviation	Term
MBAS	methylene blue active substances
MCPA	2-methyl-4-chlorophenoxyacetic acid
MPN	most probable number
NAE	National Academy of Engineering
NAS	National Academy of Sciences
NATA	National Association of Testing Authorities

Abbreviation	Term
ND	not detectable
NEPA	National Environment Protection Authority
NHMRC	National Health and Medical Research Council
NR	no recommendation
NRC	National Research Council
NRCC	National Research Council of Canada
NS	not specified
NTU	nephelometric turbidity units
NZRMA	New Zealand Resource Management Authority
OECD	Organization for Economic Co-operation and Development
PAR	photosynthetically available radiation
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCE	perchloroethylene
PAH	polycyclic aromatic hydrocarbon
PCP	pentachlorophenol
P:R	production to respiration ratio
SAR	sodium adsorption ratio
SER	State of the Environment Report
SPM	suspended particulate matter
Spp.	Species (plural)
TCA	trichloroethane
TCDD	tetrachlorodibenzo-*p*-dioxin
TCE	trichloroethene
TDS	total dissolved solids
Abbreviation	Term
THM	trihalomethane
TN	total nitrogen
TP	total phosphorus
USEPA	United States Environmental Protection Agency
VicEPA	Victorian Environment Protection Authority
VIRASC	Victorian Irrigation Research and Advisory Services Committee
WHO	World Health Organization